BIOCHAR AMENDMENTS TO FOREST SOILS: EFFECTS ON SOIL PROPERTIES AND TREE GROWTH

A Thesis

Presented in Partial Fulfillment of the Requirements for the

Degree of Master of Science

with a

Major in Natural Resources

in the

College of Graduate Studies

University of Idaho

by

Kristin M. McElligott

May 2011

Major Professor: Mark Coleman, Ph.D.

AUTHORIZATION TO SUBMIT THESIS

This thesis of Kristin McElligott, submitted for the degree of Master of Science with a major in Natural Resources and titled "Biochar amendments to forest soils: Effects on soil properties and tree growth", has been reviewed in final form. Permission, as indicated by the signatures and dates given below, is now granted to submit final copies to the College of Graduate Studies for approval.

Major Professor ______________________ Date 5/13/11
Mark Coleman

Committee Members ______________________ Date 12-MAY-11
Deborah Page-Dumroese

______________________ Date 12 MAY 2011
Paul McDaniel

Department Administrator ______________________ Date 13 May 2011
Jo Ellen Force

Discipline's College Dean ______________________ Date 5/13/2011
Kurt S. Pregitzer

Final Approval and Acceptance by the College of Graduate Studies

______________________ Date 5/24/2011
Nilsa A. Bosque-Pérez

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 (C) and altering forest site productivity. Biochar additions have been suggested as a way to mitigate soil C 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. This research evaluates the potential environmental implications of biochar application in forests by examining: (1) the potential for mobile bioenergy and biochar co-production systems in forests, (2) the influence of biochar and biochar application method on standard forest soil properties of three Inland Northwest soils, and (3) the effects of biochar and biochar application rate on poplar growth (a cultivar of *Populus trichocarpa* Torr. & Gray) in various forest soils. The results indicate that biochar contributes to notable short-term soil chemical alterations associated with blending the properties of biochar with those of various soil types, but the nature and scope of the alterations vary by soil type and application method. The soil nutrient alterations do not appear to affect tree growth in the short-term, as biochar had a neutral main effect on poplar growth. These results suggest that biochar produced from bioenergy production could be returned to forest soils to replenish soil nutrient stocks and enhance C storage, with little to no affect on tree growth in the short-term. Results from these studies provided a basic understanding of the potential for biochar in our region, and offer several primary implications for biochar management that could contribute to a comprehensive plan for continuing forest bioenergy production systems.

ACKNOWLEDGEMENTS

With great sincerity, I would like to thank the people who have guided me through all aspects of this process. Thank you to my major professor, Mark Coleman, for his continuous support and encouragement over the past two years. Thank you to Debbie Page-Dumroese and Paul McDaniel, who were always available to provide valuable ideas, edits, and input. I also want to thank the extraordinary staff of the Intermountain Forest Tree Nutrition Cooperative, including Terry Shaw, Mark Kimsey and Adam Robertson, for their invaluable support and advice. With their mentoring, I have learned lessons I will use for the rest of my career. Thank you especially to Derrick Reeves, my colleague and friend, for his input and friendship during my two years at this university, and to Joe Mascaro, my chief supporter and partner during the writing of this thesis. Thank you to my fellow graduate students, Dan Smith and Kevin White, and all the great people I've met along the way that have encouraged me. Finally, I want to especially thank my amazing family for their unwavering encouragement and support throughout my endeavors.

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CHAPTER 1: INTRODUCTION

THE NEED FOR BIOMASS REMOVALS

Removal of residual forest biomass from forest management activities is important for hazard 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). There are an estimated 73 million acres of national forest land in the Western US alone that have been identified as having unnatural or excessive amounts of woody biomass (USDA Forest Service 2003. Biomass residues associated with harvesting are also significant, further resulting in increased susceptibility to catastrophic fire. As a consequence of increasing wildfire occurrence and overstocked stands altering 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 west, the cost of biomass removals often exceeds the value of products, despite increasing interest in utilization of forest biomass. 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. There is little to no market for this 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 could reach up to $1000 per acre in the Western US for mechanical clearing alone (Stokes and Shepard 2003). In 2005, the median cost of bringing biomass to the roadside was $680/acre for mild slopes (USDA Forest Service 2005) not including costs for haul distances. The US Forest Service (2005) analyzed Western forests break-even hauling distance to be a maximum of 86 miles. This assumes 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, it can reduce greenhouse gas emissions if used as a sustainable substitute for fossil fuels. Utilizing residues from logging, pre-commercial thinning, and hazard fuel reductions for energy production will help meet US energy independence goals while promoting forest stand health, and reducing wildfire risks. It is estimated that in the 15 Western 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 et al. 2003). Yet, due to the low value of biomass, limited accessibility, and varying biomass markets, the majority of residuals are left to decay at the site or 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 results in a loss of 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 et al. 2002). Phosphorus (P) can also be lost, but in lower quantities than nitrogen and sulfur. These nutrients are frequently limiting in forest environments (Fox et al. 2007; Kishchuk and Brockley 2002), 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 the tops and limbs, and the lack of research to evaluate 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 overtime, 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 et al. 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 resulted in negative impacts (Kimmins 2004). Further, it is understood that disturbing or removing litter and displacing soil may have significant negative impacts on the subsequent stand rotations (Fleming et al. 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 systems are resilient and maintain large stocks of nutrients, increasing the frequency of biomass removal may exceed the natural capacity for nutrient replenishment between removals, making the need to understand implication of biomass removals critical and urgent.

MOBILE FAST-PYROLYSIS

A sustained bioenergy production system might include removing the energy and 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), which can be effectively used for the production of heat, power, and chemicals (Garcia-Perez et al. 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 (Laird 2008; Lehmann et al. 2006), 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 would provide an economical hazard fuels reduction system by producing alternative sources of energy which could 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 (Bridgewater 2004; Boucher et al. 2000). It involves rapidly heating the biomass (500°C /sec) to moderate reaction temperatures (400-600 °C) followed by rapid cooling with short vapor residence times (1-2 s) and includes drying the biomass feedstock to less than 10% water in to minimize the water in the resulting liquid product (Bridgewater 2004; Mohan et al. 2006) . Pyrolysis produces 60-75% w/w bio-oil, 15-25% w/w solid char, and 10-20% w/w of noncondensable gases, but exact proportions are largely dependent on the feedstock used and process temperatures (Mohan et al. 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 et al. 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 carbon intended for use as a soil amendment. It is analogous to charcoal manufactured through traditional or modern pyrolysis methods, and to black carbon found naturally in fire-ecosystems. Biochar is used as a soil amendment to improve soil nutrient status, C storage and/or filtration of percolating soil water (Lehmann and Joseph 2009). Biochar from pyrolysis and charcoal produced through natural burning share key characteristics including long residence time in soils and a soil conditioning effect (Glaser et al. 2002). Biochar has an inherent energy value which can be used to maximize the energy output of pyrolysis. However, research has shown that application of biochar to soil may be more desirable as it can increase soil organic carbon (SOC), improve the supply of nutrients to plants and therefor enhance plant growth and soil physical, chemical, and biological properties (Glaser et al 2002; Lehmann et al. 2003; Rondon et al. 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 starting 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 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} (Chan et al. 2007; Lehmann et al. 2003, Lima and Marshall 2005). Biochar also contains varying concentrations of other elements such as Oxygen (O), Hydrogen (H), N, Sulfur, P, 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 et al. 2005; Cohen-Ofri et al. 2007 The outer surfaces contain various O and H functional groups and the graphene sheets may contain O groups and free radicals (Bourke et al. 2007). Additionally, biochar has been produced with a range of pH values between 4 and 12, dependent upon the starting 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, O_2 and various soil agents (Cheng et al. 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 O_2 and water (Cheng et al. 2008; Cheng et al. 2006; Liang et al. 2006).

There are increasing concerns associated with contaminants being retained in biochar and leaching into soil once added as an amendment; however, these 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, or treated wood feedstocks (Lievens et al. 2009) and would likely not be an issue if 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 (Zhurinsh et al. 2005). Thus, it is important to understand the chemical composition of the initial feedstock and biochar to avoid potential environmental consequences prior to adding it to forest sites in a large-scale, irreversible manner.

Impacts of biochar on soil

Biochar has 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 et al. 2010), there exists potential for negative impacts. This demonstrates the need for a comprehensive understanding of biochar's origin, production, and functional properties.

Several soil benefits arise from the physical properties of biochar. The highly porous nature of biochar results from retaining the cell wall structure of the biomass feedstock. A wide range of pore sizes within the biochar results in a large surface area and a low bulk density. Biochar incorporation can alter soil physical properties such as structure, pore size distribution and density, with implications for soil aeration, water holding capacity, plant growth, and soil workability (Downie et al. 2009). Evidence suggests that biochar application into soil may increase the overall net soil surface area (Chan et al. 2007) and consequently, may improve soil water and nutrient retention (Downie et al. 2009) and soil aeration, particularly in fine-textured soils (Kolb 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 of biochar 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, porosity, and lower bulk density in mineral soil with biochar can alter water retention, aggregation, and decrease soil erosion (Piccolo and Mbagwu 1990; Piccolo et al. 1996; Mbagwu and Piccolo 1997). Water retention of soil is determined by the distribution and connectivity of pores in the soil matrix, which is largely affected by soil texture, aggregation, and soil 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 in soil. These starting physical properties in biochar occur at a range of scales and affect the proportion of water than can be retained. Kishimoto and Sugiura (1985) estimated the inner surface area of charcoal formed between 400 and 1000°C to range from 200 to 400 m^2 g^{-1}. Van Zwieten et al. (2009) reported the surface area of biochar derived from papermill waste with slow pyrolysis to be 115 m^2 g^{-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 improvement is an indirect result of alterations in soil aggregation and structure after biochar application (Brodowski et al. 2006). Biochar can affect soil aggregation through interactions with SOM, minerals, and microorganisms; however, the surface charge characteristics and their development over time determines the long-term effect on soil aggregation. Glaser et al. (2002) reported that Anthrosols 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, the addition of charcoal increased available moisture 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 additions is most commonly observed in coarse-textured or sandy soils (Gaskin et al. 2007; Glaser et al. 2002). The impact of biochar additions on moisture content may be due to increased surface area relative to that found in coarse-textured soils (Glaser et al. 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 et al. 2003). Nutrient availability can be affected by increasing cation exchange capacity, altering soil pH, or direct nutrient contributions 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 (Lehmann 2003; Liang 2006; Tryon 1948; Mbagwu and Piccolo 1997). 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 et al. 2006). Studies have shown significant increases in the availability of all major cations (Glaser et al. 2002; Topoliantz et al. 2005; Lehmann et al. 2003). 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 (AEC). Cheng et al. (2008) found biochar to exhibit an anion exchange capacity at pH 3.5, which decreased to zero over time as it aged in soil.

Biochar has a higher sorption affinity for a range of organic and inorganic compounds, and higher nutrient retention ability compared to other forms of soil organic matter (Bucheli and Gustafsson 2000, 2003; Allen-King et al. 2002; Kleineidam et al. 2002; Nguyen et al. 2004). Once 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 et al. 2008; Cheng et al. 2006). It also exhibits an ability to sorb polar compounds including many environmental contaminants (Yu et al. 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 et al. 2006, 2008; Lehmann 2007), but increase with time in soil with surface oxidation (Cheng et al. 2006). This supports the findings of high CEC observed in Amazonian Anthrosols (Liang et al. 2006).

Biochar can serve as a liming agent resulting in increased pH and nutrient availability for a number of different soil types (Glaser et al. 2002; Lehmann and Rondon 2006). The carbonate concentration of biochar facilitates liming in soils and can raise soil pH of neutral or acidic soil (Van Zweiten et al. 2007). Mbagwu and Piccolo (1997) report increases in pH of various soils and textures by up to 1.2 pH units from pH 5.4 to 6.6. Tryon (1948) report a greater increase in pH in sandy and loamy soils than in clayey soils. The pH of various soils increase after applications of hardwood charcoals (pH 6.15) than of conifer charcoals (pH 5.15) likely due to their different ash contents of 6.38% and 1.48%, respectively (Glaser 2002).

Biochar feedstocks 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 et al. 2007) depending on starting conditions. Hardwood charcoals are reported to have substantial carbonate concentrations and prove more effective in reducing soil acidity, therefore having a larger influence on soil fertility (Steiner 2007). The liming of acidic soils decreases Al 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 alone 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 pH 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; Meurisse 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, soil texture, organic matter (OM), and soil pH will be a key factor 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 et al. 2008; Sohi et al. 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 et al. 2010), but 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 OM (Wardle et al. 2008a), especially in nutrient-limited environments. Additionally, alkaline biochar may increase the pH of acidic soils and subsequently stimulate 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 pores and its high internal surface area, and increased ability to adsorb OM provide a suitable habitat to support soil microbiota that catalyze processes that reduce N loss and increase nutrient availability for plants (Winsley 2007). The pores are suggested to serve as a refuge by protecting microbes from predation and desiccation while the organic matter adsorbed to biochar provides C energy and mineral nutrient requirements (Warnock et al. 2007; Saito and Muramoto 2002). In temperate ecosystems with wildfire-produced charcoal, N mineralization and nitrification are enhanced (Berglund et al. 2004; Gundale and DeLuca 2007) by creating favorable microenvironments that enhance colonization by microbes (Warnock et al. 2007; Pietikainen et al. 2000). 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 at what rate this will take place.

Evidence supporting enhanced microbial abundance and the build-up 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 increase C storage, others have reported accelerated decomposition of soil OM (priming) after fresh biochar (charcoal) additions. Liang et al. (2010) report high stabilization of organic material added to soils from a tropical environment containing aged charcoal. They reported 25.5% less mineralization of added OM to Anthrosols compared to unamended adjacent Oxisols. While the charcoal-amended Anthrosol had more than two times the amount of microbial biomass than adjacent soils, carbon dioxide (CO_2) respiration was lower compared to unamended adjacent soils. This suggests that the microbial biomass associated with charcoal additions has higher metabolic efficiency (Liang et al. 2010). Similar findings supporting microbial proliferation and decreased soil respiration have been reported in mineral soil amended with varying rates of maize-derived biochar (Jin et al. 2008). Conversely, the potential for biochar to cause or accelerate the decomposition of soil surface OM (humus) has been reported in a 10-year study of litter bags in the boreal zone (Wardle et al. 2008a), where a more rapid loss of humus in the presence of charcoal was demonstrated. Similarly, Steinbeiss et al. (2009) showed that homogeneous biochars with or without N could stimulate the loss of soil organic C (between 8-13%) in both agricultural and forest soils. There is also evidence to suggest that the availability of soil N is a controlling factor for the priming effect of char (DeLuca et al. 2006; Gundale and DeLuca 2006; Neff et al. 2002). Whether biochar application stabilizes soil OM and soil C, or results in priming is still under speculation and warrants further investigation (Sohi et al. 2010; Lehmann and Sohi 2008; Wardle et al. 2008a; Wardle et al. 2008b).

Plant growth effects with biochar additions

Biochar can be used as a soil amendment to improve soil quality and crop productivity in a variety of soils (Blackwell et al. 2009). This has been demonstrated primarily in soils that are highly weathered or degraded through agricultural activities (Glaser et al. 2002; Kimetu et al. 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 to surrounding Oxisols (Laird et al. 2009). The soils in this region, known as *terra preta,* were created by pre-Columbian Indians (Smith 1980; Woods et al. 2000) using a slash-and-char method. Compared to the surrounding Oxisols, these Anthrosols are characterized as having enhanced levels of soil OM, higher CEC, pH, base saturation and nutrients such as N, P, K and calcium (Ca) (Sombroek 1966; Smith 1980; Sombroek et al. 1993; Glaser et al. 2001; Lehmann et al. 2003; Liang et al. 2006). Additionally, the improved nutrient retention, and enhanced soil fertility of these Anthrosols result in the production of higher crop yields relative to the adjacent Oxisols (Lehmann et al. 2003; Liang et al. 2006; Solomon et al. 2007). Their nutrient content, dark color, and greater fertility are partially attributed to their high biochar (charcoal) content (Glaser et al. 2001). These soils have C contents of up to 150 g C kg^{-1} in comparison to the surrounding Oxisols that have 20-30 g C kg^{-1} (Sombroek 1966; Lehmann et al. 2003).

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 et al. (2003) found biochar to increase rice biomass by 17% and cowpea 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 (Cu). Other studies have attributed positive plant growth to positive changes in soil biogeochemistry as a result of biochar additions (Iswaran et al. 1980; Wardle et al. 1998; Hoshi 2001; Lehmann et al. 2003b; Chan et al. 2007; Van Zwieten et al. 2007). Iswaran et al. (1980) reported a 51% increase in biomass in soybean crops with biochar additions of 0.5t ha^{-1} and Hoshi (2001) found a 20% increase in volume and 40% increase in height of tea trees with biochar additions. Chidumayo (1994) reported better seed germination (30% enhancement), shoot heights (24%) and biomass production (13%) among seven native woody plants on soils under charcoal kilns compared to the undisturbed Zambian Alfisols and Ultisols. Additionally, larger yield increases are reported with biochar additions applied together with inorganic or organic fertilizer treatments (Van Zwieten et al. 2007; Chan et al. 2007; Steiner et al. 2007; Glaser et al. 2002; Lehmann et al. 2002), with increases reported at 200% relative to unamended, unfertilized treatments (Yamato et al. 2006). A combination of biochars ability to raise soil pH (Rondon et al. 2007; Van Zwieten et al. 2007; Hoshi 2001; Yamato et al. 2006), improve physical properties such as water holding capacity (Iswaran et al. 1980) and retain soil nutrients and reduce leaching losses (Hoshi 2001; Lehmann et al. 2003; Lehmann 2007) likely contribute to its ability to increase plant productivity.

Still, not all effects on soil properties are positive and declines in plant growth have also been reported with biochar additions. Kishinmoto and Sugiura (1985) reported biochar additions at 5t ha^{-1} decreased soybean yields by 37%, while 15t ha^{-1} decreased yields by 71%. Mikan and Abrams (1995) found negative response of vegetation in >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 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 are commonly observed in association with fertilizer application, while neutral or even negative plant growth responses have been observed succeeding biochar only amendments. Much greater yields in plant growth are observed with fertilizer additions plus biochar, as opposed to fertilizer additions alone (Asai et al. 2009; Blackwell et al. 2009; Gundale and DeLuca 2007; Yamato et al. 2006). This apparent increase in fertilizer use efficiency with biochar is attributed to decreased bulk density, increased water holding capacity (Chan and Xu 2009), and the ability of biochar to retain fertilizer nutrients and reduce leaching losses (Lehmann et al. 2003). Furthermore, nutrient retention in soils amended with biochar may be attributed to the sorptive capacity of fresh biochar through charge or covalent interactions (Major et al. 2009).

It is evident that some biochar is effective at retaining nutrients due to its high adsorptive capacity as previously outlined; however, in some cases, this may prove detrimental for plant nutrient uptake. Decreased growth is frequently reported with biochar amendments when not accompanied by fertilizer additions (Gundale and DeLuca 2007; Asai et al. 2009; Gaskin et al. 2010). Furthermore, it has been demonstrated that fertilizer additions are not always capable of ameliorating the negative growth responses of fresh biochar additions (Asai et al. 2009). Both the sorptive capacity of biochar, and the high C:N ratio are proposed causes for such responses.

Biochar is suggested to cause N immobilization and could potentially cause N deficiency in plants when applied to soil alone due to high C:N ratios (Chan and Xu 2009; Lehmann and Joseph 2009; Sullivan and Miller 2001), leading to further uncertainty regarding its effect on plant growth. Additions of OM 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 a reduction in soil mineral N, and potentially reduce plant available N. However, total C and N content in biochar does not reflect the actual availability of these elements for microbes to cause immobilization. The recalcitrant nature of biochar suggests that few components contained in biochar 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). Further research is needed to understand short, mid- and long-term effects on immobilization and mineralization in conjunction with biochar additions to field environments. The varying biochar growth responses validate the need to understand the impacts of biochar application, and biochar type on various site types, especially in forests and temperate regions where data are limited.

Fresh biochar has been reported to have both direct and indirect influence on soil nutrient availability (Blackwell et al. 2009; Chan and Xu 2009), which can have impacts on 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 (Gaskin et al. 2010; Lehmann et al. 2003). Concurrently, biochar can have indirect effects on soil nutrient availability. Amendments of biochar can add 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, increase water holding capacity; Iswaran et al. 1980), and encourage the formation of mineral and microbial associations with biochar particles (Pietikainen et al. 2000, Warnock et al. 2007). Biochar typically increases pH of acidic soils (Gaskin et al. 2010; Lehmann et al. 2003; Van Zwieten et al. 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, it can be difficult to differentiate among direct and indirect factors associated with biochar application, and the combination is largely responsible for nutrient supply responses.

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 et al. 2009), which will impact soil properties (Gaskin et al. 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 (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 et al. 2002; Lehmann et al. 2002). It is likely that the optimum rate of biochar application will vary and needs to be determined for 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 et al. 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 et al. 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 et al. 2008; Lehmann et al. 2008; Liang et al. 2008). Deposits of charcoal up to 9500 years old have been found in wet tropical forest soils in Guyana (Hammond et al. 2006), up to 6000 years old in Amazonia (Soubies 1979), and up to 23,000 years old in Costa Rica (Titiz & Sanford 2007). Bird and Grocke (1997) found that components of charred material are highly oxidation resistant under laboratory treatment both with acid dichromate and basic peroxide, suggesting fractions of charcoal are long-lived. Additionally, the presence of charcoal from forest burning in soils and sediments even after thousands of years indicates the high persistence of black carbon under natural conditions (Glaser et al. 2001; Saldarriaga and West 1986). Black C has been discovered in sediments that are several million years old (Herring 1985). The age of this charred organic matter is 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 et al. 2002). The residence time of biochar is unknown and difficult to determine in part due to its heterogeneity. However, stability of biochar is substantially greater than other OM under the same environmental conditions (Baldock and Skjemstad 2000; Cheng and Lehmann 2009; Liang et al. 2008). Therefore, the transformation of labile plant organic matter into biochar through pyrolysis not only reduce CO_2 emissions from energy production, but biochar additions to the soil constitutes a net withdrawal of carbon dioxide from the atmosphere.

FOREST MANAGEMENT IMPLICATIONS AND CONCLUSIONS

A mobile fast-pyrolysis system when combined with biochar application offers a potential solution to biomass accumulation in forests ecosystems. By using the abundant forest biomass that is accumulated annually through forest harvest residues and hazard fuel reduction projects, it may be possible to generate biofuel that could reduce dependence on foreign or non-renewable energy sources. If biomass conversion occurs at biomass extraction sites, the economic and environmental impact of biomass utilization for energy production could be improved. In addition, the biochar byproduct can be redistributed to the site of energy extraction and thereby return nutrients retained from the feedstock to the site. The combined properties of biochar suggest it may be a long-term method of C sequestration on forest sites, and could potentially lead to an increase in productivity for many forest sites, particularly those with little organic matter within the mineral soil. 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.

The thesis that follows this introduction has several objectives. This research is meant to evaluate benefits, risks, and tradeoffs associated with biochar application to forest soils, specifically in the Inland Northwest. This will be of particular interest to professionals and scientists in the field of natural resources, as biochar technology is multifaceted and has numerous interdisciplinary management applications. This research has strong implications for future forest management and offers a potential mechanism for C sequestration.

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CHAPTER 2: EFFECTS OF BIOCHAR AND APPLICATION METHOD ON TEMPERATE FOREST AND AGRICULTURAL SOIL CHEMICAL PROPERTIES

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 (C) and altering site productivity. Biochar additions have been suggested as a way to mitigate soil C loss; yet, little is known about the effects of intentional biochar amendments to temperate forest soil in conjunction with biomass removals for bioenergy production. We anticipate biochar additions to modify chemical and biological properties of forest soil. To determine the impacts of adding biochar produced from woody biomass to Inland Northwest soils we applied biochar at one rate (25 Mg ha^{-1}) to an Andisol, Spodosol and Mollisol using two methods, top-dressing and incorporation. After 30 weeks of laboratory incubation, we determined soil chemical and biological properties, and considered leaching losses of nitrogen. The alteration of soil properties and nitrogen (N) retention varied by soil type and application method. Both biochar application methods significantly increased soil C, organic matter (OM), and available potassium (K) in all soils; decreased ammonium (NH_4-N) in the Andisol, and decreased leachate ammonium (NH_4-N) and nitrate (NO_3-N) concentration in of the Mollisol. The incorporated biochar treatment increased cation exchange capacity (CEC) in all soils and resulted in the greatest increase in exchangeable K in the Andisol and Mollisol, while top-dressing significantly raised pH in the Spodosol. The ability of biochar to alter the nutrient status of Inland Northwest soils during this incubation experiment appears to be a direct result of nutrients available in the biochar itself, and is also likely influenced by indirect benefits associated with biochar properties.

These results suggest that biochar produced from bioenergy production could be returned to forest soils to replenish soil nutrient stocks; however, observed reductions of NH_4-N in some forest soils could prove detrimental for plant growth. Further research into the potential benefits and risks of biochar in temperate forests is needed to understand if it is an environmentally viable tool for forest managers using bioenergy production systems.

INTRODUCTION

Biomass removed from US forests offers a critical opportunity to produce renewable energy and mitigate climate change while maintaining forest health (Richter 2009). Over 130 million tons of residual biomass is produced annually as a result of harvesting forest products, pre-commercial thinning of managed forests, and wildfire-hazard fuel removal from federally managed forests (Perlack et al. 2005). Over 36 million dry tons of this biomass is considered recoverable for energy production (Gan 2006). Leaving excess live biomass in forest stands can decrease tree vigor, increase susceptibility to pests and pathogens, and increase risk of catastrophic wildfire because of hazard fuel accumulation. Thus, removal of excess biomass can improve forest health and decrease wildfire risks (Powell 1993; Busse et al. 2009). Feasibility of thinning stands to remove woody vegetation is challenging because of economic and environmental concerns associated with removals, such as transport costs, site nutrient removals, and compromised long-term forest C sequestration potential. Mobile fast-pyrolysis is a biomass utilization approach that manages these concerns by converting residual forest biomass to biofuel and biochar near harvest sites. Pyrolysis generates valuable biofuels that can offset operating costs, while the co-product, biochar, has a market value of its own and many potential uses. From a forest management perspective, the best use for biochar produced during harvest operations may be as a forest soil amendment.

Forest biomass removals could deplete soil organic matter (SOM) and associated nutrient stocks over time, but this potential site degradation could be lessened with biochar amendments. Applying biochar to areas where forest biomass has been removed returns recalcitrant C and most of the nutrients originally held in the biomass to the soil (Gaskin et al. 2008). Furthermore, these recalcitrant amendments may contribute to long-term soil C sinks, thereby enhancing forest C sequestration potential. Biochar application may be especially beneficial in Inland Northwest forests, where many soils have low productivity and, on some sites, low total nutrient capital making them more susceptible to losses in site productivity or soil quality with removal of biomass (Garrison and Moore 1998). Low fertility sites are more likely to experience nutritional deficiencies with biomass removals (Burger 2002); therefore amending these soils with biochar may be appropriate.

Understanding the site degradation associated with long-term biomass removals is limited, largely because of the lack of long-term or appropriate studies. Some information from whole-tree harvests have resulted in reports of nutrient deficiency and growth declines (Sverdrup and Rosen 1998; Joki-Heiskala et al. 2003), whereas reviews of intensely managed stem-only and whole-tree harvesting suggest there are few long-term impacts on soil nutrients or future biomass production (Morris and Miller 1994; Johnson and Curtis 2001; Fox 2000; Hakkila 2002). Even greater uncertainty is associated with site effects from thinning harvests (Powers 2006) and partial cuts ranging from minimal short-term impacts (Sanchez et al. 2006) to significant site impacts depending on starting nutrient status and site characteristics (Henderson 1995; Grigal and Vance 2000). Removing logging slash from forest stands instead of leaving the harvest residues on site can alter nutrient availability (Sinclair 1992), and biological activity (Harvey et al. 1976; Covington 1981). Therefore, concerns of site degradation from biomass to bioenergy productions systems may be premature given the limited evidence, yet it is important that soil quality, function, and productivity potential are maintained during these thinning activities to maintain long-term productivity. Even with minimal site impacts from biomass removals, biochar amendments may still prove beneficial to forestry sites by enhancing soil quality and for C sequestration potential.

Biochar amendments have not been extensively tested in temperate forest soils, and their effects on these ecosystems are uncertain. Several studies indicate that biochar can enhance soil productivity and nutrient status in temperate and tropical agricultural systems, and improve plant productivity (Lehmann et al. 2006; Lehmann and Rondon 2006; Laird 2008; Sohi et al. 2010). Inland and Pacific Northwest soils are unique compared to many agricultural soils to which biochar has previously been applied. Differences are largely due to volcanic ash inputs and andic properties. Andic soil properties include higher concentrations of poorly crystalline minerals (e.g. ferrihydrite and allophanes) that have higher surface areas. The properties of poorly crystalline minerals cause them to be highly reactive in terms of chemicals, organic compounds, and microbial interactions, and also create distinct soil physical properties, such as low soil bulk density and high water-holding capacity (Buol et al. 1989; McDaniel et al. 2005; McDaniel and Wilson 2007), that may unpredictably alter interactions with biochar. Also, the high volcanic-ash inputs of these soils make them at higher risk of site degradation with biomass removals due to their low nutrient capital and susceptibility to erosion (Garrison and Moore 1998). Therefore, these soils may particularly benefit from biochar additions due to their unique properties and the soil enhancement potential of biochar, but because it is unclear if and how biochar affects different forest soils, testing is needed to proceed with forest bioenergy systems.

Although there is growing evidence that biochar enhances agricultural productivity (Blackwell et al. 2009), evidence for effects of biochar in temperate forest systems stems from few other sources. Fire ecology shows that charcoal enhances soil productivity, adds to stable soil C pools, and positively influences soil biological properties (Zackrisson et al. 1996; Pietikainen et al. 2000; DeLuca and Aplet 2008). However, unexpected consequences of charcoal are reported by Wardle et al. (2008), including accelerated decomposition of humus resulting in a net loss of soil C. Additionally archeological charcoal remains from historic operations (hearths) dating hundreds of years decreases forest productivity in some systems (Mikan and Abrams 1995). These studies in forest systems contradict claims of enhanced productivity as demonstrated in agricultural systems. Studying wildfire charcoal in temperate forests and the subsequent charcoal modifications by soil processes over decades or centuries allows some inference into the long-term fate of biochar in these ecosystems; however, biochar produced from pyrolysis differs physically and chemically from wildfire charcoal (Baldock and Smernik 2002; Cheng et al. 2006) due to numerous interacting factors such as the amount and variation of oxygen present, rate of heating, temperature and feedstock type. These differences among char production conditions, soils and plant responses, suggest that biochar effects in the Inland Northwest region may vary by soil type. It also remains unclear how the unique physical and chemical properties of freshly produced biochar via pyrolysis will affect short-term forest management and long-term ecosystem processes.

Another challenge involves application of biochar to biomass harvest sites. To be effective at improving soil nutrient supply and retention, CEC, and microbial associations, biochar should be present at rooting depth (0-30 cm depth in most forest soils) (Blackwell et al. 2009). However, unlike agricultural systems where biochar can be easily tilled into a plow layer, forest sites have dead branches, fallen tree stems, uprooted stumps, uneven ground, as well as live under- and over-story vegetation that make incorporation more difficult and perhaps undesirable. Further, disturbing the surface organic horizons is considered detrimental to long-term site sustainability (Page-Dumroese et al. 2010). Currently, experimental application of bulk biochar in Inland Northwest forests has been limited to manual surface applications, but other applications such as remediation of skid trails or log landings or decommissioning roads have been considered. Other concerns with mechanical incorporation of biochar include soil disturbance effects associated with forest management activities, which can cause the loss of soil OM, and can accelerate carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) emissions (Keller et al. 2005). Given the biochar application challenges associated with various sites, assessment of different methods and responses are needed.

The quality, method, and objective of biochar amendments may be dependent on its physical and chemical properties, which may also influence the desired application method or technique. For example, biochar with high quantities of soluble nutrients may be added to the surface to encourage rapid release and uptake by plants, whereas incorporating biochar throughout the soil may encourage beneficial soil-char reactions to occur at a faster rate than if top-dressing is used. It is apparent by the presence of charcoal at depth in forest soils that vertical transport readily occurs, however the rate of this occurrence, especially after manual surface application, is unclear and will largely depend on soil structure, climatic regime, and site characteristics (Blackwell et al. 2009). Therefore, incorporation of biochar after surface application would not occur immediately and may require natural mechanisms such as seasonal freeze-thaw events, transport by water, and earthworm activities (Topoliantz et al. 2005), pedoturbation (Ping et al. 2005), or root uplift (Bormann et al. 1995) , which could delay desired biochar interactions with minerals and soil OM for years. The impacts on soil properties and plant growth may subsequently be delayed. Therefore, it's important to assess the potential benefits or shortcomings associated with field application methods and identify whether timing of soil enhancements differ by application methods.

For biochar to be added on a large scale and used as a viable soil amelioration tool for land managers, we must evaluate environmental impacts associated with biochar application in regions and soils where in-woods fast pyrolysis technology may be appropriate. Therefore, the objective of this research was to evaluate chemical changes to temperate forest and agricultural soils in the Inland Northwest after either adding biochar to the surface or by incorporation. After 30 weeks of laboratory incubation we determined changes in standard soil chemical properties, microbial biomass, and assessed N losses in leachate. These studies were designed to test the following hypotheses: (1) biochar will improve standard soil chemical properties by enhancing CEC, raising soil pH, and increasing total C in all soil types, (2) responses to biochar additions will depend on soil type, and (3) responses will differ by biochar application method with incorporation than top dressing due to increased potential for biochar-soil interactions.

MATERIALS AND METHODS

Biochar and Soil

We used fast pyrolysis CQuest™ biochar produced by Dynamotive Energy Systems derived from hardwood forest residues (West Lorne Bio Oil Co-Generation L.P. division, West Lorne, Ontario, Canada N1L 2P0). The biochar used in this study had a bulk density of 0.25 Mg m^{-3}(Dynamotive Energy Systems). The biochar was analyzed for available potassium (K) and phosphorus (P) (Gavlak et al. 1994, Peech and English 1944, Murphy and Riley 1962), total C and N (LECO, St. Joseph, MI), CEC (Chapman 1965), exchangeable calcium (Ca), magnesium (Mg), potassium (K) and sodium (Na) (Gavlak et al. 1997), available NO_3-N and NH_4-N (Norwitz and Keliher 1985, Westfall et al. 1993), organic matter (OM) (Sims and Haby 1971, Walkley 1947) and pH (Gavlak 2005). All biochar analyses were conducted at the Analytical Sciences Laboratory, University of Idaho, Moscow, ID.

Three soil types were selected that typify Inland Northwest forest and agricultural soils: (1) a forested Andisol, (2) a forested Spodosol, and (3) an agricultural Mollisol, each collected in October, 2009. The forested Andisol soil was collected from the upper 20cm of the Bw horizon of a Grandad silt loam, a medial over loamy, amorphic over micaceous, frigid Alfic Udivitrand (Soil Survey Staff 2009). This soil was collected near the border of Latah County and Clearwater County, ID, 46° 48' 27" N 116° 19' 36" W and had a *Thuja plicata*/*Clintonia uniflora* (THPL/CLUN) forest type (dominant tree species present: western redcedar (*Thuja plicata* Donn ex D. Don), grand fir (*Abies grandis* [Douglas ex D. Don] Lindl.) and Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco). The forested Spodosol was collected from the E horizon only of a sandy, mixed, frigid Aquic Haplorthod (P. McDaniel, pers. comm.). This soil was collected from the Idaho Panhandle National Forest near Priest Lake, ID, 48° 36' 59" N 116° 50' 03" W and had a *Tsuga heterophylla*/*Asarum caudatum* (TSHE/ASCA) forest type (dominant tree species present: western hemlock (*Tsuga* *heterophylla* [Raf.] Sarg), lodgepole pine (*Pinus contorta* [Douglas ex.] Louden), and western redcedar (*Thuja plicata* Donn ex. D.Don) (Cooper et al. 1991). The Mollisol was collected from the upper 20cm of the Ap horizon of a fine-silty, mixed, superactive, mesic Pachic Ultic Haploxeroll (Soil Survey Staff 2010); it was collected from a winter wheat research field 2 mi southeast of Moscow, ID, 46° 43' 37.88"N, 116° 57' 34.01W". This soil is used mainly for dryland crops (*e.g.* small grains, peas, lentils, alfalfa, and grasses for hay). The Mollisol represents a cultivated comparison to the two forest soils. After collection, all soils were air-dried, sieved to 2 mm, and then stored at room temperature until use.

Treatments and Column Preparation

We conducted biochar incubations in open-top, 10-cm-diameter, 18-cm tall, schedule-40 PVC columns. Biochar-amended treatments included top-dressing, referred to as "surface," and incorporation, referred to as "mixed." Biochar was added at a rate equivalent to 25 Mg ha^{-1} for both treatments. This high rate was chosen to elicit an effect as preliminary work suggested impacts would be imperceptible with operational amounts (~2 - 6 Mg ha^{-1}). These treatments were compared to control soils with no biochar additions. Treatments were replicated six times for each soil type for a total of 54 soil columns (3 soils x 3 treatments x 6 replicates). Control treatments were constructed by pouring each soil into PVC columns and tapping to an initial bulk density of 1.0 Mg $m^{-3}$for the Spodosol, 0.7 Mg m^{-3} for the Andisol, and 1.2 Mg m^{-3} for the Mollisol. These bulk densities were chosen based on bulk density measurements of each soil type at the time of collection. Prior to top dressing, the mineral soil was prepared identically to control treatments. Surface treatments were obtained by applying 25 Mg ha^{-1} biochar to the top of each soil column. Mixed treatments were obtained by homogenously mixing each mineral soil with the equivalent of 25 Mg ha^{-1} biochar. Once mixed, amended soil was poured into columns and tapped to obtain adjusted field bulk densities. Both control and treated soils were filled to the 15 cm mark of the soil column, leaving the remaining 3 cm as head space. Organic horizons (inclusive of Oa, Oe, Oi horizons) were collected intact from each forest site and replaced on the surface of their respective soil types in each treatment. The organic horizons were added at the rate of 5g, prior to adding the surface treatment of biochar to standardize the surface horizon amounts (*e.g.,* the surface biochar was applied over the surface organic horizons). Organic horizons were included in the forest soil treatments to emulate natural field conditions and avoid excluding this essential component of the forest environment. No organic horizons were collected from the agricultural soil site as agricultural residues were not present during soil collection. Once columns were filled, they were randomized and suspended for the duration of the experiment using a custom-built rack. Soils were supported at the bottom of the columns using 20-mesh nylon screen. Treatments were irrigated to field capacity once a week with a 0.01M solution of CaCl to encourage wetting-drying cycles. The 0.01 M CaCl solution is meant to simulate the nutrient status of local rainwater. At the time of irrigation, soil columns were monitored with a TDR soil moisture meter to ensure no moisture leached through the bottom of columns during weekly watering. Treatments were laboratory incubated for 30 weeks at room temperatures.

Soil & Leachate Analysis

After 30 weeks of incubation, soil columns were leached with 150 mL of the CaCl solution. Leachate was collected and NH_4-N and NO_3-N were determined colorimetrically using Lachat Quick Chem 8500 at the Ecosystems Analysis Lab, Lincoln, NE. All six columns from each treatment and soil type were destructively sampled and the entire amount of soil—including biochar in amended treatments—was analyzed for available potassium (K) and phosphorus (P) (Gavlak et al. 1994; Peech and English 1944; Murphy and Riley 1962), total carbon (C) and nitrogen (N) (LECO, St. Joseph, MI), cation exchange capacity (CEC) (Chapman 1965), exchangeable calcium (Ca), magnesium (Mg), potassium (K) and sodium (Na) (Gavlak et al. 1997), available nitrate-nitrate (NO_3-N) and ammonium (NH_4-N) (Norwitz and Keliher 1985; Westfall et al. 1993), organic matter (OM) (Sims and Haby 1971; Walkley 1947) and pH (Gavlak 2005).

Microbial biomass was determined using chloroform-fumigation, direct extraction in 0.5 M K_2SO_4 (Anderson and Joergensen 1997; Horwath and Paul 1994) following a five day fumigation. Extracts were analyzed for DOC on a Shimadzu TOC/TN analyzer (Shimadzu TOCVCPN, Shimadzu Scientific Instruments, Columbia, MD). Chloroform labile microbial biomass C was determined by subtracting the total dissolved C in fumigated from non-fumigated samples.

Data Analysis

A general linear model was used to test for significant effects (α=0.05) of treatment type (incorporated and top-dressing reported as mixed and surface), soil type (Andisol, Mollisol, Spodosol), and their interaction on all selected soil and leachate properties. Least-squared means were generated and used to test for significant differences between model variables, followed by the Tukey's post-hoc procedure to test all pairwise comparisons among treatments and soils. All data were evaluated statistically using SAS PROC GLM (SAS Institute Inc, 2008).

RESULTS

Biochar properties were distinct from those of the three soils (Table 1). The pH of biochar was two units higher than the three soils averaged together. C content of biochar was more than ten times that of the highest soil. Both available and exchangeable K values were two to more than 30 times those found in soil. It is important to note that NH_4-N for biochar was more than half that of the soil concentrations. These different properties influenced the soil responses after biochar amendment.

Differences in application method by soil type

Biochar amendments produced variable results by application method, but effects were varied among soil types as evident by the significant interaction between application method and soil type ($P < 0.03$) for total C, available NH_4-N, pH, and exchangeable K. Carbon increased with both biochar application methods in all soils, but the magnitude of change differed among the soil types (Figure 1A). In the Spodosol, total C increased by more than 2.4 times while the Andisol only increased 1.3 times over the control soil C values. Available NH_4-N decreased with both surface and mixed application methods compared to the control. In the Andisol, NH_4-N decreased 63% with the surface treatment and 42% with the mixed treatment (Figure 1B). This decrease was not apparent in either the Spodosol or Mollisol. Soil pH increased by 8% in the surface treatment of the Spodosol (Figure 1C), and did not change with either treatment in the Andisol or Mollisol. Exchangeable K increased in all soils with both biochar application treatments ($P = 0.024$, Table 1); however, the magnitude of this increase differed among soil types. Exchangeable K increased with both surface (16%) and mixed treatments (21%) in the forest Andisol, with the mixed treatment (19%) in the Mollisol, and was unaffected in the forest Spodosol (Figure

1D). For all soil types, there were no changes ($P > 0.1$) in soil Ca, Mg, Na, available P, NO_3+NO_2-N and total N with either biochar treatment (Table 2).

Changes in soil chemical properties

Biochar additions of 25 Mg ha^{-1} had distinct effects on soil chemical properties after 30 weeks of incubation for all three soil types; however, the extent of enhancement varies by soil type and application method. Biochar application increased soil C, organic matter, CEC, and available K in all soils (Figure 2 and 1A, Table 2). Additionally the average of all three soil types had a total C increase of 75% with the surface treatment and 79% with the mixed treatment over the control (Figure 1A). Similarly, both biochar treatments increased OM by an average of 7% (Figure 2A). The surface treatment had no significant effect on CEC while the mixed treatment increased soil CEC by 5% relative to the control (Figure 2B). Available K increased with the surface treatment (13%) and it doubled with the mixed treatment (27%) as compared to the control (Figure 2C).

Leachate and microbial biomass analyses

Biochar additions considerably reduced the concentrations of N leached from the Mollisol, but not the Andisol or Spodosol. Surface treatment of the Mollisol reduced leachate NH_4-N concentrations by 38% relative to the control, and the mixed treatment nearly eliminated (99%) NH_4-N in the collected leachate (Figure 3A). Similar patterns were found for NO_3-N leaching from the Mollisol; resulting in a 56% decrease with the surface biochar treatment and 94% decrease in the mixed biochar treatment. There were no changes in leachate NH_4-N or NO_3-N concentrations from the Andisol or Spodosol (Figure 3B).

However, the concentrations of inorganic N compounds were half to two orders of magnitude lower for the forest soils than the agricultural soil.

Soil microbial biomass C (MBC) was not significantly altered by biochar additions in any soil type ($P > 0.1$) after 30 weeks of laboratory incubation. Although biochar application methods had no statistical effect on microbial biomass C, there were significant differences among soil types ($P < 0.0001$). The Andisol had significantly higher mean MBC, 2445 mg C kg^{-1} dry soil, than both the Spodosol, MBC 490 mg C kg^{-1}, and the Mollisol, MBC of 556 mg C kg^{-1}.

DISCUSSION

As hypothesized, biochar amendments to Inland Northwest soils altered nutrient status and C storage. Specifically, both biochar application methods significantly increased soil C, OM, available and exchangeable K, CEC, pH, from all soils and decreased N leaching from the Mollisol. Extractable NH_4-N decreased significantly in the Andisol after biochar additions, which is of concern since N is commonly limiting in forest soils and this soluble inorganic form of N is readily available for plant growth and commonly used as an indicator of soil quality. Ultimately, the nature and scope of the soil alterations depended on soil type and biochar application method.

In the short term, we found that increases in soil nutrients were primarily associated with the nutrient content of the applied biochar. The biochar used in this study contained more C, available K, and OM compared to all three unamended soils (Table 1). Accordingly, we observed significant increases in these components when biochar was added to each of the soil types, although the magnitude of increase varied by soil type. The nutrients in biochar depend on the nature of the starting feedstock (Gaskin et al. 2008) and pyrolysis process (*e.g.* temperature, heating rate, duration) (Tsai et al. 2007), suggesting that nutrient enhancements will vary among biochar types. Other studies have reported that soil nutrient increases due to biochar nutrient content may be short-lived, declining with plant uptake and leaching (Gaskin et al. 2010; Rondon et al. 2007; Steiner et al. 2007; Topoliantz et al. 2005). Despite these losses, C is expected to increase on a long-term basis due to the recalcitrant nature of the C contained in biochar (Lehmann et al. 2006; Sombroek et al. 2003). We report increased C among all soil types — this is not surprising given that the biochar used is predominantly C (62%) — but, the amount of increased C varied by soil type and could be a function of starting soil OM content and proportions of active clays (Lal et al. 1995). Regardless, the significant increases in C among all soil types and application methods make biochar a valuable C sequestration tool, as evident by its long residence times in many ecosystems (Agee 1996; Rackham 1980; Lehmann et al. 2006; Mann 2008).

Biochar can also affect soil nutrient availability indirectly. Amendments of biochar can add chemically active surfaces that modify the dynamics of soil nutrients or facilitate soil reaction, modify physical properties of the soil (*e.g.* reduce soil bulk density, increase porosity, improve water holding capacity; Iswaran et al. 1980), and encourage the formation of mineral and microbial associations with biochar particles (Pietikainen et al. 2000, Warnock et al. 2007). Cation exchange capacity increased in all three Inland Northwest soils but only when biochar was mixed with the mineral soil. This result agrees with previous studies, which generally find a rapid CEC response with fully-incorporated biochar amendments (Novak et al. 2009), compared to those using top-dressing approaches, which fail to produce a strong influence on CEC (Blackwell et al. 2009). We might expect greater cation retention over time due to increased CEC with biochar aging, and biochar movement into the mineral soil. Freshly produced biochar has less ability to retain cations resulting in minimal CEC (Cheng et al. 2006; Lehmann 2007; Cheng et al. 2008), but with time and incorporation in the soil, the surfaces of biochar particles oxidize and interact with soil constituents, resulting in an increase in functional groups and greater surface negative charge (Liang et al. 2006), which ultimately leads to increases in CEC. It is possible that top-dressing failed to enhance the formation of organo-mineral complexes because a majority of biochar remained on the soil surface over this 30 week study, or that the biochar did not sufficiently oxidize over this time-scale, resulting in minimal change in CEC.

Available K increases were also greatest in the mixed treatment, suggesting this response was both a direct and indirect result of biochar additions. With an increase in CEC, we would expect greater cation retention. Exchangeable K increased for both treatments in the Andisol and for the mixed treatment in the Mollisol, while there was no effect in the Spodosol. Both the Andisol and Mollisol had higher initial percent OM, CEC, and proportions of active clay (Soil Survey Staff 2010) than the Spodosol, suggesting that the starting soil colloid composition could influence biochar's capacity to impact soil properties. Together, these direct and indirect enhancements suggest biochar could be effective at altering and potentially improving soil nutrient status, however enhancements should not be expected as the extent of change may be dependent on starting soil type and site characteristics.

Biochar can indirectly affect nutrient availability by altering soil pH. Since biochar typically has higher pH than soil it can act as a liming agent resulting in an overall increase in soil pH (Glaser et al. 2002; Lehmann and Rondon 2006). Higher soil pH increases nutrient availability and decreases the proportion of Al^{+3} and H^+ ions occupying cation exchange sites, which effectively increases base saturation (Brady and Weil 2004). The starting pH of the Spodosol was the lowest among the tested soils, with a value of 3.9; whereas biochar had a pH of 6.8. The surface treatment of biochar caused a small increase in soil pH of the Spodosol. The higher pH of the biochar likely explains the increase in the Spodosol pH. The Andisol had an initial pH that was similar to the pH of the biochar, which would explain why there was no change in pH relative to untreated soil following incubation. Although the Mollisol started out with a pH of 4.4, the addition of high pH biochar resulted in no significant change in pH possibly because this soil is more highly buffered by OM (McCauley 2009). Soil texture may also play a role. Tryon (1948) reported a greater increase in pH in sandy and loamy soils than in clayey soils with biochar addition, and since the Mollisol has a 20-25% clay component, this may also play a role in limiting the change in soil pH. The liming effect associated with biochar may not be ideal for all soil types and plant communities. Increased soil pH associated with biochar additions has 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. Many forest plants and fungi thrive in lower pH soils (Meurisse 1976, Ryan et al. 1986), therefore altering forest soil pH through the addition of biochar may result in unfavorable shifts in above- and belowground biota. Determining how biochar affects soil pH, and subsequently how these alterations could affect soil function will be a key factor in determining how successful biochar application may be on forest soils.

Biochar can play a key role in nutrient cycling, potentially affecting N retention when applied to soils. In our study, the effects of biochar on soil nutrients were largely positive, but we observed significant decreases in NH_4-N in the Andisol. This was the only soil type that exhibited a negative response for both surface and mixed biochar treatments (Figure 2B). Decreases in NH_4-N may have resulted from losses due to immobilization as a result of an increased C:N ratio after carbon-rich biochar additions or from losses due to leaching, nitrification or denitrification – though it is unlikely in these well aerated acidic soils (Buol et al. 1997). The C:N ratio of the Andisol increased from 26:1 before biochar amendments to 36:1 with surface treatment, and 38:1 in mixed treatment. However, N immobilization in soil with biochar additions is unlikely because the biochar is made up of biologically recalcitrant carbon that is not easily mineralized by the soil microbial community (Chan and Xu, 2009). Reductions could instead be a dilution effect associated with biochar additions. Adding an N-depleted amendment at a relatively high rate (25 Mg ha^{-1}) to a soil with high starting rates of NH_4-N could have a notable dilution effect. The Andisol originally had 52 ug g^{-1} NH_4-N, the highest of all soils tested, whereas the biochar contained 3.3 ug g^{-1} of NH_4-N (Table 1). Furthermore, we would have expected potential increase in NH_4-N due to its positive charge and the slight increase in CEC with biochar treatments. It has been suggested that adding biochar to soil will increase NH_4 storage by enhancing CEC in soils (Clough and Condron 2010). Reductions of NH_4-N in the surface treatment could also be a result of nitrification, evident by the increase (although not statistically significant) in NO_3+NO_2-N pool following biochar additions, which increased from 79 μg g^{-1} in the control to 90.3 μg g^{-1} with biochar surface application. There was no effect on NH_4-N in the Mollisol, likely because it was a fertilized agricultural soil and it had a lower initial C:N ratio (12:1) and higher mineral N content as compared to the Andisol. Biochar application to soil has generally resulted in positive effects on soil fertility according to existing literature, particularly in sandy and infertile soils. However increasing evidence for negative consequences, particularly related to effects on soil N, stresses the importance of achieving a better understanding of potential site implications once biochar is irreversibly added to the soil.

Nutrient losses associated with leaching from the soil profile are not a large concern in most forest soils in the Inland Northwest, because these forests are typically N deficient and rarely fertilized. However, this can be a considerable problem in fertilized agricultural systems. Biochar has been found to decrease nutrient leaching when added to agricultural soil (Lehmann et al. 2003; Lehmann et al. 2006), which can improve fertilizer use efficiency and reduce pollutant leaching. Biochar is suggested to have a strong adsorption affinity for soluble nutrients such as nitrate (Mizuta et al. 2004) and ammonium (Lehmann et al. 2002). Our results show large reductions in the amounts of NH_4-N and NO_3-N leached from the agricultural Mollisol with both biochar treatments. While this may have little implication on N-deficient forest soils, it suggests when combined with nutrient additions from fertilizers or symbiotic N-fixing plants, biochar can reduce losses of mobile nutrients and lessen the negative environmental impacts of fertilizer runoff.

Biochar application strategies could have a considerable impact on soil processes and affect the fate of biochar particles in soil. Application of biochar to forest sites may be limited to top-dressing due to site characteristics and feasibility. This gives rise to a significant concern about the loss of biochar through either wind or water erosion. Husk (2009) estimated biochar losses of 30% associated with handling and surface application of biochar to a commercial agricultural field; however, it is presumed that application in forests would have reduced losses due to surface roughness resulting from irregular soil surface and the presence of surface organic horizons, fine and coarse woody debris, shrubs, and grasses that are able to retain biochar in the area it is applied. By the end of our 30-week incubation, biochar had begun moving into the upper 5 cm of the mineral soil of the column. From this, it is expected that in the forest, biochar will make its way into the mineral soil at a moderate rate depending on site conditions. Charcoal produced during wildfires has been shown to be mixed to a depth of 1 m, but a majority remains above 30 cm of the soil surface with about 70% remaining within 10 cm of the surface (Carcaillet 2001; Gavin 2003). The results from our study show that in the short-term, biochar application method has minimal impact on soil nutrient enhancement, but a dramatic effect on N retention. The long-term effects of biochar in forest soils will provide the greatest benefit for long-lived perennial trees suggesting that in forests, application method will likely have little influence on long-term effects of mineral, organic, and microbial associations with biochar particles provided biochar remains on site and is not lost to erosion. An alternative biochar application method includes incorporating biochar into the soil during skid trail or log landing rehabilitation. The low bulk density (Table 2) of biochar would help reduce compaction in these features and give an immediate boost to soil productivity. Additionally, biochar could be turned into pellets (Dumroese et al. 2011) and applied in a similar method as fertilizer. Biochar pellets would not be as subject to wind or water erosion as bulk biochar and could be either surface applied or mixed into the soil

CONCLUSIONS

Biochar amendments to forest soils were effective at increasing soil C and enhanced soil OM content, but did not improve the soil N status. Biochar and bioenergy co-production provides an innovative method for handling excess forest biomass to sequester C and potentially improve soil and plant productivity. In-woods fast pyrolysis can reduce our reliance on fossil fuels, provide a new income stream for forestry and rural communities, and generate biochar as a soil conditioner to mitigate potential nutrient losses from biomass removals. The ability of biochar to improve nutrient status of two Inland Northwest forest soils and an agricultural soil primarily appears to be a direct result of the nutrients added with the biochar. Results from this study suggest that biochar additions will result in generally improved soil qualities in both temperate forest and agricultural soils, with the possibility of lower NH_4-N concentrations depending on soil type. The magnitude of response varies depending on initial soil and site properties. Biochar is effective in significantly enhancing soil C, OM, available and exchangeable K, CEC, and pH on a relatively short timescale. It is also able to decrease inorganic N leaching losses from soils with relatively high extractable nitrogen levels. In combination, these soil chemical changes can translate into maintenance of, and potentially improved, forest site productivity. Results from an incubation study such as this must be validated through a field study to determine how realistic seasonal fluctuations in temperature and moisture, along with existing vegetation and soil microbes, might influence process by which biochar-mediated changes occur. Evidence obtained in this study on temperate forest soils indicates largely positive impacts of applying biochar to forest soils. Using biochar in forest systems provides land managers with a soil amelioration tool that increases the recalcitrant soil carbon pool and supports sustainable forest management as part of a forest bioenergy production system.

TABLES AND FIGURES

Table 2.1. Analysis of biochar and unamended soil properties

	Biochar	Andisol	Spodosol	Mollisol
pH	6.8	5.33 ± 0.07	3.87 ± 0.03	4.40 ± 0.06
CEC (cmol$^+$ kg^{-1})	30	31.67 ± 0.88	5.43 ± 0.07	20.00 ± 0.0
Base Saturation (%)	Na	56.27 ± 2.83	61.29 ± 9.73	65.93 ± 7.57
OM (%)	Na	6.67 ± 0.12	1.37 ± 0.07	2.97 ± 0.03
Total C (%)	62	4.57 ± 0.23	0.95 ± 0.05	1.83 ± 0.03
Total N (%)	0.18	0.17 ± 0.005	0.05 ± 0.003	0.14 ± 0.009
K (cmol$^+$ kg^{-1})	1.6	0.81 ± 0.0	0.05 ± 0.0	0.71 ± 0.05
Ca (cmol$^+$ kg^{-1})	2.2	16.0 ± 0.58	3.03 ± .55	11.0 ± 1.0
Mg (cmol$^+$ kg^{-1})	0.35	0.85 ± 0.04	0.16 ± 0.0	1.35 ± 0.48
Na (cmol$^+$ kg^{-1})	0.17	0.12 ± 0.03	0.09 ± 0.0	0.13 ± 0.03
NH_4+ (µg g^{-1})	3.3	52.0 ± 1.53	6.50 ± 1.05	23.67 ± 7.05
NO_3 + NO_2 (µg g^{-1})	< 1.6	79.0 ± 12.42	0.80 ± 0.0	97.73 ± 81.31
Available K (µg g^{-1})	710	263.33 ± 3.33	18.0 ± 0.0	176.67 ± 14.53
Available P (µg g^{-1})	17	6.57 ± 0.09	1.77 ± 0.03	15.0 ± 0.58

Note: Values represent arithmetic mean ± standard error of the mean (n=6).

Table 2.2. Analysis of Variance degrees of freedom (DF) and *P*-values for selected soil properties response to biochar treatments and soil type.

			---------------------%--------------------					-------------Exchangeable----------				---------------Available-----------			
Source	**DF**	**CEC**	**Base Saturation**	**Organic Matter**	**Total N**	**Total C**	**pH**	**Ca**	**Mg**	**K**	**Na**	**P**	**K**	**NO_3-N**	**NH_4-N**
Application Method	2	0.025	0.658	0.022	0.172	<.0001	0.029	0.554	0.532	0.00	0.22	0.382	<.0001	0.941	0.006
Soil	2	<.0001	0.056	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001	0.379	<.0001	<.0001	0.01	<.0001
Application Method*Soil	4	0.289	0.468	0.543	0.302	**0.012**	**0.028**	0.549	0.701	**0.024**	0.647	0.133	0.14	0.891	**0.003**

Table 2.3. Analysis of soil nutrient properties at 30 weeks

Soil	**Treatment**	**CEC** **$cmol^+ kg^{-1}$**	**Base Saturation %**	**OM %**	**Total N %**	**Ca** **$cmol^+ kg^{-1}$**	**Mg** **$cmol^+ kg^{-1}$**	**NO_3-N** **$\mu g\ g^{-1}$**	**Available K** **$\mu g\ g^{-1}$**	**Available P** **$\mu g\ g^{-1}$**
	Control	31.67 ± 0.88	56.27 ± 2.83	6.67 ± 0.12	0.17 ± 0.005	16.0 ± 0.58	0.85 ± 0.04	79.0 ± 12.42	263.33 ± 3.33	6.57 ± 0.09
Andisol	Surface	31.0 ± 0.0	61.81 ± 1.30	7.0 ± 0.0	0.18 ± 0.005	17.33 ± 0.33	0.79 ± 0.08	90.33 ± 4.91	286.67 ± 13.33	6.57 ± 0.09
	Mixed	32.33 ± 0.33	57.45 ± 2.19	6.97 ± 0.12	0.18 ± 0.003	16.67 ± 0.67	0.81 ± 0.06	66.67 ± 11.89	323.33 ± 12.02	6.60 ± 0.12
	Control	5.43 ± 0.07	61.29 ± 9.73	1.37 ± 0.07	0.05 ± 0.003	3.03 ± .55	0.16 ± 0.0	0.80 ± 0.0	18.0 ± 0.0	1.77 ± 0.03
Spodosol	Surface	6.43 ± 0.44	47.46 ± 4.46	1.43 ± 0.12	0.05 ± 0.003	2.70 ± 0.10	0.16 ± 0.0	0.80 ± 0.0	27.67 ± 1.76	1.93 ± 0.03
	Mixed	6.63 ± 0.22	54.56 ± 5.70	1.47 ± 0.09	0.05 ± 0.003	3.33 ± 0.48	0.16 ± 0.0	0.80 ± 0.0	34.0 ± 7.09	1.97 ± 0.07
	Control	20.00 ± 0.0	65.93 ± 7.57	2.97 ± 0.03	0.14 ± 0.009	11.0 ± 1.0	1.35 ± 0.48	97.73 ± 81.31	176.67 ± 14.53	15.0 ± 0.58
Mollisol	Surface	20.0 ± 0.0	62.57 ± 1.78	3.10 ± 0.06	0.16 ± 0.003	10.67 ± 0.33	0.99 ± 0.07	58.33 ± 15.45	203.33 ± 3.33	14.67 ± 0.33
	Mixed	20.67 ± 0.33	68.17 ± 5.49	3.10 ± 0.0	0.14 ± 0.003	11.67 ± 0.88	1.47 ± 0.27	98.0 ± 51.26	223.3 ± 3.33	14.0 ± 0.0

Note: Biochar Treatment rate = 25 Mg ha^{-1}. Values represent arithmetic mean ± standard deviation. OM = Organic Matter; CEC = Cation Exchange Capacity. Control = no biochar additions; Surface = biochar top-dressed on soil column; Mixed = biochar incorporated throughout the soil column. Values represent means ± sem (n=6).

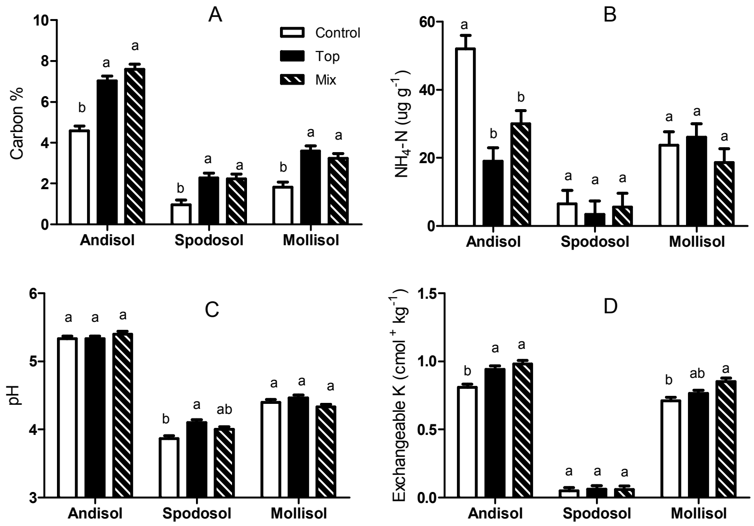

Figure 2.1. Biochar treatment*soil interaction effects observed in total C (A), available NH_4-N (B), pH (C) and exchangeable K (D) at 30 weeks. Letters denote significant differences at *P* <0.05 among treatments within soil type and treatment. Error bars represent the standard error of the mean (n=6).

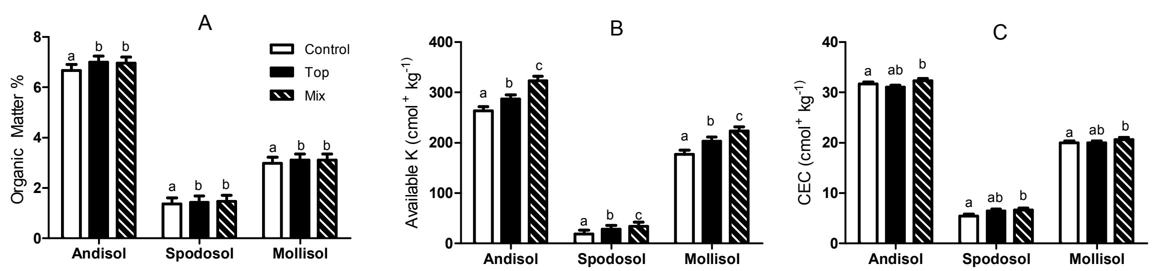

Figure 2.2. Biochar treatment effects for all soil types observed in organic matter (A), available K (B), and CEC (C) at 30 weeks. Letters denote significant differences at *P* <0.05 within soil type. Error bars represent the standard error of the mean (n=6).

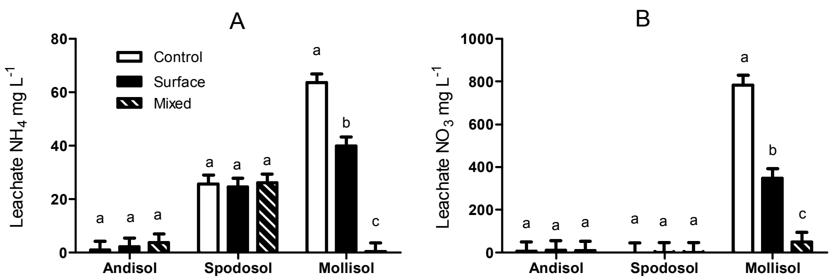

Figure 2.3. Soil leachate response to biochar treatments among various soil types (treatment*soil $P < 0.05$) for (A) NH_4 and (B) NO_3 concentrations in collected at 30 weeks. Letters above bars denote significant differences at $\alpha = 0.05$ among treatments within soil type. Error bars represent the standard error of the mean (n=6).

ACKNOWLEDGEMENTS

We thank the Intermountain Forest Tree Nutrition Cooperative for support and Dynamotive Energy Systems Corporation for biochar donations. Funding was provided by the University of Idaho Sustainability Center independent student research grant and the USDA Forest Service (08-JV-11221633-281). We thank the USDA Rocky Mountain Research Station for laboratory resources and Joanne Tirocke and Derrick Reeves for lab assistance.

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CHAPTER 3: BIOMASS RESPONSE OF POPLAR GROWN IN TWO FOREST SOILS AMENDED WITH BIOCHAR

ABSTRACT

The objective of this study was to investigate the effect of biochar application on Idaho poplar (a cultivar of *Populus trichocarpa* Torr. & Gray) biomass production. An eight-week indoor greenhouse bioassay was conducted using Idaho poplar grown in two forest soils amended with biochar derived from fast-pyrolysis hardwood mill waste. Biochar was applied at rates of 25% and 50% (v/v) to a fine-textured and coarse-textured forest Andisol collected from Idaho and Oregon. After eight weeks of growth, poplar biomass production varied by soil type, but the biochar treatments had no effect on biomass in either soil. There was a non-significant trend of decreasing biomass with increasing char concentration in above-ground biomass in the fine Andisol, and total biomass in the coarse Andisol, which was confirmed to be due to dilution as a similar, yet significant pattern was observed in pots mixed at the same volume ratios with quartz sand. However, when biochar is combined with additions of a complete fertilizer, biomass increased significantly relative to un-fertilized control treatments suggesting improved fertilizer use efficiency or retention. The analysis of leaf tissue revealed a reduction in leaf nitrogen (N) % with both biochar treatment rates. A similar effect was observed in sand-treated pots, suggesting that the amount of nutrients available for plant uptake decreased with either char or sand amendments. The results from this study demonstrate uncertainty associated with biochar effects on tree growth in forests, and potential implications need to be further verified for different soil and plant types.

INTRODUCTION

Biochar can enhance plant growth in a variety of soils by improving soil chemical characteristics (*e.g.,* nutrient retention, nutrient availability), physical characteristics (*e.g.,* bulk density and water holding capacity), and biological properties, leading to increased plant productivity (Glaser et al. 2002; Lehmann and Rondon 2006; Yamato et al. 2006). These positive reports have resulted in increased interest in using biochar as a soil amendment to improve soil quality, however these findings have mostly been demonstrated on soils degraded through agricultural activities, and plants used commonly as crops (Glaser et al. 2002). Limited information is available about the effects of biochar on woody biomass growth, which is needed if biochar soil management is to be implemented in forest ecosystems.

Forests management practices and continuous forest growth generates abundant biomass residues that can be converted to bioenergy and biochar on-site with the use of mobile fast-pyrolysis units (Coleman et al. 2009). The biochar produced could be applied back to the site of biomass extraction to potentially enhance site productivity and build soil C pools (Sohi et al. 2010; Steiner et al. 2007). Biochar applications to soil have been shown to sequester C and enhance soil productivity in temperate and tropical agricultural systems (Laird 2008; Lehmann et al. 2006; Lehmann and Rondon 2006b; Sohi et al. 2010), but have yet to be used extensively on temperate forest soils resulting in uncertainty surrounding biochar amendments after bioenergy extraction.

Understanding potential effects of biochar on forest productivity has been inferred by studying wildfire charcoal that has been modified for decades or centuries by natural soil processes (DeLuca and Aplet 2008; Pietikainen et al. 2000; Zackrisson et al. 1996). However, freshly pyrolyzed biochar differs physically and chemically from wildfire charcoal (Sohi et al. 2010). Thus, while natural forest charcoal can be used to assess the long-term fate of charcoal on soil function, it remains unclear how the unique physical and chemical properties of fresh biochar could affect tree growth in the short-term.

Direct and indirect nutrient properties of biochar are expected to increase plant productivity and growth. Numerous studies have attributed increased plant growth to changes in soil biogeochemistry as a result of biochar additions (Iswaran et al. 1980; Wardle et al. 1998; Hoshi 2001; Lehmann et al. 2003; Chan et al. 2007; Van Zwieten et al. 2007); however, few studies identify or address biochar's effect on woody plant growth, which is needed to support forest-scale application. Both positive and negative effects on soil properties and plant growth have been reported following biochar additions. For example, charcoal from hearths (similar to biochar) was found to decrease tree density and basal area by 40% compared to trees growing in non-hearth areas with limited charcoal presence (Mikan and Abrams 1995). Negative responses are attributed to unfavorable changes in soil properties from the presence of charcoal, which will likely depend on soil type and vegetation present. Conversely, Hoshi (2001) found a 20% increase in volume and a 40% increase in height of tea trees (*Camellia sinensis var. sinensis*) with biochar additions, while Chidumayo (1994) reported better seed germination, shoot heights, and biomass production among native woody plants on soils under charcoal kilns relative to plants growth on undisturbed Zambian Alfisols and Ultisols. These positive biochar responses are associated with a combination of increased soil pH of acidic soils (Chan et al. 2007; Rondon et al. 2007; Yamato et al. 2006), improved physical properties such as water holding capacity (Iswaran et al. 1980), retention of soil nutrients, and reduced leaching losses (Hoshi 2001; Lehmann et al. 2003; Lehmann 2007). These conflicting effects of biochar application by region, soil, and plant type demonstrate the possibility of variation in responses and the need for a greater understanding of all biochar-influenced factors controlling soil quality, plant growth response, and C sequestration potential. A comprehensive understanding of these factors is essential if we plan to implement forest biomass- to- bioenergy systems with biochar application on a large scale in forest ecosystems.

Greenhouse studies and bioassays have been used to investigate potential effects on plant productivity following biochar additions. From these investigations, we know biochar can improve yields and plant nutrient status. However, positive results are commonly reported when combined with fertilizer additions (Blackwell et al. 2009), resulting in uncertainties regarding nutrient supply mechanisms responsible for improvement, and expected results of biochar amendments alone. Furthermore, these studies are often conducted using growing media other than native soil in which plants would naturally be found, making extrapolation of these results to field settings difficult. Forest soils in the Inland and Pacific Northwest are unique because of the influence of recent volcanic activity (McDaniel et al. 2005). To our knowledge, these andic soils have not been investigated regarding biochar amendments.

The objective of this study was to investigate the influence of fast-pyrolysis biochar on biomass production of poplar grown in two native Andisols in a greenhouse bioassay; thus, evaluating the potential effects of biochar on tree growth when added to forests as part of a bioenergy production system. We tested the hypotheses that 1) all poplar grown in biochar-amended soil, regardless of soil type, will have greater biomass production and nutrient status than those grown in unamended soils; 2) the response to biochar is distinct compared to similar amendments with inert quartz sand; and 3) the greatest growth responses are expected in fertilized treatments.

MATERIALS AND METHODS

Biochar and soil

We used fast pyrolysis CQuest™ biochar produced by Dynamotive Energy Systems derived from clean hardwood mill residues (West Lorne Bio Oil Co-Generation L.P. division, West Lorne, Ontario, Canada N1L 2P0). The biochar used in this study had a bulk density of 0.25 Mg m^{-3} (Dynamotive Energy Systems). The biochar was analyzed for available potassium (K) and phosphorus (P) (Gavlak et al. 1994; Peech and English 1944; Murphy and Riley 1962), total C and N (LECO, St. Joseph, MI), cation exchange capacity (CEC) (Chapman 1965), exchangeable calcium (Ca), magnesium (Mg), K and sodium (Na) (Gavlak et al. 1997), available nitrate-nitrate (NO_3-NO_2-N) and ammonium (NH_4-N) (Norwitz and Keliher 1985; Westfall et al. 1993), organic matter (OM) (Sims and Haby 1971; Walkley 1947) and pH (Gavlak 2005). All analyses were conducted at the University of Idaho Analytical Sciences Laboratory, Moscow, ID.

Plant-growing media consisted of two soils that typify Northwest forest soils and differ primarily in texture: (1) a fine-textured forested Andisol from Idaho, and (2) a coarse-textured forested Andisol from Oregon, each collected in August, 2009. The fine-textured Andisol (FA) soil was collected from the upper 20cm of the Bw horizon of a Grandad silt loam, a medial over loamy, amorphic over micaceous, frigid Alfic Udivitrand. This soil was collected near the border of Latah County and Clearwater County, ID, 46° 48' 27" N 116° 19' 36" W. Dominant tree species on the site included western redcedar (*Thuja plicata* Donn ex D. Don), grand fir (*Abies grandis* [Douglas ex D. Don] Lindl.), and Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco) (Cooper et al. 1991). The coarse-textured Andisol (CA) was collected from the Umpqua National Forest, 43° 14' 5.825" N 122° 23' 47.822" W and is classified as a Ashy-pumiceous, glassy Xeric Vitricryand. Dominant tree species on the site were Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco), and lodgepole pine (*Pinus contorta* [Douglas ex.] Louden) (Soil Resource Inventory for the Umpqua NF). After collection, all soils were air-dried, sieved to 2 mm to remove the coarse fraction, and then stored at room temperature until use.

Treatment preparation

To test the impact of biochar on tree growth, the following three treatment blends were prepared with each soil type and replicated 10 times. Amendments consisted of either fast-pyrolysis biochar or sand and treatments were: (1) 0% amendment:100% soil; (2) 25% amendment:75% soil (3) 50% amendment:50% soil. The sand was meant to serve as an inert amendment to test if tree growth response was due to decreased soil volume. Treatments were randomly assigned to 0.5 L Dee cells (Stewe & Sons, Tangent, OR). Soil was mixed for 5 minutes with the amendments of sand or biochar in a cement mixer, then poured into each assigned Dee cell prior to planting poplar cuttings. Prior to adding the treatment, the bottoms of the Dee cells were filled with 30 mL turkey grit to prevent soil and biochar loss.

One-hundred hardwood cuttings of Idaho poplar (a cultivar of *Populus trichocarpa* Torr. & Gray) of approximately equal diameter (~1 cm), and 10.2 cm length were collected from the UI Pitkin Forest Nursery grounds (*lat* 46.725124, *long* -116.956307). Poplar was used as a bioassay because of its responsiveness to variable growing conditions and sensitivity to soil growth media as well as its occurrence in many forest ecosystems. Cuttings were soaked for two days to initiate rooting and then planted in the 0.5 L Dee cells that had already been randomly assigned treatments and watered to saturation. A single application of liquid fertilizer (Miracle-Gro®) was added to 50 cuttings total, 5 from each blend, at the time of planting. Plants were grown with uniform daily watering in a greenhouse with temperatures ranging from 65 – 85 degrees F for eight weeks. The study was conducted at the University of Idaho Pitkin Nursery – Center for Forest Nursery and Seedling Research.

Measurements and harvests

Two harvests were performed on two different occasions to identify growth variations during developmental stages and to evaluate temporal differences in growth response. Cuttings from each treatment were randomly assigned into two groups to be destructively harvested after 4 or 8 weeks of growth. For each harvest, leaves, and stems were clipped from the cutting and remaining soil was rinsed from the roots and cutting using gentle hose pressure. Following the harvest, leaves, roots, and stem were separated and oven-dried at 60^{o}C for 48 hours, then weighed. The initial hardwood cutting tissue was not used in the analysis. Leaf C and N were analyzed using dry combustion at 950 C on a Leco TruSpec CN determinator (St. Joseph, MI, USA).

Statistical Analysis

A general linear model was used to test for significant effects (α=0.05) of soil, treatment type (control, biochar or sand amendment), rate (0, 25, 50%), fertilizer (yes, no) and their interaction on all selected biomass, leachate, and leaf N properties. Soil type (Fine Andisol and Coarse Andisol) was significant, as expected, in each model output; therefore, to identify potential treatment effects, each soil type as analyzed separately and the soil interaction was removed. The foliar N analysis, however, combined both soil types because there was no significant soil interaction. Relative growth rate was analyzed from harvest 1 to harvest 2 (Hunt 1978). The model used tests the interaction between harvest, biochar or amendment, fertilizer, and the combination of amendment and fertilizer. Least-squared means were generated and used to test for significant differences between model variables, followed by the Tukey's post-hoc procedure to test all pairwise comparisons among treatments and soils. All data were evaluated statistically using SAS PROC GLM (SAS Institute Inc, 2008).

RESULTS

Analysis of dry biomass production is presented by soil type, total biomass, above-, and below-ground biomass (Table 2). Results revealed no significant biochar effect on total biomass, above- and below-ground biomass of poplar grown in both the FA and CA, and is reported in detail below. Both biochar treatments did, however, result in decreased leaf N content for poplar grown in both soils. Only second harvest biomass data were used because there were no interactions with harvest time (harvest 1 vs. harvest 2), and there were no changes in relative growth rate among treatments (P >0.05).

Poplar total biomass

FA:

At harvest 2, there was a positive fertilizer effect in the biochar-amended poplar (P=0.02), that resulted in 38.3% greater total biomass for poplar grown in the FA with biochar and fertilizer amendments (Figure 3). Sand amendments did not have the same fertilizer effect (P=0.137). There were no significant main treatment effects (char or sand amendments at any rate) on total biomass for poplar grown in the FA (P>0.05) relative to the control (Figure 1).

CA:

There was no biochar effect on total biomass (P=0.093) of poplar grown in the CA at harvest 2 (Figure 2). There was a fertilizer effect (P=0.0031. Table 2) in the biochar-amended poplar that resulted in 58.5% greater total biomass than unfertilized trees, but no biochar by fertilizer interaction. There was a negative sand effect on poplar total biomass (P=0.023) that resulted in a 40.4% decrease in poplar amended with 25% sand, and a 38% decrease in poplar amended with 50% sand in the CA (Figure 2). Parallel to the response in the FA, there was no beneficial fertilizer effect on sand-amended poplar grown in the CA (P=0.823).

Poplar Above-ground Biomass

FA:

There was a significant biochar fertilizer interaction (P=0.032), which showed an increasing fertilizer response as biochar increased (Figure 3) in FA. Similar to biochar, there was no sand effect on above-ground biomass in the FA (P=0.079), and there was a sand fertilizer interaction (P=0.029). However, in this case there was only a fertilizer response with 25% sand, but not without sand or with 50% sand. This interaction showed a significant difference between the fertilized control and the 25% sand-amended fertilized treatment, but fertilizer had no effect in the unamended control or 50% sand. The fertilized, 25% sand amendment resulted in a 67% increase in poplar aboveground biomass (Figure 3). However, there was no solitary fertilizer effect on above-ground biomass with sand amendments (P=0.324). There were no significant biochar effects at any rate on above-ground biomass (P=0.231, Table 2), but there is a positive fertilizer effect in the biochar-amended soils that resulted in a 39% increase in above-ground biomass relative to unfertilized treatments.

CA:

There was no biochar effect on above-ground biomass (P=0.1165) for poplar grown in the CA (Table 2). However, there was a significant increase in above-ground biomass in fertilized treatments (P=0.015), that resulted in 30.8% greater growth than unfertilized Poplar. This response did not differ among biochar treatments (Biochar x Fertilizer interaction P=0.13). There was a negative sand effect on above-ground biomass (P=0.0148) that resulted in above-ground biomass decreases of greater than 40% for both 25 and 50%

sand-amended poplar. There was no fertilizer effect on the sand-amended treatments (P=0.7209).

Poplar below-ground biomass

FA:

There were no treatment effects, biochar or sand, on belowground biomass (P>0.05) for poplar grown in FA. Additionally, there were no fertilizer effects for biochar-amended or sand-amended poplar (P>0.05).

CA:

In CA, there was no biochar effect on poplar below-ground biomass (P=0.2103), but there was a positive fertilizer effect (P=0.01) in the biochar amended treatments resulting in a 104% increase in below-ground biomass of fertilized biochar-amended poplar. There was no sand effect (P=0.106) on below-ground biomass, and no fertilizer effect (P=0.603) on sand-amended treatments.

Leaf Nitrogen

At harvest 2, there were significant biochar (P=0.001) and sand (P=0.02) treatment effects on leaf Nitrogen (N) content. In both soil types, biochar significantly reduced leaf N content by 19% with amendments of 25%, and by 24% with amendments of 50%. Sand progressively decreased leaf N by 10% with amendments of 25% and by 21% with amendments of 50%, (Figure 4). Only the 50% sand amendment was significantly different from the control.

DISCUSSION

Biochar amendments, as applied in this experiment, did not enhance poplar biomass, thereby refuting our hypothesis. Poplar biomass response to biochar amendments varied by soil type; however, biochar amendments alone did not increase biomass relative to the poplar grown in unamended soil. Additionally, the observed decrease in poplar biomass with the sand amendment suggests that high rates of biochar do not result in a nutrient dilution effect with subsequent reduction in biomass, as demonstrated in the sand-amended treatments. At harvest 2 (eight weeks), there was no main biochar effect on poplar biomass in either soil, yet there was a significant decrease in leaf N content. However, when biochar is combined with fertilizer, there is potential for increased biomass as evident in the biochar fertilizer interactions. This suggests that the physical properties of biochar may lead to increased fertilizer retention and plant growth.

Biochar has the greatest ability to enhance plant growth and nutrient content when combined with fertilizer application (Blackwell et al. 2009). Neutral and negative plant growth responses have been observed with biochar-only amendments, yet when combined with fertilizer additions, crop yields are increased to a much greater extent than with fertilizer additions in the absence of biochar (Asai et al. 2009; Blackwell et al. 2009). Decreased growth is regularly reported with biochar amendments when not associated with fertilizer additions (Asai et al. 2009; Gaskin et al. 2010). For example, Van Zwieten et al. (2009) reported no significant effects of biochar in the absence of fertilizer for certain plant and soil types, while the greatest biomass increase was observed with the application of biochar plus fertilizer. These findings support our results of no positive effect on poplar biomass grown in either biochar-amended soil, but a significant biochar*fertilizer interaction. The reported biochar-fertilizer effects suggest increased fertilizer use efficiency responses or fertilizer retention over the growth period, and could be attributed to the adsorptive capacity of biochar itself (Lehmann et al. 2003) or indirectly associated with decreased soil bulk density, or increased water holding capacity (Iswaran et al. 1980) of biochar-amended soils. Therefore, if increasing plant productivity is an objective, it is recommended that biochar be combined with inorganic or organic fertilizer (Steiner et al. 2007; Yamato et al. 2006). When applied alone these findings suggest that biochar may have little effect on plant growth, but effects will ultimately depend on numerous site factors and interactions.

The impacts of biochar on biomass and plant growth will depend upon site characteristics, soil properties and application rate (Gundale and DeLuca 2007; Asai 2009; Van Zweiten et al. 2009). Our results show varying directions in biomass trends by soil type, with a trend towards biomass reductions with increasing biochar rate in the CA, and a trend towards increasing biomass with biochar in the FA. The FA is derived from airfall pumice deposits resulting in elongated vesicles that have greater surface area and finer capillary function. The CA material vesicated without elongation (J. Archuleta pers comm 2010). Therefore, the negative biomass trends observed in the CA could be a result of the lower water holding and nutrient storage capacity than the FA. Also, the control treatments demonstrated that poplar growth was greatest in the FA. Asai (2009) reported that high rates of biochar reduced plant yield and nutrient concentrations on lower fertility sites compared to higher fertility sites, while Glaser et al (2002) indicated that high rates of biochar did not generally lead to declines in crop yields. Therefore, the optimal amount of applied biochar varies among soil and plant type, and biochar properties (Lehmann et al. 2002). Negative biomass responses associated with high rates of biochar may be ameliorated with fertilizer additions, as seen in the biochar and fertilizer interaction at the 50% rate in the FA. However, this appears to be dependent on soil type, as shown in Asai (2009) where N limitations associated with biochar were not alleviated with fertilizer additions. Ultimately, the effects of biochar on plant growth will depend on the interactions of biochar, soil, and plants that alter nutrient retention, sorption of organic molecules or minerals, pH changes, soil aggregation porosity, and surface oxidation (Major et al. 2009). The limited understanding of these processes and varying results reported in the literature make predicting effects of biochar in the field difficult.

Biochar may have a negative effect on soil N and decrease availability of soil N (Lehman et al. 2002; Asai 2009). While soil N was not measured in our study, we did find significant reductions in leaf N after amendments of both biochar rates, suggesting there was a N limitation after amendments. Decreases in N after biochar additions may result from immobilization as a result of an increased C:N ratio after C-rich biochar additions (Lehmann et al. 2002); however, the total C and N content of biochar does not reflect availability of these elements for immobilizing microbes, and reductions may result from other processes. Reductions could instead be a dilution effect associated with biochar additions. Adding an N-depleted amendment at a high rate could have a notable dilution effect on soil nutrients. This notion is supported by the similar reduction in leaf N observed in the 50% sand amendment. While reductions in leaf N could prove consequential in forests, the high rates used in this study are not realistic in forest-scale application, and the neutral biochar response of poplar biomass despite leaf N reductions suggest there will likely be little to no effect of biochar on tree growth – at least in the short term.

To gain a thorough understanding of biochar effects on forest productivity, long-term field studies are needed. The objective of this greenhouse bioassay was to infer plant growth responses to biochar application in forests, however it is apparent that soil type and application rate may influence how biochar affects soil productivity and could differ by site type and longevity in the soil. Thus, the short duration of this study may not have allotted adequate time to realize the effects of biochar, though trends in the data suggest both positive and negative biomass responses could be anticipated depending on soil type. Furthermore, high rates of biochar could dilute soil N leading to a foliar N limitation, however this would not likely be a concern in forests where reasonable biochar application rates of biochar would range between ~1-10% (v/v). Understanding the factors controlling forest growth responses after biochar additions is critical prior to making recommendations to apply biochar to forest sites.

CONCLUSIONS

The biochar used in this study did not have an effect on biomass for poplar grown in both the FA and CA soil type. The potential for negative impacts are evident by the trending decrease in biomass in the CA, and the observed reductions in leaf N for both soils. However, given the high rates of biochar used in this study, the potential for negative impacts of field application of biochar at field rates is minimal. The potential for negative effects on biomass are dependent upon soil type and appears to be remedied when biochar is combined with fertilizer application, though this practice is unrealistic in forest applications. Nonetheless, the potential for negative impacts suggests careful evaluation of biochar type and soil properties before field scale biochar application.

FIGURES AND TABLES

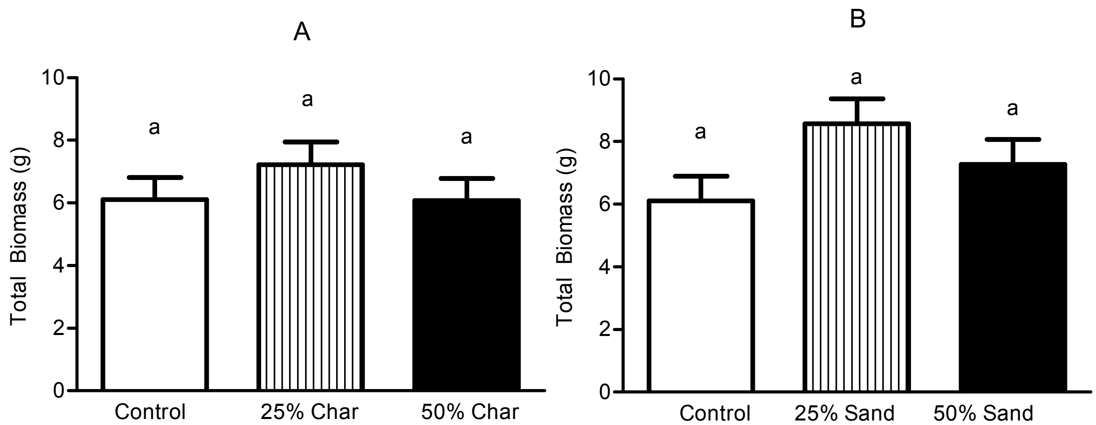

Figure 3.1. Total biomass production (g dry weight) at harvest 2 in response to biochar (A) and sand (B) amendments in the FA. Least Squares means are shown. Columns with the same letter above are not significantly different (P<0.05). n=6.

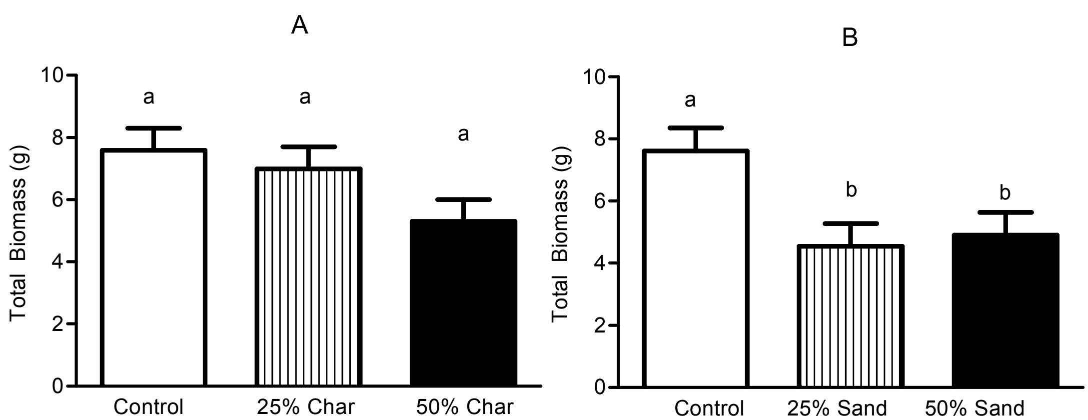

Figure 3.2. Total biomass production (g dry weight) at harvest 2 in response to biochar (A) and sand (B) amendments in the CA. Least Squares means are shown. Columns with the same letter above are not significantly different (P<0.05). n=6.

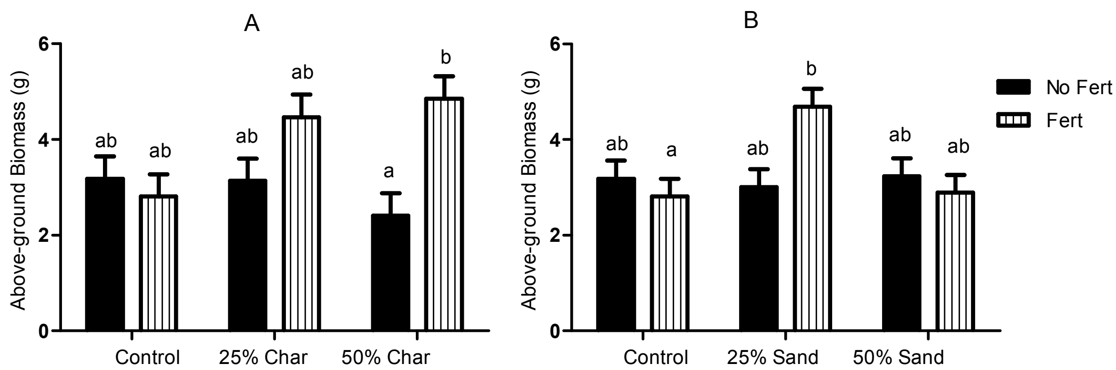

Figure 3.3. Above-ground biomass production (g dry weight) in response to biochar and fertilizer (A) and sand and fertilizer (B) amendments in the FA. Least Squares means are shown. Columns with the same letter above are not significantly different (*P*<0.05). n=6.

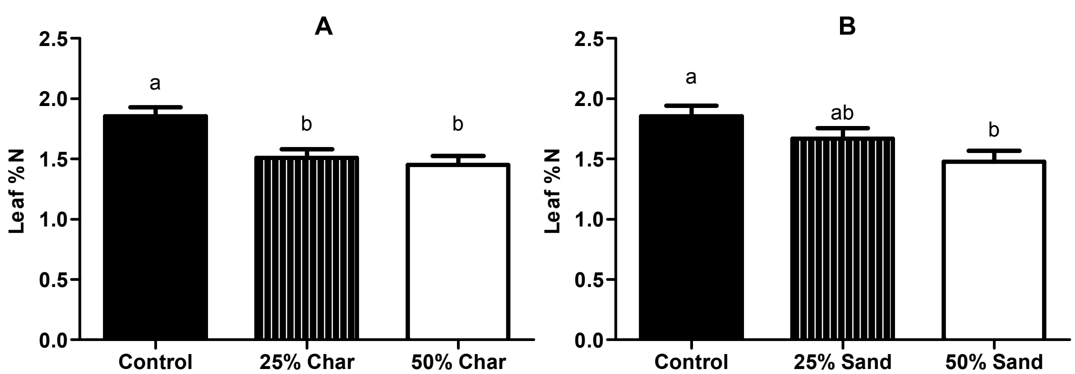

Figure 3.4. Leaf Nitrogen (%) response to biochar (A) and sand (B) amendments for both soils. Least Squares means are shown. Columns with the same letter above are not significantly different (*P*<0.05). n=6.

Table 3.1. Fresh biochar nutrient status from standard fertility analysis.

Test	**Value**
pH	6.8
CEC ($cmol^+ kg^{-1}$)	30
Total C (%)	62
Total N (%)	0.18
K ($cmol^+ kg^{-1}$)	1.6
Ca ($cmol^+ kg^{-1}$)	2.2
Mg ($cmol^+ kg^{-1}$)	0.35
Na ($cmol^+ kg^{-1}$)	0.17
NH_4^+ (µg/g)	3.3
NO_3+NO_2 (µg/g)	< 1.6
Available K (µg/g)	17
Available P (µg/g)	710

Table 3.2. Analysis of Variance degrees of freedom (DF) and *P*-values for biomass response to biochar and sand amendments separated by soil type.

Source	**DF**	**Total Biomass**	**Above-ground Biomass**	**Below-ground Biomass**
		Fine Andisol		
Biochar	2	0.4451	0.2308	0.376
Fert	1	**0.0265**	**0.0115**	0.1243
Biochar*Fert	2	0.1593	**0.0326**	0.5324
		Coarse Andisol		
Biochar	2	**0.0939**	0.1165	0.2103
Fert	1	**0.0031**	**0.0157**	**0.0112**
Biochar*Fert	2	0.2677	0.1315	0.6144
		Fine Andisol		
Sand	2	0.1326	**0.0796**	0.2023
Fert	1	0.1378	0.3239	0.1303
Sand*Fert	2	0.1726	**0.0292**	0.4673
		Coarse Andisol		
Sand	2	**0.023**	**0.0148**	0.1061
Fert	1	0.8233	0.7209	0.4976
Sand*Fert	2	0.7437	0.9223	0.6031

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CHAPTER 4 – CONCLUSION

The preceding chapters identified the potential environmental implications of biochar application in forests when combined with bioenergy production systems by examining the: (a) potential and support for bioenergy and biochar mobile co-production systems in forests, (b) influence of biochar and biochar application method on standard forest soil properties, and (c) effects of biochar and biochar application rate on woody biomass growth (Poplar) grown in forest soils. This chapter briefly summarizes major findings in this thesis and their management and research implications.

SUMMARY OF FINDINGS AND MANAGEMENT IMPLICATIONS

Research on intentional biochar application to soil typically examines 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, it has become so advanced that 'niche' or 'designer' biochars are in production, which are 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. This is not the case for forest systems. Little is known about the 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, especially when combined with bioenergy production.

The second chapter in this thesis examined fundamental questions to address a portion of this extensive knowledge gap. Primarily, we evaluated how biochar could alter forest soil chemical properties on a relatively short time scale, and demonstrated this on various soils collected from Idaho. Results showed that adding biochar to these soils altered nutrient status and C storage. However, the nature and scope of the alterations depended on soil type and biochar application method, which identifies the need for further investigation of the mechanisms affecting these variations. Biochar significantly increased soil C, OM, available and exchangeable K and CEC. Extractable NH_4-N decreased significantly in the forest Andisol after biochar additions, while biochar enhanced nutrient retention in the Mollisol by decreasing N. It is expected that most of these alterations will be short lived as it seems they are direct nutrient additions from the biochar itself, evident from the chemical analysis of the biochar (Table 1, chapter 2). Other studies have found notable nutrient additions with biochar, but these enhancements are reported to be short-lived, declining with plant uptake and leaching (Gaskin et al. 2010; Rondon et al. 2007; Steiner et al. 2007; Topoliantz et al. 2005), and would require continuous biochar re-application—similar to a fertilizer—to maintain these enhancements.

The exception to these short-lived alterations is the demonstrated increased soil C. The significant increases in C of all soils (75% – 79%) suggest biochar could be an effective C sequestration tool for forest managers. While the recalcitrance or decomposition resistance of this specific biochar was not examined in this thesis, it is well supported that biochar is more stable than any other form of soil OM. Biochar contains stabilized plant material with C stored in highly recalcitrant chemical form, making it resistant, but not inert, to abiotic and biotic decomposition once added to the soil. Furthermore, studies suggest a mean residence time for charcoal (a biochar analog) in soil on the order of millennia, compared to 50 years for bulk soil organic matter (Sohi et al. 2010). The potential to sequester C with biochar additions to soils 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. It has even been suggested that with the use of biochar as a GHG mitigation tool, biochar sequestration could exceed current emissions from fossil fuels, providing as a net soil carbon sink (Lehmann et al. 2006). In forests, the mobile fast-pyrolysis units discussed in chapter one of this thesis, could be located throughout a large region of forests. This mobility provides opportunities to reduce hazardous forest biomass while generating biofuels and biochar, thereby creating a greater opportunity to produce carbon neutral biofuels and sequester C with biochar application.

The third chapter paralleled these results by identifying whether forest soil enhancements with biochar could translate to improved forest productivity. It's important to acknowledge that long-term field investigations should be used to ascertain short-, mid-, and long-term effects of biochar on soil nutrient status and forest productivity, but given obvious temporal restraints associated with these type of studies, a greenhouse bioassay was used. The purpose of this chapter was to evaluate how biochar could alter tree growth by using a greenhouse bioassay and field collected soil as a growing media. After eight weeks of growth, biochar did not have a positive effect on poplar biomass production. However, results suggest that there is potential for improved biomass production when biochar amendments are combined with a fertilizer regime, although this is unrealistic in forests. The analysis of poplar leaf tissue showed that biochar significantly reduced leaf N content, suggesting there is potential for negative consequences associated with biochar amendments. However, these reductions are likely a result of nutrient dilutions associate with additions of an N-depleted amendment to soil.

The rate at which biochar would be applied to forests is much lower than the rates used in both the lab (25 Mg ha $^{-1}$) and greenhouse (25% and 50% v/v) chapters of this thesis. For example, when combined with forest bioenergy production systems, biochar would probably be added at a rate equivalent to the amount of biochar generated from biomass extracted from the site, which could realistically range from 2 - 6 Mg ha^{-1}, but it is dependent upon the site and forest biomass levels. Because biochar had a neutral main effect on poplar growth even when applied at high rate; it is unlikely that adding biochar to a forest at a lower rate would have any effect on tree growth, at least in the short-term. Additionally, from chapter two we understand that the biochar may contribute to notable short-term soil nutrient enhancements associated with the nutrient value of the biochar itself, or cause a reduction in ammonium; yet, these effects do not appear to translate into increased tree growth (chapter three), and will likely have little to no effect on forest soil in the short-term if applied at rates equal to, or lower than the those used in this research.

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. The biochar byproduct produced can be redistributed onto biomass extraction sites, but is expected to primarily build the recalcitrant soil carbon pool thereby sequestering carbon. Adding biochar could have long-term effects on soil and forest productivity not elucidated in this thesis, but it is likely that low-rate biochar additions will have neutral effects.

Two collaborative, long-term forest field studies have been installed in Oregon and Montana to address both short- and long-term effects of biochar application in a highly variable natural setting. Results from this thesis can be compared to emerging results from these field studies where individual tree plots received varying rates of biochar using broadcast applications. Changes in soil properties and tree growth response can be evaluated to provide realistic temporal- and spatial-scale evaluation of forest responses to land applications. Demonstrating parallel results among multi-scale approaches 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

This thesis improved understanding and advanced measurement of biochar application in temperate forests. Results provided a basic understanding of the potential for biochar in our region, and offered several primary implications for biochar management that could inform a comprehensive plan for continuing forest bioenergy production systems. Forest systems are highly variable, therefore small-scale lab and greenhouse studies, such as the two presented here, may not include important ecosystem components that could influence biochar interactions. Therefore, extrapolation of these results to a field-scale may not be appropriate depending on conditions, and should be qualified with additional, parallel studies. Nonetheless, these results can be used to gain a better understanding of processes of biochar in soils of the Inland Northwest to determine optimal biochar application rates in association with mobile fast-pyrolysis bioenergy production systems. These findings could also be useful for other regions where biochar is proposed as an amendment for forest soils, and can be compared to ongoing field studies. In summary, these findings can facilitate additional research to be applied to understanding the short- and long-term effects of biochar on the impacts on forest soil productivity. Further research is needed to provide a comprehensive assessment of the site improvement and C sequestration potential of biochar combined with forest bioenergy production using mobile fast-pyrolysis units.

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