



Bioenergy Production Systems and Biochar Application in Forests: Potential for Renewable Energy, Soil Enhancement, and Carbon Sequestration

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Abstract: Bioenergy production from forest biomass offers a unique solution to reduce wildfire hazard fuel while producing a useful source of renewable energy. However, biomass removals raise concerns about reducing soil carbon and altering forest site productivity. Biochar additions have been suggested as a way to mitigate soil carbon loss and cycle nutrients back into forestry sites; yet, little is known about the effects of intentional biochar amendments to temperate forest soil in conjunction with biomass removals for bioenergy production. In this review, we evaluate the potential for mobile bioenergy systems and the environmental implications of biochar application in forests. Using forest biomass that accumulates annually during forest harvest operations, bioenergy can be produced on-site and the biochar that is generated can be redistributed to return nutrients and help improve water holding capacity of the site. Little is known about the short- and long-term impacts of biochar application in forest ecosystems. Some sites may benefit from biochar application, while others show no or negative responses. Field studies on soil and vegetation responses combined with laboratory studies will elucidate the best sites for biochar application and sustainable bioenergy production.

The Need for Biomass Removals

Removing residual woody biomass from forest management activities is important for hazardous fuel reduction and forest

health improvement. Improper fire management during the past century has resulted in overstocked forests or excess coarse woody debris on the soil surface (Kauffman 1990). An estimated 73 million acres of National Forest land in the western United States have been identified as having unnatural or excessive amounts of woody biomass (USDA Forest Service 2003). Residues associated with timber harvesting are also significant, which further results in increased susceptibility to catastrophic fire. As a consequence of increasing wildfire occurrence and overstocked stands that alter forest health, land managers have begun to thin trees and remove residues. Removal operations are often mandated by law for both public and private land owners, but financial support and incentive is limited with these operations (Healthy Forests Initiative 2003).

In the western United States, the cost of biomass removal often exceeds its value, despite increasing interest in forest biomass utilization. Burgeoning interest in using woody biomass for heat or bioenergy is a result of rising fuel costs, greenhouse gas emissions from fossil fuels, and the threat of stand-replacing wildfires; however, the collection and transportation of woody debris and harvesting waste from forests are among many economic impediments to woody biomass utilization.

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There is little to no market for residual biomass, deeming it the lowest value material removed from the forest (Evans 2008). Consequently, it is rarely a source of income for land owners. Costs of low-grade wood such as forestry residues range from \$0.10 to \$40 per ton for chips. The median cost for removal projects that did not result in profits was \$625 per acre, but it reached up to \$1000 per acre in the western United States for mechanical clearing alone (Rummer and others 2003). In 2005, the median cost of bringing biomass to the roadside was \$680 per acre for mild slopes (USDA Forest Service 2005) not including costs for haul distances. The U.S. Department of Agriculture, Forest Service (2005) determined the western forest break-even hauling distance to be a maximum of 86 miles, assuming a price of \$30 per dry ton delivered to the mill for chips and chip transport costs of \$0.35 per dry-ton-mile, excluding treatment costs. Biomass removal costs are highly variable depending on stand conditions, locations, and markets (Lynch and Mackes 2003), making it difficult to estimate standard costs for these operations. Minimizing costs of collection and delivery of biomass to end-users is essential to effectively and economically use this resource.

An important, emerging market for woody biomass is energy production. Woody biomass is a local, renewable resource that can be used for transportation fuel, heat, and power. Additionally, if used as a sustainable substitute for fossil fuels, it can reduce greenhouse gas emissions. Utilizing residues from logging, pre-commercial thinning, and hazardous fuel reductions for energy production will help meet U.S. energy independence goals while promoting forest stand health and reducing wildfire risks. It is estimated that in the 15 western United States, more than 28 million acres of forestlands could benefit from hazardous fuel reduction treatments, yielding approximately 345 million oven-dry tons from accessible areas (Rummer and others 2003). Yet, due to the low value of biomass, limited accessibility, and varying biomass markets, the majority of residues are left to decay at the site or are incinerated in slash piles, which is a waste of potential energy. Piling and burning slash redistributes nutrients on the site, or concentrates them to localized areas, which may lead to lower average site productivity (Binkley 1986). Further, slash burning releases pollutants, including greenhouse gasses, into the atmosphere and can result in lost site nutrients. Volatile elements such as carbon (C), nitrogen (N), and sulfur (S) are readily released as gases, and some C and N is lost from the ecosystem (Hosking 1938; Knight 1966; Tiedemann 1987; Caldwell and others 2002). Phosphorus (P) can also be lost, but in lower quantities than N and S. These nutrients are frequently limiting in forest environments (Kishchuk and Brockley 2002; Fox and others 2007); therefore, it is important to retain onsite nutrient stores

instead of causing losses from volatilization. Consequently, slash burning is an unwise method for biomass removal.

Biomass removal projects raise concern over ecological impacts. Biomass typically consists of mostly fine-diameter material with high nutrient content. The concerns of consequential ecological impacts with removals are due, in part, to the high nutrient content in tree tops and limbs and to the lack of research to evaluate site-specific, short- and long-term ecological effects of removals. While the site impacts are thought to be low, there is considerable concern that an established bioenergy market would degrade site nutrient stocks over time, and over exploitation of this resource could be a negative consequence of biomass harvesting for energy production (Kimmins 1997). The dead wood left behind from harvest residues would otherwise decay and slowly recycle nutrients back to the soil and forest (Johnson and Curtis 2001; Mahendrappa and others 2006). It is understood that bole-only removal during timber harvesting has little impact on the growth of succeeding forest stands; however, whole-tree removals on nutrient-poor sites have had negative impacts (Kimmins 2004). Further, it is understood that disturbing or removing the surface organic horizons and displacing soil may have significant negative impacts on the subsequent stand rotations (Fleming and others 2006). There exists little to no field research on the impacts of removing small-diameter biomass material (e.g., twigs, small branches, and needles that contain high concentrations of nutrients), making long-term impacts associated with these removals difficult to infer. Potential removal consequences will likely depend on the initial site quality and soil properties, the frequency and intensity of harvests, and the ability of the site to replace nutrients between removals (Kimmins 2004). Although forest ecosystems are resilient and can maintain large stocks of nutrients, increasing the frequency of biomass removal may exceed the natural capacity for nutrient replenishment between removals, emphasizing the need to understand the implications of biomass removal.

Mobile Fast-Pyrolysis

A sustained bioenergy production system might include removing the energy but not the nutrients or returning the nutrients after energy is extracted from the biomass. Innovative technology allows for in-woods conversion of biomass to higher value energy products through pyrolysis, with an opportunity to retain the nutrient status at removal sites. Mobile fast-pyrolysis bioenergy production systems (Badger and Fransham 2006) may be one approach to profitable and sustainable biomass utilization. These units can be located at or near biomass removal locations and are capable of converting bulky,

low-value biomass into an easily stored and transportable fuel (bio-oil) that can be effectively used for heat, power, and chemical production (Garcia-Perez and others 2007). Additionally, these units produce a charcoal byproduct (biochar) that has market value of its own, but might best be used by returning it to the site of energy extraction as a soil amendment and as a means of soil C sequestration. Such an approach has been implemented in agricultural systems (Lehmann and others 2006; Laird 2008) but is particularly attractive in forest ecosystems where the biochar can be both produced and immediately returned to the site of energy extraction.

A mobile pyrolysis unit could provide an economical hazardous fuels reduction system by producing alternative sources of energy that can be sold to offset biomass removal costs. Mobile pyrolysis units are designed to convert biomass into bio-oil, biochar, and syngas through thermal decomposition of organic material under anaerobic conditions (Boucher and others 2000; Bridgewater 2004). Pyrolysis involves rapidly heating the biomass (500 °C/sec) to moderate reaction temperatures (400 to 600 °C) followed by rapid cooling with short vapor residence times (1 to 2 s). Biomass feedstock is dried to less than 10% water in order to minimize the water in the resulting liquid product (Bridgewater 2004; Mohan and others 2006). Pyrolysis produces 60 to 75% w/w bio-oil, 15 to 25% w/w solid biochar, and 10 to 20% w/w non-condensable gases, but exact proportions are largely dependent on the feedstock and process temperatures (Mohan and others 2006). No waste, other than flue gas and ash, is generated in the conversion process as the bio-oil and biochar can each be used as a fuel and the synthesis gas can be recycled back into the pyrolysis process, yielding an energy output and making the process sustainable (Bridgewater 2004; Mohan and others 2006). Such an energy production system would help maintain or enhance site productivity and mitigate nutrient depletion through the application of biochar.

Biochar Soil Application

Biochar, a byproduct of the pyrolysis process, is biomass-derived black C. It is analogous to charcoal manufactured through traditional or modern pyrolysis methods and to black C found naturally in fire-prone ecosystems. Biochar has been used as a soil amendment to improve soil nutrient status, C storage, and/or filtration of percolating soil water (Lehmann and Joseph 2009), primarily in agricultural systems. Biochar from pyrolysis and charcoal produced through natural burning share key characteristics, including long residence time in soils and a soil conditioning effect (Glaser and others 2002). Biochar

has an inherent energy value that can be used to maximize the energy output of pyrolysis. However, research has shown that applying biochar to soil may be more desirable as it can increase soil organic carbon (SOC) and improve the supply of nutrients to plants, therefore enhancing plant growth and soil physical, chemical, and biological properties (Glaser and others 2002; Lehmann and others 2002; Rondon and others 2007). Regardless of its commercial market value, biochar presents an opportunity to return site nutrients lost from biomass removal projects, which may overshadow other potential uses.

Biochar Composition

Biochar is produced from biomass and is predominantly composed of recalcitrant organic C with contents of plant micro- and macro-nutrients retained from the original feedstock. We know from research on wildfire occurrence and the development of Anthrosols (e.g., terra preta soils) in the Amazon that charcoal can remain in the soil for hundreds to thousands of years (Agee 1996; Lehmann and Rondon 2006). Consequently, biochar can rapidly increase the recalcitrant soil C fraction of soil. The C in biochar is held in aromatic form, which is resistant to decomposition when added as a soil amendment (Amonette and Joseph 2009), making it a C sequestration tool. However, composition varies by feedstock type and conditions of pyrolysis (Downie and others 2009). Actual C contents can range between 172g kg⁻¹ and 905g kg⁻¹. Nitrogen content ranges from 1.8 kg⁻¹ to 56.4g kg⁻¹, total P from 2.7g kg⁻¹ to 480g kg⁻¹ and total potassium (K) from 1.0g kg⁻¹ to 58g kg⁻¹ (Lehmann and others 2003; Lima and Marshall 2005; Chan and others 2007). Biochar also contains varying concentrations of other elements such as oxygen (O), hydrogen (H), S, base cations, and heavy metals (Goldberg 1985; Preston and Schmidt 2006). Freshly produced biochar consists of a crystalline phase with graphene layers and an amorphous phase of aromatic structures (Lehmann and others 2005; Cohen-Ofri and others 2007). The outer surfaces contain various O and H functional groups, and the graphene sheets may contain O groups and free radicals (Bourke and others 2007). Additionally, biochar has been produced with a range of pH values between 4 and 12, but the final pH is dependent upon the original feedstock and operating conditions (Lehmann 2007). Generally, low pyrolysis temperatures (<400 °C) yield acidic biochar, while increasing pyrolysis temperatures produce alkaline biochar. Once incorporated to the soil, surface oxidation occurs due to reactions of water, O₂, and various soil agents (Cheng and others 2006; Lehmann 2007). The cation exchange capacity (CEC) of fresh biochar is typically very low but increases with time as the biochar ages in

the presence of O₂ and water (Cheng and others 2006, 2008; Liang and others 2006).

There are increasing concerns that contaminants are being retained in biochar and may leach into the soil once added as an amendment; however, contaminant type and quantity are dependent on the origin of the pyrolysis feedstock and the conversion process. Biochar may contain contaminants such as heavy metals and organic compounds, but these are commonly associated with sewage sludge and treated wood feedstocks (Lievens and others 2009), and would likely not be an issue if biochar is produced from forest biomass. Contaminants contained in feedstocks could undergo changes during the pyrolysis process and be destroyed or transformed into benign compounds, while others could be retained in the biochar and be potentially detrimental if added to the soil. In addition, some contaminants (e.g., polycyclic aromatic hydrocarbons) can be formed during pyrolysis. Polycyclic aromatic hydrocarbons (PAH) can be formed from any carbonaceous feedstock, but concentrations are feedstock dependent (Zhurinsk and others 2005). Thus, prior to large-scale application, it is important to understand the chemical composition of the initial feedstock and biochar to avoid potential negative environmental consequences.

Impacts of Biochar on Soil

Biochar offers substantial potential for soil improvement because of its unique physical, chemical, and biological properties and their interactions with soil and plant communities. If used as a soil amendment, biochar could mitigate the possible negative impacts of forest biomass removal operations. However, uncertainties surround the potential short- and long-term effects of intentional biochar application in many regions and ecosystems (namely temperate forests) as most evidence comes from agricultural systems. While additions have largely been neutral or positive (reviewed by Sohi and others 2010), negative impacts are possible, demonstrating the need for a comprehensive understanding of biochar's origin, production, and functional properties.

The physical properties of biochar offer several soil benefits. Its highly porous nature is a result of the cell wall structure of the biomass feedstock. A wide range of pore sizes within the biochar results in low bulk density and high surface area material. Biochar incorporation can alter soil physical properties such as structure, pore size distribution, and density, and has implications for soil aeration, water holding capacity, plant growth, and soil workability (Downie and others 2009). Evidence suggests that biochar application into soil may increase the overall net soil surface area (Chan and others 2007) and, consequently, may improve soil water and nutrient retention (Downie and

others 2009) and soil aeration, particularly in fine-textured soils (Kolb and others 2007). Biochar has a bulk density much lower than that of mineral soils (-0.3 Mg m^{-3} for biochar compared to typical soil bulk density of 1.3 Mg m^{-3}); therefore, application can reduce the overall total bulk density of the soil, which is generally desirable for most plant growth (Brady and Weil 2004).

Increased surface area and porosity and lower bulk density in mineral soil treated with biochar can alter water retention and aggregation and can decrease soil erosion (Piccolo and Mbagwu 1990; Piccolo and others 1996; Mbagwu and Piccolo 1997). Soil water retention is determined by the distribution and connectivity of pores in the soil matrix, which are largely affected by soil texture, aggregation, and organic matter content (Brady and Weil 2004). Biochar has a higher surface area and greater porosity relative to other types of soil organic matter and can therefore improve soil texture and aggregation, which improves water retention. Biochar's initial physical properties occur at a range of scales and affect the proportion of water that can be retained. Kishimoto and Sugiura (1985) estimated the inner surface area of biochar formed between 400 and 1000 °C to range from 200 to 400 m²g⁻¹. Van Zwieten and others (2009) reported the surface area of biochar that was derived from papermill waste with slow pyrolysis to be 115 m²g⁻¹. These properties are expected to change over time with physical weathering but have not been explicitly examined, resulting in uncertainties associated with the longevity of these beneficial physical changes in soil.

Soil moisture retention is an indirect result of altered soil aggregation and structure after biochar application (Brodowski and others 2006). Biochar can affect soil aggregation through interactions with soil organic matter, minerals, and microorganisms; however, the surface charge characteristics and their development over time determine the long-term effect on soil aggregation. Glaser and others (2002) reported that Anthrosoles that were enriched with charcoal had surface areas three times higher than those of surrounding Oxisols and had an increased field capacity of 18%. Tryon (1948) studied the effect of charcoal on the percentage of available moisture in soils of different textures and found different response among soils. In sandy soil, available moisture increased by 18% after adding 45% biochar by volume, while no changes were observed in loamy soil, and soil available moisture decreased in the clayey soil. The high surface area of biochar can lead to increased water retention, although the effect seems to depend on the initial texture of the soil. Improved water holding capacity with biochar addition is most commonly observed in coarse-textured or sandy soils (Glaser and others 2002; Gaskin and others 2007). The impact of biochar additions on moisture content may be due to

increased surface area relative to that found in coarse-textured soils (Glaser and others 2002). Therefore, improvements in soil water retention by biochar additions may only be expected in coarse-textured soils or soils with large amounts of macropores. Additionally, a large amount of biochar may need to be applied to the soil before it increases water retention.

Biochar has the potential to increase nutrient availability for plants (Lehmann and others 2002). Nutrient availability can be affected by increasing cation exchange capacity, altering soil pH, or directly adding nutrients from biochar. One potential mechanism for enhanced nutrient retention and supply following biochar amendment is increasing CEC by up to 50% as compared to unamended soils (Tryon 1948; Mbagwu and Piccolo 1997; Lehmann and others 2002; Liang 2006). Biochar has a greater ability to adsorb and retain cations in an exchangeable form than other forms of soil organic matter due to its greater surface area and negative surface charge (Liang and others 2006). Studies have shown significant increases in the availability of all major cations (Glaser and others 2002; Lehmann and others 2002; Topoliantz and others 2005). Tryon (1948) found increasing amounts of exchangeable bases in sandy and loamy soils after adding 45% hardwood and conifer charcoals. Additionally, freshly produced biochar is reported to have an anion exchange capacity. Cheng and others (2008) found biochar to exhibit an anion exchange capacity at pH 3.5, which decreased to zero over time as the biochar aged in soil.

Biochar has a higher sorption affinity for a range of organic and inorganic compounds and a higher nutrient retention ability compared to other forms of soil organic matter (Bucheli and Gustafsson 2000, 2003; Allen-King and others 2002; Kleineidam and others 2002; Nguyen and others 2008). Once biochar is added to the soil, abiotic and biotic surface oxidation of biochar results in increased surface carboxyl groups, a greater negative charge, and subsequently, an increasing ability to sorb cations (Cheng and others 2006, 2008). Treated soil also exhibits an ability to sorb polar compounds, including many environmental contaminants (Yu and others 2006). Cation exchange capacity of biochar is highly variable depending upon the pyrolysis conditions under which it is produced. Cation exchange capacity is lower at low pyrolysis temperatures and significantly increases when produced at higher temperatures (Lehmann 2007). Freshly produced biochars have little ability to retain cations, resulting in minimal CEC (Cheng and others 2006, 2008; Lehmann 2007), but this increases with time in soil with surface oxidation (Cheng and others 2006). These results support the findings of high CEC observed in Amazonian Anthrosols (Liang and others 2006).

Biochar can serve as a liming agent, resulting in increased pH and nutrient availability for a number of different soil types

(Glaser and others 2002; Lehmann and Rondon 2006). The carbonate concentration of biochar facilitates liming in soils and can raise pH of neutral or acidic soil (Van Zwiiten and others 2007). Mbagwu and Piccolo (1997) reported increases in pH of various soils and textures by up to 1.2 pH units from pH 5.4 to 6.6. Tryon (1948) reported a greater increase in pH in sandy and loamy soils than in clayey soils. The pH of various soils increased more after applications of hardwood charcoals (pH 6.15) than of conifer charcoals (pH 5.15), probably because of different ash contents of 6.38% and 1.48%, respectively (Glaser and others 2002).

Biochar feedstock and pyrolysis conditions largely determine the resulting carbonate concentrations, making some biochar a better liming agent than others. Concentrations of carbonates can vary from 0.5 to 33% (Chan and others 2007) depending on starting conditions. Hardwood charcoals are reported to have substantial carbonate concentrations and prove more effective in reducing soil acidity and, therefore, have a larger influence on soil fertility (Steiner and others 2007). The liming of acidic soils decreases aluminum saturation, while increasing cation exchange capacity and base saturation (Cochrane and Sanchez 1980; Mbagwu and Piccolo 1997; Fisher and Binkely 2000). Additionally, nutrient availability may actually increase beyond the amount anticipated by cation exchange sites as a result of the soluble salts available in the biochar.

The liming effect associated with biochar may not be ideal for all soil types and plant communities. Increased soil pHs associated with biochar additions have caused micronutrient deficiencies in agricultural crops (Kishimoto and Sugiura 1985) and forest vegetation (Mikan and Abrams 1995); thus, it is important to acknowledge the presence of calcifuge vegetation prior to application. In addition, many forest plants, fungi, and bacteria thrive in lower pH soils (Meurisse 1976, 1985); therefore, altering forest soil pH through the addition of biochar may result in unfavorable shifts in above- and belowground flora. Understanding interactions among biochar production and application conditions and soil texture, organic matter, and pH will be key in determining long-term effects of biochar application on forest soils.

In the short term, biochar may supply a source of plant-available nutrients once applied to the soil (Gaskin and others 2008; Sohi and others 2010). A small fraction of nutrients in the feedstock, apart from N, are retained in biochar in a potentially extractable form. It is uncertain whether these soluble nutrients are released instantaneously once added to the soil environment or if they are released over time (Sohi and others 2010); this will likely depend on the starting soil physical properties. The rapid introduction of readily available nutrients and small amounts of labile C retained in biochar could

promote mineralization of soil organic matter (Wardle and others 2008a), especially in nutrient-limited environments. Additionally, alkaline biochar may increase the pH of acidic soils, subsequently stimulating microbial activity thereby further promoting mineralization or decomposition of existing soil organic matter.

Biochar properties may enhance soil microbial communities and create microenvironments that encourage microbial colonization. Biochar's pores, high internal surface area, and increased ability to adsorb organic matter provide a suitable habitat to support soil microbiota, which catalyze processes that reduce N loss and increase nutrient availability for plants (Winsley 2007). It has been suggested that pores serve as a refuge to microbes by protecting them from predation and desiccation, while the organic matter adsorbed to biochar provides C energy and mineral nutrient requirements (Saito and Muramoto 2002; Warnock and others 2007). In temperate ecosystems with wildfire-produced charcoal, N mineralization and nitrification are enhanced (Berglund and others 2004; Gundale and DeLuca 2007) by creating favorable microenvironments that enhance microbial colonization (Pietikäinen and others 2000; Warnock and others 2007). If microbial activity is able to oxidize biochar, we need to know which microbes can achieve this, the mechanism by which it occurs, and under what conditions and rates it takes place.

Evidence supporting enhanced microbial abundance and the buildup of recalcitrant soil C comes from studying charcoal-amended Anthrosols and wildfire charcoal. While many studies suggest biochar additions are beneficial for increasing microbial activity and C storage, others have reported accelerated decomposition of soil organic matter (priming) after fresh biochar or charcoal additions. Liang and others (2010) reported high stabilization of organic material added to soils from a tropical environment containing aged charcoal. They reported 25.5% less mineralization of added organic matter to Anthrosols compared to unamended adjacent Oxisols. While the charcoal-amended Anthrosol had more than two times the amount of microbial biomass as adjacent soils, CO₂ respiration was lower compared to unamended adjacent soils. This suggests that the microbial biomass associated with charcoal additions has higher metabolic efficiency (Liang and others 2010). Similar findings supporting microbial proliferation and decreased soil respiration have been reported in mineral soil that was amended with varying rates of maize-derived biochar (Jin and others 2008). Conversely, the potential for biochar to cause or accelerate decomposition of soil surface organic horizons (humus) was reported in a 10-year study of litter bags in the boreal zone (Wardle and others 2008a), where a more rapid loss of humus in the presence of charcoal was demonstrated.

Similarly, Steinbeiss and others (2009) showed that homogeneous biochars with or without N could stimulate the loss of soil organic C (between 8 and 13%) in both agricultural and forest soils. Evidence also suggests that the availability of soil N is a controlling factor for the priming effect of char (Neff and others 2002; DeLuca and others 2006; Gundale and DeLuca 2006). Whether biochar application stabilizes soil organic matter and soil C or it results in priming is still speculated and warrants further investigation (Lehmann and Sohi 2008; Wardle and others 2008a, 2008b; Sohi and others 2010).

Plant Growth Response to Biochar Additions

Biochar can be used as a soil amendment to improve soil quality and crop productivity in a variety of soils (Blackwell and others 2009). This has been demonstrated primarily in soils that are highly weathered or degraded through agricultural activities (Glaser and others 2002; Kimetu and others 2008). Much of the initial information concerning biochar effects on soil parameters and crop yields has come from studying properties of Amazon Dark Earth Anthrosols (terra preta; Laird and others 2009). Soils in this region were created by pre-Columbian Indians (Smith 1980; Woods and McCann 1999) using a slash-and-char method. Compared to the surrounding Oxisols, these Anthrosols are characterized as having enhanced levels of soil organic matter, CEC, pH, base saturation, and nutrients such as N, P, K, and calcium (Sombroek 1966; Smith 1980; Sombroek and others 1993; Glaser and others 2001; Lehmann and others 2003; Liang and others 2006). Additionally, the improved nutrient retention and enhanced soil fertility of these Anthrosols has resulted in higher crop yields relative to the adjacent Oxisols (Lehmann and others 2002; Liang and others 2006; Solomon and others 2007). Their nutrient content, dark color, and greater fertility are partially attributed to their high biochar (charcoal) content (Glaser and others 2001). These soils have C contents of up to 150 g C kg⁻¹ as compared to surrounding Oxisols that have 20 to 30 g C kg⁻¹ (Sombroek 1966; Lehmann and others 2002).

As a result of the greater fertility of these Anthrosols, numerous greenhouse and field trials have been implemented to evaluate impacts of fresh biochar on crop biomass yield and soil properties; however, the majority of the reported studies have taken place in tropical environments, resulting in little understanding of biochar potential in temperate regions. In a pot experiment, Lehmann and others (2002) found that biochar increased rice biomass by 17% and cowpea biomass by 43% when applied at rates of 68t C ha⁻¹ to 135t C ha⁻¹. This growth was attributed to direct nutrient additions from biochar of P,

K, and Copper. Other studies have attributed positive plant growth to positive changes in soil biogeochemistry resulting from biochar additions (Iswaran and others 1980; Wardle and others 1998; Hoshi 2001; Lehmann and others 2002; Chan and others 2007; Van Zwieten and others 2007). For example, Iswaran and others (1980) reported a 51% increase in biomass in soybean crops with biochar additions of 0.5 t ha^{-1} in a pot study. In a field experiment in Japan, Hoshi (2001) found a 20% increase in volume and 40% increase in height of tea trees after biochar additions. In both field and pot studies in Zambia, Chidumayo (1994) reported better seed germination (30% enhancement), shoot height (24%), and biomass production (13%) among seven native woody plants on soils under charcoal kilns as compared to undisturbed Alfisols and Ultisols. Additionally, in tropical zones, larger yield increases have been reported when biochar was applied in combination with inorganic or organic fertilizer (Glaser and others 2002; Lehmann and others 2002; Chan and others 2007; Van Zwieten and others 2007; Steiner and others 2007), with increases of 200% reported relative to unamended, unfertilized plots (Yamato and others 2006). A combination of the abilities to raise soil pH (Hoshi 2001; Yamato and others 2006; Rondon and others 2007; Van Zwieten and others 2007), improve physical properties (e.g., water holding capacity; Iswaran and others 1980), and retain soil nutrients and reduce leaching losses (Hoshi 2001; Lehmann and others 2003; Lehmann 2007) are likely reasons that biochar has contributed to increased plant productivity.

However, not all biochar effects on soil properties are positive, and declines in plant growth have also been reported with biochar additions. Kishimoto and Sugiura (1985) reported biochar additions at 5 t ha^{-1} decreased soybean yields by 37%, while 15 t ha^{-1} decreased yields by 71%. In southeastern Pennsylvania, Mikan and Abrams (1995) found negative response of vegetation in older (more than 100-year-old) charcoal hearth areas due to presence of charcoal. Tree density and basal area were reduced by 40% in charcoal hearth locations compared to non-hearth areas. Although ancient Amazonian Anthrosols have more favorable characteristics than heavily weathered Oxisols from which they were derived, fresh biochar amendments do not consistently improve soil conditions (Chan and Xu 2009).

Positive plant growth and nutrient content responses to biochar have been commonly observed in association with fertilizer application, while neutral or negative responses have been observed following biochar-only amendments (Yamato and others 2006; Gundale and DeLuca 2007; Asai and others 2009; Blackwell and others 2009; Gaskin and others 2010). Greater plant growth yields can be attributed to changes in

soil physical properties previously mentioned (Chan and Xu 2009), but also to the ability of biochar to retain fertilizer nutrients and reduce leaching losses (Lehmann and others 2002). Furthermore, nutrient retention in soils amended with biochar may be attributed to the sorptive capacity of fresh biochar through charge or covalent interactions (Major and others 2009).

However, fertilizer additions are not always capable of ameliorating negative growth responses of fresh biochar additions (Asai and others 2009). It has been suggested that biochar causes N immobilization and could potentially cause N deficiency in plants when applied to soil alone due to high C:N ratios (Sullivan and Miller 2001; Chan and Xu 2009; Lehmann and Joseph 2009), leading to further uncertainty regarding its effect on plant growth. Additions of organic matter with available C:N ratios above 20 are known to cause microbial N immobilization (Fisher and Binkley 2000). Because biochar has a high C:N ratio (up to 400), it is likely that rapid mineralization of a labile C fraction could contribute to reduced soil mineral N, potentially reducing plant available N. However, total C and N content in biochar do not reflect the actual availability of these elements for microbes to cause immobilization. The recalcitrant nature of biochar suggests that few of its components would contribute to immobilization; however, biochar may also sorb organic molecules that have high C:N from soil solution and increase mineralization (Gundale and DeLuca 2007).

Fresh biochar both directly and indirectly influences soil nutrient availability (Blackwell and others 2009; Chan and Xu 2009), which can impact plant growth. Direct effects are largely associated with the retained feedstock nutrients in biochar and are apparent when soil nutrients, plant production, and foliar nutrient concentrations are enhanced with biochar applications (Lehmann and others 2002; Gaskin and others 2010). Amendments of biochar can indirectly affect nutrient availability by adding chemically active surfaces that modify the dynamics of soil nutrients or facilitate soil reaction, modify physical properties of the soil (e.g., reduce bulk density, increase porosity, and increase water holding capacity; Iswaran and others 1980), and encourage the formation of mineral and microbial associations with biochar particles (Pietikäinen and others 2000; Warnock and others 2007). Biochar typically increases pH of acidic soils (Lehmann and others 2002; Gaskin and others 2010; Van Zwieten and others 2010) due to the liming capacity of associated carbonate salts retained in the ash component of biochar. As previously mentioned, this can improve the availability of some nutrients, which is commonly thought to be responsible for positive plant growth responses to biochar amendments (Chan and Xu 2009). However, because it can be difficult to differentiate among direct and indirect

effects of biochar application, suffice it to say the combination is largely responsible for nutrient supply responses.

Further research is needed to understand short-, mid-, and long-term effects on immobilization and mineralization in conjunction with biochar additions to field environments. We need additional research in order to understand the impacts of application rate, feedstock type, and various site types, especially in forests and temperate regions where data are limited.

Amending soils with biochar from various feedstocks will result in differing effects on soil properties and subsequent effects on plant growth. The temperature and heating rate of the pyrolysis process also has important effects on the physical and chemical attributes of the biochar produced (Amonette and Joseph 2009; Downie and others 2009), which will impact soil properties (Gaskin and others 2008). Feedstock such as poultry manure can result in biochar with high pH and P content, while sewage sludge can result in biochar with high N and heavy metal concentrations. Fresh vegetation, wood, or bark may create biochar with neutral pH and nutrient concentrations that reflect feedstock concentrations (Chan and Xu 2009). Gaskin and others (2010) compared biochar derived from peanut shells or wood chips and found peanut-shell biochar had higher nutrient concentrations and raised the pH and base cation concentrations when added to the soil, while wood chip derived biochar had little effect on these parameters. From the limited data available, no optimum range or type of biochar application has been determined to enhance plant productivity (Glaser and others 2002; Lehmann and others 2002). It is likely that the optimum rate of biochar application will depend on each soil type and target plant species.

Biochar Stability and C Sequestration Potential

The long residence time of biochar in soil makes it an important C sequestration tool (Lehmann and others 2006). During the conversion of biomass to biochar, about 50% of the original C is retained in the biochar, which offers considerable opportunity for creating a C sink (Lehmann 2007). There is ample evidence that in certain environments, charcoal is indeed recalcitrant; however, charcoal is not a homogeneous substance (Hedges and others 2000) and certain fractions will decompose at varying rates under different conditions. It has been predicted that the stable portion of biochar has a mean residence time of greater than 1000 years (Cheng and others 2008; Lehmann and others 2008; Liang and others 2008). Deposits of charcoal up to 9500 years old have been found in wet tropical forest soils in Guyana (Hammond and others 2006), up to 6000 years old in Amazonia (Soubies 1979), and up to 23,000 years old in Costa Rica (Titiz and Sanford 2007). Bird and Grocke (1997) found that components of charred material are highly resistant

to oxidation in laboratory treatments with both acid dichromate and basic peroxide, suggesting that fractions of charcoal are long-lived. Additionally, the presence of charcoal in soils and sediments from forest wildfire or prescribed burning, even after thousands of years, indicates the high persistence of black C under natural conditions (Saldarriaga and West 1986; Glaser and others 2001). Black C has been discovered in sediments that are several million years old (Herring 1985), making it up to 13,900 years older than other organic C (Masiello and Druffel 1998). Charcoal's resistance to chemical and microbiological breakdown is attributed to the polynuclear aromatic and heteroaromatic ring system structure (Haumaier and Zech 1995; Glaser and others 2002). The residence time of biochar is unknown and difficult to determine in part due to its heterogeneity. However, its stability is substantially greater than other organic matter under the same environmental conditions (Baldock and Skjemstad 2000; Liang and others 2008; Cheng and Lehmann 2009). Therefore, the transformation of labile plant organic matter into biochar through pyrolysis not only reduces CO₂ emissions from energy production, but biochar additions to the soil constitute a net withdrawal of CO₂ from the atmosphere.

Conclusions and Forest Management Implications

Existing research on intentional biochar application to soil has typically examined the effects of these amendments on crop yields and soil properties in agricultural systems throughout many regions. This pointed research has clarified many biochar uncertainties related to inherent biochar properties and has begun to identify mechanisms behind soil improvements. In fact, general knowledge has become so advanced that “niche” or “designer” biochars are being produced with the intention of providing ideal biochar for enhancing specific soil and crop types. This surge of inquiry, understanding, and new markets has brought about continued support and new project development to further advance the field. However, this is not the case for biochar additions to forest ecosystems. Little is known about the short- or long-term consequences of biochar application to forests, especially given that many of these fire-prone ecosystems have had analogous wildfire charcoal inputs for thousands of years. These ecosystems provide abundant and continual feedstocks in the form of residues that could be converted to biochar on-site, and forests may distinctly benefit from biochar application.

A mobile fast-pyrolysis system, when combined with forest ecosystem biochar application, offers a potential solution

to excess biomass accumulation. Using the abundant forest biomass that is accumulated annually through forest harvest residues and hazard fuel reduction projects may generate biofuel that could reduce dependence on foreign and/or non-renewable energy sources. However, implementation and operational recommendations must be supported by a comprehensive, mechanistic understanding of potential site consequences to infer positive and negative effects associated with biomass removals and biochar additions across the range of site types. Demonstrating parallel results among multiple temporal and spatial-scales, such as field, laboratory, and greenhouse studies, is essential to gain a better understanding of biochar, soil, and plant interactions in soils of the Inland Northwest in association with mobile fast-pyrolysis bioenergy production systems.

The potential to improve C sequestration by adding biochar to soil creates an important opportunity to mitigate greenhouse gas emissions. While this idea is not new (Seifritz 1993), it has recently gained interest with the increasing global awareness of greenhouse gas emissions and the effects of climate change. It has even been suggested that with the use of biochar as a greenhouse gas mitigation tool, biochar C sequestration could exceed current emissions from fossil fuels, providing a net soil C sink (Lehmann and others 2006). In forests, mobile fast-pyrolysis units could be located throughout a large region of forests. This mobility would provide opportunities to reduce hazardous forest biomass accumulations while generating biofuels and biochar, thereby creating a greater opportunity to produce C neutral biofuels and sequester C with biochar application.

The objective of forest bioenergy production systems should not be to enhance soil nutrient status and improve forest productivity with biochar additions, but instead to use the renewable and abundant forest biomass that is annually produced through forest harvest residues or hazard fuel reduction to generate biofuels, reduce wildfire risk, and improve forest health. A mobile fast-pyrolysis system offers a solution to biomass accumulation in forest ecosystems, and may improve the economic and environmental impact of biomass utilization for energy production.

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