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Biochar application to soil for climate change mitigation by soil organic carbon sequestration

Klaus Lorenz, Institute for Advanced Sustainability Studies e.V., Berliner Strasse 130, 14467 Potsdam, Germany*

Rattan Lal, Carbon Management and Sequestration Center, School of Environment and Natural Resources, College of Food, Agricultural, and Environmental Sciences, The Ohio State University, Columbus, OH 43210, United States of America

*Now at: Carbon Management and Sequestration Center, School of Environment and Natural Resources, College of Food, Agricultural, and Environmental Sciences, The Ohio State University, Columbus, OH 43210, United States of America

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Abstract

Pyrogenic carbon (C) is produced by incomplete combustion of fuels including organic matter (OM). Certain ranges in the combustion continuum are termed black carbon (BC). Because of its assumed persistence, surface soils in large parts of the world contain BC with up to 80% of surface soil organic carbon (SOC) stocks and up to 32% of sub-soil SOC in agricultural soils consisting of BC. High SOC stocks and high levels of soil fertility in some ancient soils containing charcoal (e.g., terra preta de Índio) have recently been used as strategies for soil applications of biochar, an engineered BC material similar to charcoal but with the purposeful use as a soil conditioner (i) to mitigate increases in atmospheric carbon dioxide (CO$_2$) by SOC sequestration and (ii) to enhance soil fertility. However, effects of biochar on soils and crop productivity cannot be generalized as they are biochar-, plant- and site-specific. For example, the largest potential increases in crop yields were reported in areas with highly weathered soils, such as those characterizing much of the humid tropics. Soils of high inherent fertility, characterizing much of the world’s important agricultural areas, appear to be less likely to benefit from biochar. It has been hypothesized that both liming and aggregating/moistening effects of biochar improved crop productivity. Meta-analyses of biochar effects on SOC sequestration have not yet been reported. To effectively mitigate climate change by SOC sequestration, a net removal of C and storage in soil relative to atmospheric CO$_2$ must occur and persist for several hundred years to a few millennia. At deeper soil depths, SOC is characterized by long turnover times, enhanced stabilization, and less vulnerability to loss by decomposition and erosion. In fact, some studies have reported preferential long-term accumulation of BC at deeper depths. Thus, it is hypothesized that surface applied biochar-C (i) must be translocated to sub-soil layers and (ii) result in deepening of SOC distribution for a notable contribution to climate change mitigation. Detailed studies are needed to understand how surface applied biochar can move to deeper soil depths, and how its application affects
organic C input to deeper soil depths. Based on this knowledge, biochar systems for climate change mitigation through SOC sequestration can be designed. It is critically important to identify mechanisms underlying the sometimes observed negative effects of biochar application on biomass, yield and SOC as biochar may persist in soils for long periods of time as well as the impacts on downstream environments and the net climate impact when biochar particles become airborne.

1 Introduction

Soils may receive black carbon (BC) and other forms of combustion-derived or pyrogenic carbon (C) (Preston and Schmidt, 2006). Aside from fossil C, combustion-derived C compounds may be the only non-mineral-associated soil organic matter (SOM) component that may be persistent in soil (Marschner et al., 2008). BC constitutes between 0% and about 80% of soil organic carbon (SOC) in surface soils (Krull et al., 2008). However, no common definition for BC exists. The BC in soils can be method-defined as a carbonaceous substance of pyrogenic origin which is resistant to thermal or chemical degradation under conditions specified by the analytical methods (Table 1; Hammes and Abiven, 2013).

Heating of solid fuels (i.e., biomass) in an oxygen-deficit environment (pyrolysis) has been traditionally used to produce charcoal, a residual form of C in solid form (Table 1; Spokas, 2010). Some charcoal particles found in soil can reach radiocarbon ages of thousands of years (Schmidt and Noack, 2000). However, it is unclear how much charcoal vs. organic waste additions have contributed to high levels of soil fertility and relatively high SOC stocks of terra preta de Índio (terra preta) of Central Amazonia, Plaggenesch soils (Plaggic Anthrosols) of North-West Europe, and Terra Preta Australis soils (Sombroek, 1966; Davidson et al., 2006; Downie et al., 2011; Glaser and Birk, 2013). Even so, engineered BC or ‘biochar’ currently receives increased attention as a soil conditioner to (i) mitigate
atmospheric increases in carbon dioxide (CO₂) by causing a net increase in SOC, and (ii) enhance soil fertility and resilience of crop land (Sohi, 2012).

Biochar can be defined as a C-enriched, fine-grained and porous by-product of slow pyrolysis when organic material (feedstock) is thermally decomposed at low-moderate temperatures during long heating times under limited supply of oxygen (Sohi et al., 2010). Feedstock may include wood chips and wood pellets, tree bark, crop residues, energy crop, organic wastes, chicken litter, dairy manure or sewage sludge. For a particular feedstock, biochar properties depend on the extent of pyrolysis (i.e., peak process temperature), and its completeness on particle size of the material and heating time. In particular, the fused aromatic ring cluster size of biochars depends on the specific production process (Brewer et al., 2009). However, no consistent definition of biochar exists as feedstock and process conditions vary widely (Kookana et al., 2011). In contrast to charcoal and other BC forms, biochar is added to soil for its benefits of C sequestration and subsequent soil quality improvements (Spokas, 2010).

Thus, biochar can be defined as charcoal for which scientific consensus exists that application to soil at a specific site is expected to substantially sequester C and concurrently improve soil functions while avoiding detrimental effects (Table 1; Verheijen et al., 2009). In essence, the new term biochar describes exactly the same material as the term charcoal while the difference is the purpose of use (IBI, 2012).

BC is considered by some as a very stable component of SOC but soil BC cannot be viewed as being generally inert (Lehmann, 2007; Czimczik and Masiello, 2007). However, BC’s decomposition pathways remain a mystery (Schmidt et al., 2011). For example, a major portion of the annual charcoal production from biomass burning may not be contributing to soil BC but be lost via dissolution and subsequent transport to the oceans (Jaffé et al., 2013). This limited understanding of soil BC loss is also indicated by simple mass balance calculations based on BC production rates since the Last Glacial Maximum and assumed BC
recalcitrance indicating that BC should compromise theoretically between 25% and 125% of
SOC (Masiello and Druffel, 2003). For the long-term storage of BC in soil, the chemical
resistance may be less important than physical protection and interaction with soil minerals
(Czimczik and Masiello, 2007; Cusack et al., 2012). However, monitoring temporal changes
in soil BC is (i) challenging as the chemistry of precursor compounds varies along the
combustion continuum, and (ii) difficult relative to SOM, i.e., the complex mixture of organic
compounds with a wide distribution of molecular properties and residence times (Behre and
Kleber, 2013). For example, char and soot may have overlapping properties such as specific
surface area and oxidative kinetics depending on formation conditions (Hammes et al., 2007).
Similarly, biochar possesses a range of chemical structures (Spokas, 2010). While some BC
quantification methods have the potential to differentiate charcoal C from soot C in soil, the
differentiation from diagenetic C (e.g., lignite and bituminous coal C) remains challenging
(Roth et al., 2012). Importantly, no single correct method exists for quantification of BC
(Hammes and Abiven, 2013)
For the first time in Europe, the Swiss Federal Ministry of Agriculture officially approved in
2013 the use of certified biochar in agriculture (http://www.ithaka-journal.net/schweiz-
bewilligt-pflanzenkohle-zur-bodenverbesserung?lang=en). Approval is based on strict,
scientifically checked requirements with regard to the sustainability of biochar production, to
biochar quality and to user protection in its application. Further, the European Biochar
Certificate has been developed to become the voluntary European industrial standard ensuring
a sustainable biochar production and low hazard use in agronomic systems
(http://www.european-biochar.org/en). However, there is currently no coherent EU policy
addressing biochar (Vereš et al., 2013), and biochars’ classification as waste blocks, de facto,
its agronomic utilization (Montanarella and Lugato, 2013). In the U.S., some biochar
production systems have been recommended for generating C offsets by soil sequestration
(De Gryze et al., 2010). Also, proposed U.S. federal legislation to comprehensively address energy and climate change (i.e., the American Power Act) included “projects for biochar production and use” to be considered for domestic C offset programs (Gurwick et al., 2012). Recently, the International Biochar Initiative (IBI) certified the first biochar material for effective use as a soil amendment for a California-based developer of small-scale bio-refineries for the conversion of non-food biomass into biofuels and biochar (http://www.biochar-international.org/certification). Commercially available in the U.S. is Maxfield’s soil conditioner, a product with about 10% biochar (Maddox, 2013). Further, large-scale biochar production from crop straw is now commercially available in China (Pan et al., 2011). However, the biochar price is high and would not be balanced by the potential economic gains based on average yield improvements and current prices for CO2 (Liu et al., 2013). Thus, biochar has not yet made a substantial entry into large-scale agricultural operations (IBI, 2014).

Globally, Woolf et al. (2010) estimated a maximum sustainable technical potential of biochar to significantly mitigate climate change. Further, biochar has also been considered as a geoengineering solution for climate-change mitigation because it is assumed to result in the net removal of CO2 from the atmosphere (Downie et al., 2012). However, recommendations regarding soil C offsets and mitigation potential cannot be generalized as biochar composition is variable, and it is not known how long a particular biochar particle remains stable in a particular soil (Sohi, 2012; Schimmelpfenning and Glaser, 2012; Spokas et al., 2012). In fact, interactions between biochar, soil, microorganisms and plant roots are biochar-, plant- and site-specific (Joseph et al., 2010). Soil addition of biochar may result in ‘SOC sequestration’ by causing a net additional long-term (i.e., >100 y) removal of CO2 from the atmosphere and C storage in the SOC pool as this process is a genuine contribution to climate change mitigation (Powlson et al., 2011; Stockmann et al., 2013). However, Mackey et al. (2013)
suggested that for climate change mitigation CO$_2$ must remain stored for much longer (>10,000 y). Also for C management, an intrinsic stability for at least 2000 y should be a key feature of biochar (Schimmelpfenning and Glaser, 2012). Aside from soil sequestration, the geological sequestration of biochar has recently been proposed for climate change mitigation (Dufour, 2013).

The objectives of the article are: (i) to briefly discuss what the term ‘SOC sequestration’ means with regard to climate change mitigation, and (ii) to collate information about both direct and indirect effects of biochar application on ‘SOC sequestration’ in agricultural soils. Examples of field experiments, where fertilizer was not applied simultaneously, will be discussed predominantly. The interactions between biochar and fertilizer are reviewed elsewhere (e.g., Biederman and Harpole, 2013). The article concludes with an overview of research gaps that need to be addressed to realize the full potential of biochar for climate change mitigation by ‘SOC sequestration’. Rigorous studies are needed on direct biochar-C inputs at deeper soil depths and indirect increases in subsoil SOC resulting from biochar application as ‘SOC sequestration’ at deeper depths may usefully contribute to climate change mitigation. As only a limited number of studies about the potential of biochar for long-term SOC sequestration is available, studies dealing with char types of comparable major properties (i.e., BC, charcoal) will also be discussed. Thus, the term biochar will be used interchangeably with the terms BC and charcoal.

2 The Meaning of Soil Organic Carbon Sequestration for Climate Change Mitigation

‘Carbon sequestration’ is one of the most important concepts in studies of climate change (Krna and Rapson, 2013). Since CO$_2$ accounts for about 60% of greenhouse gas (GHG) emissions, reducing the net increase in atmospheric CO$_2$ concentration by ‘C sequestration’ can be an effective mitigation strategy for climate change and for moderating anthropogenic alterations of the global C cycle. However, there is little consensus in the literature what the
term ‘C sequestration’ means. For example, Krna and Rapson (2013) defined ‘endogenous C sequestration’ as when non-temporarily utilized biologic C (i.e., C absent from living organism and not used in growth) fixed from the atmosphere is greater than the release of C to the atmosphere over a specified time period (minimally annual) and within a given system. A major issue is how long C must be sequestered in a system (i.e., land, soil) to usefully contribute to climate change mitigation (Mackey et al., 2013).

Some of the organic C recently fixed by photosynthesis in a terrestrial ecosystem is not rapidly returned to the atmosphere by respiration but remains in stabilized forms such as in biomass and soil. The biologically-mediated uptake and conversion of CO2 to inert, long-lived, C-containing materials is called ‘biosequestration’ (U.S. DOE, 2008). Biosequestration temporarily removes C from active cycling. Thus, ‘C sequestration’ can be defined as the uptake of C-containing substances and, in particular, CO2 into another reservoir with a longer residence time (IPCC, 2007). Any increase in the C content of a reservoir in an ecosystem might be referred to as sequestration as C is held in the reservoir and separated from other parts of the ecosystem (Powlson et al., 2011). However, it has become customary for the term C sequestration to imply a contribution to climate change mitigation. For this reason, C sequestration must slow or even reverse the increase in atmospheric concentration of CO2.

Thus, movement of C from one reservoir in the ecosystem to another should be appropriately termed accumulation whereas an additional transfer of C from the atmosphere into a reservoir should be termed sequestration as this process is a genuine contribution to climate change mitigation (Powlson et al., 2011).

The C sequestered in soil may in reality not always be locked up in a stable and inert form (Krna and Rapson, 2013). In fact, sequestered C may reside within a soil store or pool, to which C is constantly being added and removed. Effectively an individual C atom has a residence time within the C pool, with the whole pool continuously turning over at rates of up
to thousands of years (Campbell, 1967). However, there is a lack of consensus over the period
for which C has to be immobilized in soil before it is considered to be sequestered (Krna and
Rapson, 2013). For example, if C is to be usefully stored for climate change mitigation, it may
remain stored not just for 100 y, but probably for more than 10,000 y (Mackey et al., 2013).
Specifically, a ‘pulse’ or unit of CO₂ emitted to the atmosphere is only fully removed from
the atmosphere so that it no longer interacts with the climate system when it has been
completely dissolved in the deep ocean — a process requiring the concurrent dissolution of
carbonate from ocean sediments (about 5,000 to 10,000 y) and enhanced weathering of
silicate rocks (around 100,000 y) (Mackey et al., 2013). Thus, SOC sequestration requires that
C must persist for very long periods of time in soil by stabilization processes that reduce the
probability and, therefore, rate of SOC decomposition (Schmidt et al., 2011). The SOC
stabilization mechanisms possess, in particular, the ability to increase the residence time of a
given C atom within soil compared to a reference situation (Berhe and Kleber, 2013).
There is increasing evidence that not the intrinsic properties of SOC itself but rather
physicochemical and biological influences allow SOC to persist. For example, Courtier-
Murias et al. (2013) emphasized that the main mechanism by which soil C inputs are
stabilized and SOC accrues is the adsorption of microbial biomass and microbial by-products
on mineral surfaces rather than the physical and chemical protection of undecayed or partially
degraded organic structures. Organic amendments may increase more than previously thought
the microbial populations of the soil, which live, thrive, and die in close association with the
mineral surfaces. The joint physical-chemical mechanism of SOC stabilization may be
enhanced by the addition of organic materials relatively richer in compounds with molecular
structures and/or assemblies more resistant to decomposition (Courtier-Murias et al., 2013).
Thus, the association of SOC with minerals may be the most important factor in SOC
stabilization, and stability of SOC may increase with increase in soil depth, irrespective of
vegetation, soil type, and land use (Schrumpf et al., 2013). However, the reasons for the very long turnover times of SOC in subsoil horizons and its increases with increase in soil depth (common radiocarbon ages 1,000 y to >10,000 y) are not completely understood (Rumpel and Kögel-Knabner, 2011; Schmidt et al., 2011).

The alterations of the relative magnitude of soil C pools with different residence times potentially aide in SOC sequestration (Stockmann et al., 2013). This process may be enhanced by BC particles (biochar, charred biomass, charcoal, soot) as they contain C-based compounds with prolonged residence time such as condensed aromatic structures. Thus, SOC sequestration in agroecosystems potentially occurs through pyrolysis of biotically captured organic material to form biochar, which is then moved to the soil via anthropic means (Krna and Rapson, 2013). However, a biochar deemed ‘stable’ in one soil is not guaranteed to be so if deposited in another soil with different micro-climatic or environmental conditions.

Specifically, the molecular composition of organic matter (OM) and its decomposition pathways are related and coevolve over time as the ecosystem adapts and evolves to the ever changing biological, physical, and chemical conditions surrounding SOM (Eastwood et al., 2011). Consequently, soil microorganisms can only use those substrates that they are adapted to (Berhe and Kleber, 2013). Thus, at least a portion of added biochar may be decomposed by a soil microbial community when it has evolved in the presence of biochar-like materials as was indicated by an incubation experiment using two $^{13}$C-labelled biochars produced from wheat (Triticum ssp.) or eucalypt (Eucalyptus ssp.) shoots (Farrell et al., 2013). Further, soil bacteria considered well-adapted for aromatic-C degradation were capable of rapidly metabolizing pyrogenic organic matter (PyOM) made from ponderosa pine (Pinus ponderosa Douglas ex C.Lawson) wood (Santos et al., 2012). However, the question is how long-term (millennial) stability of biochar is possible despite the decomposition and mineralization capabilities of soil microorganisms. Importantly, as the molecular structure of a SOC
compound alone does not control the residence time and its stability, it is debatable how successful a climate change mitigation strategy based on SOC sequestration by application of biochar might be (Schmidt et al., 2011; Krna and Rapson, 2013). In conclusion, biochar may only contribute to climate change mitigation by SOC sequestration if biochar-C itself is stable in a soil for long periods of time (millennia) and/or the interaction of biochar with the soil results for long periods of time (millennia) in a net additional C storage in the SOC pool relative to the atmospheric CO₂ pool. Supposedly, deep soil horizons are the most important compartments where SOC sequestration for climate change mitigation by biochar application may occur.

3 Effects of Biochar Application on Soil Organic Carbon in Agricultural Soils

Soil application of biochar can directly and indirectly affect SOC dynamics as summarized in Table 2. Indirectly, biochar may alter soil C inputs by affecting net primary production (NPP) and, thus, the amount of biomass that may remain in agroecosystems. Higher belowground NPP and increased root-derived C inputs following biochar application may particularly result in an increase in SOC (Sohi et al., 2010). Directly, biochar may enhance SOC stabilization processes and contribute to SOC sequestration by increasing the mean residence time (MRT) of SOC (i.e., the mean time that a SOC-C atom spends in soil).

The MRT of biochar-C is thought by some to be in the range of millennia (Glaser and Birk, 2013). However, biochar longevity in soil is less well known and needs to be verified for a range of biochars and sites. For example, the MRTs of BC in field experiments ranged from about 8 y for BC produced by burning of forest trees during slash-and-burn agricultural practices (Nguyen et al., 2008) to 3600 y for BC produced from prunings of old mango (Mangifera indica L.) trees (Major et al., 2010). Only 7 among 311 primary research papers reviewed by Gurwick et al. (2013) reported field investigations of biochar stability in soil. Modelling approaches based on data from pyrogenic C degradation studies indicated that
pyrogenic C cannot be assumed to persist in soil for millennia (Singh et al., 2012).

Specifically, differences in climatic conditions may affect biochar longevity. For example, chemical and/or biological mineralization of natural chars produced from wood during bushfires was slower under Mediterranean compared to temperate climates in Australia (McBeath et al., 2013). In summary, assuming that biochar persists in soil 100 y or more is not supported by the very few data available to evaluate the in situ stability of biochar (Gurwick et al., 2013).

3.1 Biochar Effects on Plant Biomass Production

The amount of plant biomass produced, the exudation of C from plant roots and C transfer from plants to root symbionts are major determinants of soil C input in agroecosystems (Ciais et al., 2010). Thus, soil application of biochar causing an increase in photosynthetic C fixation, and in plant and root-derived soil C inputs may indirectly enhance the amount of CO$_2$ that is stored as SOC. For example, Oguntunde et al. (2004) reported higher grain yield for maize (Zea mays subsp. mays) growing in unfertilized charcoal kiln sites soils compared to maize growing in unfertilized control soils. However, a review by Mukherjee and Lal (2014) indicated that reductions in crop yield are also observed after biochar is applied together with fertilizer to soil.

The variability in agricultural productivity following biochar application is high, but the impacts of soil properties, climatic conditions and plant species for the yield response are less well known (Blackwell et al., 2009). The majority of biochar studies have been undertaken in tropical and subtropical regions, and extrapolation of biochar effects on yield in temperate regions is unclear (Jeffery et al., 2011). Further, the majority of data about the effects of biochar application on crop productivity have been published since 2010, and some are potentially biased due to highly skewed feedstock preferences and the fact that studies showing no significant effects are often not considered for publication (Jeffery et al., 2011;
Spokas et al., 2012; Liu et al., 2013). In particular, there are only a few studies monitoring crop yield responses (i) after soil addition of non-hardwood and crop residue biochars, (ii) produced with advanced pyrolysis systems, (iii) for medium-term to long-term (for longer than 2 y), (iv) in temperate regions, and (v) by comparing to un-treated controls in field experiments (Biederman and Harpole, 2013; Atkinson et al., 2010; Jeffery et al., 2011; Spokas et al., 2012; Liu et al., 2013). Thus, the relationship between biochar and crop productivity is not comprehensively studied in well-designed field experiments and, therefore, is poorly understood. The magnitude and relative importance of the mechanisms affecting crop productivity depend on the slow process of biological, chemical and physical modification of biochar in soil. It has been hypothesized that reduction in soil acidity by increase in soil pH (“liming effect”), and improvements in nutrient availability, cation exchange capacity (CEC), soil field capacity and habitat for soil microorganisms are major reasons for productivity improvements. Some recently published meta-analyses will be briefly discussed in the following section.

Results of a meta-analysis of 16 short-term field and pot experiments (>90% of the studies ran for only one growing season) indicate that biochar application to soils without fertilizer co-addition have small positive effects on crop productivity (both harvested yields and aboveground biomass production) with an increase of about 10% (Jeffery et al., 2011). Especially, biochar made from wood, paper pulp, woodchips and poultry (Gallus domesticus) litter had positive effects on crop productivity. The main mechanisms for yield increases discussed were a liming effect and an improved water holding capacity of the soil, along with improved crop nutrient availability (Jeffery et al., 2011).

The meta-analysis by Jeffery et al. (2011) was recently updated by a meta-analysis adding 87 more studies and 703 more pairs of data on yield and aboveground biomass (crop productivity) response to biochar in field and pot experiments (Liu et al., 2013). However,
studies without the co-addition of fertilizer were not analyzed separately. About half of the
experiments were conducted over 1 y and the longest over 4 y. On average, crop productivity
increased by 11%, consistent with the results of Jeffery et al. (2011). However, the crop
productivity response was only 9.1% in field experiments vs. 11.1% in pot experiments.
Further, crop productivity in the field increased only in years one and two following biochar
application but not in years three and four. Much higher crop productivity increase was
observed in sandy than in finer textured soils, and in acid (pH<5.0) than in neutral soils.
Manure, wood and crop residue biochars resulted in crop productivity increases by 29.0%,
12.1% and 2.6%, respectively. In contrast, applying municipal waste biochar resulted in crop
productivity reductions by 12.8%. Wood residue biochars produced at temperatures of >350
°C and crop residue biochars produced at >550 °C showed greater crop productivity
increases, respectively. Productivity was also increased for manure biochar produced at
temperatures in a range of 350-550 °C. While non-alkaline biochars (pH<7.0) caused a
reduction in crop productivity, applying alkaline biochars (pH>7.0) resulted in crop
productivity increases. The crop productivity responses were not proportional to biochar
application rates up to 20-40 Mg ha⁻¹, but the productivity increases diminished at application
rates >40 Mg ha⁻¹. Higher responses were observed in acid and sandy textured soils, and for
dry land crops than for irrigated crops or paddy rice. Thus, Liu et al. (2013) concluded that
both a liming and an aggregating/moistening effect contribute to crop productivity increases
after biochar application. However, long-term field studies in a wide range of agricultural
conditions would be needed to monitor the persistence of biochar effects. Further, crop
productivity responses observed in pot and greenhouse experiments must be critically
assessed with regard to the applicability of observations to those in field experiments (Liu et
al., 2013).
Another recent meta-analysis evaluated the short-term effects (average length 113.4 days) of ‘biochar’ on plant productivity and nutrient cycling by analyzing results from 346 experiments (Biederman and Harpole, 2013). The ‘biochar’ in this study included char, BC, charcoal and ‘agchar’. On average, aboveground biomass increased in ‘biochar’-treated soils by about 25% independently of fertilization as there was limited evidence of a synergistic effect when both ‘biochar’ and fertilizer are applied. ‘Biochars’ from grass and manure/sewage, in particular, increased aboveground productivity. However, effects on overall plant productivity of perennial species were limited compared to those of annual species probably due to higher sensitivity of perennials to toxic ‘biochar’ compounds (Biederman and Harpole, 2013). Further, belowground productivity of annual plants responded positively to ‘biochar’ indicating root-derived C inputs and, thus, the SOC pool may also increase (Ciais et al., 2010). In contrast, belowground productivity of perennials such as native and naturalized grasses and forbs, and forage crops did not respond. Also, the plant aboveground : belowground biomass ratios did not change. Thus, ‘biochar’ may enhance the belowground C inputs in the short-term but only for those of annual plants (Biederman and Harpole, 2013). However, the robustness of the conclusions drawn by this meta-analysis with regard to ‘biochar’ effects on crop productivity has been questioned as those were often based on very weak statistical relationships (Jeffery et al., 2014).

Previous meta-analyses have been hindered by missing and/or inconsistent reporting of soil properties, biochar properties, or other factors which may explain observed plant response (Crane-Droesch et al., 2013). This may have led to misleading and/or imprecise conclusions stemming respectively from correlation between grouping factors and underlying causes, and low effective sample sizes caused by dropping observations with missing covariate data. Thus, Crane-Droesch et al. (2013) employed statistical methods designed for problems with missing data, i.e., meta-analytical, missing data, and semiparametric statistical methods to...
explain heterogeneity in crop yield responses across different soils, biochars, and agricultural
management factors comparing data from 84 studies (365 crop yield response ratios). The
results were then used to estimate potential changes in yield across different soil
environments globally. Crane-Droesch et al. (2013) estimated an average crop yield increase
of approximately 10% for 3 Mg ha\(^{-1}\) of biochar addition in the first year after application but
variability in this response was high. Little evidence was found that plant response to biochar
is mediated by N additions to soil. Soil CEC and SOC content were strong predictors of yield
response, with low CEC and low SOC associated with positive response. The yield response
increased over time by approximately 7.0% and 12.3% percentage point relative increases in
crop yields in the second and fourth season after biochar application, respectively, compared
to non-biochar controls. High soil clay content and low soil pH were reportedly weaker
predictors of higher yield response. No biochar parameters in the dataset—biochar pH,
percentage C content, or temperature of pyrolysis—were significant predictors of yield
impacts. Further, the crop yield response was higher in animal-derived biochars but the result
was not significant. Globally, the largest potential yield increases to biochar were observed in
areas with highly weathered soils, such as those characterizing much of the humid tropics.
Richer soils characterizing much of the world’s important agricultural areas appear to be less
likely to benefit from biochar (Crane-Droesch et al., 2013).

Some examples of changes in soil properties possibly responsible for crop yield responses to
biochar/charcoal application mainly without co-addition of fertilizer are summarized in the
following section.

3.1.1 Liming Effect

The moderation in aluminum (Al) toxicity may be the reason why biochar application has
particularly positive effects on productivity in tropical and irrigated systems on highly
weathered and acid soils with low-activity clays (Blackwell et al., 2009). The greatest positive
crop yield responses to biochar were seen in acidic and neutral pH soils (Jeffery et al., 2011; Liu et al., 2013). The reasons for yield increases on acid soils following application of bark charcoal produced from *Acacia mangium* Wild. without co-application of fertilizer were increases in soil pH, and alleviation of Al and possibly manganese (Mn) toxicity (Yamato et al., 2006). The alkaline biochars produced at higher pyrolysis temperature are more effective in supporting increases in biomass by improved growth conditions than acidic biochars presumably through increases in soil alkalinity (Biederman and Harpole, 2013). Specifically, the acid functional group concentration in biochars produced from the biomass of rice (*Oryza sativa* L.), Valley oak (*Quercus lobata* Née), Loblolly pine (*Pinus taeda* L.) and Florida gama grass (*Tripsacum floridanum* Porter ex Vasey) decreased with increasing peak pyrolysis temperature as more fused aromatic ring structures were produced and more volatile matter was lost (Mukherjee et al., 2011; Li et al., 2013). In addition, alkalinity and the form of alkalis may be affected by peak pyrolysis temperature as was suggested by Hossain et al. (2011) based on studies with biochar produced from wastewater sludge. While carbonates were major alkaline components in biochars produced from straws of canola (*Brassica campestris* L.), corn (*Zea mays* L.), soybean (*Glycine max* L.) and peanut (*Arachis hypogaea* L.) generated at high temperatures, organic anions contributed especially to alkalinity of biochars generated at lower peak pyrolysis temperature (Yuan et al., 2011). Thus, high temperature biochars may have a great potential to raise soil pH.

Toxic effects of available Al on crop root growth in acidic soils are reduced by biochar-induced soil pH increases (Chan and Xu 2009). As a result of reduced Al toxicity, roots are able to better and more effectively explore even the acid soils to absorb nutrients and water, and this trend may contribute to an increase in crop yield. Further, reduced concentrations of Al and iron (Fe) in the soil solution after ‘biochar’ application may also enhance the
availability of previously bound phosphorus (P) to plants in acid soils and, thus, improve the
harvest index (HI; Biederman and Harpole, 2013).

The liming effect of biochar may result in SOC accumulation similar to the effects of long-
term liming of agricultural soils (Fornara et al., 2011). For example, the net increase in SOC
to 23-cm soil depth in soils limed for almost 130 yr was up to 20 times greater than that in un-
limed soils. In particular, the greater biological activity in limed soils led to plant C inputs
being processed and incorporated effectively into resistant SOC pools associated with soil
minerals (Fornara et al., 2011). However, deeper soil depths were not studied which would
have been important to assess the long-term effects of liming on SOC sequestration.

Soil application of biochar may also result in neutral and negative yield responses. Some of the
responses may be explained by strong increases in soil pH affecting pH-sensitive plants
and/or exacerbating micronutrient deficiencies similar to effects of soil application of
charcoals (Glaser et al. 2002). For example, negative yield responses to biochar applications
may occur when increase in pH exacerbates micronutrient deficiencies and calcifuge plant
species are retarded by high calcium (Ca) levels (Chan and Xu 2009).

3.1.2 Cation Exchange Capacity and Nutrient Concentrations

Biochar may improve soil CEC as it is often characterized by high CEC values, probably due
to its negative surface charges and its high specific surface area as was shown for ponderosa
pine and tall fescue (Festuca arundinacea Schreb.) derived BC (biochar) and for biochar
produced from crop residues (Keiluweit et al., 2010; Yuan et al., 2011). Thus, incorporation of
biochar into soil often but not necessarily increases CEC (Manyà, 2012). Depending on its
persistence, biochar may affect crop productivity in the long-term by providing chemically
active surfaces that modify the dynamics of soil nutrients or catalyze useful reactions, and by
modifying soil physical properties that benefit nutrient retention and acquisition (Sohi et al.,
2009). The improved plant nutrient availability by increased CEC may contribute to crop
yield increases. However, temporal changes in crop productivity through modification of soil chemistry by biochar are variable (Sohi et al., 2009). These changes depend on the mineral nutrient content of fresh biochar and complex physicochemical reactions of biochar with soil particles due to weathering processes as well as associated increases in CEC over time (Spokas et al., 2012). As CEC is indicative of the capacity to retain essential nutrient cations in plant available form and of minimizing leaching loss, increases in CEC are often regarded as key factors for crop productivity improvements following biochar application. However, CEC increases may not always be observed as, for example, no changes in CEC in soil of meager fertility characteristics were observed after application of pecan (Carya illinoinensis) shell-based biochar but soil fertility improved (Novak et al., 2009a). Otherwise, the soil fauna may also play a role in enhancing biochar effects on soil fertility. For example, activity of the earthworm Pontoscolex corethrurus potentially increases fertility in soils of the tropics under slash-and-burn practices by deposition of a reworked charcoal/soil mixture on the soil surface which favors the formation of stable ‘humus’ (Ponge et al., 2006).

Immediate beneficial effects of charcoal additions on crop productivity in tropical soils may result from increase in availabilities of Ca, Cu, K, P and Zn as was shown for secondary forest charcoal (Lehmann et al., 2003). In particular, poultry litter biochar may result in strong increases in soil extractable P (Novak et al., 2009b). Otherwise, lower crop N and Mg uptakes after charcoal addition have also been observed which may cause decrease in crop growth. However, moderate charcoal additions are not a direct supplier of plant nutrients in the long-term but other effects of charcoal on nutrient availability appear to be more important to crop yield responses (Glaser et al., 2002). For example, the reduced leaching loss by increased P and K retention on ‘biochar’s’ large and porous surface may contribute to increased soil P and K, and increased plant productivity and crop yield (Biederman and Harpole, 2013). Biochar application may also save nutrients which would have to be otherwise applied with fertilizer.
(Chan and Xu, 2009). Furthermore, the soil fauna may contribute to improved nutrient uptake efficiency. For example, the earthworm *P. corethrurus* contributed to increased yields of yardlong beans (*Vigna unguiculata* subsp. sesqui-pedalis (L.) Verdc.) after soil addition of charcoal with P-rich cassava (*Manihot esculenta* Crantz) peels (Topoliantz et al., 2005).

### 3.1.3 Soil Moisture and Physical Properties

Only limited field data are available on changes in soil physical properties of biochar-soil mixtures (Mukherjee and Lal, 2014). Less well known are, in particular, biochar effects on changes in soil aggregation and penetration resistance in field experiments. However, the effects of biochar addition on soil physical properties depend on biochar properties. For example, adding ground pecan (*Carya illinoinensis*) shells pyrolyzed at 700 ºC to a Norfolk sandy loam with poor physical characteristics reduced soil strength and improved soil water content during free drainage but neither improved aggregation nor the infiltration rate (Busscher et al., 2010). Supposedly, other biochar formulations would have been more effective in improving physical properties of the soil. Also, the water-holding capacity in this soil varied after applying biochars produced at temperatures from 250 ºC to 700 ºC from peanut (*Archis hypogaea*) hulls, pecan shells, poultry litter and switchgrass (*Panicum virgatum* L.; Novak et al., 2009b).

Changes in soil moisture retention may be among key factors in explaining positive biochar/charcoal effects on crop yield. However, experimental evidence for changes in soil water retention capacity following charcoal application is scanty (Glaser et al., 2002). Soils under charcoal kilns in Ghana had higher saturated hydraulic conductivity, higher total porosity and higher infiltration rates but lower bulk density than those under control (Oguntunde et al., 2004). These changes may result in increases in water retention and decreases in soil erosion and, thus, result in higher productivity of soils under charcoal kilns.
Amending topsoils with biochar can decrease bulk density and, thus, improve agronomic productivity but it is unclear whether a decrease in bulk density is relevant in the deeper soil profile (Mukherjee and Lal, 2013). Sometimes the improved agronomic productivity in biochar-amended soils has been attributed to increased surface area and porosity resulting in improved water retention capacity. Specifically, soil application of biochar with high specific surface area may cause a net increase in total soil-specific surface area which may improve soil-water retention and, thus, crop yield (Manyà, 2012). For example, Glaser et al. (2002) reported an increase in water holding capacity after charcoal addition possibly supported by improved soil aggregation. Also, increases in SOC after biochar application likely increase water availability, improve soil field capacity and conserve soil moisture (Atkinson et al., 2010). However, experimental evidence for biochar effects on soil-water retention is scanty as changes in plant-available soil water retention after biochar application are measured only sporadically (Manyà, 2012). Mukherjee and Lal (2014) suggested that soil moisture retention may only be improved by biochar application to coarse-textured soils.

3.1.4 Soil Organisms and other Potential Biochar-mediated Effects on Biomass Production

High-temperature ‘biochars’ were more effective at promoting aboveground productivity compared to those produced at lower temperatures possibly because the former contained less biologically-active compounds (Biederman and Harpole, 2013). Other soil biological mechanisms for yield responses following biochar application could not be assessed by meta-analysis (Jeffery et al., 2011). However, the soil fauna may play a role in enhancing biochar effects on soil fertility (Lehmann et al., 2011). In many studies, microbial biomass has been found to increase as a result of biochar additions (e.g., for ‘biochar’ Biederman and Harpole, 2013), with significant changes in microbial community composition and enzyme activities. Sorption phenomena, pH and physical properties of biochars such as pore structure, surface
area and mineral matter play important roles in determining how different biochars affect soil biota (Ameloot et al., 2013; Lehmann et al., 2011). Numerous biologically active compounds may be introduced into soil with biochar, and which may promote growth or produce toxic effects with regard to plant and, in particular, root growth. Sorption of allelopathic compounds on biochar is sometimes discussed as reason for enhanced root growth. However, the reasons for changes in root growth after biochar application are rarely well identified (Lehmann et al., 2011). Such knowledge is important for assessing biochar effects on SOC sequestration as root-derived C is the major input to SOC at deeper soil depths (Rasse et al., 2005).

Long-term toxic effects of biochar on organisms may be caused by bioaccumulation of persistent organic pollutants (POPs) such as polychlorinated biphenyls (PCBs) and dioxins adsorbed to biochar as indicated by a review on sorption of POPs on BC (Koelmans et al., 2006). Fresh biochar is a strong sorbent and may decrease the bioavailability, toxicity and mobility of organic pollutants and potentially reduce the efficacy of pesticides and herbicides (Smernik, 2009). The enhanced adsorptivity of biochar containing soils for organic contaminants indicated for studies with pine needle biochar (Chen et al., 2008), may affect the interaction of contaminants with plant growth and SOC. Altered rates and timing of seed germination, and interactions of biochar with compounds that affect plant and microbial growth are important determinants regarding their potential effects on yield. However, whether sorptive properties of aged biochars differ generally from those of fresh biochars is less well known. Also, the effectiveness of biochars on sorption of various organic/inorganic contaminants is uncertain (Ahmad et al., 2014).

It is likely that biochar-induced changes in soil microbial activity, community structure and functional diversity could impact crop yield (Jeffery et al., 2011). For example, soil microbial biomass may increase after ‘biochar’ application but may have variable effects on plant-
associated soil microorganisms (Biederman and Harpole, 2013). Changes in soil microbial
dynamics may contribute to higher nutrient availability after charcoal application (Glaser,
2007). The promotion of beneficial soil microorganisms by biochar may contribute to
improved fertilizer-use efficiency (Warnock et al., 2007). For example, a higher colonization
rate with arbuscular mycorrhizal fungi on corn roots was reported after application of charred
bark of Acacia mangium (Yamato et al., 2006). However, the direct effects of biochar on soil
microorganisms such as surface interactions with microbial cell walls or capsular materials,
and indirect effects resulting from changes in adsorption of OM and effects on plant growth
are less well known (Thies and Rillig, 2009). Changes in microorganism occurrence by
biochar and resulting direct effects on plant and, particularly, root growth are only beginning
to be explored (Lehmann et al., 2011). There is a scarcity of studies that have investigated
effects of biochars on microbial function in the rhizosphere. Further, it is unknown if there are
changes in rhizodeposition in response to biochar addition (Lehmann et al., 2011).
Biochar may moderate the environmental fate of pesticides by altering their adsorption and
desorption characteristics, and altering pesticide biodegradation and efficacy. For example,
Loganathan et al. (2009) reported that the bioavailability of atrazine [6-chloro-N-ethyl-N'-(1-
methylethyl)-1,3,5-triazine-2,4-diamine] was reduced in soil amended with wheat (Triticum
aestivum L.) straw char. Further, Pinus radiata (D. Don) wood charcoal addition to a forest
plantation soil with low SOC concentration has been shown to enhance the sorption of
terbutylazine (N2–tert-butyl-6-chloro-N4–ethyl-1,3,5-triazine-2,4-diamine; Wang et al.,
2010). In particular, weed control in biochar-amended soils may prove more difficult as pre-
emergent herbicides may be less effective (Kookana et al., 2011). How the interaction of soil
biochar with pesticides alters C inputs from plants into the soil is not known but needs to be
studied as biochar may persist for long periods of time in the soil and affect the efficiency of
pesticides. For example, mixed Fraxinus excelsior L., Fagus sylvatica L. and Quercus robur
L. biochar aged in the field for 2 y did not apparently differ in sorptive properties as it had the similar effect on sorption and mineralization of simazine (6-chloro-N,N'-diethyl-1,3,5-triazine-2,4-diamine) as did the fresh biochar (Jones et al., 2011a).

In summary, the relationship between biochar, biomass production and SOC is poorly understood. The interactions between biochar, soil organisms and biomass production must be investigated over long time-scales as, for example, biochar’s yield benefits may significantly increase over time (Crane-Droesch et al., 2013; Ameloot et al., 2013). Whether root-derived soil C inputs increase in response to biochar is less well known but such knowledge is needed to evaluate indirect effects of biochar on SOC accumulation and, in particular, on C sequestration. The main reasons for reported yield increases after biochar application may be the liming effect and an improved soil water holding capacity along with improved plant nutrient availability (CEC), i.e., P and K. Biochar properties such as percentage C, pyrolysis temperature, or pH may be poorly associated with yield response ratio (Crane-Droesch et al., 2013). However, most field studies have been conducted in often highly weathered and relatively infertile soils of tropical latitudes in which the largest potential yield increases may occur. In contrast, response on inherently fertile soils, characterizing much of the world’s important agricultural areas, may be less and these soils may not benefit from application of biochar. The longevity of biochar effects on yield is generally uncertain as well-designed field studies are hitherto short-term (Liu et al., 2013). Further, to benefit from positive biochar effects on crops, it is critically important to identify the mechanisms behind often observed but less reported negative yield responses, and also in relation to the application rate (Mukherjee and Lal, 2014; Spokas et al., 2012). Explaining mechanisms by which different biochars influence yield responses remains to be a researchable priority (Crane-Droesch et al., 2013).

3.2 Biochar Effects on Soil Organic Carbon
The inherent biochemical recalcitrance of charcoal may contribute to the stabilization of the SOC pool, especially in fire-dependent or prone ecosystems (Krull et al. 2006; 2008). However, little is known about the effects of biochar on the SOC balance as was shown for glucose-derived and yeast-derived biochars (Steinbeiss et al., 2009). In contrast to the effects on crop productivity, biochar effects on SOC have not been assessed by meta-analyses. Biochar may enhance SOC sequestration due to intrinsic stability of some biochar components but may also interact with the decomposition of specific SOC fractions. For climate change mitigation, useful SOC sequestration in agricultural soils occurs when application of biochar results in a net increase in the SOC pool relative to the atmospheric CO₂ pool in a specified area for long periods of time (millennia). However, biochar losses occur in the long-term through decomposition, degradation, erosion and leaching. Degradation occurs abiotically (e.g., chemical oxidation, photooxidation, solubilization) and biotically (e.g., microbial incorporation, oxidative respiration) as was shown for biochars made from a range of biomass types (Zimmerman, 2010). A complete mass balance is needed, but often not available, to account for the fate of soil-applied biochar as was shown for studies with BC produced from prunings of mango trees (Major et al., 2010). Specifically, the contribution of the soil fauna to biotic degradation and biochar or charcoal stability is unclear (Ameloot et al., 2013; Topliantz and Ponge, 2005). For example, activity of P. corethrurus may stabilize charcoal-derived C in soil by favoring the formation of microaggregates within macroaggregates (Ponge et al., 2006). These aggregates contain protected occluded C and their amount increases by passage through the earthworm gut (Bossuyt et al., 2005). Thus, some earthworm species potentially enhance SOC sequestration in agricultural soils, especially after biochar application. The contribution of soil fauna other than earthworms (e.g., protozoa, nematodes, collembola, microarthropods and termites) to biochar stability and SOC needs, however, additional research (Ameloot et al., 2013).
In the short term (<3 y), soil application of ‘biochar’ resulted in an increase in total soil C (Biederman and Harpole, 2013). Thus, ‘biochar’ may contribute to the sequestration of soil C but effects on SOC sequestration are unknown as inorganic C is often not analyzed separately (Biederman, pers. comm.). Further, inorganic C added to soil with biochar carbonates may either be a net CO₂ sink or source depending on whether reaction with strong acids or carbonic acid occurs similar to those following addition of agricultural lime (Hamilton et al., 2007). However, it is unlikely that biochar carbonates are stable in soil and contribute directly to soil C sequestration over millennia MRTs similar to those of pedogenic carbonates (Lal and Kimble, 2000; Schlesinger, 2006). Biochar carbonates may be rapidly lost from soil similar to agricultural lime (Ameloot et al., 2013). For example, less than 3% of lime added every 3 to 4 years since 1990 remained in a grassland soil to 23-cm depth in the year 2005 (Fornara et al., 2011). However, the cations leached by lime dissolution and biochar mineralization in the topsoil may result in inorganic C sequestration by formation of secondary carbonates at deeper depths (Nordt et al., 2000).

No review or meta-analysis on effects of biochar on SOC sequestration in field experiments has been published until early 2014. Up to the year 2012, Gurwick et al. (2013) identified only seven field studies among 74 in total related to biochar stability, transport or fate in soil which estimated biochar decomposition rates in situ. MRTs ranged from between eight to >3,000 y but the reasons for this large variation remained unexplained. Thus, generalizing claims about positive effects of biochar on SOC sequestration for climate change mitigation by increasing MRT of SOC in agricultural soils are not supported by the available research data (Gurwick et al., 2012). Some examples for observations regarding the decomposition of biochar or charcoal in soil under laboratory and field conditions, the effects of application of biochar or charcoal on the SOC balance, and losses of biochar or charcoal by erosion, leaching, gaseous emissions are discussed in the following section.
3.2.1 Decomposition of Biochar

Biochar is subject to decomposition in most surface soils as it is thermodynamically unstable under the oxidative prevailing conditions (Macías and Arbestain, 2010). However, biochar residues in soil resulting in higher stocks of oxidized char residues, usually comprising of six fused aromatic rings substituted by carboxyl groups, may contribute to SOC sequestration. This was shown by Mao et al. (2012) for char generated by presettlement fires found in grassland-derived soils in the U.S. That some components of biochar and other combustion residues are relatively resistant to decomposition is well known from the persistence of soil charcoal and its suitability for dating and paleo-environmental reconstruction (Titiz and Sanford, 2007). For example, char in residues from forest fires may be up to 10,000 y old (Preston and Schmidt, 2006). However, combustion residues in soil can also be relatively modern. For example, radiocarbon ages of <50-400 y and a median age of 652 y have been reported for BC and charcoal in boreal forest soils, respectively (Ohlson et al., 2009; Kane et al., 2010). However, radiocarbon ages provide no quantitative information about the decomposition rate of biochar (Lehmann et al., 2009). Radiocarbon ages are only indicative of the average time elapsed since atmospheric CO₂ is fixed by photosynthesis in biomass which then forms feedstock for biochar. Additional information about the amount of biochar at deposition is needed to quantify the decay rate, but this information is generally not available (Sohi et al., 2009).

Little is known about the decomposition of biochar or generally BC under field conditions (Major et al., 2010). Laboratory incubations indicate that the formation of oxygen-containing functional groups is the major mechanism leading to BC mineralization involving biotic asides some abiotic oxidation processes. This was shown by Nguyen et al. (2010) for laboratory decomposition experiments with BC materials produced from corn residues and oak (Quercus ssp.) wood. Zimmerman (2010) concluded based on incubations with biochars
made from a range of biomass types that in a sample of biochar, the C that is lost first is most likely to be aliphatic and is closer to a particle’s external surfaces. Otherwise, the residual biochar-C is more likely to be either part of highly condensed aromatic structures or condensates within protective internal pores that are more abundant in biochars pyrolyzed at higher temperature (Zimmerman, 2010). However, the relative importance of the BC structure at the micro- and nanoscale, in comparison to the role of minerals (e.g., N and K) for BC mineralization is poorly understood (Nguyen et al., 2010). Further, adding glucose to a soil containing BC produced by charring perennial ryegrass (Lolium perenne L.) residues stimulated BC decomposition for a short period (Kuzyakov et al., 2009). This response indicates co-metabolic decomposition (Hamer et al., 2004). Thus, microorganisms do not depend on BC utilization as a C or energy source but microbial enzymes produced for decomposition of other substrates such as rhizodeposits may contribute to BC decomposition (Kuzyakov et al., 2009). The importance of this priming effect on BC decomposition in the field mediated by soil organisms is largely unknown (Ameloot et al., 2013). In conclusion, laboratory experiments with BC produced from corn stover residue and oak shavings indicate that rapid BC decomposition occurs under high and consistent incubation temperatures, and by (i) mixing with sand creating an oxygen-rich environment promoting rapid oxidation, (ii) the unavailability of BC-protecting mechanisms and (iii) significant amounts of an easily decomposable BC fraction (Nguyen and Lehmann, 2009).

Studying BC or biochar decomposition in soils is extremely challenging as the quantification methods are selective for different BC phases such as for highly condensed microscopic BC particles or for low-temperature biochars (Hammes and Abiven, 2013), but no single method for quantifying solely biochar-C in soils exists. According to a ring trial involving 12 BC reference materials and seven different methods, chemical oxidation with sodium hypochlorite (NaOCl) followed by $^{13}$C Nuclear Magnetic Resonance (NMR) spectroscopy
and elemental analysis is among the most promising methods for BC quantification in soils as there is little or none potential for non-BC bias (Hammes et al., 2007). However, there is insufficient data to compare the short- and long-term decomposition of biochar under different climates and in different soils (Sohi et al., 2009). Based on NMR spectroscopy using a molecular mixing model (Nelson and Baldock, 2005), Nguyen et al. (2008) showed that BC concentrations decreased rapidly to 30% of the antecedent level during the first 30 y of cultivation of soils for corn on land cleared from previous forest by fire in western Kenya. After 100 y of cultivation, however, only small changes have been observed in charcoal stocks at the land-use conversion chronosequence based on analyses of benzene polycarboxylic acids (BPCAs; Schneider et al., 2011). Aside from physical export, decomposition of pyrogenic C also contributes to the observed losses. However, all pyrogenic C fractions may be lost in similar amounts as no indication for a changing chemical quality of pyrogenic C was observed (Schneider et al., 2011). Major et al. (2010) observed that less than 3% of BC produced from mango prunings was lost as CO$_2$ after 2 y in a soil in Colombia. However, a large portion of applied BC may have been lost by surface runoff, and the attendant erosional processes.

The readily available phase in a biochar particle may be physically protected against decomposition by entrapment in a condensed and/or crystalline phase within the particle as was indicated by NMR studies of charred peat (Almendros et al., 2003). Fresh biochar, on the other hand, may lose C abiotically in the soil by surface oxidation. This was indicated by incubation experiments with charcoal produced from barely (Hordeum vulgare L.) roots (Bruun et al. (2008). The chemical stability of biochar depends on the aliphatic portion that is more readily decomposed and is less abundant in biochar produced at higher temperatures (Lehmann et al., 2009). Chemical stability depends also on the aromatic portion that is decomposed more slowly, forming surficial, oxygen-containing functional groups including
carboxylic acids similar to stable and abundant char residues found in some grassland-derived soils with a fire history (Mao et al., 2012). Also, BC samples from historical charcoal blast furnace sites were oxidized after 130 y in soil (Cheng et al., 2008). The adsorption of non-BC to those samples was less important for surface chemistry than oxidation. In contrast, adsorption of non-BC such as humified macromolecules and/or microbes may contribute to carboxylic and phenolic C forms surrounding the BC core and its surface even after thousands of years of decomposition in soil. This was shown for biomass-derived BC isolated from subsoils near Manaus, Brazil, where other organic material may have been buried together with charcoal (Lehmann et al., 2005). The core of biomass-derived BC particles was still highly aromatic and even resembled a fresh charcoal.

Large differences have been observed for BC losses from soil, the MRT of soil BC, turnover time (i.e., MRT of BC-C if soil BC is in steady state) and half-life (Czimczik and Masiello, 2007). For example, half-lives of less than 100 y for soil elemental carbon (EC) at fire-affected savanna soils and of up to 6,623 y for BC (charcoal) in soils of temperate rainforests have been reported, respectively (Bird et al., 1999; Preston and Schmidt, 2006). Further, long-term MRTs of 1,300 and 2,600 y were estimated for soil BC from two savannah regions, respectively, where steady-state conditions of natural char production and disappearance occurred over long periods of time (Lehmann et al., 2008). Based on analyzing modern and archived profile samples from a Russian steppe soil, Hammes et al. (2008) calculated a BC turnover time of only 293 y. Vasilyeva et al. (2011) reported that both quantity and quality of pyrogenic C in a Chernozem profile in Russia remain unchanged after 55 y of extreme OM depletion under fallow management. Clay microaggregation was apparently an important process for pyrogenic C stabilization. In contrast, the MRT of the physically unprotected free light fraction containing charcoal from soils under corn and tobacco (Nicotiana tabacum
L./rye (*Secale cereale* L.) cropping was only about 20 y (Murage et al., 2007). Thus, physical protection in soil may contribute to a reduction in soil BC losses. In conclusion, the loss of BC, biochar or charcoal by decomposition is highly variable and depends on (i) inherent chemical stability, (ii) particle size and physical structure, but also (iii) on protection from microbially-produced exoenzymes through soil physical structures (Zimmerman, 2010; Keiluweit et al., 2010; Nocentini et al., 2010). The BC may potentially be sequestered in the micro- and nano-C repository soil environment through both the physical entrapment by the action of metal oxides and OM-induced microaggregation, and through molecular-level associations (Solomon et al., 2012).

### 3.2.2 Biochar and Decomposition of Soil Organic Carbon

The effect of soil-applied biochar on the decomposition of native SOC is poorly understood. Biochar may enhance SOC aggregation and, thus, reduce C losses (Liu et al., 2013). For example, no enhanced SOC loss has been observed after addition of BC produced from mango prunings in a field study (Major et al., 2010). In contrast, charcoal inputs can increase microbial activity in boreal forest surface soils and strongly stimulate SOC loss through greater respiration or greater leaching of soluble compounds (Wardle et al., 2008). Results from laboratory incubations are also variable ranging from no significant effects of BC (i.e., charred residues of perennial ryegrass) on the decomposition of native SOC (Kuzyakov et al., 2009) to the stabilization of labile SOC after addition of *Eucalyptus salinga* wood biochar by interactive priming (Keith et al., 2011). Thus, effects of biochar on decomposition of native SOC needs to be studied for a range of biochars and agroecosystems.

### 3.2.3 Biochar Losses by Mixing, Erosion and Leaching

To fully account for the fate of soil-applied biochar and its interaction with SOC, downward movement of biochar into the mineral soil by mixing and leaching and beyond into aquifers,
and physical export from soil by wind and water must be determined aside from biochar losses by decomposition on the soil surface (Lehmann et al., 2009).

Biochar applied to the soil surface layer may illuviate into the mineral soil as was indicated by studies on the mobility of household-derived BC residues in peatlands (Leifeld et al., 2007). However, only a few studies have studied the process of downward movement and quantified BC over the whole soil profile. For example, BC contents in soils under tropical slash-and-burn agriculture ranged between 5.5% of SOC in 0 to 20 cm and 4.1% of SOC in 35 to 60 cm depth (Rumpel et al., 2006). BC moved also to deeper soil depths in a Russian steppe soil and in Russian Chernozems as the maximum profile BC concentrations have been observed between 30 and 50 cm depths (Rodionov et al., 2006; Hammes et al., 2008). Further, pyrogenic C was also physically transported down a Chernozem profile and accumulated in the deepest layer studied (70-80 cm; Vasilyeva et al., 2011). Downward migration of BC is attributed to bioturbation and leaching in arable lands in Germany where BC explained up to 50% of the SOC content at a depth of 87-114 cm (Brodowski et al., 2007). Translocation of BC to sub-soil may be promoted by oxidation processes which increase the water solubility of BC. This was indicated by studies with biochars made from a range of biomass types and by studies on BC in fire-affected permafrost soils at a forested catchment (Zimmerman, 2010; Guggenberger et al., 2008). The soluble BC transport may be favored by fragmentation and dissolution through oxidation of the condensed aromatic structures. Hockaday et al. (2007) reported indirect evidence for microbial dissolution of soil charcoal derived from burning of white pine (Pinus strobus L.) and hemlock (Tsuga canadensis L.). Thus, charcoal-derived structures mostly condensed aromatic ring structures could be identified in soil pore water. High vertical transportation rates have been reported for household-derived BC residues in peatland soil profiles, and related to large pore volumes and often saturated conditions (Leifeld et al., 2007). Thus, in deeper anaerobic peat soil horizons
long-term BC accumulation may occur as microbial activity is reduced under water-saturated conditions. Furthermore, deeper horizons often contain BC of a higher thermal stability, most likely soot (Leifeld et al., 2007). Similarly, Rumpel et al. (2008) reported that under tropical slash-and-burn agriculture long-term preservation of BC occurred mostly in the deepest minerals soil horizons up to 80-cm depth. Major et al. (2010) reported that 1% of mango pruning BC applied to a soil was mobilized by percolating water over 2 y after application. Further, relatively more dissolved organic carbon (DOC) than particulate organic carbon (POC) can be lost from BC. Thus, studies limited to surface horizons may miss the location of the most concentrated BC, where it contributes most to SOC, and may miss the importance of downward migration and stabilization of BC in deeper soil horizons (Hammes et al., 2008).

Erosion and surface runoff can be among the major processes resulting in the loss of surface applied biochar. For example, up to 50% of surface applied BC produced from mango prunings was lost from field plots by surface runoff during intense rain events (Major et al., 2010). Due to low bulk density and the floating behavior, pyrogenic C produced by burning of the perennial grass species Andropogon gayanus was preferentially removed by erosional processes compared to other SOC fractions (Rumpel et al., 2009). Charcoal may even be entirely exported from watersheds and enter water bodies (Jaffe et al., 2013). Once BC becomes a component of riverine C, it is easily exported to the ocean and finally buried in deeper ocean sediments (Masiello, 2004). Similar to geological C sequestration, the burial of biomass char in deep ocean sediments may isolate C from exchange with the atmosphere for centuries to millennia and, thus, contribute to climate change mitigation (Dufour, 2013).

3.2.4 Biochar and Greenhouse Gas Emissions from Soil

Applying biochar to agricultural soils may affect SOC sequestration by altering the greenhouse gas (GHG) balance. Indirectly, radiative forcing may be altered by changes in atmospheric CH₄, CO₂, N₂O, BC (soot), and ozone concentrations resulting in changes in
temperature and precipitation with possible feedbacks on the SOC balance. However, there is little information on indirect radiative forcing effects of biochar. Annual net emissions of CO$_2$, CH$_4$ and N$_2$O may be reduced by 1.8 Pg CO$_2$-C equivalent (CO$_2$-Ce) $\text{y}^{-1}$ (1 Pg = 10$^{15}$ g), and total emissions by 130 Pg CO$_2$-Ce over a century by implementing a sustainable biochar program globally (Woolf et al., 2010). Emission savings may arise indirectly from biochar application through (i) reduced need for fertilization due to enhanced fertilizer use efficiency, (ii) avoided conversion of natural ecosystems for agriculture as crop yield may be higher on biochar amended soil, (iii) reduced need for irrigation due to improved water-holding capacity, and (iv) reduced energy need for tillage by improved soil physical properties (Sohi et al., 2010).

The soil application of biochar may alter the surface albedo (i.e., the amount of solar radiation reflected back in space) but this effect is less well studied (Meyer et al., 2012). Reductions in surface albedo of biochar-amended soils may also have consequences for soil sensible heat flux, surface temperature and evaporation. For example, Genesio et al. (2012) reported that the reduced albedo of soils mixed with charcoal produced from coppiced woodlands resulted in increases in soil temperature associated to larger soil heat flux. This temperature increase may promote decomposition and, thus, result in SOC losses. But the impacts of soil warming on decomposition have not been fully resolved (Conant et al., 2011). Verheijen et al. (2013) reported that the surface application of pine biochar in a laboratory experiment strongly reduced soil surface albedo even at relatively low application rates. For a global-scale biochar application rate equivalent to 10 Mg ha$^{-1}$, the simulated reductions in negative radiative forcings (balance between negative radiative forcings from avoided CO$_2$ emissions and positive radiative forcings from reduced soil surface albedos) were 13 and 44% for croplands and 28 and 94% for grasslands, when incorporating biochar into the topsoil or applying it to the soil surface, respectively. Thus, it is important to include changes in soil surface albedo in
studies assessing the net climate change mitigation potential of biochar (Verheijen et al., 2013). Further, if a small percentage of biochar particles become airborne this could also result in a net warming impact similar to that of BC (Bond et al., 2013). Gao and Wu (2014) reported that biochars are often ground and sieved to various sizes such as those produced from slow pyrolysis of mallee (Eucalyptus ssp.) wood, leaf, and bark with the upper size limits in the range from 0.044 to 20 mm. The particulate matter (PM) with an aerodynamic diameter of <10 μm (PM$_{10}$) and, in particular, <2.5 μm (PM$_{2.5}$) in the ground biochars will stay in air for long periods of time and be easily transported far from the application site. Considering typical biochar application rates of 5–50 Mg ha$^{-1}$, the application of the biochar after extensive grinding poses a large potential for fine PM emission (i.e., 0.25–2.5 Mg PM$_{10}$ ha$^{-1}$ and 0.1–1 Mg PM$_{2.5}$ ha$^{-1}$). Among adverse impacts of biochar loss by fine PM emission form the application site is the potential pollution of neighboring residential zones and the unacceptable health risks to workers handling the biochar (Gao and Wu, 2014). However, no published research has examined the possible net warming impact of airborne biochar particles (Ernsting et al., 2011).

The interpretation of lab incubation data on GHG fluxes and extrapolation of results to the field scale is challenging (Scheer et al. 2011). During incubation studies, GHG emissions from agricultural soil amended with biochar produced from corn stover, peanut hulls, macadamia (Macadamia ssp.) nut shells, wood chips, and turkey (Meleagris ssp.) manure plus wood chips varied widely depending on the properties of the biochar, soil type, land use and climate (Spokas and Reicosky, 2009). Only a small number of studies have assessed the direct influence of biochar on soil GHG emissions in field experiments (Gurwick et al., 2012; 2013). For example, increases, decreases and negligible effects on soil GHG emissions following application of a range of biochar types have been observed in which some soils received also fertilizer (Castaldi et al., 2011; Zhang et al., 2012; Case et al., 2013).
biochar may emit ethylene, a plant hormone which also inhibits soil microbial processes. Thus, ethylene in biochar-amended soils may contribute to GHG reductions but durations and temporal trends of those effects are uncertain (Spokas et al., 2010). However, exposure of fresh biochar made from Douglas fir [Pseudotsuga menziesii (Mirb.) Franco] chips and from hazelnut (Corylus ssp.) shells to an oxidizing environment for 3 months degassed or oxidized the entire amount of ethylene (Fulton et al., 2013). In conclusion, the specific mechanisms governing the responses of soil GHG fluxes to biochar addition are not clearly understood (Mukherjee and Lal, 2013).

**Carbon dioxide**

Lab incubations indicate that an initial increase in CO$_2$ after adding biochar to soil may come equally from breakdown of organic C and the release of inorganic C contained in biochar. This short-term release may be negligible for SOC sequestration as, for example, only about 0.1% of the C in mixed hardwood derived biochar made from Fraxinus excelsior L., Fagus sylvatica L. and Quercus robur L. was released in total (Jones et al., 2011b). During a short-term field study, no changes in CO$_2$ emissions have been observed after application of biochar which was a by-product of birch (Betula ssp.) charcoal production probably as biochar effects needed a longer time to develop or higher biochar application rates would have been required (Karhu et al., 2011). The labile content of biochar may be the reason for increased CO$_2$ emissions as was shown for a calcareous and infertile soil amended with biochar produced from wheat straw (Zhang et al., 2012). However, this effect may be a transient and decrease when labile biochar-C is no longer readily available (Zimmerman et al., 2011). In the long-term, increased belowground NPP after biochar application are probably causing increased CO$_2$ emissions (Major et al., 2010). Suppression of soil CO$_2$ emissions observed over 2 y in a bioenergy crops system after application of a biochar produced from thinnings of hardwood trees [oak, cherry (Prunus ssp.) and ash (Fraxinus ssp.)] may be due to a combined effect of
reduced enzymatic activity, the increased carbon-use efficiency from the co-location of soil microbes, SOM and nutrients and the precipitation of CO₂ onto the biochar surface (Case et al., 2013). However, the mechanism of GHG sorption/desorption on biochar may have only small effects on GHG fluxes as indicated by incubation studies with a range of different biochar types (Spokas and Reicosky, 2009). Further, it is unclear whether the long-term CO₂ balance of soils is affected by reduced or enhanced decomposition (negative or positive priming effect) of SOC sometimes observed initially after biochar addition, for example, for biochar produced from wood (Keith et al., 2011). Also, based on meta-analysis of eight studies assessing mineralization of ¹⁴C or ¹³C-labelled biochar, Ameloot et al. (2013) concluded that how biochar mineralization rates may change over years and decades remains largely unknown.

In summary, short-term increase in soil CO₂ emissions may occur after biochar addition but the long-term effects are uncertain.

**Methane**

Similar to the effects of biochar addition on the soil CO₂ flux, responses of CH₄ fluxes to biochar in field experiments vary and mechanisms are also poorly understood (Van Zwieten et al., 2009). For example, improved soil aeration and porosity after soil application of a by-product of birch charcoal production may be reasons for reduced CH₄ emissions observed in a short-term field study either due to a decrease in methanogenesis, increase in CH₄ oxidation or both (Karhu et al., 2011). Otherwise, CH₄ emissions increased weakly in soils amended with wheat straw biochar under corn and strongly under rice (Oryza sativa L.) cultivation, respectively, during the whole growing season but the reasons remain unknown (Zhang et al., 2010; 2012).
In summary, some agricultural soils may change from a net \( \text{CH}_4 \) sink into a net \( \text{CH}_4 \) source by the addition of some biochars when the \( \text{CH}_4 \) production increases and/or the \( \text{CH}_4 \) oxidation by methanotrophs decreases (Mukherjee and Lal, 2013).

**Nitrous oxide**

A combination of biotic and abiotic factors may be involved in effects of biochar on \( \text{N}_2\text{O} \) emissions from soil (Van Zwieten et al., 2009). Based mainly on prior knowledge of the requirements of nitrifiers and denitrifiers, proposed effects suppressing \( \text{N}_2\text{O} \) emissions include (i) enhanced soil aeration (reduced soil moisture) inhibiting denitrification due to more oxygen being present, (ii) labile C in the biochar promoting complete denitrification, i.e., dinitrogen (N\(_2\)) formation, (iii) the elevated pH of the biochar creating an environment where \( \text{N}_2\text{O} \) reductase activity is enhanced thus promoting \( \text{N}_2 \) formation and higher \( \text{N}_2/\text{N}_2\text{O} \) ratios, and (iv) a reduction in the inorganic-N pool available for the nitrifiers and/or denitrifiers that produce \( \text{N}_2\text{O} \), as a result of \( \text{NH}_4^+ \) and/or \( \text{NO}_3^- \) adsorption, greater plant growth, \( \text{NH}_3 \) volatilisation loss, or immobilisation of N (Clough et al., 2013). Further, Cayuela et al. (2013a) proposed that biochar facilitates the transfer of electrons to soil denitrifying microorganisms, which together with its liming effect would promote the reduction of \( \text{N}_2\text{O} \) to \( \text{N}_2 \). The quinone-hydroquinone moieties and/or conjugated \( \pi \)-electron systems associated with condensed aromatic (sub-) structures of biochar may be involved in this electron shuttling (Klüpfel et al., 2014). Otherwise, increases in \( \text{N}_2\text{O} \) emissions have been attributed to (i) the release of biochar embodied-N or priming effects on SOM following biochar addition, (ii) biochar increasing soil water content and improving conditions for denitrification, and (iii) biochar providing inorganic-N and/or C substrate for microbes. However, rigorous field experiments to test the proposed mechanisms are lacking (Clough et al., 2013).
Biochar interactions with \( \text{N}_2 \text{O} \) emissions may vary depending on soil type, land use, climate and biochar characteristics. For example, Karhu et al. (2011) observed no effect of biochar (i.e., by-product of birch charcoal production) on \( \text{N}_2 \text{O} \) emissions during the growing period associated with the highest \( \text{N}_2 \text{O} \) emissions probably as biochar effects needed more time to develop. Otherwise, even over 2 y the effects of application of mixed hardwood biochar on soil \( \text{N}_2 \text{O} \) emissions were negligible (Case et al., 2013). In contrast, adding wheat straw biochar to corn and rice soils in field experiments reduced \( \text{N}_2 \text{O} \) emissions (Zhang et al., 2010; 2012).

Cayuela et al. (2013b) performed a meta-analysis on the effects of ‘biochar’ (i.e., biochar, charcoal or BC) on soil \( \text{N}_2 \text{O} \) emissions, comparing 261 experimental treatments. Overall, ‘biochar’ reduced soil \( \text{N}_2 \text{O} \) emissions by 54% in laboratory and field studies. The ‘biochar’ feedstock, pyrolysis conditions and C/N ratio were key factors influencing emissions of \( \text{N}_2 \text{O} \) while a direct correlation occurred between the ‘biochar’ application rate and \( \text{N}_2 \text{O} \) emission reductions. Interactions between soil texture and ‘biochar’ and the chemical form of N fertilizer applied with ‘biochar’ also had a major influence on soil \( \text{N}_2 \text{O} \) emissions. However, there is still a significant lack in understanding of the key mechanisms which alter \( \text{N}_2 \text{O} \) emissions (Cayuela et al., 2013b).

In summary, most studies on \( \text{N}_2 \text{O} \) emissions from biochar-amended soils were short-term, and most laboratory experiments indicate emission reductions. However, long-term field studies are lacking, as is a mechanistic understanding of the biochar’s effects on soil \( \text{N}_2 \text{O} \) fluxes and, in particular, the role of ethylene on \( \text{N}_2 \text{O} \) emissions (Mukherjee and Lal, 2013; Spokas et al., 2010; Clough et al., 2013).

4 Research Needs

The effects of soil application of biochar on SOC sequestration, biomass, yield and other agronomic benefits are highly variable, and biochar-, plant- and site-specific. Less is known
how to engineer the pyrolysis process conditions to produce the desired biochar properties (e.g., fused aromatic ring structure) for SOC accumulation in agricultural soils (Brewer et al., 2009). Before large-scale biochar commercialization is implemented, long-term field research is needed to optimize biochar systems targeted to maximize agronomic benefits (Sohi, 2012). Well-designed studies must report consistently biochar chemistry and soil characteristics (Biederman and Harpole, 2013). Aside evaluating ecological effects, an economic evaluation is needed as subsidies will be required for widespread biochar implementation but it is unclear if and how subsidies will be financed through the C markets or trading C credits (Ernsting et al., 2011). Thus, to get the most benefit from biochar application, environmental and social circumstances must both be considered (Abiven et al., 2014).

Little attention has been paid to potential unintended environmental effects of biochar (e.g., biomass and yield reduction, SOC loss, increase in harmful compounds, changes in radiative forcing) following soil application (Kookana et al., 2011). Biochar field trials have been conducted since 1980 but mostly in tropical and subtropical regions, and only recently have the field experiments been initiated elsewhere (Jeffery et al., 2011; Liu et al., 2013). Thus, a large number of long-term field studies are needed in all climatic regions and, in particular, in temperate regions. There is a paucity of data concerning biochar produced from feedstocks other than wood and crop residues, and from feedstocks produced by technologies other than pyrolysis (Sohi et al., 2010; Liu et al., 2013).

The maximum allowable amount of biochar that can be incorporated into soils for C offset purposes must be established (De Gryze et al., 2010). The scientific knowledge about fundamental mechanisms by which biochar affects SOC dynamics needs to be improved by studying: (i) contribution of biochar to fused aromatic ring structure of soil BC and SOC, (ii) functional interactions of biochar with soil fauna and microbial communities, (iii) surface interactions, (iv) nutrient use efficiency, (v) soil physical effects, (vi) fate of biochar in the
soil profile, watershed and agricultural landscape, (vii) effects on GHG emissions, and (viii) plant physiological responses. Biochar studies must, in particular, include a systematic appreciation of different biochar-types and basic manipulative experiments that unambiguously identify the interactions between biochar and soil biota (Ameloot et al., 2013; Lehmann et al., 2011).

To assess the contribution of soil application of biochar to climate change mitigation by SOC sequestration, biochar and its effect on SOC must be studied in soil profiles and not only in surface soils as biochar and SOC sequestration may occur specifically in deeper soil horizons as indicated by very long SOC turnover times that increase with increase in soil depth (Schmidt et al., 2011). This includes studies on how biochar application affects organic C input to subsoils in dissolved form following preferential flow pathways, as aboveground or root litter and exudates along root channels and/or through bioturbation (Lorenz and Lal, 2005; Rumpel and Kögel-Knabner, 2011). In particular, biochar effects on microbial products in subsoils must be studied in detail as those contribute more to SOC at deeper soil depths than plant compounds (Schmidt et al., 2011; Courtier-Murias et al., 2013).

Also, biochar losses through physical export in dissolved, gaseous and particulate forms needs to be quantified for a range of sites to fully address the contribution of biochar to SOC sequestration in an agricultural landscape. Currently, only degradation/mineralization (abiotic and biotic) as mechanisms for biochar loss from soils have been subject of considerable research while studies on losses by erosion, illuviation, leaching/solubilization, volatilization and consumption by later fires are scanty (Saiz et al., 2014; Zimmerman and Gao, 2013).

Thus, a large number of studies of agricultural watersheds are needed to assess the environmental fate of biochar in agroecosystems as soil biochar may persist for long periods of time (Sohi et al., 2010). Further, biochar’s impact on downstream environments is less well known and field research on those effects is urgently needed. Studies are also needed to
strengthen the limited understanding of mechanisms by which biochar interacts with organisms (Biederman and Harpole, 2013). Modelling the coupled C and N (H₂O and P?) cycles in soil with and without biochar is essential to understanding the fundamental mechanisms through which biochar affects SOC and the impact on soil GHG emissions. Most importantly, improved methods of quantification of biochar in soil are needed, along with the standardization of the pyrolysis process (Sohi et al., 2010).

Finally, the hypothesis that biochar can only make a useful contribution to climate change mitigation in soils by affecting subsoil SOC must be tested by rigorous experiments accompanied by modeling studies. The hypothesis is based on the (i) long turnover times of SOC at depth, (ii) the reported preferential accumulation of BC at deeper depths, and (iii) the increase in stabilized SOC fractions with depth. If biochar itself should remain in soil for long periods of time (millennia) it must be removed from the soil surface and moved to deeper soil depths as it is otherwise prone to losses by decomposition and surface erosion. Over 60% of the global land area is composed of landscapes with >8% slope (Staub and Rosenzweig, 1992), and topsoil and with it SOC and biochar may be distributed laterally over the Earth’s surface by water, wind, and through gravity-driven diffusive mass transport (Berhe and Kleber, 2013). On the millennial time scale needed for a useful contribution to climate change mitigation (Mackey et al., 2013), the topsoil C including biochar buried at depositional sites may be lost by decomposition and leaching aside losses occurring during transport (Van Oost et al., 2012). Thus, biochar must be buried at eroding sites at deep soil depths to reduce the risk of erosion-induced C losses. The subsoil biochar may also contribute indirectly to SOC sequestration by affecting soil fertility and, thus, agronomic productivity. There is strong evidence that subsoil can contribute to more than two-thirds of the plant nutrition of N, P and K in agricultural soils of temperate regions (Kautz et al., 2013). The improved subsoil fertility following biochar addition to deeper soil depth can, thus, potentially enhance crop
productivity and soil C inputs. In conclusion, further evaluation of biochar effects on terrestrial C sequestration is needed before large quantities of biochar are applied to achieve useful goals for SOC sequestration (Post et al., 2012). Crucial will be to understand biochar and SOC dynamics at deeper soil depths, and how they can be managed for climate change mitigation.

5 Conclusions

Soil application of biochar results in a moderate increase in crop productivity (yield and aboveground biomass) for up to 2 y but it is unclear how long this enhancement will persist and whether soil C inputs may also increase. Not comprehensively assessed are, in particular, biochar effects on the SOC balance. However, biochar can be a useful contribution to climate change mitigation by SOC sequestration at deeper soil depths. Mitigation requires that biochar results in a net removal of C relative to the atmospheric CO$_2$ pool in soil for long periods of time (millennia) to reduce the interaction of atmospheric CO$_2$ with the climate. However, the importance of chemical recalcitrance vs. physical protection and interaction with soil minerals, and of processes at deeper soil depths for SOC stabilization is less well known. Specifically, biochar is not uniform but its properties vary widely and its fate and direct and indirect effects on SOC dynamics depends on feedstock, pyrolysis production systems and site properties. These factors must be studied by a large number of field studies accompanied by modeling before biochar can be commercialized on a large scale. It is critically important to identify the mechanisms behind unintended consequences of soil application of biochar such as reduction in biomass, yield and SOC, the effects of harmful compounds, impacts on downstream environments, and the net warming impact of airborne biochar particles.

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<tr>
<th>Pyrogenic carbon form</th>
<th>Definition</th>
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<td>Black carbon</td>
<td>Carbonaceous substance of pyrogenic origin resistant to thermal or chemical degradation by applying specific methods</td>
<td><em>Hammes</em> and <em>Abiven</em> (2013)</td>
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<tr>
<td>Biochar</td>
<td>Charcoal for which scientific consensus exists that soil application at a specific site is expected to substantially sequester carbon and concurrently improve soil functions while avoiding detrimental effects</td>
<td><em>Verheijen</em> et al. (2009)</td>
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<td>Pathway</td>
<td>Observed effect</td>
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<td>Direct</td>
<td>Increase in stabilized soil organic carbon fraction</td>
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<td>Physical entrapment by metal oxides; increase in microaggregation and molecular-level associations</td>
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<td>Deepening of soil organic carbon distribution</td>
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<td>Indirect</td>
<td>Higher crop yield and/or aboveground productivity</td>
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<td>Increased aggregation; increased soil alkalinity</td>
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<td>Effect</td>
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<td>Increased soil P and K concentrations; increased tissue K concentration</td>
<td><em>Biederman and Harpole</em> (2013)</td>
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<td>Increased cation exchange capacity</td>
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<td>Higher total and belowground productivity of annual plant species</td>
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<td>Reduced sensitivity to toxic biochar compounds</td>
<td><em>Biederman and Harpole</em> (2013)</td>
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