Biochar and its Importance on Nutrient Dynamics in Soil and Plant: A Review

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Contents

Highlights

- Biochar's role as a potential source of plant macro- and micro-nutrients reviewed.
- Biochar increases plant nutrient availability in soils.
- Biochar influences soil physical, chemical and biological properties.
- Biochar enhances nutrient use efficiency and uptake by plants by decreasing nutrient loss.

Abstract

Biochar, an environmentally friendly soil conditioner, is produced using several thermochemical processes. It has unique characteristics like high surface area, porosity, and surface charges. This paper reviews the fertilizer value of biochar, and its effects on soil properties, and nutrient use efficiency of crops. Biochar serves as an important source of plant nutrients, especially nitrogen in biochar produced from manures and wastes at low temperature (≤ 400 °C). The phosphorus, potassium, and other nutrient contents are higher in manure/waste biochars than those in crop residues and woody biochars. The nutrient contents and pH of biochar are positively correlated with pyrolysis temperature, except for nitrogen content. Biochar improves the nutrient retention capacity of soil, which depends on porosity and surface charge of biochar. Biochar increases nitrogen retention in soil by reducing leaching and gaseous loss, and also increases phosphorus availability by decreasing the leaching process in soil. However, for potassium and other nutrients, biochar shows inconsistent (positive and negative) impacts on soil. After addition of biochar, porosity, aggregate stability, and amount of water held in soil increase and bulk density decreases. Mostly, biochar increases soil pH and, thus, influences nutrient availability for plants. Biochar also alters soil biological properties by increasing microbial populations, enzyme activity, soil respiration, and microbial biomass. Finally, nutrient use efficiency and nutrient uptake improve with application of biochar to soil. Thus, biochar can be a potential nutrient reservoir for plants and a good amendment to improve soil properties.

Keywords: Biochar, Nutrients, Manure, Soil properties, Nutrient use efficiency

1. Introduction

In recent decades, application of biochar to soil has drawn attention from the scientific community. Research has focused on its cost-effectiveness and environmentally friendly features, such as enhancing carbon sequestration and remediating contaminated soil. Biochar can influence nutrients in soil in several ways: (i) as a source of nutrients for plants and soil microorganisms (Li et al. 2017b); (ii) as a nutrient sink, thereby impacting the mobility and bioavailability of nutrients (Gul and Whalen 2016); and (iii) as a soil conditioner, thereby altering soil properties that influence the reactions and cycling of nutrients in the soil (Lusiba et al. 2017). As a source, biochar can supply nutrients such as nitrogen (N), phosphorus (P), potassium (K), and other trace elements inherently present in the original feedstock used for biochar production (Purakayastha et al. 2019). While some nitrogen and sulfur in the feedstock materials are lost through gaseous emission during pyrolysis (Al-Wabel et al. 2013; Leng et al. 2020), most nutrients are released during the weathering of biochar in soil, and they become available for plant uptake (Zhao et al. 2018). The nutrient content of biochar depends on the nature of the feedstock materials and the pyrolytic conditions. Biochars derived from manure- and biosolid-based feedstock materials generally contain higher levels of N and P than those derived from wood- and straw-based feedstock materials (El-Naggar et al. 2019a; Purakayastha et al. 2019). While the N content decreases with increasing pyrolytic temperature through gaseous emission (Leng et al. 2020), the P and K contents increase due to an increase in ash content (Christel et al. 2016; Tomczyk et al. 2020; Wang et al. 2013). As a nutrient sink, biochar can retain nutrients, thereby reducing their losses through leaching and gaseous emission. The nutrient retention capacity of biochar depends on its porosity and surface charge (cation and anion exchange capacity) (Yu et al. 2018). Biochar application reduces the loss of N, P, and K through leaching, and N through nitrous oxide emission (Beusch et al. 2019; Yao et al. 2012; Yuan et al. 2016). However, the loss of N through ammonia emission depends mainly on the pH of the biochar; biochar with a slightly acidic or near-neutral pH reduce ammonia volatilization from soil (Mandal et al. 2019; Mandal et al. 2018).

Biochar application influences various soil properties including pH, bulk density, cation exchange capacity, water retention, and biological activity. These changes in soil properties are likely to impact nutrient reactions on soil particles and microbial transformation of nutrients (Mandal et al. 2018). Upon application to the soil, biochar improves soil fertility and crop productivity by increasing the soil nutrient contents and the mobility of nutrients. It enhances microbial activity (Meier et al. 2019), improves aeration, and water retention (Kambo and Dutta 2015; Razzaghi et al. 2020), buffers soil reactions (Laghari et al. 2016), reduces bulk density (Yan et al. 2019a), and maintains soil aggregate structure (Zhang et al. 2020). Moreover, biochar reduces nutrient leaching and loss of nutrients by volatilization through altering the soil pH and by enhancing the ion exchange capacity (DeLuca et al. 2015). Biochar can change the soil microbial community composition (Ducey et al. 2013), and thus it impacts nutrient cycling and uptake by plants (Lehmann et al. 2011). Biochar decreases nitrification in soil resulting in reduced nitrate leaching (Igalavithana et al. 2016). Fig. 1 shows a conceptual framework depicting various impacts of biochar on soil and plants.

Many reviews have been published about the importance of biochar for soil health, crop production, and problem soils (Agegnehu et al. 2017; Al-Wabel et al. 2018; Dai et al. 2017; Ding et al. 2017; Ding et al. 2016; El-Naggar et al. 2019b; Juriga and Šimanský 2018; Laghari et al. 2016; Lone et al. 2015; Muhammad et al. 2018; Munoz et al. 2016; Palansooriya et al. 2019; Shaaban et al. 2018; Yu et al. 2019), soil carbon sequestration (Sarfraz et al. 2019), availability of N, P, and K (Liu et al. 2019a), and decreasing drought and salinity stress in plants (Ali et al. 2017). Reviews and meta-analyses also have been published focussing on soil N dynamics such as available N (Nguyen et al. 2017b), leaching and gaseous emissions of N (Borchard et al. 2019; Cai and Akiyama 2017), and the overall soil-N cycle (Liu et al. 2018). However, there is no review concerning the ability of biochar to retain multiple nutrients in soil through reducing gaseous and leaching losses and, thus, enhance plant growth. This paper focusses on: (i) effect of biochar on soil properties, (ii) biochar as a nutrient source, and (iii) impact of biochar on nutrient reactions in soil and uptake by plants.

2. Production and characteristics of biochar

The term *char* means output from disintegration of organic and inorganic materials. Biochar and charcoal have been synonymously used but can be differentiated by their use, because charcoal is used for energy, whereas biochar is considered for carbon sequestration and environmental applications. Biochar is also called as 'pyrochar,' because it is produced by the pyrolysis of biomass (Ralebitso-Senior and Orr 2016). The typical definition of biochar, as stated by the International Biochar Initiative (IBI), is 'a solid material obtained from the thermochemical conversion of biomass in an oxygen-limited environment' (IBI 2015). The production and soil application of biochar are related to the *'terra-preta'* (black earth) soils of Amazon region, which are important because of their high productivity. After the characterization of these soils, the scientific community recognized that biochar has properties similar to the *terra-preta* soils. Thereafter, much work was done related to biochar and its application in the soil. Generally, biochar is produced from a range of biomasses (e.g., manure, wood, crop, and industrial residues) at temperatures less than 900 °C and under oxygen-limited pyrolytic conditions (Zhang et al. 2019e). However, recent studies have shown that biochar can also be produced by other thermochemical processes, e.g., hydrothermal carbonization, gasification, torrefaction, and microwave-assisted pyrolysis (Kambo and Dutta 2015; Vithanage et al. 2017; Yuan et al. 2017).

The characteristics of biochar are influenced by the feedstock and heating conditions (Joseph and Taylor 2014; Laghari et al. 2016; Li et al. 2017b; Ralebitso-Senior and Orr 2016; Yuan et al. 2017). The physical and chemical properties also depend on other factors such as heating rate, kiln pressure, the composition of the atmosphere (N or CO_2 atmosphere in the kiln), and the type of pre- or post-treatment of biochar (Joseph and Taylor 2014). The important properties of biochar are presented in Fig. 2. Based on the ash composition and its properties, biochar can be divided into the following three main groups (Joseph and Taylor 2014).

i) Biochar produced from biomass with minimum ash content (<3–5%), such as wood, nut shells, bamboo, and some seeds (e.g., apricots). These hard biochars have large porosity, surface area (SA), and hold more water than biochars in other groups.

ii) Biochar produced from biomasses containing medium ash content between 5 to 13%, which include most agricultural wastes, bark, and high-quality green waste (i.e., with low contamination of plastics, soil, and metals).

iii) Biochar produced from biomasses with high ash contents (>13%), such as manures, sludges, wastepaper, municipal waste, and rice husks.

The physical characteristics of biochar, especially the surface area and pore size/volume/distribution, are controlled by the pyrolytic conditions and the nature of feedstock. For example, under high-temperature pyrolytic conditions (>550° C), biochar is characterised by having a large surface area and a high aromaticity (Ralebitso-Senior and Orr 2016). However, at pyrolysis under low temperatures (200–400°C), biochar is characterised by having more oxygen-containing functional groups, such as –COOH, –OH, C=O, phenolic –OH and –CHO groups, which stimulate nutrient exchange and, thus, improve soil fertility (Mandal et al. 2020; Ralebitso-Senior and Orr 2016). The characteristics of biochar are important for its uses. For example, biochar with a low surface area is less suitable for soil health improvement than that with a high surface area.

3. Effect of biochar on soil properties

The changes in soil properties resulting from biochar application are likely to impact nutrient reactions and microbial transformation of nutrients. Fig. 3 summarizes these processes.

3.1 Physical properties

Owing to special characters (such as high surface area and porosity), biochar application influences soil physical properties (Fu et al. 2019; Greenberg et al. 2019; Horák et al. 2019; Oladele 2019; Zhang et al. 2020). The effect of biochar on various soil physical properties that are likely to impact nutrient interactions in soil are summarized in Table 1. For example, in a 4-year field study, peanut-shell biochar altered soil properties by increasing water-stable aggregates (WSA) (Du et al. 2018), and rice straw biochar increased aggregate stability from 1 to 17% (Peng et al. 2011). In addition, biochar rate is positively correlated with WSA. For instance, Oladele (2019) reported that addition of rice husk biochar increased WSA at various soil depths over three years. The author found that with 3, 6, and 12 t ha^{-1} of biochar application, WSA increased by 10, 18, and 23%, respectively, at the 0-10 cm depth, and by 16, 20, and 26%, respectively, at the 10–20 cm soil depth compared to no biochar application in the first year. After three years, WSA increased by 22 and 24% at the 0–10 and 10–20 cm depths, respectively. Moreover, the application of rice husk biochar (10 t ha^{-1}) increased soil porosity by decreasing bulk density and increased available water in a sandy clay loam soil (Laghari et al. 2016). Li et al. (2018) said that maize straw biochar reduced soil bulk density and improved soil porosity in a semi-arid region. In a pot study, Prapagdee and Tawinteung (2017) concluded that cassava stem biochar increased soil porosity, which was in line with Fu et al. (2019) who found in a field trial that biochar dose was positively correlated with soil porosity. Li et al. (2018) conducted a study on the impact of maize straw biochar on soil properties in a tomato field in a semi-arid region of China. The authors found that application of biochar at 10, 20, 40, and 60 t ha^{-1} increased the soil porosity from 42.5% to 48, 50, 55, and 56%, respectively, and reduced the bulk density of a sandy loam soil. The application of biochar reduces bulk density of soil regardless of soil types, study environments, biochar application rate, or production conditions (Table 1).

Addition of biochar has been shown to increase the ability of soil to hold water (Yadav et al. 2018). Razzaghi et al. (2020) did a meta-analysis on the effect of biochar on soil water retention and found that the ability of soil to hold water increased, especially in coarse-textured soils, Peake et al. (2014) reported that biochar had a positive impact on the ability of loamy sand and sandy loam soils to hold water. The ability of soil to hold water has increased with increasing biochar application rates (Greenberg et al. 2019; Oladele 2019). Biochar reduced the tensile strength and cracks of a surface soil (Mandal et al. 2020), and suppressed soil shrinkage by increasing the ability of the soil to hold water; thus, soil structure was improved (Fu et al. 2019). Nair et al. (2017) observed that biochar improved soil water retention, reduced bulk density, and stabilized soil organic matter. Additionally, it was confirmed that there were hydrophilic functional groups on the surface and pores of biochar with a high affinity for water; biochar application was shown to increase soil water retention more in a sandy soil than a loamy soil or a clay soil (Mandal et al. 2020). Biochar also showed a positive impact on surface area of soil (Anawar et al. 2015), which varied with biochar types (Tomczyk et al. 2020). For example, biochar (10%) amended soil had 3 times higher surface area than untreated soil (Tomczyk et al. 2019). Therefore, irrespective of soil types, experimental conditions, biochar types, pyrolytic temperatures, and application rates, biochar has positive impacts on soil physical properties. Moreover, the above discussion shows that the soil physical properties are interlinked and influence each other.

3.2 Chemical properties

Biochar application has been shown to impact soil chemical properties such as pH, electrical conductivity (EC), and cation exchange capacity (CEC). These soil chemical properties influence nutrient interactions in soil. The impacts of biochar on selected chemical properties of soils are summarised in Table 2. Soil pH can be altered by incorporation of biochar into soil, thereby contributing to alterations in nutrient availability. The pH of biochar is an important character for its use in agriculture as a soil conditioner. Biochar pH is dependent on the rate of the carbonization process, pyrolytic temperature, and feedstock type (Weber and Quicker 2018). Biochar also generates organic acids during pyrolysis of biomasses that influence the pH of the final product (Cheng et al. 2018). Biochars generally have a pH range of 6.52–12.64 (Table 4), and the pH values positively correlate with the pyrolytic temperature (Fig. 4). Biochar has an alkaline nature due to the presence of alkali and alkaline metals in feedstocks that are not volatilized during pyrolysis (Yang et al. 2018). Application of alkaline biochar tends to increase the pH of acidic and neutral soils (Buss et al. 2016). The alkalinity of biochar depends on three important factors: a) organic functional groups; b) carbonate content, and c) inorganic alkali content (Lee et al. 2013). The concentration of base cations in biochar is strongly correlated with biochar alkalinity, which is not a simple function of biochar's soluble ash content (Fidel et al. 2017). Alkaline biochar can be used as a liming material for neutralizing acid soils (Taskin et al. 2019). However, the soil liming potential of biochar is not consistent across soil and biochar types. For example, application of biochar (at 1% and 2% rate) generated from various types of crop straws (pH value of biochar ranging from 7.69–10.26) in a three-month incubation study decreased the pH of an acidic Ultisol (pH = 4.31) over time (Laghari et al. 2016). However, in a field study, application of a paddy straw-derived biochar (biochar pH was 10.50) to a sandy soil (soil pH = 5.24) increased the pH of the soil by 4.5 units compared to the control (El-Naggar et al. 2018b). Moreover, a high dose (50 and 100 t ha^{-1}) of biochar (pH = 9.40) increased the pH of an Alfisol and, consequently, reduced exchangeable Al concentration in the soil (Tomczyk et al. 2020). Li et al. (2018) observed that application of biochar (10, 20, 40, and 60 t ha^{-1}) had no impact on soil pH in a semi-arid region, which was consistent with the results reported by Werner et al. (2018) who found that the pH of a sandy loam soil was not changed with addition of biochar. Therefore, biochar application to soil could either increase or decrease soil pH based upon the original soil properties (e.g., pH, texture) and biochar pH and alkalinity, as well as the species of crop grown in the biochar-amended soil (Table 2).

Most biochars contain high amounts of soluble salts, and, hence, the EC of biochar is generally higher than most agricultural soils (Igalavithana et al. 2018). Availability of soluble nutrient ions such as NO^{3-}, K^+, and Ca^{2+} could be directly related to the soluble salt content and, hence, the EC of biochar when applied to soil. Excess salts or high EC in soil is harmful for plants, because of a decrease in osmotic potential. Therefore, the EC of the soil must be maintained low for desirable nutrient availability and plant growth. Nevertheless, the EC of soil was reported to increase with increasing application rates of biochar (Li et al. (2018). Prapagdee and Tawinteung (2017) found that the EC of soil increased when cassava stem-derived biochar was applied at a rate of 10% (w/w). In a sandy soil (EC = 0.07 dS m^{-1}), the EC was increased by 385, 100, and 71% with the addition of paddy straw, silver grass residue, and umbrella tree residue biochar (30 t ha^{-1}), respectively (El-Naggar et al. 2018b). However, rice husk biochar (EC = 2.56 dS m^{-1}) had no impact on increasing the EC in the soil (Jatav et al. 2018).

The CEC of most biochars is higher than that of typical agricultural soils (Sohi et al. 2009; Sohi et al. 2010). The CEC of biochar is attributed to the generation of various functional groups, such as carboxyl and hydroxyl groups, during the pyrolysis of biomass (Tomczyk et al. 2020). Biochar CEC is governed by two important factors: (a) surface oxidation, and (b) adsorption of highly oxidized organic matter onto the biochar surface (Tomczyk et al. 2020). Like pH, CEC of soil can also be altered by biochar application. For instance, in a short-term (11 d) incubation study using an Ultisol, the addition of rice straw-derived biochar at 2.4 t ha^{-1} increased the CEC of soil (Peng et al. 2011). In another study, El-Naggar et al. (2018b) showed that the CEC of a sandy soil (CEC = 0.5

cmol kg^{-1}) increased by 3.00, 1.00, and 0.75 cmol kg^{-1} with the application of biochars (at 30 t ha^{-1} rate) derived from paddy straw, silvergrass residue, and umbrella tree residue, respectively. However, in a sandy loam soil (initial CEC = 10 cmol kg^{-1}), the paddy straw-biochar (at 30 t ha^{-1} rate) increased the CEC by 1.0 cmol kg^{-1} only. In another study, biochar derived from wood was found to increase the CEC by as much as 190% in an Anthrosol (initial CEC = 2.81 cmol kg^{-1}) compared to the control treatment (Tomczyk et al. 2020). Therefore, various types of biochars produced from various feedstocks change the CEC of soils to a different extent (Table 2), and the CEC affects nutrient availability and water retention of soil (Yadav et al. 2018). Moreover, biochar is known to increase the organic carbon content in soil (Table 2) and stimulate C sequestration by suppressing the long-term turnover of soil organic matter (Schofield et al. 2019). The increased organic carbon content, together with improved chemical properties due to biochar application, positively affect the nutrient status in soil.

3.3 Biological properties

Effects of biochar on various soil biological properties, such as soil respiration, microbial biomass carbon, microbial activity and functions, and soil enzymatic activity, are presented in Table 3. Owing to its porous system, biochar can be a favourable habitat for soil microorganisms including bacteria, mycorrhizal fungi, and actinomycetes (Compant et al. 2010; Prapagdee and Tawinteung 2017). Du et al. (2018) found that peanut-shell biochar (1%) increased microbial populations, microbial biomass, and actinomycetes. However, Wang et al. (2020) reported that a high dose of biochar could show a negative impact and a low dose could have a positive impact on soil microbial communities. The authors suggested that such variation of biochar's effects was due to the toxic effect (chemical stress) of biochar on soil microorganisms when applied at a high rate. However, in numerous studies biochar application exhibited positive effects on soil microbial activities. For example, in a coastal wetland soil, biochar application boosted the soil microbial biomass C and resulted in a low metabolic quotient (Zheng et al. 2018). Zheng et al. (2018) also found a shift of the bacterial community towards low C turnover bacterial taxa (e.g., Actinobacteria and Deltaproteobacteria), which stabilised soil aggregates. In another study over 90 d by growing tobacco plants with biochar application, Cheng et al. (2017) reported that, as the result of biochar application to soil with tobacco, the average populations of Sphingomonadaceae and Pseudomonadaceae bacteria were increased by 18 and 63%, respectively. In the same study, when tobacco plants were not grown, populations of the two bacterial groups in the soil were increased by 46 and 110%, respectively. Moreover, biochar was reported to increase microbial biomass N by 12% (Liu et al. 2018). The effects of biochar on soil microbial community structure and N-cycling bacteria depends on several factors, such as soil type, C/N ratio, nutrients, pH, and biochar addition rates (Abujabhah et al. 2018). Biochar application increased biological N fixation by 63% (Lu et al. 2018). Schofield et al. (2019) tested horticultural green waste biochar to retain N in a sandy loam soil. They found that biochar increased the microbial activity by 73, 84, 214% when applied at rates of 2, 5 and 10%, respectively.

Biochar showed positive impacts on soil enzymatic activities (Mierzwa-Hersztek et al. 2016; Ouyang et al. 2014). For instance, addition of biochar (5 and 10 t ha^{-1}) in an Inceptisol increased the dehydrogenase and urease activity by 19 and 44%, respectively (Ameloot et al. 2013; Mierzwa-Hersztek et al. 2016). Similarly, a greenhouse study concluded that biochar improved soil enzymatic properties with the application rate up to 6% (Yadav et al. 2018). Biochar also increased P-solubilizing bacterial populations such as *Burkholderia-Paraburkholderia, Planctomyces, Sphingomonas*, and *Singulisphaera*, which contributed to improving P availability in a forest soil (mountain acidic red loam soil) (Zhou et al. 2020). However, Haefele et al. (2011) found a negative effect on earthworm populations with the addition of rice residue biochar (41.3 Mg ha^{-1}). Similarly, Weyers and Spokas (2011) observed a negative effect (short-term) or no effect (long-term) of poultry litter biochar on earthworm activity in soil, which was attributed to a rapid pH change or high ammonia concentration in the soil due to the addition of the biochar (Liesch 2010). Earthworms are highly sensitive to soil pH and ammonia concentration (Saleh et al. 1970).

4. Biochar as a source of nutrients

Biochar can be a nutrient source for crop plants. The nutrient content of biochar depends mainly on the nature of the feedstock materials and the pyrolytic conditions (pyrolytic temperature, residence time, gaseous environment) (El-Naggar et al. 2019a). Feedstock materials containing high nutrient contents result in nutrient-enriched biochars. For example, manure and sewage sludge produce nutrient-rich biochars (Table 4).

4.1 Primary nutrients

4.1.1 Nitrogen

Nitrogen is one of the most limiting nutrients in soils for plant growth and productivity due to high crop demand for it and to chances of losses by leaching, runoff, and volatilization (Nguyen et al. 2017b). A continuous application of N in available forms is essential for many agricultural soils to maintain production in cropping seasons (Fageria and Baligar 2005). Biochar can be a potential source of N for plants. In addition to organic forms of N (e.g., hydrolyzable‒N, water-soluble‒N, and nonhydrolyzable‒N), biochar also contains inorganic N forms such as NH_4^+-N, NO_3^--N, and N_2O-N (Liu et al. (2019a). Although N content is low in most biomasses, the N content is mostly increased after pyrolysis due to reducing the mass (mainly the moisture) of the biomass. In the case of N, there could be some losses also during the pyrolysis of biomass due to gaseous emissions of the element. Hence not all forms of N present in the feedstock can be found in the biochar. For example, some amino acids, such as arginine containing amide groups, are mostly converted to ammonia or other gaseous forms of N during biomass pyrolysis, and, consequently, they are lost (Leng et al. 2020). Nitrogen conversion pathways from feedstock-N to biochar-N through the process of pyrolysis are presented in Fig. 4. The existence of metal elements in feedstock can influence the conversion of N-containing compounds and, thus, the amount and forms of N species in final biochar products (Xiao et al. 2018). Table 4 shows that the N content of biochar can be of a wide range (0.24 to 6.8 %). Although, most biochars have low N content (below 1.5 %) (Table 4), the N content is high in a few biochars such as those derived from sewage sludge (6.8%), poultry litter (5.85%), grass waste (4.9%), and microalgae (14.12%) (Chang et al. 2015). Biochar produced from sewage sludge (at 350 °C) had more N (3.17%) than that produced from sugarcane and eucalyptus wastes (1.4 and 0.4%, respectively) (Figueredo et al. 2017). Furthermore, N content of biochar decreases with an increase in the pyrolytic temperature (Fig. 5), due to conversion of parts of amino acids into pyridine-N and pyrrolic-N (Leng et al. 2020). Ultimately, the loss of NH_4^+-N as NH_3 occurs through volatilization during pyrolysis (El-Naggar et al. 2019a). For instance, N contents of chicken manure biochar were found to be 2.79, 2.45, and 1.81% when the material was produced at 250, 350 and 550 °C, respectively (Xiao et al. 2018). Similarly, N content of maize-straw biochar decreased from 1.25% (300 °C) to 1.20% (500 °C) (Song et al. 2018), and that of elephant-grass biochar decreased from 3.87% (400 °C) to 2.15% (600 °C) (Ferreira et al. 2018), due to a rise of the pyrolytic temperature. Acidified biochar (pre-pyrolysis) decreased the total N content, which was attributed to volatilization loss of N during pyrolysis (Sahin et al. 2017). However, salt impregnated (chicken manure with $CaCl_2$ and $FeCl_3{\cdot}6H_2O$) biochar slightly increased the total and available NH_4^+-N contents when pyrolyzed at a low temperature (250°C), but at 350 and 550°C, the NH_4^+-N content decreased (Xiao et al. 2018). Xiao et al. (2018) found 0.48, 0.30, and 0.17 g kg^{-1} available NH_4^+-N (KCl extractable) in chicken manure biochar following pre-pyrolysis impregnation of the biomass with $CaCl_2$, $MgCl_2.6H_2O$, and $FeCl_3.6H_2O$ mineral salts, respectively. Chang et al. (2015) found that N content in a *Chlorella*-based algal residue biochar increased from 10.23 to 14.12% when the residence time of pyrolysis was increased from 20–60 min at 500 °C. However, the effect of rising pyrolytic temperature ranging from 300 to 700 °C on the N content of algal biochar was not consistent (Chang et al. 2015). The N-containing components of biochar can be present on the biochar surfaces and/or inside the pores as nitrates, ammonium salts, or heterocyclic compounds (Grierson et al. 2011). These N components of algal biochar were much higher than other common biochars such as manure and biosolid/sewage sludge derived biochars. Among the inorganic forms of N, NO_3^--N and N_2O-N were increased at a high temperature (800 °C) for pyrolysis, NH_4^+-N and NO_3^--N were decreased drastically at 300 °C, and all inorganic N remained stable at 600 °C (Zhu et al. 2016). Therefore, when producing N-enriched biochar, special care should be taken to decide the pyrolytic temperature and feedstock type.

4.1.2 Phosphorus

Like the N content in different biochars, the P content varies over a wide range (0.005–5.9 %) (Table 4). While the N content decreases with pyrolytic temperature, the P content is positively correlated with the pyrolytic temperature (Fig. 5). The increased P content in biochar with increasing pyrolytic temperature can be attributed to the ‘concentration effect’ resulting from decreased biochar yield with increasing temperature. For example, Xiao et al. (2018) produced biochar from chicken manure at 250, 350, and 550 °C and found corresponding P contents of 1.91, 2.15 and 2.96%, respectively (Table 4). Moreover, the P content also depends on the type of biomass. For instance, P contents in biochar derived from swine solid (5.9%) (Cantrell et al. 2012), chicken manure (2.96%) (Xiao et al. 2018), and poultry litter (2.57%) (Brantley et al. 2016) were greater than those derived from rice husks (0.15%) (Bu et al. 2017) and apple branches (0.18%) (Li and Shangguan 2018). Thus, feedstock selection is an important aspect for producing P-enriched biochar. In addition, the P content of chicken manure biochar increased from 1.91 to 2.96% by increasing the pyrolytic temperature from 250 to 550 °C (Table 4). Biochar with a high ash content contained a high P content (Laghari et al. (2016). In a review on the mineral contents of biochar, Xu et al. (2017) stated that biochar from sewage sludge and poultry litter had higher P contents than biochar from crop residues, animal manures, and woody biochar. They also found that available P (i.e., Olsen-P) in biochar increased from 280 to 676 mg kg^{-1} when the pyrolytic temperature increased from 300 to 600 °C. Li et al. (2020) found that Olsen-P increased in both pristine and P-laden biochar by 43 and 15%, respectively, when the pyrolytic temperature increased from 350 to 600 °C. The authors also observed that the amount of Olsen-P increased in KH_2PO_4 biochars with increase in temperature. In addition, Xiao et al. (2018) found that water-extractable P was negatively correlated with the pyrolytic temperature for both pristine and modified biochars, while the Olsen-P was positively correlated with increasing temperature. The authors also observed that the Olsen-P decreased when pre-treatment of chicken manure was conducted with different types of salts, because of the formation of insoluble phosphate compounds such as $(CaMg)_3(PO_4)_2$ and $Fe_4(PO_4)_2O$. Zhang et al. (2019d) found that Olsen-P and water soluble-P contents were 775.45 and 495.21 mg kg^{-1}, respectively, in an acidified biochar (700 °C) derived from maize straw.

4.1.3 Potassium

The K content in biochar also varies both with the feedstock type and temperature of pyrolysis (Table 4). For example, poultry litter, chicken manure, rice straw, and bamboo biochar contained more K than biochars made from rice husks, corn stalks, and apple branches. As in the case of P, K content of biochar also increases with increasing pyrolytic temperature (Fig. 5), which can be attributed to the 'concentration effect'. Xiao et al. (2018) found that the K content in chicken manure biochar was increased from 4.16–5.93% when the pyrolytic temperature was increased from 250 to 550 °C (Table 4). Poultry litter-derived biochar contained 3.88 and 5.88% K at pyrolytic temperatures of 400 and 600 °C, respectively (Subedi et al. 2016). Similarly, Vaughn et al. (2018) produced biosolid-biochar at 300, 400, 500, 700, and 900 °C, and the K contents were 3.89, 3.98, 4.06, 4.02, 8.12, and 9.83%, respectively. Karim et al. (2017) evaluated the K-enrichment of banana peduncle biochar produced in the presence of different gases (Ar and O_2) and plasma with processing times of 3, 5, 7, and 9 min. They found that plasma processing for up to 7 min enriched the biochar with K in both Ar and O_2 environments. For instance, due to Ar gas loading for 7 min, K increased from 8.6 to 28.6% for available K, 3.5 to 11.2% for water soluble K, and 5.1 to 14.7% for exchangeable K. Amin (2016) reported that soluble-K content was 6.05 g kg^{-1} in corn cob biochar, and Nguyen et al. (2020) found 8.50 g kg^{-1} exchangeable K in rice husk biochar.

4.2 Secondary nutrients

As shown in Table 4, contents of secondary nutrients including S, Ca, and Mg are high in animal manure biochar, as reported by Xiao et al. (2018) and Brantley et al. (2016). The Ca contents of animal-manure biochar ranged from 0.40 to 6.15% and that of industrial and municipal waste-derived biochar ranged from 0.37–6.57% (Table 4). Biochar derived from crop residues had concentrations of Ca ranging from 0.20–1.57% and that of woody biochar was in the range of 0.05–2.42% (Table 4). However, biochar produced from apple branches had a higher Ca content (2.42%) (Li and Shangguan 2018) than other feedstocks such as barley straw (0.20%) (Jatav et al. 2018), sugar maple sawdust (0.50%) (Noyce et al. 2017), and acacia (0.27%) (Arif et al. 2016). The Mg contents of biochar produced at 250-750 °C from various types of biomasses (e.g., animal manure, woody biomass, crop residue) ranged from 0.001–3.78% (Table 4). Most of the animal-manure derived biochars and grass waste biochar contained higher Mg contents than crop-residue biochar and woody biochar (Table 4). Generally, the S content was lowest (0.001–0.32%) in biochar produced from woody biomass followed by waste-derived biochar (0.005–0.63%) and crop residue-derived biochar (0.07–0.32%) (Table 4). Animal manure biochar contained more S (0.02–1.36%) than orchard-pruning-biomass-derived biochar (0.005%) (Table 4). The effects of pyrolytic temperature on the S content of biochars are inconsistent (Table 4), because high temperatures can either increase S content by the incorporation of S into complex structures or decrease S content due to volatilization loss (Al-Wabel et al. 2013).

4.3 Trace elements

Biochar also contains a significant amount of trace element nutrients (micronutrients) such as Fe, Cu, B, Zn, Mn, and Mo. Most of the published literature reports only Fe, Zn, and Cu contents of biochar; few of them mention Mn content; and only few report Mo and B contents (Table 4). Table 4 shows that Fe content in biochar of animal manure was higher (311–7480 mg kg^{-1}) than biochar from crop residues and woody materials. The Fe content in biochars produced from waste materials was in the range of 0.009–380 mg kg^{-1} (Table 4). Like Fe, animal manure biochar contained more Zn (131–4981 mg kg^{-1}) and Cu (99–2446 mg kg^{-1}) than waste- and crop-residue derived biochars (Table 4). The contents of the micronutrient elements depend on the feedstock type and biochar production temperature. However, the effect of these factors is not consistent for micronutrient contents of biochar products, which can be attributed mainly to the low micronutrient contents in feedstock materials. For instance, eucalyptus green waste biochar produced at 650–750 °C had 7000 mg kg^{-1} Fe (Abujabhah et al. 2016), whereas willow wood waste biochar produced at 550 °C had only 0.05 mg kg^{-1} Fe (Agegnehu et al. 2016a). Several other studies (Brantley et al. 2016; Chen et al. 2018; Li and Shangguan 2018; Miranda et al. 2017; Noyce et al. 2017) also reported that biochar contains a low but significant amount of micronutrients.

5. Effect of biochar on nutrient reactions in soil and uptake by plants

As a sink, biochar can retain nutrients, thereby reducing their losses through leaching and gaseous emission. Biochar application influences various soil properties including pH, bulk density, CEC, water retention, and biological activity (section 3), which in turn affect nutrient retention of soils.

5.1 Nutrient Retention

Biochar can contribute in improving nutrient retention capacity of soil due to its large surface area, porosity, and presence of both nonpolar and polar surface sites (Ahmad et al. 2014; Hussain et al. 2017; Mukherjee et al. 2011;

Yu et al. 2018). The polar sites are likely to increase the soil CEC (Mukherjee et al. 2011). For example, biochar with a high CEC retains more nutrients in soil by reducing nutrient loss through leaching (Tomczyk et al. 2020). Application of biochar also enhances nutrient retention by increasing the soil pH and soil organic matter (Mendez et al. 2012). Nutrient retention and release depend on soil pH (Fig. 6). For instance, Gao et al. (2016) reported that addition of biochar increased NO_3^--N and NH_4^+-N retention in soil by 33 and 53%, respectively. Sorrenti et al. (2016) also observed a similar effect of biochar application on soil N. Liu et al. (2017b) proposed three important mechanisms for N retention after biochar application in soil: (i) adsorption of NH_4^+-N due to the high CEC of biochar, (ii) reduced leaching of NO_3^--N due to increased ability of the soil to hold water, and (iii) increased microbial immobilization of N in soil by the supply of labile C. Schofield et al. (2019) suggested that high cation and anion exchange capacities of biochar and its ability to retain ions and molecules within the pores further contribute to biochar's enhanced nutrient retention capacity. Hence, biochar produced at high temperature might have a high ability to retain NO_3^--N without its leaching to ground water. Sometimes biochar has reduced nutrient retention due to quick decomposition of biochar C (e.g., by 51% within 16 months of application) (Beusch et al. 2019). The impacts of various types of biochar and nutrient availability changes in different soils are summarized in Table 5.

Owing to porous structure and NH_4^+-N adsorption ability, biochar can play a vital role in slowing down N release from the soil. This statement was supported by Zhang *et al.* (2017) who reported that the pore space of biochar can facilitate water and nutrient transfer at initial stage of biochar application. The hydrophobic nature of biochar can hinder water transport and thus limit N diffusion (Dong *et al.*, 2020). Moreover, NO_3^--N adsorption capacity of biochar also influence N release in soil (Hagemann *et al.*, 2017). In recent years, several studies reported that biochar can be used as a slow-release fertilizer. For example, Shi *et al.* (2020) conducted a pot study and found that biochar-urea composite release N slowly than conventional urea fertilizer and thus it was more effective in NH_4^+-N retention. This agreement was supported by Sashidhar *et al.* (2020) who also reported that biochar-based slow-release fertilizer (BSRF) release N slowly by 69.8% over a period of 30 d. Similarly, Hu *et al.* (2019) and Liu *et al.* (2019d) reported that 59.32% N was released after 84 d and 69.8% N released within 28 d of BSRF application, respectively.

Biochar plays a role for N availability in soil due to two main mechanisms: biotic (fixation, mineralization, immobilization, denitrification, plant uptake) and abiotic (sorption, volatilization, leaching) (Clough et al. 2013; Nguyen et al. 2017b). The increase of N availability in soil from biochar application is, therefore, beneficial for plant growth (Esfandbod et al. 2017; Igalavithana et al. 2016). In addition, negative and neutral impacts of biochar on soil N availability have been reported (Mukherjee and Lal 2014; Nguyen et al. 2017b). For example, addition of rice husk biochar reduced the available N content by 21% (sole biochar) and 15% (biochar + fertilizer) compared to a control soil (Arenosol), which was due to immobilization of N (Werner et al. 2018). Liu et al. (2018) did a meta-analysis and concluded that biochar application decreased NH_4^+-N and NO_3^--N contents in soil by 6 and 12%, respectively. Therefore, the effects of biochar application on N availability in soil are not consistent as the N availability is governed by rate and type of biochar as well as the soil type (Table 5). For example, under field conditions, the addition of biochar (10 Mg ha^{-1}) plus organic and chemical fertilizers increased N availability in a silty clay loam soil (Arif et al. 2017). In addition, modified biochar (calcium alginate impregnated) also increased the nutrient (N and K) retention in soil, as reported by Wang et al. (2018). Moreover, combined application of biochar and farm yard manure (FYM) improved the nutrient (N and P) retention in soil (Arif et al. 2017).

Biochar can be a reserve stock for P in soils (Dai et al. 2016; Zhang et al. 2016). For instance, with the incorporation of sugar maple and red pine biochar, available P was found to be three times higher in a sand than in sandy loam and silty sand soils (Noyce et al. 2017). Several studies showed that soil amended with biochar increases P bioavailability and plant growth (Arif et al. 2017; Beheshti et al. 2017; Biederman et al. 2017; Brantley et al. 2016; Efthymiou et al. 2018; Houben et al. 2017). The changes of P availability in soil, as impacted by biochar application, are presented in the Table 5. Like N, the availability of P is changed with the addition of biochar and it depends on the biochar and soil. The majority of the studies report that the availability of P is increased with the application of biochar. However, some researchers showed decreased availability of P after biochar addition (Table 5). Modified or fortified biochars increase the P retention capacity of soil. For instance, Wu et al. (2019a) studied the mechanism of inorganic P adsorption under field conditions in saline-alkaline soil. The authors found that MgO-biochar showed 1.46 times more phosphate adsorption than pristine biochar due to electrostatic attraction, precipitation, and exchangeable anions. Thus, modified biochar increased the availability of P in soil. Several studies (Atkinson et al. 2010; Glaser et al. 2002; Major et al. 2010) reported that application of alkaline biochar to acidic soils increased K content in soils. This is in agreement with DeLuca et al. (2015) and Lehmann et al. (2003) who reported that the bioavailability of K was increased with addition of biochar. Usually the availability of K in soil is increased with the addition of biochar irrespective of the study, although some negative impacts of biochar on the availability of K in soil have been reported (Table 5). The addition of biochar (10 t ha^{-1}) increased the Mg content in a loamy-sand soil (Lusiba et al. 2017).

The impacts of biochar on nutrient retention in soil are mostly positive. For instance, biochar increased Ca and Mg availability in soil and, thus, boosted crop yield (Hussain et al. 2017) which was previously supported by Abujabhah et al. (2016) who found that woody biochar had a significant impact on exchangeable Ca, Mg, and Na in black clay loam, red loam, and brown sandy loam soils. Moreover, the Ca availability increased in soil even at a low rate of biochar application (1.25%); however, no change in S availability was observed (Eykelbosh et al. 2014). The availability of Ca, Mg, and S increased or decreased due to incorporation of biochar in soil, as shown in Table 5. A few studies (Lu et al. 2014; Zhang et al. 2013) state that biochar alters the bioavailability of trace elements in soils (Beesley et al. 2011). For example, woody biochar improved the availability of micronutrients (B and Mo) (Hussain et al. 2017), whereas the addition of mixed hardwood-derived biochar did not influence the Cu and Zn content (Cai and Chang 2016). The Fe and Al contents were decreased by biochar addition in sandy soils, but biochar had no impact in silt or clay soils (El-Naggar et al. 2018c). However, addition of hardwood-derived biochar increased Fe and Mn availability, but it had no effect on Zn and Cu availability (Ippolito et al. 2014). Noyce et al. (2017) showed a positive effect of biochar on Mn and Na contents in sand, sandy loam, and silty sand soils. The availability of micronutrients is influenced by the application of biochar to soil (Table 5), and feedstock and type of soil are important in determining micronutrient availability.

5.2 Nutrient Leaching

5.2.1 Nitrogen

Nitrate (NO_3^-) leaching is a major reason for loss of N from soils and causes groundwater pollution (Cheng et al. 2018). Surface properties of biochar facilitate the adsorption of ions in the soil solution. Electrostatic and capillary forces on the surface of biochar reduce nutrient leaching from soils. For instance, the application of Brazilian pepperwood biochar reduced NO_3^- leaching by 34% through adsorption (Yao et al. 2012). Soil amended with biochar can adsorb NO_3^- through its anion exchange sites, thereby reducing N losses and increasing NO_3^- retention. Moreover, woody biochar application can decrease nutrient leaching through increasing water retention, as reported by Lehmann et al. (2003). Biochar has the capacity to retain inorganic N ions and, therefore, it reduces N leaching and runoff in soils (Steiner et al. 2008). Fig. 7 shows that the application of biochar reduced NO_3^- leaching by 26%. Cao et al. (2019) showed that biochar derived from apple branches reduced leaching of NO_3^--N by 9.9–68.7% and nitrogen-oxide flux by 6.3–19.2%. Application of mixed hardwood biochar decreased N leaching by 11% in Midwestern agricultural soils (Laird et al. 2010), 72% in sub-alkaline soils of an apple orchard

(Ventura et al. 2013), and 46% in a tropical Arenosol (Beusch et al. 2019). Cheng et al. (2018) conducted an incubation study and found that NO_3^--N leaching was decreased, but NH_4-N leaching was increased, in biochar-amended soil due to reducing the CEC in biochar with increasing temperature.

5.2.2 Phosphorus

Excessive application of P fertilisers has resulted in the leaching of P from agricultural fields to aquatic systems (Karunanithi et al. 2015; Loganathan et al. 2014). Biochar has proven to alter P availability in soils by reducing P leaching through sorption/adsorption. In a column study, biochar produced from Brazilian pepperwood at 600 °C reduced the total amount of phosphate by about 20.6% in biochar-amended soil (Yao et al. 2012). Doydora et al. (2011) found that the application of peanut hull biochar increased the amount of phosphate in the soil solution by 39%. The possible mechanisms suggested for the influence of biochar on P availability are change in soil pH and subsequent influence on the interaction of P with other cations and enhanced retention through anion exchange and P precipitation (Atkinson et al. 2010). In natural environments, P is strongly adsorbed onto the surface of Fe(III)-(hydr)oxides in soils (Jaisi et al. 2010). Cui et al. (2011) showed that addition of biochars reduced the amount (30-40%) of P sorbed onto ferrihydrite (the most effective Fe-oxide for P adsorption), which likely improved in P availability in soil. The biochars magnetized with Fe^{3+}/Fe^{2+} enhanced phosphate sorption, compared to non-magnetic char (Chen et al. 2011). Leaching of P is reduced by absorbing it on the surface of biochar (Biederman and Harpole 2013). Biochar with a large surface area has high adsorption capacity for the ionic forms of P. So, biochar can reduce ortho-P leaching from nutrient-rich soil and influences P availability (Gul and Whalen 2016; Hussain et al. 2017).

5.2.3 Other nutrients

Leaching of nutrients depends on soil type, physico-chemical properties of the biochar, and the pyrolytic temperature (Cheng et al. 2018; Yuan et al. 2016). For example, sewage sludge biochar produced at 500 and 700 °C reduced the leaching loss of K in a Typic Plinthudult soil more than that of biochar produced at 300 °C (Yuan et al. 2016). Biochar can increase leaching of K in crop fields for the short term (Angst et al. 2014; Guo et al. 2013), which results in ground water pollution. For example, application of wood biochar in an acidic and low fertile soil resulted in leaching of K, Ca, and Mg to the 60 cm depth, but concentrations gradually decreased to the 120 cm depth (Major et al. 2012). This might be related to variation in nutrient uptake by plants at different depths. Addition of biochar resulted in increased K leaching by 65% below the A1 horizon (Hardie et al. 2015), which was attributed to a high amount of soluble K in the biochar. Biochar-induced leaching loss of Ca decreased with increasing temperature of biochar production (Cheng et al. 2018). Thus, leaching of nutrients in biochar amended soil depends of several factors, including biochar type and rate of application, soil type, and depth of soil. Long-term field studies are needed to investigate the effect of biochar on nutrient leaching.

5.3 Gaseous emission

Nitrogen in soil is lost through leaching and gaseous emission of ammonia (NH_3) and nitrous oxide (N_2O). Inorganic N is reduced in soil mainly through NH_3 volatilization (Liu et al. 2017b). More than 85% NH_4^+-N is lost from soil due to gaseous emission (Esfandbod et al. 2017). It is necessary to reduce the loss of N from soil for plant growth and development. The physical and chemical characteristics of biochar influence their effectiveness in controlling NH_3 volatilization. Biochar addition to a highly alkaline soil decreased soil pH thereby reducing NH_3 volatilization (Mandal et al. 2016). The NH_3 adsorbed by biochar can, subsequently, become available for plants (Taghizadeh-Toosi et al. 2012). Biochar addition has often been shown to decrease total N_2O emission from soils treated with N sources such as manure, urea, and compost (Bruun et al. 2011; Singh et al. 2010; Spokas et al. 2009). Denitrification is the biological process leading to increased N_2O emission from soil. A decrease in denitrification is likely to occur due to adsorption of inorganic N (NH_4^+, NO_3^-) to biochar surfaces, thus reducing the substrate for denitrification (Taghizadeh-Toosi et al. 2012). Complete denitrification leading to N_2 emission due to biochar addition was explained by enhanced anaerobic conditions (Taghizadeh-Toosi et al. 2012), presence of labile C in biochar, elevated soil pH, and enhanced microbial activity (Anderson et al. 2011). Lehmann et al. (2006) hypothesized that biochar could reduce N_2O emissions by inducing microbial immobilization of mineral N in the soil. According to Lu et al. (2018) and Nguyen et al. (2016) biochar inhibited denitrification and thus decreased NO and N_2O emission by 32%. However, biochar could temporarily increase volatilization of N by 19% as NH_3, which will be ultimately deposited into the soil (Fig. 7). However, Cayuela et al. (2014) carried out a meta-analysis and showed about a 54% reduction in N_2O emissions with biochar application. Biochar reduced the cumulative N_2O emissions, the N_2O–N emission factor, and the yield-scaled N_2O emissions by 5–39, 16–67, and 14–53%, respectively (Li et al. 2017a). The addition of biochar reduced N_2O emissions by 15% from acidic soil in a vegetable field (Wang et al. 2015). In a study by Fungo et al. (2019), addition of biochar reduced cumulative emissions of NH_3 and N_2O by 47% and 22%, respectively, over three years, which indicated that biochar has a residual effect on gaseous emissions of N.

5.4 Uptake and assimilation of nutrients

5.5 Nitrogen

The impact of biochar on nutrient concentration, uptake, and crop growth and development are presented in Table 6. Biochar application to soil influences N uptake in plants. For example, Amin and Eissa (2017) studied the impact of biochar on N and P use efficiency of zucchini plants (*Cucurbita pepo*) grown in a calcareous soil. They found that the fruit N content increased by 39.23% over the control with the lowest (6.3 g/pot) biochar rate, whereas, with increasing the rate of biochar addition by 12.6 and 25.5 g/pot, the N content decreased by 7.45% and 13.73%, respectively, which was attributed to 'dilution' effect caused by increased yield. However, Werner et al. (2018) showed that sole biochar and biochar with NPK fertilizer decreased N concentration in plants by 20 and 15%, respectively, which they attributed to immobilization of N in soil. In the USA, Sistani et al. (2019) investigated the effect of hardwood biochar on corn yield and greenhouse gas emission under field conditions in silt loam soil. They found higher N concentration in biomass in the first year of the study, which was a dry period, whereas in the second and third years, which had favourable moisture conditions, N concentration was lower than in the control treatment. Application of biochar has been shown to increase N uptake by 11% (Fig. 7). However, a few studies (Akoto-Danso et al. 2018; Kang et al. 2018) stated the negative impacts of biochar on N concentration and uptake by plants. Results are variable. Mandal et al. (2016) reported that biochar increased N uptake by 76.11% over the control soil, while Nguyen et al. (2016) found no impact on N uptake with the addition of rice husk biochar up to 30 t ha^{-1}.

5.6 Phosphorus

Plants take up P as monovalent or divalent anions ($H_2PO_4^-$ or HPO_4^{2-}), but the availability of these ions may be below the required level for plant growth if they are physically and chemically bonded in soils (Noyce et al. 2017). Addition of biochar increased the P concentration of lettuce leaves (Biederman and Harpole 2013; Gunes et al. 2014). Other studies support this observation (Arif et al. 2017; Shepherd et al. 2017; Werner et al. 2018). Residual biochar plus microbial inoculation with and without P-fertilizer increased by 20–52% the P content of maize (Rafique et al. 2020). The impact of biochar on P uptake is mostly positive and few studies show a negative impact (Table 6). For instance, incorporation of various types of biochars (empty fruit bunch, sewage sludge, and chicken litter) at different levels (5–40 t ha^{-1}) increased P uptake by 23–2096% (Table 6). Biochar plus chemical fertilizer increased P and K uptake more than biochar alone (Sistani et al. 2019). However, biochar has been shown to reduce P uptake by plants (Kang et al. 2018; Liu et al. 2017a) and thus decrease crop yield, which might be due to the phytotoxic effects of wood biochar (Liu et al. 2017a). Table 5 gives information on P uptake with different biochars.

5.7 Potassium

Biochar addition plus N-fertilizer was positively correlated with K content in sunflower plants, and the treatments improved plant growth and development (Pfister and Saha 2017). Fazal and Bano (2016) did an experiment under axenic conditions in a growth chamber to evaluate the role of biochar, *Pseudomonas* sp., and chemical fertilizer on uptake of K by maize. They observed that K content was increased in maize by 46, 47, and 3% with addition of only biochar, biochar + *Pseudomonas* sp., and biochar + chemical fertilisers, respectively. Biochar can be used as an effective K-fertilizer in terms of its economic, environmental, and slow-release properties (Oh et al. 2014). The concentration of K in plants grown in soil with biochar application has increased up to 112.27% (Table 6). Addition of biochar at 10% increased K in stems, leaves, nut shells, and roots (Prapagdee and Tawinteung 2017). Mycorrhizal inoculation in biochar amended soil increased K content by 11–20% and K uptake by 69% (Rafique et al. 2020). Most studies report that the uptake of K is stimulated due to the addition of biochar (Table 6). However, a few negative impacts of K uptake are presented in the Table 6.

5.7.1 Other nutrients

Addition of poultry manure biochar decreased Ca and Mg concentrations in lettuce (Gunes et al. 2014). But, biochar (1%) increased Ca and Mg concentration in chicory (*Cichorium intybus*). Concentration of Ca, Mg, and S increased after 50 t ha^{-1} biochar addition (Noyce et al. 2017). Application of woody biochar increased the uptake of micronutrients (iron, copper, zinc and manganese) in soil (Gao et al. 2016). Table 6 shows concentrations of Ca, Mg, and micronutrients after biochar addition.

5.8 Nutrient use efficiency

The nutrient use efficiency can be defined as yield or biomass per unit input (fertilizer, nutrient content) (Reich et al. 2014; Sarkar and Baishya 2017). It depends upon the soil, plant, and environment (Reich et al. 2014). Biochar can contribute to nutrient use efficiency in plants, both directly through increased nutrient uptake and indirectly by decreasing the loss of nutrients through leaching and gaseous emissions. Several studies (Cao et al. 2019; Coelho et al. 2018; Li et al. 2017a; Nguyen et al. 2017a; Yu et al. 2017; Yu et al. 2018) report that application of biochar increases N uptake, thereby increasing N use efficiency (NUE) in crops. Addition of wood biochar (10 t ha^{-1}) in an alkaline soil improved P use efficiency (PUE) of both wheat and maize (Arif et al. 2017). Zhang et al. (2020) reported that biochar increased NUE (20–53%) and PUE (38–230%), compared to N fertilization, in a rice-wheat rotation during a 6-year field experiment. Application of woody biochar (20%) increased NUE of green bean crops (Prapagdee and Tawinteung 2017). Indirectly, biochar increased NUE by reducing leaching of nutrients (Cheng et al. 2018), decreasing gas emissions (Li et al. 2017a), and increasing soil organic carbon (Arif et al. 2017). Addition of biochar (up to 20 t ha^{-1}) increased NUE and PUE by 90 and 191%, respectively (Table 6). Application of several types of biochars (coffee waste, *Dalbergia sissoo*, acacia prunings, maize stalk, chicken litter, mixed wood, and cuttings of acacia) at different levels (2–30 t ha^{-1}) increased the NUE (65–90%) and PUE (44–150%) (Table 6). Nonetheless, application of mixed (70% Norway spruce + 30% European beech) biochar in field crops reduced NUE by 6.09-8.01%, (Table 6) which was due to the presence of polyaromatic hydrocarbons (PAHs) in biochar that reduced the N availability for plants (Haider et al. 2017). Usually, biochar improves NUE in plants (Li et al. 2017a).

6. Conclusion and Future Research Recommendations

Biochar can be an important source of plant nutrients and can supply macro-nutrients, secondary nutrients, and micronutrients to plants. Biochar has unique physical and chemical properties that influence nutrient interactions in soil by altering soil properties including pH and CEC. The availability of nutrients in soil with biochar mainly depends on the feedstock type of the biochar, pyrolytic conditions, rate of biochar addition to soil, and the type of soil. Animal manures and waste-derived biochars have higher N, P, and K contents than crop residues and woody biochars. Moreover, manure and waste (municipal and industrial) derived biochars contain more micronutrients than crop residues and woody biochars. Availability of most nutrients are positively correlated with the pyrolytic temperature, except N and S, and that is because of volatilization loss. The effect of biochar on Ca, Mg, and micronutrient (Zn, Cu, Fe, Mn) uptake show inconsistent results. Biochar can retain P, K, and other nutrients in soil by decreasing their leaching loss. Biochar usually improves nutrient use efficiency in plants.

The following are recommendations for future research:

- ✓ Long-term field studies are needed rather than pot or column studies to understand the impact of biochar in soil.
- ✓ The feedstock selection and application rate should be studied in relation to availability of nutrients.

- ✓ Methods to increase the N content of biochar should be considered, for example by adjusting the pyrolytic conditions, because N is reduced by increasing the pyrolysis temperature.
- ✓ The availability of P as a result of different pyrolytic temperatures needs to be studied.
- ✓ Studies are needed to understand the interaction of biochar and microbes and how they affect nutrient transformation.

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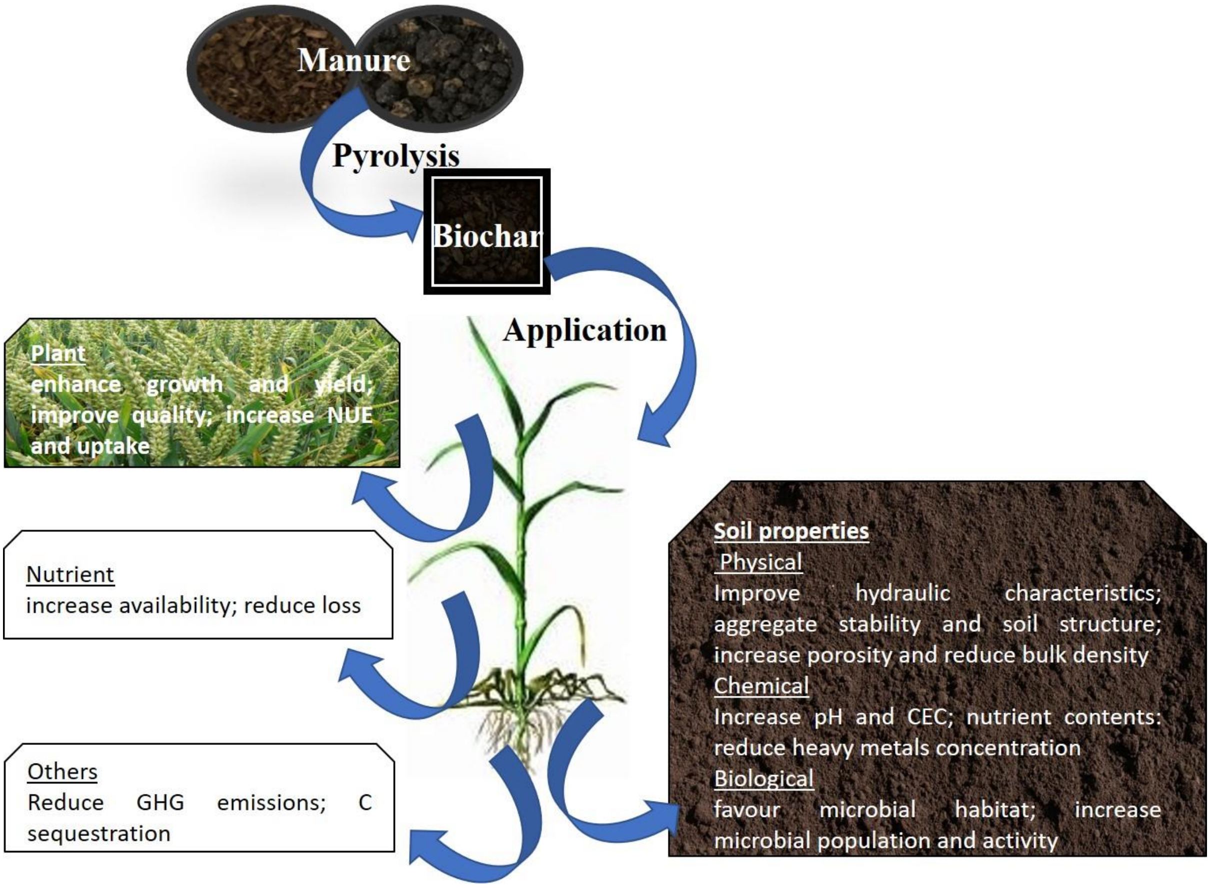

Figure 1: Conceptual framework for impact of biochar on soils and plants

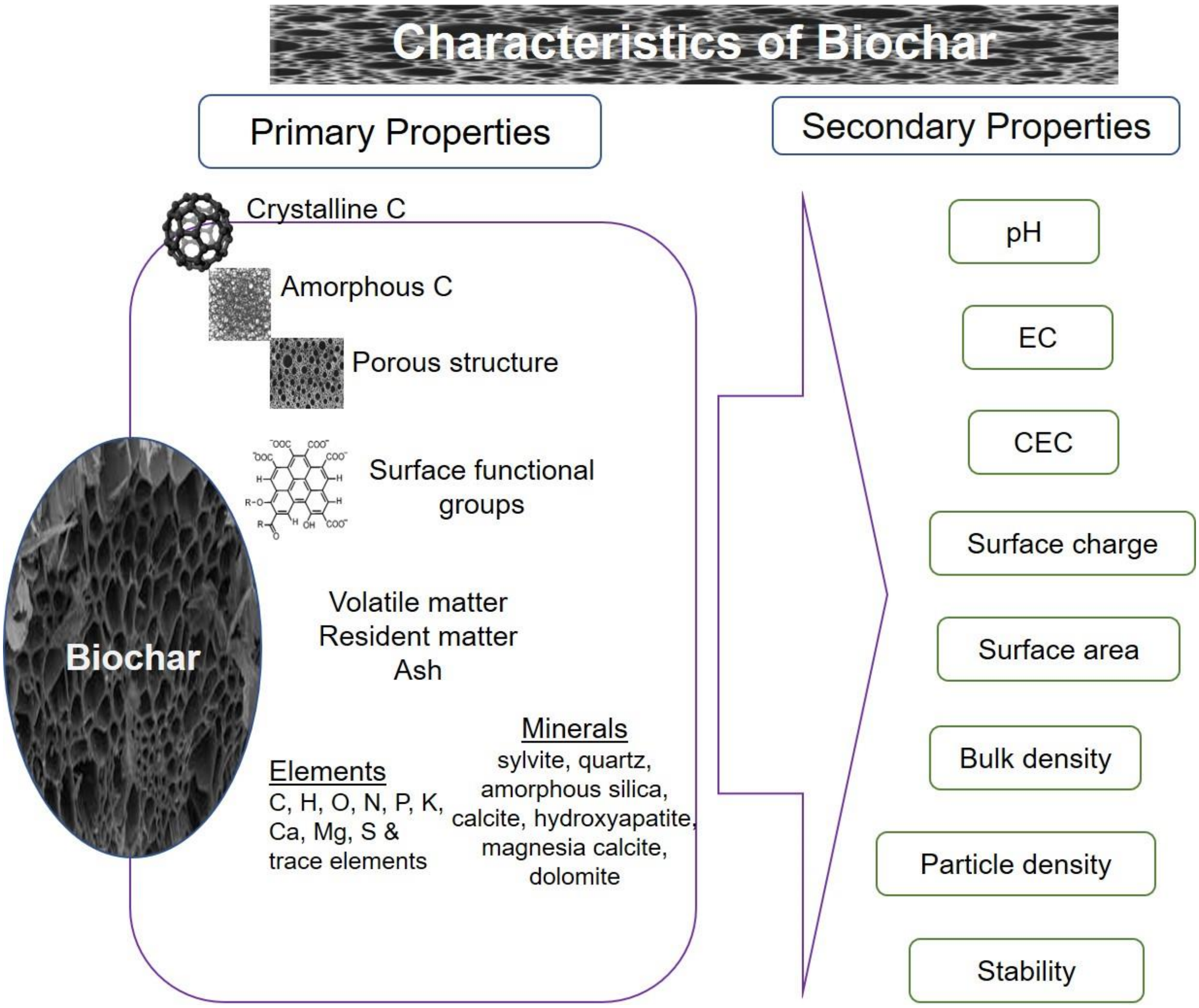

Figure 2: Properties of biochar [modified from (Igalavithana et al. 2018; Xu et al. 2017)]

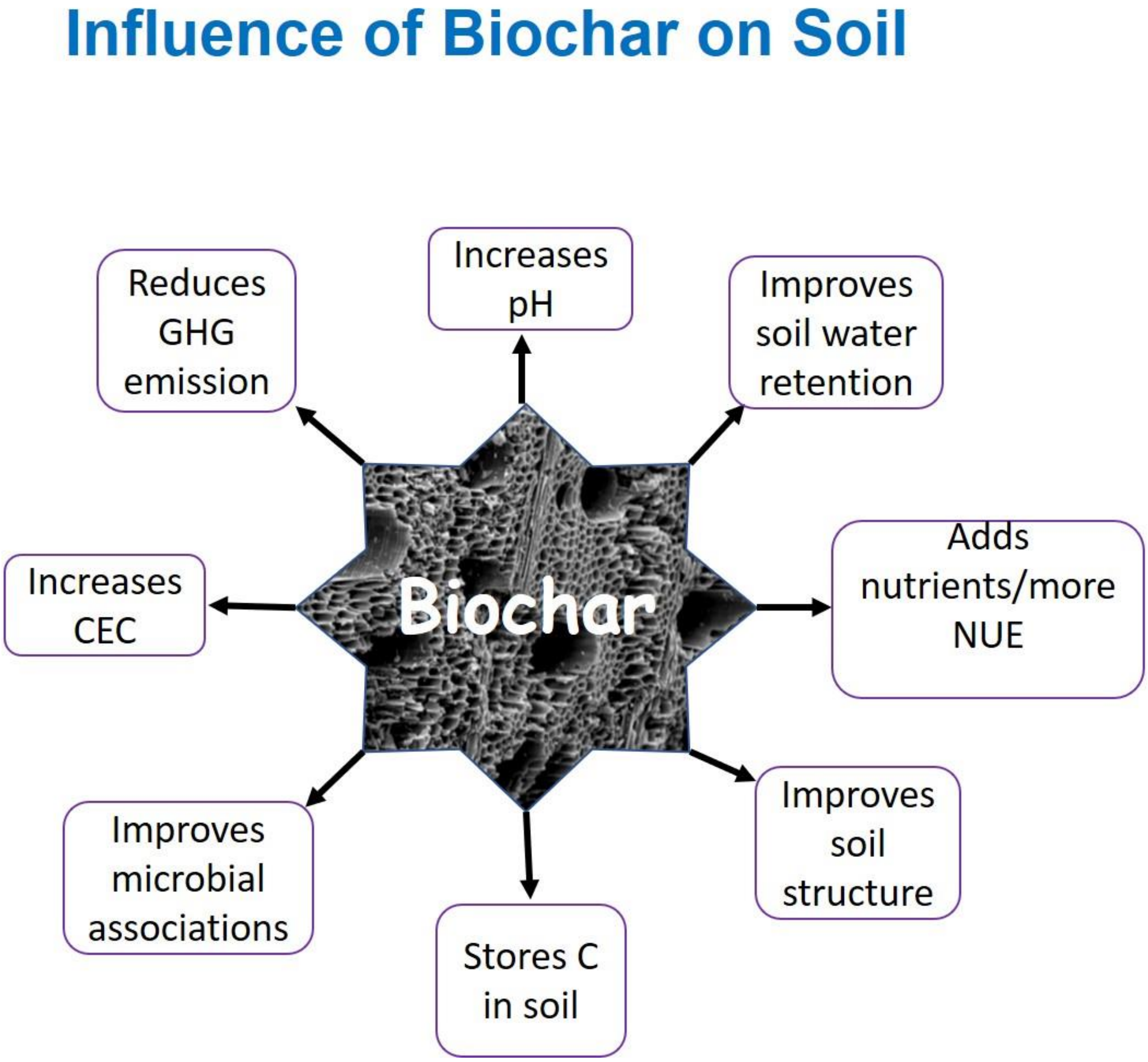

Figure 3: Influence of biochar on soil properties [adapted from Lopez-Capel et al. (2016)]

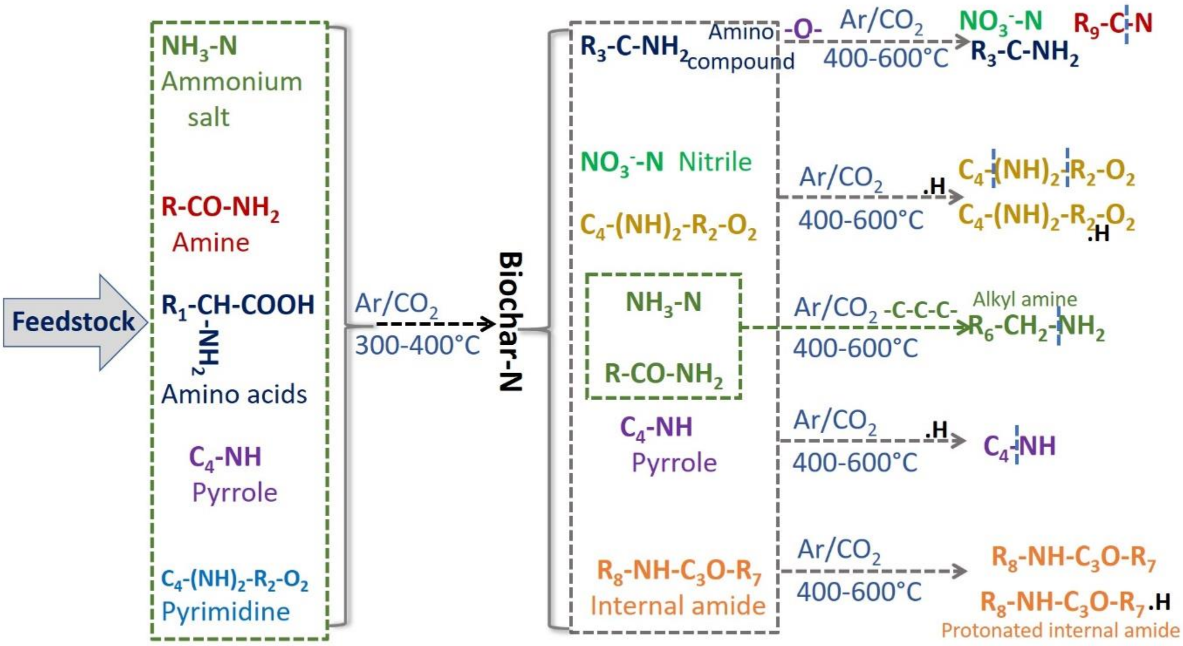

Figure 4: Nitrogen conversion pathways from feedstock-N to biochar-N through pyrolysis process (Leng et al. 2020)

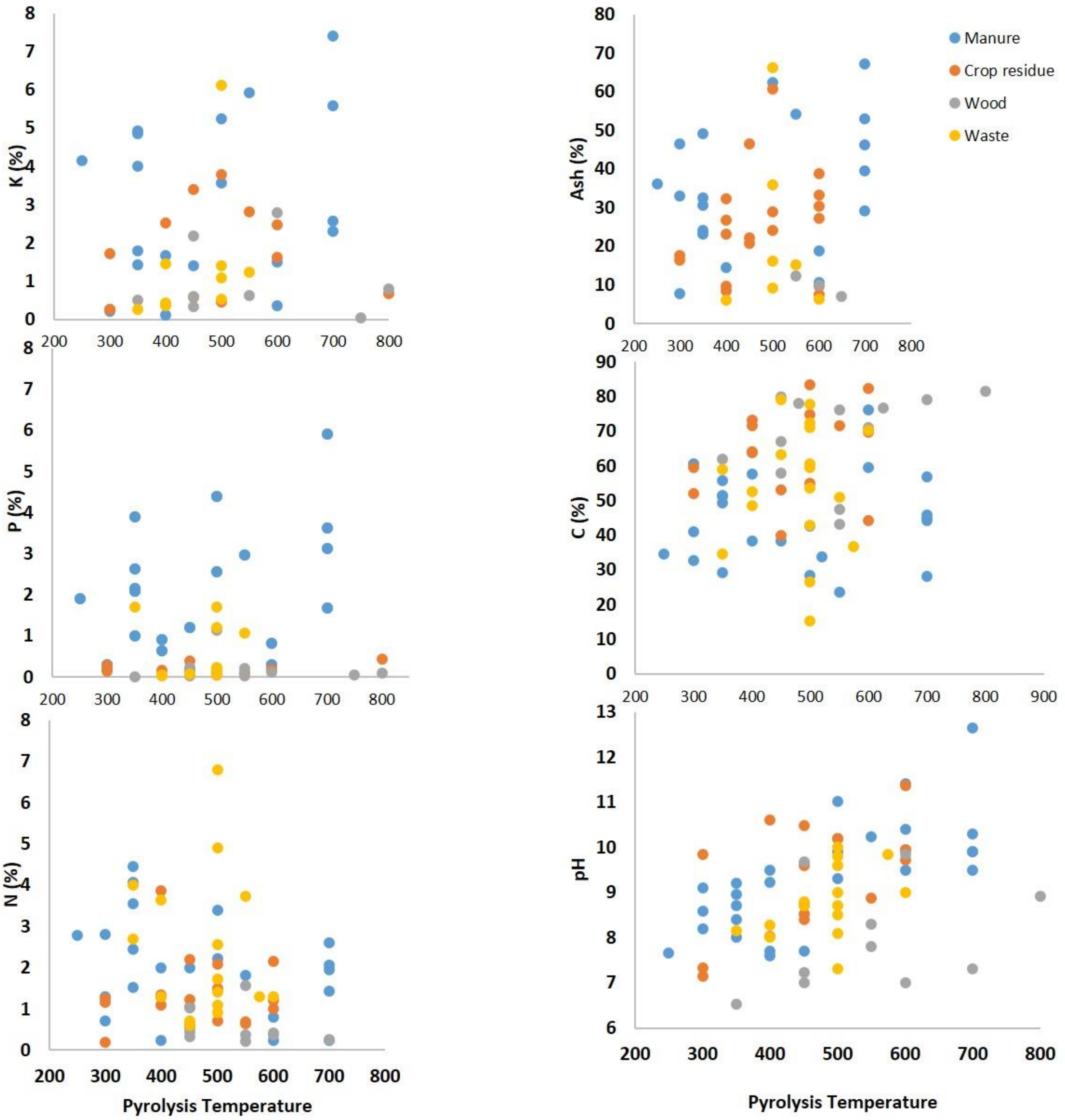

Figure 5: Impact of feedstock and pyrolysis temperature on biochar chemical properties (data obtained from Table 1)

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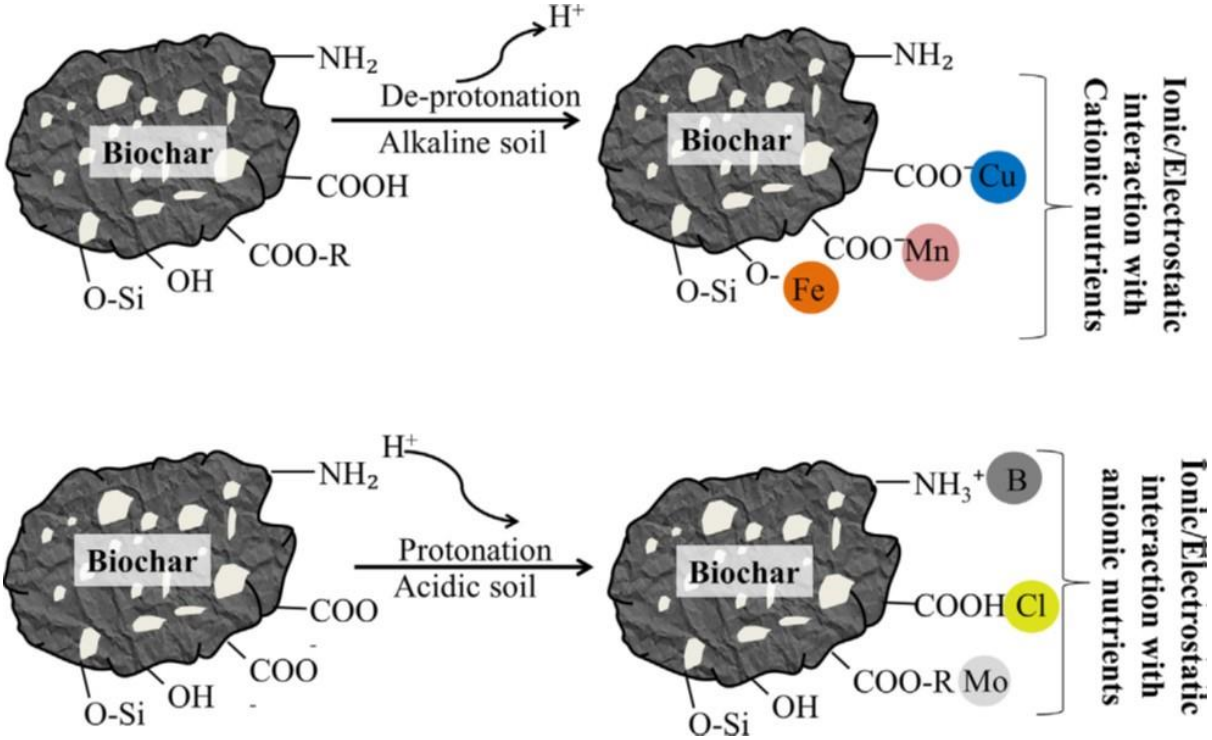

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1507 Figure 6: pH-dependent association and dissociation of nutrients from biochar (Sashidhar et al. 2020)

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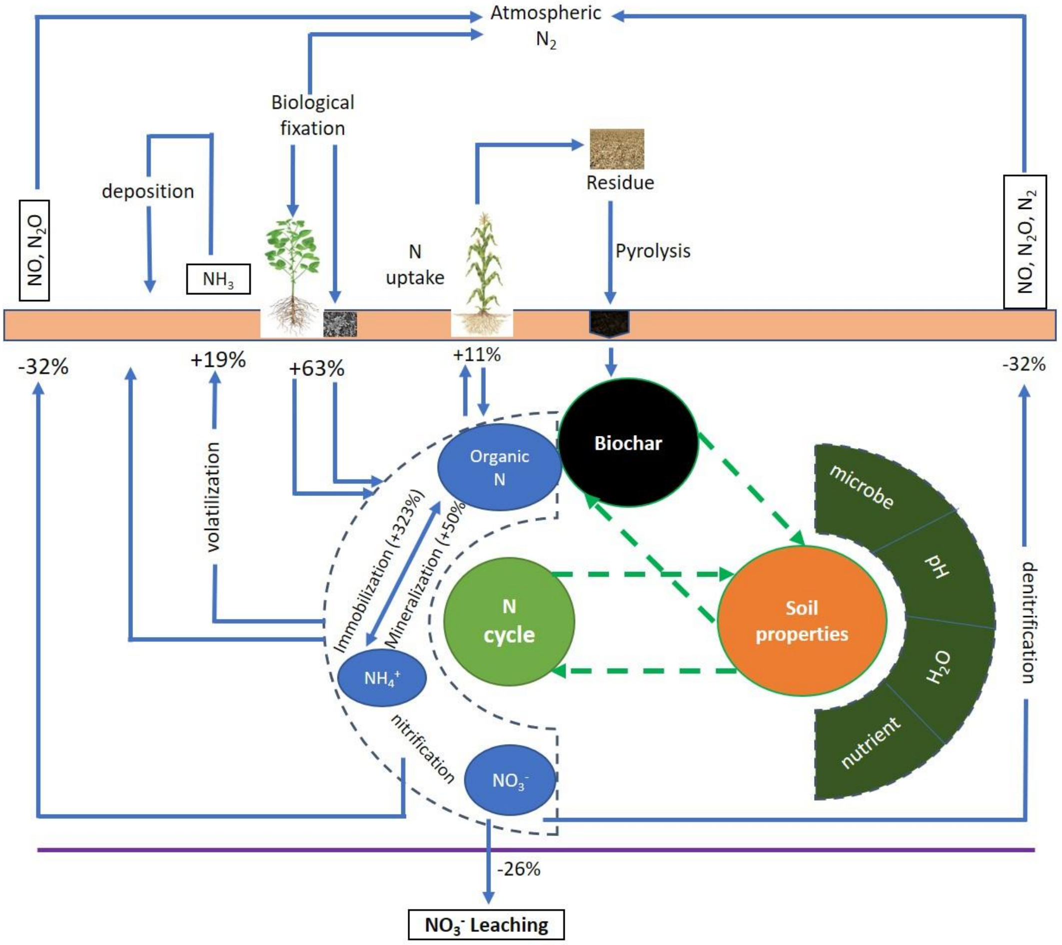

Figure 7: Conceptual framework of biochar mediated N cycle [modified from Liu et al. (2018)]

Tables

Table 1: Effect of biochar on soil physical properties

Biochar	PT(°C)	Application rate (t/ha)	Soil type	Aggregate stability (%)	Temperature (°C)		Porosity (%)		Water content (%)		Bulk density (g cm^{-3})		Reference
					Control	Amended	Control	Amended	Control	Amended	Control	Amended	
Wheat straw	400	20 40	Irragric Anthrosols	43-48							1.21	1.15 1.14	Zhang et al. (2020)
Maize straw	550	20 40 60	Silt loam		8.5	8.8 8.9 9.0			21.5	23.3 23.4 23.1	1.34	1.29 1.26 1.22	Yan et al. (2019)
Rice husk	350-400	3 6 12	Alfisol	10 18 23							1.60	1.55 1.51 1.44	Oladele (2019)
Paper fiber sludge + grain husks	550	10 20	Haplic Luvisol		17.2	17.5 17.3			15	15.5 17.2			Horák et al. (2019)
Green cuttings	650	3 40	Sandy Cambisol								1.30	1.30 1.27	Greenberg et al. (2019)
Corn straw	500	3 6 9 12	-				49	50 52 54 58					Fu et al. (2019)
Date palm residue	300 400 500 600	8	Loamy sand						25	50 42.5 35 32.5			Alotaibi and Schoenau (2019)
Macadamia nutshell		3	Sand								1.68	1.61	Lim and Spokas (2018)
Pine chip	-	3	Sand								1.68	1.58	Lim and Spokas (2018)
Maize straw	400-500	10 20 40 60	Sandy loam				42.5	48 50 55 56			1.53	1.40 1.32 1.20 1.18	Li et al. (2018)
Barley straw	400	10	Loam				51	55			1.4	1.3	Kang et al. (2018)
Poultry manure	450	5	Alfisol	18							1.60	1.44	Are et al. (2018)

Table 2: Effect of biochar on selected soil chemical properties

Biochar	PT(°C)	Application rate (t/ha)	Soil type	pH		CEC (cmol/kg)		OM (%)		Reference
				Control	Treatment	Control	Treatment	Control	Treatment	
Wheat straw	500	20 40	Irragric Anthrosols	7.00	7.10 7.40			2.57	3.28 3.97	Wu et al. (2019)
Bamboo	450	11.25 45 180	-	4.67	4.80 4.95 5.30			0.7	1.25 1.90 3.55	Tarin et al. (2019)
Hardwood	420	11.25 45 180	-	4.67	4.70 4.90 5.15			0.7	1.13 2.25 4.50	Tarin et al. (2019)
Rice straw	500	11.25 45 180	-	4.67	4.90 4.95 5.45			0.7	1.00 1.90 2.55	Tarin et al. (2019)
Rice straw	400	72	Ultisol	5.00	4.80					Shi et al. (2019)
Peanut straw	400	72	Ultisol	5.80	5.30					Shi et al. (2019)
Rice husk	500	22.5 67.5	Typic Hapludalfs	6.71	6.84 7.20	12.17	13.28 14.44	1.90	2.33 3.22	Ghorbani et al. (2019)
Rice husk	500	22.5 67.5	Typic Hapludepts	4.36	4.76 5.06	5.71	6.87 7.40	0.91	2.03 2.45	Ghorbani et al. (2019)
Chicken manure	535	6.43	Aquic Hapludults	6.69	6.81	6.28	7.01			Clark et al. (2019)
Chicken manure	535	4.23	Typic Hapludalfs	5.10	5.61	11.3	12.1			Clark et al. (2019)
Winter grass	450	45 90 135 180	Entisol	7.70	7.80 7.80 7.90 7.90	14.3	18.2 23.9 27.4 29.6	0.86	1.21 3.45 3.97 6.55	Yadav et al. (2018)
Winter grass	850	45 90 135 180	Entisol	7.70	7.90 8.00 8.10 8.30	14.3	17.2 20.2 24.3 27.1	0.86	0.86 2.07 3.97 6.03	Yadav et al. (2018)

Table 3: Effect of biochar on soil biological properties

Biochar (rate)	Temp. (°C)	Soil type	Study	Biological properties or microbial response	References
Wheat straw (1%)	400	Fimi-Orthic Anthrosol Ferralic Cambisol	Incubation	• fresh biochar reduced ammonia-oxidizing archaea (AOA) but increased ammonia-oxidizing bacteria (AOB) gene populations in acidic soil • aged biochar increased AOA- and AOB- in both soils	Zhang et al. (2019c)
Peanut shell (2%)	400	Yellow-brown Fluvo-aquic Luo Black	Incubation	• increased bacterial diversity but decreased fungal diversity • *Fusurium* population reduced by biochar plus chemical fertilizers	Zhang et al. (2019b)
Rice straw	500	Sandy loam	Field	• no effect on AOA but AOB abundance and diversity increased	Zhang et al. (2019a)
Rice straw	350 480	Clinosol	Field	• Lactobacillales and Bacteroidales population increased	Yan et al. (2019)
Corn straw (1.33%)	500	Sandy loam	Pot	• improved antagonistic percentage and antagonistic ability of *Bacillus* spp. And *Pseudomonas* spp.	Wang et al. (2019)
Straw of reed, smooth grass and rice	450	Clay	Pot	• increased microbial biomass • decreased microbial activity and soil respiration	Tian et al. (2019)
Moso bamboo (20 and 40 t/ha)	600	Ferrisol	Field	• reduced urease and acid phosphatase activities	Peng et al. (2019)
Chicken manure, oat hull, pine bark (3%)	300 500 600	Alfisol		• increased basal respiration and dehydrogenase (DHA) activity and modified microbial communities.	Meier et al. (2019)
Wheat straw (40 t/ha)	350-550	Anthrosol	Incubation	• fresh biochar increased microbial biomass C (MBC) • aged biochar decreased Gram-positive/Gram-negative ratio	Liu et al. (2019b)
Rice straw (4 and 20 t/ha)	550-650	Vertisol	Field	• increased the nifH (nitrogenase iron protein) gene abundance and altered the community structure of soil diazotrophs.	Liu et al. (2019a)
Corn straw (2.4, 6 and 12 t/ha)	400	Inceptisol	Field	• improved growth of Gram-positive bacteria and fungi • increased MBC and influenced the soil microbial community structure	Li et al. (2019)
Wheat stalk (1 and 5%)	650	Ge-Eutric Gleysols		• strengthened network connectivity among rhizosphere bacteria • improved linkage between rhizosphere bacteria and soil C	Huang et al. (2019)

Bamboo biomass (5, 10 20 t/ha)	350-400		Field	• reduced the Proteobacterial community in soils	Herrmann et al. (2019)
Sewage sludge (15 t/ha)	300 500	Red-Yellow Latosol	Field	• increased mycorrhizal colonization in corn plant	de Figueiredo et al. (2019)
Conifer wood chips (5 and 10%)	280	Cambisol	Incubation	• decreased DHA, β-glucosidase and phosphatase activities	Cordovil et al. (2019)

Table 4: pH and nutrient contents of biochar produced at different pyrolysis temperature

Feedstock	PT^1(°C)	pH	C	N			C/N	P			K	Ca	Mg	S	Zn	Cu	Fe	Mn	Mo	B	Reference
				TN	Available NH$_4$-N (g kg^{-1})			TP	Available P (g kg^{-1})												
					H$_2$O extract	KCl extract			H$_2$O extract	Olsen-P											
										%					mg/kg						
Manure																					
Chicken manure	250	7.66	34.55	2.79	0.07	0.16	12	1.91	5.08	6.76	4.16	1.98	2.14	-	-	-	-	-	-	-	Xiao et al. (2018)
Chicken manure	350	8.95	29.21	2.45	0.07	-	12	2.15	4.57	8.42	4.93	2.17	2.84	-	-	-	-	-	-	-	Xiao et al. (2018)
Chicken manure	550	10.24	23.65	1.81	-	-	13	2.96	2.93	8.74	5.93	3.03	3.78	-	-	-	-	-	-	-	Xiao et al. (2018)
Chicken manure-Ca*	250	7.84	30.00	2.85	0.27	0.48	11	1.83	2.49	3.17	4.14	4.05	1.67	-	-	-	-	-	-	-	Xiao et al. (2018)
Chicken manure-Ca*	350	9.32	26.68	2.44	0.01	0.03	11	2.21	1.23	8.68	4.87	4.91	2.18	-	-	-	-	-	-	-	Xiao et al. (2018)
Chicken manure-Ca*	550	10.61	24.73	1.96	-	-	13	3.06	-	1.22	6.03	5.91	2.67	-	-	-	-	-	-	-	Xiao et al. (2018)
Chicken manure-Mg$^{\#}$	250	7.35	26.40	2.43	0.40	0.30	11	2.05	5.65	6.98	3.92	2.24	4.17	-	-	-	-	-	-	-	Xiao et al. (2018)
Chicken manure-Mg$^{\#}$	350	9.17	26.22	2.42	-	-	11	2.67	3.33	8.36	5.03	2.81	4.73	-	-	-	-	-	-	-	Xiao et al. (2018)
Chicken manure-Mg$^{\#}$	550	10.32	27.04	2.06	-	-	13	3.03	0.05	1.27	5.88	3.09	5.22	-	-	-	-	-	-	-	Xiao et al. (2018)
Chicken manure-Fe$^{@}$	250	5.75	28.26	2.91	0.44	0.17	10	2.01	1.27	1.23	3.92	2.03	2.03	-	-	-	-	-	-	-	Xiao et al. (2018)
Chicken manure-Fe$^{@}$	350	5.72	26.44	2.45	0.02	-	11	2.44	1.24	1.23	5.06	2.53	2.88	-	-	-	-	-	-	-	Xiao et al. (2018)
Chicken manure-Fe$^{@}$	550	6.68	27.13	2.17	-	-	13	3.10	0.09	0.51	5.95	3.18	3.87	-	-	-	-	-	-	-	Xiao et al. (2018)
Chicken manure	450	7.7	38.3	2.0			19	1.2			1.4	-	-	-	-	-	-	-	-	-	Madiba et al. (2016)
Poultry litter	500-520	9.30	33.72	3.39			10	2.57			5.24	4.54	1.26	1.36	829.50	583	-	715			Brantley et al. (2016)
Poultry Litter	400	9.5	52.1	5.85			9	1.22			3.88	2.83	1.73	0.08	-	-	-	-	-	-	Subedi et al. (2016)
Poultry Litter	600	10.4	52.8	4.0			13	1.54			5.88	3.59	2.4	0.08	-	-	-	-	-	-	Subedi et al. (2016)
Poultry litter	350	8.70	51.1	4.45			11	2.08			4.85	2.66	0.94	0.61	712	213	13200	640	11	-	Cantrell et al. (2012)
Poultry litter	400	7.70	38.3	2.0			19	0.90			1.0	2.5	0.30	-	238	57	2695	265	5	-	Macdonald et al. (2014)
Poultry litter	700	10.3	45.9	2.07			22	3.12			7.4	0.40	1.45	0.63	1010	310	18900	948	13	-	Cantrell et al. (2012)
Turkey litter	350	8.00	49.3	4.07			12	2.62			4.01	4.04	0.85	0.55	690	535	27800	710	7.16		Cantrell et al. (2012)

Turkey litter	700	9.90	44.8	1.94			23	3.63			5.59	5.61	1.24	0.41	909	762	36500	986	10.1		Cantrell et al. (2012)
Cow manure	300	8.59	41.02	0.71			58	0.19			0.26	-	-	-	-	-	-	-	-	-	Beheshti et al. (2017)
Bull manure	300	8.20	60.6	1.3			47	0.30			0.20	0.94	0.40	0.11	162	-	376	137	-	-	Enders et al. (2012)
Bull manure	600	9.50	76.0	0.8			95	0.30			0.36	0.94	0.51	0.10	193	-	311	165	-	-	Enders et al. (2012)
Digested dairy manure	400	9.22	57.7	0.24			240	0.65			1.66	2.26	0.97	0.27	131	-	1656	145	-	-	Enders et al. (2012)
Digested dairy manure	600	9.94	59.4	0.23			258	0.83			1.49	2.65	0.85	0.29	200	-	2356	191	-	-	Enders et al. (2012)
Dairy manure	350	9.2	55.8	1.51			37	1.0			1.43	2.67	1.22	0.11	361	99	26700	525	7.8	-	Cantrell et al. (2012)
Dairy manure	700	9.9	56.7	0.24			236	1.69			2.31	4.48	2.06	0.15	423	163	44800	867	10	-	Cantrell et al. (2012)
Swine manure	300	9.11	32.58	2.80			12	-			-	-	-	-	-	-	-	-	-	-	Xu et al. (2019)
Swine manure	500	11.02	28.43	2.21			13	-			-	-	-	-	-	-	-	-	-	-	Xu et al. (2019)
Swine manure	700	12.64	28.23	1.42			20	-			-	-	-	-	-	-	-	-	-	-	Xu et al. (2019)
Swine manure	400	7.6	54.9	2.23			24.6	0.98			1.62	2.03	1.57	0.02	-	-	-	-	-	-	Subedi et al. (2016)
Swine manure	600	11.4	57.9	1.79			32.4	1.55			3.53	2.89	2.13	0.04	-	-	-	-	-	-	Subedi et al. (2016)
Pig manure	500	9.90	42.7	-				4.39			3.56	3.47	2.80	-	1010	780	6960	1230	-	-	Zhao et al. (2018)
Swine solids	350	8.40	51.5	3.54			15	3.89			1.78	3.91	2.44	0.80	3181	1538	48400	1453	18.3	-	Cantrell et al. (2012)
Swine solids	700	9.50	44.1	2.61			17	5.9			2.57	6.15	3.69	0.85	4981	2446	74800	2240	27.4		Cantrell et al. (2012)
Crop residue																					
Rice husk	450	8.53	39.90	0.54			74	0.16			0.58	-	-	-	-	-	-	-	-	-	Bu et al. (2017)
Rice husk	-	9.50	-	0.1			-	0.15			0.20	-	-	-	-	-	-	-	-	-	Jatav et al. (2018)
Barley straw	400	8.02	71.50	1.3			55	-			-	0.20	-	-	-	-	-	-	-	-	Kang et al. (2018)
Rice straw	550-650	9.71	44.27	0.64			69	0.09			2.82	-	-	0.24	-	-	-	-	-	-	Si et al. (2018)
Wheat straw	300	7.15	52.12	0.2			261	0.27			0.25	-	-	-	-	-	-	-	-	-	Beheshti et al. (2017)
Wheat straw	350-550	9.60	-	1.05			-	-			-	-	-	-	-	-	-	-	-	-	Zheng et al. (2017)
Wheat chaff	450	8.40	53.1	2.2			24	0.40			3.40	-	-	-	-	-	-	-	-	-	Madiba et al. (2016)
Maize straw	300	9.84	-	1.25			-	-			-	-	-	-	-	-	-	-	-	-	Song et al. (2018)
Maize straw	450	10.47	-	1.22			-	-			-	-	-	-	-	-	-	-	-	-	Song et al. (2018)
Maize straw	600	11.37	-	1.21			-	-			-	-	-	-	-	-	-	-	-	-	Song et al. (2018)
Corn stalks	500-600	8.87	71.50	0.69			104	-			1.61	-	-	-	-	-	-	-	-	-	Yao et al. (2017)

Wheat straw and peanut shell	500	10.20	83.40	1.5			56	-			-	-	-	0.30	-	-	-	-	-	-	El-Naggar et al. (2018)
Elephant grass	400	-	63.86	3.87			17	-			-	-	-	-	-	-	-	-	-	-	Ferreira et al. (2018)
Elephant grass	500	-	74.85	2.08			36	-			-	-	-	-	-	-	-	-	-	-	Ferreira et al. (2018)
Elephant grass	600	-	82.23	2.15			38	-			-	-	-	-	-	-	-	-	-	-	Ferreira et al. (2018)
Kunai grass	500	10.20	55.00	0.7			79	0.10			0.46	-	-	-	-	-	-	-	-	-	Baiga and Rao (2017)
Switch grass	400	-	73.10	1.35			54	-			-	-	-	0.32	-	-	-	-	-	-	Purakayastha et al. (2016)
Corn stover	300	7.33	59.5	1.16			51	0.14			1.71	0.65	0.59	0.07	132	-	963	142	-	-	Enders et al. (2012)
Corn stover	600	9.95	69.80	1.01			69	0.18			2.46	0.94	0.86	0.08	70	-	1362	226	-	-	Enders et al. (2012)
Soybean	500	-	-	-			-	0.06			3.78	1.57	1.17	0.11	28	-	699	58	-	-	Enders et al. (2012)
Pearl millet	400	10.60	64	1.1			58	0.16			2.52	1.47	1.06	0.22	-	-	-	-	-	-	Purakayastha et al. (2015)
Wood																					
Sugar maple sawdust	450	7.22	80.00	0.32			250	0.02			0.32	0.50	0.06	-	23.90	5.01	49.70	368	-	-	Noyce et al. (2017)
Eucalyptus camaldulensis Traditional kiln	350	6.52	61.86	-			-	0.005			0.51	0.54	0.04				500				Butnan et al. (2015); Butnan et al. (2018)
Eucalyptus camaldulensis Flash carbinization	800	8.92	81.50	-			-	0.09			0.78	1.04	0.06	-	-	-	229	-	-	-	Butnan et al. (2015)
Apple branch	450	9.67	67.01	0.57			118	0.18			0.60	2.42	0.32	-	37.30	9.90	5745.80	91.50	-	-	Li and Shangguan (2018)
Castor stalk	550	-	43.18	1.57			27	0.22			0.62	0.90	-	-	-	-	-	-	-	-	Hilioti et al. (2017)
Bamboo	600	9.84	70.90	0.41			173	0.11			2.78	-	-	0.46	-	-	-	-	-	-	Lu et al. (2018)
Hardwood	550	7.80	76.00	0.22			345	0.02			-	-	-	0.08	-	-	-	-	-	-	Nguyen et al. (2018)
Cashew wood residue	-	-	-	0.94			-	0.01			0.13	-	-	-	18.45	10.21	185.04	32.27	-	-	Miranda et al. (2017)
Hardwood	600-650	7.00	76.60	0.38			201	-			-	-	-	-	-	-	-	-	-	-	Aller et al. (2017)
Eucalypt green waste	650-750	7.30	79.00	0.26			303	0.04			0.03	0.05	0.02	0.08	200	12	7000	180	-	6.10	Abujabhah et al. (2016)
Willow wood waste	550	8.30	47.50	0.38			125	-			-	-	-	0.19	83.50	2.55	0.05	110	<0.30	9.25	Agegnehu et al. (2016)
Acacia	400-500	7.01	57.80	1.02			57	1.14			-	0.27	0.001	-	-	-	-	-	-	-	Arif et al. (2016)

Macadamia shell	450-480	8.76	78.03	0.43			181	0.24			2.19	0.37	0.17	-	-	-	1211	-	-	-	Wrobel-Tobiszewska et al. (2015)
Industrial and Municipal Waste																					
Sugarcane bagasse	450-500	8.79	63.27	0.67			94	0.07			-	-	-	-	-	-	-	-	-	-	Gerdelidani and Hosseini (2018)
Sugarcane bagasse	350	-	59	4			15	-			-	-	-	-	-	-	-	-	-	-	Batista et al. (2018)
Castor cake	550	-	50.81	3.73			14	1.07			1.23	0.37	-	-	-	-	-	-	-	-	Hilioti et al. (2017)
Sewage sludge	350	8.15	34.56	2.7			13	1.70			0.26	-	-	-	-	-	-	-	-	-	Khanmohammadi et al. (2017)
Sewage sludge	500	7.30	43.0	6.8			6	0.11			-	-	-	-	-	-	-	-	-	-	Gonzaga et al. (2019)
Sewage sludge	500	8.10	26.6	-			-	1.70			0.52	6.57	0.64	-	1520	380	22100	450	-	-	Zhao et al. (2018)
Sewage sludge	500	8.70	15.26	1.73			-	-			-	-	-	-	-	-	-	-	-	-	Yue et al. (2017)
Orchard pruning biomass	500	9.80	77.8	0.91			63.5	0.23			1.39	2.5	2.87	0.005	0.01	0.009	0.033	0.008	-	-	Baronti et al. (2014)
Leave waste	500	9.00	60.7	1.1			55	0.21			1.08	5.46	0.36	0.10	70	-	1504	555	-	-	Enders et al. (2012)
Grass waste	500	9.60	53.5	4.9			11	1.20			6.13	2.06	0.63	0.63	150	-	1557	360	-	-	Enders et al. (2012)
Food waste	400	8.27	52.4	3.65			14	0.05			1.46	5.17	0.53	0.08	39	-	4431	179	-	-	Enders et al. (2012)
Orange bagasse	500	10.00	72.3	2.55			28	0.05			-	-	-	-	-	-	-	-	-	-	Gonzaga et al. (2019)
Coffee waste	400-500	8.7	79	0.7			113	0.03			0.35	0.40	0.08	0.03	45	15	150	40	-	-	Prakongkep et al. (2015)
Bagasse	400-500	8.7	71	0.6			118	0.08			0.43	1.20	0.21	0.03	400	15	4800	300	-	-	Prakongkep et al. (2015)
Sugarcane filtercake	575	9.85	36.7	1.3			28	-			-	-	-	-	-	-	-	-	-	-	Eykelbosh et al. (2014)
Municipal solid waste	400	8.00	48.6	1.3			37	-			-	-	-	0.1	149	63	-	-	-	-	Jin et al. (2014)
Municipal solid waste	500	8.50	59.5	1.4			43	-			-	-	-	-	213	101	-	-	-	-	Jin et al. (2014)
Municipal solid waste	600	9.00	70.1	1.3			54	-			-	-	-	0.1	356	157	-	-	-	-	Jin et al. (2014)

1= Pyrolysis temperature *= $CaCl_2$ # = $MgCl_2.6H_2O$, @ = $FeCl_3.6H_2O$

Table 5: Biochar and nutrient availability changes in different soils

Expt. condition	Soil type/test crop	BC source	PT (°C)	BC rate	Nutrient availability changes over control (%)									References
					C	NO$_3$-N	NH$_4$-N	Tot. N	Avail. P	Tot. P	K	Secondary	Minor	
		Woody												
Incubation	Silty clay loam	Yellow pine	550	10 Mg/ha		2.04 (-)								(Baechle et al., 2018)
Pot	Khorat and Wahiawa/maize	upper branches of eucalyptus trees	350 and 800	1, 2, 3, 4%	708.54 (+) 271.90 (+)						145.24(+) 13(+)	Ca(268) (133.53) (+) Mg(106.14)(+) Mg(89.49)(-)	Mn(311.61) (+)	Butnan et al. (2018)
Field	Silt-clay/wheat	Apple branch	450	0, 1, 2, 4, 6%		48.66–256.49(+) 72.52-85.53(-)	46.42(+) 18.38(-)		32.34–51.41(+) 13.70–35.13(0-)					Li and Shangguan (2018)
Pot	Yellow loamy/rice	Bamboo	600	0.16 kg/pot	228.41 (+)	22.61(+)	41(+)	9.51(+)	3.54(-)		191.13(+)			Lu et al. (2018)
Field	Ferralsol/Forage peanut	Hardwood	550	10 t/ha	21.60(+)	42.75(+)	24.06(-)	2.63(+)						Nguyen et al. (2018)
Field	Silty loam/maize-mustard	*Eupatorium adenophorum*	450-500	5, 10, 15, 20, 40 t/ha	175.69 (+)			11(+)	422.4(+)		80.95(+)	Ca(78.26)(+) Mg(60.66)(+)		Pandit et al. (2018)
Pot	*Brassica juncea*	Oak wood	400	5%							61535.07 (+)	Mg(1158.92)(+)	Mn(2702)(+) Cu, Fe(-)	Rodriguez-Vila et al. (2017)
Pot	Sand, sandy loam and silty sand/sugar maple and red pine	Sugar maple sawdust	450	5, 20, 50 t/ha	64.5(+)				0.97(-)		92.5(+)	Ca(3)(-) Mg(4.05)(-)	Mn(17.1)(+)	Noyce et al. (2017)
Greenhouse	Yellow Latosol/rice and cowpea	Cashew wood residue		3.5, 7, 10 t/ha	19.4(-)					28.97(+)	46.12(+)	Ca and Mg(3.39)(-)		Miranda et al. (2017)
Field	Clay and loamy sand/chickpea	*Acacia nilotica* *Eucalyptus obliqua*	450-550	0, 5, 10, 20 t/ha	57.52(+)			33.33(+)		42.17(-)	5.13(+)	Ca(3.38)(-) Mg(3.57)(+)		Lusiba et al. (2017)
Greenhouse	Courval sandy loam	Softwood chips	500	20 Mg/ha			15.16(+)							Backer et al. (2017)
Field	Fine-loamy, mixed, superactive/corn, soybean and switchgrass	Hardwood	600-650	22.4 t/ha	37(+)			26(+)			11 (+)			Aller et al. (2017)

Pot	Loam; sandy loam; clay loam	Eucalypt green waste	650-750	2.5, 5, 10%		103(+) 110(+) 207(+)	53 (-) 32 (-) 61 (-)				8(+) 16(+)	Ca(19)(+) Mg(+)	Na(28) (+) Al(68)(-) Fe(13)(-) Cu(16) (-) B(40)(-)	Abujabhah et al. (2016)
Field	Dark reddish brown Ferrasol; Red Ferrosol/Maize	Willow wood waste	550	10 t/ha	43-73(+)	10(+)	36(+)		59-117(+)			Ca (31-54)(+)	Al (37.5)(-)	Agegnehu et al. (2016a)
Field	Acidic Eutric Nitisol/Barley	Stem, bark and branches of Acacia wood	Earth klin	2, 10 t/ha	30 (+)			15(+)		29(+)	17(+)	Ca (23)(+) Mg (16) (+)		Agegnehu et al. (2016b)
Field	Silty clay loam/maize	Acacia	400-500	25, 50 t/ha	483.33 (+)			66.67(+)		200 (+)				Arif et al. (2016)
		Crop residue												
Field	Loam/Chinese cabbage	Barley straw	400	10 t/ha				20.86(+)	9.76(+)		24(+)	Ca(9.81)(+) Mg(32.26)(+)		Kang et al. (2018)
Field	Silt loam/rice	Rice straw	550-650		0.40(+)			1.90(+)	32(+)		22.79(+)	Ca(2.47) (+) Mg(4.80)(-)		Si et al. (2018)
Pot	Calcareous	Maize straw	300 450 600	1%	247.41 (+)			42.37(+)	105.32(+)		469.73(+)			Song et al. (2018)
Field	Sandy loam/wheat-maize	Wheat straw	350 550	40, 50, 100%	7.66(+)			16.46(+)	119.10(+)					Zheng et al. (2017)
Field	Clay loam/soybean-maize	Corn stalks	500-600	0, 2, 4, 8%	349.26 (+)	119.35(+)	2.22(-)	120.39(+)	15.78(+)	17.86(+)	9.11(+)			Yao et al. (2017)
Greenhouse	Alluvial soil/rice	Rice husk		2.5, 5, 7.5, 10, 15, 20 t/ha	70(+)								Fe(54.63)(+) Cu(192.99)(+) Zn(162.64)(+) Mn(87.37)(+)	Jatav et al. (2018)
Soil column	Riparian	Rice husk	450	1, 2, 3, 10%		88.11(+)	53.35(-)	58.64(+)	85.05(+)					Bu et al. (2017)
Pot	Calcareous sandy/wheat	Corn cobs		20, 40, 60 Mg/ha	166.67 (+)				25.51(+)		75.78(+)			Amin (2016)
Incubation		Elephant grass	400 500 600	5, 15 g/L	2.11(+)					16619.75(+)	3122.03(+)	Ca(45.6)(+) Mg(460)(+) S(1579.71)(+)	Zn(17.72)(-) Cu(20.25)(-) B(114.28)(-)	Ferreira et al. (2018)

													Mn(60.98)(+)	
Microcosm	Silty clay loam	Rice straw, rice hull, and Maize stover	500	1.5, 3%	16.86(+)		161.90(+)			140 (+)	122.61(+)			Bashir et al. (2018)
Field	Fluvisol/wheat-maize	Rice husks (70%) and cotton seed hulls (30%)	400	30, 60, 90 t/ha	29-41.5(-)									Dong et al. (2018)
Field	Farmland	Wheat straw and peanut shell	500	8 t/ha							10.53(-)	Ca(1.80)(-) Mg(3.37) (+)		El-Naggar et al. (2018)
Incubation	Clay loam, loam and sandy loam	Sugar cane bagasse	450-500						23.72-63.67(+)					Gerdelidani and Hosseini (2018)
Greenhouse	Loamy, kaolinitic, thermic Grossarenic Kandiudult/maize	Coconut husks, orange bagasse and pine woodchips	500	5, 10, 20, 60 t/ha				30.39(+)	21.88(-) 13.28(+)	18.51(+)				Gonzaga et al. (2018)
Pot	Loamy/turf grass	Sewage sludge	500	0, 1, 5, 10, 20, 50%	4443.65(+)			6209.09(+)	3819.88(+)		740.77(+)			Yue et al. (2017)
Greenhouse	Calcareous	Sewage sludge	350	7.3, 14.5, 29 Mg/ha		16.11(+)		1.18(+)	32.79(+)		2.79(+)			Khanmohammadi et al. (2017)
Incubation	Silt loam	*Miscanthus* straws, coffee husks and woody material	600	1, 3%					75(+)					Houben et al. (2017)
Growth chamber	Commercial/tomato and castor bean	Castor cake and castor stalks	550	1, 5%				59.30(-) 9.05(+)		81.20(+)			Zn(32.74)(+) Mn(25.19)(-) Cu(25.52)(-) Fe(6.50)(-) Fe(17.51)(+)	Hilioti et al. (2017)
Incubation and greenhouse	Sandy clay loam/Chinese cabbage	Kunai grass	500		1900(-)	75(+)								Baiga and Rao (2017)
		Others												
Incubation	Calcareous	Wheat straw and cow manure	300 and 500	5, 10 t/ha						290.91 (+)				Beheshti et al. (2017)
Greenhouse	Loam/maize	Poultry litter	500-520	5, 10 Mg/ha				55.77(+)		27.27(+)		Ca(4.35)(-) S(75)(+)	Mn (52.67)(-)	Brantley et al. (2016)
Pot	Sandy/wheat	Wood and peanut shell –	450	1, 2 %					208(+)					Madiba et al. (2016)

		Chicken manure – wheat chaff												
Screen house	Organic/Rice	unknown		0, 2 t/ha		1.72(+)		4.45(-)						Dewi et al. (2018)

Table 6: Impact of biochar on different crops

Expt./crop	Source of biochar and application rate	PT (°C)	Changes over control (%)					Crop growth (%)	References
			Nutrient concentration	Nutrient uptake	NUE	Crop yield			
						grain	biomass		
Field/various	Rice husk (20 t/ha)	500	N(5.20)(-), (2.11)(+), P(11)(+)				9-15 (+)		Akoto-Danso et al. (2018)
Incubation and germination/rice and tomato	Rice husk (0.5, 1, 5, 10, 25, 50%)	480						Seedling emergence(17-20)(-)	Anyanwu et al. (2018)
Germination/*Quercus serrata* and *Prunus sargentii*	Oak tree and bamboo()	700-800 1200						Seedling quality index(8.3-19.9)(+)	Aung et al. (2018)
Roof and ground/Ryegrass, *Sedum lineare* and cucumber	Sewage sludge(5, 10, 15, 20%,)						54-54.2(+)	Promoted plant growth	Chen et al. (2018)
Screenhouse/Rice	Unknown(2 t/ha)						6.3-13.3(+)		Dewi et al. (2018)
Pot/Chinese melon	Pinewood(5%)							Plant height, No. of leaves and stem dia (43, 192. 60, 66.5)(+), respectively	Elbashier et al. (2018)
Greenhouse/Okra	Wheat straw(5, 10%)	350-550						Plant growth increased; salinity threshold level(81.2)(+)	Elshaikh et al. (2018)
Pot/Bean	Maple residue(5, 10%)	560	K(28.57), Ca(6.20), Mg(10)(+)				20.22(+)	Plant growth increased	Farhangi-Abriz and Torabian (2018)
Spinach	Cattle manure(1.25, 2.5, 5%)	600					51(+)	Stomatal conductance(11-63)(+)	Gavili et al. (2018)
Greenhouse/Maize	Coconut husks, orange bagasse and pine wood chip(5, 10, 20 and 60 t/ha)	500	N(0.88), P(0.15)(+)				90(+)		Gonzaga et al. (2018)
Greenhouse/Rice	Rice husk((2.5, 5.0, 7.5 10, 15, 20 t/ha))			Fe(480), Cu(570), Zn(336), Mn(322)(+)		8.5(+)	7.5(+)	Panicle length, grain/panicle, test weight, (78.37), (85.33), (34.55)(+), respectively	Jatav et al. (2018)
Field/Chinese cabbage	Barley straw(10 t/ha)	400	N(0.43), P(0.08), K(0.28)(+)	N, P, K (-)			64.9(+)		Kang et al. (2018)
Mesocosm/ Broadleaf cattail	Alder(95%), birch, oak, linden and willow(10%)						170(+)		Kasak et al. (2018)

Field/Spring barley	paper fiber sludge and grain husks (10, 20 t/ha)	550				77.78(+)	44(+)	Plant height(23.79)(+)	Kondrlova et al. (2018)
Field/Corn, cotton, peanut,	Re oak(22.4, 44.8 t/ha)	450-600				33(+)			Lamb et al. (2018)
Pot/Wheat	Apple branch(1,2,4,6%)	450				7.4-12(+) 6.25-21.83(-)			Li and Shangguan (2018)
Greenhouse/Maize	Coffee ground and coffee husk(4, 8, 12 and 16 t/ha)	530			N(71), P(44)(+)			Improve plant growth	Lima et al. (2018)
Pot/Rice	Bamboo(0.16kg/pot)	600				81.82(+)	58.82(+)		Lu et al. (2018)
Pot/Wheat-maize	Rice residue(10, 20, 40 t/ha)						40(+)		Mavi et al. (2018)
Pot/Bean	Biosolid(4, 8, 16, 32 t/ha)	190		P, Ca, Zn (+)			96-112(+)		Melo et al. (2018)
Growth chamber/Crabgrass	Mixture of softwoods and loblolly pine + switchgrass (2%)	450					72.72(+)		Mitchell et al. (2018)
Field/Maize-Mustard	*EupIatoriumadenophIorum*(5,10,15,25,40 t/ha)	400-500				50-134(+)			Pandit et al. (2018)
Greenhouse/Rice	Rice straw and sugarcane bagasse(0.3, 0.9%)	350				260-321(+)			Sadegh-Zadeh et al. (2018)
Pot/Sunflower, Maize	Miscanthus (25, 50, 75%)	350				33-50(+)	42-70(+)	Physiology, biochemistry and antioxidant defense(+)	Shahbaz et al. (2018)
Field/Rice	Rice straw(2.25 t/ha)	550-650				33(+)	20-29.4(+)	Grains/panicle (72.7)(+)	Si et al. (2018)
Greenhouse/Maize	Cotton husks, eucalyptus residue, sugarcane filtercake, swine manure (1, 2, 3, 4%)						20(+)		Speratti et al. (2018)
Field/Wheat	Wood of *Dalbergia sissoo* (1, 2, 3, 4, 5, 6, 7, 8, 9 and 10%)	500-700	N (25-48)(+)	N(50)(+)	N(65)(+)	38(+)	19(+)		Abbas et al. (2017)
Pot/Zuccini	Maize stalk(6.3, 12.6, 25.5 g/pot)	400			N(90.03)(+)	26.7-195(+)			Amin and Eissa (2017)
Field/Maize-wheat	Acacia prunings(10 t/ha)	1000			P(69.23-150)(+)	18-24(+)		Plant height, grains/panicle, 1000-grain weight and harvest index (+)	Arif et al. (2017)
Hydroponics/leafy vegetables	Rice husk(1:1 ratio)	500	Ca, Mg, Mn, Zn(120-350)(+)	N(12)(+)			100-140(+) 55.8-87.1(-)	Shoot length (49)(-) Shoot and leaves number(200)(+)	Awad et al. (2017)
Greenhouse/Corn	Softwood chips(20 t/ha)	500	N(15.5)(+)				17(+)	Total root length (18)(+) Specific root length(5)(+)	Backer et al. (2017)

								Tissue density(7)(-)	
Pot/Chinese cabbage	Kunai grass(10 t/ha)	500		N(14.89)(+)		48.92(+)	35.67(+)		Baiga and Rao (2017)
Field/Maize	Chicken litter(10, 20 t/ha)	550	N(31.90), P(256.25), K(112.27), Ca(20.82), Mg(11.76)(+) Fe(72.5)(-)	N(706.62), P(2096.34), K(1189.68), Ca(674.15), Mg(550.63), Fe(212.93)(+)	P(190.96)(+)	339.23(+)	512.70(+)		Ch'ng et al. (2017)
Field/Rice	Rice straw(2, 40 t/ha)	400-500				10(+)		Grains/panicle (5.20)(+) Seed setting rate (3.05)(+) 1000-grain weight (1.05)(+) No. of effective tillers/hill(1.95)(+)	Cui et al. (2017)
Field/Corn	Sewage sludge(15 t/ha)	300 and 500		N(49.27), P(98.73), K(31.83), Ca(58.92), Mg(96.90), S(53.93), Cu(85.71), Zn(127.27), Fe(14.89), Mn(50)(+)		33.33-46.67(+)			Faria et al. (2017)
Field/major crops & cover crops	Norway spruce (70%) +European Beech (30%) (15, 30 t/ha)	550-600	K(16)(+) Mn(25-42)(-)		N(6.09-8.01)(-)	9.18-11.00(-)	11.68-25.68(-)	Plant height (7-14)(-)	Haider et al. (2017)
Greenhouse/Maize	Sewage sludge(7.3, 14.5, 29 t/ha)	350	N(6.14), P(15.39)(-) K(1.46)(+) Fe(10.07), Zn(17.52), Cu(12.22), Mn(1.54)(-)	N(9.66), P(23.26), K(2.84)(-)			11.67(-)		Khanmohammadi et al. (2017)
Pot/ Sugar maple and Red pine	Maple sawdust and wood ash(5, 20, 50 t/ha)	450	N(1.5), P(28.03), K(46.96), Ca(1.83), Mg(7.22), S(28.57)(+)				20(+)		Noyce et al. (2017)
Glasshouse/Corn	Empty fruit bunch (0, 5, 10, 15 and 20 t/ha)	350-450	N(148), P(236), K(185) Ca(181), Mg(154)(+)	N(564), P(666), K(678) Ca(600), Mg(500)			67-150(+)		Abdulrahman et al. (2016)
Field/Maize	Maize cobs (4 t/ha)	350				154-425(+)	152-246(+)		Abiven et al. (2015)
Field/Maize	Willow wood waste (2.5, 10 t/ha)	550	N(5-14), P(11-41)(+) $\delta^{15}N$ (1.3-2.2 times)	$\delta^{13}C$ (10.9-11 times)		10-29(+)	12-18(+)		Agegnehu ,Bass , et al. (2016)
Field/Barley	Stem, bark and branches of Acacia wood (2, 10 t/ha)	350-450	N(6.5-11)(+) $\delta^{15}N$ (1.2 times)	N(37-64)(+)	N(45)(+)	30-79(+)	56-176(+)		Agegnehu ,Nelson , et al. (2016b)

Field/Barley	Stem, bark and branches of Acacia wood (2, 10 t/ha)		N(39), P(11), K(11)(+)			48(+)	52(+)		Agegnehu ,Nelson , et al. (2016a)