Chapter 15
Biochar in Soil for Climate Change Mitigation and Adaptation

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15.1 Introduction

Global environmental change, including land degradation, loss of biodiversity, changes in hydrology and changes in climate patterns resulting from enhanced anthropogenic emission of greenhouse gases, will have serious consequences for world food security, particularly affecting the more vulnerable socio-economic sectors (Ericksen et al. 2009; Lal 2010). The World Bank suggests that at least a doubling of cereal yields and a 75% increase in meat production by 2030 are required to maintain the current level of nutrition globally (Fresco 2009). This poses a quandary. To significantly increase food production when large areas of agricultural lands will be adversely affected by climate change or converted into

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forestry for C sinks may not be possible unless new technologies and sustainable practices are rapidly adopted. The application of biochar to agricultural soils may play a crucial role in global climate change mitigation through the reduction of greenhouse gas production and the sequestering of atmospheric carbon in soils (Gaunt and Cowie 2009; Lehmann 2007; McCarl et al. 2009; McHenry 2009; Read 2009). The agronomic benefits of biochar in soils (Chan et al. 2007; Steiner et al. 2008a, b) could assist in the adaptation of agriculture to meet rising demands for food and fibre. Furthermore, improving soil health with biochar application may increase resilience of agricultural systems and enable the continuation of farming on marginal lands. Application of biochar to soil has been shown to have many advantages including enhanced soil health characteristics, reduced metal contamination risks and consequently increased plant growth (Chan et al. 2007; Namgay et al. 2010; Reichenauer et al. 2009); as well as reduced greenhouse gas emissions from soil (Singh et al. 2010a; Van Zwieten et al. 2009; Yanai et al. 2007).

The competing and often conflicting demands of land use primarily stem from growing populations requiring housing and food, coupled with community desires for greater allocation of land to ecological reserves and the increasing production of energy crops to displace greenhouse gas emitting fossil fuels (Koomen et al. 2005; Simon and Wiegmann 2009). In addition to the challenge of a changing climate, the increasing claim for this scarce land use resource will force the necessity for greater productivity from less land, meaning farmers will need to undertake activities that result in significant yield increases. Land managers from more developed countries have historically had greater access to technological innovations and training, thereby improving the productivity of agricultural systems compared with those from developing countries. With the escalating effects of climate change technological adaptation will become increasingly vital to sustainably augment production systems globally (Bryan et al. 2009; Jones and Thornton 2009).

"Black carbon" (BC), a heterogeneous mix of carbonaceous materials formed from the incomplete combustion of biomass (Hammes et al. 2008; Schmidt and Noack 2000), is found in the most stable pool of soil organic carbon (SOC) (i.e. the component that resists microbial decomposition and mineralisation to CO_2). This BC may be derived from natural events such as biomass burning in wildfires (producing charcoal, consisting of partly charred organic matter through to completely carbonised submicron particles of soot) or through human activities (referred to as biochar).

Biochar can be manufactured through the pyrolysis of biomass (Lehmann and Rondon 2006), which condenses aliphatic carbon into more stable aromatic carbon, while releasing combustible gases (H_2, CH_4, CO) that can be used to heat the kiln with surplus for bioenergy. Rudimentary biochar production systems have been used for over 2,000 years, and when applied to soils biochar has demonstrated sustained productivity increases. A well-known example of ancient soil amendment with charcoal is the Terra Preta – dark earth-soils of the Amazon. These low fertility tropical soils were amended in pre-Columbian times by indigenous Amerindians through the addition of carbonised organic matter, believed to be from their cooking hearths (Glaser et al. 2001).

Modern biochar production uses a range of technologies including fast pyrolysis, gasification and/or carbonisation (Bridgwater 2003). These processes can be applied at different scales from small cooking stoves often used in developing countries through to more advanced industrial systems which include full gas recovery for integrated bioenergy production (Brown 2009). The sustainability credentials of each of these systems including efficiency of resource utilisation, emissions control, life cycle greenhouse gas balance and environmental sustainability need to be assessed on a case-by-case basis. The production process will influence the properties of the biochar and therefore the way it behaves and interacts in a soil (Downie et al. 2009; Glaser et al. 2002; Joseph et al. 2010; Novak et al. 2009; Singh et al. 2010a). To date agronomic benefits from biochar application have been demonstrated for biochars produced from a limited range of production systems (mainly small industrial scale, demonstration-level, pyrolysis or gasification units involving co-production of biochar, bio-oil and/or syngas) and applied to limited soil/plant systems. However, further research is required to quantify the impacts of biochar produced from a range of small (including mobile units) to large industrial scale biochar production systems and conditions, and then applied to contrasting soil/plant systems.

The global potential for annual sequestration of atmospheric CO_2 through biochar application has been estimated at the billion-tonne scale (Gt/year) under present day scenarios (Laird et al. 2009). The greenhouse gas mitigation potential from the application of biochar to agricultural systems may vary widely with variation in biomass feedstock, production technologies, product utilisation methods and environmental conditions.

15.2 Biochar Properties for Soil Health and Climate Change

15.2.1 Biochar Stability

The stability of organic matter in soils is determined by its ability to resist microbial and/or chemical decomposition, through chemical transformations and physical interactions with soil minerals (Lehmann et al. 2007; Rasse et al. 2006; Skjemstad et al. 1996). BC, as either charcoal or biochar, has a predominantly condensed aromatic structure that is known to be highly resistant to microbial decomposition (Baldock and Smernik 2002). Additionally, interactions of biochar with soil minerals could further increase stability of biochar in soil (Brodowski et al. 2006), further contributing to long-term carbon sequestration (Lehmann et al. 2009), while also adding to the health and production outcomes of soil systems.

Published studies have reported soil residence time of charcoal and biochars in timescales ranging from decades to centuries to millennia (Cheng et al. 2008b; Hamer et al. 2004; Hammes et al. 2008; Kuzyakov et al. 2009; Major et al. 2010; Skjemstad et al. 1996; Titiz and Sanford 2007; Zimmerman 2010). The stability of biochar depends on the type of biomass feedstock, charring conditions (temperature, heating time), biochar particle size, and edaphic and climatic conditions under which biochar oxidises (Kuzyakov et al. 2009; Lehmann et al. 2009; Nguyen and Lehmann 2009; Nguyen et al. 2010; Singh and Cowie 2008, 2010; Zimmerman 2010). In general, the proportion of aryl-C to aliphatic-C in biochar increases with increasing charring or pyrolysis temperature (Baldock and Smernik 2002; McBeath and Smernik 2009; Nguyen et al. 2010). The lability and density of the biomass feedstock and its mineral content may also influence the decomposition rate of biochar in soil (Nguyen et al. 2010; Singh and Cowie 2008, 2010).

Spectroscopic and surface chemistry analyses have proven useful to evaluate biochar–mineral interactions and oxidation status of biochar along a decomposition continuum (Cheng et al. 2006, 2008b; Liang et al. 2008). However, these approaches do not quantify turnover time, necessary to evaluate the residence time of biochar in soil. The rate of biochar decomposition may vary according to the stability of the oxidisable component, i.e. initial rapid decomposition of surface-oriented labile components of the biochar particle (e.g. aliphatic-C) followed by slow decomposition of condensed aromatic-C, which dominates the core structure of biochar. This warrants long-term studies to accurately estimate the mean residence time of biochar in soil (Kuzyakov et al. 2009; Nguyen and Lehmann 2009). Furthermore, biochars can potentially stimulate decomposition of native soil organic matter (i.e. humic and labile components) possibly by enhancing microbial activity (Hamer et al. 2004; Wardle et al. 2008). However, application of biochar may also lead to a decline in the decomposition of other organic matter components, through the possible enhancement of soil aggregation (Liang et al. 2010). The “priming effect” of biochar on organic matter decomposition in soil needs to be accounted for to determine the magnitude of biochar decomposed. Carbon isotope methods (δ^{13}C, or ^{14}C/^{13}C labelling) can be used to identify sources of C decomposed in biochar–soil systems (Kuzyakov et al. 2009). These methods can be relatively easy to manage in the laboratory, providing optimal conditions for biochar decomposition. However, in the field, presence of plant roots, rhizosphere processes and variable environmental conditions provide challenges to identifying C sources with a limited number of isotopes (Major et al. 2010).

15.2.2 Nutrient and Liming Values of Biochar

Some biochars are a potential source of nutrients (Table 15.1). The nutrient content of biochar is largely determined by biomass feedstocks (Gaskin et al. 2008; Singh et al. 2010b; Table 15.1). Feedstocks with higher nutrient contents such as animal manures will produce biochars with greater nutrient value, compared with plant feedstocks (Singh et al. 2010b). Pyrolysis temperature also affects nutrient value: for example, analysis of two biochars produced under different temperatures (400 and 500°C) from the same poultry litter feedstock revealed a higher N percentage (3.47%) and lower P percentage (3.01%) for the lower temperature product

Table 15.1 Nutrient content of selected biochars

Biochar source	N (%)	P (%)	K (%)	Ca (%)	CEC (cmol/kg)	C (%)	pH water	C:N	EC (dS/m)	Production temperature	References
Green wastes	0.18	0.07	0.82	<0.01	24	36	9.4[a]	200	3.2	450°C	Chan et al. (2007)
Poultry litter	2	2.5	–	–	–	38	9.9[a]	19	5.6	450°C	Chan et al. (2008)
Poultry litter	3.47	3.01	5.11	4.27	61.1	39.2	10.1	11.3		400°C	Gaskin et al. (2008)
Poultry litter	3.09	3.59	5.86	5.04	38.3	39.2	9.74	12.7		500°C	Gaskin et al. (2008)
Bark of *Acacia mangium*	1.04				37.14	39.8	7.4	38		260–360°C	Yamato et al. (2006)
Paper mill sludge and wood (1:1)	0.48		0.22	6.2	9.0	50	9.4	104		550°C	Van Zwieten et al. (2010a)
Paper mill sludge and wood (1:2)	0.31		1.0	11.0	18.0	52	8.2	168		550°C	Van Zwieten et al. (2010a)
Soybean cake	7.82					58.81		7.5		550°C	Uzun et al. (2006)
Pinus ponderosa bark	<0.01	<0.01	–	–	34.5	71.5	4.81	–	1.12	350°C	Gundale and DeLuca (2007)
Cow manure/*Pinus* spp. (3:1)	1.2	0.3	1.9	1.0	–	73.3	9.4	61	–	500°C	Kolb et al. (2009)
Pinus taeda chips	0.255	0.015	0.145	0.171	7.27	73.9	7.55	290		400°C	Gaskin et al. (2008)
Pinus taeda chips	0.223	0.014	0.145	0.185	5.03	81.7	8.3	366		500°C	Gaskin et al. (2008)
Eucalyptus deglupta wood	0.57	0.06	–	–	4.7	82.4	7.00	144	–	350°C	Rondon et al. (2007)
Eucalyptus saligna wood	0.22	0.03	0.27	0.98	–	85.1	9.4	387	–	400–500°C	Kimetu et al. (2008)
Eucalyptus saligna wood	0.26	0.02	0.24	2.13	3.48	83.6	8.82	322	0.32	550°C	Singh et al. (2010a)
Tectona grandis, *Pterocarpus macrocarpus*	0.3		3.1	4.4	10.7	87	7.5	290		Earth mound	Asai et al. (2009)

[a] pH measured in 1:5 soil/0.01 M $CaCl_2$ extract

compared with the higher temperature product (3.09% and 3.59% respectively, Table 15.1) (Gaskin et al. 2008). Furthermore, the concentration of C and N may increase with increasing pyrolysis temperature in plant-based biochars, but the C and N concentrations may decrease with increasing pyrolysis temperature for mineral-rich feedstocks, such as manure or papermill sludge, because less-volatile elements, including P, K, Ca and Mg, are concentrated as the volatiles are lost (Gaskin et al. 2008; Singh et al. 2010b). However, information on forms and bioavailability of nutrients present in biochars is scarce, and some research has shown that feedstock type and pyrolysis temperature can significantly influence bioavailable fraction of nutrients in biochars (Gaskin et al. 2008; Singh et al. 2010b).

Many biochars have a neutral to alkaline pH value (Table 15.1) and can provide some benefit in neutralising acidic soils. Van Zwieten et al. (2010a) reported liming values of 33 and 29% for two papermill waste biochars (compared to carbonate). Singh et al. (2010b) found that the $CaCO_3$ equivalence of biochars increased with increasing pyrolysis temperature.

15.2.3 Surface Charge Properties

Cation exchange capacity (CEC) is a measure of the ability of a substrate to retain positively charged ions through electrostatic forces. Biochar has been associated with the enhancement in CEC of some amended soils (Glaser et al. 2001; Van Zwieten et al. 2010a), thereby increasing the availability and retention of plant nutrients in soil and potentially increasing nutrient use efficiency. However, biochars from different feedstocks and produced under differing pyrolysis conditions may differ in surface charge properties. Furthermore, the method for determining CEC of biochars is far from standardised, and methods applied to soils may not be appropriate to biochars (Singh et al. 2010b). Of the two biochars from the same peanut hull biomass, the biochar produced at 500°C had a lower CEC (4.63 cmol/kg) compared with that produced at 400°C (14.2 cmol/kg) (Gaskin et al. 2008). The reduction in surface functional groups was suggested as the cause of lower CEC in the biochar produced at higher temperatures. The decline in the acidic functional groups on biochar surfaces has been reported to be greatest between 300 and 400°C (Guo and Rockstraw 2007). Liang et al. (2006) reported that the high charge density (CEC/specific surface area) of "aged" biochar resulted from oxidation of the particles and adsorption of organic matter to biochar surfaces. An increase in the charge density on biochar surfaces as biochar interacts with soil over time (e.g. Cheng et al. 2008a) could be responsible for enhanced cation retention and consequently reduced leaching from amended soils (Singh et al. 2010a). However, more research on the chemical interactions of differing biochars and soils, as well as the implications for soil nutrient retention, is needed.

15.3 Impacts of Biochar on Soil Health and Plant Growth

Biochar application can potentially influence a number of physical, chemical and biological properties of soil due to the inherent characteristics of biochar, and properties that develop over time through oxidation of biochar surfaces and interaction with plant–soil–microbial components. Some potential impacts of biochar application on soil health, soil carbon dynamics, nutrient use efficiency and plant growth are described below, and the benefits to plant–soil systems are summarised in Fig. 15.1.

15.3.1 Soil Physical Health

Increases in SOC contents often contribute to enhanced soil aggregate stability (e.g. Albiach et al. 2001; Chan et al. 2003; Neufeldt et al. 2002) which can result from interactions of carbon functional groups and clay mineral surfaces (Lehmann et al. 2008). Evidence suggests a close interaction between biochar particles and clay mineral surfaces, which may aid in the occlusion of biochar particles within newly formed soil aggregates (Brodowski et al. 2006; Liang et al. 2008). Implications that

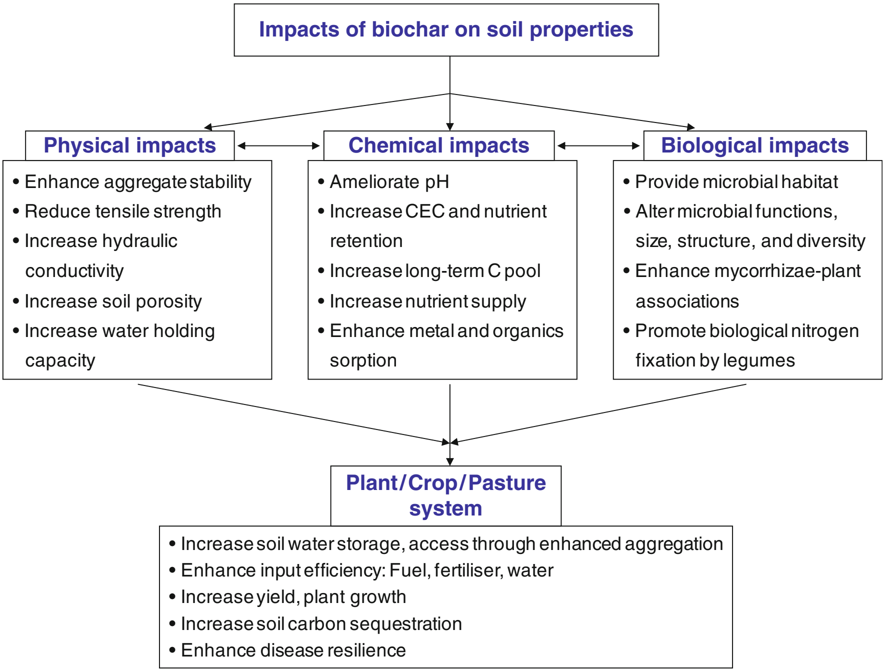

Fig. 15.1 Potential impacts of biochar application to plant–soil systems

biochar may contribute to the physical stabilisation of other soil organic matter, through aggregation (Liang et al. 2008), could also suggest an enhanced soil structure.

Biochar has also been associated with the enhancement of other soil physical properties such as soil water retention, saturated hydraulic conductivity and porosity. A study of available soil moisture in three soil types (sand, loam, clay) with 15, 30 and 45% wood biochar additions reported increases in the sandy soils, no change in the loam and a decrease in the clay soil (Tryon 1948). Similarly, Glaser et al. (2002) reported an 18% increase in field capacity for high BC Anthrosol soils compared to low BC surrounding soils, and attributed this to the increased surface area and porous structure of the char particle. In a study of soils under charcoal kilns in Ghana, saturated hydraulic conductivity and total porosity were increased and bulk density decreased compared to adjacent field soils (Oguntunde et al. 2008). Biochar was reported to enhance saturated hydraulic conductivity and water-holding capacity in upland rice production in Northern Laos (Asai et al. 2009), indicating a greater potential for efficient water use and improved soil productivity. The improvement in aggregation, water retention, saturated hydraulic conductivity and porosity from different biochar-amended soils could lead to better plant water use efficiency and consequently more resilient plant systems, and needs greater investigation.

15.3.2 Soil Chemical Health

The addition of biochar to soils can have a positive effect on soil chemical processes. Studies of the Terra Preta soils in the Amazon Basin have revealed significantly higher CEC per unit of SOC, attributed to the high level of "aged" biochar-like carbon in the Anthrosol soils (Glaser et al. 2001; Liang et al. 2006). In a pot trial, two papermill biochars (10 t/ha) increased the CEC and pH of a Ferralsol; however, there was no effect on a calcarosol (Van Zwieten et al. 2010a). The addition of a pecan biochar to a sandy Norfolk soil at rates of 1 and 2% did not change the soil's inherent CEC, although soil pH was raised more than one unit over the two incubation periods (Novak et al. 2009). These latter authors suggested minimal surface oxidation due to high pyrolysing temperatures may be the reason for the unchanged CEC.

Increased retention of plant available nutrients in soils as a result of biochar application could have significant agronomic and environmental benefits. Increased retention of inorganic nutrients such as ammonium and potassium within the soil profile may reduce fertiliser requirements. Furthermore, reducing nutrient losses from leaching may slow soil acidification (Helyar et al. 1990) and eutrophication of waterways. Although mechanisms for increasing soil nutrient retention have recently been explored (Liang et al. 2006; Major et al. 2009), greater understanding of the impacts of biochar on different soil types and in different climatic conditions is still required.

15.3.3 Soil Biological Health

Many studies have reported a positive response of soil micro-organisms to biochar amendments (O'Neill et al. 2009; Pietikäinen et al. 2000; Steiner et al. 2008a; Thies and Rillig 2009; Warnock et al. 2007; Zackrisson et al. 1996), although overall soil productivity outcomes from these interactions are mostly undocumented. Microbe/biochar interactions could include the attraction of microbes to the products of biochar adsorption such as other organic matter fractions, soil mineral components and nutrients, and extracellular enzymes (Thies and Rillig 2009).

Several studies have reported increased N mineralisation and nitrification through biological processes with charcoal amendment in forest soils (Berglund et al. 2004; DeLuca et al. 2002; MacKenzie et al. 2008). It has been suggested that the adsorption of phytotoxic phenolic compounds by charcoal in forest soils reduces the inhibition of nitrifying micro-organisms in these soils (Berglund et al. 2004; MacKenzie and DeLuca 2006; Zackrisson et al. 1996), or reduces the presence of organic compounds that could stimulate N immobilisation (DeLuca et al. 2006). In agricultural soils, N mineralisation and nitrification may be reduced by biochar addition due to either N immobilisation by N-poor and labile biochar (i.e. a high C/N ratio), or adsorption of ammonium (Lehmann et al. 2006). A study on the effect of a manure-pine biochar in four soils from Wisconsin reported enhanced microbial biomass and activity, as well as decreased extractable N with increasing biochar rates in the three agricultural soils (Kolb et al. 2009). However, Kolb et al. (2009) recorded the highest extractable N in the coniferous forest soil with the highest biochar rates. Pietikäinen et al. (2000) reported that charcoal adsorbed up to 42% of dissolved organic carbon from a litter extract, which consequently attracted and harboured micro-organisms.

Biochar may enhance the symbiotic associations of mycorrhizal fungi (MF) and terrestrial plants. Demonstrations of the positive response of plant growth and nutrient availability as a result of enhanced MF colonisation following BC additions in soils have been reported (Makoto et al. 2010). Root growth and aboveground biomass of *Larix gmelinii* (Gmelin larch) both increased with applied BC alone, and were greatest when BC was applied with MF. Phosphorus concentration in needles of the larch seedlings was also highest from the application of biochar with MF, indicating increased plant uptake, due to the utilisation of phosphate by the MF and seedling root/BC contact. A trial of maize amended with Acacia bark charcoal in Indonesia recorded increases in plant root mass and colonisation rates of MF (Yamato et al. 2006). A review of biochar–mycorrhizal interactions reported numerous positive responses, such as increases in soil nutrient availability and enhanced disease resistance, but also noted that a few studies reported a negative effect on MF with biochar addition, possibly from a reduction in plant available nutrients (Gaur and Adholeya 2000; Warnock et al. 2007).

Biochar has also been implicated in the enhancement of biological N_2 fixation (BNF) of *Phaseolus vulgaris* (Rondon et al. 2007). This study reported a BNF increase of 49% and 78% with 30 and 60 g/kg biochar additions, respectively.

However, a 90 g/kg biochar application increased BNF only by 30% above the control due to lower total biomass production and plant N uptake. Rondon et al. (2007) stated that greater boron and molybdenum availability were the main reasons for the increase in BNF. While some evidence exists for the improvement of plant–soil systems from BC/microbe interactions, this field of research is currently largely unexplored.

15.3.4 Turnover of SOC

Biochar addition to soils may influence the net carbon balance of systems. A stepwise increase in total soil carbon due to direct biochar addition is expected (Chan et al. 2007; Novak et al. 2009; Van Zwieten et al. 2010a). For example, a study of incubations of a Norfolk loamy sand amended with four rates of pecan shell biochar (0, 0.5, 1.0 and 2.0%) revealed increases in total SOC with increasing biochar rates (Novak et al. 2009). In a pot trial of *Raphanus sativus* with the addition of two poultry manure biochars (10, 25, 50 t/ha), total SOC increased compared to the controls (Chan et al. 2008). Furthermore, Liang et al. (2010) reported a greater incorporation of added plant carbon (sugarcane residue) into the intra-aggregate fraction in the *terra preta* soils as compared to the control soil (oxisol), indicating enhanced stabilisation of added carbon in the soil enriched with biochar-like organic matter. Additionally, in the studied *terra preta* soils, biochar-like carbon was found to reside primarily in organo-mineral (heavy) rather than free (light) fractions (Liang et al. 2010). However, another study reported that 72–75% of the light fraction of organic matter in an agricultural soil in Ontario was BC from the previously burnt C_3 forest, and that the turnover of the light fraction with BC was 2.5 times slower than without BC (Murage et al. 2007), suggesting a net reduction in the turnover rate of the light fraction in the presence of BC. In a cropping trial from Brazil, the loss of SOC over 20 months was reduced from biochar-amended soils (4–8% C) in comparison to soils amended with chicken manure, compost, or non-amended control plots (27, 27, and 25% C loss) (Steiner et al. 2007). In a study of historical charcoal blast furnace sites across the eastern half of the USA (Cheng et al. 2008b), organic carbon in the BC-containing soils was more stable, with a lower labile fraction and longer half-life of the recalcitrant component, compared to adjacent non-BC soils.

However, as noted previously in the stability section (Sect. 15.2.1), the overall increase of SOC due to biochar addition may sometimes be partly offset or even negated by the increased turnover of native/labile C (Hamer et al. 2004; Steinbeiss et al. 2009; Wardle et al. 2008). A 10-year study of mesh bags mixed with biochar and humus in a boreal forest site recorded a greater loss of carbon mass, compared with mesh bags of biochar or humus alone (Wardle et al. 2008). However, it was unclear as to the exact source of the carbon losses, or to their specific fate (i.e. leaching or emission). These losses occurred predominantly in the first year of mesh mixing and in the absence of a mineral component and soil profile; so there is some uncertainty as to the effect of biochar on humus in this instance (Lehmann and Sohi 2008). Another study investigating the influence of biochar on decomposition rates of litters of different quality mixed in a cambisol found no difference in the rate of decay between separate and combined mixtures of these substrates over 240 days of incubation (Abiven and Andreoli 2010). Clearly, further research is needed to generalise the effect of biochar on decomposition of relatively labile forms of organic carbon in soil and to advocate the role of biochar in offsetting global CO_2 emissions (Woolf et al. 2010).

The complexity of interactions between biochar and soil, and consequences of these for carbon sequestration, appear to revolve around the type of biochar, its degree of ageing and the extent of interaction with minerals and organic matter components in soil (Brodowski et al. 2005; Liang et al. 2008). It may well be that as biochar ages in a soil, increasing interactions with soil mineral components may help protect the labile and recalcitrant components of biochar from further biotic and abiotic oxidation. The occlusion of biochar particles within soil mineral aggregation has also been demonstrated in a study of a long-term agricultural field experiment in Germany (Brodowski et al. 2006). It was suggested that biochar could act as a binding agent in micro-aggregation. Further studies involving the identification and influence of specific biomass feedstocks and biochar production conditions to the mechanisms of biochar–soil interactions, as well as processes leading to stabilisation of biochar and other forms of organic matter in biochar-amended soil, are needed to assess the overall influence of biochar on the net soil carbon balance. In particular, the biochemical (e.g. microbial activity, aromaticity) and physicochemical (aggregation, sorption) factors affecting turnover of various forms of SOC need further investigation.

15.3.5 Nutrient Use Efficiency

There have been several reports of increases in fertiliser use efficiency with the addition of biochar to soils. A glasshouse study of the agronomic response of wheat, soybean and radish to the application of paper mill waste biochar in a ferrosol and calcarosol, revealed an increase in biomass of wheat (250% of fertilised control), as well as soybean and radish, with fertiliser plus biochar in the ferrosol (Van Zwieten et al. 2010a). The authors reported significantly increased N uptake for the wheat treatment and suggested an improvement in fertiliser use efficiency. However, the results of biochar and fertiliser amendments in the calcarosol were variable, with increased soybean growth but reduced wheat and radish growth. In an upland rice production system in Northern Laos, treatments with wood biochar reported higher grain yields and improved response to fertiliser treatments (Asai et al. 2009), although the authors noted that the positive yield response was dependent on adequate soil nitrogen. In another study, the application of a low nutrient biochar derived from timber increased the retention of N in soil and increased uptake of N into crop biomass (Steiner et al. 2008a, b).

When biochar amendments are combined with fertilisers, the effect is often synergistic, most likely due to increased plant nutrients and nutrient use efficiency from greater retention (Hossain et al. 2010). In a cropping trial in Brazil, wood charcoal and NPK fertiliser together significantly improved plant growth and doubled grain production of *Oryza sativa* and *Sorghum bicolor*, compared with NPK fertiliser alone (Steiner et al. 2007). These authors also reported higher plant available nutrients for following crops, despite the greater nutrient export from the higher plant yields of the biochar-amended plots. In a pot trial of *R. sativus*, a combination of 50 or 100 t/ha addition of green waste biochar and N fertiliser increased dry matter by approximately two times, compared with the N fertiliser treatment only, and 3.7 times compared to the biochar treatment only (Chan et al. 2007).

15.3.6 Plant Growth and Yield

The production of plant biomass through photosynthesis removes CO_2 from the atmosphere, and therefore any increase in plant biomass (carbon stock) due to biochar additions in soil systems will contribute to the mitigation of rapidly rising atmospheric CO_2 levels. Specifically, biochar either increases plant nutrient availability or enhances the soil environment (e.g. CEC, soil pH, aeration) and therefore may indirectly contribute to enhanced plant growth (e.g. Chan et al. 2008; Lehmann et al. 2003; Steiner et al. 2007; Zackrisson et al. 1996).

Some studies have reported increased plant nutrient availability and crop yield with the addition of BC alone. In a cropping trial (*Vigna unguiculata* and *O. sativa*) in Amazon Basin archaeological Anthrosol soils with high carbon levels and Ferralsols with added wood biochar, significantly increased phosphorus, calcium, manganese and zinc availability was found, with a 38–45% increase in biomass of the two crops in the Anthrosol (Lehmann et al. 2003). In a *Zea mays* trial of degraded cropping soils in Western Kenya, the authors noted that the application of biochar doubled crop yield, and furthermore suggested that the improvement could not be explained by biochar nutrient availability alone (Kimetu et al. 2008). Hence, despite the low nutrient status of some biochars, biochars generally appear to increase nutrient availability through increased ion retention in soils (Liang et al. 2006; Tryon 1948) and therefore potentially enhance plant yields. Biochar applications produced from manures may directly contribute high levels of nutrients to soils. Chan et al. (2008) reported yield increases of *R. sativus* with the application of 10, 25 and 50 t/ha of poultry manure biochar alone.

However, there have been variable results from the addition of some biochar types in particular soils. Van Zwieten et al. (2010a) reported reduced growth in wheat and radish with the addition of a paper mill sludge biochar in a calcarosol. In a pot trial of *R. sativus* in an Alfisol, a 10 t/ha green waste biochar and N fertiliser amendment resulted in a biomass decrease of 30%. This latter study also reported biomass increases at higher biochar rates. Other studies have reported a decline in soil N availability with wood biochar addition, potentially causing reduced yields (Asai et al. 2009). The inconsistency of plant response, ranging from small declines to large increases, would indicate a need for further research to verify the different plant responses to different biochars under varying soil conditions.

15.4 Role of Biochar in Climate Change Mitigation and Adaptation

The previous sections of this chapter have demonstrated the considerable potential for biochar to enhance the fertility and productivity of agricultural systems, as well as provide a stable form of carbon for sequestration in soil. As the sustainability of agriculture becomes increasingly threatened by climate change (Chap. 1), tools such as biochar will be needed to enhance resilience and productivity of these systems, so that world food supply can satisfy demand. Changes to world rainfall patterns may see declines in some of the major food producing areas of the world (Howden et al. 2007); the role of biochar in enhancing moisture retention may prove critical to maintaining production in these locations. Increases in soil health and crop productivity may have a range of resultant environmental, social and greenhouse gas balance implications. For example, higher crop productivity due to improved soil health could result in less use of land for the same yield, thereby reducing the need to produce food on more marginal land, and potentially increasing production per unit of gaseous emission. Enhanced crop productivity from biochar application may also reduce the rate of land clearing and deforestation, or encourage the rejuvenation of degraded land, again with significant positive ecological, social and economic consequences. A schematic of potentially interrelated ecosystem benefits of biochar production/application systems, including enhanced climate change mitigation and adaptation and improved performance of plant–soil systems, is presented in Fig. 15.2, and evidence for some of these benefits is described below.

15.4.1 Mitigation of N_2O Gas Emissions from Soil

Soil represents a significant source of the greenhouse gas nitrous oxide (N_2O). The microbial processes nitrification and denitrification are largely responsible for production of N_2O in soil (Chen et al. 2008; Dalal et al. 2003; Yanai et al. 2007). As the global warming potential of N_2O is 298 times greater than the equivalent mass of CO_2 in the atmosphere (Forster et al. 2007), technologies to minimise soil N_2O emissions need to be implemented to meet demands for climate change mitigation. Some recent studies have provided evidence that emissions of N_2O

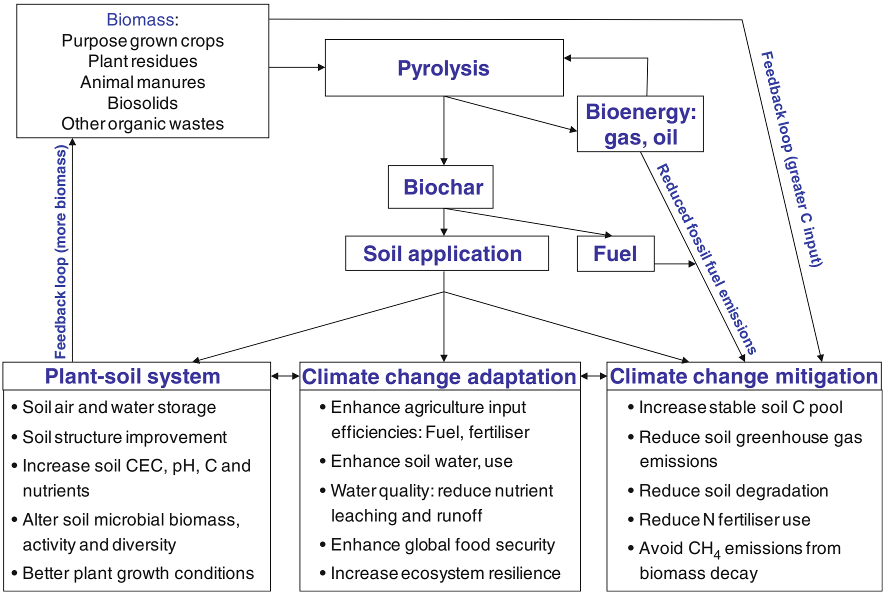

Fig. 15.2 Potential ecosystem benefits from biochar production/application systems

may be reduced by biochar application to soil (Singh et al. 2010a; Spokas and Reicosky 2009; Van Zwieten et al. 2010b; Yanai et al. 2007). The magnitude of reduction in N_2O emissions is dependent on soil type, biochar type and application rate, soil moisture content, and biochar ageing (Singh et al. 2010a; Spokas and Reicosky 2009; Van Zwieten et al. 2010b). However, in some cases, emissions of N_2O from soil can also be increased or not affected by the presence of biochar. For example, Clough et al. (2010) showed short-term increases in N_2O emissions in a pasture soil following biochar application (at 4.3% w/w) in the presence of ruminant urine; however, no significant differences were observed in cumulative N_2O emissions over the 53-day laboratory incubation between the biochar plus urine and urine-only treatments. Likewise, Spokas and Reicosky (2009) found that application of a high nitrogen compost-amended biochar (at 10% w/w) resulted in high N_2O emissions from three different soils and Spokas et al. (2009) found no significant differences in soil N_2O emissions at biochar application rates of 2–10% (w/w). The exact mechanisms for observed effects of biochar on N_2O emissions remain largely unexplored (Van Zwieten et al. 2009). Singh et al. (2010a) found that effectiveness of biochars in reducing soil N_2O emissions can increase over time, and hypothesised that this may be due to increased sorption capacity of biochars through oxidative reactions on biochar surfaces with ageing. Thus, in addition to its potential long-term soil carbon sequestration value, biochar application could provide considerable greenhouse gas mitigation benefit if reductions in N_2O emissions are found to apply broadly (Van Zwieten et al. 2009).

15.4.2 Reduced N Fertiliser Requirements

As the demand for food increases through wealth and population pressures, so too does the need for resources such as fertilisers and water. Nitrogen in particular is a resource that is poorly managed (Spiertz 2010), and more effort is needed to ensure that N supply matches N demand (see Chap. 6). As less than 50% of soil nitrogen can be used by the crop (Baligar et al. 2001), technologies that improve N use efficiency will have implications for productivity and emissions. A large portion of N is lost through mechanisms such as leaching (Olarewaju et al. 2009), or loss by denitrification and ammonia (NH_3) volatilization (Khalil et al. 2009). As the manufacture of nitrogen fertiliser releases more than 3 t CO_2e per t N (West and Marland 2002), technologies that can reduce the frequency and quantity of N application will result in lower emissions from the resulting reduction in fertiliser application. The evidence for increases in N fertiliser use efficiency with biochar amendments is reviewed in Sect. 15.3.5.

15.4.3 Biofuel Production

With mounting evidence for global warming from anthropogenic emissions of greenhouse gases, alternative forms of energy to reduce society's dependence on fossil fuels are required. The production of biofuels, from the chemical or thermal conversion of biomass (Bridgwater 2003), is currently being promoted as an alternative energy source that may help to reduce reliance on fossil fuel and avoid CO_2 emissions. The chemical and thermal pathways that produce biochar result in the co-production of combustible gas and/or oil which can be used for bioenergy production. It has been estimated that agricultural lands in the USA could provide enough manure through feedlot and intensive dairies to supply 0.7 billion US dollars of energy in terms of barrel of oil equivalents, based on a 20% thermochemical conversion factor of biomass (Ro et al. 2009). The energy output of pyrolysis has been favourably compared to that of the production of ethanol from corn. Even when pyrolysis is optimised for biochar production, energy output is 2–7 MJ, per MJ of fossil energy input (Gaunt and Lehmann 2008) compared with 1–2 MJ for corn to ethanol (Cherubini et al. 2009). The future decline in world fossil fuel reserves may enhance the relative merits of sustainable energy technologies such as pyrolysis with the added benefits of biochar application to soils.

15.4.4 Soil Structure Improvements

Well-structured soils are generally characterised by stable aggregation, high saturated hydraulic conductivity, low tensile strength and often high water-holding capacity. These qualities are all desirable as they assist in maintaining soil and plant productivity. Amendments such as biochar that may assist in the efficient capture, storage and utilisation of water in soils through structural improvement will become increasingly vital with any decline in rainfall as a consequence of a changing climate.

A poorly structured soil can present a substantial challenge for plant root development due to physical constraint associated with higher bulk densities and high soil tensile strength. Soil structure can be improved through the accumulation of soil organic matter (e.g. Perie and Ouimet 2008; Ruehlmann and Korschens 2009), with the more labile forms increasing the stability of macro-aggregates and less labile forms increasing the stability of micro-aggregates (Tisdall and Oades 1982). The evidence for better soil structure and enhanced physical properties (see Sect. 15.3.1) suggests that biochar may be a useful tool to mitigate climate change outcomes such as reduced rainfall, or extreme weather events (e.g. floods). Increased soil water use efficiency can help mitigate the impact of reduced annual rainfall on plant growth, while soil and nutrient losses from erosion during extreme weather events can be greatly reduced by increased soil aggregate stability and decreased surface runoff through enhanced infiltration.

As soil tensile strength and compaction increase, so does the requirement for greater cultivation draught capacity and frequency of tillage (O'Sullivan and Simota 1995). It could thus be anticipated that as biochar amendments can reduce soil tensile strength in a hard setting soil (Alfisol) as reported by Chan et al. (2007), and in a Norfolk loamy sand (Busscher et al. 2010), it would be reasonable to suggest that biochar could, in some soils, also reduce cultivation requirements, and hence reduce fuel usage. However, there is little direct evidence for overall enhancement of soil aggregation by biochar application, and the timeframe required; this aspect needs further research, especially as part of long-term assessment of potential agronomic and environmental benefits of biochar application in field studies.

15.4.5 Ecological Resilience

Appropriate biological functioning in soil systems can contribute to climate change adaptation through improvements in nutrient availability (Geisseler et al. 2009; Lavelle 1988), disease suppression (Larkin 2008) and aggregate stability (Lee and Foster 1991; Rillig and Mummey 2006). Many studies have reported increased microbial biomass in response to biochar amendments (O'Neill et al. 2009; Steiner et al. 2008a; Warnock et al. 2007; see Sect. 15.3.3). Biochar may enhance the symbiotic associations between MF and terrestrial plants, strengthening the plant's adaptability to climate change. Furthermore, biochar could provide long-term storage of carbon in soils while enhancing soil productivity, thereby enhancing the sustainability of agro-systems.

15.4.6 Net Mitigation Benefits

As indicated above, biochar may deliver mitigation benefits to terrestrial systems through several routes: stabilisation of soil organic matter, thus reducing its rate of oxidation while also decreasing soil erosion through improved aggregation; production of bioenergy that can displace fossil energy emissions (see Chap. 16); reduction in N_2O emissions from soil and fertilisers; reduction in fuel requirement for cultivation; increased carbon stock in plants and soil (Woolf et al. 2010). Furthermore, some biomass feedstocks, when used in biochar production, may deliver added benefit through avoided emissions: biomass that would have been deposited in landfill would have released methane (CH_4), while decomposition of manures can release CH_4 and N_2O gases (Gaunt and Cowie 2009). Therefore, the production and sequestration of biomass C in the form of biochar (with co-production and utilisation of bioenergy to offset fossil fuel emissions) could help slow climate change through the net removal of CO_2 from the atmosphere and avoiding emissions in the order of 1.0–1.8 Mt CO_2e/year at current levels of feedstock availability (Woolf et al. 2010). Gaunt and Cowie (2009) estimated net emissions reduction of 130–5,900 kg CO_2e/t feedstock for biomass residues (straw, manure and greenwaste), with variation arising from differences in feedstock properties, conventional use of feedstock and fossil energy source displaced. Roberts et al. (2010) also calculated an emissions reduction of 800–900 kg CO_2e/t biomass for similar biomass feedstocks (corn stover and yard waste). However, the mitigation benefit was much reduced for purpose grown biomass: Roberts' estimates ranged from a reduction of 440 kg CO_2e/t feedstock to an increase of 36 kg CO_2e/t feedstock, depending on the method used to estimate emissions from land use change (Roberts et al. 2010).

The sequestration benefit coupled with the creation of carbon neutral fuel (emissions from the burnt fuel are balanced by C sequestered in its production) could potentially reduce American emissions of CO_2 by 10% (Laird 2008). Globally, the potential mitigation benefit from biochar has been estimated at between 0.7 and 2.6 Gt C/year by 2050 (Laird et al. 2009).

15.5 Implementing Biochar Globally

The biochar supply chain includes biomass sourcing, conversion technology, product distribution and use. Due to the widely distributed nature of many biomass sources, biochar will need to be converted efficiently and economically from local biomass resources for distribution into regional agricultural soils.

The conversion of biomass to energy on a global scale will require a range of systems that use available resources while recognising regional socio-economic constraints and desired outcomes. For example, while up-scaling of "industrialised" biochar systems will be attractive to investors in developed countries, there is potential to implement clean, efficient biochar solutions at a small scale in developing nations to improve community welfare and reduce greenhouse gas emissions (Bailis 2009; Ewing and Msangi 2009). Existing thermal conversion technologies may be enhanced through the development of more sustainable sources of biomass (e.g. plantation biomass) and the implementation of modern kilns. Advantages of these more efficient systems may include rapid carbonisation, reduced gaseous emissions and higher yield of biochar from a greater potential range of feedstocks. The production outcomes from these systems will vary according to local resources and needs, ranging from small-scale production of biochar for fuel to larger industrialised production of liquid/gas energy and biochar for soil amendment.

The quantity of wood charcoal traded by the global forestry industry in 2008 was 49.35 million tonnes, of which more than half was produced in Africa and only 0.6% in Europe (FAO 2010). While the sustainability of this biomass source needs to be secured to remove the threat of net deforestation, with technological modernisation, growth in this industry could supply the energy needs of communities while meeting the expanding demand for biochar in agriculture. The relatively small amounts of charcoal produced in industrialised nations, under strict environmental regulatory control, are generally for specialised applications which can afford the more expensive process technology to ensure product meets specification.

The large-scale production of biochar for a low-value agricultural market requires the commercialisation of a new generation of clean and safe thermal conversion technology. The additional costs of the various regional environment and planning regulations can perhaps be offset through the lower cost of feedstock that comes with the efficient conversion of waste residue feedstocks to biochar. Commercial viability may also be assisted if the technology enables the utilisation of co-products such as bio-oils and gas for energy generation. Examples of commercial biochar production systems that have been able to demonstrate that large-scale reliable, economically viable and environmentally sustainable supply of product are largely non-existent at this time.

15.6 Future Directions

Biochar research is in its infancy; further investment in research is needed to understand the mechanisms of its impacts, particularly in relation to N_2O emissions, nutrient retention and interactions with soil constituents such as native organic matter and minerals in a range of soil type, vegetation systems and climatic conditions. As the impacts of biochar on soil processes may change over time, there is a need for long-term studies to assess biochar's potential to provide the projected benefits. Measures to secure sustainable feedstock supply and novel biochar processing technologies are needed to ensure that biochar production delivers net environmental benefits. Measures could include certification against an agreed standard, similar to the sustainability certification undertaken in the forestry sector and being developed for bioenergy. Government incentives for commercial demonstration are needed to enable the technologies to become an acceptably low-risk proposition in a free market economy. With the possibility of multiple environmental benefits from its use, biochar-amended systems may become a vital tool to mitigate climate change and enhance the sustainability and productive capacity of global terrestrial systems.

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