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Chapter 17

Role of Biochar in Mitigation of Climate Change

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Introduction

By virtue of the large fraction of the terrestrial carbon (C) cycle controlled by human activities (Haberl *et al.*, 2007), agroecosystems are both sources and sinks for greenhouse gases. Their potential role in mitigation of climate change thus depends on a dual strategy of decreasing greenhouse gas emissions while increasing sinks so that the net impact on climate warming is less than at present. Emissions of carbon dioxide, methane and nitrous oxide arise from various agricultural activities, ranging from land clearing to ploughing, fertilization, and animal husbandry (Denman *et al.*, 2007). Reductions in these emissions can be achieved by decreasing the heterotrophic conversion of organic C to carbon dioxide, and by better management of agricultural waste streams to minimize release of methane and nitrous oxide. Current sinks include C stored in standing biomass and soil organic matter, and the oxidation of atmospheric methane by soil bacteria. These sinks can be enhanced by increasing net primary productivity, thereby actively withdrawing more carbon dioxide from the atmosphere, and by promoting more oxidation of methane by soils. Judicious biochar management (Lehmann and Joseph, 2009a) may contribute to both strategies, reductions of emissions by agriculture and active withdrawal of atmospheric carbon dioxide, as part of a comprehensive carbon management scheme in agricultural and forestry watersheds.

Biochar is a carbon-rich organic material generated by heating biomass in the absence, or under a limited supply, of oxygen. This so-called charring or pyrolysis process has been used to produce charcoal as a source of fuel for millennia (Harris, 1999; Antal and Grønli, 2003). Recently, interest has grown in understanding the potential of this process to improve soil health by adding biochar as an amendment to soil, to manage agricultural and forestry wastes, to generate energy, to decrease net emissions of nitrous oxide and methane, and to store carbon (C) (Lehmann and Joseph, 2009a).

The main incentive of biochar systems for mitigation of climate change is to increase the stability of organic matter or biomass. This stability is achieved by the conversion of fresh organic materials, which mineralize comparatively quickly, into biochar, which mineralizes much more slowly. The difference between the mineralization of uncharred and charred material results in a greater amount of carbon storage in soils and a lower amount of carbon dioxide, the major greenhouse gas, in the atmosphere (Fig. 1).

The principle of creating and managing biochar systems may address multiple environmental constraints. Biochar may help not only in mitigating climate change, but also fulfill a role in management of agricultural and forestry wastes, enhancement of soil sustainability, and generation of energy (Lehmann and Joseph, 2009a). Pyrolysis is a comparatively low-technology intervention. Deployment on a global scale, however, must be done carefully if the full mitigation potential is to be reached. Critical aspects of a successful implementation are that:

— the biochar is sufficiently stable to reduce greenhouse gases in the atmosphere for an appropriate length of time. A critical question is what level of stability is sufficient to lead to real emission reductions.
— the storage of carbon as biochar in soil is not offset by greenhouse gas emissions along the value chain of the system, such as mineralization of soil carbon or emissions of other greenhouse gases (e.g., methane and nitrous oxide).
— net emission reductions are achieved for the entire life cycle of the system including indirect land use. Greenhouse gases that are generated as a result of biochar production and application must be smaller than the emission reductions.
— the biochar product does not cause unwanted side effects in soil. Biochar application must not lead to soil degradation or decreases in soil fertility but should rather be used to enhance soil quality.
— the handling and production of biochar are in compliance with health and safety standards and do not pose hurdles to implementation.
— the biochar system is financially viable.

This chapter discusses these issues in separate sections, identifies knowledge gaps, and proposes a road map to fully evaluate an environmentally and socially safe

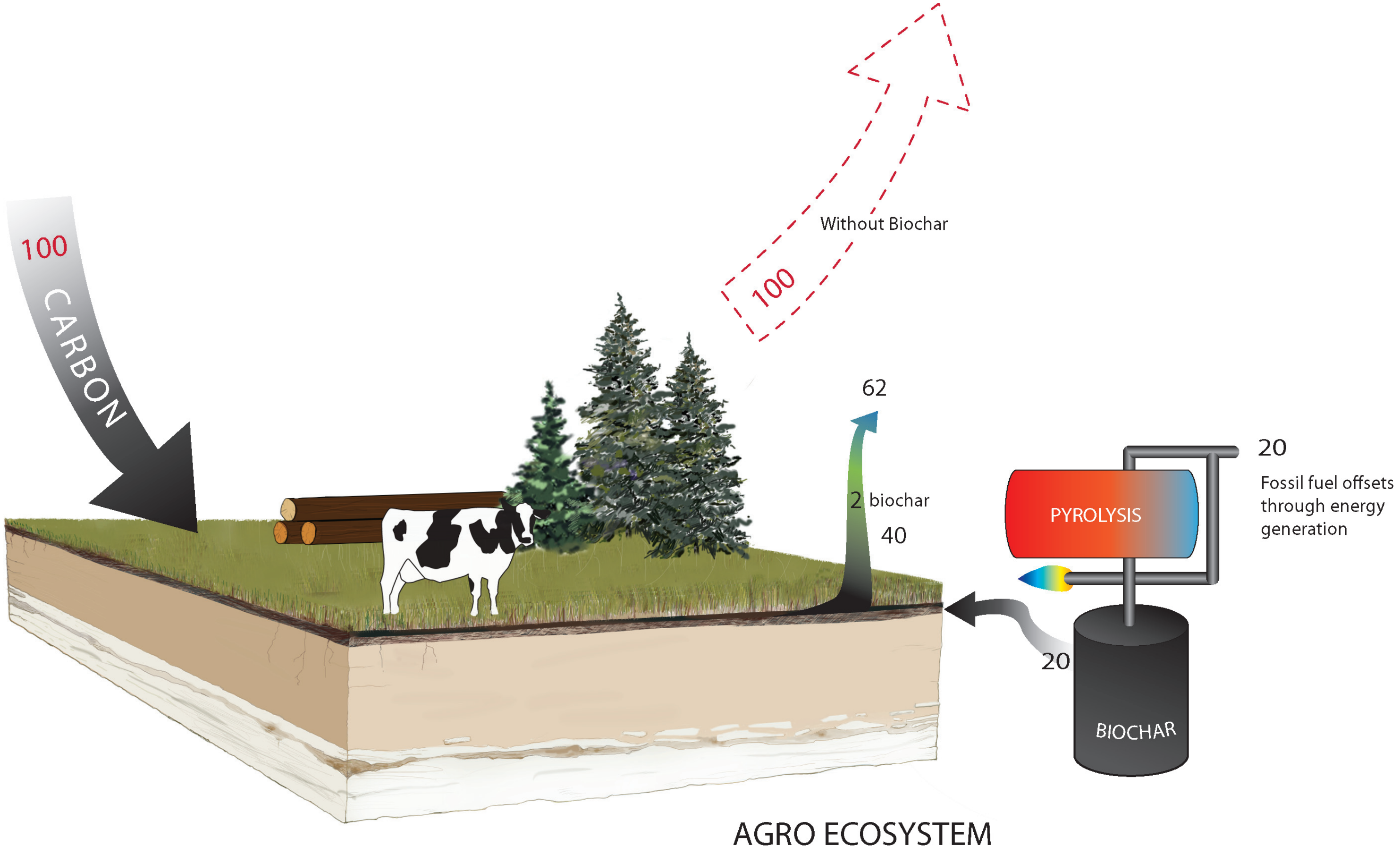

Fig. 1. Principle of the carbon flows in an agricultural watershed with (filled arrows) and without (dashed arrow) biochar systems approach.

exploration of the biochar potential to mitigate climate change if adopted widely around the world.

Stability of Biochar

The stability of biochar is fundamental to its efficiency in reducing greenhouse gas emissions. It is undisputed that biochar will eventually decay as does any other organic matter, but the key difference is that it decays *much* more slowly. It is therefore important to establish the mineralization rates of biochar to prove that the carbon thus sequestered remains in the soil for sufficiently long periods of time. Over the past two years, a series of publications indicate mean residence times in soil of several hundred to several thousand years (Bruun *et al.*, 2008; Cheng *et al.*, 2008; Hammes *et al.*, 2008; Lehmann *et al.*, 2008; Liang *et al.*, 2008; Kuzyakov *et al.*, 2009; Major *et al.*, 2010; Zimmerman, 2010). However, a universal mean residence time for biochars does not exist. In the same way as the turnover of plant residues depends on a variety of factors (including the substrate quality, moisture, temperature or soil texture and mineralogy), biochar decomposition will also be influenced by these factors. What is important to realize is that biochar is much more stable than uncharred organic matter (Baldock and Smernik, 2002), and that this difference is the relevant measure for its ability to prevent carbon from being returned rapidly to the atmosphere.

The primary reason for the stability of biochars in soils is their chemical recalcitrance (Liang *et al.*, 2008) which is due to aromatic structures of varying properties that are subject to current investigation (Fig. 2; Lehmann *et al.*, 2009). However, stabilization mechanisms due to reactions with soil constituents are likely underestimated. Similar to plant residues such as leaves or roots, biochar undergoes reactions with mineral surfaces and dissolved species in soil. At present, little is known about the nature and the quantitative importance of such reactions. The chemical properties of biochar likely facilitate the formation of such protection mechanisms because of their intrinsic chemical recalcitrance but also because of their specific surface properties. This is an area of ongoing research that will help explain the empirical evidence for the long lifetimes that biochar-type substances can attain, as shown from C-14 measurements. These naturally occurring carbon forms reach radiocarbon ages of hundreds to several thousands of years in soils (Glaser *et al.*, 2001; Pessenda *et al.*, 2001; Gouveia *et al.*, 2002; Gavin *et al.*, 2003) and many tens of thousands of years in ocean sediments (Herring, 1985; Masiello and Druffel, 1998; Middelburg *et al.*, 1999).

An important consideration is the question of how stable biochar needs to be for effective climate change mitigation. The answer depends, in part, on the biochar

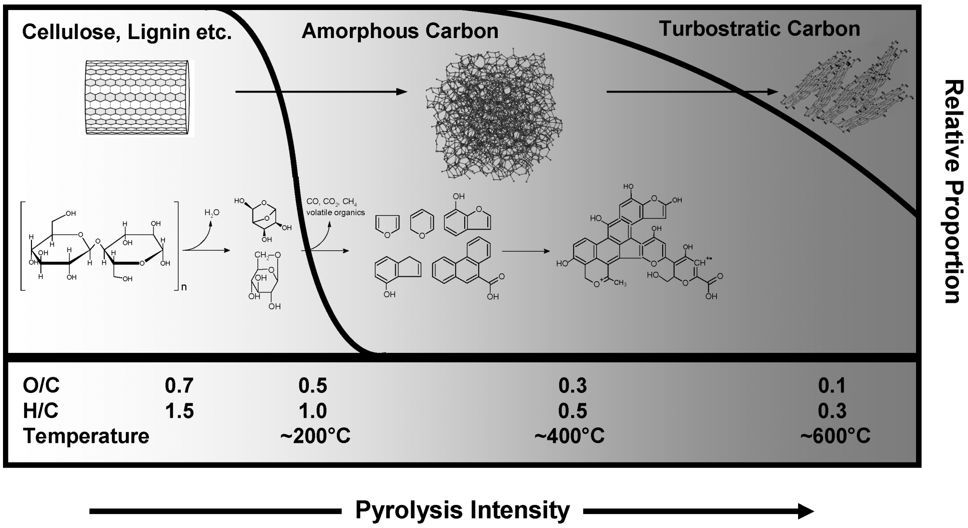

Fig. 2. Schematic example of transformation of organic materials (using the example of cellulose) during pyrolysis at progressively higher temperatures. Initially, levoglucosan and anhydride are produced by dehydration (Mok and Antal, 1983); at higher temperatures, structures dominated by poly-aromatic and heterocyclic C that are rich in oxygen (Knicker, 2007) form as large quantities of C in gaseous and liquid forms are released; at even higher temperatures, these structures collapse to yield amorphous C (Kercher and Nagle, 2003), which subsequently reorganizes to yield new highly ordered structures rich in radicals (Bourke *et al.*, 2007) and heterocycles (Harris, 2005). (Data summaries from Hammes and Schmidt, 2009; Krull *et al.*, 2009; Keiluweit *et al.*, 2010).

system that is used. While all systems rely on the stability of biochar in soil relative to uncharred biomass, total emission reductions depend on the efficiency by which biochar is produced, and the degree to which energy is captured during the pyrolysis process. Additional emission reductions can also be derived from the changes in soil properties induced by biochar amendments that affect net soil fluxes of other greenhouse gases such as methane and nitrous oxide and net primary productivity. Thus a biochar system with low biochar-production and energy-capture efficiencies, such as traditional biochar production methods, would have a climate-change mitigation impact much lower (perhaps by 2–3 fold) than a highly-efficient biochar system that maximized biochar production and the capture of energy to offset fossil-C emissions.

If one assumes a modern slow-pyrolysis biochar system in which 50% of the carbon in the biomass is converted to biochar, a mean residence time of greater than 100 years may be sufficient to be able to claim near-maximum emission reductions for common trading schemes (Gaunt and Lehmann, 2008). To achieve true long-term sequestration, however, the biochar may need to have a mean residence time exceeding a few hundred years (Fig. 3, left). A mean residence time of 50 years returns 96% of the sequestered carbon within 200 years, whereas a mean residence

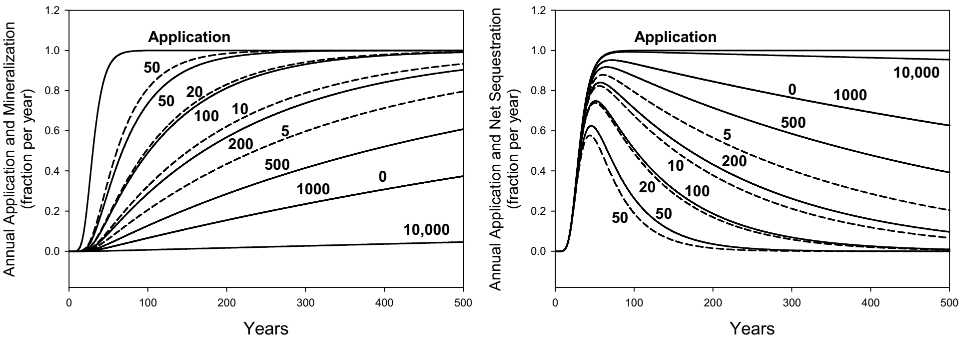

Fig. 3. Comparison of biochar application rates (solid black line) and either mineralization (left) or net sequestration (right, calculated as application minus mineralization from left graph). Blue lines and numbers are results from a one-pool model with varying mean residence times (MRT); red dashed lines and numbers are results from a two-pool model with fixed MRT but varying proportion of the labile fraction (labile fraction with a MRT of 20 years; recalcitrant fraction with a MRT of 1000 years); all calculations assume half of total adoption within 30 years and 90% within 50 years.

time of 500 years is merely 28%. The calculation of a mean residence time assumes that the carbon forms in biochar are homogeneous. In fact, the carbon consists of a mixture of highly recalcitrant forms and relatively labile forms. The essential impact of these forms on the climate-change mitigation potential of biochar may be captured as a first approximation by a two-pool model (i.e., "recalcitrant" and "labile") (Lehmann *et al.*, 2009). It is evident that uncertainties about the proportion of labile carbon in biochar have a greater effect on mineralization than uncertainties about mean residence times (Fig. 3, left). Over centennial time scales, an increase in labile carbon from 0 to 5% augments mineralization to a greater extent than a decrease in mean residence time from 1000 to 500 years.

As the labile fraction of the biochar increases or the mean residence time decreases, the net carbon stored annually per unit biochar produced is smaller because more carbon is mineralized from the biochar added in previous years (Fig. 3, right). And, for a given initial labile-carbon fraction, the amount of carbon mineralized in a given year eventually increases to the point that net carbon storage begins to decrease. The period before "peak biochar" is reached is proportional to the recalcitrance of the biochar and, under the modeled assumptions, occurs 44 to 95 years after the start of a biochar program. Therefore, long mean residence times are desirable for effective sequestration over the next few centuries. Clearly, biochar cannot be used to mitigate climate change in perpetuity, unless mean residence times on the order of thousands of years are obtained for all the biochars produced. However, even comparatively short mean residence times of several hundred years will allow enough carbon to be stored in biochar over the next 100 years to have a significant impact on climate change while other opportunities for carbon management are being developed.

Effects of Biochar on Gas Exchange from Soil

Biochar may also have an influence on the stability of soil organic matter and crop residues, and on nitrous oxide and methane emissions from soil. Those emissions may enhance or counteract the emission reductions achieved by the conversion of relatively labile biomass into much more recalcitrant biochar.

Current research indicates that biochar additions may not increase the loss of organic matter already present in soil to a greater extent than other soil additions. Some short-term losses may occur (Hamer *et al.*, 2004; Wardle *et al.*, 2008; Spokas *et al.*, 2009) due to a stimulation of microbial activity, possibly as a result of nutrients and labile carbon in the biochar or an increase in pH values of acid soils. Where carbon dioxide measurements were done, experiments could not detect an increase and, in several cases, even detected a decrease of mineralization of residues or soil organic carbon (Kuzyakov *et al.*, 2009; Spokas *et al.*, 2009; Liang *et al.*, 2010; Singh *et al.*, 2010). Further research is required to fully explore the interactions between biochar and other organic matter in soil.

Biochar appears to also affect the emissions of other greenhouse gases such as nitrous oxide and methane. Only limited results are currently available, and, as a result, the underlying mechanisms for the observed effects are still unclear. Empirical evidence indicates that in most cases a significant reduction of nitrous oxide emissions from soil can be expected (Yanai *et al.*, 2007; Spokas *et al.*, 2009; Spokas and Reicosky, 2009; Singh *et al.*, 2010). Greater emissions of nitrous oxides were observed in one study at high moisture contents (Yanai *et al.*, 2007) or temporarily at the beginning of the experiment possibly due to high nitrogen additions (Singh *et al.*, 2010). The underlying mechanisms must be examined to explain these remarkable reductions in order to base the empirical evidence on sound scientific footing. Possible explanations are changes in water-filled pore space or nitrogen and carbon availability, but changes in microbial populations and carbon dioxide consumption cannot be excluded.

For methane emissions, mixed results have been obtained. For tropical soils used to grow maize and a forage crop, net methane emissions by soils decreased with biochar amendments (Rondon *et al.*, 2005; 2006). Biochar had no impact on methane emissions by rice paddy soils (Knoblauch *et al.*, 2008) and some biochars decreased methane oxidation in temperate-zone soils (Spokas *et al.*, 2009; Spokas and Reicosky, 2009). Only an understanding of the processes involved will enable us to predict the magnitude of the methane and nitrous oxide emissions to be expected in the landscape and to optimize the effects.

Life-Cycle Emission Reduction

It is important to quantify the emission reductions along the entire life cycle of a biochar system. The emission reductions by converting labile biomass into much more stable biochar may be partly or entirely canceled by emissions associated with producing and collecting the feedstock, building and operating the pyrolysis unit or handling and applying the biochar to soil. On the other hand, bioenergy can be captured from the gases that evolve during pyrolysis in a variety of ways and may offset fossil fuel use. This can be counted as an additional emission reduction. The systems view of biochar management (Lehmann and Joseph, 2009a; 2009b) is a critical way towards achieving net emission reductions (Roberts *et al.*, 2010).

Life-cycle or related assessments of the total greenhouse gas emissions indicate large variations between different biochar systems. This is expected as many parameters change depending on feedstock collection requirements, whether fertilizers are used or not, the moisture content of the feedstock and the ability to capture the bioenergy (Gaunt and Lehmann, 2008; Gaunt and Cowie, 2009; Roberts *et al.*, 2010). The limited information available to date allows this tool to be mainly used as a way to identify those practices that have the greatest influence on the emission balance. The largest proportion of the emission reduction typically stems from the biochar sequestration, e.g., varying between 50–65% of total emission reductions for bioenergy crops replacing winter wheat, and 41–46% for crop residues in the UK (Gaunt and Lehmann, 2008). Nitrous oxide or possibly methane emission reductions from soil can play a role, but are dwarfed by potential emission reductions from avoided decomposition of biomass in landfills using current accounting approaches (Gaunt and Cowie, 2009). Pyrolysing green waste (e.g., yard waste) that would otherwise be land filled, was calculated to reduce emissions by more than $3\,\mathrm{t\,CO_2e\,t^{-1}}$ feedstock compared to the biochar sequestration of only $0.7\,\mathrm{t\,CO_2e\,t^{-1}}$ feedstock (Gaunt and Cowie, 2009).

Potential leakage has to be carefully assessed. Leakage is a term used to describe emissions generated at a different place due to a change in practice that is not captured in the accounting approach. If indirect land use change occurs, biochar systems may not reduce emissions over a 30-year period (Roberts *et al.*, 2010) as argued for biofuels from cropland in general (Searchinger *et al.*, 2008).

Another factor is the moisture of the feedstock. Very wet yard wastes or animal manures require innovative biomass handling to minimize the energy costs to achieve the degree of dryness necessary for pyrolysis. Whether hydrothermal conversion

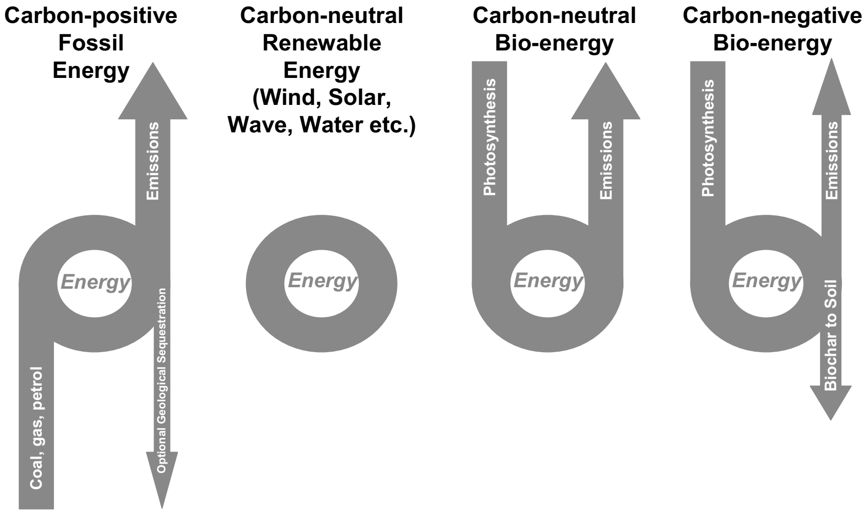

Fig. 4. Schematic carbon flows of different energy scenarios.

of these high-moisture wastes is a solution for biochar production remains to be fully evaluated, although we note that the products evaluated so far are much less recalcitrant than biochars produced by pyrolysis of dry biomass (Steinbeiss *et al.*, 2009).

A notable difference from other energy approaches is the potential ability of biochar systems to withdraw carbon dioxide from the atmosphere (Fig. 4). Such a carbon negative energy is achieved because the emission reductions are greater than the emissions generated during the life cycle of a biochar system (Gaunt and Lehmann, 2008; Roberts *et al.*, 2010). Although such sequestration may not be sufficient to completely offset global fossil-fuel emissions (Amonette *et al.*, 2008; Laird *et al.*, 2009; Roberts *et al.*, 2010), it may serve as a complementary strategy for a transition into a low-carbon economy.

The ability to generate carbon-negative energy comes at a cost, however, in the form of unrecovered energy from the biomass feedstock that remains in the biochar. If this energy must be supplied by some other means, such as combustion of fossil fuel, then questions arise as to the net avoided emissions from the production of biochar and its storage in soils relative to the complete combustion of the biomass to offset fossil-fuel emissions (bearing in mind that combustion is not always possible or desirable). One can view this from a variety of different perspectives. The first consideration is whether the alternative to biochar addition to soil is combustion of biochar after pyrolysis or combustion of biomass without going through a pyrolysis step. The second consideration is whether emissions are expressed per unit area, per unit biomass or per unit energy produced. Such data are still being refined as the science and technology of bioenergy production matures. The relative benefit of biochar also depends on the carbon intensity (i.e., the amount of carbon dioxide emitted per unit of energy produced) of the fossil-fuel emissions being offset. Calculations by Fowles (2007) suggest that for offsetting all but the most carbon-intensive fuel (coal), the net avoided emissions of biochar exceed those of bioenergy through direct combustion of biomass. However, Roberts *et al.* (2010) calculated that biochar systems seem to yield about the same emission reduction as direct combustion of biomass per unit feedstock. Also per unit of energy produced, emissions reductions for a crop residue scenario with natural gas as the fossil fuel offset (calculated from data in Roberts *et al.*, 2010) were similar with 0.08 kg CO_2e MJ^{-1} for both combustion and pyrolysis systems. Per unit of land, the relative emissions benefit of adding biochar to soil exceeded that of combusting the biochar for both natural gas and coal as fossil-fuel offsets and for biomass sourced from bioenergy crops as well as crop residues (Gaunt and Lehmann, 2008). None of these comparisons considered the possible positive feedback of increased plant productivity stemming from application of biochar to soils. Local conditions can be decisive in tipping the balance between biochar and bioenergy one way or the other, and research to determine whether biochar systems may have a significant role in tackling global warming is vital (Fowles, 2007).

Not all biochar systems may make use of the energy, either because the costs of the required pyrolysis equipment are too high or the energy production occurs in a location without a need for the generated energy. Careful evaluation is required to assess the options for emission reductions in such cases. Suitable entry points to achieve emission reductions are situations where large emissions occur due to burning of crop or forest residues and due to landfill of green wastes.

Biochar Effects on Soil Health

Historically, the primary incentive for evaluating biochar as a soil amendment stemmed from its beneficial effects on soil fertility and productivity, not its potential to reduce greenhouse gas emissions. The most recent interest was sparked by in-depth investigations into the so-called "Terra Preta de Indio" soils in the Amazon that have very high concentrations of biochar and maintained an unusually high fertility over millennia (Lehmann *et al.*, 2003). Single scientific studies of Terra Preta have been conducted for some time (Sombroek, 1966) and sporadic research into soil fertility effects of biochar stretches back to the beginning of the 20th century and before (Lehmann and Joseph, 2009a).

Published information generally indicates no detrimental effects of the tested biochars on soil productivity but rather yield increases (reviewed by Glaser *et al.*, 2002; Lehmann and Rondon, 2006; Blackwell *et al.*, 2009). The reasons for this may vary and can include pH improvements in acid soils, increased nutrient availability (Glaser *et al.*, 2002) and soil water infiltration (Ayodele *et al.*, 2009) or possibly even microbial effects (Thies and Rillig, 2009). However, biochars may be produced that are detrimental to soil productivity if they contain excessive amounts of salts, a fraction that immobilizes nitrogen during heterotrophic respiration, or simply a pH that decreases availability of nutrients. Such effects are specific to soil, crop and biochar type. Similarly, biochar may have no effect on productivity at all. For example, a very productive soil may not show any improvements in crop yield in response to biochar additions and only show positive responses after severe degradation (Kimetu *et al.*, 2008). After addition of biochar with pH above 8, a soil with a low pH may show large yield increases, whereas a soil with a high pH may rather show yield decreases (van Zwieten *et al.*, 2010). This site-specificity is well known for any soil amendment management, be it organic (Kimetu *et al.*, 2008) or inorganic (Bullock *et al.*, 2002; Dobermann *et al.*, 2002) fertilizer applications. Additionally, crop rotations play an important role in affecting the best practice; for example, after a fallow with alfalfa nitrogen additions can typically be significantly reduced (Bruulsema and Christie, 1987). These examples show that we would not expect blanket recommendations to be useful for biochar.

A point of careful evaluation is the question whether biochars may add organic or inorganic pollutants. Some feedstocks may contain heavy metals that will be retained in the biochar. If the uncharred organic material is currently applied to soil, the biochar conversion may possibly prove to be beneficial by decreasing the availability of the heavy metals and resulting in lower levels of soluble metals (Shinogi *et al.*, 2003). However, in situations where biochar is not added as an alternative to the uncharred organic matter, total loadings must be carefully evaluated and monitored, as the heavy metals will eventually be released (albeit after long periods of time). While heavy metals are already contained in the original feedstocks, polycyclic aromatic hydrocarbons (PAH) and dioxins may be formed during heating. Current understanding is that the precursors of dioxins such as chlorinated benzenes and phenols are formed at around 750°C (Froese and Hutzinger, 1996), which is above the temperature of 350 to 600°C at which biochars are commonly produced. Secondly, the absence of oxygen in pyrolysis and the absence of chlorine in biomass used for making biochar limit the production of dioxins (Conesa *et al.*, 2009). These are the reasons why dioxins have not been reported in biochars (Garcia-Perez and Metcalf, 2008). PAH may already be produced above 400°C (Hajaligol *et al.*, 2001), even though experimental evidence indicates that the more hazardous tertiary PAH forms are primarily formed above 700°C (Ledesma *et al.*, 2002). However, monitoring is required to prove compliance for both PAH and dioxins, especially if feedstocks are used that already contain these compounds or their direct precursors, and if pyrolysis temperatures exceed 600°C.

Long-term detrimental effects of the most commonly used biochar materials produced from agricultural and forest residues are extremely unlikely. Biochar-type substances are found in almost all soils globally (Krull *et al.*, 2008) and make up approximately 20% of the soil organic carbon on the Australian continent (Lehmann *et al.*, 2008). Terra preta and other anthropogenic soils with high biochar contents are several thousand years old and have rather shown soil health improvements (Lehmann *et al.*, 2003).

Nonetheless, there is a limit to the total amount of biochar that can be stored in agricultural soil, even though values for such a limit are currently not apparent and likely depend on biochar, crop and soil type (Lehmann and Rondon, 2006). A biochar application of 10 Mg C ha^{-1} to all 1.5 Gha of the world's croplands translates into a global addition of 15 Pg. Assuming an upper ceiling of between 50–100 Mg C ha^{-1} to the plow layer (as some crop growth reductions may emerge at high rates, Rondon *et al.*, 2007), it is apparent that a biochar approach has large technical potential, albeit with a possible maximum storage of about 75–150 Pg under such a scenario. Deeper incorporation of biochar may be possible in some situations, and this could increase the potential storage reservoir.

Another possibility is the application of biochar to managed forests, which could improve productivity and sequester C. However, logistical issues of transporting the biochar and application to forests limit this option to forest lands with existing road access and possibly to establishment of regrowth. Systems such as these may benefit from small, mobile pyrolysis units such that forest residues and biochar can be utilized and applied on site. It is conceivable that biochar could be stored in other locations (Seifritz, 1993), such as abandoned mine sites, once agricultural soil becomes saturated. Such an approach, however, would eliminate the positive feedbacks obtained when biochar is stored in soils, and could present risks associated with the potential for combustion of the stored C. Thus there would be little incentive to undertake biochar production for this purpose rather than bioenergy production.

Short-term negative effects may arise from a variety of properties and processes as outlined above. The burden is on the scientific community to provide a knowledge base that allows identification of suitable properties of biochar and their production conditions. This can then be incorporated into guidelines or even a regulatory framework to ensure that only those biochars are produced and applied to soil that are beneficial to soil health and productivity. It is logical to examine existing regulations for the application of composts or municipal wastes as guidance for the development of regulations for safe application of biochar.

Handling and Production

The general challenges for handling material flows of biomass feedstocks have been well recognized during recent examinations of bioenergy (Rentizelas *et al.*, 2009; Sokhansanj *et al.*, 2006). Constraints posed for example by varying availability over time will also apply to feedstock used for biochar production. Specific to biochar is the transport requirement back to the fields and its application to soil. Dustiness and combustibility are significant challenges that require intelligent materials handling procedures (Blackwell *et al.*, 2009). For some agricultural systems, simple modifications may result in safe and inexpensive solutions such as in combinations with the application of liquid dairy manure. For other systems the hurdles may be greater. This is an area of ongoing exploration as the science of biochar effects in soil and its production matures. Optimization and full evaluation are still pending since the amounts of biochar necessary to achieve the relevant development have not been available.

Biochars have dark colors and their application to soil can be expected to darken the soil. This may decrease the albedo of the land surface which could then increase warming of the atmosphere. Any carbon accrual will have a similar effect (Ben-Dor *et al.*, 1999), though one may hypothesize a somewhat greater effect of biochar per unit carbon applied. Greater soil organic matter or biochar contents may often change plant growth thereby also changing the albedo. The question of albedo is linked to the performance of the agroecosystem and not just the soil, which includes multiple coupled processes such as the water balance or plant growth (Pielke *et al.*, 2002).

Aeolian transport and deposition of black carbon particles can have a large influence on the albedo of snowy regions and, consequently, on global radiative forcing (Koch *et al.*, 2009). Currently, most of these particles are derived as soot from cooking fires, diesel-engine exhaust, and open biomass burning (Bond *et al.*, 2004). Several pathways by which biochar particles can become entrained in the atmosphere can be envisioned (during production, incorporation into soils, or during subsequent soil erosion). Although expected to be small relative to the degree to which biochar decreases radiative forcing, the potential of these pathways to further exacerbate climate change in polar and other snowy regions needs to be determined.

Investigations into the production of charcoal and even of bioenergy by pyrolysis have been conducted for some time (Brown, 2009). However, optimization for biochar is a fairly recent approach and has not been sufficiently developed for small-scale and distributed application. Specifically, simple and effective ways to achieve minimal emissions from the pyrolysis process are not readily available. In biochar systems that capture bioenergy, the fossil-fuel offsets make up a significant proportion of total emission reductions (Amonette *et al.*, 2008; Gaunt and Lehmann, 2008; Gaunt and Cowie, 2009; Laird *et al.*, 2009; Roberts *et al.*, 2010). Therefore, uncertainties or variability of energy yields due to site-specific conditions can make a large difference to the overall emission reductions.

The costs and revenues are the least explored aspects of biochar systems. Although a biochar industry has been developed in Japan for some years and several demonstration projects have been implemented to date globally (www.biochar-international.org), the economic costs and benefits will likely emerge only over the coming years. This is in part due to the nascent state of the industry but also the complexity of the way in which biomass is currently utilized and the variations in soil responses (as discussed above). The most economically viable biochar systems are likely to develop using biomass feedstocks that have costs associated with the current management.

Carbon Trading

Biochar systems may lend themselves to carbon trading. However, to date no relevant trading activities have been carried out. The reasons may include the lack of sufficient biochar being produced and applied to soil, the early stages of methodology development, and perceived uncertainties in biochar stability. The principles of biochar appear to make systems design suitable to carbon trading likely. A conversion of relatively labile organic materials that would emit greenhouse gases within a short period of time into biochar that decomposes much more slowly constitutes a reliable, predictable and measurable emission reduction. The emission reductions may be reliable because there are no known causes of a rapid release of the captured carbon dioxide, as compared to, for example, no-till which can be easily reversed. The emissions reductions may be predictable, if sufficient information is available about the mean residence time of a specific biochar and its effects on greenhouse gas emissions from soil.

Significant challenges include the prediction and quantification of emissions along the life cycle of the entire suite of possible biochar systems. Development of models for standard practices supported by a limited number of measurements may help in establishing a typology of biochar systems, similar to what is proposed for agricultural carbon in general (Paustian *et al.*, 2009). Verification is possible through measurements of biochar contents in soil, if desired. Even though such analyses can be done by relatively inexpensive mid-infrared tools using proper calibration (Janik *et al.*, 2007), it may still be too costly as part of a routine methodology but rather implemented as a learning tool at early stages of development. One challenge with verification may be the erodibility of biochar, which can be significant

(Guggenberger *et al.*, 2008; Major *et al.*, 2010). Even though the biochar may be more stable in sediments of rivers and oceans (Masiello and Druffel, 1998), its presence may not be readily identifiable without large investments into monitoring. To allow full verification, erosion must be minimized.

In many situations, soil fertility benefits may be a large part of the incentive to adopt biochar management. This may trigger questions about additionality, i.e., whether the emission reduction was additional to the baseline scenario or whether the emission reduction would have been implemented without the financial incentive from carbon credits. The fact that biochar systems have to date not been implemented to any significant extent may be an argument for an additional emission reduction. Such an argument will likely be challenged on a case-by-case basis under current rules. One may argue that it is desirable to allow multiple sustainability outcomes and therefore allow stacking benefits of biochar (Lehmann, 2009). Being able to combine soil fertility benefits with carbon trading will also support biochar systems that do not contribute to, but rather mitigate climate change. The development of carbon trading programs based on avoided soil carbon losses will help in preventing unintended consequences of a biochar soil management program. Unsustainable land management practices such as the utilization of primary forests for biochar production or bioenergy plantations would then be discouraged due to indirect land use change impacts.

Safe Exploration of the Global Biochar Potential — A Road Map

Climate change mitigation by using biochar rests on (i) the conversion of biomass to reduce emissions of greenhouse gases either from soil or landfill, and (ii) fossil fuel offsets due to bioenergy production. In order to calculate the full carbon abatement of a given biochar system, however, all emissions and emission reductions must be determined. This makes it at present impossible to predict with high accuracy the emission reductions with biochar as a global strategy. It is possible, however, to probe the technical or theoretical potential depending on the availability of biomass that can be accessed by following sustainability guidelines and without competing with existing uses. Preliminary evaluations have concluded that emission reductions in the range of a few gigatons of CO_2 equivalents may be possible based on feedstock availability (Lehmann *et al.*, 2006; Amonette *et al.*, 2008; Laird *et al.*, 2009; Roberts *et al.*, 2010). Such conclusions of the purely technical potential justify an exploration of biochar as a global strategy. A full and more realistic picture of the biochar potential will only emerge after sufficiently wide-spread implementation.

There are several principal obstacles to wide-spread implementation of biochar systems. The requirement for changes in practices is a hurdle to any adoption.

Biochar requires building, purchasing and running equipment for the conversion of biomass. This costs money and time which in many cases will only be invested if the return on investment occurs in a feasible timeframe. The multiple revenue streams such as soil fertility enhancement, waste management, bioenergy production or emission reduction may aide in sustainability but may also be obstacles to designing biochar solutions for any given location if they are all required to generate financial benefits (Lehmann, 2009). Implementation will rest on local solutions to environmental constraints and on wider agricultural activities. Biochar cannot be an alternative but should be perceived as complementary to established best agronomic practices such as appropriate tillage, nutrient supply or crop rotation.

It is therefore wrong to ask "whether" biochar systems will be successful. One should rather ask "where" they may be successful. Biochar has traditionally proven to be a viable strategy as a niche approach. Once an exploration of its utility has been conducted under a wide variety of situations, the global potential will become apparent. It is still useful to probe the theoretical potential of biochar. But it is not useful to make its examination contingent upon proof of its universal applicability. More realistic expectations on the one hand, and more practical approaches to innovations in sustainability on the other hand, will be supportive of appropriately placing biochar in the mix of climate mitigation tools in research, development and policy.

Many critical pieces of information have been gathered to launch demonstration projects that should be accompanied by credible research and monitoring activities. Before wide-spread adoption can be planned, however, several knowledge and development gaps must be filled. Two main groups of research and development goals need to and can be addressed in the near future:

(1) Development of pyrolysis units for a variety of biochar systems with:
    - Increased efficiency
    - Lower emissions
    - Improved safety
    - Lower costs

(2) Site, crop and biochar-specific information about the magnitude and mechanisms to manipulate:
    - Soil fertility
    - Greenhouse gas emissions
    - Nutrient leaching

The successful development of biochar on a global scale will require setting standards for the performance of the pyrolysis process, the sourcing of biomass, the properties and the application of biochar. The International Biochar Initiative (IBI, under www.biochar-international.org) has provided leadership in assembling an international body of scientists to develop and refine guidelines that governments and international organizations can adopt for a safe deployment of biochar based on a full set of sustainability principles. The interest in biochar systems as an approach to mitigate climate change by various governments, the United Nations Framework Convention on Climate Change (UNFCCC) and the United Nations Convention on Combating Desertification (UNCCD) among other organizations are a testament to the growing need for a concerted effort to provide the information that allows a global strategy to be developed.

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