Use of Biochar in Organic Farming

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Chapter Overview

There are currently relatively few studies on the use of biochar in organic farming systems, yet there is much that can be learned from historical use charcoal in agriculture and contemporary research in conventional agriculture. From the citrus fields of Japan to basket willow stands of north Great Britain to the famous *Terra Preta* soils of Amazon Basin, farmers have used biochar, the practice of burying charcoal in soil to improve fertility and tilth for centuries. Biochar has recently had a revival in modern agriculture with this carbon (C)-rich material being widely used as a means of improving soil tilth and promote a more sustainable agriculture. The purpose of this chapter is to briefly describe the nature and properties of biochar and its potential impact on the fertility and function of soils following incorporation with an emphasis on organic agriculture. We briefly review biochar generation and limitations associated with centralized production and distribution. We then discuss in detail the influence of biochar application on soil properties and crop production using organic examples where possible. Finally, we discuss the specific use of biochar in organic farming systems and highlight the San Juan Island experience wherein replicated studies were conducted on ten independent organic farms to assess the influence of locally produced wood biochar on soil properties and processes and crop productivity on the San Juan Islands, WA, USA.

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Introduction: Biochar History and Use in Agricultural Systems

Biochar is a carbon (C)-rich, stable solid material that is generated from the pyrolysis or thermochemical decomposition of organic material in an oxygen-limited environment under controlled condition, and it differs from charcoal generated during wildfires (DeLuca and Aplet 2008) or that produced for fuel as biochar is specifically generated for use as a soil amendment, while charcoal is commonly produced as an energy carrier (Lehmann and Joseph 2015). Biochar can be made from a variety of materials including forest or crop residues, municipal solid waste, or biosolids (Brown et al. 2015). The C-rich nature of biochar combined with its unique resistance to decomposition has resulted in it being discussed as a means of abating climate change by sequestering C when applied to soils (Lehmann et al. 2006). Besides, the morphological characteristics of biochar might also alter soil hydrological properties and subsequently affect soil nutrient transformations (DeLuca et al. 2015b). It has therefore become a topic of unique interest in soil science (Atkinson et al. 2010), and the numbers of papers published annually on the subject have increased exponentially over the last 20 years (Gao and DeLuca 2016).

Despite the fact that the term "biochar" was introduced only recently, the original idea for using charcoal in agriculture dates back thousands of years. The "Amazon Dark Earth" or *Terra Preta* soils found in the Amazon River Basin was reported to have been established by aboriginal cultures thousands of years ago yet remain some of the most fertile and high biodiverse soils in the Amazon today. The origin of *Terra Preta* remains unclear but was ascribed to the large proportion of char that remains in these soils makes it unlikely that it was a product of biomass burning (slash-and-burn farming), but it is not clear whether the "biochar application" was intentional (Glaser and Birk 2012) or a means of sanitary waste management in populated areas of the Amazon basin. Olarieta et al. (2011) indicated that an ancient method named "formiguer," the structure of which is somehow similar to a charcoal kiln, was largely used in the Mediterranean region to produce "soil-fertilizing material" with dried woody vegetation up to the 1960s. Pioneering work on the agricultural use of biochar in combination with composting techniques was shown to have been performed by farmers in Japan since early twentieth century (Ogawa and Okimori 2010). Farmers would use rice husks and other farming residues to produce charcoal using traditional earthen kilns and use them largely as soil improvers or odor absorbents (Nishio 1996). However, in-depth investigation of the beneficial effects of biochar on agricultural soils received little attention by Japanese scientists until the early 1980s (Saito 1990).

As noted above, the number of papers addressing the use of biochar in agricultural ecosystems have increased dramatically, with the focus largely being soil C storage and sequestration (Lehmann et al. 2006), management of greenhouse gas emissions (He et al. 2017), soil fertility and nutrient management (Nguyen et al.

2017), and crop productivity (Jones et al. 2012; Griffin et al. 2017). Given the broad interest in achieving more sustainable agricultural ecosystems while maintaining food security, there is increasing interest in understanding how biochar application fits into this framework, particularly for organic farming systems that rely on natural soil amendments and seek to minimize environmental impacts (Wezel et al. 2014; Reganold and Wachter 2016). Herein we describe the nature and properties of biochar, its potential impact on the fertility and function of soils following incorporation, and highlight recent research using biochar in on-farm organic field trials.

Biochar Generation and Properties

Biochar Generation

Charcoal production through wood carbonization has been practiced for thousands of years; however, the ancient method for producing *Terra Preta* by earthen-pit burning may have released a large amount of greenhouse gases and volatiles back into the atmosphere (Brown et al. 2015). Modern biochar production involves some form of pyrolysis, a thermal-chemical conversion process, of agricultural or forestry biomass residues. A variety of carbonization technologies associated with pyrolysis or gasification reactors have been developed to pyrolyze organic material and produce biochar, and this production can be done on either large or small scale (Boateng et al. 2015).

Large-scale centralized biochar generation typically involves reactors that can process 2000 metric tons of dry biomass per day, either through pyrolysis under relatively low heating rate (approximately 100 °C min^{-1}), namely, slow pyrolysis, or high heating rate (on the order of several hundred ºC s^{-1}) such as fast pyrolysis or gasification (Wright et al. 2010; Verma et al. 2012). Slow pyrolysis reactors can be further classified as kilns or retorts where kilns are typically used in traditional charcoal making without recovering the subsequent liquid fractions, whereas retorts capture gaseous and liquid fractions during pyrolysis process (Boateng et al. 2015). Fast pyrolysis or gasification typically has lower percentage of biochar yield (15–20%) compared to slow pyrolysis (20–50%) where those reactors are intended to maximize the production of high-value energy product (bio-oil or syngas) with biochar as a by-product. Although large-scale centralized pyrolysis systems have higher efficiency in processing agricultural or forestry residues, long-haul distances can more than double the break-even price of biochar (Schackley et al. 2015) reducing attractiveness of biochar to agricultural operations. Further, monetizing the value of biochar applications to agricultural operations is challenging given the variable and perhaps long-term benefits of biochar to landowners or land managers.

Therefore, distributed biochar production by low-tech pyrolysis kilns or simple mobile units may increase the attractiveness of biochar to farms that generate small quantities of biochar using local resources.

Typically, small-scale systems for biochar production utilize slow pyrolysis which involves longer processing time yet higher yields of biochar (Odesola and Owoseni 2011). Such systems can process 0.5–1 metric ton of biomass per hour and can be distributed on small properties and operated on farms (Nsamba et al. 2015). Schmidt and Taylor (2014) described a "Kon-Tiki" method which follows the principle of pyrolyzing biomass layer after layer in an open, conically built metal kiln (Schmidt and Taylor 2014). Briefly, a fire is started in the kiln to burn the first layer of biomass into embers on the bottom of the kiln; a thin layer is then added on top of the embers and being heated quickly to be carbonized. When ash starts to appear and the fire becomes hot, the next layer of biomass is homogenously spread on top. Energy from both the flames above and the layer below will start to pyrolyze fresh biomass. The manual layering of biomass is repeated until the kiln is filled, and the reaction is stopped by quenching with water or a layer of soil on top. The generated biochar below the upper pyrolyzing layer is shielded from oxygen flow and thus oxidation. Syngas generated during the process will simultaneously react with combustion air entering from the top of the kiln, producing heat and partially self-sustaining the system. This fast, easy-to-operate biochar production method has been reported to be low in greenhouse gas emissions and can produce roughly 750–850 L of biochar within 4–5 h (Schmidt and Taylor 2014). It has been continuously improved and widely used in many small-scale farming operations (Cornelissen et al. 2016; Gao et al. 2016, 2017; Pandit et al. 2017; Hagemann et al. 2018).

Physical and Structural Properties of Biochar

Various feedstock types combined with a diverse range of pyrolysis conditions can strongly influence the structure and physical properties of a biochar (Zhao et al. 2013). Scanning electron microscope (SEM) images (Fig. 1) have revealed that the pore structure of a biochar will generally represent the cellular structure of its feedstock (Lee et al. 2013). On the other hand, as the highest treatment temperature (HTT) of biochar increases, the biochar exhibits a greater percentage of crystallinity, where the percentage of aromatic C is increased and the entire structure of the biochar becomes more graphitic (Chia et al. 2015).

Typically, with the increased ordering of turbostratic aromatic C sheets, the interplanar distances of aromatic C forms will decrease, creating high surface area per total volume of a biochar (Lehmann et al. 2011). Coarse sand typically has very low surface area (0.01 $m^2\ g^{-1}$), whereas clay can have exceptionally high surface areas (100–1000 $m^2\ g^{-1}$) (Heilman et al. 1965). Biochar has been widely reported to have similar or higher surface area than clays, for example, biochar produced from Douglas-fir wood by fast pyrolysis at 900–1000 °C was reported to have a surface

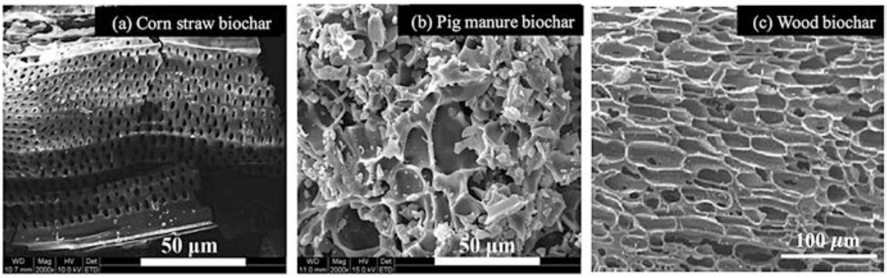

Fig. 1 Scanning electron microscopy images of biochar produced from different feedstocks: (**a**) corn straw, (**b**) pig manure, and (**c**) wood (Source of (**a**) and (**b**): Wang et al. 2017b, Article link (open access): https://www.nature.com/articles/s41598-017-12503-3. License: Creative Commons Attribution 4.0 International License https://creativecommons.org/licenses/by/4.0/. Reprinted with permission; source of (**c**): Jaafar et al. 2014, Article link: http://www.sciencedirect.com/science/article/pii/S2095311913607030?via%3Dihub. License: under Copyright's Clearance Center's Rightslink service. Reprinted with permission)

area of 745 m^2 g^{-1} by the N_2 BET method (Karunanayake et al. 2017). The micropore (diameter less than 2 nm) density of a biochar has contributed to this high surface area, leading to higher adsorptive capacities and hydrophobic effect potentials (Yang et al. 2018). Biochar can be used to remediate contaminated agricultural soils through the adsorption of heavy metals (Lu et al. 2017) and organic pollutants such as polycyclic aromatic hydrocarbons (PAHs) (Cao et al. 2016) or pesticides (Jones et al. 2011a). Biochar also has the potential to absorb some organic molecules that are involved in chelation, forming organo-mineral-biochar complexes which can potentially aid soil soluble P availability in organic farming systems (DeLuca et al. 2015b).

Macromolecular Properties of Biochar

Specific chemical changes occur in biomass when it is heated in an environment lacking electron acceptor such as oxygen (Kleber et al. 2015). Biochar generation starts with water loss at low heating temperatures, but as temperature increases, molecules such as lignin, cellulose, and hemicellulose are lost, and amorphous C begins to form. With further heating, turbostratic crystallites start to form as aromatic rings begin to condense and grow into sheets (Keiluweit et al. 2010). Eventually, feedstock biomass C is "compressed" into new solid phases with higher proportions of C, and some amount of its original C lost as volatiles during the heating process. It has been proposed that the nature of these C structures formed during the heating processes is a primary reason for biochar's high stability in soils (Nguyen et al. 2010; Lehmann et al. 2011). Although degradation of some labile components of biochar may occur in soils, soil microorganisms will generally be less likely to utilize these aromatic C compounds as an energy source, potentially contributing to the biochemical stability of biochar (Wang et al. 2016). Recent studies have also demonstrated the chemical stability and thermal stability of biochar (Chen et al. 2016a; Conti et al. 2016; Suárez-Abelenda et al. 2017).

Influence of Biochar on Soil Properties

Soil Physical Properties

The addition of biochar to agricultural soils can lead to unique interactions that influence soil physical properties including changes in soil porosity, water holding capacity (WHC), bulk density, aggregation, and drainage (Lehmann and Joseph 2015). As mentioned above, biochar is highly porous and possesses great surface area, thereby enhancing the total surface area, porosity, and water- or nutrient-holding capacities when added to soil. Głąb et al. (2016) demonstrated that the application of winter wheat straw biochar significantly improved the total porosity of a sandy agricultural soil, with the most volume increment increase being in small pores (less than 50 μm in diameter). The changes in soil porosity were also reflected in the water retention properties of the investigated soil with the finer biochar particles causing a greater increase in soil WHC. Similarly, Liu et al. (2017a) reported a 17% increase of soil porosity and a 28% increase in soil WHC of a silt loam agricultural soil following maize biochar application. Biochar was also reported to increase the retention of water at field capacity by 1.3% in an organically managed loamy soil (Ulyett et al. 2014).

Soil aggregation determines the soil pore network and thus contributes to root elongation, water infiltration, aeration, drainage, and diffusion of nutrients. Wang et al. (2017a) reported a significant improvement of wet aggregate stability in a silt loam agricultural soil following application of either walnut shell or softwood biochar with a 126% and 217% average increase of mean weight diameter observed for walnut shell and softwood biochar treatments, respectively. Du et al. (2017) also observed an increase in the stability of soil macroaggregates with increasing biochar doses. Biochar additions to soil generally lead to the creation of aggregate bridges and large void spaces, therefore potentially reducing soil bulk density, which describes the mass of soil per unit volume (Jones et al. 2011b; Agegnehu et al. 2016a). The reduced bulk density mediated by biochar could further alleviate soil compaction stress and possibly transition into promotion of crop growth (Liu et al. 2017a). Although there is some variation in the literature, soil physical properties are generally improved with the addition of biochar to agricultural soils.

Soil Biochemical Properties

In organic farming systems, soil fertility and plant production depend on the mineralization of nutrients from plant and animal residues, soil minerals, and resident soil organic matter involving a variety of soil biochemical processes (Mäder et al. 2002). It is therefore essential to understand how biochar applications influence soil biochemical properties and processes including C storage, soil nutrient capital and cycling, and microbial and associated enzyme activities. The use of biochar generated from local feedstocks in organic farming systems has been reported to either increase or have no significant impact on soil nutrient availability (Arif et al. 2016; Cavoski et al. 2016; Usman et al. 2016); and the mechanisms behind these shifts have been argued as both abiotic (such as adsorption or desorption of nutrients) or biotic factors associated with nutrient transformation processes, particularly N cycling (Nguyen et al. 2017). Gao et al. (2016) demonstrated that locally produced wood biochar had the ability to enhance the availability of soil NH_4^+-N in agricultural sandy soils of an organically managed system when applied alone or in combination with an organic fertilizer. The enhanced N availability was potentially due to increased adsorption capacity associated with the wood biochar as well as increased N mineralization rates following biochar application (Gao et al. 2016). With similar rates of biochar application in the following year, the authors subsequently detected a significant increase in both soil available inorganic P (citrate-extractable P) and potentially available organic P (enzyme-extractable P) pools five months after biochar amendment at six organic farms (Gao et al. 2017). Similarly, in a field study, Agegnehu et al. (2016b) reported that an increment of soil exchangeable cations including K, Na, Ca, and Mg following the addition of *Acacia* spp. produced biochar on an organic barley field, and they attributed these nutrient alterations to the direct effect of biochar on soil cation exchange capacity (CEC) due to its various surface charges and high surface area. By contrast, Sánchez-García et al. (2016) reported that biochar applied alone did not alter soil mineral N content in a two consecutive year of field study with organic olive crop growing in a calcareous arid land.

Nitrogen is considered as one of the most limiting nutrients in temperate agroecosystems; hence its transformation following biochar application to soils has been widely investigated for the last 10 years. Relatively thorough reviews of biochar influence on nutrient cycling can be found elsewhere (DeLuca et al. 2015b; Gao and DeLuca 2016; Gul and Whalen 2016; Nguyen et al. 2017). The influence of biochar or natural charcoal on N cycling and specifically nitrification appears to be more pronounced in forest soils than in N-amended agricultural soils. In forest soils, charcoal presence appears to stimulate net nitrification potentially as a result of charcoal adsorption of phenolics or terpenes that otherwise may interfere with this process (DeLuca et al. 2006). In organic agricultural soils, several recent studies also demonstrated increased nitrification following biochar addition which might be explained by a stimulated nitrifier activity due to an alteration in soil moisture and aeration (Ulyett et al. 2014; Pereira et al. 2015). For acidic agricultural soils, biochar-induced pH rise might also accelerate nitrification process (Teutscherova et al. 2017a). Nitrogen mineralization, the process by which organic N is converted to inorganic forms, was reported to increase in response to a ryegrass biochar application at the first week and decrease over time (Maestrini et al. 2014). It has been suggested that the short-term enhanced soil N mineralization rates with biochar addition to soil might be related to the H/C ratio of the biochar, where a higher ratio represents less recalcitrant biochar which is more likely to be decomposed and thereby release N trapped in the char into the mineral pool (Mukherjee and Zimmerman 2013; Pereira et al. 2015). Alternatively, the biochar additions may adsorb organic compounds associated with litter decomposition thereby enhancing net N mineralization (DeLuca et al. 2015b). Regardless of the mechanism, accelerated N mineralization with biochar addition would be particularly beneficial for organic farming systems as they tend to be challenged by a slower mineral-N release from the decomposition of organic material throughout the season when compared to conventional farming. On the other hand, biochar can affect soil N losses via denitrification process or direct leaching, both of which are commonly found to be reduced when biochar presents (Gao et al. 2016; Pereira et al. 2017). Biochar-mediated reductions in N_2O emissions can possibly be explained by changes in a variety of factors including soil pH, aeration, and substrate availability such as organic C or inorganic N (Gao and DeLuca 2016), while biochar could reduce N leaching by altering soil physical properties, or via altering total soil cation-exchange capacity and increasing NH_4^+ retention in surface soils (DeLuca et al. 2015b). Biochar additions to agricultural soils is often cited as means of increasing total C storage in soils (Lehmann et al. 2006). As mentioned above, a large proportion of biochar on a mass basis is aromatic C which tends to be resistant to microbial decomposition. When biochar is mixed into soil, this portion of C in biochar can immediately enhance soil total organic C content and, due to the resistance of biochar to decomposition, subsequently contribute to the long-term C storage and sequestration (Lehmann et al. 2011). Conventional agricultural systems tend to have reduced soil organic C content compared to forest soils as their topsoils have been constantly disturbed (van Wesemael et al. 2010); therefore, biochar amendment might provide a beneficial yet low-cost means of retaining more organic C into soil sink (Lehmann et al. 2006). While organic farming systems are having relatively higher organic C content than conventional farming (Gattinger et al. 2012), biochar addition was demonstrated to significantly contribute more to this C pool in a couple of studies associated with organic farming (Schulz et al. 2013; Sánchez-García et al. 2016).

Most soil nutrient transformations are enzyme-mediated reactions, and many of these have been found to be influenced by biochar additions to soil (Thies et al. 2015). Unfortunately, few of these studies have specifically been conducted in association with organic farming systems (Gao et al. 2017). Soil enzyme activity in response to biochar addition largely depends on the alterations in the interaction of substrate and enzyme through sorption and desorption, which subsequently is related to substrate availability. β-Glucosidase, an enzyme involved in cellulose degradation process, was shown to generally be nonresponsive or respond negatively with biochar addition on agricultural soils (Wu et al. 2013; Abujabhah et al. 2016), whereas peroxidase which is involved in the degradation of recalcitrant C forms in soil was positively responsive to char addition (Ng et al. 2014; García-Delgado et al. 2015). This trend potentially reflects or could be explained by the dominance of persistent forms of C in char-amended soils that would or would not be preferred substrates for specific enzymes (Chen et al. 2013). An opposite trend for β-glucosidase activity occurs in studies where the biochar used in the study temporarily contributed labile C (Al Marzooqi and Yousef 2017; Gao et al. 2017). Soil enzymes associated with N or P mineralization (urease, amidases, phosphatase, etc.) have been reported to generally respond neutrally or positively to biochar additions (Gao et al. 2017; Huang et al. 2017; Liu et al. 2017b; Teutscherova et al. 2017b). Enzyme response to biochar partly depends on how the enzyme active site or substrate interacts with biochar and its local chemical environment (Thies et al. 2015), yet it is important to note that enzyme activity does not always directly dictate microbial activity, and a considerable amount of activity detected in biochar amended soils may be from enzymes stabilized in soil matrix that are no longer associated with viable cells (Nannipieri et al. 2018). Overall, organic farming systems tend to have less readily available inorganic forms of nutrients that compared to that in conventional farming systems. Therefore, it is likely that biochar would play a potentially more important role in nutrient turnover and availability in organic farming systems.

Soil Microorganisms

Soil microorganisms play an integral role in virtually all soil processes, such that microbial abundance, activity, and composition will largely determine sustainable productivity of agricultural land (Paul 2014). Studies examining soil biota following biochar addition to agricultural soils are relatively abundant (Lehmann et al. 2011), yet little attention has been paid to this response within organic cropping systems (Gao et al. 2017; Gao and DeLuca 2018). Soil microbial communities can be influenced by biochar through several mechanisms: (1) the biochar itself could serve as a habitat or surface for soil microorganisms (Quilliam et al. 2013; Jiang et al. 2016); (2) biochar can serve as a substrate or loci of substrate accumulation for microbial consumption (Lehmann et al. 2011; Quilliam et al. 2013); (3) biochar can adsorb soil toxins and chemical signals that will otherwise inhibit microbial growth (Kasozi et al. 2010); and (4) biochar can alter the abundance of soil microorganisms through changing abiotic factors such as moisture, pH, or the concentration of specific elements or compounds possibly via adsorption (DeLuca et al. 2015b; Pingree and DeLuca 2017; Yu et al. 2018). For instance, Dumontet et al. (2017) observed evaluated biochemical and microbial activity in biochar-amended soils with or without organic fertilizer additions, and their results showed that both treatments had higher C oxidizing potential and greater diversity of cellulose-degrading bacteria than the control, suggesting a positive biochar effect in microbial heterotrophic metabolism possibly through inputs of C substrate. Similarly, Teutscherova et al. (2017a) recorded higher microbial activity and subsequent enhanced N mineralization rates following biochar addition to a degraded acidic soil in a microcosm experiment. The authors attributed this finding to the biochar alteration of the soil microenvironment, where biochar addition resulted in a significant increase in soil pH throughout the incubation period.

A number of studies in recent years have investigated the abundance and diversity of soil microbial populations in biochar-amended soils with respect to soil bacteria and archaea, fungi, and fauna (Abujabhah et al. 2017; Lucheta et al. 2017; Teutscherova et al. 2017b). Results of these studies vary widely, and the differences in these responses are likely related to interactions between biochar and microenvironmental factors including soil pH and soil moisture content. Most studies have reported no significant change or slight decrease in microbial abundance in biochar-treated soils (Quilliam et al. 2013; Gao et al. 2016). Recently, Teutscherova et al. (2017b) reported a decrease of microbial biomass in biochar-treated soils in a short-term incubation study using the substrate-induced respiration (SIR) method and attributed this decrease to the biochar-induced shift in soil pH which altered the balance between fungal and bacterial biomass. A similar argument was forwarded by Yao et al. (2017) where the authors detected higher soil fungal abundance (compared to bacteria or archaea) following 3 years of biochar addition by using quantitative PCR. Lucheta et al. (2017) used high-throughput DNA sequencing to observe elevated fungal abundance and richness in Amazon Dark Earth compared to unamended surrounding soils. These Amazonian dark earth soils are characterized by high levels of charred black carbon (Lucheta et al. 2017). As noted above, the porous physical structure of biochar and its high surface area can potentially contribute to water retention and the sorption of soil organic molecules, making it suitable for fungal colonization both internally and externally (Thies et al. 2015). However, an opposite trend has been observed where bacterial abundance was significantly increased by 28% with the application of 20 t biochar ha^{-1}, while fungal abundance decreased by 35% in a rice paddy soil (Chen et al. 2013). It was speculated that the neutral soil pH was unresponsive to biochar addition, therefore favoring a diverse bacterial community compared to acid soils that would likely be preferred by fungi (Fierer and Jackson 2006; Rousk et al. 2009).

Given that biochar may induce changes in microbial biomass (Gao et al. 2019), such overall changes in abundance will likely to cause some microbial groups to become more dominant and thus lead shifts in community structure of microorganisms (Lehmann et al. 2011). Studies associated with the influence of biochar on soil bacterial, fungal, or faunal diversity have also demonstrated varied results. Soil bacterial diversity was generally found to decrease or have no change in short-term studies (Imparato et al. 2016; Song et al. 2017) but generally increase in long-term studies and in the Terra Preta soils (O'Neill et al. 2009; Zheng et al. 2016; Abujabhah et al. 2017). The labile substances in biochar may stimulate activity (Jones et al. 2012) and induce shifts in microbial communities (Lehmann et al. 2011); however these resources are quickly mineralized in and present a transient effect, whereas long-term effect of biochar on soil microbial communities are likely achieved by multiple direct and indirect mechanisms and related to physicochemical and biochemical properties (Gul et al. 2015). In a long-term study examining microbial community structure following corncob biochar additions to a soybean-cultivated agricultural soil, researchers detected greater activity and diversity of bacteria in biochar-treated soils compared to the control, where the bacterial communities shifted from preferring metabolizing carbohydrates to xenobiotics (Sun et al. 2016). On the phylum level, the relative abundance of *Proteobacteria* and *Actinobacteria* increases with biochar amendment, while that of *Acidobacteria* decreased (Ahmad et al. 2016; Xu et al. 2016), and the overall shift was attributed to the high dissolved organic C present in biochar (Ahmad et al. 2016; Sun et al. 2016; Xu et al. 2016). On the other hand, fungal diversity exhibited very different responses to biochar application across various functional types and study conditions (Chen et al. 2016b; Lucheta et al. 2017; Yao et al. 2017). Using the phospholipid fatty acid (PLFA) technique, Luo et al. (2017) found that the proportion of arbuscular mycorrhizal fungi and the ratio of arbuscular mycorrhizal fungi/saprotrophic fungi were both enhanced by biochar addition and were correlated with biochar application rates (Luo et al. 2017). On a phylum level, biochar has been reported to increase the relative abundance of the *Basidiomycota* with high fungal diversity index observed in biochar-amended soils (Awasthi et al. 2017). More commonly fungal diversity observed in long-term studies was found to be unchanged although fungal community structure found to be significantly correlated with soil total C, N, or K (Dai et al. 2016; Lucheta et al. 2016; Yao et al. 2017).

The addition of biochar to soils also appears to influence the relative abundance of soil fauna, with a focus on earthworms (Bamminger et al. 2014; Kamau et al. 2017; Pingree et al. 2017). In fact, nearly all biochar-mediated changes in soil properties could directly or indirectly influence the soil faunal community (Sauvadet et al. 2016). Biochar generally directly affects soil faunal communities by improving habitat or indirectly through the biochar-mediated alterations at the lower trophic levels within the soil food web, such as shifts in the abundance of fungi and bacteria (Paz-Ferreiro et al. 2015). A recent study demonstrated that the abundance of earthworms in soil was not only related to soil charcoal content but to the nature of the biochar feedstock (Kamau et al. 2017). And in a short-term microcosm study, Pingree et al. (2017) reported a significantly greater biologically available P pool in both biochar-treated and biochar- and earthworm-treated soils, suggesting an interactive effect of biochar and earthworms in mediating soil P cycling. Earthworms have also been demonstrated to directly ingest biochar particles and thus could contribute to the stability or decomposition of biochar in soil (Lehmann et al. 2011).

Influence of Biochar on Crop Productivity in Organic Agriculture

Organic farming aims at creating a closed nutrient cycle on the farm to produce food with no soluble mineral or synthetic pesticide inputs and minimal harm to ecosystems (Mäder et al. 2002). However, critics argue that agriculture based on these principles typically result in relatively lower yields compared to conventional farming systems (Seufert et al. 2012). Therefore, while the goal of organic operations also includes building soil fertility over time, one must explore effective crop and nutrient management practice including initiating biochar amendments to surface soils.

Although a large number of studies in recent years have examined the influence of biochar on crop nutrient uptake and yield (see Lehmann and Joseph 2015), few have focused on its use in organic farming systems and especially associated with field studies (Table 1). Broadly speaking, aboveground production and yield have been widely reported to increase in biochar-treated agricultural soils (Biederman and Harpole 2013), and the response of crop to biochar addition primarily depends

Table 1 Recent biochar studies associated with organic farming systems

Study	Study type	Study period	Study focus and details
Dumontet et al. (2017)	Field (1 farm)	Two months	Metabolic and genetic patterns of soil microbial communities following olive mill waste biochar (commercial) and compost amendments on an organic farm
Gao et al. (2016)	Field (10 farms)	One growing season	Wood biochar (80% Douglas fir, locally produced on-site) amendment on soil nutrient availability (particularly N, P), retention, and dry beans nutrient uptake
Gao et al. (2017)	Field (6 farms)	One growing season	Wood biochar (80% Douglas fir locally produced on-site) amendment on soil nutrient availability (particularly N, P), and winter squash yield and nutrient uptake
Pereira et al. (2015)	Greenhouse mesocosm	42 days	Effect of different types of biochar (Douglas fir, pine, or hog waste wood produced) on soil N transformations (with molecular and stable isotope techniques) and lettuce growth performance
Pereira et al. (2016)	Field (1 farm)	One growing season	Walnut shell biochar (locally produced on-site) amendment on CO_2 abatements and emissions on an organic walnut farm
Pereira et al. (2017)	Greenhouse mesocosm	Two growing seasons	Pine chip and walnut shell biochar (commercial) with organic N fertilizer on soil N leachate, N_2O emission, and plant N uptake
Sánchez-García et al. (2016)	Field (1 farm)	Two years	Oak biochar (commercial) and compost amendments on soil C buildup, N dynamics, and plant nutritional status in a drip-irrigated organic olive crop
Ulyett et al. (2014)	Field (2 farms)	Two months	Deciduous mixed wood biochar (commercial) amendment on water retention and nitrification processes in sandy loam soils under organic and conventional management
Ye et al. (2016)	Pot trial at experimental station	1.5 months	Biochar-mineral complexes (commercial) and compost amendments on soil physicochemical properties, bacterial abundance, and Pakchoi nutritional status and yield

on biochar's effect on soil physical and biochemical properties that is later transformed to soil-plant interaction (Gao and DeLuca 2016). The overall responses were found to vary with crop types, soil types, biochar types, residence time of biochar in soil, and a combination of these factors (Jeffery et al. 2011). The black color of biochar will enhance surface albedo and subsequently influence thermal dynamics that are associated with soil physical conditions, and this may possibly influence the germination process (Genesio et al. 2012). Generally, biochar additions improve soil physical properties including WHC thereby reducing nutrient leaching and possibly promoting soil nutrient availability and biomass gain (Gao et al. 2017). However, crop productivity increase was shown to be less responsive under wood- and crop-derived biochar additions than that under manure biochar; and crops growing on acidic soil with a coarse texture tend to respond more rapidly and efficiently to biochar additions in their productivity (Liu et al. 2013).

In a short-term field study (see San Juan case study below) examining nutrient uptake by dry bean on organic farming systems, Gao et al. (2016) found higher P, iron (Fe), magnesium (Mg), and zinc (Zn) concentrations in whole dry bean plant following biochar application over one growing season, and the responses were aligned with reduced resin-sorbed accumulations of these nutrients below dry beans rooting zone, suggesting an alteration of biochar-soil-plant interaction through its effect on soil nutrient leaching. A greenhouse experiment involving biochar amendments to an organically managed soil was also found to significantly reduce cadmium (Cd) availability in soil solution as well as Cd accumulation in all parts of the wheat plant (root, shoot, grain, or husk) due to the sorption of Cd onto biochar surface (Yousaf et al. 2016). The potential of biochar to remove heavy metal and associated pollutants is of importance to organic farming systems since there is potential for introducing contaminants from municipal and industrial organic wastes which would need to be managed without the use of synthetic chemicals (Alloway 2013). In addition, a significant synergistic effect of biochar and organic fertilizer or compost has been found to improve soil nutrient availability and organic C content, subsequently promoting crop nutrient uptake and yield in biochar-treated soils (Ye et al. 2016; Gao et al. 2017). This indicates a biochar-induced priming effect could potentially provide an additive effect in promoting organic fertilizer use efficiency in organic farming systems (Plaza et al. 2016). Another agricultural benefit of biochar in agriculture that has been commonly explored is the influence of biochar on biological N_2 fixation, root nodulation, and legume crop growth (DeLuca et al. 2015b) which are uniquely important in organic farming systems. This effect has been widely proposed to be closely related to the greater boron (B) and molybdenum (Mo) availability by biochar additions (Rondon et al. 2007; Güereña et al. 2015). Although organic farming systems have been reported to generate 5–34% lower yields than conventional farming (Seufert et al. 2012), the incorporation of biochar into an organic management system might help reduce nutrients loss and aid reducing the yield gap between the two farming systems while aiding in the buildup of soil C and fertility (Jeffery et al. 2011; Liu et al. 2013; Gao et al. 2017; Gao et al. 2019).

Biochar in Organic Agriculture: The San Juan Experience

As we mentioned above, a great number of studies have examined the role of biochar in agricultural soils in general, but few have focused on its use in organic farming systems, particularly associated with biochar generation using on-site feedstock. To our knowledge, the following case study is the first and only published field trial that has investigated the effect of locally produced wood biochar on soil fertility and crop performance in association with well-replicated established plots on multiple small-scale organic farming systems to date (Gao et al. 2016, 2017). Aiming at creating a closed-loop system that recaptures the value of local logging biomass that would otherwise be pile burned and generate net loss of nutrients, our study leveraged the existing resources and community readiness to create sustainable forest restoration and agriculture practices.

Background and Biochar Generation

Fire is a major form of disturbance in forests ecosystems of the western US (Heyerdahl et al. 1995). Active fire suppression and a shift in forest management objectives over the last few decades have led to an increased occurrence of heavily stocked second-growth forests that potentially change wildfire behavior (Naficy et al. 2010). Forest restoration and fuel reduction treatments, such as selection harvest combined with prescribed fire, are being practiced in the western USA to rebuild a more resilient forest structure (Agee and Skinner 2005). Forest residues from timber harvests are normally piled and burned resulting in emissions of gaseous air pollutants and volatiles, net loss of nutrients, and no net environmental benefit. Therefore, generating a value-added approach to managing timber harvest residues might help catalyze restoration activities on private and public forest lands.

We conducted an extensive study at six to ten organic farms located on the near-shore islands of San Juan County, WA, USA. Since the region is largely covered by heavily stocked, second-growth forests, thinning treatments have become a common practice for foresters and landowners on the islands. However, the dominant small-diameter timber in these forests has relatively low value and high transportation costs to get the timber to the market resulting in the timber mostly being piled and burned. At the same time, a critical part of San Juan County's economy rests on small-scale organic farming on sandy loam soils formed in glacial till and outwash across the islands. Creation of a system that simultaneously generates less pollution from forest thinning while contributing to the soil fertility of local organic farms food production would be highly desirable. Biochar generation from local timber harvest residues in this region may offer a sustainable means of reducing wildfire hazard fuel loading while improving soil health on neighboring organic farms.

With the formation of local nonprofit organization (http://restorechar.org/team/), environmental consulting (http://www.rainshadowconsulting.com/), forest service company (http://www.nnrg.org/), and county conservation district (https://www.sanjuanislandscd.org/), biochar was produced on-site by "cylinder burn" method tested by a group of local farmers and foresters and proved to be a highly efficient technique on the island (http://restorechar.org/make-charcoal/). The production cylinder was set up in close proximity to farm sites using logging residues which on average consisted of a mixture of 80% Douglas-fir (*Pseudotsuga menziesii*), 15% white fir (*Abies concolor*), and 5% western red cedar (*Thuja plicata*). The kiln was 1.5 m in height by 1.5 m diameter, and the production method operated in a similar manner to the traditional method called the "Kon-Tiki" kiln. Briefly, the cylinder burn operated with an open lid and relied on regular additions of feedstock to fill the cylinder (Gao et al. 2016). As the flame wall climbing up and feedstock being added throughout the burning, the material below was kept in a low-oxygen environment. Pyrolysis took approximately 7 h with temperature being kept at 450–550 °C. Approximately 55 L of water was later used to douse the flame once the fire reached the top of the cylinder. A floating metal lid was then placed on top and sealed with mineral earth. After 48 h, the char was removed, allowed to dry, ground by crushing under a polyvinyl tarp, and then sieved to 2 cm diameter.

Study Design and Results

The study region has a large percentage of forest land cover, consisting mostly of Douglas fir, western hemlock, and western red cedar. Most of the remaining land in the county is used for organic agriculture. The climate of the region is influenced by the Olympic Mountains and Vancouver Island, Canada, creating a "rain shadow" effect producing less rainfall and experiencing significantly drier and brighter weather than the surrounding locations. The soils of this region are predominately sandy loam soils formed in glacial till and outwash with a naturally high leaching capacity. Organic farms involved in our field study are dominated by Xerepts and Xeralfs as soil suborders (USDA soil survey: https://websoilsurvey.sc.egov.usda.gov/).

This field study was started in summer 2015, biochar amendment practice and associated examination of soil and crop performances have been conducted for three continuous growing seasons, and the sites are currently still under management by local farmers. The four treatments used in this study included (1) control with no additional amendment, (2) poultry litter applied at 70 kg N ha^{-1}, (3) wood biochar applied at 20 t ha^{-1}, and (4) a mix of poultry litter and biochar (70 kg N ha^{-1} + 20 t ha^{-1}). Local pond water was used to create a slurry of dry poultry litter and biochar in treatment (4), resulting in a moist "charged biochar," while the same volume of pond water was also applied with the poultry litter in treatment (2) (see Gao et al. 2016 for more details). In May 2015, the study was conducted on ten organic farms located on three islands in the region with cover crops being dry beans (*Phaseolus vulgaris L.*); three to five replicated blocks were established on each farm, and four treatments were randomly applied within each block with treatment plot size of 1 ×

1 m and 30 cm buffer in between. The following growing season (May 2016), six organic farms on Waldron Island were set up semipermanently for this study, a larger size of treatment plot (2 × 2 m) and buffer (1.5 m) was used, and four treatments were replicated three times and applied randomly at each farm with cover crops being Kobocha squash (*Cucurbita maxima*). The same layout and design were continuously applied on those six farms that all grew dry beans in summer 2017. Biochar were produced in the same manner for 2015 and 2016, and all treatments were applied before any plantation of crops in May 2015 and 2016.

Composite soil samples were collected from each treatment plot both at the mid-growing season (3 months after biochar application) and the end-growing season (6 months after biochar application). Soil samples were analyzed for a series of physical and biochemical variables including pH, bulk density, WHC, total C and N content, NH_4^+-N, NO_3^- -N, biologically based P status (DeLuca et al. 2015a), other soil macro- and micronutrient concentrations, potentially mineralizable N (PMN), microbial biomass C or N, basal respiration, and enzyme activities associated with C, N, and P cycling. Other soil analyses are described in details in Gao et al. (2017). Whole plant samples were taken when harvested for nutrient concentration determination. Given the fact that the plots were incorporated into the normal farming operations by farmers at individual farms, we were only able to get plot-size crop yield data at the second growing season (summer 2016). Ionic resin capsules were installed below crop rooting zone during each growing season to capture those accumulated nutrients that were leaching down or lost, and the resins were retracted at the end-growing season and extracted for nutrient concentrations.

Here we only present the data from the first two growing seasons (summer 2015 and 2016) that have been published. Biochar addition to soils significantly enhanced soil WHC in both growing seasons, implying an improved hydrological function of sandy soils by biochar. A significant increase in soil total C content following biochar additions was observed both growing seasons across all farms (30% on the ten farms in 2015 and 45% across the six farms in 2016) thereby enhancing soil C sequestration. The practice of biochar amendment was also found to alter soil N dynamics in both growing seasons, where soil PMN and NH_4^+-N were found to largely increase in biochar-treated plots at both midseason sampling points, but no differences were observed for soil NO_3^- -N pools. This finding implied a stimulated N mineralization process and associated NH_4^+-N pool being built up by biochar amendment, possibly through its adsorption of resident organic N compounds (such as amino acids, small proteins, and peptides) that added to the total mineralizable N pool or through its effect on soil moisture retention which may have improved conditions for mineralization process of regional sandy soils. The lack of change in NO_3^- -N pool with biochar addition was likely due to an already active nitrifier community that does not benefit from biochar additions (DeLuca et al. 2006, 2015b). Synergistic effects of poultry litter and biochar were found in both seasons. In organic farming systems, N is added in organic forms requiring net mineralization into plant available forms compared to conventional farms where N is applied in soluble (e.g., NH_4NO_3) or easily mineralizable (urea) forms. Our finding that biochar imparts a short-term increase in mineralization of applied organic N in these organic farming systems is of significance. However, it is also important to note that the observed effect of biochar on soil N appears to be transient given that no significant differences were observed among treatment plots at harvesting time in both seasons.

Soil available P status was also shown to be altered by biochar additions in both seasons. Citrate-extractable P (which represents a chelation-based acquisition strategy) was observed enhanced by charcoal at the first growing season (29% increase); and both citrate- and enzyme-extractable P (which represents an enzyme hydrolysis-based acquisition strategy) was found to be higher in biochar-treated plots at the second growing season (by 25% and 54%, respectively). Hydrophobic or charged biochar was demonstrated to be able to surface adsorb organic molecules involved in the chelation of specific ions forming organo-biochar or organo-mineral-biochar complexes (Joseph et al. 2013; DeLuca et al. 2015b), thereby they can modify soil P solubility and the pool of bioavailable P. Further, regarding the observed P status shifts in the second growing season, we proposed that wood-based biochar added to regional sandy soils were able to increase the phytoavailability of both organic and inorganic P pools through stimulating the P-solubilizing bacterial communities (PSB) and plant or microbial phosphatase activity, given the fact that an enhanced microbial biomass, bulk soil phosphatase activity, and abundance of PSB were observed with char addition. These biochemical variables were also found to share a significant percentage of variance with soil physicochemical properties, potentially suggesting that these changes in soil nutrient status were largely mediated by biochar-stimulated soil microbial communities. Again, similar to soil N, P inputs to organic farming systems are largely as manures or other organic P sources; thus the enhanced enzyme activity may potentially play a key role in supplementing the bioavailable P through mineralization process.

Significantly lower levels of NO_3^- -N, NH_4^+-N, P, Ca, and Fe were detected in ionic resins buried below the rooting zone in biochar plots compared to controls during both growing seasons, suggestion that biochar reduced leaching potentials in these sandy soils. Among those nutrients, Fe and P were reflected in cover crops where higher concentrations were observed in plants growing in biochar-treated plots, both dry beans of the first year and winter squash of the second year. An approximately 20% increment of squash fresh fruit yield was reported for the second growing season, posing a rather promising view of biochar use in these farming systems.

Linking Sustainable Agroforestry to Organic Farming

Our on-farm biochar study over the past two growing seasons has demonstrated the beneficial role of biochar in nutrient cycling and uptake by cover crops in these active organic farming systems associated with sandy soil of a glacial till origin. Biochar effect on soils that were observed in our study was primarily the significant increase in soil total C storage, alterations of N dynamics, biologically based P status, and significant less accumulated nutrients below crop rooting zone. These benefits on local soils were later reflected in cover crops across multiple organic farms on the islands. Concomitantly, the study region has an urgent need for forest health management or fuel reduction treatments to reduce fire risk on the isolated dry-forest ecosystem, but dealing with those on-site logging residues remains a problem for resident landowners. Therefore, linking the utilization of local woody residues to the creation of a closed-loop organic farming system with the need of improving soil fertility, our study has served as a unique example of sustainable agriculture practice and a community cooperative effort that represented operational, on-farm research trials that are of value to the broader research community as well as the regional farming community.

With small-scale regional biochar producers charging approximately $30 per cubic foot of biochar, selling almost exclusively to high-end gardeners and garden stores, biochar is currently not an economically feasible option for many small-scale organic farmers. In regions where forests and agricultural activities are close together, there is a great potential to create partnerships between the local forest industry, forest landowners, and farmers to create and utilize lower-cost production methods for biochar while driving forward forest restoration simultaneously. In addition, organic farming aims at emphasizing fewer negative environmental impacts, higher system resilience and ecological services provisions, soil sustainability, and quality food production while reducing external inputs cost and enhancing social capacity (Jouzi et al. 2017). By using local feedstock, relatively low-tech biochar production methodology, with minimal transportation costs, and decentralized yet less human labor in applying biochar by farmers on neighboring lands, this practice potentially minimized the net system nutrient loss and catalyzed local agricultural industry. It is possible that this type of biochar-associated sustainable agroforestry strategy could be exported to other agroecosystems that are in locales where forest biomass residues are abundant in and distributed across a landscape with small-scale farming operations that would benefit from biochar additions to surface soils.

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