Environmental Benefits of Biochar

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Understanding and improving environmental quality by reducing soil nutrient leaching losses, reducing bioavailability of environmental contaminants, sequestering C, reducing greenhouse gas emissions, and enhancing crop productivity in highly weathered or degraded soils, has been the goal of agroecosystem researchers and producers for years. Biochar, produced by pyrolysis of biomass, may help attain these goals. The desire to advance understanding of the environmental and agronomic implication of biochar utilization led to the organization of the 2010 American Society of Agronomy–Soil Science Society of America Environmental Quality Division session titled "Biochar Effects on the Environment and Agricultural Productivity." This specialized session and sessions from other biochar conferences, such as the 2010 U.S. Biochar Initiative and the Biochar Symposium 2010 are the sources for this special manuscript collection. Individual contributions address improvement of the biochar knowledge base, current information gaps, and future biochar research needs. The prospect of biochar utilization is promising, as biochars may be customized for specific environmental applications.

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The interest in the effects of biochar application on plant growth, soil properties, and environmental contaminants has spurred a significant amount of research in recent years. This introductory paper to the special section on the environmental benefits of biochar provides an overview of a select group of papers presented at three 2010 biochar meetings: the 2010 American Society of Agronomy–Soil Science Society of America (ASA–SSSA) Environmental Quality Division session titled "Biochar Effects on the Environment and Agricultural Productivity" (Long Beach, CA, Oct. 31–Nov. 3; http://scisoc.confex.com/scisoc/2010am/webprogram/Session7428.html); the 2010 U.S. Biochar Initiative Conference (Ames, IA, June 27–30); and the Biochar Symposium 2010 (Bayreuth, Germany, July 8–9; organized by Dr. Bruno Glaser; http://www.limno.uni-bayreuth.de/biochar2010/en/program/bayconf/programm.php?). It also points to areas in which research indicates biochar may be useful and identifies critical knowledge gaps in understanding the nature of biochar and its utilization in environmental settings.

Introduction to the Environmental Impacts of Biochar

The review by Spokas et al. (2012) presents a compelling introduction to the environmental impacts of biochar. The authors provide insight into the pyrolysis process and the resultant biochars based on chosen feedstock; they go on to present a summary of the types of biochars that provide positive yield responses and the economics of biochar creation, transportation, and utilization, following a whole-systems approach. Their examination of 44 previously published biochar research articles showed that approximately half observed yield increases whereas the other half showed no or even negative yield responses. This led Spokas et al. (2012) to conclude that not all biochars are created equal and that biochars should be designed with special characteristics for use in specific environmental or agronomic settings (Novak and Busscher, 2012). Spokas et al. (2012) also conclude that although current economics may not be favorable for large-scale production agriculture utilization of biochar, the potential exists for biochar to provide environmental quality benefits and to improve nonproductive or degraded soils.

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Abbreviations: GHG, greenhouse gas; PAH, polycyclic aromatic hydrocarbon.

Biochar Characterization for Potential Environmental Use

Feedstocks utilized for biochar production (e.g., woody biomass, crop residues, grasses, and manures) influence biochar characteristics, including concentrations of elemental constituents, density, porosity, and hardness (Spokas et al., 2012). Optimizing biochar for a specific application may require selection of a feedstock as well as pyrolysis production technique and conditions to produce biochars with specific characteristics. Several papers in this special section target the relationships between biochar production conditions, biochar characteristics, and potential end-uses of biochar.

Kloss et al. (2012) characterize biochars produced from three feedstocks pyrolyzed at various temperatures. In general, straw-based biochars had greater soluble elemental concentrations than two woody-based biochars, although nutrient concentrations were not high enough to promote the direct use as a soil fertilizer. Increasing pyrolysis temperature increased biochar specific surface area, which may benefit sandy soils by increasing sorption sites or may improve the retention of nonpolar pollutants in soils (Kloss et al., 2012). The authors also show that increasing pyrolysis temperatures affected the polycyclic aromatic hydrocarbon (PAH) content of biochar; PAHs are relatively recalcitrant and potentially toxic aromatic hydrocarbons formed during incomplete combustion. In the study by Kloss et al. (2012), straw-based biochar PAH content increased with increasing pyrolysis temperatures whereas wood-based biochar PAH content tended to decrease with increasing temperature.

Schimmelpfennig and Glaser (2012) used 16 different feedstock materials to create 66 biochars produced from five different pyrolytic processes (traditional charcoal stack, rotary kiln, Pyreg reactor, wood gasifier, and hydrothermal carbonization) to derive a minimum analytical dataset for assessing the potential use of biochar as a soil amendment and for carbon sequestration. On the basis of their results, the authors suggest that biochars containing the following will be effective C sequestration agents when applied to soils: O:C ratio < 0.4, H:C ratio < 0.6 (O:C:H ratios serve as an indicator for the degree of carbonization that influences the stability of biochar in soil environments); black carbon content > 15% C (black C is resistant to degradation). They further suggest other standards, including N_2–Brunauer–Emmett–Teller surface area > 100 m^2 g^{-1} (this may help predict the biochar effect on soil moisture levels) and recommend that biochar PAHs be less than background levels in soils for its utilization as a soil amendment. Rogovska et al. (2012) used aqueous biochar extracts to conduct standard germination tests with measurements of early seedling growth to identify biochars that contain phytotoxic compounds. Using corn (*Zea mays* L.) for the bioassays, they observed a decrease in shoot length and radical length (compared with controls) associated with biochars produced by high temperature gasification and pyrolysis, but not for biochars produced at lower temperatures. The authors hypothesize that the decrease in shoot and radical length was probably due to the presence of identified PAHs, primarily naphthalene, in the biochar extracts of high-temperature biochars.

The results presented by Schimmelpfennig and Glaser (2012) and Rogovska et al. (2012) point to the need for identifying inexpensive and easy biotoxicity tests that can be utilized to identify biochars with negative effects, which was the goal of Busch et al. (2012). The authors utilized one biochar and one hydrochar, a material produced by hydrothermal carbonization of feedstock in an aqueous suspension under moderate pressures and elevated temperatures (Funke and Ziegler, 2010), in four quick (<2 wk) toxicity evaluation tests (cress germination, barley germination and regrowth, lettuce germination, and earthworm avoidance tests). Negative effects were not observed with the biochar, but the hydrochar produced negative responses in all four tests. Although verification will be required on a larger set of biochars, the results are promising because they were based on quick, easy, and relatively inexpensive procedures that could be used universally by producers and other end users.

In addition to some of the basic analytical tests suggested in the above papers, Busch et al. (2012) suggest a minimum characterization dataset for biochars intended for use in environmental settings. Quantification of the effects of feedstock on production will require future dose–response studies coupled with testing of a large range of biochars.

Biochar Effects on Plants

Several papers in this special collection describe plant growth responses to biochar. Lentz and Ippolito (2012) applied a one-time application of hardwood biochar at 22.4 Mg ha^{-1} to an Aridisol, observing no change in corn (*Zea mays* L.) silage yield as compared to a control 1 yr following application; however, they observed a 36% yield decrease, as compared to the controls, during year 2. On the basis of corn silage nutrient concentrations, the suppression in yield was due either to reduced nutrient (N, S, Mn, and Cu) availability or uptake. The response was similar to a priming effect observed in low organic C–containing soils (Zimmerman et al., 2011) where the biochar may have induced a reduction in soil C mineralization, which in turn limited at least soil N and S availability.

Schnell et al. (2012) applied up to 3 Mg ha^{-1} of a sorghum [*Sorghum bicolor* (L.) Moench] biochar to an Alfisol and then grew sorghum for 45 d. No difference in biomass production was noted between the control and biochar applications, likely due to the modest biochar application rates utilized. However, similar to Lentz and Ippolito (2012), the authors also suggest that low nutrient recovery in plants grown in biochar-treated soil could have contributed to a lack of yield response.

Kammann et al. (2012) added peanut (*Arachis hypogaea* L.) hull biochar at 50 Mg ha^{-1} to a German Luvisol and then grew ryegrass (*Lolium perenne* L.). The authors observed a significant increase in biomass yield when compared to controls. The cause of the increase in yield was unknown, but it could have been a function of reduced N loss to denitrification and hence greater N uptake by plants grown in the presence of biochar. Gajić and Koch (2012) also utilized a German Luvisol, growing sugar beet (*Beta vulgaris* L.) in soil amended with 10 Mg ha^{-1} of either sugar beet pulp or beer draff hydrochar. In both studies, plant growth ceased or was drastically reduced immediately after emergence and final crop yield was reduced compared with the control. The authors performed a similar study using a German Cambisol and hydrochar applied at 30 Mg ha^{-1}, observing similar results. This report is consistent with previous reports that suggest that materials containing significant amounts of bioavailable C (such as hydrochar) can decrease plant available N and yields, due to immobilization (e.g., Rillig et al., 2010; Leifeld et al., 2002; Kuzyakov and Bol, 2006).

These results indicate that some biochars and hydrochars can be detrimental to crop yields, whereas others can increase crop yields. What is lacking is a complete mechanistic understanding of how biochars cause yield reductions or increases. Future research should focus on a better understanding of biochar–plant interactions to develop production protocols that produce biochars optimized for specific crop–soil–environmental systems.

Biochar Use and Soil Nutrient Dynamics

Current global interest in biochar has been built largely on research conducted using highly weathered and infertile soils (Lima et al., 2002; Lehmann et al., 2003; Glaser et al., 2004; Steiner et al., 2007; Kimetu et al., 2008; Asai et al., 2009; Novak et al., 2009b; Gaskin et al., 2010; Major et al., 2010). Challenges in these highly weathered systems include prevention of nutrient loss via leaching and retention of nutrients in the root zone. Major et al. (2012) studied nutrient leaching in a Columbian Oxisol following a 20 Mg ha^{-1} biochar application. In general, nutrient leaching with biochar applications relative to unamended soils was greater at 0.6 m than at the 1.2 m in depth. Leaching differences were evident even though no differences in net water flux were present between the two treatments. The authors suggest that biochar may have influenced nutrient retention throughout the root zone. Schomberg et al. (2012) added nine different biochars to a South Carolina Ultisol at a rate equal to ~40 Mg ha^{-1}, incubating and leaching the soils over a 127-d period. The authors found that some biochars reduced N leaching losses, but soil N fractions were not increased with biochar application. Much of the apparent reductions in leaching were due to NH_3 volatilization loss from high ash biochars. Biochar ash content and pH are dependent on feedstock and pyrolysis temperature (Gaskin et al., 2008; Novak et al., 2009b).

Hass et al. (2012) incubated a West Virginia Ultisol mixed with 0, 5, 10, 20, or 40 g kg^{-1} chicken manure biochar (equivalent to 0, 10, 20, 40, and 80 Mg ha^{-1}) for 8 wk. The authors noted a decrease in S, K, and P, and an increase in Cu and Zn availability associated with an increase in biochar production temperature. Hass et al. (2012) noted that biochar application increased leachate PO_4 concentrations compared to controls, suggesting that chicken manure biochar applications may be limited by environmental P concerns. Similarly, Schnell et al. (2012) found that topdressing up to 3 Mg of sorghum-derived biochar per hectare (with no incorporation) on an eastern Texas Alfisol caused significant surface runoff P losses compared with control soils and that incorporating the biochar into soil reduced runoff P losses by 78%.

Research on biochar effects in different soil types have extended to less-weathered soils in both temperate and arid climates. In a California Alfisol, Sarkhot et al. (2012) added the equivalent of 20 Mg ha^{-1} biochar as is or enriched in nutrients from dairy manure effluent. Nitrogen leaching losses with both biochars were similar to unamended soil, suggesting that biochar either acts as a slow-release source of N or that it caused N immobilization. Brewer et al. (2012) amended a sandy Mollisol with 0.5% (w/w; ~10 Mg ha^{-1}) biochar made under various pyrolysis conditions, generally observing an increase in soil extractable P, K, Mn, and Fe compared with unamended soil. The authors also noted little change in soil NO_3–N concentrations with biochar amendment compared with control soil, suggesting that even incompletely pyrolyzed biochars did not cause significant N immobilization. Lentz and Ippolito (2012) studied hardwood-derived biochar application (22.4 Mg ha^{-1}) to an Aridisol, noting an increase in soil-extractable Mn over a 2-yr period. More important, results showed that biochar applied with manure (42 Mg ha^{-1}) reduced manure organic C losses. In this calcareous system, the authors did not observe a change in soil pH, cation, or P availability as is typically noted in more acidic, weathered soils. Ippolito et al. (2012a) added switchgrass biochar pyrolyzed at two different temperatures (250 and 500°C) to two Aridisols (2% by weight; equivalent to 40 Mg ha^{-1}). Similar to the observations of Hass et al. (2012) in weathered soil systems, the authors noted a two- to threefold decrease in leachate P concentrations with the lower- versus higher-temperature biochar. This was probably due to retention of orthophosphate by surface functional groups, Fe and Al (hydr)oxide sorption, and Ca and Mg phosphate precipitation (Novak et al., 2009a). Compared with biochars produced at high temperatures, application of biochars produced at low pyrolysis temperatures showed less Ca, Mg, and NO_3–N leaching, likely due to immobilization because biochars produced at 250°C contain substantial amounts of bioavailable C similar to the hydrochar reported by Gajić and Koch (2012). Gajić and Koch (2012) suggested that using low-temperature biochars and hydrochars containing lower C-to-N ratios, and microbially degradable C, should help reduce immobilization. Nutrient immobilization by higher-temperature biochars is generally not a problem. For example, Kameyama et al. (2012) studied NO_3–N retention by calcareous Japanese soils amended with biochar produced from bagasse at 400 to 800°C. The authors show that NO_3–N was weakly sorbed to biochar but sorption increased with greater temperatures due to the formation of base functional groups, and that increased retention of nutrients and water in biochar micropores decreased NO_3 leaching and provided a greater opportunity for crops to utilize available NO_3–N.

The above findings show that further research needs to fully elucidate the effects of interactions between biochar characteristics, climatic conditions, and soil properties on nutrient leaching, retention, and immobilization. Almost no information is currently available on how biochar characteristics affect microbial-mediated nutrient cycling and soil microbial communities. However, the use of tracer techniques to differentiate nutrient pathways and pools hold promise for addressing these questions (Major et al., 2012). Single-year-yield responses to biochar applications may not be large enough to justify the expense; however, if a one-time application results in a long-term improvement in nutrient and water-use efficiency, then the expense could be amortized over many years (Spokas et al., 2012). Longer-term field research focusing on nutrient and water use efficiency is needed to quantify the legacy of biochar applications and assess their true value.

Biochar Use for Sequestering Inorganic and Organic Contaminants

Charcoal has long been used to remove impurities from aqueous systems (Ippolito et al., 2012b). With this in mind, research presented in this special section has explored the potential use of biochar as a media for heavy metal, phosphorus, and antibiotic sorption. Uchimiya et al. (2012) point out that for stabilization to occur, metals need to interact with biochar via electrostatic interactions, ionic exchange, sorption via proton exchange, or specific ligand binding. The authors studied sorption of Pb, Cu, Ni, and Cd to a South Carolina Ultisol amended with five different manure biochars (dairy, paved feedlot, swine solids, poultry litter, and turkey litter) pyrolyzed at either 350 or 700°C. Biochar applications enhanced heavy metal retention as measured by heavy metal concentrations in equilibrium aqueous extracts, which were substantially lower (for Pb, Cu, and Cd) or slightly lower (for Ni) than metal concentrations in aqueous extract of the control soils (i.e., no biochar addition). Although Pb sorption by biochar has previously been attributed to phosphate and carbonate phases (Cao et al., 2009), these relationships could not explain Pb sorption in the current study. Copper sorption appeared to be positively correlated with pyrolysis temperature, likely a function of both increased pH and electron donor-acceptor complexes with condensed aromatic phases. Sorption of Cd showed no clear effect of pyrolysis temperature, possibly due to feedstock related variations in the density of nitrogen-containing surface functional groups. Nickel, the metal least impacted by biochar applications, may have been influenced by competition from other more strongly retained metals such as Cu.

Ippolito et al. (2012b) observed that biochar could adsorb up to 42,000 mg Cu kg^{-1} biochar from aqueous solutions depending on initial solution pH. Similar to the findings of Uchimiya et al. (2012), the authors showed (via X-ray absorption fine structure spectroscopy) that Cu was sorbed as an organic phase at lower pH values. At higher pH values, however, Cu was retained by binding to organic ligands on the biochar surface and by precipitation as separate carbonate and oxide mineral phases. Buss et al. (2012) grew quinoa (*Chenopodium quinoa* Willd.) in sandy soil in the presence of 0, 50, or 200 mg Cu kg^{-1} and 0, 2, or 4% forest green waste biochar (material left following forest harvesting) by weight. Without biochar, plants showed severe stress symptoms at 50 mg Cu kg^{-1} and were completely dead at 200 mg Cu kg^{-1}. Increasing the amount of biochar reduced plant stress and Cu uptake by the plants. Plant biomass in the presence of 200 mg Cu kg^{-1} and 4% (~80 Mg ha^{-1}) biochar was nearly as great as that of the controls. The authors reported that the improvement in plant growth resulted from reduced Cu toxicity, which was probably due to Cu binding on biochar negatively charged carboxyl groups, an increase in soil pH, or an increase in volumetric soil water content essentially diluting the soil solution Cu concentrations.

Streubel et al. (2012) utilized biochar produced by pyrolysis of anaerobic digester dairy fiber to remove P from dairy lagoon effluent. Using a closed cycle system, they continuously cycled effluent across pelletized biochar, with the biochar removing 380 mg P L^{-1} (initial dairy lagoon effluent P concentration was approximately 550 mg L^{-1}). This was equal to approximately a 70% reduction in waste stream P content. Furthermore, the P retained on the biochar was a combination of adsorbed orthophosphate and Ca-PO_4 precipitates, indicating that biochar effluent filters could be useful for recovering P in plant-available forms.

Choppala et al. (2012) studied the effect of biochar on reduction of Cr(VI), the bioavailable and toxic form, to Cr(III), the strongly bound and nontoxic form. The authors compared results for the chicken manure biochar with acid-activated black carbon from a weedy species (*Solanum elaeagnifolium* Cav.). Two different soils received 0 or 50 mg kg^{-1} of biochar or activated black carbon, along with 0 or 500 mg kg^{-1} Cr(VI). Soils were incubated at field capacity for up to 14 d. Results showed that the activated black carbon reduced all of the Cr(VI) to Cr(III) within 6 to 10 d, whereas the chicken manure biochar reduced between 198 and 219 mg kg^{-1} over the 14-d incubation; the estimated half-life for Cr(VI) reduction by biochar was between 10.7 and 11.4 d. Although biochar did not fully reduce Cr(VI) to Cr(III) within the timeframe of the study, results appear promising that both biochar and acid activated black carbons could play a role in reducing Cr(VI) in contaminated soils.

Animal manures may contain substantial quantities of unmetabolized antibiotics (up to 90% of the parent compounds may be excreted; Sarmah et al., 2006), and the use of antibiotic laden manures as soil amendments could lead to the development of antibiotic resistant soil bacteria. Jeong et al. (2012) showed that both hardwood and softwood biochar applications significantly reduced tylosin (a common veterinary antibiotic) movement through soils. The amount of tylosin leaching from forest and cornfield soils decreased as the biochar amendments increased from 0 to 10% by weight. The quantity of tylosin desorbed from the soils was also reduced with increasing rate of biochar application, possibly due to irreversible surface binding or entrapment of tylosin within biochar particles (Spokas et al., 2009).

The above results suggest that biochar may be of value for sequestering bioavailable metals and antibiotics in contaminated soils, as well as for capturing and recycling nutrients in effluent streams. More research targeting a broad spectrum of soils affected by environmental degradation is required. Specifically designed biochars could sorb less easily stabilized metals such as Cd and Ni or sequester mobile organic phases. Contaminant sorption mechanisms and kinetics need to be identified, loading capacities of biochars need to be quantified, and the ultimate fate of contaminants in biochar-amended soils needs to be documented before large-scale biochar field applications in contaminated soil begin. Cost and efficacy comparisons for biochars relative to other contaminant mitigation technologies would also be helpful.

Biochar Affects Greenhouse Gas Emissions

Biochars are effective agents for sequestering C in soils. Although hydrochars and low-temperature biochars contain some bioavailable C, it is generally more stable in soils than C in the original biomass; the C in moderate- and high-temperature biochars is overwhelmingly stabilized against microbial decomposition and hence will persist for hundreds if not thousands of years in soils. However, the net greenhouse gas (GHG) impact due to biochar applications to soil is also influenced by changes in net primary crop productivity, increases in the efficiency of residue mineralization or humification, soil organic matter cycling, and emissions of CH_4 and N_2O. Furthermore, the overall impact of biochar amendments must also include GHG emissions resulting from biochar production, transport, and soil application itself.

Yoo and Kang (2012) added biochar (~20 Mg ha^{-1}) made from either swine manure (pyrolyzed at 600–800°C) or barley stover (pyrolyzed at 320°C) to two different soils. They then measured CO_2, CH_4, and N_2O emissions over a 36-d period.

Differences were evident depending on type of biochar as well as soil condition. Nitrogen-limited soil released less CO_2 and CH_4 when biochar containing elevated available N was added (i.e., swine manure biochar); however, the reduced CO_2 and CH_4 emissions were likely offset by increased N_2O emissions. Greenhouse gas emissions were not increased when barley stover biochar was added to either soil; thus, Yoo and Kang (2012) suggest that this biochar may be a more appropriate amendment material. In several laboratory incubation studies, Kammann et al. (2012) investigated the effect of peanut hull biochars (produced between 500 and 800°C) and hydrochar (200°C), in the absence or presence of organic amendments or fertilizer, on soil GHG emissions. Except for the case combining long-term elevated soil water contents and inorganic N fertilizer amendments, all other biochar amendment scenarios exhibited significant reductions in N_2O emissions. Biochar applications caused equal or less release of CO_2, N_2O, and CH_4 compared with control soils. Hydrochar applications caused larger releases of CO_2, N_2O, and CH_4 compared with the biochars and thus may not be a suitable material if soil C sequestration is a goal.

Augustenborg et al. (2012) applied either peanut hull (500°C) or *Miscanthus* (550°C) biochar to low- or high-organic matter soils, with or without endogeic (i.e., soil feeding) earthworms, and measured N_2O and CO_2 emissions. The authors found that biochar additions significantly reduced both CO_2 and N_2O emissions in the absence of earthworms compared with no-biochar controls. The endogeic earthworms increased N_2O emissions coming from the controls by as much as 12.6-fold; however, both biochars drastically reduced N_2O emissions in the presence of earthworms.

Yoo and Kang (2012) and Kammann et al. (2012) both found that biochar created at higher pyrolysis temperatures caused a greater reduction in cumulative CO_2 release compared with biochars produced at lower temperatures. Over a 365-d period, Qayyum et al. (2012) measured cumulative CO_2 released from three soils amended with either nothing, wheat straw, hydrochar (200°C), low-temperature biochar (sewage sludge pyrolyzed at 400°C), or charcoal (550°C). Cumulative CO_2 released generally followed the order: wheat straw > hydrochar > low temperature biochar > charcoal = control. The authors concluded that the biochar utilized for an application should match the aim of the use, with high-temperature biochars being good for soil C sequestration and low-temperature biochars perhaps better for enhancing soil fertility. These results follow that of Brewer et al.'s (2012) study, which shows that biochar-amended soil CO_2 losses are inversely related to the extent of pyrolysis.

The above studies illustrate that biochar type, pyrolysis conditions, and environmental factors all play a role in GHG emissions from biochar-amended soils. Future studies are necessary to understand the mechanisms and interactions among plants, soils, microbes, and climate, as well as their impact on GHG emissions. Short-term laboratory incubations, as performed in the studies above, may be useful for screening biochars and could lead to guidelines for longer-term use; however, long-term field research will be necessary to quantify the effects of the interactions noted above on net GHG emissions from agricultural soils.

Summary

The papers within this special section greatly advance our understanding of interactions between biochar and the environment but also point out gaps in our current knowledge and understanding. Several authors report that the type of feedstock (e.g., hardwood, softwood, crop residue, manure, biosolids), pyrolytic process (e.g., traditional charcoal kiln, rotary kiln, industrial fast pyrolysis, Pyreg reactor, wood gasifier, hydrothermal carbonization), and pyrolytic conditions (e.g., temperature, pressure, steam-activation) influence the properties of the resulting biochar and that biochar properties influence the environmental and agronomic impacts of biochar applications. Ideally, we would design specific biochars with properties optimized for a specific environmental or agronomic application. Although this idea may be attainable for a few high-value applications, economics will limit the available options for most applications. In this latter realm, the principal of "first, do no harm" must be foundational, and along with economic optimization, the balance between cost and benefits will ultimately control the availability and options for using biochar in environmental and agronomic applications.

Because biochar plays a role in plant productivity, the authors in this special section have provided evidence that biochar use can be detrimental, neutral, or positive in terms of plant growth. The negative or neutral yield responses due to biochar applications appear to be the result of low application rates, or N immobilization due to the use of low-temperature biochar and hydrochars containing appreciable bioavailable C. One paper observed an increase in plant yield due to biochar-reducing denitrification N losses, leading to greater N availability and plant uptake. Not all soils will benefit from biochar applications; putting biochar on degraded or sandy soils where productivity is limited by low nutrient or water holding capacity is likely to be far more beneficial than adding biochar on highly productive soils. Placing biochars where they will intercept effluent runoff or groundwater laden with nutrients or other contaminants will likely yield far greater positive environmental impacts than spreading biochar uniformly across fields. Future studies need to target specific biochars to these types of specific environmental or agronomic applications.

The majority of research presented herein focuses on short-term impacts of biochar applications on soil nutrient, metal, gaseous, and organic contaminant dynamics. In the future, long-term field research focusing on nutrient use efficiency, water use efficiency, net C sequestration, net GHG emissions, and changes in soil quality, plant, and microbial community dynamics is needed. Once biochar is applied and incorporated into soil, it will leave a lasting legacy. It is thus imperative that we understand the long-term impacts.

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