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ENVIRONMENTAL BENEFITS OF BIOCHAR

Biochar: A Synthesis of Its Agronomic Impact beyond Carbon Sequestration

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Biochar has been heralded as an amendment to revitalize degraded soils, improve soil carbon sequestration, increase agronomic productivity, and enter into future carbon trading markets. However, scientific and economic technicalities may limit the ability of biochar to consistently deliver on these expectations. Past research has demonstrated that biochar is part of the black carbon continuum with variable properties due to the net result of production (e.g., feedstock and pyrolysis conditions) and postproduction factors (storage or activation). Therefore, biochar is not a single entity but rather spans a wide range of black carbon forms. Biochar is black carbon, but not all black carbon is biochar. Agronomic benefits arising from biochar additions to degraded soils have been emphasized, but negligible and negative agronomic effects have also been reported. Fifty percent of the reviewed studies reported yield increases after black carbon or biochar additions, with the remainder of the studies reporting alarming decreases to no significant differences. Hardwood biochar (black carbon) produced by traditional methods (kilns or soil pits) possessed the most consistent yield increases when added to soils. The universality of this conclusion requires further evaluation due to the highly skewed feedstock preferences within existing studies. With global population expanding while the amount of arable land remains limited, restoring soil quality to nonproductive soils could be key to meeting future global food production, food security, and energy supplies; biochar may play a role in this endeavor. Biochar economics are often marginally viable and are tightly tied to the assumed duration of agronomic benefits. Further research is needed to determine the conditions under which biochar can provide economic and agronomic benefits and to elucidate the fundamental mechanisms responsible for these benefits.

Black carbon (BC) is the name given to the spectrum of chemical–thermal solid conversion products formed from carbonaceous materials, which could be biomass or fossil fuels (Goldberg, 1985; Masiello et al., 2002). The BC continuum contains all charred residues, with a lack of consistency over terminology ranging from char, charcoal, bone char, carbon ash, carbon black, black carbon, carbonized carbon, coke, and soot (Jones et al., 1997; Masiello, 2004). Recently, biochar has been added to this BC terminology mixture. In this review, the term *BC* is used for the carbonaceous solid byproduct of the chemical–thermal conversion of any carbon-containing material that may or may not be biomass. Biochar refers to BC that is produced as a vehicle of carbon sequestration from renewable and sustainable biomass (Lehmann, 2007). Therefore, biochar is BC, but not all BC is biochar.

Black carbon has been applied to soils virtually from the dawn of civilization, since fire pits were built on soil, and associated research can be documented to the start of modern science (Lefroy, 1883; Hall, 1910). In addition, BC use in agriculture dates back at least to the early 1600s in Japan and potentially earlier in China (cited in Ogawa and Okimori, 2010). These purposeful BC applications, combined with the natural deposition of BC (e.g., forest fires, prairie fires, volcanoes), have resulted in the widespread presence of BC in the soil organic matter pool (Skjemstad et al., 2002). The first use of the term *biochar* was around 1998 for the solid residual of biomass pyrolysis (Bapat and Manahan, 1998). In the late 1980s, there was an immense shift in the intended purpose for biomass pyrolysis—from an energy and chemical resource to a means of atmospheric carbon sequestration (Goldberg, 1985; Kuhlbusch and Crutzen, 1995). This alteration of purpose has prompted a shift in referring to BC that is produced

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Abbreviations: BC, black carbon; CEC, cation exchange capacity; GHG, greenhouse gas; VOC, volatile organic compound.

for carbon sequestration purposes (assumed to be applied to soils) as biochar (Karaosmanoglu et al., 2000; Lehmann, 2007; Laird, 2008).

The current application of biochar to soil is modeled after the Amazonian Terra Preta soils, which have higher soil fertility believed to result from intentional BC additions from "slash and char" agricultural practices (Mishra and Ramakrishnan, 1983; German, 2003; Mann, 2005). However, BC additions to soils have not uniformly resulted in soil fertility improvements. Charcoal spots (historical charcoal production sites) in Zambian forests possess slower plant regeneration rates than surrounding areas without BC remnants (Chidumayo, 1988), and the long-term application of pyrolysis products to soils at historical US wood pyrolysis plants (wood distillation) that occurred from 1800 to 1960 raised soil organic chemical concentrations to such high levels that some sites are included in the U.S. Environmental Protection Agency's Superfund program (Erstfeld and Snow-Ashbrook, 1999; Edenborn and Severson, 2007). Due to these diverse potential environmental impacts arising from soil application of biomass pyrolysis products, it is important to summarize existing observations and to document current gaps in our understanding of biochar's potential impacts, both positive and negative, to guide future use.

Biochar Production

Pyrolysis refers to the process of thermochemical decomposition of organic material at elevated temperatures in the absence of oxygen (Bridgwater, 1994). There are three product streams from pyrolysis: (i) noncondensable gases, (ii) a combustible bio-oil representing the condensable liquids (tars), (iii) and biochar, a solid residual coproduct. Pyrolysis of biomass dates back at least 5000 yr when the Egyptians formed pyroligneous acid (wood vinegar, tars, and smoke condensates or bio-oil) used for embalming (Baumann, 1960). Biomass pyrolysis products served as chemical and energy sources for the next ~5000 yr. Wood pyrolysis peaked in the early 1900s with the "standard distillation apparatus" (retort) that processed 10 cords of wood (25 Mg) within a 24-h period (Hawley, 1926). Biomass pyrolysis was replaced by coal in the 1800s as an energy source and by crude oil (petroleum) in the 1920s as a chemical source due to the improved economics of nonrenewable fossil fuels, the improved distillation efficiency of the higher-purity petroluem products, and the decline of woody resources in proximity to pyrolysis plants (Owen, 1975; Edenborn and Severson, 2007).

Pyrolysis can be an endothermic or exothermic reaction depending on the temperature of the reactants, becoming increasingly exothermic as the reaction temperature decreases (Mok and Antal, 1983). The exothermicity of the slow pyrolysis reaction per unit of biochar yield is reported to range from 2.0 to 3.2 kJ g^{-1} char (Mok and Antal, 1983; Milosavljevic et al., 1996). Hence, because the fixed carbon content of biomass is high, biochar formation commences at low temperatures where autogeneous pyrolysis begins (Mok and Antal, 1983; Milosavljevic et al., 1996). The self-sustaining nature of the low-temperature reaction explains why traditional methods of making charcoal, where biomass was buried underground, could carry on for days.

Current biochar production is focused on advanced pyrolysis systems (Bridgwater et al., 1999; Boateng et al., 2007; Zhang et al., 2007; Boateng et al., 2010b; Lima et al., 2010; Lima and Marshall, 2010). However, traditional charcoal production technologies are still used (Major et al., 2010b). Advanced conversion systems (Table 1) allow precise control of operating conditions, which, coupled with feedstock selection, can regulate the physical and chemical properties of biochar (Table 2) and allow for customization of biochar properties. Available biochar nutrients are linked to the differences in the composition of the original parent feedstock (Abdullah et al., 2010; Song and Peng, 2010; Yip et al., 2010), the moisture content of the feedstock (Yip et al., 2007), pyrolysis production conditions (Bridgwater, 1994; Bridgwater et al., 1999; Antal and Grønli, 2003; Hossain et al., 2011), and postproduction handling and processing (Azargohar and Dalai, 2008) because these factors control the degree of aromaticity and potential entrapment of noncarbon atoms in BC carbon ring structures. The resulting biochar nutrient contents are variable based on feedstock and production conditions, but this has been known for some time (Anonymous, 1840) (Table 2). Furthermore, differences have been noted in the chemistries of various biochars, despite the similarity in production conditions (DeGroot et al., 1991; Bustin and Guo, 1999). Recent advancements in pyrolysis conversions, particularly the process controls, have reduced this variability (Cantrell and Martin, 2011).

Further discussion of the various production techniques and impacts on the product yields are found in the associated references in Table 1. These are average ranges of the physical properties and yields, but quantities vary widely as a function of feedstock and presence of impurities (e.g., soil) (Table 2) (Spokas et al., 2011). Slow pyrolysis is the conversion technique that maximizes biochar yield, but the other variants of hydrothermal carbonization and microwave-assisted pyrolysis are also appealing due to their ability to handle wetter biomass sources, which reduces biomass drying costs (Table 1). Torrefaction had been established in the forestry industry for fuel upgrading of woody biomass, primarily for densification, reducing transportation costs due to moisture removal and increasing heating values (Bourgois and Guyonnet, 1988), which generates an energy product amenable to coal-blending and subsequent co-combustion (Bridgeman et al., 2010; Repellin et al., 2010; Phanphanich and Mani, 2011). Torrefaction is a mild form of pyrolysis, with production temperatures ranging between 200 and 300°C (Prins et al., 2006). However, torrefaction and hydrothermal carbonization are not optimal choices for biochar production because the oxygen to carbon (O/C) ratio of the resulting charred material is high (between 0.4 and 0.6) (Bridgeman et al., 2008), indicating lower aromatic character in the biochar (Fuertes et al., 2010). When biochar is produced at higher pyrolysis temperatures (400–700°C), its carbon is distributed in more polycondensed aromatic structures (Keiluweit et al., 2010). These polycondensed aromatic structures have low O/C ratios that are resistant to microbial degradation (Glaser et al., 2002; Kimetu and Lehmann, 2010; Zimmerman, 2010). These properties are important because biochar that resists microbial mineralization may be best suited for long-term soil carbon sequestration.

Characteristics of the raw feedstock biomass impart specific properties to the resulting biochar, such as ash content and its elemental constituents, density, and hardness. Biomass with high elemental contents usually produces an even higher elemental concentration in the ash portion of biochar (Abdullah et al., 2010). Particularly, biochars from grass, corn stover, and manure feedstocks contain higher amounts of ash than biochars from other biomass sources (Brewer et al., 2009) (Table 2). However, the ash content also can be reduced by post-production activities, such as acid washing (DeGroot et al., 1991). Specific biochar nutrient concentrations may be greater (Abdullah et al., 2010) or lower (DeGroot et al., 1991) than original feedstock nutrient concentrations. These results suggest that occasional volatilization and loss of nutrients during pyrolysis may be linked to higher production temperatures (Cantrell and Martin, 2011). The large range of operational maximum temperatures common to slow pyrolysis processes determines the extent of volatilization taking place and therefore the final composition of the resulting biochar in terms of residual volatile content. During pyrolysis, a series of cleavage and polymerization reactions occurs, resulting in the formation of fixed carbon (aromatic) structures that are thermally stable. Devolatilization and carbonization processes are involved, as described elsewhere (Aiman and Stubington, 1993; Sjöström, 1993; Alén et al., 1996; Drummond and Drummond, 1996). Biochars created from grasses and manures (including poultry litter) appear to possess higher nutrient contents than other feedstocks (Table 2).

Historically, BC has been an undesirable waste product because the primary focus has been on optimizing the liquid and gas products for energy conversion and not on biochar for carbon sequestration. Despite the long research history of pyrolysis, further research is needed into the techniques to optimize biochar yields. Biochar is not homogenous or a single material; differing nutrient and chemical structures and differing positions in the BC spectrum lead to the lack of a "one-size-fits-all" biochar (Novak and Busscher, 2011). A holistic and objective approach to the production of biochar is vital for economic success where profits from all co-products are optimized. Thus, for a biochar to deliver an agronomic benefit, it is important to understand how biochar quality (physical and chemical properties) is influenced by the choice of feedstock and the pyrolysis conditions used in its production (Antal and Grønli, 2003; Lehmann and Joseph, 2009; Novak et al., 2009b).

Biochar Impacts on Agronomic Yields

Soil fertility is influenced by a number of soil properties and involves a complex balance of biotic and abiotic reactions that are spatially and temporally dynamic. Adding biochar to soils may produce immediate effects on properties such as soil nutrition, water retention, or microbial activity (Atkinson et al., 2010; Lehmann et al., 2011), although these effects vary

Table 1. Description of advanced thermal conversion processes.†

Conversion type	Temp. range	Residency time	Heating rate	Product production (% of original feedstock mass)			Solid proximate analysis‡				Other notes
				Solid	Liquid	Gas	Moisture	VM	Ash	Fixed C	
	°C		°C s^{-1}	%							
Torrefaction	200–320	hours	<1	40–90	–	10–60	0–1	50–85	2–10	13–38	densification and increase energy value of solid residuals; energy generation
Slow pyrolysis	350–700	hours	1–100	15–40	20–55	20–60	0–5	5–20	2–10	40–90	devolatilization and polymerization reactions occur; maximum solid (biochar) yield
Fast pyrolysis	450–550	<1 min	>1000	10–30	50–70	5–15	0–5	40	30	40–60	typically fine powders (300–400 μm); dust problems maximize bio-oil production
Flash pyrolysis	300–800	<1 s	Similar to fast (>1000)	30–40	–	60–70	0–5	5–26	0–40	40–60	high VM potential negatives for soil/plant gas/solid production; no liquid (oil); elevated (~1 MPa)
Gasification	>800	seconds to minutes	Variable	0–10	–	90–100	n/a	n/a	n/a	n/a	ashes: pH and potential toxicity issues; conversion of biomass to energy (no biochar or liquid products)
Hydrothermal carbonization	150–400	Minutes to hours	n/a	5–40	20–40	2–10	10–40	50–90	5–15	4–10	chars less stable (higher O:C ratios); very high pressures (>5 MPa) handles wet biomass
Microwave-assisted pyrolysis	300–500	minutes to hours	n/a	20–30	0–20	50–70	10–25	20–30	20–25	50–60	higher gas yields from microwave assisted pyrolysis; handles wet biomass

† References: Torrefacation (Bridgeman et al., 2008; Repellin et al., 2010; Phanphanich and Mani, 2011); slow pyrolysis (Apaydin-Varol et al., 2007; Pütün et al., 2007; Boateng et al., 2010b; Lima and Marshall, 2010); fast pyrolysis (Boateng, 2007; Boateng et al., 2010a; Boateng et al., 2010b; Lima et al., 2010; Mullen et al., 2010); flash (Antal and Grønli, 2003; Deenik et al., 2010); gasification (Masclet et al., 1987; Ptasinski, 2008; Salleh et al., 2010; Fernández-Pereira et al., 2011); hydrothermal carbonization (Molton et al., 1981; Karagöz et al., 2005; Steinbeiss et al., 2009; Yuan et al., 2009; Rillig et al., 2010); microwave-assisted pyrolysis (Menéndez et al., 2006; Huang et al., 2008; Lei et al., 2009).

‡ Volatile matter (VM), ash content, and fixed carbon expressed on a dry weight basis.

depending on soil type (Tryon, 1948; Shneour, 1966; Spokas and Reicosky, 2009; Van Zwieten et al., 2010b). Nonetheless, because of its generally recalcitrant nature, biochar may also have long-term impacts on soil environments. Soil formation is the net result of several external and internal factors that influence or drive pedogenic processes (Jenny, 1946; Buol et al., 2003). Biochar potentially can influence soil-forming processes that govern the accumulation, transformation, and translocation of soil constituents and hence in the long term can modify soil pedogenic activity, morphology, and productivity (Richter, 2007). For biochar to serve a beneficial role in revitalizing nutrient-impoverished soils, there should be a noted increase in the quantity of plant-available nutrients and its nutrition retention capacity (McLauchlan, 2006; Sohi et al., 2010). To understand soil–biochar interactions, we must consider how these effects vary geographically and temporally.

Assemblages of soil microbial communities and their interaction with organic and inorganic plant nutrient turnover processes are complex (Ingham et al., 1985; Zak et al., 2003) and have a profound impact on soil functions and its fertility. Microbial diversity is altered in response to organic amendments (Pérez-Piqueres et al., 2006; Sullivan et al., 2006; Khodadad et al., 2011). Research has suggested that soil applications of biochar can have a significant impact on microbial C metabolism and population dynamics (Warnock et al., 2007; O'Neill et al., 2009; Ball et al., 2010; Warnock et al., 2010; Zimmerman et al., 2011). A number of explanations for these impacts have been offered, such as biochar sorption, including the presence of volatile organic compounds (VOCs) that can inhibit or stimulate microbial mineralization reactions or affect plant–microbial interactions (Graber et al., 2010; Spokas et al., 2010), variability in biochar's susceptibility to mineralization

Table 2. Biochars chemical and fertilizer equivalent ratios.†

Reference	Feedstock	Pyrolysis	% Ash (dry weight basis)	pH	Fertilizer equivalent ratio		
					N	P	K
		°C			kg per tonne of biochar		
Novak et al., 2009b	peanut hull	400	8.2	7.9	30	3	20
	peanut hull	500	9.3	8.6	30	3	20
	pecan shell	350	2.4	5.9	3	0.3	2
	pecan shell	700	5.2	7.2	5	0.5	5
	poultry litter	350	35.9	8.7	50	30	60
	poultry litter	700	52.4	10.3	30	40	90
	switchgrass	250	2.6	5.4	4	1	5
	switchgrass	500	7.8	8.0	10	2	10
	hardwoods	450–600	8.9	5.7	3	0.3	6
	pine chips	465	5.6	6.1	3	0.8	4
Brewer et al., 2009	corn stover	500	49.7	n/a‡	16	3	12
	hardwood	500	13.9	n/a	12	0.2	2
Singh et al., 2010a	wood (*Eucalyptus saligna*)	400 (activated)	4.2	7.7	2	0.1	2
	wood (*E. saligna*)	550 (activated)	4.4	9.5	2	0.2	2
	wood (*E. saligna*)	400	3.2	6.9	2	0.1	1
	wood (*E. saligna*)	550	4.4	8.8	3	0.2	2
	leaves (*E. saligna*)	400 (activated)	10	9.2	16	2	13
	leaves (*E. saligna*)	550 (activated)	11.8	9.8	17	3	15
	paper sludge	550 (activated)	65.4	9.2	2	0.4	0.5
	poultry manure	400	42.3	9.2	52	6	25
	cow manure	400	70.3	9.0	14	4	26
	cow manure	550	76.2	8.9	11	5	23
Cantrell et al., unpublished	dairy manure	350	24.2	9.2	30	10	14
	dairy manure	700	39.5	9.9	17	17	23
	feedlot manure	350	28.7	9.1	34	11	32
	feedlot manure	700	44.0	10.3	17	18	49
	poultry litter	350	30.7	8.7	53	21	49
	poultry litter	700	46.2	10.3	22	31	74
	turkey litter	350	34.8	8.0	43	26	40
	turkey litter	700	49.9	9.9	20	37	56
Cantrell and Martin, 2011	swine manure	350	32.5	8.2	37	39	18
	swine manure	700	52.9	8.2	26	59	26
van Zwieten et al., 2010a	sludge + wood chip (49%)	550	n/a	9.4	5	n/a	0.4
	sludge + wood chip (69%)	550	n/a	8.2	3	n/a	19

† These fertilizer equivalent ratios were based on the total element concentration and likely do not reflect true effective plant availability following soil application.

‡ Data not available in the reference.

(Novak et al., 2009b; Zimmerman, 2010), microbial habitat through pH modifications (Atkinson et al., 2010), beneficial micropores on the charcoal for microbial habitat (Warnock et al., 2007), or the presence of critical nutrients for microbial growth and metabolic energy transfer reactions (Garcia-Montiel et al., 2000). These and other microbial impacts have been reviewed elsewhere (Atkinson et al., 2010; Lehmann et al., 2011). The previous list highlights the importance of understanding the interactions of biochar with soil microbes, and this knowledge is vital to improve soil quality while raising crop productivity.

The past literature indicates an early interest in the use of BC to improve soil and crop growth (Lefroy, 1883). Past studies report that BC's effect on agronomic crop yield is variable, with production improvements ranging from negative to more than twofold over nonamended controls (Table 3). In 1833, there was a recommendation to slowly smother burning biomass under a soil cover and then to rapidly collect the BC and immediately apply it to improve agronomic performance (application rate ~0.54 kg charcoal m^{-2}) (referenced in Lefroy, 1883).

More recent biochar studies have yielded contrary results in soil quality and yield improvements (Table 3). A meta-analysis by Verheijen et al. (2009) predicted a short-term yield improvement of 12% from biochar additions, although this analysis included a limited subset of nine recent biochar specific studies (since 2007). However, there was limited accountability for the different biochar types across the different studies because biochar itself possesses a wide range of chemistries (Table 1). Biochar and BC additions have not consistently resulted in increased yields (Gundale and DeLuca, 2007; Rajkovich, 2010; Van Zwieten et al., 2010b). Without knowledge of the fundamental driving factors resulting in these decreased yields, our ability to extract statistically significant conclusions from existing studies is limited. From laboratory incubations, grass and nonwoody biomass biochar is more easily mineralized than wood-derived biochar, resulting in longer predicted soil residency times for wood biochar (Zimmerman, 2010). From a soil fertility perspective, this increased mineralization could provide nutrient resources to plants. On the other hand, food waste biochar (Rajkovich, 2010) and biochar with high volatile matter contents (Deenik et al., 2010) have also suppressed plant growth.

Approximately 50% of the compiled studies observed short-term positive yield or growth impacts, 30% reported no significant differences, and 20% noted negative yield or growth impacts (Table 3). However, due to potential publication biases, these percentages should only be taken as reflective of the studies presented here and not as evidence of an overall biochar likelihood of producing positive impacts (Møller and Jennions, 2001). There are a greater number of increased yield results reported for biochar additions that occurred in weathered or degraded soils having limited fertility and productivity (Table 3). Of the 50% of the compiled studies with positive yield improvements, a majority of the yield improvements have been realized from (i) traditional kiln-formed hardwood charcoal or (ii) chars that possess plant nutrients (e.g., high N content in poultry manure biochar). This observation was also recently stated by Haefele et al. (2011), who observed yield increases in rice of 16 to 35% with rice hull biochar in a nutrient-poor soil compared with larger increased rice yields reported using wood biochar in similar soils (Table 3).

Numerous potential reasons exist for this apparent improved performance of traditional hardwood charcoal biochar. First is the low availability of advanced pyrolysis units. This limited availability results in a bias in the literature, with a majority of the studies using traditional charcoal techniques for the creation of biochar (Table 3).

Second, biochars from fast pyrolysis units have been extremely variable. Recently, it has been suggested that this variability could result from the incomplete conversion of the biomass feedstock due to thermal limitations and irreproducibility of heat transfer (Bruun et al., 2011). Deenik et al. (2010) also noted variable volatile matter content in fast pyrolysis biochars. This translates to differences between batches of biochar, making them potentially unique despite similar production conditions.

Last, there are differences not only in biochar quality as a function of the production process but also linked to the postproduction storage or activation (Azargohar and Dalai, 2008; Nuithitikul et al., 2010). Activation can occur by simply cooling the biochar with water or exposing the hot biochar to atmospheric oxygen during cooling. Surface oxidation of BC, even at ambient conditions, alters surface chemical groups (Puri et al., 1958; Allardice, 1966; Cheng et al., 2006), which correspondingly influences the potential interactions with soil nutrient cycles (Bohn et al., 1985). Traditional soil kiln charcoal can be oxidized due to the exposure of the hot biochar to atmospheric air. However, often the postproduction handling of the biochar is not documented, which highlights the need for improved reporting of biochar postproduction handling and storage conditions.

Potential Responsible Mechanisms for Biochar Yield Responses

Recent studies have indicated a complex biochar and fertilizer interaction with respect to yield response (Chan et al., 2007). However, alterations in soil nutrient concentrations have not been able to fully predict yield increases (Turner, 1955; Gundale and DeLuca, 2007; Kimetu et al., 2008; Graber et al., 2010), suggesting involvement of other soil processes or properties. Biochar additions to infertile soils have been cited to improve soil cation exchange capacity (CEC) properties (Cheng et al., 2006; Liang et al., 2006; Grossman et al., 2010; Inyang et al., 2010; Lee et al., 2010). However, not all biochar–soil combinations cause an increase in CEC because no or minimal changes in CEC have also been observed after certain biochar additions to soils (Novak et al., 2009a; Nguyen et al., 2010) that have been linked to biochar production parameters (Singh et al., 2010a). Other studies have found that biochar addition may alter pH levels and the availability of soil nutrients such as Ca or Mg, which were found to limit maize growth in highly weathered tropical soils (Major et al., 2010a), or the availability of B and Mo, which are important cofactors in biological N fixation (Rondon et al., 2007), while decreasing exchangeable Al^{3+} and H^{+} concentrations (Novak et al., 2009a).

Table 3. Impacts of black carbon and biochar additions on the yield of various crops.

Reference	Country	Soil type	Crop	Addition/rate	Yield results (compared with control)
Asai et al., 2009	Laos	Laotian paddy soils; field plots	rice	charcoal (various)	higher grain yields at sites with low P availability with biochar
	Laos	Laotian paddy soils; field plots	rice	charcoal (various)	reduced grain yields in soils with a low indigenous N supply
Bovey and Miller, 1969	U.S.	Toa silty clay + sand	beans (*Phaseolus vulgaris* L.)	activated charcoal (640 mg kg^{-1})	+26% (yield increase)
	U.S.	sand	cucumbers	activated charcoal (640 mg kg^{-1})	−15% (yield reduction)
	U.S.	Toa silty clay	cucumbers and oats	activated charcoal (0–1% w/w)	+54% oats +77% cucumbers
Chen et al., 2010	Japan	Shimajiri maji soil (heavy clay)	sugarcane	sugarcane bagasse biochar (3% + fertilizer)	increased sugarcane yield
Colauto et al., 2010	Brazil	compost/soil	mushroom (*Agaricus brasiliensis*)	charcoal (?) (charcoal as casing layer)	−50%
Constantin et al., 1977	U.S.	culture media	tobacco (*Nicotiana tabacum*)	activated charcoal	sorbs plant hormones, inhibiting callus and shoot development (negative effects observed)
de Keijzer and Hermann, 1966	U.S.	laboratory/field various	conifer species	charcoal (various)	summarizes impact on germination of conifer species (positive, negative, and no impact)
		field plot	douglas-fir	charcoal (various)	increased germination tied to increased soil temperature
Deenik et al., 2010	U.S.	greenhouse/lab	lettuce and corn	fast pyrolysis macadamia nut shell (0–20% by wt)	yield decreases observed
Devonald, 1982	U.K.	growing media	garden peas (*Pisum sativum*)	activated charcoal (5% w/w)	significant decrease in shoot height/rot mass and nodulation in peas
Gaskin et al., 2010	U.S.	Tifton loamy sand soil (Plinthic Kandiudult)	corn	pine chip biochar (0, 11.2, 22.4 Mg ha^{-1})	2006: decrease with increasing BC†; 2007: increase with BC amounts
	U.S.	Tifton loamy sand soil (Plinthic Kandiudult)	corn	peanut hull biochar (0, 11.2, 22.4 Mg ha^{-1})	decreases/increases; no statistically significant pattern
Gundale and DeLuca, 2007	U.S.	sandy-skeletal, mixed, frigid Typic Dystrustepts	perennial grass (*Koeleria macrantha*)	laboratory produced charcoal (350°C, 2 h); various rates	yield suppressions (Conclusion: Differences existed between wildfire charcoal and laboratory created charcoal.)
				wildfire charcoal, various	yield increases (correlated with amount of charcoal)
Haefele et al., 2011	Philippines	anthraquic Gleysols	rice	rice husk biochar (traditional)	initial negative; after fourth season no significant effect
	Philippines	Humic Nitisols	rice	rice husk (traditional)	no significant effect
	Thailand	Gleyic acrisols	rice	rice husk biochar (traditional)	positive effects; poorest soil and most draft stress 16–35% yield increase
Herr et al., 1999	U.S.	forest soil	white pine (*Pinus strobus L.*)	lab burnt wood ash	no differences
Hossain et al., 2010	Australia	chromosol	tomato (*Lycopersicon esculentum*)	wastewater sludge biochar (10 t ha^{-1})	+64% with fertilizer additions
Iswaran et al., 1980	India	Delphi agricultural soil (no description)	moong (*Vigna radiata*)	charcoal (500 kg ha^{-1})	+20%
		soybean (*Glycine max*)	charcoal (500 kg ha^{-1})	+50%	
		pea (*Pisum sativum*)	charcoal (500 kg ha^{-1})	+60%	
		moong (*V. radiata*)	coal (500 kg ha^{-1})	+30%	
		soybean (*G. max*)	coal (500 kg ha^{-1})	+140%	
		pea (*P. sativum*)	coal (500 kg ha^{-1})	+70%	

Table 3. Continued.

Reference	Country	Soil type	Crop	Addition/rate	Yield results (compared with control)
Kadota and Niimi, 2004	Japan	potting mix	bedding plants	charcoal + PA	negative growth shown in French marigold and scarlet sage; positive effects for melampodium, scarlet sage, and zinnia
Kim et al., 2003	Korea	unknown	red pepper	charcoal	small particle size: increased yield; large particle charcoal: decreased root growth; no yield differences
Kimetu et al., 2008	Kenya	Ultisol	corn	biochar (traditional kiln) *Eucalyptus saligna* (7 tons BC ha^{-1})	+80 to +100%
Kratky and Warren, 1971	U.S.	vermiculite + activated carbon (greenhouse)	cucumbers (*Cucumis sativus L.*)	activated charcoal (7% w/w)	no differences
Kratky and Warren, 1971	U.S.	vermiculite + activated carbon	tomatoes (*Lycopersicum esculentum*)	activated charcoal 7% (w/w)	no differences
Kratky and Warren, 1971	U.S.	field plot soil (Indiana soil)	tomatoes (*L. esculentum*)	activated charcoal 7% (w/w)	no differences
Kulmatiski and Beard, 2006	U.S.	coarse-loamy, mixed mesic typic haploxerolls (field plots)	native and exotic grassland vegetation	activated charcoal (1% w/w)	no differences first year; second year: increased plant cover, heterotrophic bacteria differences noted
Lamb et al., unpublished	U.S.	Greenville fine sandy loam (fine, kaolinitic, thermic Rhodic Kandiudults)	peanut, corn, and cotton	fast pyrolysis, hardwood (22,500 & 45,000 kg ha^{-1})	no difference (year 1-ongoing)
Lau et al., 2008	various	various	various	various activated charcoals, various rates	positive and negative; mostly positive effects observed
Leibundgut, 1960	Germany	litter compost	conifer	charcoal (beech wood) (0.5–2 kg m^{-2})	no significant effect on germination; some inhibition on conifer seedling growth observed
Linscott and Hagin, 1967	U.S. (NY)	Lima silty clay loam	alfalfa	activated charcoal (8 lbs per acre)	50% increase; not statistically significant
Mohamed-Yasseen, 2001	Egypt	culture media	corn	activated charcoal (5 g L^{-1})	longer shoots and roots in the presence of charcoal
Namgay et al., 2010	Australia	Quartzipsamment	corn	wood; slow pyrolysis (550°C) (0, 10, and 50 mg kg^{-1})	no significant differences in shoot dry matter yield (10 wk)
Noguera et al., 2010	Colombia	inceptisol	rice	charcoal (wood) (0.2 and 0.5% w/w) (lab produced)	+30 to +200%
Nutman, 1952	U.K.	test tube/no soil	clover	charcoal (0.5–2% w/w)	beneficial increases in nodulation; does not occur if charcoal is ashed
O'Toole, 2010	Norway	Fluvic cambisol	rye grass (*Lolium perenne L*)	wheat-straw biochar (up to 30% w/w)	no effect with fertilizers; without or low fertilization: negative yield impacts
Oguntunde et al., 2004	Ghana	compared charcoal kiln soils	corn	compared wood charcoal kiln soils with non kiln soils (unknown rates)	+90%; observed differences were not fully explainable by nutrient availability
Rajkovich, 2010	U.S.	silt loam and loam	corn	variety of feedstocks examined; food wastes, paper mill wastes, wood, and manures at various temperatures (0.2, 0.5, 2, and 7% w/w)	decreased biomass seen in about one third of the tested mixtures: food wastes biochar (−18 to −85%), papermill biochar (−85%), +17% increase in poultry manure biochars (+17%)
Rondon et al., 2007	Columbia	clay–loam oxisol (Typic Haplustox)	beans (*P. vulgaris* L.)	kiln charcoal (0, 30, 60, and 90 g kg^{-1} soil)	46% (≤60); >90 resulted in yield decrease
Rutto and Mizutani, 2006	China	growing media	peach	activated charcoal (unknown)	no differences

Table 3. Continued.

Reference	Country	Soil type	Crop	Addition/rate	Yield results (compared with control)
Solaiman et al., 2010	Australia	sandy clay loam (field)	wheat (*Triticum aestivum*)	biochar (0, 1.5, 3.0, and 6 t ha^{-1})	increased yields at low fertilizer rates; no significant differences at recommended levels
Sorensen et al., unpublished	U.S.	Greenville fine sandy loam (fine, kaolinitic, thermic Rhodic Kandiudults)	cotton	fast pyrolysis hardwood (22, 45, 89, and 135 Mg ha^{-1})	no difference (year 1)
Spokas, unpublished	U.S.	Waukegan silt loam (fine-silty over skeletal mixed, super active, mesic Typic Hapludoll) Field plots	corn	fast pyrolysis sawdust biochar (22,460 kg ha^{-1})	no difference (year 1)
			corn	slow pyrolysis woodwaste (22,460 kg ha^{-1})	no difference (year 1)
			corn	slow pyrolysis wood pellet biochar (22,460 kg ha^{-1})	no difference (year 1)
		potting soil mix (greenhouse experiment)	lettuce, spinach, radish	fast pyrolysis macadamia nut (10% w/w)	significant decrease in growth rate, germination timing, and biomass production observed with biochar
		potting soil mix (greenhouse experiment)	lettuce, spinach, radish	slow pyrolysis wood pellet biochar (10% w/w)	
Steiner et al., 2007	Brazil	Xanthic Ferralsol (weathered)	rice (*Oryza sativa* L.) and sorghum (*Sorghum bicolor* L.) rotation	charcoal (forest wood) (11 Mg ha^{-1} charcoal (reference has 11 mg ha^{-1} , assumed Mg)	+170% with fertilizer; charcoal additions alone did not increase production
Suhardi et al., 2006	Indonesia	Bukit Suharto experiment field	*Shorea leprosula*	charcoal + fertilizer (0–100 g of charcoal per plot)	no impact on height, plant diameter or mycorrhizal formation
Tagoe et al., 2008	Africa		soybean	chicken manure BC	+41%
			cowpea	chicken manure BC	+146%
			soybean	municipal organic waste biochar	+20%
			cowpea	municipal organic waste biochar	+59%
Topoliantz et al., 2005	French Guiana	Oxisol	bean (*V. unguiculata sesquipedalis*)	charcoal (wood)	increased yield with green manure (manioc peels)
Vaccari et al., 2011	Italy	silty loam	wheat	wood charcoal	20–40% increase (only significant at $P = 0.10$)
Vantsis and Bond, 1950	U.K.	sand	clover	wood charcoal (0.5–2% w/w)	increases in dry weight and nitrogen fixation
			clover	animal charcoal (bone) (0.5–2% w/w)	inhibition of growth
			clover	activated charcoal (0.5–2% w/w)	increases in dry weight & N fixation
Wang and Huang, 1976	Taiwan	culture media	various	activated charcoal (3 g L^{-1})	improves growth by sorbing toxic metabolites
Yamato et al., 2006	Indonesia	Indonesia farmland soils (weathered)	corn	*Acacia mangium* (kiln bark charcoal), 10 L m^{-2}	site A: +190%; site B: no difference
		Indonesia farmland soils (weathered)	cowpea	*A. mangium* (kiln bark charcoal), 10 L m^{-2}	no differences
		Indonesia farmland soils (weathered)	peanut	*A. mangium* (kiln bark charcoal), 10 L m^{-2}	site A: +100%; site B: no difference
Zhang et al. 2011	China	Entic halpudept	rice	wheat straw biochar (10 and 40 t ha^{-1})	8–14% increase

† Black carbon.

Other explanations for biochar's crop yield impact have ranged from N immobilization leading to decreased N availability due to the high C/N biochar ratios (Rondon et al., 2007), liming effects of the biochar (Verheijen et al., 2009), reduced plant availability of macronutrients due to pH alterations (Hiradate et al., 2007; Makoto et al., 2010), and direct sorption of soil nutrients (Asai et al., 2009). Asai et al. (2009) tested the influence of biochar additions on a variety of soil types at 10 different locations and observed yield increases in soils with low P availability and improved plant response to

additional fertilizers with biochar additions. However, these findings are not universal; even fertilizer plus biochar additions have resulted in suppressed yields in some cases (e.g., Table 3). Therefore, soil nutrient status alone is not sufficient to explain all the crop responses observed but could be important after biochar amendments to weathered and low N- and P-containing soils due to fertilization (Table 2).

There are additional potential mechanisms responsible for biochar's effect on agronomic yield. Studies have shown altered rates and timing of seed germination as a function of biochar additions (e.g., de Keijzer and Hermann, 1966; Rillig et al., 2010). Differences in germination and consequentially plant emergence could influence plant growth and yield due to the timing of precipitation and accumulation of thermal time. In other words, plant seeds that are simultaneously sown in biochar-amended and non–biochar-amended soils that emerge at different times are also temporally equivalent to a varying planting date. Differences in planting date in field plots have been observed to affect plant growth and yield due to the timing of precipitation and accumulation of growing degree days (Egli and Bruening, 1992).

Biochar can also sorb, release, or catalyze transformations of compounds that affect plant and microbial growth. Black carbon has been observed to catalyze abiotic transformations of nitrogen-containing compounds (i.e., ammonia, nitric oxide, nitrate, or nitrous oxide) even at ambient conditions (Chang and Novakov, 1975; DeGroot et al., 1991; Aarna and Suuberg, 1997), particularly linked to ammonium formation from other N forms (Chang and Novakov, 1975). Black carbon has also been observed to catalyze the transformation of sulfur compounds, leading to the formation of sulfates (Novakov et al., 1974). Despite the fact that some of these studies investigated nonbiomass BC forms, biochar is BC and could possess chemistries similar to nonbiomass source BC as a function of production and processing conditions (Spokas, 2010). In addition to direct abiotic transformation of soil nutrients, the sorption of soil inhibitory chemicals by BC was hypothesized to be responsible for alterations in clover nodulation (Turner, 1955). The role of volatile organics in soil microbial and plant signaling is an emerging field (Insam and Seewald, 2010). In some cases, these VOCs may be sorbed by biochar particles (Turner, 1955; Warnock et al., 2007), whereas at other times VOC may be emitted from biochars (Spokas et al., 2010). This release or sorption of VOCs may cause plant allelopathic reactions and may inhibit or stimulate microbial functionality and positive or negative plant effects (Deenik et al., 2010; Graber et al., 2010). However, these chemical effects would be dependent on soil, microbial, plant, and biochar properties. Therefore, this role of biochar sorbing or releasing inhibitory chemicals could explain seemingly contradictory results because the effect would be a function of the respective concentration thresholds for the specific microbe or plant. However, the commonality of this hypothesis is lacking because characterization of the sorbed compounds is not a typical analysis conducted on biochar.

Biochar-induced yield improvements are further complicated by the occasional delayed response, with negative or no impact in the initial year followed by yield increases of varying degrees in subsequent years (Kulmatiski and Beard, 2006; Gaskin et al., 2010; Major et al., 2010b). These delayed responses are hypothesized due to aging of the biochar (e.g., oxidation or other chemical alteration) (Puri et al., 1958; Allardice, 1966; Cheng et al., 2006; Singh et al., 2010b). Chemical or thermal biochar activation drastically alters the surface chemistry (Azargohar and Dalai, 2006, 2008; Nuithitikul et al., 2010). Chemisorption of oxygen by biochar also alters the surface chemistry (Puri et al., 1958) and microbial degradability (Cheng et al., 2008), which could affect biochar nutrient availability. These abiotic chemisorption reactions occur at ambient conditions (Itay et al., 1989), which have only received limited attention in the biochar literature (Cheng et al., 2006; Spokas et al., 2009; Zimmerman, 2010; Jones et al., 2011). These postprocessing reactions can drastically alter the biochar and resulting observed impacts, which again highlights the need for documenting postproduction handling and storage of biochar.

The influence of biochar on soil fertility may be positive or negative depending on the quality and rate of biochar applied, with some uncertainty as to the exact mechanisms. As with soil application of other byproducts (Sumner, 2000; Ippolito et al., 2011), an application of high-nutrient biochar that exceeds recommended fertilization rates may unbalance soil nutrient levels, produce little improvement in soil nutrient retention, and increase nutrient leaching potentials. Some biochar nutrients are leachable despite the observations of nutrient sorption (Ding et al., 2010). Although control soils had no detectable soluble N or P in the leachate, the first leachate collected from soils treated with poultry litter biochar contained 3 and 8 μg mL^{-1} of NH_4–N and NO_3–N, respectively, and 36 μg mL^{-1} dissolved P concentrations (Novak et al., 2009b). The addition of hardwood charcoal to a typical Midwestern agricultural soil (Hapludoll) in a laboratory column study substantially reduced P and N nutrient leaching from a subsequent manure addition compared with controls, with greater reductions correlated with increasing charcoal rates (Laird et al., 2010a). However, the amounts of K, Mg, Zn, Ca, and total N leached from the columns only receiving charcoal were greater than the control columns, hence the conclusion that some charcoal nutrients are partially leachable (Laird et al., 2010b).

Soil nutrient improvements may take some time to be observed. One could envision a delay if the particular element is enclosed in a chemical ring structure because the kinetics of surface functional group oxidation and cleavage of ring structures would be rate limiting (Glaser et al., 2002; Liang et al., 2006; Yao et al., 2010). However, a majority of the existing studies have been limited to less than 3 yr, which may not be enough time for the soil nutrient cycle to be affected.

In conclusion, plant responses to biochar additions are the net result of production (e.g., feedstock and pyrolysis conditions) and postproduction (storage or activation) conditions. These processes can infer potentially unique properties to each batch of biochar, even from the same pyrolysis unit and biomass feedstock. The mechanisms resulting in negative plant impacts need to be fully understood due to the significant plant biomass reductions (35–87% of nonamended soil controls) that have been cited as a function of feedstock and production temperatures (Table 3). However, production temperature alone cannot describe the variability in laboratory biochar microbial impact assessments (Spokas, 2010).

Due to the lack of universal properties and characterization reported in biochar studies, full elucidation of the responsible processes from literature studies is unfeasible and requires additional detailed studies (Lehmann et al., 2011), particularly documenting biochar production and postproduction handling. Based on the available data, the processes by which biochar improves plant growth and yield are debatable, but this knowledge is critical to fully optimize biochar for agronomic purposes.

Biochar Economics

Assessing the economic feasibility of biochar requires evaluation of the entire system, which starts with feedstock production and preparation, transportation to the pyrolysis unit, net energy consumption or production of the pyrolysis process itself, storage, and biochar utilization, including transportation and application costs. Some of these costs could be offset by the other products that are typically coproduced with biochar and by potential sustained biochar benefits. At the pyrolysis facility that targets energy production, biochar represents a loss in energy production, which is a loss of revenue. As an aside, biochar can be used as an energy source (Boateng, 2007; Abdullah and Wu, 2009; Abdullah et al., 2010). However, this use of biochar as a fuel would not fall under the definition of biochar because there is no carbon sequestration. Direct energy use of biochar will be a competing use to carbon sequestration.

Increasing soil aggregation, water infiltration, and water-holding capacity may also reduce irrigation costs, especially in semiarid environments. Drought and subsequent crop stress is common in the arid western United States, and thus irrigation is commonly used to meet crop water demands. Soils in the arid to semiarid regions of the U.S. Pacific Northwest are typically fine-textured Aridisols and Mollisols that inherently have a high water-storage capacity. However, irrigation water quickly evaporates; consequently, the length of time that soil water is available for crops after irrigation or precipitation is of paramount concern. In a laboratory column leaching study, Brockhoff et al. (2010) observed an increase of 370% in water-holding capacity under gravity drainage after a 25% (v/v) biochar addition to sand. An increase in soil moisture storage due to biochar application could be appealing to crop producers in the semiarid to arid regions of the United States and in other drought-plagued regions around the world with limitated irrigation water availability. For the agricultural producer, the cost of biochar must be low enough that the benefits of soil application exceed the total cost of biochar application or the cost of applying additional irrigation water.

Select economic analyses have been conducted looking at the entire system. Roberts et al. (2010) conducted a life cycle assessment of biochar systems using corn stover, switchgrass, or yard wastes as feedstocks for greenhouse gas (GHG) offset values of \$20 Mg^{-1} CO_2 equivalents (CO_2e) and \$80 Mg^{-1} CO_2e. Only the yard waste feedstock could profitably be used for pyrolysis with \$20 Mg^{-1} CO_2e. However, all feedstocks were profitable at \$80 Mg^{-1} CO_2e, with the exception of one switchgrass scenario, where switchgrass production led to large indirect land use changes that negated the GHG benefits. This analysis assumed that 80% of the C in biochar was stable, with the remaining 20% released as CO_2 within the first few years of application (Roberts et al., 2010). This assumption is questionable given the fact that biochar is not a singular homogeneous product and that the degradability is directly tied to the overall chemistry (Spokas, 2010), which varies greatly as a function of biochar type (Zimmerman, 2010) as well as the variability of the biochar itself (Hedges et al., 2000). The analysis also assumed that there were no yield benefits to biochar application but that N, P, and K fertilizer use was reduced by 7.2% with biochar application and N_2O emissions were reduced by 50%. These may or may not be realistic assumptions due to the complex interaction of biochar and soil N-cycling and due to variations in the responses due to biochar and soil types as previously described.

McCarl et al. (2009) showed that both fast and slow pyrolysis production of biochar from corn stover was not economically feasible. The results were sensitive to pyrolysis facility costs, GHG offset price, energy prices, and impacts of biochar on crop yields. The analysis showed that pyrolysis would be profitable with GHG offset prices of more than \$58 Mg^{-1} CO_2e or \$71 Mg^{-1} CO_2e for fast or slow pyrolysis, respectively. The analysis also showed that pyrolysis would be profitable for biochar prices greater than \$450 Mg^{-1} or \$246 Mg^{-1} for fast or slow pyrolysis, respectively. These prices were substantially greater than the 2008 biochar energy value from combustion (\$55 Mg^{-1}) or soil application value (\$47 Mg^{-1}) (McCarl et al., 2009). The soil application value was based on the assumption of a persistent 5% maize yield increase and annual input cost savings of \$73 ha^{-1} due to reductions in need for nutrients, lime, and seed but did not include the value of any GHG offsets. The estimated soil application value might be conservative if the short-term 12% yield improvement indicated in the Verheijen et al. (2009) meta-analysis was obtained with maize production and was present for many years after biochar application. The magnitude and persistence of yield benefits are key to understanding the potential economic value of biochar application.

Granatstein et al. (2009) conducted an economic analysis on pyrolyzing biomass from forest thinning. Results showed that the lowest breakeven cost for a facility producing biochar was \$87 Mg^{-1} biochar for a stationary facility and up to \$1,742 Mg^{-1} biochar for a mobile unit. The potential impacts of GHG offsets were not included in the analysis. In a related study, Galinato et al. (2010) analyzed the profitability of applying biochar instead of agricultural lime to winter wheat fields in eastern Washington. The only crop production impact of the biochar included in the analysis was the effect on crop yields through changing soil pH. Without GHG offset payments, analysis showed that the biochar price would need to be less than \$4.34 Mg^{-1} for biochar application to be more profitable than lime. The breakeven biochar price for the farmer, or the price farmers would be willing to pay, will increase with increasing GHG offset payments, assuming the GHG offset is paid to the farmer. For example, given a GHG offset payment of \$31 Mg^{-1} CO_2e, biochar application is more profitable than lime application for biochar prices less than \$96 Mg^{-1} (Galinato et al., 2010). At this level, the agricultural value exceeds the lowest breakeven cost for a stationary facility, indicating the potential for profitable production of biochar.

Galinato et al. (2010) also alluded to an important aspect of evaluating the value of biochar for field application. Comparisons must be made with the next most profitable alternative, not simply a no-biochar alternative. The break-even price of biochar using this comparison was always higher than the price relative to the alternative with agricultural lime because this was the most profitable practice. Therefore, omitting relevant biochar alternatives from the analysis can falsely skew the predicted value of biochar.

Focusing on the value of CO_2 reductions, Gaunt and Lehmann (2008) determined that biochar application to agricultural land would provide greater GHG emission reductions than using biochar for electricity generation. However, the costs of achieving the additional GHG emission through soil application ranged from 9 to \$16 Mg^{-1} CO_2e. This cost was based on the value of lost electricity generation and did not include the value of any agricultural production benefits. The calculated GHG emission reductions with biochar addition did, however, assume a fertilizer reduction of 10%, which would result in a 50% reduction in N_2O emissions, and that the effect of biochar would remain for 10 yr after application.

A combined biochar life cycle and cost-benefit analysis was conducted by Tejerina (2010). This analysis used monetary, energy, and GHG balances (assuming a 10% fertilizer savings and a 5% yield increase with biochar) in the context of a developing country under three different management scenarios: (i) biochar for soil application, (ii) bio-coal production, or (iii) current management (residue left on the field). The results showed corn stover used for biochar production and soil application as providing the greatest net energy and positive revenue potential over bio-coal production or leaving the residue in the field. However, the results for a developed country could be drastically different due to economic differences in the assumed costs for labor, machinery, transportation, and energy.

In each of these above cases, the value of biochar for field application was sensitive to uncertainties associated with the annual value of benefits (or costs) that result from soil application and the duration of these benefits (or costs). Annual benefits may accrue if a biochar application increased yields or reduced production costs. However, our ability to extract a defensible yield improvement prediction from the existing data is hampered by insufficient biochar characterization data coupled to the lack of uniform yield responses (Table 3). Wood-based biochars, which typically increase agronomic yields, are the biochars with the lowest nutrient contents (Table 2). The GHG offset payments associated with avoided emissions due to reduced fertilizer, pesticide, or other inputs use may also be included as annual benefits once the duration of these benefits is known. Long-term, small annual biochar benefits can produce high value to farmers over time. This means that biochar application is economically feasible at the field-scale if it is available to farmers at a cost less than a break-even value. However, the amount farmers are willing to invest in biochar applications will be directly related to the certainty of attaining the degree and persistence of predicted annual benefits. On the other hand, any potential persistent negative cost effects can substantially reduce the value of biochar to the farmer.

The value of GHG offset payments is typically based on permanent GHG offsets, so the soil carbon sequestration value of biochar application would have to be included as a one-time benefit in the above analysis and not as a repeating annual benefit. In addition, the value would be reduced if sequestration was not long-term. Furthermore, the lack of standardization in the accounting of renewable energy system GHG assessments creates systematic biases that can lead to over- or underestimation of the net GHG impacts (Whitman and Lehmann, 2011).

Potential Specialized Markets

The possibility of niche specialized uses of biochar could substantially improve the economic picture for biochar utilization, particularly engineering "designer biochars" for improving a specific soil deficiency (Novak and Busscher, 2011). For example, there have been efforts at impregnating biochar with inorganic fertilizers to serve as a slow-release fertilizer (Khan et al., 2008) and to provide localized crop protection from herbicides, pesticides, and other chemicals (Fink, 1934; Linscott and Hagin, 1967). Biochar could be blended with compost (Rosenfeld, 2001; Rosenfeld and Henry, 2001; Steiner et al., 2010), which could increase biochar's value, particularly if biochar application for bedding plants is beneficial (Kadota and Niimi, 2004).

Biochar might be useful for controlling selected invasive or nuisance species of animals (Mason and Clark, 1994, 1995) and plants (Kulmatiski and Beard, 2006). Biochar may offer the potential as a means for contaminated site cleanup (Cao and Harris, 2010). An example could be reducing soil copper contamination resulting from irrigation with water from spent copper sulfate ($CuSO_4$) dairy hoof bath lagoons (Ippolito et al., 2012) or as a potential remediation tool for acid mine soils (Novak and Busscher, 2011).

Biochar use does not have to be limited to soil application. Biochar could be suitable as a precursor to generate activated carbon, which is commonly used in industrial filtration processes (Azargohar and Dalai, 2006). However, the production of activated charcoal requires extra conditioning steps, which reduces the economic return (Lussier et al., 1994). Several studies have examined biochar and activated biochar use in municipal wastewater treatment (Ng et al., 2002, 2003; Bansode et al., 2004; Lima and Marshall, 2009), in mercury removal from flue gas (Klasson et al., 2010), and in other water filtering systems (van Duck and van de Voorde, 1984). However, for biochar to be used for potable water filtration, the potential for bacterial growth and organic contaminates on the biochar needs to be further investigated (Wallis et al., 1974). Other potential uses of biochar include use as a nutrient recovery agent (Streubel et al., 2010), as an additive for reducing the bioavailability and mobility of toxic trace metals (Beesley and Marmiroli, 2011; Uchimiya et al., 2011), as a contaminant mitigation agent (Beesley et al., 2010), or as a material for iron and steel production (de Beer et al., 1998). Potential markets in laboratory settings exist for biochar, including solid-phase microextraction fibers (Wan et al., 1994), electroanalytical chemistry electrodes (Tavares and Barbeira, 2008), or biochar-based DNA hybridization biosensors (Wang and Kawde, 2001). Spent biochar could be examined for uses in landfill covers for odor and bird control (Mason and Clark, 1994). Furthermore, the potential use of biochar as a sorbent media

for scrubbing CO_2 from fossil fuel stack emissions warrants additional research because this could offer additional carbon sequestration potential for biochar (Mercedes Maroto-Valer et al., 2008). All of the above applications preserve the carbon sequestration potential. In addition to direct applications, the use of biochar as a green product label also offers potential targeted economic advantages (Yoder and Galinato, 2009).

We acknowledge that biochar is expensive as a carbon sequestration agent or as a soil supplement for crop yield improvements. However, the high production cost for biochar could be offset if these specialty or boutique markets are more fully developed. The key is to diversify biochar applications to other sectors, which could result in reduced costs for production.

Future Needs

Just as economics caused the shift from biomass to fossil fuels in the early 1920s, civilization is currently at the cusp where environmental stewardship is returning the pendulum back to biomass as the source for human's energy, chemical, and agronomic needs. Given this return to biomass, there are several factors that should be considered as we move forward:

- There is a fundamental need to recognize that biochar does not refer to one singular product; it refers to the range of potential products produced for carbon sequestration purposes.
- There is a vital need to fully document the production style, biomass conditions, and pyrolysis conditions of the biochar production, but almost equally important is the time since biochar production, postproduction handling, and storage conditions. This is especially evident given the importance of the surface chemistry on overall biochar chemistry and resulting environmental interactions.
- As seen in several aspects of this review, the lack of adequate data has limited the ability to use the historic data in meta-analyses to elucidate driving variables. This is due to the lack of consistency in biochar characterization and documentation, which is partially a result of the incomplete understanding of the mechanisms responsible for agronomic yield improvements. In other words, what properties do we want biochar to possess?
- The studies using traditional kilns and production systems without industrial controls are further suspect because there is no documentation on production parameters (e.g., pyrolysis temperature, oxygen status, etc.) or repeatability of the biochar production style.
- There is also a need to overcome the terminology and definition hurdles and to separate the soil impacts of biochar additions from the biochar itself. This is needed to continue the development of biochar as a stable carbon form that could be considered for carbon sequestration markets. The potential end uses for biochar are virtually limitless, and these specialized market niches potentially could increase the economic value of biochar.
- There is a need for the creation of an international pool of well characterized biochars for systematic research (Lehmann et al., 2011). Furthermore, these biochars need to come from a variety of sources with well documented production systems to achieve a cross-sectional sampling required for such an effort. This effort could be coordinated by the American Society of Agronomy's Biochar Community (https://www.agronomy.org/membership/communities/biochar-agronomic-and-environmental-uses).
- We need to better understand how biochar's production conditions influence its quality and subsequent effects on the soil–plant systems, which would lead to more accurate guidance for biochar amendment management.

A tipping point is being approached where global demand for food will exceed production. Despite the current forecasted economics, the fact that biochar has the potential to improve soil quality and provide benefits to nonproductive and degraded soils justifies continued research efforts into biochar's soil quality impacts. Continued research can optimize biochar production for the overall health and fertility of our soil resources, which represent our most vital asset in the current bioenergy renaissance and are the fundamental foundation for food security.

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