8 Biochar for Climate Change Mitigation

Navigating from Science to Evidence-Based Policy

Dominic Woolf, Johannes Lehmann, Annette Cowie,
Maria Luz Cayuela, Thea Whitman, and Saran Sohi

CONTENTS

8.1 INTRODUCTION

8.1.1 Overview of Biochar Systems as a Climate Mitigation Strategy

Biochar is the carbon-rich solid formed by heating biomass in an anaerobic environment (a process called pyrolysis). This pyrogenic carbonized material is typically known as biochar when it is intended as a soil amendment or to provide related environmental benefits. The concept of using carbonized biomass on a large scale as a climate-change mitigation approach can be traced back to two independent fields of study: (1) Seifritz (1993) discussed the climate-mitigation potential of industrial production and the burial of charcoal in landfills but did not consider its use as a soil amendment. (2) Charcoal was found to contribute a major component of the stable carbon in Amazonian dark earth soils known also as Terra Preta de Indio (Glaser et al. 2001, 2004). Sombroek et al. (1993) described the historical use of organic matter additions by indigenous peoples in the Amazon region to improve soil fertility, the resulting large accumulation of carbon (C), and its consequent potential as a climate-change mitigation strategy. The conjunction of these strands—that large scale charcoal production can be used to sequester C, and that charcoal can be used as a soil fertility amendment—has driven a rapid expansion and interest in biochar research over the past two decades, with only one journal article in the period 1990–1999 using the term "biochar," rising to 78 in 2000–2009 and soaring to over 4000 since 2010 (Web of Science citations). It has been estimated that biochar systems can mitigate up to 1.8 Pg CO_2C equivalent yr^{-1} (12% of anthropogenic CO_2, CH_4, and N_2O emissions), without endangering food security, habitat or soil conservation—a larger climate-change mitigation potential than using the same biomass for bioenergy (Woolf et al. 2010).

8.1.2 Main Impacts of Biochar Systems on Climate Change

Biochar's climate-change mitigation potential stems primarily from its slower decomposition than the raw biomass from which it is generated, thus lowering the rate at which photosynthetically-fixed C is returned to the atmosphere (Wang et al. 2016) (Figure 8.1). It is this *difference* in decomposition rates that is critical in determining how net carbon stocks evolve over time (Figure 8.2 and Whitman et al. 2010). Although approximately half of the carbon in a biomass feedstock is emitted as CO_2 during biochar production; by comparison, more readily-decomposed un-pyrolysed biomass will rapidly return most of its carbon to the atmosphere if allowed to decompose. Therefore, the carbon stocks remaining over time are larger for biochar than for raw biomass, leading to a net increase in soil carbon stocks. Thus, although embedding carbon in biochar is, in one sense, a redistribution of biomass carbon rather than newly fixed carbon, nonetheless the greater persistence of the biochar drives a net sequestration of carbon. Most studies have concluded that this persistence-derived carbon sequestration is the largest influence of biochar on net greenhouse gas balances, while other mechanisms serve to mediate this primary influence (Fowles 2007; Gaunt & Lehmann 2008; Roberts et al. 2010; Whitman et al. 2010; Woolf et al. 2010; Cowie et al. 2015).

In addition, biochar's overall impact on climate change mitigation depends also on a range of other secondary mechanisms. It can reduce nitrous oxide (N_2O) emissions from soil (Cayuela et al. 2013). It can alter methane emission or oxidation rates in soil (Jeffery et al. 2016). Conversion of biomass to biochar can avoid emissions of N_2O and/or methane (CH_4) that would have arisen from the decomposition or combustion of that biomass (Woolf et al. 2010). Biochar can enhance plant growth (Crane-Droesch et al. 2013; Jeffery et al. 2016), this enhanced productivity providing a positive feedback that further enhances the amount of CO_2 removed from the atmosphere, particularly if that increased biomass increases the feedstock available for further biochar production (Woolf et al. 2010). Biochar can alter the turnover rate of native soil organic matter (SOM), thus potentially either increasing or decreasing stocks of non-pyrogenic soil C (Zimmerman et al. 2011).

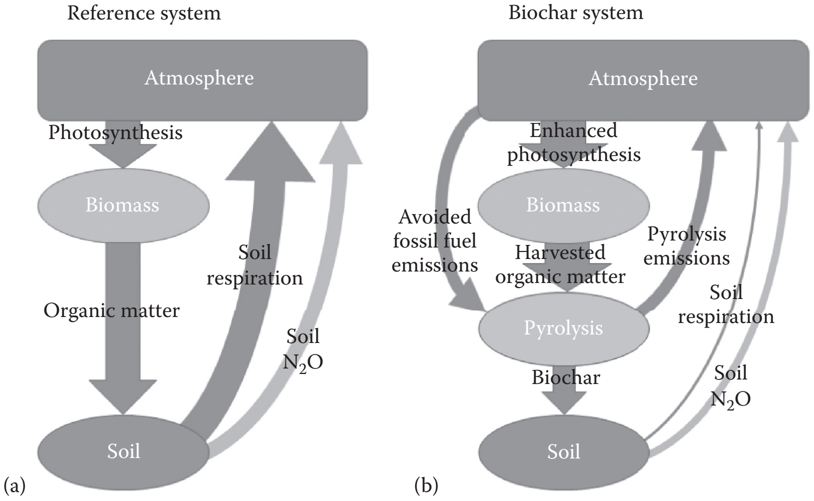

FIGURE 8.1 Main impacts of biochar on greenhouse gas (GHG) fluxes. In the reference system without biochar (a), plants remove CO_2 from the atmosphere by photosynthesis with fixed carbon being returned to the atmosphere as plants decompose in the soil. A fraction of the reactive nitrogen in the soil is also released to the atmosphere in the form of nitrous oxide, a potent GHG. In the biochar system (b), approximately half of the biomass carbon is returned to the atmosphere during pyrolysis, but the remaining carbon in the biochar decomposes more slowly than raw biomass, leading to an overall reduction in the rate of CO_2 emission to the atmosphere. Simultaneously, nitrous oxide emissions from soil can be reduced by up to 80%. Draw down of CO_2 by photosynthesis may be increased by improved soil fertility leading to an increase in net primary production. Finally, co-production of bioenergy in the pyrolysis process can displace fossil fuels that would otherwise have provided that energy (indicated as a net reduction in atmospheric CO_2 on the diagram).

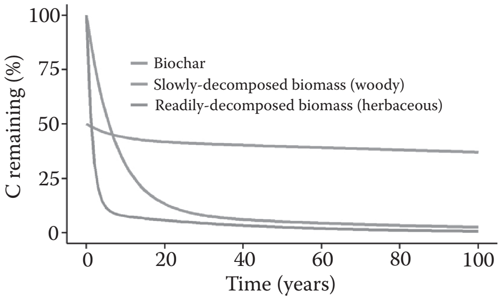

FIGURE 8.2 **(See color insert.)** Conceptual comparison of un-mineralized biomass carbon (C) remaining from different grades of organic matter, as a function of time. The lines are modeled using a two pool exponential decay model, comparing slow-turnover woody biomass (green line), fast-turnover herbaceous biomass (blue line), and biochar (red line). Assumed representative fast and slow fraction half lives, respectively, were 4 and 25 years for woody biomass, 1 and 25 years for herbaceous biomass, and 5 and 500 years for biochar. It was also assumed that half of the initial biomass carbon is lost during biochar production, hence the carbon remaining in biochar starts at 50% at time equal to zero.

As with *any* mechanism that increases the soil's organic C (SOC) stocks, biochar reduces soil albedo, thus increasing radiative forcing on bare soils (Verheijen et al. 2013). Slow decay of biochar in soils, together with tillage and transport activities, return a small amount of CO_2 to the atmosphere (Roberts et al. 2010). Reduced fertilizer requirements from improved nutrient-use efficiency (NUE) can reduce associated greenhouse gases (GHGs) from fertilizer production, transport and

N_2O emissions. Coproduction of bioenergy can offset fossil-fuel emissions, while returning about half of the C fixed by photosynthesis to the atmosphere (Woolf et al. 2014).

Although these secondary mechanisms are typically smaller than persistence-derived carbon sequestration, they are, nonetheless, pivotal in determining whether the climate-change abatement from biochar is greater or smaller than can be achieved by other ways (such as bioenergy) the same biomass could be used to offset or displace GHG emissions. Depending on the carbon intensity of the energy supply that is displaced, the reduction in fossil fuel emissions from bioenergy may be comparable to the carbon sequestration that biochar provides (Fowles 2007; Gaunt & Lehmann 2008; Roberts et al. 2010; Woolf et al. 2010; Cowie et al. 2015; Woolf, Lehmann & Lee 2016). Thus, the question of whether biochar provides greater or lesser abatement than bioenergy hinges largely on the size of these secondary climate-change mitigation impacts and the carbon intensity of any displaced fossil fuels.

The net impact of all these mechanisms is highly variable, depending on the characteristics of the entire biochar production system (from feedstock provision, through pyrolysis conditions, to soil biogeochemistry) and also on the characteristics of the assumed reference system to which it is compared (i.e., how the land, biomass and energy systems would be managed in the absence of biochar production). Life cycle assessments (LCAs) have indicated that the net mitigation impact of biochar systems commonly range from –0.6 to +1.75 Mg CO_2e Mg^{-1} feedstock (–0.3 to +1.3 Mg C Mg^{-1} feedstock-C) (Cowie et al. 2015). This wide variability is only partly attributable to uncertainties that remain in the size and longevity of some effects (especially, on plant growth, N_2O emissions from soil, and stabilization of non-pyrogenic SOC). The variability in impacts is mainly dependent on differences in the scenarios that are modeled (type of feedstock, pyrolysis conditions, and the reference energy system). Although some options are estimated to provide minimal or even negative climate-change abatement, one thing emerges clearly from these LCA studies: provided care is taken to avoid controllable detrimental impacts (such as indirect land use change), then the net GHG abatement is typically greater than the C sequestration alone. That is, the positive feedbacks on GHG abatement from factors such as reduced CH_4 and N_2O emissions, increased net primary production (NPP), reduced fossil fuel consumption, and increased soil C stabilization tend to outweigh the negative feedbacks from factors such as energy required for transporting biochar and incorporating it into the soil.

Thus, the overall picture that emerges is nuanced with both knowns and unknowns. While much remains to be learned about how long different types of biochar persist in different environments, nonetheless enough is known to be confident that woody biochars produced at high temperatures are sufficiently persistent to justify their use in managing the C cycle over the centennial timescales that are most relevant to navigating humanity through the challenges of the coming century. While much uncertainty remains in the impact of biochar on yield of different crops in different soils and agroecological zones, and it is expected to be as predictably variable as any other soil management such as fertilizer or lime applications, nonetheless, it is known that sandy, low cation exchange capacity (CEC) and highly acidic soils typically benefit the most from biochar applications; whereas, already-fertile soils will see little or no improvement, and adding alkaline biochar to alkaline soils can depress yields.

In the remainder of this chapter we investigate what is currently known and what remains to be researched about the major impacts of biochar on climate change. Finally, the policy implications of the current state of knowledge about biochar and climate are also discussed to draw recommendations about what types of biochar systems can already be confidently deployed, under what conditions biochar systems may be suboptimal or other forms of organic matter (OM) management would be preferred, and what are the most pressing research directions that need to be addressed to improve the decision making tools for biochar systems.

8.2 PERSISTENCE

A greater persistence of charred than uncharred OM is the foundation of biochar systems' capacity to reduce GHG emissions, and thereby mitigate climate change (Lehmann et al. 2006;

Lehmann et al. 2015). Such greater persistence means that the C in the charred biomass is returning to the atmosphere much slower than that in charred biomass, leading to more C in soil (or wherever the biochar is applied to) than in the atmosphere (Lehmann 2007). In addition to this most obvious impact, increased persistence has three other main effects: (1) the C in biochar does not generate CH_4, a much more potent GHGs than CO_2, if landfilled or added to soil (Houghton et al. 1997); (2) biochar added to soil (or compost) may reduce CH_4, N_2O, or CO_2 production from C or nitrogen (N) otherwise originating from soil (Van Zwieten et al. 2015; Whitman et al. 2015a,b)—the longevity of these impacts being contingent on the persistence of the charred material; and (3) biochar added to soil may increase plant growth through a variety of mechanisms (Jeffery et al. 2014), whose longevity also depends on biochar persistence. Except for the remediation of CH_4 emission from biomass (point 1 above) which is instantaneous and quasi permanent, greater persistence is the basis for greater emission reductions and therefore the basis for the role of biochar systems in climate change mitigation. However, it is important to recognize that, while greater persistence inferred by charring is a necessary precondition for climate change mitigation by biochar systems, it is not sufficient, because life-cycle emission reductions will hinge on additional factors such as the fate of biomass under business as usual, transportation, and energy use among other factors discussed in the sections below.

The importance of persistence with respect to C trading and dangerous climate change over the coming decades and century hinges not so much on the absolute persistence than on the persistence relative to the scenario were the biomass not charred. When considering centennial assessment horizons, the extent of emission reductions is remarkably insensitive to the extent of biochar persistence, both for emissions of N_2O (Gaunt and Lehmann 2008), as well as the C in the biochar (Roberts et al. 2010), provided that the mean residence time (MRT) remains above 100 years. This importance over the coming century has to be distinguished from the millennial C balance (Lehmann et al. 2010), which requires MRT in the range of millennia for biochar systems to safeguard against return of C to the atmosphere on the long term. Since biochar may have a role to play in an energy and C management transition (Woolf, Lehmann, and Lee 2016), MRT of above 100 years appear to be sufficient in a first approximation for contribution to climate stabilization over the coming century.

Evidence of biochar persistence has been forthcoming from naturally occurring pyrogenic organic C generated by vegetation fires, from anthropogenic forest clearing and by-product management, from field trials with deliberate additions of biochar, and from incubation studies of biochars. Each of these approaches to the quantification of its persistence has advantages and disadvantages (Lehmann et al. 2015). As with uncharred OM, both material properties as well as environmental factors play a role in the decomposition rate of biochar. Material properties play a greater role in the short term of months and decades, while interaction with the soil matrix emerges as an important control over the long term, with factors such as temperature and moisture being the most important control during any period. Over the period of decades and a century that is most relevant to the current discussion and need for climate change mitigation, the most important determinants of biochar persistence are material properties—specifically the extent of thermal transformation to a higher degree of what is called aromatic condensation, i.e., to what extent and in which configuration C atoms in the biochar are fused together during charring (Lehmann et al. 2015). Different biochars therefore have vastly different mineralization rates, with a MRT of about 556 years of 97% of its mass (Wang et al. 2016) for production conditions that typically generate biochars with greater aromatic condensation. The degree of aromatic condensation can be studied using spectroscopic techniques (Wiedemeier et al. 2015) but is often estimated simply by their elemental oxygen-to-organic carbon (Spokas 2010) or hydrogen-to-organic carbon ratios (Lehmann et al. 2015) that are to a certain degree interchangeable (Enders et al. 2012). An important nuance to recognizing and analytically establishing the material properties is that biochars (as any plant residue) are not homogeneous but typically contain some portion that is extremely easily metabolized to CO_2, while others require enzymatic expenditures by microorganisms that are energetically costly and are therefore typically mineralized only very slowly.

Two points deserve special attention: first, no matter what MRT is measured for a certain biochar, more important than the absolute mineralization is the difference in mineralization between the uncharred OM and the charred biochar. While biochar mineralization varies depending on all the factors mentioned above including environmental factors, under identical soil and climatic conditions charring has always been shown to confer greater persistence to organic residues, typically by more than one order of magnitude. Second, the charring process itself releases CO_2 (or gases and particulates with even greater climate forcing than CO_2) to the atmosphere (Whitman et al. 2013) that has to be compensated by slower mineralization of the biochar compared to the uncharred feedstock. In addition, the released gases and volatiles can be utilized as an energy or bio-product and offset other bioenergy or even fossil energy use, which would create a very favorable emission balance depending on the fuel that is offset (Woolf et al. 2010). Third, both biochar persistence in absolute terms (such as through calculation of its MRT) as well as the effect that charring has on organic residues in relative terms will always be highly variable in time and space, simply because of the multitude of controls that are at play (as discussed above). This variability must not be confused with uncertainty about biochar persistence (Lehmann and Rillig 2014). While the variability in mineralization rates between different biochars or between sites can be very large, such variability is predictable given the knowledge of mineralization processes.

Therefore, it is evident that knowing the production conditions (mainly pyrolysis temperature and duration) and the feedstock type will be sufficient to be certain of the degree of aromatic condensation that will reduce mineralization of a given plant residue under otherwise identical soil and climatic conditions. This aromatic condensation can be verified using said elemental ratios (oxygen-to-organic C or hydrogen-to-organic C). For example, woody biomass residues pyrolysed above 500°C for at least 10 min will have molar hydrogen-to-organic C ratios of below 0.4 (Enders et al. 2012) and therefore under climate and soil conditions where biochar is applied in an agricultural context show MRT of more than 1000 years (Lehmann et al. 2015). The myriad of possible different feedstocks, conversion technologies and conditions, soil properties and management, as well as climate and weather conditions and their combinations will generate a wide range of persistence values. Information about all possibilities will emerge as science progresses, but sufficient information is already available for certain combinations of factors that have been studied in greater detail and for which natural analogs exist (e.g., from vegetation fires or historic anthropogenic additions).

From a scientific perspective, even though it is a foundational and necessary property of biochar, persistence can be managed and verified, and will not constitute a constraint for designing biochar systems for climate change mitigation. From an industry, policy and public perspective, this is different. Trust in the extent of biochar persistence remains a constraint that will likely only be mitigated by improved communication and additional experimentation, with a focus on long-term field experiments.

The persistence of emission changes of other greenhouse gases from soils (CO_2, CH_4, N_2O) and growth responses by plants that could all increase or decrease the life-cycle emission balance, also benefit from further and long-term (decadal) field experimentation. However, sufficient information is available for many different combinations of factors that appropriate estimates can be made to sufficiently constrain projections over the coming century (Woolf et al. 2010). Therefore, it is appropriate that the first publically available C trading methodology has been established (Placer 2015). For a truly long-term C sequestration (aka millennial or semi-permanent from a geological perspective), more ambitious criteria to biochar persistence have to be applied than for arresting dangerous climate change over the coming century (Lehmann et al. 2010). The scientific desire and justification to establish a global network of biochar field experiments to generate long-term (more than 20 years) data on full-field emission balances should not be seen as an argument against implementing biochar systems for climate change mitigation now, in order to start gathering relevant data beyond but including biochar persistence at scale of implementation.

8.3 SOIL NITROUS OXIDE EMISSIONS

Nitrous oxide (N_2O) is now considered the third most important long-lived GHG after CO_2 and CH_4 (Davidson and Kanter, 2014). The emission projections considering business-as-usual scenarios estimate that anthropogenic N_2O emissions could double in 2050 (from 5.3 to 9.7 Tg N_2O-N y^{-1}). Importantly these emissions are mostly associated with food production, because agriculture currently accounts for 56%–81% of gross anthropogenic N_2O emissions (Davidson and Kanter, 2014). Since arable soils are by far the dominant N_2O source, reducing N_2O fluxes from fertilized soils would represent a substantial mitigation opportunity in the agricultural sector (Paustian et al., 2016).

N_2O is produced in soils by microbiological processes and it is generally believed that the principal factor responsible for agricultural N_2O emissions is a lack of synchronization between crop N demand and soil N supply (Venterea et al. 2012). Microorganisms utilize N compounds, not only to incorporate N into their cell structures, but also in many catabolic reactions that imply N redox transformations. It is within these transformations where several trace N gases (including N_2O and NO) are formed and emitted to the atmosphere (Conrad 1996). Thus, whenever N enters the soil matrix a competition starts between plant roots and soil microorganisms to use that N (Hodge et al. 2000; Kuzyakov & Xu 2013). This implies that some N_2O emissions associated with fertilized croplands is inevitable, and that even when engineered high N-use-efficiency cultivars are used and N is applied in synchronization with plant N demand, there will always be a vigorous competition for N in the rhizosphere. Depending on the circumstances, this can lead to substantial N_2O emissions. Ultimately, mitigating N_2O emissions without limiting N supplied to plants can be achieved only by methods that either (1) increase the proportion of added N that is assimilated by crops, so that less N-fertilizer inputs are needed (Venterea et al. 2012), or (2) promote the consumption of N_2O in soil, i.e., its reduction to N_2 (Richardson et al. 2009)—both routes where biochar has been found to have a substantial impact (Cayuela et al. 2013; Zheng et al. 2013; Quin et al. 2015).

Soils emit more N_2O to the atmosphere than any other source (Paustian et al. 2016), which is largely the result of the extensive use of N-based fertilizers in agricultural lands (Stein & Yung 2003; Smith et al. 2008; Hu et al. 2015; van Groenigen et al. 2015). In spite of the numerous research efforts over the past decades, N_2O mitigation remains a serious challenge, since few practices have been found to be both effective and consistent across agroecosystems. Recently, the use of biochar is being investigated as a means to mitigate N_2O emissions from arable soils (Spokas & Reikosky 2009; Van Zwieten et al. 2009), and there is strong evidence of the potential of biochar to decrease N_2O emissions from fertilized soils. A recent meta-analysis summarizing results from 56 studies (published between 2009 and 2014) found that biochar reduced soil N_2O emissions by an average of 49% (Cayuela et al. 2015). However, the N_2O mitigation was highly contextual, with some soils undergoing 90% reductions (Spokas & Reikosky 2009; Cayuela et al. 2013; Nelissen et al. 2014) and others none or even an increase in N_2O emission (Sánchez-García et al. 2014; Wells & Baggs, 2014). This fact is clearly linked to different N_2O formation pathways in different soils, since N_2O is both an intermediary and a by-product in several N chemical reactions. To date, the individual N_2O formation pathways on which biochar operates have not been studied to a sufficient extent. Aligned to a paucity of data on biochar mechanisms are gaps in the scientific understanding on the processes leading to N_2O production (and consumption) themselves irrespective of biochar additions, with new studies challenging previous assumptions being published every year (Sanford et al. 2012; Yang et al. 2012; Zhu et al. 2013; Phillips et al. 2016).

In spite of the complexity of the subject, there is some evidence about which types of biochar might work best for N_2O mitigation. So far, the highest N_2O reductions have been found with biochars made of lignocellulosic or woody feedstocks by slow pyrolysis and at relatively high temperatures (500–700°C). These biochars generally have low ash content (Cayuela et al. 2013), medium-to-high organic C to N ratios (C_{org}:N >30) (Cayuela et al. 2014), and low atomic H:C_{org} (<0.5) (Cayuela et al. 2015). The dose of application is also relevant and needs to be at least 1% in dry weight (equivalent to approximately 10 Mg ha^{-1}) in order to reach significant reductions (Cayuela et al. 2014), beyond which threshold N_2O response increases linearly with dose at 0.17% to 0.91% of initial emissions per Mg ha^{-1} biochar applied (Woolf et al. 2016; Cayuela et al. 2014). As regards environmental settings, the best results have been reported with conditions promoting denitrification, i.e., high soil NO_3^- concentration and moisture content (Nelissen et al. 2014; Thomazini et al. 2015). Interestingly a correlation has been found between N_2O total cumulative emissions and biochar mitigation: the higher the N_2O emissions, the higher the proportion of reductions with biochar (Cayuela et al. 2013; Thomazini et al. 2015).

Several hypotheses have been suggested about the potential mechanisms underpinning N_2O mitigation with biochar. However, the number of studies focusing on unveiling the involvement of a particular mechanism remains conspicuously low. A brief summary of the most studied mechanisms to date is given below. It is important to note that different mechanisms could be more or less relevant depending on the specific soil and prevailing environmental conditions.

8.3.1 Increasing Soil pH

Lower soil pH is generally associated to higher N_2O emissions from both nitrification and denitrification processes (Mørkved et al. 2007; Liu et al. 2010). The role of biochar alkalinity on N_2O mitigation was studied by Cayuela et al. (2013), who demonstrated that the alkalizing effect of a series of biochars could explain between 6-65% of the N_2O decrease in an organic acid soil. Subsequently, Obia et al. (2015) quantified NO, N_2O and N_2 production at high temporal resolution under full denitrification conditions and found a clear link between biochar alkalizing effect and the reduction in N_2O/N_2 ratio in acidic soils. However, they acknowledged that pH alone could not explain the observed reductions and that other factors may have contributed to the suppression of N_2O in addition to the pH effect. In a field experiment Hüppi et al. (2015) compared biochar with lime amendment (adjusting to the same pH) to a slightly acid soil (pH 6.3) and found 52% reductions with biochar whereas no reductions with lime. They concluded that there is no evidence that reduced N_2O emissions with biochar is merely caused by a higher soil pH.

8.3.2 Altering Microbial Composition and/or Functioning of Communities

Biochar changes microbial abundance and community composition, with known impacts on soil N transformations (Lehmann et al. 2011). It was initially speculated that biochar's toxicity (due to its polyaromatic hydrocarbon (PAH) or dioxin content) could decrease the total activity of denitrifiers in soil (Wang et al. 2013). Nonetheless, this hypothesis was rebutted by Alburquerque et al. (2015), who demonstrated that the presence of PAHs at typical biochar concentrations did stimulate, rather than inhibit, N_2O emissions and could not explain N_2O reductions with biochar. Harter et al. (2014) were the first to report an increase in the relative gene and transcript copy numbers of the *nosZ*-encoded bacterial N_2O reductase in a biochar amended soil, a fact that was accompanied by a decrease in the N_2O/N_2 ratio. In a more recent study, Harter et al. (2016) found that biochar not only altered the 16S rRNA gene-based community composition and structure, but it also led to the development of distinct functional traits capable of N_2O reduction containing typical and atypical *nosZ* genes.

8.3.3 Favoring Biotic/Abiotic Electron Transport

Several recent articles underline the importance of biochar electron transport properties, which might have a bigger impact on soil biogeochemical processes than previously thought (Chen et al. 2014; Kappler et al. 2014; Prévoteau et al. 2016; Saquing et al. 2016; Sun et al. 2017). Abiotic interactions would imply a purely chemical redox interaction of biochar with N compounds. For instance, Quin et al. (2015) measured N2O reduction by injecting 15N-N_2O in sterilized soil columns and demonstrated that biochar took part in abiotic redox reactions reducing N_2O to dinitrogen

(N_2), in addition to adsorption of N_2O. Biotic interactions would imply the ability of biochar to directly accept or donate electrons from/to soil microorganisms. This ability has been demonstrated with microorganisms carrying out N redox transformations, like dissimilatory nitrate reduction to ammonium (Saquing et al. 2016), but to date they have not been explored in N_2O studies.

8.3.4 SORPTION OF C AND/OR N COMPOUNDS

Since denitrification requires organic C as an electron donor, any modification of the concentration of easily mineralizable C_{org} in soil will have an impact on denitrification rates. Biochar (especially when produced at high temperature) provides little easily mineralizable C_{org}. On the contrary, it is known to interact with native or added C_{org}, reducing its availability for soil microorganisms (Kasozi et al. 2010). Although this mechanism has been suggested (Borchard et al. 2014), its importance for N_2O emission reductions has not been systematically evaluated. On the other hand, the C_{org}:NO_3^- ratio is crucial in determining the stoichiometry of denitrification products, with higher ratios favoring the last step of denitrification and therefore decreasing N_2O emissions. The adsorption of NO_3^- by biochar could potentially increase this ratio and this mechanism has also been postulated to explain N_2O mitigation. This hypothesis is supported by Hagemann et al. (2017b) who reported that biochar is able to capture nitrate (NO_3^-), although no isotope labeling was employed in this study leaving the question of the source of sorbed N unresolved. Other studies have indicated that NO_3^- is typically not retained by biochar (Hollister et al. 2013; Wang et al. 2016).

Few studies have looked at the long-term N_2O mitigation capacity of biochar and the results to date are contradictory. The first study investigating how biochar aging might modify its ability to decrease N_2O emissions was carried out by Spokas (2013). This study concluded that 3 years of weathering negated the suppression of N_2O production that was originally observed from the fresh biochar. Subsequently Felber et al. (2014) found a decreasing capacity of biochar to reduce N_2O emissions after one year in grassland. Conversely, Hagemann et al. (2017a) reported that N_2O emissions were still effectively reduced by biochar in the third year after application in a field experiment with corn. In summary, the knowledge on long-term N_2O mitigation with biochar is clearly insufficient, as the number of long-term studies is still low.

8.4 PRIMING

In order to fully characterize the net climate impact of biochar in a given system, its effects on non-biochar soil organic carbon (SOC) stocks must also be accounted for. Although biochar itself usually decomposes very slowly in soils, it can affect existing SOC stocks, changing their rate of mineralization. These changes in mineralization rate are often referred to as “priming,” where “positive priming” indicates that SOC is mineralized faster with biochar additions than without, and “negative priming” indicates that SOC is mineralized more slowly than it would have been without biochar additions (Bingeman et al. 1953; Woolf & Lehmann 2012). This section provides an overview of observations of priming in biochar systems, possible mechanisms that may drive priming, interactions with other C sources in soils, including plant roots and added OM, and implications for the expected magnitude and duration of priming.

A number of papers have recently summarized biochar effects on SOC stocks (Wang et al. 2016; Whitman et al. 2015b; Maestrini et al. 2015; Sagrilo et al. 2015). A meta-analysis of 21 studies reported a mean decrease of 3.8% in SOC mineralization with biochar additions (Wang et al. 2016), although the 95% confidence interval included zero. This study builds on a meta-analysis conducted two years earlier, which included 16 studies, and found a mean increase of 15% in SOC mineralization 1 year after biochar additions to soil (Maestrini et al. 2015). The apparent contrast between these two studies reflects both the larger number of observations and the increase in the number of longer-term studies available to the more recent (Wang et al. 2016) meta-analysis. The inclusion of more long-term data is particularly relevant here, as both positive and negative priming effects can coexist, with initial positive priming often transitioning to later negative priming. (Zimmerman et al. 2011; Woolf & Lehmann 2012; Singh and Cowie 2014; Weng et al. 2015; Maestrini et al. 2015; Wang et al. 2016). These metastudies also highlight the wide range of potential SOC responses to biochar, and the challenge of succinctly summarizing effects from a relatively small number of studies, given the wide diversity of potential soil–biochar systems and timescales. Factors such as soil texture or mineralogy, study duration, and biochar feedstock and temperature, all influence the net effect of biochar on SOC mineralization (Maestrini et al. 2015; Wang et al. 2016; Whitman et al. 2015b). To thoroughly characterize and understand these effects, further studies will be necessary, covering a wider range of systems, including more field studies and long-term studies, and focusing on understanding the mechanisms behind these effects.

Part of the challenge of predicting the long-term effects of biochar on SOC stocks is that there are diverse mechanisms by which biochar may affect SOC mineralization. The brief summary given below is in the context of theories of SOC stabilization (Six & Paustian 2014; Cotrufo et al. 2013). Determinants of SOC persistence in soils include environmental controls (e.g., optimal moisture, temperature, oxygen), physicochemical stabilization (e.g., SOC stabilization on mineral surfaces and occlusion by soil aggregation), chemical recalcitrance of SOM (e.g., lignin vs. simple sugars), and biological factors (e.g., microbial community composition and functional potential). A brief discussion follows on how biochar could affect each of these factors. Any impact of biochar on soil moisture or temperature could be predicted to have an impact on soil microbial activity, and SOC mineralization rates. For example, if biochar increases water retention in soils (Abel et al. 2013) or shifts soil pH to be more favorable for microbial activity, priming could occur. Physicochemical stabilization of SOC could be affected in at least two ways by biochar additions. First, a commonly cited explanation for longer-term negative priming is that biochar surfaces sorb SOM, making it less easily available to microbes (Weng et al. 2015; Woolf & Lehmann 2012; Zimmerman et al. 2011; Kerré et al. 2017). Second, in theory, if biochar additions to soil affected aggregation dynamics (e.g., by promoting or inhibiting fungal growth or other aggregate-stabilizing factors), then SOC protection by aggregates could be affected. While biochar wouldn't likely directly affect the chemical composition of SOM, it could change the relative "appeal" of a given substrate (Whitman et al. 2014b). For example, Whitman et al. (2014a) observed very short-term (less than one week) negative priming, and suggested that substrate switching may have occurred, where the small but relatively easily-mineralizable fraction of biochar was preferentially used as a C source by soil microbes. Conversely, that same fraction of biochar has been postulated to be responsible for short-term positive priming, where the input of easily-mineralizable C temporarily increases total soil microbial activity, increasing SOC mineralization as well (Zimmerman et al. 2011; Whitman et al. 2014b). Finally, biochar additions could shift the microbial community composition (Whitman et al. 2016; Xu et al. 2014), which could have positive or negative implications for microbial activity.

Modeling can play a key role in helping us predict the long-term effects of biochar on SOC mineralization. While short-term experimental studies may give us a rate of change in SOC mineralization, predicting the long-term net effects of biochar additions to soil is not simply a matter of extrapolating from the short-term rate. Depending on the mechanism by which biochar affects SOC mineralization, it is possible to predict whether the effects of biochar on SOC will saturate or will persist over time. For example, if biochar itself is responsible for directly sorbing and stabilizing SOC, thereby decreasing its mineralization rate (negative priming), the net potential effect on SOC will depend on the number of potential sorption sites on the total added biochar but would be expected to persist while the biochar remains in the soil. In contrast, if biochar additions increase SOC mineralization (positive priming) due to the stimulation of the microbial community by the relatively small easily-mineralizable fraction of biochar, this effect might be expected to persist only as long as the easily-mineralizable fraction of biochar exists. Woolf & Lehmann (2012) modeled these assumptions, modifying the RothC model (Coleman & Jenkinson 2008) to represent positive priming by changing mineralization rates, and to represent negative priming by changing the fraction of C entering the stable carbon pool. The model predicted that negative priming would have a greater net impact on SOC stocks than positive priming, over 100 years. Archontoulis et al. (2015) used a similar modeling approach and identified the predicted duration of priming effects as a key knowledge gap for predicting net effects of biochar on SOC stocks.

In addition to direct biochar–SOC interactions, researchers have begun to investigate interactions between biochar, SOC, and other OM inputs, including litter and plant root C inputs. While there are still relatively few studies, this research is essential, as complex three- or more-way interactions likely occur in these more realistic systems. For example, Cui et al. (2017) found that biochar caused positive priming of SOM, but that priming was less than was caused by fresh OM additions, and when the two were added together, SOM decomposition rates were lower than with fresh OM alone. Ventura et al. (2015) found that the presence of roots increased biochar decomposition rates. Weng et al. (2015) found that biochar induced negative priming of SOC when plants were present but had no effect on SOC mineralization in soils without plants. Similarly, Keith et al. (2015) found that biochar reduced positive priming effects on SOC by roots. However, this effect was limited to only one of two soils studied. In contrast, Whitman and Lehmann (2015) found evidence that biochar may enhance positive priming of SOC by plants, although this effect was just observed at a single timepoint.

The challenge for researchers in the coming years will be to continue to disentangle the myriad potential mechanisms that could affect biochar–SOC interactions in real-world systems. Isolating and testing for specific mechanisms, controlling for or quantifying all potential drivers of priming in an experiment, and determining when each factor is most important will be key to developing a predictive understanding of biochar's effects on SOC stocks. Understanding the underlying mechanisms will improve the ability to predict long-term priming effects of biochar on SOC.

8.5 ALBEDO

The GHG emissions are not the only means by which human activity can influence climate change. Another forcing mechanism arises from anthropogenic alteration of the Earth's albedo (the fraction of incident solar radiation that is reflected). Soil darkening from increased soil C stocks can reduce the local surface albedo, particularly during bare fallows and while leaf area index is low. While this is true of any mechanism that increases soil carbon stocks (for example Meyer et al. 2012 found that compost reduced soil albedo by the same amount as biochar, per unit C), it is nonetheless worthy of discussion in this chapter due to the greater attention that albedo has received in the biochar literature than in relation to other methods of SOC sequestration.

The earliest published discussion of the impacts of biochar on albedo probably appears in Woolf (2008), who noted the potential for biochar amendments to lower local surface albedo, and also for airborne dust from biochar storage, transport, application, and wind erosion to contribute to tropospheric aerosols and black carbon (BC) deposition on snow and ice. Woolf (2008) also suggested that measures should be taken to reduce airborne dust (such as pelletization of biochar) and to mitigate potential impacts on surface albedo by limiting applications in regions with light soils and extended periods of bare soils.

Woolf (2011) provided a first order estimate of the impact of biochar application on albedo at the global scale, by assuming that the albedo of a soil-biochar mixture is approximately linearly interpolated between the albedos of biochar and soil, on a volumetric mixing ratio basis. In practice, the combined albedo of a soil–biochar mixture will also depend on their particle size distributions, physical interactions such as the coating of biochar particles with mineral deposits or the coating of soil particles with organic compounds from the biochar, biological effects such as the effect of biochar on soil microfauna and also on the extent of vertical transport of biochar within the soil column (Serbin et al. 2009). Nonetheless the linear mixing model provides at least an indication of the expected order of magnitude and also provides a useful baseline to compare measured albedo impacts to.

The global mean albedo of cropland soils is 0.144, derived by combining spatial datasets of soil albedo (Wilson & Henderson-Sellers 1985) and cropland density (Erb 2007). The albedo of charcoal is 0.04 (Serbin et al. 2009). Woolf (2011) thus estimated that a 5% v/v biochar-to-soil mixture (equivalent to 25 g kg^{-1}, or 50 Mg BC ha^{-1} to a depth of 0.15 m) would reduce the mean soil albedo by approximately 5%. The associated radiative forcing (*RF*) arising from this change in soil albedo can then be calculated according to (Lenton and Vaughan 2009):

$$RF = -0.579 S_0 f_e f_s \quad \alpha_s, \tag{8.1}$$

where S_0 is the annual mean flux of solar radiation at the top of the atmosphere (342 W m^{-2}); $\Delta\alpha_s$ is the change in soil albedo; f_e is the fraction of the Earth's surface over which the change in albedo occurs; and f_s is the fraction of surface albedo attributable to soil albedo (which depends on the vegetation canopy density), estimated to be 0.2 according to Serbin et al. (2009). A 5% reduction in mean soil albedo thus corresponds to a net radiative forcing of 6×10^{-3} W m^{-2}, if biochar were applied to all 1.5 Gha (giga hectare) of global cropland (Woolf 2011). This is equivalent to 1.1% of the −0.4 W m^{-2} radiative forcing from avoided CO_2 emissions attributable to the same biochar applications (Lenton & Vaughan 2009).

Relatively few studies have quantified the change in soil albedo from biochar additions experimentally, with somewhat variable results. Genesio et al. (2012) measured surface albedo in a winter durum wheat crop (*Triticum durum*) following application of biochar (0, 30 and 60 Mg ha^{-1}), and incorporation in the top 0.1 m with a rotary hoe. They found that, in the first year, soil albedo in both the 30 and 60 Mg ha^{-1} treatments was 0.062 ± 0.001 compared to the contrsol albedo of 0.208 ± 0.004. This marked reduction in albedo ($\Delta\alpha_s$) on the biochar plots is approximately one order of magnitude greater than would be predicted by the linear mixing model and corresponds to a soil albedo that is almost as low as that of pure charcoal (0.04; Serbin et al. 2009). However, Genesio et al. (2012) also found that $\Delta\alpha_s$ declined to 0.05 after 18 months, and completely disappeared following a second tillage operation in the second year. The observation that soil albedo was almost as low as pure charcoal in the first year but became indistinguishable from soil albedo after tillage in the second year suggests that the profound reduction in albedo in the first year may have been caused by incomplete mixing, with a higher concentration of biochar remaining at the soil surface. This hypothesis is compatible with the use of a rotary hoe for biochar incorporation, which is designed for weeding rather than pedoturbation.

Other authors have reported significantly lower values of $\Delta\alpha_s$ than observed in the first year by Genesio et al. (2012). Oguntunde et al. (2008) found that surface deposits of charcoal remaining at charcoal production sites 2 to 14 months after cessation of operation produced a $\Delta\alpha_s$ of 0.03 – 0.06. Meyer et al. (2012) measured $\Delta\alpha_s$ to be 0.01 from 31.5 Mg ha^{-1} biochar tilled to 0.1 m depth in a wheat field trial in Donndorf, Germany. Notwithstanding the much lower value observed for $\Delta\alpha_s$ in their field trial, Meyer et al. (2012) ("conservatively") estimated the long term radiative forcing of albedo by assuming that $\Delta\alpha_s$ in the first year was 65% of the value measured by Genesio et al. (2012) in the first year, declining to 22% by year 3 and thereafter declining only slowly with a half-life of 500 years. Based on this assumption, Meyer et al. (2012) estimated that the overall climate change mitigation of biochar could be reduced by 13%–22% by the albedo impact.

Verheijen et al. (2013) found a 0.2% reduction in soil albedo per Mg ha^{-1} biochar application mixed to 0.15 m depth in wet soils, and a 0.35% albedo reduction per Mg ha^{-1} biochar in dry soils. Based on these measurements, they estimated a 5%–11% reduction in the mitigation potential of biochar due to the albedo impact. Verheijen et al. (2013) also found that surface application of biochar without incorporation caused a much more profound in decrease in soil albedo that reduced its mitigation impact by 11%–23%. Similar to Verheijen et al. (2013), Bozzi et al. (2015) measured a 0.25% reduction in soil albedo per Mg ha^{-1} biochar application. Combining this with time-averaged satellite measurements of surface albedo over agriculture areas with different soil albedos, they estimated the biochar application rates that would result in the same soil albedo difference as that present between the different soil types. Based on these inferred equivalent biochar application rates Bozzi et al. (2015)

calculated that the measured time-averaged radiative forcing for dark soils relative to bright soils of 0.1–2.0 W m^{-2} would imply a 1.1%–29.7% reduction in the mitigation potential of biochar.

Zhang et al. (2015) found that 45 Mg ha^{-1} biochar reduced surface albedo by 23% at seeding of maize (0.5% per Mg ha^{-1}), declining to 20% (0.4% per Mg ha^{-1}) in the jointing stage, and no significant difference from the heading stage to mature stage. Zhang et al. (2013) observed that 4.5 Mg ha^{-1} biochar caused a 2%–7% *increase* in soil reflectance in the 350–495 nm (ultraviolet to blue) range and a 1%–6% decrease in the 496–2474 nm range.

Usowicz et al. (2016) found that surface application (without incorporation into the soil profile) of 30 Mg ha^{-1} biochar on grassland reduced surface albedo by 30%, from 0.17 to 0.10 (1% per Mg ha^{-1} biochar) when the grass was short (0.06 m). When the grass had grown to 0.10–0.15 m height there was no significant difference in albedo between biochar and control plots, and when the grass reached 0.15–0.50 m, the effect was reversed with surface albedo increased 15% from 0.20 to 0.23 (0.5% per Mg ha^{-1} biochar). On bare fallow plots, in which 30 Mg ha^{-1} biochar was incorporated to 0.115 m by rotary tillage, Usowicz et al. (2016) observed a reduction in soil albedo of 0.02–0.03 (0.5% per Mg ha^{-1} biochar).

The large variability in these results indicates that the effects on albedo are mediated by a number of both controllable and uncontrollable factors. These include:

- Initial soil albedo, with the albedo of lighter colored soils being reduced more than those of darker ones.
- Hydrology, with dry soil albedo being reduced more than wet soils.
- Vegetation, with increasing leaf area reducing the impact of soil albedo on surface albedo. The interaction between vegetation cover, time, and atmospheric diffusivity is also important, with bare soil being less important to dark sky albedo at lower solar declination.
- Time, with some evidence that initial reductions in albedo may become negligible after as little as one year. This may be caused either by vertical transport of biochar into the soil column away from the surface, and/or coating of biochar surfaces with minerals or OM.
- Depth of incorporation, with mixing of biochar into deeper soil reducing its albedo impact. As noted by Verheijen et al. (2013), below surface application of biochar would prevent any albedo reduction but is more expensive and future mechanical soil operations could yet expose biochar to the surface.

Given that under unfavorable conditions the albedo impact might be a significant part (possibly up to 30%) of the mitigation potential of biochar, whereas under favorable conditions the albedo impact is negligible (Figure 8.3), it would be prudent while more long-term data are acquired to take

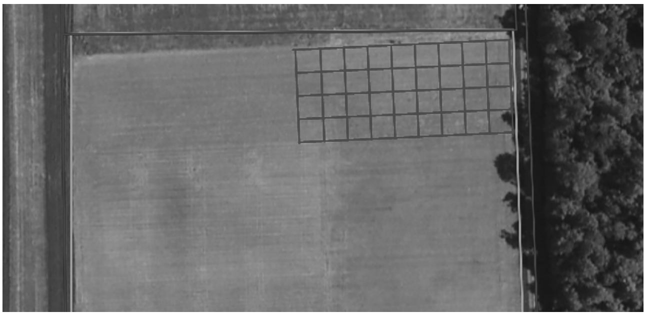

FIGURE 8.3 Satellite image (April 14, 2017) of bare soil on a biochar field trial in a continuous maize cropping system in New York State (42.7305 N, 76.655 W). Biochar was applied in 2006 at rates of 0 to 50 Mg ha^{-1} in a randomized block design in the blocks outlined in purple. The field was plowed annually in spring. No treatment differences in albedo are discernable by the naked eye 10 years after application.

active measures to limit its importance when planning and implementing biochar systems. Steps that will mitigate albedo impacts include avoiding surface application without incorporation (particularly where soil is likely to be bare for an extended period), ensuring thorough mixing of biochar into the soil, and mixing biochar deeper into the soil. These steps are consistent with optimal use of biochar to improve soil fertility. Although albedo impacts can also be mitigated by preferentially using biochar in soils with already lower albedo and/or high vegetation cover, this must be balanced against the fact that the more degraded soils that can most benefit from biochar typically have lower OM content that can give them a lighter color. In addition to the potential impacts of biochar on surface albedo, Genesio et al. (2016) remind us that airborne BC aerosols from wind erosion of fine particulate BC may not be entirely negligible and note that quantification of this effect has not received sufficient attention. Pelletization of biochar prior to field application (Woolf et al. 2010) or charring of pelleted feedstock has been suggested as one way to limit the production of airborne dust.

It is clear that the albedo impacts of biochar warrant further research, particularly into finding optimal strategies to mitigate the impact, and into the how changes in soil albedo develop over time as biochar is translocated vertically or horizontally, and the surface properties of the biochar change due to coating with minerals and organic matter and occlusion within soil aggregates.

As a final note, it is important to reiterate that albedo reduction is an issue for all types of soil C sequestration, not only biochar. Although there is little research to quantify the impact of non-pyrogenic SOM on albedo, Meyer et al. (2012) found no difference between the reduction of soil albedo by compost or biochar, per unit mass of C added.

8.6 PLANT RESPONSE TO BIOCHAR

Many studies have shown positive responses of crop or biomass yield to biochar amendment, although negative responses are also sometimes seen (Figure 8.4 and Jeffery et al. 2011; Macdonald et al. 2014; Mukherjee & Lal 2014). Even though the focus of this chapter is on the climate-change impacts of biochar rather than on its agronomic use, nonetheless it is pertinent to include a discussion of biochar's impact on plant growth, because altered rates of plant growth are not only of

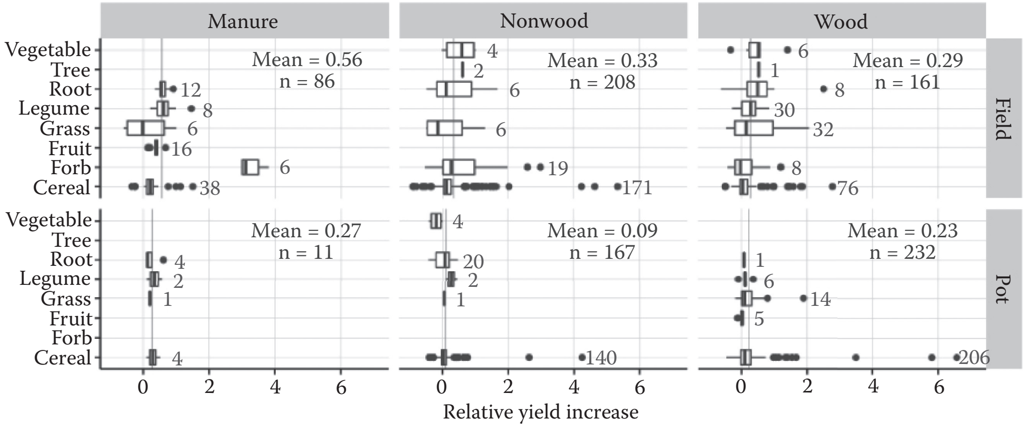

FIGURE 8.4 **(See color insert.)** Yield response of crops to biochar additions (difference between biochar amended yield and control, expressed as a fraction of control yield). Data from 865 treatments from 74 published articles are broken down by field trial versus pot trial, by feedstock type (manure, wood or non-wood), and by crop type. Vertical red line on each panel shows the mean crop response. Mean response and number treatments for each panel are also given in red text. Numbers shown in blue adjacent to each box indicate the number of treatments in the sub-category.

interest from a crop production standpoint, but also affect climate. There are five primary ways in which the soil fertility and crop productivity impacts of biochar relate to climate-change mitigation:

1. Sustainable intensification of cropping systems from increased yields may reduce pressure on land use and have a positive impact on indirect land use change (Wicke et al. 2012) (see also Section 7.4 below).
2. Biochar can improve fertilizer use efficiency, thereby reducing emissions associated with fertilizer manufacture and transport (Sohi et al. 2010). The manufacture of N fertilizer emits approximately 3.2 Mg CO_2 Mg^{-1} N (West & Marland 2002), which is approximately equal to the 3.0 Mg CO_2e Mg^{-1} N direct N_2O emissions from fertilizer application (Del Grosso et al. 2006).
3. Increased NPP can increase biomass available for mitigation through increased inputs available for building SOC stocks (Roberts et al. 2010), producing more biochar (Woolf et al. 2010), generating bioenergy, or other biomass-based mitigation approaches (Woolf et al. 2016).
4. In perennial (particularly woody) systems, increased NPP can increase biomass carbon stocks (Scharenbroch et al. 2013; Ghosh et al. 2015; Thomas & Gale 2015).
5. Last, but not least, the value of increased yield and reduced fertilizer requirement is critical in making biochar systems economically viable (McCarl et al. 2009; Field et al. 2013). Unless biochar provides a long-term improvement in soil fertility it is not economically competitive with other uses of the same biomass to provide climate-change mitigation, such as bioenergy or bioenergy with carbon capture and storage (Woolf et al. 2016).

There are several mechanisms by which biochar can affect crop yields. These include direct provision of nutrients; altering soil pH; increasing CEC, which in turn can improve fertilizer use efficiency and thus nutrient uptake for a given fertilizer application rate; and by increasing the water holding capacity (WHC) of sandy soils or the drainage of clayey soils (Jeffery et al. 2011; Sohi et al. 2010; Atkinson et al. 2010). The impacts of biochar's nutrient content and liming potential are likely to be short-lived effects, whereas CEC and WHC impacts are associated with the C matrix provided by biochar and are likely to persist while the biochar remains. Indeed, the CEC of biochar increases over time as its surface develops oxygenated functional groups through oxidation from exposure to oxygen and water (Liang et al. 2006). Effect on WHC varies with soil, biochar, and application rate. Biochar has been shown to increase WHC by up to 84%, with the greatest increases being on sandy soils, or to reduce WHC by up to 45%, with the greatest decreases being on clayey soils (Masiello et al. 2015).

The complex interaction of the various mechanisms by which crop yields are affected means that biochars have variable impacts on plant growth, depending on the biochar physical and chemical properties (which in turn depend on both feedstock and production conditions; Enders et al. 2012), properties of the soil, requirements of the target crop, application rate, depth of incorporation into soil, and time since application.

Although many studies have measured the impact of biochar on crop yields, the number of studies that directly investigate the underlying mechanisms driving these responses remains low. To fill this gap, some meta-analyses have attempted to discern general relationships from the corpus of published data. Jeffrey et al. (2011) found that biochar altered yields by –28% to +39%, with a mean increase of 10%. In a similar updated meta-analysis with more recent data, Jeffrey et al. (2015) found the mean yield increase was 18% relative to controls without biochar. Negative yield responses, where they have been observed, can generally be attributed to use of biochar with chemical properties that are inappropriate for the soil, crop or production system. Most often this involves the application of high pH biochar to soils that already have a neutral or alkaline pH, or the use of biochar whose high C:N ratio and high easily mineralizable -C content can give rise to N immobilization. These types of negative impacts can be avoided through the provision of appropriate decision-making tools or guidelines to agronomists and farmers. The greatest positive effects on crop yield were seen in acidic to neutral pH soils, and coarse to medium textured soils. Jeffrey et al. (2011) suggested, on this basis, that the main mechanisms for yield increase may be liming, improved WHC, and improved nutrient availability. WHC has been shown to The biochar feedstocks that showed the greatest yield increase was poultry litter, whereas biosolids were the only feedstock showing a statistically significant negative effect (Jeffery et al. 2011). Crane-Droesch et al. (2013) found soil CEC and organic C were the strongest predictors of yield response, with low cation exchange and low C associated with positive response, and that yield response increased over time since initial application. Biochar characteristics, on the other hand, were not significant predictors of yield impact (Crane-Droesch et al. 2013). Woolf et al. (2016) conducted a meta-analysis on a subset of published data that excluded biochars made from manures, thus excluding nutrient provision as a mechanism for yield impact, with the intention to better understand the extent to which longer lasting mechanisms affect yield. They found that soil pH and CEC were overwhelmingly the most important predictors of yield response, with soil texture, biochar carbon fraction, and fertilizer and biochar application rates of secondary importance.

While these meta-analyses are helpful to shed some light on the mechanisms by which biochar affects yield, they are of necessity limited by the extent of available published data, with most studies being only short term (relating to the first or second cropping season following biochar application), and the full range of soil and crop types and environmental and management conditions remaining relatively unexplored.

Notwithstanding these shortcomings in available data and lack of studies specifically designed to test for mechanisms, some clear patterns are already apparent. It is important in this regard to take care to distinguish between uncertainty and manageable variability (Lehmann & Rillig 2014), with much of the observed variation in crop response being attributable to predictable biochar–soil–crop interactions. Yield benefits are generally greatest in poor soils, especially light-textured acidic or degraded soils, particularly those with low soil CEC. Although a correlation between higher crop response and low soil CEC suggests that biochar may offer long term improvement to the fertility of low CEC soils (Woolf et al. 2016), experimental data on long term effects remain sparse. Furthermore, in some studies, unfavorable changes in soil chemical, physical and biological properties, and reductions in crop yields have been reported (Mukherjee & Lal 2014). An improved understanding of the mechanisms underlying crop response to biochar will help to direct biochar applications into systems they can most benefit, and also avoid the application of unsuitable types of biochar in cropping systems where they may do harm.

In conclusion, although there is a substantial body of evidence indicating that biochar can improve yields in infertile or degraded soils, a high uncertainty remains in the expected long-term (>3 years) response of specific soil-crop systems to biochar amendments. Further research is needed to provide long term data on crop response in a variety of soils, cropping systems and agroecological zones. A greater focus on research that aims to understand the mechanistic causes of observed impacts should also be a priority. This will support the development of decision-support models that can ensure use of different biochar types in cropping systems they can most benefit.

8.7 GHG ACCOUNTING FOR BIOCHAR

Methods for quantifying GHG emissions fluxes from biochar systems are required for GHG inventory, for GHG accounting against performance targets, for C footprint quantification and for calculating abatement at project level. Applicable methods depend on the purpose and the specific sources and sinks that are included.

8.7.1 GHG Inventory and Reporting

GHG inventory may include national level inventory undertaken for reporting under the United Nations Framework Convention on Climate Change (UNFCCC), for which the Intergovernmental

Panel on Climate Change (IPCC) prepares guidance, and may also refer to organizational inventory, undertaken for sub-national reporting or for regulatory agencies and corporations to establish compliance or to track progress towards emission reduction goals.

A GHG inventory is usually undertaken on annual basis, estimating all GHG emissions and removals that occur within the national or organizational boundaries within a specified period. To avoid double-counting, the inventory is usually confined to those facilities for which the country or organization has control, that is, to emissions that occur within their boundaries.

National GHG inventory reporting is sector-based, and GHG fluxes associated with biochar systems will be counted in different sectors: fuel use in the energy sector; C stock changes in biomass and soil, in the land use, land use change and forestry (LULUCF) sector; and emissions of N_2O and CH_4 from soils and manure management in agriculture (Cowie et al. 2012). Methods for inventory and reporting for biochar systems would need to integrate with existing quantification methods for each of these sectors. Quantification of some sources and sinks is straight forward, while others require adaptation to recognize the mitigation benefits of biochar. The IPCC publishes Tier 1 default methods (equations and emissions factors), and many countries have developed Tier 2 (country-specific emissions factors) or Tier 3 (measured or comprehensively modelled) methods for their key source categories.

It is important that the national inventory is accurate at the national scale. High spatial precision is not necessary for national scale assessment of net emissions and progress towards mitigation targets, although monitoring and verification of spatially heterogeneous C stocks and GHG fluxes may be facilitated by spatially-explicit reporting. Whether or not high spatial resolution is provided, comprehensive coverage of emissions sources and sinks is desirable to assist policy-makers in gauging the success of our collective efforts, globally, to curb emissions.

8.7.2 GHG Accounting

GHG accounting refers to assessing performance with respect to a GHG emissions reduction target, such as the commitments made by Annex I countries under the Kyoto Protocol. GHG accounting may be less comprehensive than GHG inventory reporting, as it may focus on certain sources or sinks targeted by the policy. As with GHG inventory reporting, GHG accounting is usually conducted on an annual basis, and the same quantification methods can be applied. For accounting, the objective is to ensure that results reflect effort in pursuing agreed mitigation activities, and high precision is less important.

8.7.3 Project and Product Accounting

Project-level accounting is undertaken to assess the mitigation benefits of an activity, such as within emissions trading schemes where abatement activities earn offset credits. The C footprinting quantifies the net GHG emissions associated with a specific product and may be used for product labelling to inform consumers, or for business-to-business communication to inform downstream customers. In both cases the intention is to encourage a change in behavior to reduce net GHG emissions. For each of these applications it is necessary to take a full life cycle approach, considering emissions and removals across the supply chain and also indirect effects, so that the full climate change impacts of the biochar system can be quantified. Ideally, all significant emissions sources and sinks directly or indirectly affected by the activity should be included.

For project- and product-level GHG accounting the most important criteria are that the method is cost-effective whilst being sufficiently accurate to ensure credibility that abatement has occurred, to encourage maximum participation and therefore maximum abatement. Methods for project- and product-level accounting should therefore be as simple as possible, conservative (i.e., tending to underestimate rather than to overestimate GHG abatement), and readily audited under a verification process. Methods based on emissions factors linked to specific practices are much more readily applied and verified than methods based on physical measurements at the site where biochar is applied. Project methods must also be widely applicable, or at least their applicability should be readily specified and assessed. For example, a GHG source may be excluded if the project activity is restricted to ensure that the source is not a high risk. For example, methane emitted from biomass decomposition during storage could be minimized by requiring that the biomass is dried before storage and stored in dry conditions under cover, thereby avoiding the need to quantify CH_4 emissions.

To quantify the climate change impacts of a biochar system for carbon footprinting and project-level accounting it is necessary to quantify the life cycle emissions of the biochar system and the processes and products that it displaces. For projects, this involves comparing the biochar system with the "no-biochar" reference scenario (Figure 8.5). Note that Figure 8.1 shows a reference and biochar system from the perspective of the fate of the biomass. Figure 8.5, in contrast, illustrates the services supplied by biochar system, and the corresponding provision of the same services in the reference system. The reference, also known as baseline, could be the situation at start of project, a BAU (business as usual) projection, or a forward counterfactual that envisages the no-biochar future. In research, or to inform policy development, it can be enlightening to compare several alternative scenarios.

Carbon footprinting of products is based on LCA methodology, which is standardized through ISO (ISO 2006a,b, 2013). LCA is commonly undertaken using proprietary or open source LCA software that facilitates the construction of models, access to databases of inventory data, and agreed models for analysis.

LCA approaches can be distinguished as consequential (CLCA) or attributional (ALCA). CLCA considers the direct and indirect effects of producing an additional unit of the product, while ALCA quantifies the impacts of the average unit of production, focusing on the direct supply chain emissions. CLCA applies system expansion and substitution to handle co-products, giving the studied product a credit for emissions avoided due to products displaced by the co-product (ISO 2015). ALCA handles co-products through allocation, dividing the supply chain emissions between the different products according to economic value or physical features such as energy content or mass. CLCA gives more accurate results, though often with greater uncertainty, and is recommended for

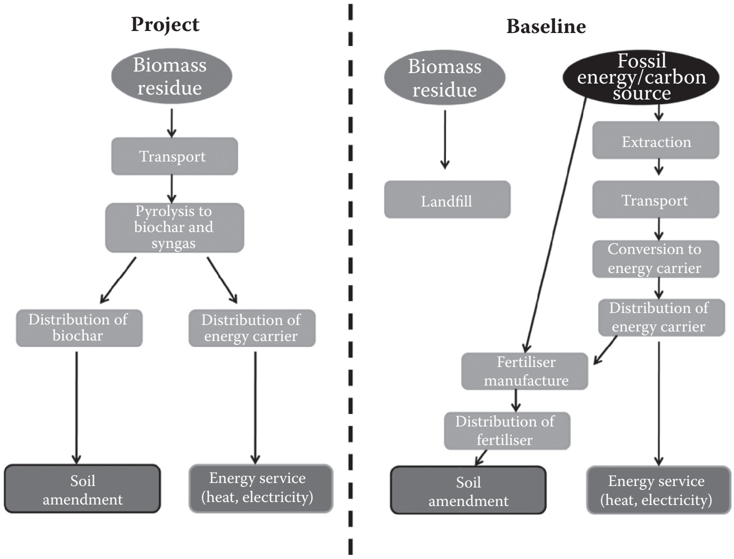

FIGURE 8.5 Project-level GHG accounting compares the life cycle emissions from the biochar system with those of the reference system providing the same services.

policy development (Brandão et al. 2014). ALCA is often employed for policy implementation, as it is considered easier, and more appropriate because it reflects the emissions and removals over which the operator has control.

8.7.4 METHODS FOR QUANTIFYING GHG FLUXES FROM BIOCHAR SYSTEMS

Biochar systems can affect sources and sinks, and can contribute to abatement through carbon sequestration, direct reduction in GHG emissions, and indirect reduction in GHG emissions. Below, each of these processes is considered with respect to its readiness for inclusion in GHG accounting, and, for abatement processes considered ready, a method for quantification is briefly described.

8.7.4.1 Carbon Sequestration

Increased persistence: Biochar systems that include the production of biomass and production of biochar remove CO_2 from the atmosphere and sequester C in pyrolysed biomass, which is more persistent than the raw biomass, thus delaying emissions for tens to thousands of years, as described in Section 8.2. Stabilization of biomass C is the most important contributor to abatement in most LCA studies of biochar systems (Cowie et al. 2015). Budai et al. (2013) reviewed alternative methods to assess the persistence of biochar, and concluded that $H:C_{org}$ ratio is a suitable measure for application in project-level accounting. They proposed the metric $BC_{+100,}$ the quantity of biochar C remaining after 100 years, and defined cut-offs at 0.4 and 0.7 in order to identify stability classes: for an $H:C_{org}$ value lower than or equal to 0.4, which is considered "highly stable", at least 70% of the biochar C is predicted to remain in soil for ≥100 years ($BC_{+100} = 70\%$), whereas for an $H:C_{org}$ value greater than 0.4 and lower than or equal to 0.7, considered "stable," a BC_{+100} of 50% can be conservatively expected.

This method for estimating abatement has several advantages. First, the method can be applied to any biochar technology and production conditions (kiln temperature, heating rate and residence time); these factors will affect biochar persistence, and these impacts will be reflected in the measured persistence. Second, it can be applied at the point of production of the biochar, and so does not require a mechanism to track the fate of the biochar. However, it is necessary to ensure that the biochar is not combusted, such as for cooking fuel. This can be assured when the biochar is produced from biomass mixtures that include manure, for example. With clean feedstocks, measures may be required to confirm application to soil.

Negative priming: As discussed in Section 8.4, biochar addition to soil can delay the decomposition of native SOM and newly added add OM, such as plant litter and root exudates. While negative priming has been demonstrated in several soil types, agricultural systems and environments, knowledge is currently too immature to allow this benefit to be predicted in GHG accounting. A change in SOC stock could be measured directly, through soil sampling and analysis such as applied to other soil C enhancement projects (for example, the Australian Emissions Reduction Fund methodology, Australian Government 2014). Care should be taken to avoid double-counting: if C persistence is estimated as in 1a, then the amount of C added in biochar should be deducted from the measured C stock. If an offsets credit is granted on the basis of measured changes, then the permanence of that sequestration needs to be assured through a program that requires maintenance of the sequestered C. Permanence measures can include a make-good requirement in the event of loss; a buffer of unsold abatement; or measures that allow the maintenance obligation to be transferred to other sequestration activities.

Increased plant growth: Biochars have variable impacts on plant growth, depending on the feedstock, properties of the receiving soil and requirements of the target crop, and negative responses are sometimes seen (Section 8.4 above and Macdonald et al. 2014).

Yield benefits are generally greatest in poor soils, especially light-textured acidic or degraded soils. Because the result varies with soil type, biochar type, crop, and rate of application (see Section 6), it is not possible to provide a generically-applicable method to estimate response. Measured increase in plant growth could be included in the calculation of project abatement, expressed as the increase in average C stock.

8.7.4.2 Emissions Reduction

Reduced nitrous oxide emissions from soil: Application of biochar can reduce emissions of N_2O from soil, through a range of mechanisms discussed above (section 3), with meta-analysis showing an average reduction of 54% (Cayuela et al. 2014). Cayuela et al. (2015) have identified a relationship between decrease in N_2O emissions and the molar H/C_{org} and N/C_{org} ratios of the biochar, which hold promise as a basis for a quantification method similar to that used to estimate carbon stability. However, given that the long-term persistence of the N_2O abatement has not been well established, conservative accounting of N_2O impacts would assume that it is a short-lived effect (Section 8.3).

Reduced fuel use: Biochar can enhance water holding capacity, thereby reducing the need for pumping irrigation water, and may reduce soil strength, decreasing fuel use. These effects cannot readily be generalized. Other factors could increase fuel use in the biochar system compared to the reference system: collection and processing of biomass, construction of pyrolysis facilities, transport of biochar. To the extent that reduction in use of electricity and/or liquid fuels relative to the reference system can be documented, a reduction in GHG emissions could be included in project accounting.

Reduced emissions during composting: Adding biochar to compost can reduce emissions of CH_4 and N_2O during composting (e.g., Agyarko-Mintah et al. 2016). However, there is currently insufficient knowledge to develop a model to estimate this reduction, and direct measurement is impractical on a routine basis.

8.7.4.3 Indirect Avoided Emissions

Avoided fossil fuel use: Pyrolysis gases can be used to displace fossil fuels for heat and electricity. IPCC Tier 1 or national Tier 2 emission factors can be used with activity data to quantify avoided emissions.

Avoided biomass decomposition: Pyrolysis of biomass may avoid emissions of CH_4 from the decomposition of biomass such as manure that would otherwise be stockpiled, crop residues that would be burned or processing residues that would be landfilled. IPCC methods (equations and emission factors) used to quantify these emissions sources can be used to estimate emissions avoided.

Reduced fertilizer manufacture: Biochars can reduce the requirement for chemical fertilizer by enhancing NUE. If a reduction in chemical fertilizer use per unit crop produced can be demonstrated, avoided emissions from fertilizer production could be calculated using life cycle inventory data for the specific fertilizer products avoided.

There are also factors that reduce the abatement that must be included:

Emission from pyrolysis: CH_4 and N_2O emissions are produced during pyrolysis at variable rates depending on the design of the facility, feedstock properties and skill of the operator. For small scale facilities such as cookstoves and drum ovens, a conservative estimate of emissions based on published literature (MacCarty et al. 2008; Whitman et al. 2011; Sparrevik et al. 2015) should be included. For larger scale facilities, gases should be flared or combusted to produce heat or electricity. Furthermore, emissions should be measured to establish the average emissions for each feedstock processed at the facility, and these emissions should be deducted from the calculated abatement.

Positive priming: Positive priming, that is, the stimulation of turnover of native SOC has been suggested to reduce the climate-change mitigation benefits of biochar systems (Wardle et al. 2008; Luo et al. 2011). However, more recent research (reviewed by Whitman et al. 2015b and discussed above in section 4), suggests that positive priming is not likely to cause a significant loss of abatement in mineral soils. However, acknowledging the small risk of positive priming it is conservatively suggested that the quantity of biochar C stabilized should be discounted by 5%, as proposed by Koper et al. (2013).

Carbon stock loss in biomass and soil: If additional biomass is harvested for biochar, this may lead to a reduction in biomass C or soil C at the harvest site. Any reduction in average C stock should be included the calculation of abatement.

Indirect land use change (iLUC): If biomass crops are grown to produce biochar the biomass crops may displace other crops producing food, feed or fiber. The continued demand for these crops may lead to land use change elsewhere to provide land for these crops. iLUC is challenging to prove or quantify, so it is difficult to include in the quantification of abatement at project level. Because of this difficulty in quantification, current international standards for LCA and climate finance do not require the inclusion of iLUC (Finkbeiner 2014). Nevertheless, some biochar studies have estimated iLUC (e.g., Roberts et al. 2010), and methods for inclusion of iLUC in LCA have been proposed (Schmidt et al. 2015). iLUC risk can be avoided by using biomass residues for biochar production. Risk of iLUC should be assessed and iLUC should be included in a sensitivity analysis in studies conducted to inform policy (Muñoz et al. 2015). Ideally, iLUC associated with biochar should be investigated in comprehensive global modelling using partial or general-equilibrium economic models that take into consideration market-mediated interactions between land use, energy sector, food prices and other macroeconomic indicators of production, consumption and trade. Equilibrium models such as FAPRI-CARD (Food and Agricultural Policy Research Institute - Center for Agricultural and Rural Development), GCAM (Global Change Assessment Model), and GTAP (Global Trade Analysis Project) that account for factors such as land, food, fiber and energy prices, maps of land suitability, proximity to transport infrastructure and existing crop production have been applied previously to the question of quantifying iLUC due to biofuels (Searchinger et al. 2008; Hertel et al. 2008; Schmidt et al. 2015; Flugge et al. 2017), and could be applied also to biochar systems. Despite the inherent, and possibly intractable, difficulties in rigorously quantifying iLUC impacts of specific projects, the general philosophy outlined above that GHG accounting should be conservative indicates that either (a) iLUC should be accounted for with a sensitivity analysis that includes conservative estimates in its range, or (b) projects should demonstrate that iLUC is not a risk, because non-competitive biomass such as unused crop or forestry residues is being sourced as feedstock.

8.7.5 SUMMARY OF GHG ACCOUNTING FOR BIOCHAR

Quantification of GHG fluxes differs depending on the purpose. National inventory applies IPCC methods to quantify annual emissions and removals. In contrast, GHG accounting for biochar projects compares the life cycle emissions and removals for a biochar system with those for a reference system providing the same services. Ideally, methods for project-level accounting are based on cost-effective methods utilizing scientifically-based models. Biochar knowledge has improved markedly over the last decade, providing the basis for estimation methods for the key processes that contribute to abatement. Thus, the molar H:C_{org} ratio has been proposed as a simple indicator of C persistence, with the $BC_{+100,}$ the quantity of biochar C remaining after 100 years, proposed as the metric to distinguish “highly stable” and ”stable” biochars. The molar H:C_{org} and N:C_{org} ratios also appear to provide a suitable basis for estimating the reduction in N_2O emissions from soil, although the longevity of such impacts remains uncertain. Other abatement processes including reduction in emissions from manure handling, fuel use in irrigation and cultivation, and fertilizer manufacture can be estimated using emissions factors, life cycle inventory data and activity data. Inclusion of negative priming and enhanced biomass production will require on-site sampling and measurement, which will substantially increase transaction costs and effort required for GHG estimation and verification.

8.8 DISCUSSION AND CONCLUSIONS

8.8.1 Evidence-Based Policy

The discussion presented in this chapter shows that, although many uncertainties remain, nonetheless a large body of evidence has been developed in the scientific literature over the last decade which allows us to move forwards with some clear policy guidelines. Although some data are still lacking to predict the climate-change mitigation performance of biochar in all environmental conditions, soils types and management systems, enough is known now to design biochar systems that adopt best practice recommendations and are applied in soils and cropping systems where a positive outcome is expected. The data presented in this chapter provides clear guidance on what types of best management practices (BMPs) will ensure that biochar projects avoid the potential pitfalls of negative impacts. These recommendations include:

1. Use of pyrolysis temperatures and/or reaction times that are high enough to create biochar that is sufficiently persistent to guarantee carbon sequestration over centennial timescales or longer. The $H:C_{org}$ ratio of the biochar providing a simple and reliable proxy for its persistence, with values less than 0.4 indicating high persistence.
2. The same pyrolysis conditions that offer improved biochar persistence also tend to offer the best mitigation of soil N_2O emissions, although the longevity of this impact remains to be proven.
3. Surface application of biochar without incorporation should be avoided, as this may lead to reductions in albedo that have a warming effect on the climate and may give rise to airborne dust. Additionally, incorporation of biochar will increase its agronomic effectiveness.
4. Soil fertility and crop productivity benefits of biochar will be optimized by applying biochar in soils that have some combination of low CEC, low SOM, low pH and low WHC, as these are the primary constraints that can be alleviated by biochar. The corollary is that adding high rates of biochar to soils with high pH can suppress yields and should be avoided, and adding biochar to soils that already have high CEC is unlikely to show any benefits in cycling or availability of nutrients.
5. The same pyrolysis conditions that optimize persistence and N_2O abatement will also eliminate the risk of lowering crop yield through nitrogen immobilization arising from easily mineralizable OM with a high C:N ratio.
6. Use of biomass feedstocks that are non-competitive with other demands will avoid emissions associated with direct or indirect land use change.

Existing GHG accounting life cycle assessment methodologies provide a sound basis to develop accounting methodologies for biochar, and there is sufficient evidence to parameterize the methodologies for biochar systems using conservative estimates. Thus, despite remaining uncertainties, use of conservative values means that we can confidently move forwards with biochar assessment protocols that estimate the lower range of potential mitigation impacts. Using this approach is prudent in that it provides assurance that biochar projects should perform better rather than worse than predicted, thus reducing or eliminating the risk that anticipated climate abatement goals will not be achieved.

It is also clear from the evidence provided in this chapter that policy development must be informed by comprehensive analyses that include biophysical, economic and social factors, so that market-mediated effects are also included. Policy development should also model a range of alternative scenarios to fully understand the range of possible outcomes for a given policy and also to compare predicted outcomes from a range of alternative policy options.

With regard to informing climate change mitigation policy at the regional or global scales, there is a clear need to include biochar in Integrated Assessment Models (IAMs) (Smith 2016; Woolf et al. 2016). Use of IAMs is the best way to understand where biochar fits into an overall portfolio of mitigation options in terms of both its technical and economic performance. IAMs and/or general equilibrium models will also provide the best means to assess potential impacts of biochar policy frameworks on broader outcomes such as land use and land use change, and on food and energy prices.

8.8.2 CONCLUSION

In summary, it is concluded that biochar offers high potential as a climate change mitigation technology, but that careful design and monitoring of projects, policy frameworks, and agricultural extension advice will be required to optimize results and to avoid negative outcomes from poor implementation practices. Robust and conservative GHG accounting methodologies can already be applied based on the existing evidence base and LCA. These accounting methodologies can be improved over time by ongoing research, particularly with respect to improving predictability of long-term impacts on priming, soil N_2O emissions, and crop yield responses. However, the large body of scientific evidence that has been accumulated over the last decade means that well-designed biochar projects can already be deployed at low risk while we continue to learn more about the mechanisms involved.

ACRONYMS

C	Carbon
C_{org}	Organic carbon
N	Nitrogen
H	Hydrogen
O	Oxygen
N_2O	Nitrous oxide
CH_4	Methane
CO_2	Carbon dioxide
CO_2e	Carbon dioxide equivalent
SOM	Soil organic matter
SOC	Soil organic carbon
NUE	Nutrient-use efficiency
GHG	Greenhouse gas
LCA	Life cycle assessment
NPP	Net primary production
CEC	Cation exchange capacity
OM	Organic matter
MRT	Mean residence time
PAH	Polyaromatic hydrocarbon
NO_3	Nitrate
BC	Black carbon
RF	Radiative forcing
WHC	Water holding capacity

UNFCCC	United Nations Framework Convention on Climate Change
LULUCF	Land use, land use change, and forestry
IPCC	Intergovernmental Panel on Climate Change
BAU	Business as usual
ISO	International Standards Organization
CLCA	Consequential LCA
ALCA	Attributional LCA
iLUC	Indirect land use change
LUC	Land use change
FAPRI	Food and Agricultural Policy Research Institute
CARD	Center for Agricultural and Rural Development
GCAM	Global Change Assessment Model
GTAP	Global Trade Analysis Project
IAM	Integrated Assessment Model

REFERENCES

Abel, S., Peters, A., Trinks, S., Schonsky, H., Facklam, M., and Wessolek, G. 2013. Impact of Biochar and Hydrochar Addition on Water Retention and Water Repellency of Sandy Soil. *Geoderma* 202–203 (July): 183–91. doi.org/10.1016/j.geoderma.2013.03.003.

Agyarko-Mintah, E., Cowie, A., Singh, B. P., Joseph, S., Van Zwieten, L., Cowie, A., Harden, S., and Smillie, R. 2016. Biochar Increases Nitrogen Retention and Lowers Greenhouse Gas Emissions When Added to Composting Poultry Litter. *Waste Management* 61: 138–149.

Alburquerque, J. A., Sánchez-Monedero, M. A., Roig, A., Cayuela, M. L., 2015. High Concentrations of Polycyclic Aromatic Hydrocarbons (Naphthalene, Phenanthrene And Pyrene) Failed to Explain Biochar's Capacity to Reduce Soil Nitrous Oxide Emissions. *Environmental Pollution* 196: 72–77.

Archontoulis, Sotirios V., Huber, I., Miguez, F. E., Thorburn, P. J., Rogovska, N., and Laird, D. 2015. A Model for Mechanistic and System Assessments of Biochar Effects on Soils and Crops and Trade-Offs. *GCB Bioenergy*. http://onlinelibrary.wiley.com/doi/10.1111/gcbb.12314/pdf.

Atkinson, Christopher J., Fitzgerald, Jean D., and Hipps, Neil A. 2010. Potential Mechanisms for Achieving Agricultural Benefits from Biochar Application to Temperate Soils: A Review. *Plant and Soil* 337 (1–2): 1–18.

Australian Government. 2014. Carbon farming initiative. Soil Sampling Design—Method and Guidelines. Australian Government Department of the Environment. http://www.environment.gov.au/system/files/pages/b341ae7a-5ddf-4725-a3fe-1b17ead2fa8a/files/cfi-soil-sampling-design-method-and-guidelines.pdf

Bingeman, C. W, Varner, J. E., and Martin, W. P. 1953. The Effect of the Addition of Organic Materials on the Decomposition of an Organic Soil. *Soil Science Society of America Journal* 17 (1): 34–38.

Borchard, N., Spokas, K., Prost, K., Siemens, J., 2014. Greenhouse Gas Production in Mixtures of Soil with Composted and Noncomposted Biochars Is Governed by Char-Associated Organic Compounds. *Journal of Environmental Quality* 43: 971–979.

Bozzi, E., Genesio, L., Toscano, P., Pieri, M., and Miglietta, F.. 2015. Mimicking Biochar-Albedo Feedback in Complex Mediterranean Agricultural Landscapes. *Environmental Research Letters* 10 (8): 084014.

Brandão, M., Clift, R., Cowie, A., and Greenhalgh, S. 2014. The Use of Life Cycle Assessment in the Support of Robust (Climate) Policy Making: Comment on "Using Attributional Life Cycle Assessment to Estimate Climate-Change Mitigation." *Journal of Industrial Ecology* 18: 461–463.

Budai, A., Zimmerman, A., Cowie, A., Webber, J., Singh, B., Glaser, B., Masiello, C., Andersson, D., Shields, F., and Lehmann, J. 2013. Biochar Carbon Stability Test Method: An Assessment of Methods to Determine Biochar Carbon Stability. *Carbon Methodology, IBI*. http://www.biochar-international.org/sites/default/files/IBI_Report_Biochar_Stability_Test_Method_Final.pdf.

Cayuela, M. L., van Zwieten, L., Singh, B. P., Jeffery, S., Roig, A., and Sánchez-Monedero, M. A. 2014. Biochar's Role in Mitigating Soil Nitrous Oxide Emissions: A Review and Meta-Analysis. *Agriculture, Ecosystems & Environment* 191: 5–16.

Cayuela, M., Jeffery, S., and Van Zwieten, L. 2015. The Molar H: Corg Ratio of Biochar Is a Key Factor in Mitigating N_2O Emissions from Soil. *Agriculture, Ecosystems & Environment* 202: 135–138.

Cayuela, M. L., Sánchez-Monedero, M. A., Roig, A., Hanley, K., Enders, A., and Lehmann, J. 2013. Biochar and Denitrification in Soils: When, How Much and Why Does Biochar Reduce N_2O Emissions? *Scientific Reports* 3: 1732.

Chen, S., Rotaru, A.-E., Shrestha, P. M., Malvankar, N. S., Liu, F., Fan, W., Nevin, K. P., and Lovley, D. R. 2014. Promoting Interspecies Electron Transfer with Biochar. *Scientific Reports* 4: 5019.

Coleman, K., and Jenkinson, D. S. 2008. *ROTHC-26.3.* Rothamsted Research, Harpenden, Herts, UK.

Conrad, R., 1996. Soil Microorganisms as Controllers of Atmospheric Trace Gases (H2, CO, CH4, OCS, N2O, and NO). *Microbiological Reviews* 60: 609–640.

Cotrufo, M. Francesca, Wallenstein, Matthew D., Boot, Claudia M., Denef, Karolien, and Paul, Eldor. 2013. The Microbial Efficiency-Matrix Stabilization (MEMS) Framework Integrates Plant Litter Decomposition with Soil Organic Matter Stabilization: Do Labile Plant Inputs Form Stable Soil Organic Matter? *Global Change Biology* 19 (4): 988–995.

Cowie, A., Woolf, D., Gaunt, J., Brandão, M., Anaya de la Rosa, R., Cowie, A., Lehmann, J., and Joseph, S. 2015. Biochar, Carbon Accounting and Climate Change. In *Biochar for Environmental Management: Science, Technology and Implementation*, 763–794. Lehmann and Joseph (Eds), Routledge (Oxon, UK): 763–794.

Cowie, A., Eckard, R., and Eady, S. 2012. Greenhouse Gas Accounting for Inventory, Emissions Trading and Life Cycle Assessment in the Land-Based Sector: A Review. *Crop and Pasture Science* 63: 284–296.

Crane-Droesch, Andrew, Abiven, Samuel, Jeffery, Simon, and Torn, Margaret S. 2013. Heterogeneous Global Crop Yield Response to Biochar: A Meta-Regression Analysis. *Environmental Research Letters* 8 (4): 044049.

Cui, Jun, Ge, Tida, Kuzyakov, Yakov, Nie, Ming, Fang, Changming, Tang, Boping, and Zhou, Chunlin. 2017. Interactions between Biochar and Litter Priming: A Three-Source 14 C and δ 13 C Partitioning Study. *Soil Biology and Biochemistry* 104: 49–58.

Davidson, E. A., and Kanter, D. 2014. Inventories and Scenarios of Nitrous Oxide Emissions. *Environmental Research Letters* 9: 105012.

Del Grosso, S. J., Parton, W. J., Mosier, A. R., Walsh, M. K., Ojima, D. S., and Thornton, P. E. 2006. DAYCENT National-Scale Simulations of Nitrous Oxide Emissions from Cropped Soils in the United States. *Journal of Environmental Quality* 35 (4): 1451–1460. doi.org/10.2134/jeq2005.0160.

Enders, Akio, Hanley, Kelly, Whitman, Thea, Joseph, Stephen, and Lehmann, Johannes. 2012. Characterization of Biochars to Evaluate Recalcitrance and Agronomic Performance. *Bioresource Technology* 114: 644–653.

Erb, K. H. 2007. A Comprehensive Global 5 Min Resolution Land-Use Data Set for the Year 2000 Consistent with National Census Data. *Journal of Land Use Science* 2 (3): 191–224.

Felber, R., Leifeld, J., Horák, J., and Neftel, A. 2014. Nitrous Oxide Emission Reduction with Greenwaste Biochar: Comparison of Laboratory and Field Experiments. *European Journal of Soil Science* 65: 128–138.

Field, John L., Keske, Catherine M. H., Birch, Greta L., DeFoort, Morgan W., and Cotrufo, M. Francesca. 2013. Distributed Biochar and Bioenergy Coproduction: A Regionally Specific Case Study of Environmental Benefits and Economic Impacts. *GCB Bioenergy* 5 (2): 177–191.

Finkbeiner, Matthias. 2014. Indirect Land Use Change—Help beyond the Hype? *Biomass and Bioenergy* 62 (March): 218–221.

Flugge, M., Lewandrowski, J., Rosenfeld, J., Boland, C., Hendrickson, T., Jaglo, K., Kolansky, S., Moffroid, K., Riley-Gilbert, M., and Pape, D. 2017. A Life-Cycle Analysis of the Greenhouse Gas Emissions of Corn-Based Ethanol. Report prepared by ICF under USDA Contract No. AG-3142-D-16-0243. January 30, 2017.

Fowles, M. (2007). Black Carbon Sequestration as an Alternative to Bioenergy. *Biomass and Bioenergy*, 31 (6): 426–432.

Gaunt, J. L., and Lehmann, J. 2008. Energy Balance and Emissions Associated with Biochar Sequestration and Pyrolysis Bioenergy Production. *Environmental Science & Technology* 42 (11): 4152–4158.

Genesio, L., Miglietta, F., Lugato, E., Baronti, S., Pieri, M., and Vaccari, F. P. 2012. Surface Albedo Following Biochar Application in Durum Wheat. *Environmental Research Letters* 7 (1): 014025.

Genesio, Lorenzo, Vaccari, Francesco Primo, and Miglietta, Franco. 2016. Black Carbon Aerosol from Biochar Threats Its Negative Emission Potential. *Global Change Biology* 22 (7): 2313–2314.

Ghosh, S., Fern Ow, L., and Wilson, B. 2015. Influence of Biochar and Compost on Soil Properties and Tree Growth in a Tropical Urban Environment. *International Journal of Environmental Science and Technology* 12 (4): 1303–1310.

Glaser, B., Haumaier, L., Guggenberger, G., and Zech, W. 2001. The ‘Terra Preta’ Phenomenon: A Model for Sustainable Agriculture in the Humid Tropics. *Naturwissenschaften* 88 (1): 37–41.

Glaser, B., Zech, W., and Woods, W. I. 2004. History, Current Knowledge and Future Perspectives of Geoecological Research Concerning the Origin Of Amazonian Anthropogenic Dark Earths (Terra Preta). In *Amazonian Dark Earths: Explorations in Space and Time* (pp. 9–17). Springer, Berlin, Heidelberg.

Hagemann, N., Harter, J., Kaldamukova, R., Guzman-Bustamante, I., Ruser, R., Graeff, S., Kappler, A., and Behrens, S. 2017a. Does Soil Aging Affect the N2O Mitigation Potential of Biochar? A Combined Microcosm and Field Study. *GCB Bioenergy* 9 (5): 953–964.

Hagemann, N., Kammann, C. I., Schmidt, H.-P., Kappler, A., Behrens, S. 2017b. Nitrate Capture and Slow Release in Biochar Amended Compost and Soil. *PLOS ONE* 12: e0171214.

Harter, J., Krause, H.-M., Schuettler, S., Ruser, R., Fromme, M., Scholten, T., Kappler, A., and Behrens, S., 2014. Linking N2O Emissions from Biochar-Amended Soil to the Structure and Function of the N-Cycling Microbial Community. *ISME Journal* 8: 660–674.

Harter, J., Weigold, P., El-Hadidi, M., Huson, D.H., Kappler, A., and S., 2016. Soil Biochar Amendment Shapes the Composition of N2O-Reducing Microbial Communities. Sci*ence of The Total Environment* 562: 379–390.

Hertel, T., Rose, S., and Tol, R. 2008. Land Use in Computable General Equilibrium Models: An Overview. GTAP Technical Paper. https://www.gtap.agecon.purdue.edu/resources/download/3659.pdf.

Hodge, A., Robinson, D., and Fitter, A, 2000. Are Microorganisms More Effective Than Plants at Competing for Nitrogen? *Trends in Plant Science* 5–308.

Hollister, C. C., Bisogni, J. J., and Lehmann, J. 2013. Ammonium, Nitrate, and Phosphate Sorption to and Solute Leaching from Biochars Prepared from Corn Stover (L.) and Oak Wood (spp.). *Journal of Environmental Quality* 42 (1): 137–144.

Houghton, J. T., Meira Filho, L. G., Lim, B., Treanton, K., Mamaty, I. et al. 1997. Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories. v. 1: Greenhouse Gas Inventory Reporting Instructions.—v. 2: Greenhouse Gas Inventory Workbook.—v. 3: Greenhouse Gas Inventory Reference Manual. http://agris.fao.org/agris-search/search.do?recordID=XF2015041434.

Hu, H.-W., Chen, D., and He, J.-Z. 2015. Microbial Regulation of Terrestrial Nitrous Oxide Formation: Understanding the Biological Pathways for Prediction of Emission Rates. *FEMS Microbiology Reviews* 39: 729–749.

Hüppi, R., Felber, R., Neftel, A., Six, J., and Leifeld, J. 2015. Effect of Biochar and Liming on Soil Nitrous Oxide Emissions from a Temperate Maize Cropping System. *SOIL* 1: 707–717.

ISO (2006a). ISO 14040:2006 Environmental Management-Life Cycle Assessment-Principles and Framework. International Organisation for Standardization, Geneva, Switzerland.

ISO (2006b). ISO 14044:2006 Environmental Management—Life Cycle Assessment—Principles and Framework. Vol. ISO 14040:2006(E), pp. 28. International Organization for Standardization, Geneva, Switzerland, Switzerland.

ISO (2013). ISO/TS 14067:2013 Greenhouse Gases—Carbon Footprint of Products—Requirements and Guidelines for Quantification and Communication. International Organization for Standardization, Geneva, Switzerland.

ISO (2015). ISO 13065:2015 Sustainability Criteria for Bioenergy. International Organization for Standardization, Geneva, Switzerland.

Jeffery, S., Abalos, D., Spokas, K. A., and Verheijen, F. G. 2015. Biochar Effects on Crop Yield. In *Biochar for Environmental Management: Science, Technology and Implementation*, Vol. 2. Lehmann and Joseph (Eds), Routledge (Oxon, UK).

Jeffery, S., Verheijen, F. G., Van Der Velde, M., and Bastos, A. C. 2011. A Quantitative Review of the Effects of Biochar Application to Soils on Crop Productivity Using Meta-Analysis. *Agriculture, Ecosystems & Environment* 144: 175–187.

Jeffery, S., Verheijen, F. G. A., Kammann, C., and Abalos, D. 2016. Biochar Effects on Methane Emissions from Soils: A Meta-Analysis. *Soil Biology and Biochemistry* 101: 251–258.

Jeffery, S., Verheijen, F. G. A., Bastos, A. C., and Van Der Velde, M. 2014. A Comment on “Biochar and Its Effects on Plant Productivity and Nutrient Cycling: A Meta-Analysis”: On the Importance of Accurate Reporting in Supporting a Fast-Moving Research Field with Policy Implications. *GCB Bioenergy* 6 (3): 176–179.

Kappler, A., Wuestner, M. L., Ruecker, A., Harter, J., Halama, M., and Behrens, S. 2014. Biochar as an Electron Shuttle between Bacteria and Fe(III) Minerals. *Environmental Science & Technology* Letters 1: 339–344.

Kasozi, G. N., Zimmerman, A. R., Nkedi-Kizza, P., and Gao, B. 2010. Catechol and Humic Acid Sorption onto a Range of Laboratory-Produced Black Carbons (Biochars). *Environmental Science & Technology* 44: 6189–6195.

Keith, A., Singh, B., and Dijkstra, F. A. 2015. Biochar Reduces the Rhizosphere Priming Effect on Soil Organic Carbon. *Soil Biology and Biochemistry* 88: 372–379.

Kerré, B., Willaert, B., and Smolders, E. 2017. Lower Residue Decomposition in Historically Charcoal-Enriched Soils Is Related to Increased Adsorption of Organic Matter. *Soil Biology and Biochemistry* 104: 1–7.

Koper, T., Weisberg, P., Lennie, A., Driver, K., Simons, H., Rodriguez, M., Reed, D., Jirka, S., and Gaunt, J. 2013. Methodology for Biochar Projects, version 1.0. Public Comment Draft submitted to the American Carbon Registry (ACR). http://americancarbonregistry.org/carbon-accounting/methodology-for-biochar-projects

Kuzyakov, Y., and Xu, X. 2013. Competition between Roots and Microorganisms for Nitrogen: Mechanisms and Ecological Relevance. *New Phytologist* 198: 656–669.

Lehmann, J., Rillig, M. C., Thies, J., Masiello, C. A., Hockaday, W. C., and Crowley, D. 2011. Biochar Effects on Soil Biota—A Review. *Soil Biology and Biochemistry* 43: 1812–1836.

Lehmann, J., and Rillig, M. 2014. Distinguishing Variability from Uncertainty. *Nature Climate Change* 4 (3): 153–153.

Lehmann, J., Amonette, J. E., and Roberts, K. 2010. Role of Biochar in Mitigation of Climate Change." *Handbook of Climate Change and Agroecosystems: Impacts, Adaptation, and Mitigation* (pp. 343–363). Imperial College Press, London.

Lehmann, J., Gaunt, J., and Rondon, M. 2006. Bio-Char Sequestration in Terrestrial Ecosystems – A Review. *Mitigation and Adaptation Strategies for Global Change* 11 (2): 395–419.

Lehmann, J., Abiven, S., Kleber, M., Pan, G., Singh, B. P., Sohi, S. P., Zimmerman, A. R., Lehmann, J., and Joseph, S. 2015. Persistence of Biochar in Soil. *Biochar for Environmental Management: Science, Technology and Implementation*, 233–280.

Lehmann, J. 2007. A Handful of Carbon. *Nature* 447 (7141): 143–144.

Lenton, T. M., and Vaughan, N. E. 2009. The Radiative Forcing Potential of Different Climate Geoengineering Options. *Atmospheric Chemistry and Physics Discussions* 9: 2559–2608.

Liang, B., Lehmann, J., Solomon, D., Kinyangi, J., Grossman, J., O'Neill, B., Skjemstad, J. O. et al. 2006. Black Carbon Increases Cation Exchange Capacity in Soils. *Soil Science Society of America Journal* 70 (5): 1719–1730.

Liu, B., Mørkved, P. T., Frostegård, Å., and Bakken, L. R. 2010. Denitrification Gene Pools, Transcription and Kinetics of NO, N_2O and N_2 Production as Affected by Soil pH. *FEMS Microbiological Ecology* 72: 407–417.

Luo, Y., Durenkamp, M., De Nobili, M., Lin, Q., and Brookes P. C. 2011. Short Term Soil Priming Effects and the Mineralisation of Biochar Following Its Incorporation to Soils of Different pH. *Soil Biology and Biochemistry* 43: 2304–2314.

MacCarty, N., Ogle, D., Still, D., Bond, T., and Roden, C. 2008. A Laboratory Comparison of the Global Warming Impact of Five Major Types of Biomass Cooking Stoves. *Energy for Sustainable Development* 12: 56–65.

Macdonald, L. M., Farrell, M., Van Zwieten, L., and Krull, E. S. 2014. Plant Growth Responses to Biochar Addition: An Australian Soils Perspective. *Biology and Fertility of Soils* 50: 1035–1045.

Maestrini, B., Nannipieri, P., and Abiven, S. 2015. A Meta-Analysis on Pyrogenic Organic Matter Induced Priming Effect. *GCB Bioenergy* 7 (4): 577–590.

Masiello, C., Dugan, B., Brewer, C., Spokas, K., Novak, J. M., Liu, Z., and Sorrenti, G. 2015. Biochar Effects on Soil Hydrology. *Biochar for Environmental Management*, 2nd ed. Earthscan, Routledge.

McCarl, B., Peacocke, C., Chrisman, R., Chih-Chun, K., and Sands, R. 2009. Economics of Biochar Production, Utilisation and Emissions. In *Biochar for Environmental Management: Science and Technology*, Lehmann, J. & Joseph, S. (Eds). Earthscan.

Meyer, S., Bright, R. M., Fischer, D., Schulz, H., and Glaser, B. 2012. Albedo Impact on the Suitability of Biochar Systems To Mitigate Global Warming. *Environmental Science & Technology* 46 (22): 12726–12734.

Mørkved, P.T., Dörsch, P., and Bakken, L.R. 2007. The N2O Product Ratio of Nitrification and Its Dependence on Long-Term Changes in Soil pH. *Soil Biology and Biochemistry* 39: 2048–2057.

Mukherjee, A., and Lal, R. 2014. The Biochar Dilemma. *Soil Research* 52 (3): 217–230.

Muñoz, I., Schmidt, J. H., Brandão, M., and Weidema, B. P. 2015. Rebuttal to "Indirect Land Use Change (iLUC) within Life Cycle Assessment (LCA)—Scientific Robustness and Consistency with International Standards." *GCB Bioenergy* 7 (4): 565–566.

Nelissen, V., Saha, B. K., Ruysschaert, G., and Boeckx, P. 2014. Effect of Different Biochar and Fertilizer Types on N2O and NO Emissions. *Soil Biology Biochemistry* 70: 244–255.

Obia, A., Cornelissen, G., Mulder, J., and Dörsch, P. 2015. Effect of Soil pH Increase by Biochar on NO, N_2O and N_2 Production during Denitrification in Acid Soils. *PLoS ONE* 10, e0138781.

Oguntunde, P. G., Abiodun, B. J., Ajayi, A. E., and van de Giesen, N. 2008. Effects of Charcoal Production on Soil Physical Properties in Ghana. *Journal of Plant Nutrition and Soil Science* 171 (4): 591–596. doi.org/10.1002/jpln.200625185.

Paustian, K., Lehmann, J., Ogle, S., Reay, D., Robertson, G.P., and Smith, P. 2016. Climate-Smart Soils. *Nature* 532: 49–57.

Phillips, R. L., Song, B., McMillan, A. M. S., Grelet, G., Weir, B. S., Palmada, T., and Tobias, C., 2016. Chemical Formation of Hybrid Di-Nitrogen Calls Fungal Codenitrification into Question. *Scientific Reports* 6: 39077.

Placer County. 2015. Biochar Production Project Reporting Protocol: GHG Emission Reduction Accounting. Version 3.4. September 10. https://www.placer.ca.gov/~/media/apc/documents/apcd%.

Prévoteau, A., Ronsse, F., Cid, I., Boeckx, P., and Rabaey, K., 2016. The Electron Donating Capacity of Biochar Is Dramatically Underestimated. *Scientific Reports* 6: 32870.

Quin, P., Joseph, S., Husson, O., Donne, S., Mitchell, D., Munroe, P., Phelan, D., Cowie, A., and Van Zwieten, L., 2015. Lowering N2O Emissions from Soils Using Eucalypt Biochar: The Importance of Redox Reactions. *Scientific Reports* 5: 16773.

Richardson, D., Felgate, H., Watmough, N., Thomson, A., and Baggs, E., 2009. Mitigating Release of the Potent Greenhouse Gas N2O from the Nitrogen Cycle—Could Enzymic Regulation Hold The Key? *Trends in Biotechnology* 27: 388–397.

Roberts, K. G., Gloy, B. A., Joseph, S., Scott, N. R., and Lehmann, J. 2010. Life Cycle Assessment of Biochar Systems: Estimating the Energetic, Economic, and Climate Change Potential. *Environmental Science & Technology* 44 (2): 827–833. doi:10.1021/es902266r.

Sagrilo, E., Jeffery, S., Hoffland, E., and Kuyper, T. W. 2015. Emission of CO2 from Biochar-Amended Soils and Implications for Soil Organic Carbon. *GCB Bioenergy* 7 (6): 1294–1304.

Sánchez-García, M., Roig, A., Sanchez-Monedero, M. A., Cayuela, M. L., 2014. Biochar Increases Soil N2O Emissions Produced by Nitrification-Mediated Pathways. *Frontiers in Environmental Science* 2: 25.

Sanford, R. A., Wagner, D. D., Wu, Q., Chee-Sanford, J. C., Thomas, S. H., Cruz-García, C., Rodríguez, G., Massol-Deyá, A., Krishnani, K. K., Ritalahti, K. M., Nissen, S., Konstantinidis, K. T., and Löffler, F.E. 2012. Unexpected Nondenitrifier Nitrous Oxide Reductase Gene Diversity and Abundance sn Soils. *Proceedings of the National Academy of Science* 109: 19709–19714.

Saquing, J. M., Yu, Y.-H., and Chiu, P. C., 2016. Wood-Derived Black Carbon (Biochar) as a Microbial Electron Donor and Acceptor. *Environmental Science & Technology Letters* 3: 62–66.

Scharenbroch, B. C., Meza, E. N., Catania, M., and Fite, K. 2013. Biochar and Biosolids Increase Tree Growth and Improve Soil Quality for Urban Landscapes.” *Journal of Environmental Quality* 42 (5): 1372–1385. doi.org10.2134/jeq2013.04.0124.

Schmidt, J. H., Weidema, B. P., and Brandão, M. (2015). A Framework for Modelling Indirect Land Use Changes in Life Cycle Assessment. *Journal of Cleaner Production* 99: 230–238.

Searchinger, T., Heimlich, R., Houghton, R. A., Dong, F., Elobeid, A., Fabiosa, J., Tokgoz, S., Hayes, D., and Yu, T-H. 2008. Use of US Croplands for Biofuels Increases Greenhouse Gases through Emissions from Land Use Change. *Science* 319 (5867): 1238–1240.

Seifritz, W. 1993. Should We Store Carbon in Charcoal? *International Journal of Hydrogen Energy* 18 (5): 405–407.

Serbin, G., Daughtry, C. S. T., Hunt, E. R., Reeves, J. B., and. Brown, D. J. 2009. Effects of Soil Composition and Mineralogy on Remote Sensing of Crop Residue Cover. *Remote Sensing of Environment* 113 (1): 224–238.

Singh, B. P., and Cowie, A. L. 2014. Long-Term Influence of Biochar on Native Organic Carbon Mineralisation in a Low-Carbon Clayey Soil. *Scientific Reports*, 4. https://doi.org/10.1038/srep03687.

Six, J., and Paustian, K. 2014. Aggregate-Associated Soil Organic Matter as an Ecosystem Property and a Measurement Tool. *Soil Biology and Biochemistry* 68: A4–A9.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O’Mara, F., Rice, C., Scholes, B., Sirotenko, O., Howden, M., McAllister, T., Pan, G., Romanenkov, V., Schneider, U., Towprayoon, S., Wattenbach, M., and Smith, J. 2008. Greenhouse Gas Mitigation in Agriculture. *Philosophical Transactions of the Royal Society of London B: Biological Sciences* 36: 789–813.

Smith, P. 2016. Soil Carbon Sequestration and Biochar as Negative Emission Technologies. *Global Change Biology*, January, 22(3): 1315–1324. doi.org/10.1111/gcb.13178.

Sohi, S. P., Krull, E., Lopez-Capel, E., and Bol, R. 2010. A Review of Biochar and Its Use and Function in Soil.” In *Advances in Agronomy*, Vol. 105. D. L. Sparks (Ed.) Academic Press (Cambridge, MA). (pp. 47–82).

Sombroek, W. G., Nachtergaele, F. O., and Hebel, A. 1993. Amounts, Dynamics and Sequestering of Carbon in Tropical and Subtropical Soils. *Ambio*. 22 (7): 417–426.

Sparrevik, M., Adam, C., Martinsen, V., and Cornelissen, G. 2015. Emissions of Gases and Particles from Charcoal/Biochar Production in Rural Areas Using Medium-Sized Traditional and Improved “Retort” Kilns. *Biomass and Bioenergy* 72: 65–73.

Spokas, K. 2010. Review of the Stability of Biochar in Soils: Predictability of O:C Molar Ratios. *Carbon Management* 1 (2): 289–303. doi.org/10.1016/j.carbpol.2010.10.007.

Spokas, K. A., 2013. Impact of Biochar Field Aging on Laboratory Greenhouse Gas Production Potentials. *GCB Bioenergy* 5: 165–176.

Spokas, K. A., and Reikosky, D. C., 2009. Impacts of Sixteen Different Biochars on Soil Greenhouse Gas Production. *Annals of Environmental Science* 3: 179–193.

Stein, L. Y., and Yung, Y. L., 2003. Production, Isotopic Composition and Atmospheric Fate of Biologically Produced Nitrous Oxide. *Annual Review of Earth and Planetary Sciences* 31: 329–356.

Sun, T., Levin, B. D. A., Guzman, J. J. L., Enders, A., Muller, D. A., Angenent, L. T., and Lehmann, J. 2017. Rapid Electron Transfer by the Carbon Matrix in Natural Pyrogenic Carbon. *Nature Communications* 8: 14873.

Thomas, S. C., and Gale, N. 2015. Biochar and Forest Restoration: A Review and Meta-Analysis of Tree Growth Responses. *New Forests* 46 (5–6): 931–946.

Thomazini, A., Spokas, K., Hall, K., Ippolito, J., Lentz, R., and Novak, J. 2015. GHG Impacts of Biochar: Predictability for the Same Biochar. *Agriculture, Ecosystems & Environment* 207: 183–191.

Usowicz, B., Lipiec, J., Łukowski, M., Marczewski, W., and Usowicz, J. 2016. The Effect of Biochar Application on Thermal Properties and Albedo of Loess Soil under Grassland and Fallow. *Soil and Tillage Research* 164: 45–51.

van Groenigen, J. W., Huygens, D., Boeckx, P., Kuyper, T. W., Lubbers, I. M., Rütting, T., and Groffman, P. M., 2015. The Soil N Cycle: New Insights and Key Challenges. *SOIL* 1: 235–256.

Van Zwieten, L., Kammann, C., Cayuela, M. L., Singh, B. P., Joseph, S., Kimber, S., Clough, T., and Spokas, K. 2015. Biochar Effects Emissions of Non-CO 2 GHGs from Soil. In *Biochar for Environmental Management: Science and Technology II,* Lehmann and Joseph (Eds), Routledge (Oxon, UK), (Chapter 17, pp. 489–520).

Van Zwieten, L., Singh, B., Joseph, S., Kimber, S., Cowie, A., and Chan, K., 2009. Biochar and Emissions of Non-CO2 Greenhouse Gases from Soil. In *Biochar for Environmental Management: Science and Technology*, Lehmann and Joseph (Eds), Routledge (Oxon, UK): 227–249.

Venterea, R. T., Halvorson, A. D., Kitchen, N., Liebig, M. A., Cavigelli, M. A., Grosso, S. J. D., Motavalli, P. P., Nelson, K. A., Spokas, K. A., Singh, B. P., Stewart, C. E., Ranaivoson, A., Strock, J., and Collins, H., 2012. Challenges and Opportunities for Mitigating Nitrous Oxide Emissions from Fertilized Cropping Systems. *Frontiers in Ecology and the Environment* 10: 562–570.

Ventura, Maurizio, Alberti, G., Viger, M., Jenkins, J. R., Girardin, C., Baronti, S., Zaldei, A. et al. 2015. Biochar Mineralization and Priming Effect on SOM Decomposition in Two European Short Rotation Coppices. *GCB Bioenergy* 7 (5): 1150–1160.

Verheijen, F. G. A., Jeffery, S., van der Velde, M., Penížek, V., Beland, M., Catarina Bastos, A., and Keizer, J. J. 2013. Reductions in Soil Surface Albedo as a Function of Biochar Application Rate: Implications for Global Radiative Forcing. *Environmental Research Letters* 8 (4): 044008.

Wang, Jinyang, Xiong, Z., and Kuzyakov, Y. 2016. Biochar Stability in Soil: Meta-Analysis of Decomposition and Priming Effects. *GCB Bioenergy* 8 (3): 512–523.

Wang, Z., Zheng, H., Luo, Y., Deng, X., Herbert, S., and Xing, B. 2013. Characterization and Influence of Biochars on Nitrous Oxide Emission from Agricultural Soil. *Environmental Pollution* 174: 289–296.

Wardle, D. A., Nilsson, M.-C., & Zackrisson, O. 2008. Fire-Derived Charcoal Causes Loss of Forest Humus. *Science* 320: 629.

Wells, N. S., and Baggs, E. M. 2014. Char Amendments Impact Soil Nitrous Oxide Production During Ammonia Oxidation. *Soil Science Society of America Journal* 78: 1656–1660.

Weng, Z. H., Van Zwieten, L., Pal Singh, B., Kimber, S., Morris, S., Cowie, A., and Macdonald, L. M. 2015. Plant-Biochar Interactions Drive the Negative Priming of Soil Organic Carbon in an Annual Ryegrass Field System. *Soil Biology and Biochemistry* 90: 111–121.

West, T. O., and Marland, G. 2002. A Synthesis of Carbon Sequestration, Carbon Emissions, and Net Carbon Flux in Agriculture: Comparing Tillage Practices in the United States. *Agriculture, Ecosystems & Environment* 91 (1–3): 217–232.

Whitman, T., Nicholson, C. F., Torres, D., and Lehmann, J. 2011. Climate Change Impact of Biochar Cook Stoves in Western Kenyan Farm Households: System Dynamics Model Analysis. *Environmental Science & Technology* 45: 3687–3694.

Whitman, T., Scholz, S. M., and Lehmann, J. 2010. Biochar Projects for Mitigating Climate Change: An Investigation of Critical Methodology Issues for Carbon Accounting. *Carbon Management* 1 (1): 89–107.

Whitman, T., Enders, A., and Lehmann, J. 2014a. Pyrogenic Carbon Additions to Soil Counteract Positive Priming of Soil Carbon Mineralization by Plants. *Soil Biology and Biochemistry* 73 (June): 33–41.

Whitman, T., and Lehmann, J. 2015a. A Dual-Isotope Approach to Allow Conclusive Partitioning between Three Sources. *Nature Communications* 6: 8708.

Whitman, T., Pal Singh, B., Zimmerman, A. R., Lehmann, J., and Joseph, S. 2015b. Priming Effects in Biochar-Amended Soils: Implications of Biochar-Soil Organic Matter Interactions for Carbon Storage. In *Biochar for Environmental Management: Science, Technology and Implementation*, Lehmann and Joseph (Eds), Routledge (Oxon, UK): 455–488.

Whitman, T., Pepe-Ranney, C., Enders, A., Koechli, C., Campbell, A., Buckley, D. H., and Lehmann, J. 2016. Dynamics of Microbial Community Composition and Soil Organic Carbon Mineralization in Soil Following Addition of Pyrogenic and Fresh Organic Matter. *The ISME Journal* 10 (12): 2918–2930.

Whitman, T., Hanley, K., Enders, A., and Lehmann, J. 2013. Predicting Pyrogenic Organic Matter Mineralization from Its Initial Properties and Implications for Carbon Management. *Organic Geochemistry* 64: 76–83.

Whitman, T., Zhu, Z., and Lehmann, J. 2014b. Carbon Mineralizability Determines Interactive Effects on Mineralization of Pyrogenic Organic Matter and Soil Organic Carbon. *Environmental Science & Technology* 48 (23): 13727–13734.

Wicke, B., Verweij, P., van Meijl, H., van Vuuren, D. P., and Faaij, A. P. C. 2012. Indirect Land Use Change: Review of Existing Models and Strategies for Mitigation. *Biofuels* 3 (1): 87–100. doi.org/10.4155/bfs.11.154.

Wiedemeier, D. B., Abiven, S., Hockaday, W. C., Keiluweit, M., Kleber, M., Masiello, C. A., McBeath, A. V. et al. 2015. Aromaticity and Degree of Aromatic Condensation of Char. *Organic Geochemistry* 78: 135–143.

Wilson, M. F., and Henderson-Sellers, A. 1985. A Global Archive of Land Cover and Soils Data for Use in General Circulation Climate Models. *Journal of Climatology* 5 (2): 119–143.

Woolf, D. 2011. The Potential for Sustainable Biochar Systems to Mitigate Climate Change." Thesis for the Degree of Doctor of Philosophy, Swansea University.

Woolf, D. 2008. Biochar as a Soil Amendment: A Review of the Environmental Implications. *Organic Eprints* 13268.

Woolf, D., Amonette, J. E, Street-Perrott, F. A., Lehmann, J., and Joseph, S. 2010. Sustainable Biochar to Mitigate Global Climate Change. *Nature Communications* 1 (5): 1–9.

Woolf, D., and Lehmann, J. 2012. Modelling the Long-Term Response to Positive and Negative Priming of Soil Organic Carbon by Black Carbon. *Biogeochemistry* 111 (1–3): 83–95.

Woolf, D., Lehmann, J., and Lee, D. R. 2016. Optimal Bioenergy Power Generation for Climate Change Mitigation with or without Carbon Sequestration. *Nature Communications* 7: 13160.

Woolf, D., Lehmann, J., Fisher, E. M., and Angenent, L. T. 2014. Biofuels from Pyrolysis in Perspective: Trade-Offs between Energy Yields and Soil-Carbon Additions. *Environmental Science & Technology* 48 (11): 6492–6499.

Xu, H.-J., Wang, X.-H., Li, H., Yao, H.-Y., Su, J.-Q., and Zhu, Y.-G. 2014. Biochar Impacts Soil Microbial Community Composition and Nitrogen Cycling in an Acidic Soil Planted with Rape. *Environmental Science & Technology* 48 (16): 9391–9399.

Yang, W. H., Weber, K. A., and Silver, W. L. 2012. Nitrogen Loss from Soil through Anaerobic Ammonium Oxidation Coupled to Iron Reduction. *Nature Geoscience* 5: 538–541.

Zhang, Q., Wang, Y., Wu, Y., Wang, X., Du, Z., Liu, X., and Song, J. 2013. Effects of Biochar Amendment on Soil Thermal Conductivity, Reflectance, and Temperature. *Soil Science Society of America Journal* 77 (5): 1478–1487.

Zhang, Y., Hu, X., Zhang, D., Chen, W., and Zou, J. 2015. Effects of Biochar on Soil Surface Albedo, Temperature and Moisture in Agricultural Soil. *Research of Environmental Sciences* 8: 008.

Zheng, H., Wang, Z., Deng, X., Herbert, S., and Xing, B. 2013. Impacts of Adding Biochar on Nitrogen Retention and Bioavailability in Agricultural Soil. *Geoderma* 206: 32–39.

Zhu, X., Burger, M., Doane, T. A., and Horwath, W. R., 2013. Ammonia Oxidation Pathways and Nitrifier Denitrification Are Significant Sources of N_2O and NO under Low Oxygen Availability. *Proceedings of the National Academy of Science USA* 110: 6328–6333.

Zimmerman, A. R., Gao, B., and Ahn, M.-Y. 2011. Positive and Negative Carbon Mineralization Priming Effects among a Variety of Biochar-Amended Soils. *Soil Biology and Biochemistry* 43 (6): 1169–1179.