GLOBAL CHANGE BIOLOGY
BIOENERGY

GCB Bioenergy (2015) **7**, 1–13, doi: 10.1111/gcbb.12132

The way forward in biochar research: targeting trade-offs between the potential wins

SIMON JEFFERY*, T. MARTIJN BEZEMER†, GERARD CORNELISSEN‡§, THOMAS W. KUYPER*, JOHANNES LEHMANN¶, LIESJE MOMMER‖, SARAN P. SOHI**, TESS F.J. VAN DE VOORDE‖, DAVID A. WARDLE†† and JAN WILLEM VAN GROENIGEN*

**Department of Soil Quality, Wageningen University, PO BOX 47, Wageningen 6700 AA, The Netherlands, †Department of Terrestrial Ecology, Netherlands Institute of Ecology (NIOO-KNAW), PO Box 50, Wageningen 6700 AB, The Netherlands, ‡Norwegian Geotechnical Institute (NGI), P.O. Box 3930 Ullevål Stadion, Oslo NO-0806, Norway, §Department of Plant and Environmental Sciences (IPM), University of Life Sciences (UMB), P.O. Box 5003, 1432 Ås, Norway, ¶Cornell University, 909 Bradfield Hall, Ithaca 14853, NY, USA, ‖Nature Conservation and Plant Ecology group, Wageningen University, PO Box 47, Wageningen 6700 AA, The Netherlands, **UK Biochar Research Centre, School of GeoSciences, University of Edinburgh, Edinburgh EH9 3JN, UK, ††Department of Forest Ecology and Management, Swedish University of Agricultural Sciences, Umeå SE901-83, Sweden*

Abstract

Biochar application to soil is currently widely advocated for a variety of reasons related to sustainability. Typically, soil amelioration with biochar is presented as a multiple-'win' strategy, although it is also associated with potential risks such as environmental contamination. The most often claimed benefits of biochar (i.e. the 'wins') include (i) carbon sequestration; (ii) soil fertility enhancement; (iii) biofuel/bioenergy production; (iv) pollutant immobilization; and (v) waste disposal. However, the vast majority of studies ignore possible trade-offs between them. For example, there is an obvious trade-off between maximizing biofuel production and maximizing biochar production. Also, relatively little attention has been paid to mechanisms, as opposed to systems impacts, behind observed biochar effects, often leaving open the question as to whether they reflect truly unique properties of biochar as opposed to being simply the short-term consequences of a fertilization or liming effect. Here, we provide an outline for the future of soil biochar research. We first identify possible trade-offs between the potential benefits. Second, to be able to better understand and quantify these trade-offs, we propose guidelines for robust experimental design and selection of appropriate controls that allow both mechanistic and systems assessment of biochar effects and trade-offs between the wins. Third, we offer a conceptual framework to guide future experiments and suggest guidelines for the standardized reporting of biochar experiments to allow effective between-site comparisons to quantify trade-offs. Such a mechanistic and systems framework is required to allow effective comparisons between experiments, across scales and locations, to guide policy and recommendations concerning biochar application to soil.

Keywords: biochar, controls, soil, standardization, trade-offs

Received 1 July 2013 and accepted 4 September 2013

Introduction

Biochar is produced from the thermal degradation of organic material in the absence of oxygen. It differs from charcoal in that it is produced with the intention of application to soil rather than as fuel (Lehmann & Joseph 2009). Biochar is often promoted as having several potential benefits or 'wins', including carbon (C) sequestration, soil fertility enhancement, provision of biofuels, pollutant immobilization and disposal of organic wastes, among others. However, in many instances, it is not possible for all benefits to be simultaneously maximized and negative effects may also occur such as priming of soil organic matter (Cross & Sohi 2011; Zimmerman *et al.* 2011), or the introduction of contaminants into the soil (Chan & Xu 2009). Gains over one timeframe could also turn into losses over the longer term, when considering all aspects. The longevity of biochar implies that negative effects can endure in soil, potentially for thousands of years (Glaser *et al.* 2002). The type of feedstock and production conditions affect the properties of the resulting biochar and its associated effects (Jeffery *et al.* 2011; Zimmerman 2010). The question remains whether other effects of biochar application could be as long-lasting as the C storage;

Correspondence: Simon Jeffery, tel. +31 0 317 485749, fax +31 0 317 426101, e-mail: simon.jeffery@wur.nl

clarification of risks and testing across a broad spectrum of soil-crop combinations is vital before large-scale applications can be advocated.

Proponents of biochar deployment frequently draw parallels with historical practices of soil improvement conducted by Amerindians in the Amazon basin (creating the Amazonian Dark Earth or '*terra preta de Indio*', generally referred to as '*terra preta*') (Sombroek 1966). *Terra preta* appears to achieve two 'wins' through achieving high levels of fertility compared to surrounding unamended soils, while demonstrably sequestering C in the soil. Extrapolating to the current day, this suggests circumvention of the 'carbon dilemma' described by Janzen (2006). Janzen stated that a 'paradox' exists with regard to soil organic matter (SOM). SOM stocks should be conserved to sequester C, but at the same time decomposition of SOM is the driving force for increasing overall soil quality through activation of the soil food web and mineralization of nutrients. *Terra preta* appears to store large amounts of C at the same time as achieving relatively high levels of fertility for which the cycling of SOM is conventionally assumed necessary, thereby supposedly achieving the 'win–win' stated earlier.

For biochar precisely the same 'win–win' was proposed as was claimed for *terra preta* (Glaser *et al.* 2002). This led to the proposition that biochar application to soil outside of the Amazon basin has the same potential to sequester carbon and improve soil fertility in the manner described for *terra preta*. However, despite the link between biochar and *terra preta* being almost routinely made in the introductory sections of biochar reports, the justification for this association is often poorly outlined and there is as yet little to no evidence that adding biochar to soils will create *terra preta* like soils.

Recent work has identified further 'wins' that are often associated with biochar. Laird (2008) described biochar as a 'win–win–win' technology due to the production of biofuel during biomass pyrolysis. Other potential benefits or 'wins' include the suppression of greenhouse gas emission such as N_2O and CH_4 from soils (Karhu *et al.* 2011; Cayuela *et al*., 2013a, 2013b); the remediation of contaminated soils as biochar strongly binds to most organic pollutants (Cornelissen *et al.* 2005); and waste disposal (Cascarosa *et al.* 2013).

In this article, we consider the potential benefits that are associated with biochar before discussing why it is not possible for all such benefits to be simultaneously maximized and some negative effects may occur; trade-offs, therefore, are inevitable. We then provide a conceptual framework to allow identification of the potential benefits of biochar application in a given situation and hence the trade-offs between them. Finally, we make a call for standardized experimental techniques and reporting of results to allow robust and policy-relevant judgements to be made. Key to any advances is to acknowledge that biochars often have very different properties (Schimmelpfennig & Glaser, 2012), to the extent that the characterization of a material as 'biochar' can be insufficient and must be connected to a description of the biochar that has been studied or discussed. This critical point should be kept in mind when the term 'biochar' is used in the following sections.

Trade-offs

To conceptualize the trade-offs associated with different uses of biochar, we present a graphical framework that is focused on the most frequently reported potential benefits of biochar (Fig. 1). This framework will aid the identification of the best biochar to apply to a soil in a given situation. The identification of trade-offs could result in maximization of a particular benefit and/or minimization of trade-offs. In combination with life-cycle assessment and other information to determine how to weigh multiple biochar benefits and trade-offs, optimization of feedstock, pyrolysis technology and application modes should be possible (Roberts *et al.* 2010; Sparrevik *et al.* 2013). While we focus here on trade-offs between the most often reported benefits of biochar application, we acknowledge that there are other potential benefits that are not explicitly considered here (e.g., water retention, effects on soil biodiversity through increased refugia, decreasing greenhouse gas emission). Furthermore, currently the axes on Fig. 1 are qualitative owing to insufficient data to produce quantitative axes. There is currently no single metric by which trade-offs can be quantified. Our aim here was to present the concept of trade-offs and the figure is a way of conceptualizing them. Furthermore, the figure will help to direct future research as researchers can aim to produce the necessary data and analyses to allow conversion of the axes from qualitative to quantitative.

Here, we present three different scenarios, represented by Fig 1 a–c. Figure 1a shows an idealized biochar where all five of the most often reported benefits or 'wins' are maximized (solid black line). Figure 1b shows that when biochar is produced for maximum soil fertility effects, this trades-off against biofuel production for reasons discussed below. The residence time of such 'agronomic' biochars in the soil are relatively high and so they still count as a win for climate change mitigation. Figure 1c shows a biochar made from waste products (solid black line) that maximizes the waste disposal win but at a cost of not maximizing soil fertility effects as much as may be possible if a different feedstock was used. As stated above, these figures are only conceptual

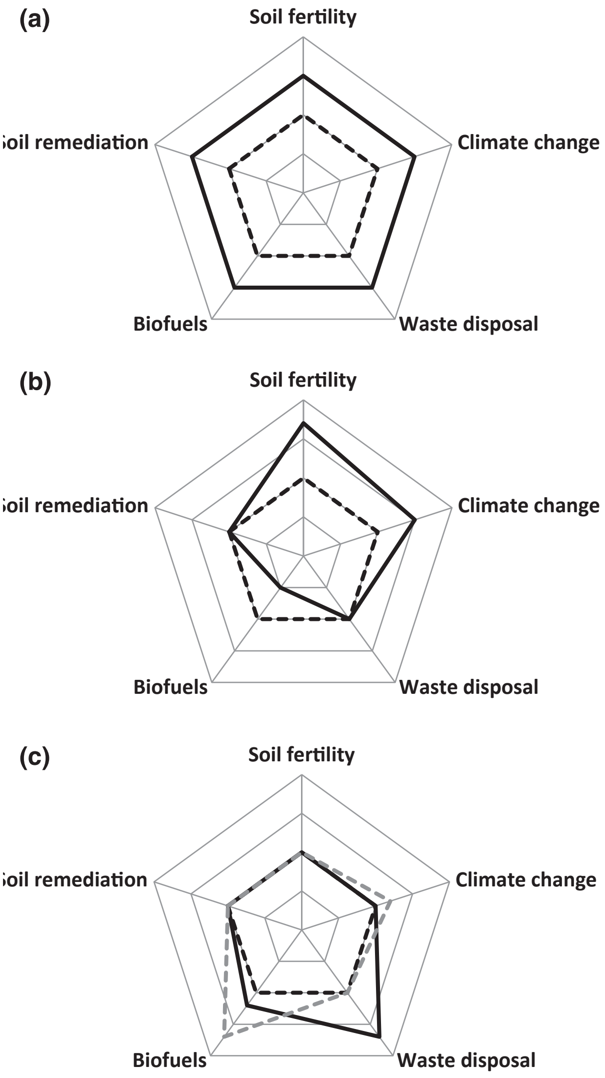

Fig. 1 Conceptualization of important examples of biochar trade-offs: (a) An idealized biochar in which five potential benefits of biochar production and soil application are maximized (black polygon). The dotted black polygon represents the soil's potential with regard to all four factors in the absence of biochar. Therefore, when the black polygon falls within or outside of the border of the dotted polygon this represents a negative effect or positive effect, respectively, when compared to that which would occur in the absence of biochar. (b) Climate change mitigation and crop productivity potential are maximized at the cost of biofuel provision. Other factors remain unaffected compared to what would exist in the absence of biochar. (c) Biochar made from waste products maximizes the waste disposal win of biochar but at a cost of not maximizing soil fertility effects as much as may be possible if a different feedstock were used. The dotted grey polygon allows for comparison with a situation in which the crops are all removed and burned for biofuel as shown by increases in biofuel provision and improvements in climate change mitigation (compared to the energy provided by the biofuel being obtained from fossil fuels).

and there are currently insufficient data to make the axes quantitative. This will change as new data become available.

Climate change mitigation vs. soil fertility

A substantial proportion of biochar can remain undecomposed in the soil over centuries to millennia (Spokas 2010; Zimmerman 2010). This implies that biochar has the potential to mitigate climate change through sequestering C for extended periods, in addition to other mechanisms that reduce greenhouse gas emissions (see below). However, the timeframe over which C remains sequestered in the soil is uncertain as biochar properties vary and depend on feedstock type and processing conditions (e.g., fast vs. slow pyrolysis, pyrolysis temperature) (Spokas 2010; Zimmerman 2010), local climatic conditions, soil type and native soil biota. The main factor affecting the turnover rate of uncharred C in soils is its interaction with the organo-mineral fraction of the soil that can lead to physical and physicochemical stabilization (Liang *et al.* 2008; Schmidt *et al.* 2011; Dungait *et al.* 2012). However, charring confers a greater stability of residues. Direct oxidative ageing methods that compare biochar of various types with natural char in soil suggest that >60% of the C in fresh biochar will remain after 100 years (Cross & Sohi 2013). The chemical composition of the biochar must, therefore, play some role. It is yet to be determined whether the stabilization mechanisms proposed by Schmidt *et al.* (2011), such as physical disconnection and organo-mineral associations, apply to all instances of biochar and their relative importance.

In addition to C sequestration, other effects on the soil greenhouse gas balance have been reported (Sohi *et al.* 2009). Biochar addition resulted in reductions in methane (CH_4) emission from the soil, possibly through increasing soil aeration (Rondon *et al.* 2005; Karhu *et al.* 2011). Furthermore, biochar decreased nitrous oxide (N_2O) emissions from soils (e.g., Schouten *et al.* 2012) via a number of mechanisms (Singh *et al.* 2010; Cayuela *et al.*, 2013a, 2013b). For example, biochar addition might change the microbial community including N_2O-producing organisms, or alter soil structure and thereby the anaerobic volume of the soil in which denitrification takes place (Van Zwieten *et al.* 2009). It may also cause a shift of the product ratio N_2O: N_2 towards N_2 after acid neutralization by alkaline biochar (Dörsch *et al.* 2012), probably alleviating the posttranscriptional inhibition of N_2O reductase in acid soils (Liu *et al.* 2010). Biochar has also been suggested to function as an 'electron shuttle' that facilitates the transfer of electrons to denitrifying microorganisms (Cayuela *et al.*, 2013a, 2013b). A last possible mechanism is that biochar could sorb N_2O

sufficiently to suppress emissions (Van Zwieten *et al.* 2009; Cornelissen *et al.* 2013a). However, it is important to note that the time frame over which these effects persist remains to be determined. Such information will be vital for inclusion in life-cycle assessments that aim to quantify trade-offs.

Increased soil fertility has been reported following biochar application with results being highly variable (Jeffery *et al.* 2011; Biederman & Harpole, 2013). Elucidation of several of the mechanisms behind these observed effects, and quantification of their longevity, is still required. Evidence suggests that the underlying mechanisms may include: increased cation exchange capacity (CEC) (Liang *et al.* 2006); increased plant-available water contents (Karhu *et al.* 2011), improved drainage of excess water (Ayodele *et al.* 2009); a liming effect in acidic soils (Yamato *et al.* 2006) or acid neutralization through the addition of organic anions in biochars produced at low temperatures (Yuan *et al.* 2011); the presence of available nutrients in the biochar (Angst & Sohi, 2013); and increased abundance of soil microbes (O'Neill *et al.* 2009; Liang *et al.* 2010), including mycorrhizal fungi (Warnock *et al.* 2007; Solaiman *et al.* 2010) and decomposers (Zackrisson *et al.* 1996).

The time range over which these effects operate varies considerably. Liming effects and direct nutrient addition effects are likely transient as nutrients are utilized or leached from the system. Such effects likely result from the addition of ash inclusions with the biochar (Mukherjee *et al.* 2011) rather than from the biochar C itself. Other effects may be longer lived, but slower to develop, such as increased CEC and associated nutrient-binding effects through surface oxidation of biochar particles (Liang *et al.* 2006) or increased plant-available water due to the high porosity of biochar particles increasing the water-holding capacity of soils (Karhu *et al.* 2011).There is evidence that some types of biochar have phytotoxic effects depending on the original feedstock and temperature of pyrolysis (Gell *et al.* 2011), but evidence also exists that charcoal can adsorb and inactivate phytotoxic compounds (Hille and Ouden 2005).

It is probable that there will be a trade-off between these two established 'wins'. This could be the definitive case of 'hoarding' rather than 'using' in the context of Janzen's 'Carbon Dilemma' (Janzen 2006). As mentioned above, *terra preta* apparently achieves beneficial results with regard to both C sequestration and maintenance of enhanced levels of fertility compared to surrounding soils (Glaser *et al.* 2002). However, whether such concurrent benefits can also be achieved with biochar addition to soil, and at what level the benefits may trade-off against each other, remains to be determined, particularly in the long term and in temperate regions. For example, Quilliam *et al.* (2012) reported that double dosing and extra loading of biochar in a temperate field plot only provided transient effects on soil fertility over 4 years. However, Mao *et al.* (2012) found that the relatively high CEC of Mollisols in Iowa were due to the high black C contents of the soils. This suggests that biochar (which contains large proportions of black C, its defining chemical feature) has the potential to increase soil fertility through CEC effects, even in temperate soils. It should be noted, however, that mollisols have a high pH with relatively high Ca and P contents. Hence, it is possible that this is an exceptional case. A trade-off may occur between producing biochar that maximizes C sequestration potential vs. biochars with desired agronomic properties, due to the use of oxidation to 'age' biochars (i.e. accelerate the formation of biochar properties that develop over time in the soil) and increase their CEC. Such increased CECs will help the biochar to adsorb cations and reduce nutrient leaching, but the oxidation process leads to a loss of C, thereby reducing C sequestration potential. Evidence suggests that oxidation of biochars may concur with reduced recalcitrance of the biochar in soil, further reducing the biochar's C sequestration potential (Nguyen *et al.* 2010).

Biochar has been reported to cause priming of SOM (both positive and negative), specifically over short periods of time (Steinbeiss *et al.* 2009; Zimmerman *et al.* 2011; but see Cross & Sohi, 2011; Jones *et al.* 2012). This suggests another potential trade-off of biochar even when focusing on C sequestration. Potential short-term losses of native SOM are smaller than the C gain of biochar and might be negligible in many cases (Woolf & Lehmann 2012). However, a trade-off may also exist with faster cycling SOM potentially leading to reductions in the quality of C available in the soil for use by the biota (i.e. as SOM is reduced), even if the overall quantity of C actually increases (i.e. in the form of biochar).

Provision of biofuel vs. production of biochar

Growing crops for biofuel production have given rise to concern about the trade-off regarding the land that is needed to grow these crops, as this land could otherwise be used for food crops or conservation (e.g. Tilman *et al.* 2009). Pyrolysis of waste products to produce biochar while concurrently producing biofuel is thought to at least partially circumvent this problem. However, some trade-offs between stabilized and actively cycling SOM are likely unavoidable because plant material that is removed from the field and converted to biofuel is no longer available for decomposition. Although biochar C when returned to the soil would more than compensate for the C removed in plant material, most of which would decompose, there is an important temporal delay with respect to the soil C balance and soil fertility (Whitman *et al.* 2010, 2011). Furthermore, the quality of the C will differ as discussed previously, and allocation of biochar in different locations than those from where biomass was removed may create additional trade-offs.

Life-cycle assessment has demonstrated that, if potential soil effects are not included, greenhouse gas (GHG) reduction effects are similar when biomass is used to produce biochar as when it is subject to complete combustion for energy production (Roberts *et al.* 2010; Woolf *et al.* 2010). When potential effects such as increased plant growth, reduced N_2O emissions, etc., are factored in, biochar production can be favourable compared to combustion of biomass (Hammond *et al.* 2011). However, there is an inherent trade-off in the pyrolysis process between production of energy and production of biochar; increasing biochar production will always decrease energy production within the same energy pathway (Gaunt & Lehmann 2008). There is thus a trade-off in policy objectives between bioenergy and energy security vs. C abatement (Sohi 2012).

Different pyrolysis conditions and feedstocks lead to the production of different proportions of biochar, condensable gas and bio-oil. This gas and bio-oil can then be collected and used as a biofuel (Mahinpey *et al.* 2009). Slow pyrolysis of feedstock has been calculated to be more energy efficient, in terms of energy input vs. energy output, than production of biofuel through fermentation of feedstock to produce ethanol. Gaunt & Lehmann (2008) reported that where slow pyrolysis technology is optimized to produce biochar for soil application, a reduction in energy output of ca. 30% occurs compared to fast pyrolysis optimized for biofuel production. Thus, maximizing the production of biofuel through fast pyrolysis reduces the amount of biochar that is produced (IEA 2006). If all biochars are similarly stable in the soil (stability is critical for C sequestration potential), this would suggest lower overall C abatement. Slow pyrolysis favours biochar production and by the same token could maximize the C sequestration potential, at the cost of diminished output of biofuel products. Recent work has demonstrated that biochar produced through pyrolysis has beneficial effects when compared to the solid residue (i.e. solid by-products) of bioethanol production in terms of CO_2 and N_2O emissions (Cayuela *et al.* 2013a). Also, a smaller quantity of biochar of high stability can have the same C abatement value as a larger quantity of less stable biochar (Crombie *et al.* 2012). Further work is needed to compare energy pathways to allow quantification of trade-offs that inevitably occur between energy and biochar production.

Changing pyrolysis conditions to maximize either biochar or biofuel production is likely to affect the C : N : P stoichiometry of the resulting biochar. However, while the majority of the beneficial effects of biochar on crop productivity stems from pH effects and the ability of biochars to retain N and P from other sources, evidence suggests that N and P can be available from some types of biochars (De la Rosa and Knicker, 2011; Wang *et al.* 2012; Chintala *et al.* 2013). Differential losses of C and N during pyrolysis occur as a function of temperature (Enders *et al.* 2012). Additional properties (such as those affecting soil pH) may impact the bioavailability of nutrients, particularly P, present in both biochar and bulk soil. This confounds prediction of the likely effects of a given biochar on the stoichiometry of the soil solution. However, it is evident that trade-offs will occur between maximizing biofuel production from pyrolysis and producing biochar with optimal C : N : P stoichiometry for a given soil/crop/climate combination (Gaunt & Lehmann 2008). Therefore, the effects of different pyrolysis conditions on the C : N : P stoichiometry of biochars and their effects on bioavailability in the soil is an important area of future research.

Feedstock selection vs. the use of wastes

One readily apparent trade-off regarding choice of feedstock for biochar production is that of stability of the resulting biochar vs. its nutrient content. For example, evidence suggests that biochars made from poultry litter support greater increases in crop productivity than those made from wood (Jeffery *et al.* 2011) which probably at least partly result from higher nutrient contents in this feedstock. However, biochars made from poultry litter are less stable in the soil than those made from wood (Singh *et al.* 2012).

The paucity of feedstocks in many parts of the world suggests that another trade-off may occur. In such areas, such as much of Africa, all of the aboveground crop biomass produced, and not just the grain or fruit, is utilized for a range of purposes such as animal feed, roofing materials, mulch to reduce water requirements or incorporation into the soil to improve organic matter content. Therefore, it may be difficult to reserve significant amounts of feedstock for biochar production, and environmental degradation may result when alternative feedstocks for biochar production are sought (e.g., through deforestation). As such, entry points for adoption of biochar technologies may occur through substitution of current technologies, such as the switch from burning fire wood to pyrolysis stoves for cooking (Torres-Rojas *et al.* 2011). However, the implementation of biochar technology in Africa can be a contentious issue for a variety of social reasons (for further discussion see Leach *et al.* 2012).

One method that potentially circumvents the problem of paucity of feedstocks is the use of waste materials to produce biochar. In theory, any C-based feedstock can be pyrolysed to produce biochar, and so biochar production has the potential to mitigate the increasing global problem of waste disposal. To date, a wide range of waste streams have been considered and tested, including biosolids (Chan & Xu, 2009), tannery wastes (Muralidhara, 1982), paper sludge (Rajkovich *et al.* 2011) and sewage and wastewater sludge (Bridle & Pritchard 2004; Hossain *et al.* 2010). The type of feedstock affects the properties of the resulting biochar (Kloss *et al.* 2012) in terms of crop yield effects (Jeffery *et al.* 2011) and recalcitrance in the soil (Zimmerman 2010; Singh *et al.* 2012). Furthermore, it is likely to affect whether the resulting biochar is classified as a waste product, with implications regarding its permissibility for soil application (Sohi *et al.* 2010). Legislative issues surrounding biochar application to soils produced from waste products, and the classification of such biochar in terms of policy, is vital before its large-scale application can be implemented. Such a discussion is beyond the scope of this article but has been covered in terms of European policy implications by Van Den Bergh (2009).

The type of feedstock and pyrolysis conditions also affect the types and concentration of contaminants in the resulting biochar. For example, heavy metals, which are generally found in high concentrations in sewage sludge and biosolids, are increased in concentrations following pyrolysis (Chan & Xu 2009). To assess potential trade-offs, an assessment of whether heavy metals pose a greater risk in the sewage/biosolids stream or in the soil biochar stream is required. Such an assessment would necessarily examine whether such contaminants are more or less bioavailable in biochar compared to the original feedstock. Biochar can reduce the bioavailability of contaminants such as heavy metals that are already present in the soil (Park *et al.* 2011). In addition, while biochar can contain polycyclic aromatic hydrocarbons (PAHs; by-products of incomplete combustion) these amounts are low and hardly bioavailable such that biochars leach PAHs far below water quality criteria (Hale *et al*., 2012). However, as discussed previously biochars can vary considerably in their physical and chemical properties and as such bioavailability of PAHs (and other contaminants) should be monitored. Owing to the high cost of such monitoring this presents a potential hurdle to wide-scale implementation of biochar application to soil.

The use of waste products to produce biochar can lead to biochar with suboptimal properties. One example is the high sodium (Na) content that biochars produced from food wastes sometimes contain (Rajkovich *et al.* 2011). However, Na is mobile in soil and will leach out relatively quickly. Nevertheless, such biochars, which could reduce crop growth initially, need to be applied in smaller amounts, or they need to be applied well in advance of planting so that the Na has time to leach out. This leads to a further trade-off between applying biochar when possible (e.g., when labour is available, at an appropriate time to plough the field) compared to when it would function optimally (e.g., sufficiently before planting to allow time for Na to be leached).

The fact that any C-rich feedstock can be turned into biochar has also led to suggestions by the popular media that other waste materials, such as plastic, could be used for biochar production too (e.g. Harrabin 2009; Lovelock 2009; Black 2010). However, 'plastic' denotes a wide range of different compounds, and what is suitable for use as a feedstock needs further research both in terms of biofuel production and suitability of the produced biochar for soil application. While pyrolysing plastics may release compounds such as syngas that could be used for energy, from a climate change mitigation viewpoint, plastics are generally highly recalcitrant and it seems probable that subjecting them to pyrolysis will release more C into the atmosphere than would occur if they were buried in a landfill. If C sequestration is the goal, this is perhaps not a sensible option.

Contaminated-soil remediation vs. soil fertility

Addition of activated carbon to contaminated soil and sediment is sometimes used as a remediation strategy. Biochar also has the potential to perform this role as it has a high sorption capacity for persistent organic pollutants and pesticides (although generally lower than activated carbon) (Luthy *et al.* 1997; Cornelissen *et al.* 2005). While the use of activated carbon is an established practice for soil remediation, amending biochar to soil or sediment can also lower the bioavailable concentrations of pollutants and pesticides by one to two orders of magnitude. This is similar to what can be achieved through adding activated carbon, but is potentially less costly (Yang & Sheng 2003; Ghosh *et al.* 2011; Jakob *et al.* 2012).

Soil fertility increases resulting from biochar addition are often relatively modest in well-fertilized soils in temperature regions (Jeffery *et al.* 2011). As the effect of biochar on pollutant immobilization can be rather strong, the use of biochar in the temperate zone may be more relevant for alleviating soil and sediment contamination than for promoting crop growth. Life-cycle assessment (LCA) is required to examine the relative benefits of both uses. For example, one LCA study showed that activated biochar was greatly superior to an anthracite-based activated carbon for remediating a dioxin-contaminated fjord system in Norway, even though the biochar was slightly less chemically active in immobilizing dioxins, owing to its C sequestration potential (Sparrevik *et al.* 2011).

Other Trade-Offs

Further to the above-mentioned trade-offs between the most often reported 'wins' (also see Fig. 1), other trade-offs related to biochar use are evident. Here, we provide three further examples of such trade-offs.

Biochar and conservation tillage

A seldom acknowledged trade-off that may occur when applying biochar to soil is that it is necessary to bury the biochar in order to prevent it from being eroded by wind or water. This is usually done by mixing the biochar into the topsoil, either mechanically or by hand. Such mixing requires disturbance and cultivation of the soil, which promotes native SOM loss as well as potentially other side effects in those situations where incorporation is not part of on-going management. Therefore, wide-scale biochar application may trade-off against the benefits that no-till farming brings. Direct surface application (e.g., added to slurry or in muck spreading) is also possible, although such application techniques run the risk of the biochar being eroded by wind or rain (Rumpel *et al*., 2006). A possible response to mitigate this potential trade-off is to combine biochar with minimum tillage conservation farming practices, and apply biochar, for example, in the hoe basins or rip lines where cultivation takes place (Hobbs *et al.* 2008; Giller *et al.* 2009). This would also reduce the amount of biochar needed for fertility effects (Cornelissen *et al*., 2013b) and so helps circumvent some of the trade-offs that occur due to competition for feedstocks. However, such a strategy is labour intensive and unlikely to be implemented in large-scale arable systems, unless it can be combined with application of manures which may already be part of on-going soil management.

Biochar production and human health

Another potential trade-off exists between soil fertility and human health effects. For example, a biochar produced from maize cobs proved to be very effective for soil fertility in Zambia, increasing harvests by up to a factor four (Cornelissen et al., 2013b). However, the only realistic way for these farmers to produce biochar in the near term is through traditional kilns that emit particles <10 μm (PM10), CH_4 and carbon monoxide (CO), as they cannot afford cleaner retort pyrolysis technologies. Charcoal particles (i.e. soot) have adverse effects on human health such as causing the lung disease pneumoconiosis, in addition to contributing to global warming (Baveye 2007). Thus, while beneficial to the farmers in terms of crop yield, a complete LCA using technology that was available to farmers showed that biochar implementation may have adverse health effects and was only slightly beneficial for climate change mitigation (since the emitted CH_4 is a strong GHG, offsetting C sequestration) (Sparrevik *et al.* 2013). The areas that are most likely to experience these problems are also those where the potential benefit of biochar to crop production appears to be the highest: smallholder farms in developing countries (Sparrevik *et al.* 2013). The level of this trade-off is likely to vary and could be reduced or eliminated with the use of lower emission pyrolysis cook stoves.

Increased resistance to pests vs. decreased pesticide effectiveness

Elad *et al.* (2010) and Meller Harel *et al.* (2012) have reported that biochar can induce systemic resistance in some plant species (peppers, tomatoes and strawberries) to some fungal pathogens and other pests such as the broad mite (*Polyphagotarsonemus latus*). Should this effect also be found for field-grown crops, biochar may reduce the need for fungicides and potentially other pesticides. However, a trade-off likely exists as some evidence also suggests that pesticides are less effective when applied following biochar application (Yu *et al.* 2009; Graber *et al.* 2012). The issue of whether increased resistance to pests is sufficient to counter the reduced efficacy of pesticides and the wider-scale implications of biochar application to soil on pest populations requires further work.

Experimental Set-Up

There is now a large body of pertinent and robust literature on the effects of biochar on soil properties and processes. These include crop yields from multi-year field experiments (e.g. Kimetu *et al.* 2008; Gaskin *et al.* 2010; Major *et al.* 2010; Jones *et al.* 2012; Güereña *et al*., 2013), possible priming effects on soil organic matter (e.g. Zimmerman *et al.* 2011) and disease resistance (Meller Harel *et al.* 2012). However, most research is still phenomenological and descriptive; there is comparatively little research studying mechanisms behind observed effects (Sohi *et al.* 2010; Güereña et al., 2013). Such a mechanistic understanding is imperative to allow the assessment of potential trade-offs that is needed before large-scale application of biochar should be promoted. For this reason, we provide recommendations for experimental design to guide progress towards on the one hand (i) a mechanistic (i.e. reductionist) understanding and on the other hand; (ii) a systems (i.e. holistic) understanding.

Use of experimental controls

In order to allow elucidation of underlying mechanisms in comparison to the life-cycle effects of implementing a biochar system, quite different experimental designs are required. Traditionally, biochar effects have been assessed through comparisons with negative controls (i.e. with no addition). However, considerable scope also exists for comparing biochar addition with positive controls to address systems-level questions. The choice of controls (negative and/or positive) should depend on the situation and the hypothesis being tested. One way forward in this field could involve the use of multiple positive controls; having only one control allows the testing of only one hypothesis, whereas multiple controls allow the testing of several competing hypotheses. In addition, resolving questions about the systems impact of implementing biochar may require different controls than resolving questions about soil and plant processes. Therefore, in many cases, comparisons with positive controls that contain the uncharred feedstocks from which the biochar was produced are desirable. For example, many of the beneficial effects of adding biochar produced from poultry manure to a soil may also occur when adding the un-pyrolysed poultry litter (Chan *et al.* 2008). Inclusion of positive controls in experimental designs is vital to allow for the effects of biochar *per se* to be quantified and to move towards a systems understanding of the effects of biochar application to soil.

When soil fertility effects are the main subject of investigation, a positive control might involve the addition of the un-pyrolysed and/or ashed feedstock (i.e. a positive control). This would allow quantification of the effects of pyrolysis on increasing the availability of nutrients relative to fresh organic matter, and controlling the provision of soluble nutrients present in ash. It would reflect direct soil incorporation and combustion as two alternative uses for the same biomass. To establish trade-offs in feedstock use, rates of addition should probably be based on equivalent mass of the starting material. This would allow quantification of the effects of reduced residue incorporation on soil processes that are likely to occur if such residue is removed for biochar production. Such approaches make sense in systems and life-cycle studies, whereas process-based investigations may benefit from other sets of controls. The intent and purpose of the research is therefore important for driving decisions on choice of controls.

In addition to a systems understanding, a mechanistic understanding is also critical. It has been reported repeatedly that biochar application reduces CH_4 and N_2O emissions from the soil (Singh *et al.* 2010; Liu *et al.* 2011; Cayuela *et al.*, 2013a, 2013b). However, without understanding the mechanisms behind these results it is not possible to robustly extrapolate to other environments, soil types or soil and crop combinations. As mentioned above, possible mechanisms include biochar reducing soil N availability directly, or indirectly through increasing P availability and thereby plant N uptake and immobilization. Such hypothesized mechanisms could be tested experimentally through comparison of biochar addition with a carefully selected positive control (i.e. with addition) as well as a negative control (i.e. without addition). For example, if the reduced N_2O emissions occur as a result of reduced N availability due to changes in the C : N ratio, comparisons with positive control treatments containing high C : N ratio uncharred residues such as wood or straw may be informative. This would be pertinent if N-limited crops such as cereals were grown. Alternatively, increasing P availability in the positive controls and monitoring N_2O emissions would allow testing of the hypothesis that changes in P availability increases plant N uptake, reducing N surplus within the soil thereby reducing N_2O emissions.

Types of positive control

Biochar often leads to acid neutralization or a liming effect in acidic soils, although the strength of the liming effect will vary between biochar types. Soil pH plays an important role in the availability of nutrients and other ions in the soil, some of which can increase to toxic levels in acidic soils (e.g. Al), (Liang *et al.* 2006). Therefore, a control treatment in which the pH is adjusted to bring the soil pH in line with the biochar treatments would often be useful (Hass *et al.* 2012) to allow distinguishing those biochar effects that occur beyond pH effects. Other types of positive controls may include the use of plastic chips of the same size as biochar particles, or perlite to increase the soil WHC. Again, the choice of positive control will depend on the hypothesis being tested.

Finally, additions of nutrients to positive controls at the same rate as those present in leachable ash would allow differentiation between effects resulting from changes in nutrient availability relative to direct or indirect effects of biochar on the soil microbial community, water retention, etc. Quantification of nutrients could take place as a pilot experiment in which the amount of available nutrients such as N, P and K following biochar addition are analysed, and then this amount applied in the main experiment as a control. While the rate of release may be different, it would at least allow some control of nutrient effects and so aid identification of biochar effects without the influence of nutrients. While use of such a control has several potential weaknesses that would need to be overcome (such as different release rates of nutrients from biochar compared to fertilizer) efforts in this direction are necessary to allow identification of biochar effects beyond nutrient effects. Inclusion of positive controls in an experimental design should not (and indeed, must not in most instances) preclude the inclusion of negative controls as well.

Time

Positive controls seek to match the projected, initial effects of a biochar addition. Over time the residual effects of a biochar and positive control are likely to diverge. Thus, an alternative approach is to remove a property of the biochar such as the potential nutrient content. This could be achieved by understanding the pattern of nutrient release from biochar and repeated leaching (Angst & Sohi 2013) or, for the effects of time on CEC, biochar 'ageing' (Cross & Sohi 2013).

A tiered approach to choice of controls

To achieve a mechanistic understanding of the effects of biochar, we suggest that moving forward with biochar research may benefit from a three-tiered approach, a framework for which is provided in Fig. 2. We suggest that, when the question at hand warrants it, a useful approach for experiments looking at the effects of biochar application could involve the use of both positive (i.e. addition of un-pyrolysed feedstock) and negative (i.e. no addition) controls (Level One controls; Fig. 2). It may also be useful to include 'subtractive' controls (Fig. 2). This would consist of controls in which the amount of un-pyrolysed feedstock added is reduced by an equivalent amount to that which is removed to produce the biochar. This would allow investigation of the effects of reduced C inputs into the soil due to removal of crop residues for biochar production.

However, to achieve a full mechanistic understanding of biochar effects it is necessary to distinguish between the effects of the biochar itself and effects such as nutrients added through ash, which are likely to be short-lived. For such experiments, we suggest that further controls would likely be necessary; examples of which are given in Level Two of Fig. 2. Inclusion of such controls has the added advantage of overcoming some of the shortcomings of relatively short-term experiments. For example, nutrient or liming effects associated with biochar are likely to be relatively short-lived as nutrients are utilized or leached from the soil. Addition of nutrients to controls at the quantities as present in biochar will allow for effects beyond nutrient additions to be quantified. While the rate of release between biochar and fertilizer (including slow-release fertilizer) is likely different, this may give an indication of what benefits remain once such nutrients are no longer present or available.

Finally, extrapolation of results from biochars produced from the same feedstocks but under different pyrolysis conditions should be undertaken with caution, despite being a potentially appealing short-cut in maximizing our understanding of biochar and its effects within the soil and wider environment. Biochars produced under different conditions likely show differences in terms of ash content, and therefore differences in liming effects, nutrient concentrations and ratios. It is vital that studies reporting the effects of biochar application to soil also include as much information as possible about the biochar characteristics and manufacturing or preparation conditions.

Reporting of Results

Comprehensive reviews and meta-analyses are important for aggregating effects and allowing more robust extrapolation, and hence guidance of policy and directing of future research. They are particularly pertinent for areas in which experimental results at a relatively small scale need to be up-scaled due to proposed large-scale implementation, as is the case for biochar (similar to other areas such as compost research which suffers from many of the same problems as biochar research). It would therefore be useful if data presented in primary research manuscripts comply with the guidelines described by the Cochrane Collaboration (Higgins and Green 2009) concerning the production of systematic reviews and meta-analyses and as first suggested for biochar research by Verheijen *et al*. (2010).

To maximize the utility of studies for meta-analyses, as many auxiliary variables as possible should be reported. These should include both biochar and soil properties; e.g. CEC and pH of soil before and after application, nutrient content of biochar, pyrolysis conditions (especially temperature), feedstock, soil properties, crop or plant type, and fertilizer used (type and dose), as well as climatic information, particle size, and mode and depth of incorporation where possible. Supporting such advances, approaches to provide standards or specification have emerged, for example, by the International Biochar Initiative (IBI 2012). Use of a reference biochar, produced from the same feedstock under the

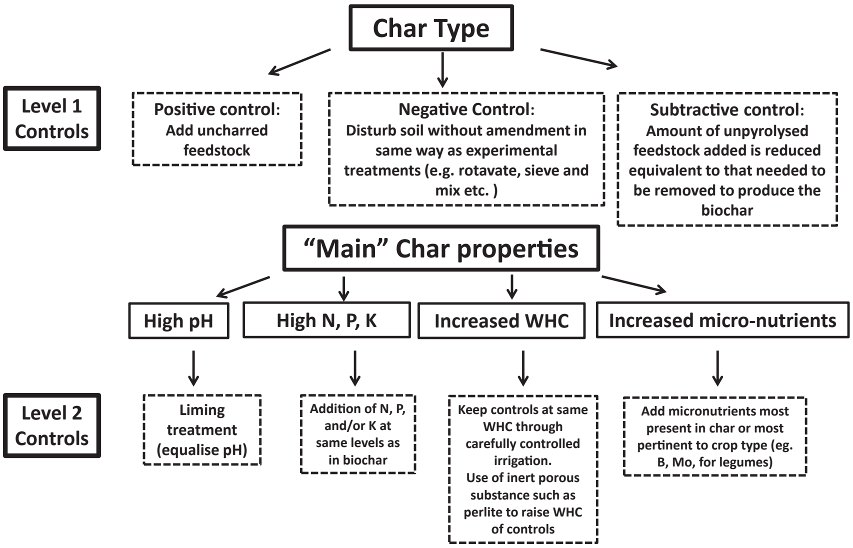

Fig. 2 Flow chart for identification of appropriate controls. Level 1 controls represent the minimum controls to allow the impacts of biochar to be identified and quantified. Level 2 controls represent the controls necessary to gain a mechanistic understanding of the impacts of biochar application to soil.

same conditions, will aid cross-site comparisons and therefore quantification of differences in interactions of a given biochar with contrasting soil types.

Furthermore, it is desirable that such information is reported in a standardized way. In many experiments, particularly those involving pots or mesocosms, application rate has been variably reported as% w/w (e.g. Meller Harel *et al.* 2012), as t ha^{-1} of C equivalence (e.g. Woolf *et al.* 2010) or as t ha^{-1} of biochar mass equivalence (e.g. Karhu *et al.* 2011). This complicates comparison of between-site results and standardization of units across papers as much as is possible is therefore useful. We recommend that application rates should be reported as t ha^{-1} mass equivalents, because not all researchers have the necessary equipment to analyse their biochar in units of t C ha^{-1}. This recommendation is in addition to study-specific reporting (such as % w/w in the case of microcosm experiments) where different units may be warranted such as volume if targeting the physical impact on bulk soil, or concentration if targeting chemical effects.

Future Research and Conclusions

Owing to the extensive range of combinations of biochar, soils and plants, much research still needs to be undertaken to understand the large variety of resulting interactions and their effects. As research progresses, it will be possible to make extrapolations with increasing robustness as, for example, the database upon which meta-analysis can be performed grows. Such information is vital to guide the development of certification schemes such as that proposed by the International Biochar Initiative, and The European Biochar Certificate, which is already implemented in part of Europe, as well as to guide policy. However, as discussed above, trade-offs will almost inevitably occur between the potential 'wins' following biochar application to soil; such trade-offs are generally not yet quantified, or even identified. Experimental designs that consider such trade-offs between the wins should therefore be a priority.

To effectively guide future research, a representativity analysis (i.e. an analysis of which soil, biochar, crop combinations have been studied and whether it is representative of combinations ultimately deployed) is urgently needed to allow identification of the gaps in current research. For example, the meta-analysis of Jeffery *et al.* (2011) indicates that relatively few biochar experiments have taken place in temperate regions, or have focused on major crops such as potato. A representativity analysis would be a very useful tool for researchers as well as providing guidance to policy makers as to where to direct research funding.

In conclusion, the large and growing body of research reported in the literature demonstrates the potential of biochar application to soil to provide a range of benefits. However, such benefits are unlikely to be maximized in all situations and trade-offs will inevitably occur between them. Furthermore, there are currently insufficient data in the literature to draw conclusions concerning biochar production and application to soil in all situations. Published long-term experiments in particular are lacking and are vital to assess the long-term implications of biochar application.

To quantify and predict such trade-offs it is necessary to move towards a mechanistic understanding of the effects of biochar application. One way to move towards such an understanding is through judicious choice of controls. Standardized reporting of results will aid cross-site comparisons and aid quantitative reviews. Such steps will allow biochar research to move forwards while remaining firmly grounded in robust science, and will allow policy to be effectively developed to maximize the potential benefits of biochar while concurrently avoiding or minimizing negative effects.

Acknowledgements

We gratefully acknowledge funding from the Graduate School Production Ecology and Resource Conservation in Wageningen, the Netherlands for financial support and we thank the participants of the biochar workshop 'Biochar: The Soil is the Limit' in Wageningen, May 2012 for their part in the interesting discussions that have led to the production of this manuscript.

References

Angst T, Sohi S (2013) Establishing release dynamics for plant nutrients from biochar. *Global Change Biology and Bioenergy*, **5**, 221–226.

Ayodele AP, Oguntunde A, Joseph S, de Souza Dias M (2009) Numerical analysis of the impact of charcoal production on soil hydrological behavior, runoff response and erosion susceptibility. *Revista Brasileira de Ciência do Solo*, **33**, 137–145.

Baveye PC (2007) Soils and runaway global warming: terra incognita. *Journal of Soil and Water Conservation*, **62**, 139A–139A.

Biederman LA, Harpole WS (2013) Biochar and its effects on plant productivity and nutrient cycling: a meta-analysis. *Global Change Biology and Bioenergy*, **5**, 202–214.

Black R (2010) Delivering biochar's triple win. In: Earth Watch. BBC. Available at: http://www.bbc.co.uk/blogs/thereporters/richardblack/2010/08/last_year_you_could_hardly.html. (accessed 23 March 2013).

Bridle TR, Pritchard D (2004) Energy and nutrient recovery from sewage sludge via pyrolysis. *Water Science Technology*, **50**, 169–175.

Cascarosa E, Boldrin A, Astrup T (2013) Pyrolysis and gasification of meat-and-bone-meal: energy balance and GHG accounting. *Waste Management*, **33**, 2501–2508.

Cayuela ML, Kuikman P, Bakkerd R, Van Groenigen JW (2013a) Tracking C and N dynamics and stabilization in soil amended with wheat residue and its corresponding bio-ethanol by product: a 13C/15N study. *Global Change Biology and Bioenergy*. doi:10.1111/gcbb.12102.

Cayuela ML, Sánchez-Monedero MA, Roig A, Hanley K, Enders A, Lehmann J (2013b) Biochar and denitrification in soils: when, how much and why does biochar reduce N2O emissions? *Scientific Reports*, **3**, 1732.

Chan KY, Van Zwieten L, Meszaros I, Downie A, Joseph S (2008) Using poultry litter biochars as soil amendments. *Soil Research*, **46**, 437–444.

Chan KY, Xu Z (2009) Nutrient properties and their enhancement. In: *Biochar for Environmental Management: Science and Technology* (eds Lehmann J, Joseph S), pp. 67–84. Earthscan, London.

Chintala R, Schumacher TE, McDonald LM, Clay DE, Malo DD, Papiernik SK, Clay SA, Julson JL (2013) Phosphorus sorption and availability from biochars and soil/biochar mixtures. *Clean – Soil, Air, Water*. doi:10.1002/clen.201300089.

Cornelissen G, Gustafsson O, Bucheli TD, Jonker MTO, Koelmans AA, Van Noort PCM (2005) Extensive sorption of organic compounds to black carbon, coal, and kerogen in sediments and soils: mechanisms and consequences for distribution, bioaccumulation, and biodegradation. *Environmental Science & Technology*, **39**, 6881–6895.

Cornelissen G, Rutherford DW, Arp HPH, Dörsch P, Kelly CN, Rostad CE (2013a) Sorption of pure N_2O to biochars and other organic and inorganic materials in anhydrous systems. *Environmental Science & Technology*, **47**, 7704–7712.

Cornelissen G, Martinsen V, Shitumbanuma V, et al. (2013b) Biochar effect on maize yield and soil characteristics in five conservation farming sites in Zambia. *Agronomy*, **3**, 256–274.

Crombie K, Mašek O, Sohi SP, Brownsort P, Cross A (2012) The effect of pyrolysis conditions on biochar stability as determined by three methods. *Global Change Biology and Bioenergy*, **5**, 122–131.

Cross A, Sohi SP (2011) The priming potential of biochar products in relation to labile carbon contents and soil organic matter status. *Soil Biology and Biochemistry*, **43**, 2127–2134.

Cross A, Sohi SP (2013) A method for screening the relative long-term stability of biochar. *Global Change Biology and Bioenergy*, **5**, 215–220.

De la Rosa JM, Knicker H (2011) Bioavailability of N released from N-rich pyrogenic organic matter: an incubation study. *Soil Biology and Biochemistry*, **43**, 2368–2373.

Dungait JJ, Hopkins DW, Gregory AS, Whitmore AP (2012) Soil organic matter turnover is governed by accessibility not recalcitrance. *Global Change Biology*, **18**, 1781–1796.

Dörsch P, Braker G, Bakken LR (2012) Community-specific pH response of denitrification: experiments with cells extracted from organic soils. *FEMS Microbiology Ecology*, **79**, 530–541.

Elad Y, David DR, Harel YM, Borenshtein M, Kalifa HB, Silber A, Graber ER (2010) Induction of systemic resistance in plants by biochar, a soil-applied carbon sequestering agent. *Phytopathology*, **100**, 913–921.

Enders A, Hanley K, Whitman T, Joseph S, Lehmann J (2012) Characterization of biochars to evaluate recalcitrance and agronomic performance. *Bioresource Technology*, **114**, 644–653.

Gaskin JW, Speir RA, Harris K, Das KC, Lee RD, Morris LA, Fisher DS (2010) Effect of peanut hull and pine chip biochar on soil nutrients, corn nutrient status, and yield. *Agronomy Journal*, **102**, 623–633.

Gaunt JL, Lehmann J (2008) Energy balance and emissions associated with biochar sequestration and pyrolysis bioenergy production. *Environmental Science & Technology*, **42**, 4152–4158.

Gell K, Van Groenigen JW, Cayuela ML (2011) Residues of bioenergy production chains as soil amendments: immediate and temporal phytotoxicity. *Journal of Hazardous Materials*, **186**, 2017–2025.

Ghosh U, Luthy RG, Cornelissen G, Werner D, Menzie CA (2011) In-situ sorbent amendments: a new direction in contaminated sediment management. *Environmental Science & Technology*, **45**, 1163–1168.

Giller KE, Witter E, Corbeels M, Tittonell P (2009) Conservation agriculture and smallholder farming in Africa: the heretics view. *Field Crops Research*, **114**, 23–34.

Glaser B, Haumaier L, Guggenberger G, Zech W (2001) The '*Terra preta*' phenomenon: a model for sustainable agriculture in the humid tropics. *Naturwissenschaften*, **88**, 37–41.

Glaser B, Lehmann J, Zech W (2002) Ameliorating physical and chemical properties of highly weathered soils in the tropics with charcoal-a review. *Biology and fertility of soils*, **35**, 219–230.

Graber ER, Tsechansky L, Gerstl Z, Lew B (2012) High surface area biochar negatively impacts herbicide efficacy. *Plant and Soil*, **353**, 95–106.

Güereña D, Lehmann J, Hanley K, Enders A, Hyland C, Riha S (2013) Nitrogen dynamics following field application of biochar in a temperate North American maize-based production system. *Plant and Soil*, **365**, 239–254.

Hale SE, Lehmann J, Rutherford D, et al. (2012) Quantifying the total and bioavailable polycyclic aromatic hydrocarbons and dioxins in biochars. *Environmental Science and Technology*, **46**, 2830–2838.

Hammond J, Shackley S, Sohi S, Brownsort P (2011) Prospective life cycle carbon abatement for pyrolysis–biochar systems in the UK. *Energy Policy*, **39**, 2646–2655.

Harrabin R (2009) Biochar: Is the hype justified? In: *Science and Environment*. Available at: http://news.bbc.co.uk/2/hi/science/nature/7924373.stm (accessed 21 March 2013).

Hass A, Gonzalez JM, Lima IM, Godwin HW, Halvorson JJ, Boyer DG (2012) Chicken manure biochar as liming and nutrient source for acid Appalachian soil. *Journal of Environmental Quality*, **41**, 1096–1106.

Higgins JPT, Green S (eds) (2009) *Cochrane Handbook for Systematic Reviews of Interventions Version 5.0.2* [updated September 2009]. The Cochrane Collaboration, www.cochrane-handbook.org. (accessed 19 January 2013).

Hille M, Ouden JD (2005) Charcoal and activated carbon as an adsorbate of phytotoxic compounds – a comparative study. *Oikos*, **108**, 202–207.

Hobbs PR, Sayre K, Gupta R (2008) The role of conservation agriculture in sustainable agriculture. *Philosophical Transactions of the Royal Society B*, **363**, 543–555.

Hospido A, Moreira MT, Martin M, Rigola M, Feijoo G (2005) Environmental evaluation of different treatment processes for sludge from urban wastewater treatments: anaerobic digestion versus thermal processes. *International Journal of Life Cycle Analysis*, **5**, 336–345.

Hossain MK, Strezov V, Yin Chan K, Nelson PF (2010) Agronomic properties of wastewater sludge biochar and bioavailability of metals in production of cherry tomato (*Lycopersicon esculentum*). *Chemosphere*, **78**, 1167–1171.

IBI (2012) *Standardized Product Definition and Product Testing Guidelines for Biochar that is Used in Soil*. International Biochar Initiative, IBI-STD-01. http://www.biocharinternational.org/sites/default/files/Guidelines_for_Biochar_That_Is_Used_in_Soil_Final.pdf. (accessed 15 June 2013).

IEA (2006) Annual Report - IEA Bioenergy. Task 34 Pyrolysis of Biomass. International Energy Agency, Paris, France.

Jakob L, Hartnik T, Henriksen T, Elmquist M, Brändli RC, Hale SE, Cornelissen G (2012) PAH-sequestration capacity of granular and powder activated carbon amendments in soil, and their effects on earthworms and plants. *Chemosphere*, **88**, 699–705.

Janzen HH (2006) The soil carbon dilemma: shall we hoard it or use it? *Soil Biology and Biochemistry*, **38**, 419–424.

Jeffery S, Verheijen FGA, Van der Velde M, Bastos AC (2011) A quantitative review of the effects of biochar application to soils on crop productivity using meta-analysis. *Agriculture, Ecosystems and Environment*, **144**, 175–187.

Jones DL, Rouska J, Edwards-Jones G, DeLucaa TH, Murphy DV (2012) Biochar-mediated changes in soil quality and plant growth in a three year field trial. *Soil Biology and Biochemistry*, **45**, 113–124.

Karhu K, Mattila T, Bergström I, Regina K (2011) Biochar addition to agricultural soil increased CH4 uptake and water holding capacity – Results from a short-term pilot field study. *Agriculture, Ecosystems & Environment*, **140**, 309–313.

Kimetu JM, Lehmann J, Ngoze SO, et al. (2008) Reversibility of soil productivity decline with organic matter of differing quality along a degradation gradient. *Ecosystems*, **11**, 726–739.

Kloss S, Zehetner F, Dellantonio A, et al. (2012) Characterization of slow pyrolysis biochars: effects of feedstocks and pyrolysis temperature on biochar properties. *Journal of Environmental Quality*, **41**, 990–1000.

Laird DA (2008) The Charcoal Vision: a win - win - win scenario for simultaneously producing bioenergy, permanently sequestering carbon, while improving soil and water quality. *Agronomy Journal*, **100**, 178–181.

Leach M, Fairhead J, Fraser J (2012) Green grabs and biochar: revaluing African soils and farming in the new carbon economy. *Journal of peasant studies*, **39**, 285–307.

Lehmann J, Joseph S (2009) *Biochar for Environmental Management: Science and Technology*. Earthscan, Oxford.

Lehmann J, Rillig MC, Thies J, Masiello CA, Hockaday WC, Crowley D (2011) Biochar effects on soil biota - A review. *Soil Biology and Biochemistry*, **43**, 1812–1836.

Liang B, Lehmann J, Solomon D, et al. (2006) Black Carbon Increases Cation Exchange Capacity in Soils. *Soil Science Society of America Journal*, **70**, 1719–1730.

Liang B, Lehmann J, Solomon D, et al. (2008) Stability of biomass-derived black carbon in soils. *Geochimica et Cosmochimica Acta*, **72**, 6069–6078.

Liang BQ, Lehmann J, Sohi SP, et al. (2010) Black carbon affects the cycling of non black carbon in soil. *Organic Geochemistry*, **41**, 206–213.

Liu B, Mørkved PT, Frostegård Å, Bakken LR (2010) Denitrification gene pools, transcription and kinetics of NO, N_2O and N_2 production as affected by soil pH. *FEMS Microbiology Ecology*, **72**, 407–417.

Liu Y, Yang M, Wu Y, Wang H, Chen Y, Wu W (2011) Reducing CH_4 and CO_2 emissions from waterlogged paddy soil with biochar. *Journal of Soils and Sediments*, **11**, 930–939.

Lovelock J (2009) James Lovelock on biochar: Let the Earth remove CO_2 for us. In: Environment. The Guardian. Available at: http://www.guardian.co.uk/environment/2009/mar/24/biochar-earth-c02. (accessed 22 March 2013).

Luthy RG, Aiken GR, Brusseau ML, et al. (1997) Sequestration of hydrophobic organic contaminants by geosorbents. *Environmental Science & Technology*, **31**, 3341–3347.

Mahinpey N, Murugan P, Mani T, Raina R (2009) Analysis of bio-oil, biogas, and biochar from pressurized pyrolysis of wheat straw using a tubular reactor. *Energy Fuels*, **23**, 2736–2742.

Major J, Rondon M, Molina D, Riha SJ, Lehmann J (2010) Maize yield and nutrition during 4 years after biochar application to a Colombian savanna oxisol. *Plant and Soil*, **333**, 117–128.

Mao JD, Johnson RL, Lehmann J, Olk DC, Neves EG, Thompson ML, Schmidt-Rohr K (2012) Abundant and stable char residues in soils: implications for soil fertility and carbon sequestration. *Environmental Science & Technology*, **46**, 9571–9576.

Meller Harel Y, Elad Y, Rav-David D, Borenstein M, Shulchani R, Lew B, Graber ER (2012) Biochar-induced systemic response of strawberry to foliar fungal pathogens. *Plant and Soil*, **357**, 245–257.

Mukherjee A, Zimmerman AR, Harris W (2011) Surface chemistry variations among a series of laboratory-produced biochars. *Geoderma*, **163**, 247–255.

Muralidhara HS (1982) Conversion of tannery waste to useful products. *Resources and Conservation*, **8**, 43–59.

Nelson NO, Agudelo SC, Yuan W, Gan J (2011) Nitrogen and phosphorus availability in biochar-amended soils. *Soil Science*, **176**, 218–226.

Nilesen MN, Winding A (2002) *Microorganisms as indicators of soil health*. National Environmental Research Institute, Roskilde, Denmark.

Nguyen B, Lehmann J, Hockaday WC, Joseph S, Masiello CA (2010) Temperature sensitivity of black carbon decomposition and oxidation. *Environmental Science and Technology*, **44**, 3324–3331.

O'Neill B, Grossman J, Tsai MT, et al. (2009) Bacterial community composition in Brazilian Anthrosols and adjacent soils characterized using culturing and molecular identification. *Microbial Ecology*, **58**, 23–35.

Park JH, Choppala GK, Bolan NS, Chung JW, Chuasavathi T (2011) Biochar reduces the bioavailability and phytotoxicity of heavy metals. *Plant and Soil*, **348**, 439–451.

Quilliam RS, Marsden KA, Gertler C, Rousk J, DeLuca TH, Jones DL (2012) Nutrient dynamics, microbial growth and weed emergence in biochar amended soil are influenced by time since application and reapplication rate. *Agriculture Ecosystems and Environment*, **158**, 192–199.

Rajkovich S, Enders A, Hanley K, Hyland C, Zimmerman AR, Lehmann J (2011) Corn growth and nitrogen nutrition after additions of biochars with varying properties to a temperate soil. *Biology and Fertility of Soils*, **48**, 271–284.

Roberts KG, Gloy BA, Joseph S, Scott NR, Lehmann J (2010) Life cycle assessment of biochar systems: estimating the energetic, economic, and climate change potential. *Environmental Science & Technology*, **44**, 827–833.

Rondon MA, Ramirez JA, Lehmann J (2005) Greenhouse gas emissions decrease with charcoal additions to tropical soils. In: *3rd USDA Symposium on Greenhouse Gases and Carbon Sequestration in Agriculture and Forestry* (March 21–24 2005). Baltimore, MD, USA.

Rumpel C, Chaplot V, Planchon O, Bernadou J, Valentin C, Mariotti A (2006) Preferential erosion of black carbon on steep slopes with slash and burn agriculture. *Catena*, **65**, 30–40.

Schimmelpfennig S, Glaser B (2012) One step forward toward characterization: some important material properties to distinguish biochars. *Journal of Environmental Quality*, **41**, 1001–1013.

Schmidt MWI, Torn MS, Abiven S, et al. (2011) Persistence of soil organic matter as an ecosystem property. *Nature*, **478**, 49–56.

Schouten S, Van Groenigen JW, Oenema O, Cayuela ML (2012) Bioenergy from cattle manure? Implications of anaerobic digestion and subsequent pyrolysis for carbon and nitrogen dynamics in soil. *Global Change Biology and Bioenergy*, **4**, 751–760.

Singh BP, Hatton BJ, Balwant S, Cowie AL, Kathuria A (2010) Influence of biochars on nitrous oxide emission and nitrogen leaching from two contrasting soils. *Journal of Environmental Quality*, **39**, 1224–1235.

Singh BP, Cowie AL, Smernik RJ (2012) Biochar carbon stability in a clayey soil as a function of feedstock and pyrolysis temperature. *Environmental Science and Technology*, **46**, 11770–11778.

Sohi SP, Loez-Capel E, Krull E, Bol R (2009) *Biochar's Roles in Soil and Climate Change: A Review of Research Needs*. CSIRO, Highett, Victoria, Australia.

Sohi SP, Krull E, Lopez-Capel E, Bol R (2010) A review of biochar and its use and function in soil. *Advances in Agronomy*, **105**, 47–82.

Sohi SP (2012) Carbon storage with benefits. *Science*, **338**, 1034–1035.

Solaiman ZM, Blackwell P, Abbott LK, Storer P (2010) Direct and residual effect of biochar application on mycorrhizal root colonisation, growth and nutrition of wheat. *Australian Journal of Soil Research*, **48**, 546–554.

Sombroek W, Nachtergaele FO, Hebel A (1993) Amounts, dynamics and sequestering of carbon in tropical and subtropical soils. *Ambio*, **22**, 417–426.

Sombroek WG (1966) *Amazon soils. A reconnaissance of the soils of the Brazilian Amazon region*. Verslag landbouwkundig onderzoek, Wageningen, the Netherlands.

Sparrevik M, Field JL, Martinsen V, Breedveld GD, Cornelissen G (2013) Life cycle assessment to evaluate the environmental impact of biochar implementation in conservation agriculture in Zambia. *Environmental Science & Technology*, **47**, 1206–1215.

Sparrevik M, Saloranta T, Cornelissen G, Eek E, Fet AM, Breedveld GD, Linkov I (2011) Use of life cycle assessments to evaluate the environmental footprint of contaminated sediment remediation. *Environmental Science & Technology*, **45**, 4235–4241.

Spokas KA (2010) Review of the stability of biochar in soils: predictability of O:C molar ratios. *Carbon Management*, **1**, 289–303.

Steinbeiss S, Gleixner G, Antonietti M (2009) Effect of biochar amendment on soil carbon balance and soil microbial activity. *Soil Biology and Biochemistry*, **41**, 1301–1310.

Tilman D, Socolow R, Foley JA, et al. (2009) Beneficial biofuels—The food, energy, and environment trilemma. *Science*, **325**, 270–271.

Torres-Rojas D, Lehmann J, Hobbs P, Joseph S, Neufeldt H (2011) Biomass availability, energy consumption and biochar production in rural households of Western Kenya. *Biomass and Bioenergy*, **35**, 3537–3546.

Van Den Bergh C (2009) Biochar and waste law: a comparative analysis. *European Energy and Environmental Law Review*, **18**, 243–253.

Van Zwieten L, Singh BP, Joseph S, Kimber S, Cowie A, Chan Y (2009) Biochar and emissions of non-CO_2 greenhouse gases from soil. In: *Biochar for Environmental Management: Science and Technology*, (eds Lehman J and Joseph S). Earthscan, Oxford.

Verheijen FGA, Jeffery S, Bastos AC, Van Der Velde M, Diafas I (2010) *Biochar Application to Soils - A Critical Scientific Review of Effects on Soil Properties. Processes and Functions*. European Commission, Luxembourg.

Wang T, Camps Arbestain M, Hedley M, Bishop P (2012) Chemical and bioassay characterisation of nitrogen availability in biochar produced from dairy manure and biosolids. *Organic Geochemistry*, **51**, 45–54.

Wardle DA, Zackrisson O, Nilsson MC (1998) The charcoal effect in boreal forests: mechanisms and ecological consequences. *Oecologia*, **115**, 419–426.

Warnock DD, Lehmann J, Kuyper TW, Rillig MC (2007) Mycorrhizal responses to biochar in soil - concepts and mechanisms. *Plant and Soil*, **300**, 9–20.

Whitman T, Scholz S, Lehmann J (2010) Biochar projects for mitigating climate change: an investigation of critical methodology issues for carbon accounting. *Carbon Management*, **1**, 89–107.

Whitman T, Nicholson CF, Torres D, Lehmann L (2011) Climate change impact of biochar cook stoves in Western Kenyan farm households: system dynamics model analysis. *Environmental Science & Technology*, **45**, 3687–3694.

Woolf D, Amonette JE, Alayne Street-Perrott F, Lehmann J, Joseph S (2010) Sustainable biochar to mitigate global climate change. *Nature Communications*, **1**, 1–9.

Woolf D, Lehmann J (2012) Modelling the long-term response to positive and negative priming of soil organic carbon by black carbon. *Biogeochemistry*, **111**, 83–95.

Yamato M, Okimori Y, Wibowo IF, Anshori S, Ogawa M (2006) Effects of the application of charred bark of *Acacia mangium* on the yield of maize, cowpea and peanut, and soil chemical properties in South Sumatra, Indonesia. *Soil Science & Plant Nutrition*, **52**, 489–495.

Yang YN, Sheng GY (2003) Enhanced pesticide sorption by soils containing particulate matter from crop residue burns. *Environmental Science & Technology*, **37**, 3635–3639.

Yu XY, Ying GG, Kookana RS (2009) Reduced plant uptake of pesticides with biochar additions to soil. *Chemosphere*, **76**, 665–671.

Yuan JH, Xu RK, Zhang H (2011) The forms of alkalis in the biochar produced from crop residues at different temperatures. *Bioresource Technology*, **102**, 3488–3497.

Zackrisson O, Nilsson MC, Wardle DA (1996) Key ecological function of charcoal from wildfire in the boreal forest. *Oikos*, **77**, 10–19.

Zimmerman AR (2010) Abiotic and microbial oxidation of laboratory-produced black carbon. *Environmental Science & Technology*, **44**, 1295–1301.

Zimmerman AR, Gao B, Ahn MY (2011) Positive and negative carbon mineralization priming effects among a variety of biochar-amended soils. *Soil Biology and Biochemistry*, **43**, 1169–1179.