Biochar physicochemical properties: pyrolysis temperature and feedstock kind effects

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Abstract Biochar is a pyrogenous, organic material synthesized through pyrolysis of different biomass (plant or animal waste). The potential biochar applications include: (1) pollution remediation due to high CEC and specific surface area; (2) soil fertility improvement on the way of liming effect, enrichment in volatile matter and increase of pore volume, (3) carbon sequestration due to carbon and ash content, etc. Biochar properties are affected by several technological parameters, mainly pyrolysis temperature and feedstock kind, which differentiation can lead to products with a wide range of values of pH, specific surface area, pore volume, CEC, volatile matter, ash and carbon content. High pyrolysis temperature promotes the production of biochar with a strongly developed specific surface area, high porosity, pH as well as content of ash and carbon, but with low values of CEC and content of volatile matter. This is most likely due to significant degree of organic matter decomposition. Biochars produced from animal litter and solid waste feedstocks exhibit lower surface areas, carbon content, volatile matter and high CEC compared to biochars produced from crop residue and wood biomass, even at higher pyrolysis temperatures. The reason for this difference is considerable variation in lignin and cellulose content as well as in moisture content of biomass. The physicochemical properties of biochar determine application of this biomaterial as an additive to improve soil quality. This review succinctly presents the impact of pyrolysis temperature and the type of biomass on the physicochemical characteristics of biochar and its impact on soil fertility.

Keywords Biochar · Feedstock kind · Soil quality · Physicochemical properties · Pyrolysis temperature

1 Introduction

Biochar is a carbon-rich material. It can be used not only as a renewable fuel, but also as an additive for improvement of soil quality (Lehmann and Joseph 2009). The nature of carbon structures is the key reason for the high stability (Lehmann et al. 2011; Nguyen et al. 2010). The most pronounced chemical difference between biochar and other organic matter is much higher proportion of aromatic C and condensed aromatic structures, in contrast to other aromatic structures of soil organic matter, such as lignin (Schmidt and Noack 2000). The condensed aromatic structure of biochars can have varying forms, including amorphous C (which dominates at lower pyrolysis temperatures), turbostratic C (formed at higher temperatures) and graphite C (Keiluweit et al. 2010;
Biochar exhibits high biodegradability, high contents of total and organic carbon, as well as optimal concentrations of micro- and macroelements (potassium, sodium, magnesium, calcium, copper, zinc, iron etc.) (Malińska 2012). It is generally characterized by a high specific surface area, high content of surface functional groups, pH and porosity (Hernandez-Mena et al. 2014; Lehmann et al. 2011). Hernandez-Mena et al. (2014) showed that biochar exhibits high porosity, with longitudinal pores of sizes ranging from micro- to macropores. Large pores, originating from the vascular bundles of the raw biomass, are important for improving soil quality because they can provide habitats for symbiotic microorganisms (Thies and Rillig 2009). Porous structures can also act as release routes for pyrolytic vapours (Lee et al. 2013a).

Biochar is created through the pyrolysis of biomass. Pyrolysis represents the oldest known method of biomass thermal processing. The use of pyrolysis dates back at least to ancient Egypt (Mohan et al. 2006). Biowastes from agriculture, the food industry, and forestry are the main sources of feedstock (Shakya and Agarwal 2017). The most popular substrates include wood chips and pellets, tree cuttings, bagasse, distiller grains, press cakes from the oil and juice industry, rice husks and crop residues (Parmar et al. 2014). However, production can also be based on biomass sources other than lignocelluloses matter, such as sewage sludge, poultry litter, excrement, bones, dairy manure etc. (Kumar et al. 2016). The selection of suitable conditions for producing a char with the desired properties therefore requires knowledge of dependencies and influencing factors, both quantitatively and qualitatively (Weber and Quicker 2018; Zhang et al. 2019). The products of pyrolysis are an oil (a mixture of hydrocarbons), synthetic gas (mixed hydrocarbon gases) and biochar (Lewandowski et al. 2010; Verheijen et al. 2010). The proportions of these individual products depend on the temperature range, pressure, residence time, etc. (Brewer 2012; Cheah et al. 2016; Lewandowski et al. 2010). It is summarized in Table 1. The slow pyrolysis is the most effective for biochar production with a typical biochar yield of 35.0% from dry biomass weight. The fast pyrolysis is the most efficient method for producing biofuels and the gasification is the most efficient for producing syngas and hence is usually used to generate heat and energy (Cheah et al. 2016). As a result of high heating rates and short residence times, fast pyrolysis tends to yield higher proportions of oils. In contrast, slow pyrolysis tends to yield higher proportions of biochars because of slow heating rates and longer residence times (Daful and Chandraratne 2018). Furthermore, the pyrolysis of biomass may contribute to the formation of phytotoxic and potentially carcinogenic compounds under some conditions (type of feedstock, pollution in feedstock, and pyrolysis parameters) (Ndriangu et al. 2019). Some of the heavy metals are transformed into less toxic forms, pathogens are eliminated as the result of the pyrolysis process (Paz-Ferreiro et al. 2018), but nitrogen and sulphur are lost during pyrolysis (Maguire and Agblevor 2010). Therefore, biochar has been researched as a soil modifier.

Biochar has a long history as a soil modifier (Brewer 2012). Early Japanese farmers used unique manure called “haigoe”, which was prepared by adding human waste to rice husk biochar and applying it to fields some time before planting crops (Shakya and Agarwal 2017). Moreover, the addition of biochar may affect the biological community composition of soil, as demonstrated for the ‘Terra Preta’ soils in the Amazon (Lehmann et al. 2011). Soil is a complex material and comprises minerals, soil organic matter, water, and air. These fractions greatly influence soil texture, structure, and porosity (Baghdadi and Zribi 2016). These properties subsequently affect air and water movement in the soil layers and thus the soil’s ability to function (Naga Raju et al. 2017). Therefore, soil physicochemical properties have a great influence on the soil quality. Biochar generally increases carbon sequestration in soil (Sohi et al. 2010), reduces the emission of ammonia and carbon dioxide (Cabeza et al. 2018), lowers soil compactness, optimizes compost (Liang et al. 2010), improves water retention and the sorption of heavy metals, increases the availability of micronutrients for plants and increases the pH of soils (Van Zwieten et al. 2010). Biochar also stimulates the growth of rhizosphere microorganisms and mycorrhizal fungi (Głuszek et al. 2017). These bacteria and fungi may also promote plant growth (Compant et al. 2010).

The authors provided a more comprehensive and insightful scientific overview. The objective of this review presented herein is to assess: (1) the physicochemical characteristics of biochar formed at different temperatures and from different biomass, (2) the
mechanisms of change in biochar structure, surface and main physicochemical properties, (3) discussion of biochars effect on the physicochemical properties of the soil and biological properties of the soil and (4) the application and potential effects of biochar as a soil adsorbent. The explanation of the main mechanisms of changes in physicochemical properties of biochar during pyrolysis at various temperatures and the types of feedstock is necessary to determine the ability of biochar to remediate various soils, which is useful in future studies.

### 2 Effects of pyrolysis temperature

Process of biochar production had three stages: pre-pyrolysis; main-pyrolysis and formation of carbonaceous soil products (Lee et al. 2017). The first stage (from ambient temperature to 200 °C) is attributed to evaporation of moisture and light volatiles. The moisture evaporation causes breakage of bonds and formation hydroperoxide, –COOH and –CO groups (Ca´rdenas-Aguiar et al. 2017). The second stage (from 200 to 500 °C) was a devolatilized and decomposed of hemicelluloses and cellulose at a fast rate (Ding et al. 2014). The last stage (above 500 °C) is degradation of lignin and other organic matter with stronger chemical bonds (Ca´rdenas-Aguiar et al. 2017). The pyrolysis temperature is strongly correlated with changes in the structure and physicochemical properties of biochar (Asadullah et al. 2007; Chen et al. 2008; Jindo et al. 2014; Mukherjee et al. 2011). Data on these relationships are presented in Table 2. Pyrolysis temperature had a strong influence on biochars physicochemical properties (e.g. surface area, pH and functional groups) and it affected on the functions of biochar as a soil amendment (Ding et al. 2014). Higher pyrolysis temperature resulted in an increase of surface area, carbonized fractions, pH and volatile matter and a decrease of CEC and content of surface functional groups.

#### 2.1 Specific surface area

It has been found that increasing pyrolysis temperature causes changes in biochar surface area and porosity (Bonelli et al. 2007). This is most likely due to the decomposition of organic matter and the formation of micropores (Katyal et al. 2003). Moreover, the destruction of aliphatic alkyls and ester groups as well as the exposure of the aromatic lignin core under higher pyrolysis temperatures may result in increased surface area (Chen and Chen 2009). Ghani et al. (2013) have shown that at lower temperatures (less than 500 °C), lignin is not converted into a hydrophobic polycyclic aromatic hydrocarbon (PAH) and biochar becomes more hydrophilic. At temperatures higher than 650 °C, biochar is thermally stable and becomes more hydrophobic (Ghani et al. 2013).

However, the hydrophobicity/hydrophilicity of the surface is not an unambiguous factor conditioning the sorption process of aromatic compounds. The surface area of the biochar increases with increasing temperature. This is because with increasing pyrolysis temperature, pore-blocking substances are driven off or are thermally cracked, increasing the externally accessible surface area (Rafiq et al. 2016). Pyrolysis may increase the surface area and pore volumes through progressive degradation of the organic materials (cellulose, lignin) and the formation of vascular

### Table 1 Products of pyrolysis processes in a variety of conditions

| Process                        | Pyrolysis temperature (°C) | Pressure            | Residence time   | The proportion of products in the pyrolysis process (%) |
|--------------------------------|---------------------------|---------------------|------------------|-------------------------------------------------------|
| Fast Pyrolysis                 | 400–600                   | Vacuum-atmospheric  | Seconds          | Bio-oil: 75.0, Synthetic gas: 13.0, Biochar: 12.0     |
| Biocarbonization (slow pyrolysis) | 350–800                  | Atmospheric         | Seconds–hours    | Bio-oil: 30.0, Synthetic gas: 35.0, Biochar: 35.0     |
| Gasification                   | 700–1500                  | Atmospheric–elevated| Seconds–minutes  | Bio-oil: 5.0, Synthetic gas: 85.0, Biochar: 10.0      |

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| BF                        | PT (°C) | PY (%) | pH  | SSA (m²/g) | VM (%) | A (%) | CEC (cmol/kg) | C (%) | References                  |
|--------------------------|---------|--------|-----|------------|--------|-------|---------------|------|-----------------------------|
| Peanut shell             | 300     | 36.9   | 7.8 | 3.1        | 60.5   | 1.2   | –             | 68.3 | Ahmad et al. (2012)         |
| Peanut shell             | 700     | 21.9   | 10.6| 448.2      | 32.7   | 8.9   | –             | 83.8 |                             |
| Peanut straw             | 700     | –      | 11.2| –          | 38.5   | 254.0 | –             |       |                             |
| Dairy Manure             | 350     | –      | 9.2 | 1.6        | 53.5   | 24.2  | –             | 55.8 | Cantrell et al. (2012)      |
| Dairy Manure             | 700     | –      | 9.9 | 186.5      | 27.7   | 39.5  | –             | 56.7 |                             |
| Feedlot manure           | 350     | –      | 9.1 | 1.34       | 47.9   | 28.7  | –             | 53.3 |                             |
| Feedlot manure           | 700     | –      | 10.3| 145.2      | 19.8   | 44.0  | –             | 52.4 |                             |
| Poultry litter           | 350     | –      | 8.7 | 3.9        | 42.3   | 30.7  | –             | 51.2 |                             |
| Poultry litter           | 700     | –      | 10.3| 50.9       | 18.3   | 46.2  | –             | 45.9 |                             |
| Separated swine solids   | 350     | –      | 8.4 | 0.9        | 49.8   | 32.5  | –             | 51.5 |                             |
| Separated swine solids   | 700     | –      | 9.5 | 4.1        | 13.4   | 52.9  | –             | 44.0 |                             |
| Turkey litter            | 350     | –      | 8.0 | 2.6        | 42.1   | 34.8  | –             | 49.3 |                             |
| Turkey litter            | 700     | –      | 9.9 | 66.7       | 20.8   | 49.9  | –             | 44.8 |                             |
| Dairy Manure             | 100     | 97.0   | 8.0 | 1.8        | –      | 37.0  | –             | 36.8 | Cao et al. (2009)           |
| Dairy Manure             | 200     | 58.0   | 6.8 | 2.7        | –      | 44.0  | –             | 31.1 |                             |
| Dairy Manure             | 350     | 27.0   | 10.5| 7.1        | –      | 62.0  | –             | 25.2 |                             |
| Dairy Manure             | 500     | 25.0   | 10.5| 13.0       | –      | 95.0  | –             | 1.7  |                             |
| Prunings of fruit trees  | 500     | –      | 10.8| –          | 58.8   | 4.7   | –             | –    | Castellini et al. (2015)    |
| Cattle manure            | 300     | –      | 8.0 | –          | 47.3   | 20.2  | 66.3          | –    | Cely et al. (2015)          |
| Cattle manure            | 500     | –      | 10.2| –          | 13.2   | 43.7  | 70.9          | –    |                             |
| Cattle–straw manure      | 300     | –      | 10.1| –          | 24.9   | 38.3  | 65.5          | –    |                             |
| Cattle–straw manure      | 500     | –      | 10.1| –          | 11.9   | 51.3  | 58.4          | –    |                             |
| Chicken manure           | 300     | –      | 8.1 | –          | 23.9   | 34.8  | 137.6         | –    |                             |
| Chicken manure           | 500     | –      | 10.6| –          | 11.9   | 38.0  | 81.4          | –    |                             |
| Chicken–sawdust manure   | 300     | –      | 10.6| –          | 13.0   | 34.6  | 81.7          | –    |                             |
| Chicken–sawdust manure   | 500     | –      | 10.3| –          | 8.3    | 35.9  | 65.8          | –    |                             |
| Pig manure               | 300     | –      | 7.8 | –          | 31.3   | 50.3  | 35.6          | –    |                             |
| Pig manure               | 500     | –      | 8.2 | –          | 6.50   | 73.9  | 32.7          | –    |                             |
| Corn straw               | 600     | –      | 9.5 | 13.1       | –      | 60.2  | –             | 35.9 | Chen et al. (2011)          |
| Hardwood                 | 450     | –      | 5.6 | 0.4        | –      | 38.6  | –             | 53.4 | Chen et al. (2011)          |
| Municipal sewage sludge  | 900     | 53.3   | 12.2| 67.6       | 87.5   | 88.1  | 247.5         | 15.9 | Chen et al. (2015)          |
| Sugarcane bagasse        | 400     | 31.6   | 7.0 | 0.8        | –      | 3.8   | –             | –    | Ding et al. (2014)          |
| Sugarcane bagasse        | 600     | 22.9   | 7.7 | 14.1       | –      | –     | 4.2           | –    |                             |
| Chicken Manure           | 350     | 69.7   | 9.7 | –          | 36.9   | 52.0  | –             | 31.2 | Domingues et al. (2017)     |
| Chicken Manure           | 450     | 63.0   | 10.2| –          | 30.6   | 55.3  | –             | 27.2 |                             |
| Chicken Manure           | 750     | 55.9   | 11.7| –          | 26.5   | 56.4  | –             | 24.7 |                             |
| Eucalyptus sawdust       | 350     | 42.5   | 5.9 | –          | 36.9   | 0.9   | –             | 70.4 |                             |
| Eucalyptus sawdust       | 450     | 36.0   | 8.0 | –          | 28.5   | 0.7   | –             | 78.6 |                             |
| Eucalyptus sawdust       | 750     | 28.2   | 9.7 | –          | 6.5    | 1.1   | –             | 90.9 |                             |
| Coffee husk              | 350     | 43.5   | 9.7 | –          | 34.6   | 12.9  | –             | 60.5 |                             |
| Coffee husk              | 450     | 37.7   | 9.8 | –          | 26.2   | 12.9  | –             | 61.3 |                             |
| Coffee husk              | 750     | 31.6   | 9.9 | –          | 17.6   | 19.6  | –             | 66.0 |                             |
| BF                          | BF (°C) | PT (%) | PY (%) | pH | SSA (m²/g) | VM (%) | A (%) | CEC (cmol/kg) | C (%) | References                  |
|-----------------------------|---------|--------|--------|-----|------------|--------|-------|---------------|-------|----------------------------|
| Sugarcane bagasse          | 350     | 37.5   | 7.2    | –   | 35         | 1.9    | –     | –             | 74.7  | Elaigwu et al. (2014)       |
| Sugarcane bagasse          | 450     | 33.2   | 8.8    | –   | 24         | 2.1    | –     | –             | 81.6  |                            |
| Sugarcane bagasse          | 750     | 26.9   | 9.7    | –   | 7.7        | 2.2    | –     | –             | 90.5  |                            |
| Pine bark                  | 350     | 59.6   | 7.8    | –   | 38.5       | 8.3    | –     | –             | 67.6  |                            |
| Pine bark                  | 450     | 49.3   | 8.3    | –   | 29.3       | 7.9    | –     | –             | 75.2  |                            |
| Pine bark                  | 750     | 38.9   | 9.9    | –   | 6          | 14.5   | –     | –             | 86.3  |                            |
| **Prosopis Africana shell**| 350     | 60.1   | 6.5    | 3.1 | –          | 25.2   | –     | –             | 70.8  | Jin et al. (2016)           |
| Swine manure               | 400     | 49.3   | 11.0   | 4.9 | 35.5       | 49.8   | 65.6  | –             | 74.9  |                            |
| Rapeseed plant             | 400     | 39.4   | –      | 16.0| 27.1       | 12.2   | –     | –             | 71.3  | Karaosmanoglu et al. (2000)|
| Rapeseed plant             | 700     | 29.6   | 19.3   | 9.0 | 14.4       | –      | –     | –             | 79.5  |                            |
| Fescue straw               | 100     | 99.9   | 1.8    | 69.6| 6.9        | –      | –     | –             | 48.6  | Keiluweit et al. (2010)    |
| Fescue straw               | 700     | 28.8   | –      | 139.0| 9.1       | 19.3   | –     | –             | 94.2  |                            |
| Cow manure                 | 400     | –      | –      | 2.5 | 27.4       | 15.3   | –     | –             | 60.2  | Kolodynska et al. (2012)   |
| Cow manure                 | 600     | –      | –      | 8.0 | 13.0       | 18.8   | –     | –             | 58.8  |                            |
| Pig manure                 | 400     | –      | –      | 15.6| 19.1       | 46.5   | –     | –             | 44.1  |                            |
| Pig manure                 | 600     | –      | –      | 15.9| 15.1       | 50.3   | –     | –             | 42.3  |                            |
| Tire rubber                | 200     | 93.5   | –      | –   | –          | 15.0   | –     | –             | 74.7  | Lian et al. (2011)          |
| Tire rubber                | 800     | 43.0   | –      | 50.0| –          | 10.5   | –     | –             | 86.0  |                            |
| Oak wood                   | 450     | –      | –      | 1.9 | 15.6       | 64.5   | –     | –             | 71.3  | Mohan et al. (2011)         |
| Corn cobs                  | 500     | 18.9   | 7.8    | 0   | –          | 13.3   | –     | –             | 77.6  | Mullen et al. (2010)        |
| Corn stover                | 500     | 17.0   | 7.2    | 3.1 | –          | 32.8   | –     | –             | 57.3  |                            |
| Poultry litter             | 500     | –      | –      | 1.0 | 17.7       | 41.9   | –     | –             | 48.3  | Novak et al. (2009)         |
| Pine chip                  | 500     | –      | –      | 6.2 | 22.4       | 2.6    | –     | –             | 88.9  | Novak et al. (2016)         |
| Soybean stover             | 700     | 29.6   | 11.3   | 420.3| 14.7      | 17.2   | 59.2  | –             | 82.0  | Karunanithi et al. (2017)   |
| Soybean straw              | 700     | –      | 11.1   | –   | –          | 23.7   | 222.0 | –             | –     |                            |
| Corn stover                | 300     | 66.2   | 7.7    | 3.2 | 54         | 5.7    | –     | –             | 45.5  | Rafiq et al. (2016)         |
| Corn stover                | 400     | 37.1   | 8.8    | 3.2 | 45.5       | 12.5   | –     | –             | 64.0  |                            |
| Corn stover                | 500     | 29.2   | 9.8    | 4.6 | 33.8       | 18.7   | –     | –             | 64.5  |                            |
| Orange pomace              | 350     | 71.9   | 9.9    | 1.2 | 32.3       | 11.3   | 35.2  | –             | 56.8  | Tag et al. (2016)           |
| Orange pomace              | 600     | 44.6   | 10.5   | –   | 17.3       | 16.3   | 25.6  | –             | 68.1  |                            |
| Vine pruning               | 350     | 64.6   | 10.3   | 8.1 | 30.2       | 8.3    | 47.4  | –             | 64.7  |                            |
| Cottonseed hull            | 350     | 36.8   | 7.0    | 4.7 | 34.9       | 5.7    | –     | –             | 77.0  | Uchimiya et al. (2011b)     |
| Cottonseed hull            | 800     | 24.2   | 9.2    | 322.0| 11.4      | 9.2    | –     | –             | 90.0  |                            |
| Orange peel                | 700     | 22.2   | –      | 201.0| –         | 2.8    | –     | –             | 71.6  |                            |
| Black wattle               | 475     | 9.7    | 241.0  | –   | 4.8        | 101.0  | –     | –             | 66.5  | Uras et al. (2012)          |
| Sugarcane bagasse          | 475     | 8.6    | 259.0  | –   | 12.1       | 122.0  | –     | –             | 57.3  |                            |
| Vineyard prunings          | 475     | –      | 10.4   | 92.0| –          | 8.1    | 65.0  | –             | 66.5  |                            |
| Tree barks                 | 400     | –      | 8.9    | –   | –          | 23.0   | –     | –             | 80.0  | Venegas et al. (2015)       |
| Rice straw                 | 300     | 50.1   | 9.3    | –   | 48.4       | 25.4   | 60.6  | –             | 72.5  | Wu et al. (2012)            |
| Rice straw                 | 700     | 33.5   | 10.8   | –   | 14.9       | 28.2   | 23.1  | –             | 90.6  |                            |
| Bamboo                     | 450     | 26.3   | 5.2    | 18.2| –          | –      | –     | –             | 76.9  | Yao et al. (2012)           |
| Bamboo                     | 600     | 24.0   | 7.9    | 470.4| –         | –      | –     | –             | 80.9  |                            |
| Canola straw               | 300     | 6.5    | –      | –   | 10.7       | 199.0  | –     | –             | 81.1  | Yuan et al. (2011)          |
bundles or channel structure (Li et al. 2013; Zhao et al. 2017). Some amorphous carbon structures also form during pyrolysis due to the degradation of cellulose (Zhao et al. 2017). It has been reported that micropores may be formed by amorphous carbon structures (Vamvuka and Sfakiotakis 2011). A higher pyrolysis temperature causes the release of volatile matter and creates more pores (Shaaban et al. 2014). Low specific surface areas with low ash content were observed in biochars produced from cotton seed hull (4.7 m²/g) (Uchimiya et al. 2011b), poultry litter (17.7 m²/g) (Novak et al. 2009) and dairy manures (13.0 m²/g) (Cao and Harris 2010). The type and concentration of surface functional groups have been reported to play an important role in adsorption capacity and the removal mechanism of the adsorbates (Yenisoy-Karakas¸ et al. 2004). Moreover, an increase in the structure aromaticity with an increase in pyrolysis temperature can also enhance resistance to microbial decomposition (Xie et al. 2015). Uchimiya et al. (2010) discovered that biochar produced at a temperature higher than 400 °C was more effective for organic and inorganic contaminant sorption due to its high surface area and considerable micropore development. However, Chen et al. (2008) proved that the partitioning of organic and inorganic contaminants into non-carbonized biochar fractions derived from pine needles was the major sorption mechanism at low pyrolysis temperatures (100–300 °C), whereas adsorption onto porous carbonized fractions was dominant at high temperatures (400–700 °C).

### 2.2 Surface Functional Groups and CEC

The heating to temperatures of 350–650 °C breaks and rearranges the chemical bonds in the biomass, forming new functional groups (e.g. carboxyl, lactone, lactol, quinine, chromene, anhydride, phenol, ether, pyrone, pyridine, pyridone, and pyrrole) (Mia et al. 2017). Figures 1 and 2 shows example structures on the outer surface of the graphene sheets (Harris 1997; Harris and Tsang 1997) and pores (Van Zwieten et al. 2010; Zheng et al. 2010). The FTIR spectra indicate that biochar is dominated by functional groups typical of oxygenated hydrocarbons, reflecting the carbohydrate structure of cellulose and hemicelluloses (Ghani et al. 2013). The pyrolysis process can cause the disappearance of absorption bands characteristic of raw material and the appearance of new bands typical of biochar samples. Ghani et al. (2013) demonstrated that the biochar from sawdust exhibited a broad band between 3000 and 3600 cm⁻¹ (peak maximum at 3339 cm⁻¹).

### Table 2

| BF             | PT (°C) | PY (%) | pH  | SSA (m²/g) | VM (%) | A (%) | CEC (cmol/kg) | C (%) | References     |
|----------------|---------|--------|-----|------------|--------|-------|---------------|-------|----------------|
| Canola straw   | 700     | –      | 10.8| –          | 28.6   | 179.0 | –             | –     | Zama et al. (2017) |
| Buckwheat husk | 350     | 46.3   | 9.2 | 11.4       | 4.0    | 11.2  | 70.1          |       |                 |
| Buckwheat husk | 450     | 42.3   | 9.7 | 10.7       | 25.4   | 11.5  | 76.5          |       |                 |
| Buckwheat husk | 550     | 34.2   | 10.0| 17.0       | 5.8    | 10.1  | 82.8          |       |                 |
| Buckwheat husk | 650     | 28.5   | 9.1 | 17.8       | 33.1   | 11.7  | 83.9          |       |                 |
| Mulberry wood  | 350     | 37.5   | 10.2| 16.6       | 7.5    | 23.3  | 67.9          |       |                 |
| Mulberry wood  | 450     | 32.7   | 11.1| 31.5       | 7.7    | 22.1  | 70.8          |       |                 |
| Mulberry wood  | 550     | 26.2   | 10.6| 58.0       | 9.8    | 19.0  | 77.0          |       |                 |
| Mulberry wood  | 650     | 22.8   | 10.6| 24.5       | 9.8    | 21.8  | 80.1          |       |                 |
| Peanut shells  | 350     | 45.7   | 10.4| 14.0       | 7.06   | 26.5  | 64.3          |       |                 |
| Peanut shells  | 450     | 38.1   | 11.1| 14.0       | 16.9   | 23.7  | 70.8          |       |                 |
| Peanut shells  | 550     | 32.5   | 10.6| 18.6       | 7.1    | 19.7  | 73.7          |       |                 |
| Peanut shells  | 650     | 29.4   | 10.6| 28.1       | 24.4   | 17.4  | 74.6          |       |                 |

BF Biochar feedstock, PT pyrolysis temperature, SSA specific surface area, VM Volatile matter, A Ash, PY product yield, C total carbon content, CEC cation exchange capacity.
with a smaller band from 2700 to 3000 cm\(^{-1}\) (maximum at 2907 cm\(^{-1}\)). The band centred at 3339 cm\(^{-1}\) was attributed to the presence of OH functional groups (alcoholic and phenolic) (Pretsch et al. 2009), while the band at \(\sim 2907\) cm\(^{-1}\) was attributed to alkyl C–H stretching (Claoston et al., 2014). The band occurring at 1600 cm\(^{-1}\) was attributed to aromatic C–C and C–O stretching of conjugated ketones and quinones (Ruthiraan et al. 2015), and the band occurring at 1735 cm\(^{-1}\) was attributed to C=O stretching of ketones, aldehydes and esters (Uchimiya et al. 2011a). The band centred at 1238 cm\(^{-1}\) was attributed to the presence of C–O–C groups and aryl ethers, phenolic associated with lignin (Kardam et al. 2012). The intense band occurring at 1130 cm\(^{-1}\) was characteristic of C–O–C stretching of ester groups in cellulose and hemicelluloses (Jeba Jeevitha et al. 2015). All of these bands are typical for the FTIR of biochars (Claoston et al. 2014; Ghani et al. 2013; Liu et al. 2015; Kardam et al. 2012; Zhao et al. 2017).

The effect of temperature on biochar structure and functional groups is shown in Fig. 3. Biochar produced at high temperatures (600–700 °C) exhibits a highly hydrophobic nature with well-organized C layers (Uchimiya et al. 2011a). However, it is characterized by lower contents of H- and O-containing functional groups due to dehydration and deoxygenation of the biomass (Ahmad et al. 2014a; Uchimiya et al. 2011a). Surface groups can act as electron donors or electron acceptors, which leads to the formation of coexisting areas whose properties can range from acidic to basic and from hydrophilic to hydrophobic (Amonette and Joseph 2009). Consequently, such a product exhibits potentially lower ion exchange capacity (Novak et al. 2009). On the other hand, biochar produced at lower temperatures (300–400 °C) displays more diversified organic character due to the occurrence of aliphatic and cellulose type structures (Glaser et al. 2002; Novak et al. 2009). As a result, the structure of biochar appears to have more organized C layers (like graphene structure) and less content of surface functional groups when pyrolysis temperature increases (Ahmad et al. 2014b).

The decrease in CEC is due to the removal of surface functional groups and the formation of aromatic carbon (Joseph et al. 2010). Numerous studies have reported that the cation exchange capacity (CEC) of biochar decreases with increasing pyrolysis temperature (Mukherjee et al. 2011; Song and Guo 2012; Yao et al. 2012). The detectable CEC suggested that when biochar was produced at temperatures up to 480 °C, some acidic oxygenated functional groups such as phenolic acid and carboxyl groups were retained (Mitchell et al. 2013). Banik et al. (2018) reported that the CEC of biochar is dependent on the nature and distribution of O-containing functional groups on the biochar surface. The negative charge sites on biochar surfaces are attributed

![Fig. 1](image1.png) **Biochar surface functional acidic groups**

![Fig. 2](image2.png) **Biochar surface functional basic groups**
to carboxylate and phenolate functional groups (Mia et al. 2017). They assume that negative surface charge can only come from carboxylate and phenolate groups and positive charge from oxonium groups (heteroatoms in aromatic rings) (Banik et al. 2018). However, other studies have found that biochars with higher specific surface area (obtained at temperatures above 600 °C) have greater surface microporosity and increased CEC (Gomez-Eyles et al. 2013; Kasozi et al. 2010). This was caused by the loss of volatile matter (Cely et al. 2015; Song and Guo 2012).

2.3 Volatile matter

Pyrolysis temperature has an influence on the structure of biochar due to the release of volatiles and the formation and volatilization of intermediate melts (Shaaban et al. 2014). Increasing the temperature leads to a decreased content of volatile matter (VM) (Crombie et al. 2013; Tag et al. 2016). Zhao et al. (2017) reported that an increase in the pyrolysis temperature decreased the content of VM (by 60.8–14.9%) for biochars obtained from apple tree branches. This was observed because the increasing temperature resulted in further cracking of the volatile fractions into low-molecular-weight liquids and gases instead of biochar (Ronsse et al. 2012). Furthermore, increasing temperature might result in the dehydration of hydroxyl groups and thermal degradation of cellulose and lignin (Zhang et al. 2015). Low-temperature biochars contain labile hydroxyl, carbonyl, carboxyl and hemiacetal compounds, while high-temperature biochars contain pyranones, ethers and quinines (Bourke et al. 2007). Antal and Grønli (2003) reported the stepwise chemical changes of cellulose subjected to increasing temperature. Biochars were dominated by oligosaccharides during the first stage of the pyrolysis process (temperatures up to 250 °C). Phenols and furans appeared in biochars as the temperature increased to 290 °C. At temperatures above 290 °C, the biochar consisted primarily of alkyl furans, benzenoid aromatics, and condensed aromatics. The volatile matter content of a biochar affects the stability of the material (Zimmerman 2010), its N availability (Deenik et al. 2010), plant growth (Deenik et al. 2010; Denyes et al. 2014) and its sorption capacity (Mukherjee et al. 2011). The volatile components fill micropores, dominating the surface of biochars, and are released from pores at higher production temperatures, making them accessible to ions (Mukherjee et al. 2011). The content of volatile matter can affect plant growth in two ways: toxic compounds such as phenol can inhibit root growth, while oligosaccharides, which are produced during the first two stages, can serve as a labile carbon for microbial decomposition (Fernandes and Brooks 2003).

2.4 Carbon content and ash content

The carbon and ash contents of biochar increase with increasing pyrolysis temperature (Chen et al. 2008;
High carbon content suggests that biochars probably still contain a certain amount of original organic plant residues such as cellulose (Chun et al. 2004). Rafiq et al. (2016) reported that increased pyrolysis temperature caused an increase of 5.7–18.7% in ash content. The increase in the ash content resulted from progressive concentration of inorganic constituents and OM combustion residues (Cao and Harris 2010; Chen et al. 2014; Zhao et al. 2017). Also, Zama et al. (2017) explained the increases in Mg, Ca, K, and P on biochars pyrolysed at high temperatures as being due to increased ash content (ranging from 4.0 to 33.1%). Mineral matter forming ash remains in biochar following carbonization (Domingues et al. 2017; Özcimen and Ersoy-Mericiboyu 2010). Increased carbon content (ranging from 62.2 to 92.4%) with an increase in pyrolysis temperature occurs due to a higher degree of polymerization (Domingues et al. 2017), leading to a more condensed carbon structure in the biochar (Lehmann and Joseph, 2009). For example, the carbon content of orange pomace biochar increased with increasing pyrolysis temperature (ranging from 56.8 to 68.1%) (Tag et al. 2016). Cantrell et al. (2012) observed that the carbon content of poultry litter biochar decreased with increasing pyrolysis temperature (ranging from 27.0 to 35.5%). The greater the degree of formation of aromatic structures, the higher the resistance of the biochar to microbial degradation (Keiluweit et al. 2010). Biochars with high ash contents also tend to have greater amounts of PAHs and trace metals (Yargicoglu et al. 2015). Preliminary studies have indicated that rigorous control of the feedstock materials and pyrolysis conditions contributed to substantial reductions of the emission levels of atmospheric pollutants (e.g. PAHs, dioxins) and particulate matter associated with biochar production (Verheijen et al. 2010).

2.5 pH

The pH values of biochars are positively correlated with the formation of carbonates and the contents of inorganic alkalis (Ding et al. 2014). These groups are the main cause of alkaline pH (Yuan et al. 2011). The contents of total base cations and carbonates have been reported to increase with increasing temperature, contributing to increased pH (ranging from 6.5 to 10.8) (Yuan et al. 2011). Higher pH with increasing temperature has been associated with the increases in ash content and oxygen functional groups that occur during pyrolysis (Ronsse et al. 2012; Spokas et al. 2012; Zhao et al. 2017). The disappearance of acidic functional groups (–COOH) and appearance of basic functional groups are additional contributors (Al-Wabel et al. 2013). However, increases in pH occur primarily due to the separation of alkali salts from organic materials due to increased pyrolysis temperature (Ding et al. 2014; Yuan et al. 2011). Above 300 °C, alkali salts begin to separate from the organic matrix, increasing the pH of the product. In contrast, cellulose and hemicelluloses decompose around 200–300 °C, yielding organic acids and phenolic substances that lower the pH of the products (Yu et al. 2014). The pH becomes constant at a temperature around 600 °C when all of the alkali salts are released from the pyrolytic structure (Shinogi and Kanri 2003).

3 Effects of feedstock material

Biomass is a complex biological, organic or non-organic solid material derived from living or recently living organisms (Mohan et al. 2006). Various types of wastes, such as animal manure, waste paper, sludge and many industrial wastes, are also treated as biomass because, like natural biomass, these waste materials are also a mixture of organic and non-organic compounds and can be processed to obtain energy (Tripathi et al. 2016). Biomass is categorized into woody and non-woody biomass. Woody biomass primarily comprises residues from forestry and trees (Jafri et al. 2018). The characteristics of woody biomass are low moisture, low ash, high calorific value, high bulk density and less voidage (Jafri et al. 2018). Non-woody biomass consists of agricultural crops and residues, animal waste, urban and industrial solid waste (Jafri et al. 2018). It is considered to have high moisture and high ash content, lower calorific value, low bulk density and higher voidage (Jafri et al. 2018).

The moisture content has influence on biochar formation (Kloss et al. 2012; Sun et al. 2014). Moisture contained in biomass not only increases the energy required to reach the pyrolysis temperature, it also inhibits char formation (Tripathi et al. 2016). Biomass is always associated with some amount of water/moisture content. This water inside a biomass
can exist as water vapour, chemically bound water (adsorbed within the pores of biomass) and free liquid water (Vassilev et al. 2013). Low moisture is advisable for the biochar production due to considerable reduction the heat energy and time required for the pyrolysis making the process economically viable as compared to pyrolysis involving biomass with high moisture content (Tripathi et al. 2016). Wide range of moisture content in biomass promotes production biochars with different physicochemical characteristics (Tripathi et al. 2016). For example, the moisture content of hardwood and softwood bark samples had a pronounced effect on the surface chemistry of the pyrolytic charcoals (Darmstadt et al. 2000). With decreasing maple bark moisture the charcoal surface becomes more polyaromatic and graphite-like, probably due to the longer effective pyrolysis time after the water has been evaporated (Darmstadt et al. 2000).

3.1 Volatile matter, carbon content and ash content

The lignin and cellulose content have influence on biochar formation (Kloss et al. 2012). Cellulose, present in the biomass, helps in the formation of tar (is a mixture of discrete ketones, aldehydes, organic liquids, and char), while high lignin content is favourable for char production during pyrolysis (Tripathi et al. 2016; Yu et al. 2014). El-Gamal et al. (2017) reported that the content of cellulose and hemicelluloses as well as moisture content of sugarcane bagasse were higher than rice husk, but rice husk biomass had higher content of lignin and ash. These can be attributed to the presence of different organic constituents in feedstock (El-Gamal et al. 2017). Lignin is amorphous and hydrophobic polymer with high molecular weight and numerous functional groups of aromatic substructure (Lee et al. 2013b). Cellulose and hemicelluloses are consisting of simple sugar monomer, which decompose at a temperature lower than 450 °C (Lee et al. 2013b). These two compounds have lower molecular weight than lignin and are easily released as pyrolytic vapors (Lee et al. 2013a), while lignin is very resistant to thermal degradation. Shariff et al. (2016) shows that coconut frond feedstock had higher cellulose (39.1%) and hemicelluloses (22.5%) content than coconut husk-biochar (33.6 and 22.0%). Coconut husk-feedstock had also higher lignin content (28.2%) than coconut frond feedstock (21.5%). Shariff et al. (2016) observed that the variations of lignocellulosic component in the feedstock influence biochar production. The high lignin composition in the feedstock will result in higher char formation (Demirbas 2004). This indicated that production of biochar will be increased with increased lignin content in feedstock (Shariff et al. 2016).

The higher lignin content in plant biomass has been reported to promote carbonization and to increase biochar carbon content and ash content (Sohi et al. 2010; Wang et al. 2015). Other studies of biomass structure have revealed that cellulose and hemicelluloses also have a significant influence on the carbon and ash contents (Rauber et al. 2018; Tripathi et al. 2016). Keiluweit et al. (2010) investigated different biochars derived from grass and wood biomass. Woody biomass often has higher cellulose, hemicellulose and lignin contents than biomass from herbaceous or grass species (Keiluweit et al. 2010; Lupoi and Smith 2012). However, apart from the lignin, cellulose and hemicellulose, a high yield of pyrolysis is also related to a high content of inorganic constituents of the feedstock materials, as indicated by their relatively high ash content and low content of volatile matter (Keiluweit et al. 2010). For example wood derived biochars have lower ash content (< 7.0%) in comparison to non-wood-derived biochars (> 50.0%) (Mukome et al. 2013). Singh and Cowie (2010) observed lower ash content in eucalyptus-derived biochar compared to poultry litter and cow manure. Manure and grass biochars typically have higher ash contents due to the presence of silica from soil contamination (Mukome et al. 2013). The low ash content makes biochar more amenable to transportation and incorporation into soils, as there is less windblown loss (Mukome et al. 2013). Additionally, Zhang et al. (2019) shows that the ash content was the lowest in woody biochars (1.5–3.0%) and the highest in peanut shell biochars (7.0–12.0%). Selection of wood-biochars would limit the increase in soil ash content, which has been associated with increased hydrophobicity (Kookana et al. 2011). An increase in hydrophobicity causes potential retention of hydrophobic agrochemicals, such as the herbicides (Sopena et al. 2012). Zielinska et al. (2015) shows that sewage sludge derived–biochars were characterized by high ash content (ranged from 64.1 to 79.1%). This results
from complexity and diversity of components contained in biomass (Zielinska et al. 2015).

Different biochars clearly demonstrate that plant-based biomass undergoes dehydration and depolymerization into smaller dissociation products of lignin and cellulose with an increase in pyrolysis temperature (Keiluweit et al. 2010). However, biochars derived from poultry manure and sewage sludge do not undergo depolymerisation due to the absence of lignocellulosic compounds (Liu et al. 2014; Lu et al. 2012). Biochars produced from animal litter and solid waste feedstocks exhibit lower surface areas compared to biochars produced from crop residue and wood biomass, even at higher pyrolysis temperatures (Lu et al. 2012). This may be due to the low C content, low volatile matter content and high molar H/C and O/C ratios in the latter biomass samples, leading to the formation of extensive cross-linkages (Bourke et al. 2007). Chen et al. (2008) showed that decreased H/C and O/C ratios were related to a higher aromaticity and lower polarity of biochars derived from pine needles.

Tag et al. (2016) observed the highest carbon content in the lignocellulosic biochar (vine pruning—72.3%; orange pomace—68.1% and poultry litter—56.6%) and the lowest in the algal biochar (45.1%) at 600 °C. Zielinska et al. (2015) shows that sewage sludge-biochar had lower carbon content (ranged from 18.1 to 27.8%). An increase of carbon content is associated with the loss of –OH surface functional groups as a result of dehydration (Zielinska et al. 2015) however different types of biomass behave differently, which is result of a different kind of graphitization of carbon into well-organized layers (Uchimiya et al. 2011b). Volatile matters of biochars decreased and fixed carbon of biochars increased differently depending on ash content of biomass and biomass type (Tag et al. 2016). This is due to the fact that the volatile matter fraction of combustible carbon inside the biomass decreased. This occurred because the organic compounds in animal waste are more labile and (compared to wood biochars) are more rapidly lost as pyrolysis temperature increases, before the formation of biochar recalcitrant compounds (Domingues et al. 2017).

3.2 Specific surface area

Specific surface area is one of the most important properties of biochar and is dependent on feedstock type (Chen et al. 2011; Wang et al. 2015). Tag et al. (2016) discovered that biochars obtained from orange pomace and vine pruning, with low ash content, had low surface area (1.2 m²/g and 8.1 m²/g). Probably this means that the pores within those biochars were dead-ended (Sharma et al. 2004). Type of feedstock causes the release of volatile matter and creates more pores (Shaaban et al. 2014). The increase in the porosity of biochar is due to the decomposition of lignin, the quick release of H₂ and CH₄ and the reaction of aromatic condensation as the temperature increases (Chen et al. 2012; Zhao et al. 2017). Kajina and Rousset (2018) reported that biochar produced from sugar cane leaves had a higher pore size (0.1 m²/g) and specific surface area (253.2 m²/g) than biochar produced from coconut shell (total pore size = 0.1 m²/g and specific surface area = 25.8 m²/g). El-Gamal et al. (2017) observed also that sugarcane-biochar had higher pore size (0.1 m²/g) and specific surface area (185.6 m²/g) than rice husks-biochar (total pore size = 0.1 m²/g and specific surface area = 154.7 m²/g). This indicates different thermal degradation and
content of lignin and cellulose (El-Gamal et al. 2017). This was also confirmed by Apaydın-Varol and Pütüne (2012). They reported that pine cone-biochars (1.8 m²/g) and peanut shell-biochars (2.0 m²/g) have higher surface area compared with soybean cake-biochar (0.5 m²/g) and corn stalk-biochar (0.8 m²/g). This can be also attributed to the presence of high amounts of lignin in the biomass samples (Raveendran and Ganesh 1998). Özçimen and Ersoy-Meriçboyu (2010) shows that porosity, total pore volume and specific surface area values of biochar derived from apricot stone (0.1%, 0.2 ml/g and 11.6 m²/g), derived from hazelnut shell (0.1%, 0.1 ml/g and 14.7 m²/g) and derived from grape seed (0.1%, 0.1 ml/g and 14.5 m²/g) were found higher than their biomass from apricot stone (0.1%, 0.1 ml/g and 10.5 m²/g), from hazelnut shell (0.1%, 0.1 ml/g and 5.8 m²/g), and from grape seed (0.1%, 0.1 ml/g and 10.6 m²/g). The difference can be attributed to the different degradation of cellulose or lignin.

Surface area increases during pyrolysis due to the decomposition of cellulose and hemicelluloses, and the formation of channel structures (Ahmad et al. 2012). The non-combustible component content depends on the feedstock material and has an influence on the specific surface area (Wang et al. 2015). The non-combustible component corresponds to the moisture and ash content (Pichtel 2014). The non-combustible component content of the herbaceous biochar was higher than that of the woody biochar, which can be attributed to the fact that grass biochar, has a lower specific surface area (Wang et al. 2015). The lower surface area is also probably due to inorganic material that partially fills or blocks the micropores (Lee et al. 2010). Ronsse et al. (2012) observed that higher amount of inorganic (i.e. ash content) in the biomass feedstock negatively correlate with specific surface area in the biochars. Wood-biochar offers the highest potential of surface area (127.0 m²/g) as all other biochar types (straw-biochar: 22.0 m²/g; green waste-biochar: 46.0 m²/g; algae-biochar: 19.0 m²/g). Wood-biochar had the lowest ash content (0.2%) than straw-biochar (7.9%), green waste-biochar (3.5%) and algae-biochar (38.4%). This is possibly explained by fusion of molten ash filling up pores in the biochar, thereby decreasing accessible surface area (Ronsse et al. 2012). Darmstadt et al. (2000) observed higher surface area for the softwoods (326.0 m²/g) than the hardwoods (221.0 m²/g) feedstock. The higher surface area of the charcoal produced from softwood bark may be explained by a partial activation of this sample by the water present in the feedstock and by creation of voids by evolving steam (Darmstadt et al. 2000). The less dense composition of the softwoods renders the more susceptible to thermal decomposition, resulting in more vesicles and pores throughout the wood structure, which effectively increases the surface area (Mukome et al. 2013).

3.3 CEC

The CEC of biochar depends on the type of feedstock. For example, the CEC of biochars derived from pig manure (32.7 cmol/kg) was lower than that of biochars produced from chicken manure (81.4 cmol/kg) at 500°C (Cely et al. 2015). However, the CEC of biochars derived from paper mill waste (9.0–18.0 cmol/kg) (Van Zwieten et al. 2010) was significantly lower than the CEC of biochars derived from sugar cane bagasse (122.0 cmol/kg) (Carrier et al. 2012). This difference was observed because biomass with a high content of ash can produce biochar with a higher CEC (Yang et al. 2015). The reason for this ability might be that alkali and alkali metals in biomass promote the formation of O-containing surface functional groups (Cely et al. 2015; Tag et al. 2016). Thus, the CEC of manure-derived biochar is higher than that of woody biochar (Tag et al. 2016). Tag et al. (2016) observed that CEC was the highest in algal (57.5 cmol/kg) and poultry litter (48.4 cmol/kg) derived biochars and the lowest in biochars produced from orange pomace (29.9 cmol/kg) at 500°C. Non-wood-derived biochars had increased CEC and surface acidity when compared to the wood biochars (Mukome et al. 2013). Similar results were reported by Gaskin et al. (2008), who found a significantly higher CEC for biochar from poultry litter (38.3 cmol/kg) than for biochars produced from peanut hulls (4.6 cmol/kg) and pine chips (5.0 cmol/kg) at 500°C. This decrease could be in part due to the reduction of the content of oxygenated functional groups on the biochar surface (Singh and Cowie 2010). Suliman et al. (2016) observed that biochars derived from Douglas fir wood had higher CEC than Douglas fir bark-biochar and hybrid poplar-biochar. This difference could be attributed to the combination of carboxylic functional groups, which contribute most of the CEC among the acidic
functional groups, and specific surface area (Singh and Cowie 2010).

3.4 pH

The pH of biochars is generally alkaline (from 7.1 to 10.5) (Inyang et al. 2010; Lehmann et al. 2011). Yuan et al. (2011) reported that biochars produced from corn straw, peanut and soybean (at a temperature of 300 °C) were alkaline (9.4, 8.6 and 7.7, respectively), while the pH values of biochar produced from canola straw were acidic (around 6.5). Differences in pH can result from biomass type. Biochar produced from wood has an average pH lower by 2 pH units than the values for other biomasses formed under similar pyrolysis conditions (Tag et al. 2016). Mukome et al. (2013) reported that non-wood-derived biochars have higher pH values (about 3.4 units). The basicity of the non-wood-derived biochar arises from the presence of salts (carbonates and chlorides of potassium and calcium in the ash) (Montes-Morán et al. 2004). El-Gamal et al. (2017) reported that the pH value of sugar cane-biochar (8.6) was lower than rice husks-biochar (8.9). This result could be due to the sugar cane-biochar (19.1%) had lower ash content than rice husks-biochar (40.2%). Increase of the pH value could be attributed to the concentration of non-pyrolyzed inorganic elements and also to decomposition of organic matrix (Garcia-Jaramillo et al. 2015).

The pH of a biochar is likely to be correlated with its contents of lignin, hemicelluloses or cellulose and the presence of oxygen functionalities (Ronsse et al. 2012). Yuan et al. (2011) reported that the –COO– and –O– groups and the carbonate content of the biochars were responsible for the alkaline properties. Mukome et al. (2013) showed that biochar pH correlated best with O content ($R^2 = 0.7$), corroborating previous findings that biochar basicity resulted from oxygen-rich functional groups such as $\gamma$-pyrone-type, chromene, diketone, or quinine groups (Montes-Morán et al. 2004). The formation of compounds such as levoglucosan (from pyrolysis of cellulose material) and its by-products (levoglucosenone, furfural, 2,3-butanediol and 5-methylfurfural) results in oxygen functional groups during the pyrolysis (Kawamoto and Saka 2003). This process is associated with the polymerization/condensation reactions of aliphatic compounds and with the effect of the dehydration of the feedstock (Zielinska et al. 2015). Additionally, Li et al. (2013) reported that pH of biochars derived from rice straw and rice bran was negatively correlated with aliphatic O-alkylated carbons and anomeric O–C–O carbons, but positively correlated with fused-ring aromatic structures and aromatic C–O groups. Consequently, a larger amount of carboxyl groups in the obtained biochar that are reduced during pyrolysis and/or acidic groups that become deprotonated to the conjugated bases result in a more alkaline pH of the biochar (Tag et al. 2016).

4 Biochar influence on soil quality

Biochar interacts physically with the soil fractions (Verheijen et al. 2010). Sub-molecular interactions with clay and silt particles, as well as with SOM (soil organic matter), occur through van der Waals forces and hydrophobic interactions (Xueyong et al. 2018). Interactions at this scale determine the influence of biochar on soil psychochemical properties and also on the interactions with cations, anions and other organic compounds in the soil (Zhu et al. 2017). These interactions are very specific for biochar, with the exact properties being influenced by feedstock type and pyrolysis conditions (Janus et al. 2015; Verheijen et al. 2010). Application of biochar can have positive or negative effects on soil properties, including water holding capacity (Revell et al. 2012), CEC (Alburquerque et al. 2014), bulk density (Cabeza et al. 2018), and specific surface area (Tomczyk et al. 2019). Data from a few experiments concerning the application of biochar as a soil amendment in different doses on different soil types are presented in Table 3. Higher biochar stability is not only useful from a climate mitigation point of view but also for maintaining possible positive agronomic effects over longer periods of time (Enders et al. 2012).

4.1 Physicochemical properties of soil

Biochar application to soils has been proposed as one of the best techniques for climate change mitigation via C sequestration in soil (Lehmann et al. 2006). The long-term stability of biochar in soil is a key factor affecting the decrease of CO$_2$ emissions into the atmosphere (Cheng et al. 2008). A recent long-term experiment estimated that the mean residence time of C in biochars varies from 90 to 1600 years depending...
on the labile and intermediate stable C components (Singh et al. 2012). The changes in functional groups and their distribution in soil microaggregates are influenced by the presence of biochar, this being indicative of changes in the physical protection and processing of C in soil (Hernandez-Soriano et al. 2016). A few recent studies have shown that biochar can reduce nitrous oxide (N\textsubscript{2}O) and methane (CH\textsubscript{4}) emissions from soil via both biotic and abiotic mechanisms (Jha et al. 2010; Sun et al. 2018; Van Zwieten et al. 2009). Woolf et al. (2010) proposed a sustainable biochar concept, through which the emission of greenhouse gases including CH\textsubscript{4} and N\textsubscript{2}O can be avoided. Additionally, the bioenergy produced during the pyrolysis process offsets fossil energy consumption (Woolf et al. 2010).

Furthermore, biochars are abundant in mineral elements such as Na, K, Ca, Fe and Mg (Jha et al. 2010). Their concentrations increase with the pyrolysis temperature (Saletnik et al. 2016) and vary with the type of biomass (de la Rosa et al. 2014). In one study, the highest contents of P, K and Mg (4.3, 9.9 and 2.8 g/kg, respectively) were observed in the biochar obtained at the temperature of 500 °C, while the temperature of 400 °C resulted in the maximum contents of carbon and nitrogen (73.6% and 1.9%, respectively) (Saletnik et al. 2016). Cantrell et al. (2012) suggested that the various metals inherent in

### Table 3 Experiments about using of biochar as soil amendment in different doses on different soil type

| References            | Feedstock kind          | Pyrolysis temperature | Soil type (layer depth) | Biochar dose |
|------------------------|-------------------------|-----------------------|-------------------------|--------------|
| Ameloot et al. (2013)  | Willow wood, Swine manure | 350 °C, 700 °C        | Sandy loam soil          | 10.0 Mg/ha   |
| Jin et al. (2016)     | Swine manure            | 400 °C                | Silt loam soil (0–15 cm) | 0.5 and 1.5% |
| Mierzwa-Hersztek et al. (2016) | Poultry litter          | 300 °C                | Eutric Cambisol (0–10 cm) | 2.3 and 5.0 t/ha |
| Novak et al. 2009     | Peanut hulls, Pecan shells, Poultry litter, Switchgrass | 250–700 °C, 500 °C | Loamy sand (0–15 cm), Fine loamy kaolinitic thermic Typic Kandiudult (20–40 cm) | 2.0% (40–44 t/ha) |
| Novak et al. (2016)   | Poultry litter, Pine chip, Blends of the pine chip, Poultry litter | 500 °C | Forest loamy sand | 20.0 g/kg |
| Ouyang et al. (2014)  | Fresh dairy manure, Pine tree, Woodchip | 300 °C, 500 °C, 700 °C | Forest loamy sand | 5.0% |
| Tomczyk et al. (2019) | Wood waste             | 650 °C                | Haplic Luvisol, Haplic Podzol, Haplic Luvisol (0–20 cm) | 0, 0.01, 0.05, 0.1% |
| Usowicz et al. (2016) | Wood waste             | 360 °C                | Haplic Luvisol (0–20 cm) | 10.0, 20.0, 30.0 Mg/ha |
| Yao et al. (2012)     | Sugarcane bagasse, Peanut hull, Brazilian pepperwood, Bamboo | 300 °C, 450 °C, 600 °C | Sandy soil | 2.0% |
| Zheng et al. (2010)   | Corn cobs, Wood chips  | 450 °C                | Silt loam (0–10 cm)      | 40.0 t/ha    |

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animal litter may protect against the loss of volatile material by changing the dissociation energies of organic and inorganic C bonds. The addition of biochar to the soil should increase the concentrations of micronutrients that are easily available to plants (Alburquerque et al. 2014; Jha et al. 2010). Improvement of soil physical, chemical, and biological properties promotes plant productivity through increasing the amount and availability of nutrient elements, reducing nutrient leaching and mitigating losses of gaseous components (Ding et al. 2016). It is widely known that a high CEC corresponds to high nutrient contents (Liang et al. 2006). Guo et al. (2012) showed that biochar has a high CEC and is expected to retain more nutrients in soil and to decrease nutrient leaching.

Glaser et al. (2002) suggested the oxidation of aromatic C and formation of carboxyl groups to be the main reason for high CEC. This formation of carboxyl groups or other functional groups with a negative charge in the pH range of soils can be the result of two principally different processes: (1) surface oxidation of the biochar particles themselves and (2) adsorption of highly oxidized organic matter onto biochar surfaces (Lehmann et al. 2005). Low-temperature biochars are often employed due to more enhanced soil-biochar interactions in relation to high-temperature biochars (Joseph et al. 2010). Such low-temperature biochar also yields a greater recovery of C and other nutrients (feedstock dependent), which are usually lost at higher temperatures (Keiluweit et al. 2010). The low-temperature product, which has been pyrolysed between 400 and 500 °C, has its main advantage in increasing CEC. Furthermore, this biochar type sequesters soil C, however, not to the same extent as high-temperature biochar (Antal and Grønli 2003). High-temperature biochars consequently have lesser reactivity in soils than lower-temperature biochars, which tend to have a better impact on soil fertility (Antal and Grønli 2003; Steinbeiss et al. 2009). Liang et al. (2006) reported that the CEC was up to 1.9 times higher in Anthrosols with a high biochar concentration than in control soils. The same results were obtained by Sombroek et al. (1993). The changes in soil properties that occur with biochar amendments, such as increases in organic carbon, the contents of mineral elements and values of CEC have an influence on soil pH (Rutkowska et al. 2014). Hass et al. (2012) observed that the effect of biochar on soil pH increased with application rate and varied among different types of biochar. The increases in pH and the corresponding reduction in exchangeable Al could also have improved the chemical environment (e.g. soil pH, soil organic matter, phosphorus (P) or potassium (K) contents) of the biochar-amended soils for radish plants (Abdulaha-Al Baquy et al. 2017). Chan et al. (2007) reported that the pH increases were accompanied by a significant reduction in exchangeable Al by > 50.0% at the higher rates of biochar application, i.e., 50.0 and 100.0 t/ha.

Application of biochar can have a positive effect on another physicochemical property of soil: specific surface area (Anawar et al. 2015). Liang et al. (2006) reported that Anthrosols had a (up to 4.8 times) higher surface area than other soils due to their higher biochar concentrations. Similarly, Tomczyk et al. (2019) showed that the specific surface area of the non-modified silty Haplic Luvisol soil is almost three times higher than that of the sandy Haplic Podzol soil; amendment with biochars led to an increase of the specific surface area values in both soils. Modified Haplic Luvisol had a higher surface area due to higher biochar content (approx. 12.7–21.9 m²/g) than the non-modified loamy soil and modified Haplic Podzol also had higher surface area due to higher biochar content (approx. 2.5–11.6 m²/g) than the non-modified sandy soil. This suggests that Haplic Podzol had weaker interactions with than Haplic Luvisol. The larger specific surface area of Haplic Luvisol soil may be the result of the higher amounts of organic carbon as well as clay and silt materials in Haplic Luvisol in comparison to Haplic Podzol soil. Effect of biochar on soil specific surface area varies among biochar types. Lei and Zhang (2013) reported that wood biochar has a higher specific surface area (124.0 m²/g) than biochar from the dairy industry (83.4 m²/g). The application of biochar has been observed to increase the amount of macropores (with a maximum increase of 59.0%), and the amount of macropores in biochar increases with the pyrolysis temperature (Kutilek et al. 2006; Lei and Zhang 2013). The increase of specific surface area and porosity caused better water sorption (Nair et al. 2017).

Fertilization with biochar also has a positive effect on water holding capacity (Duong et al. 2017). Biochar can absorb water up to 5.0 times its own weight (Ga˛sior and Tic 2017). Biochar can increase the moisture and content of organic and inorganic
nitrogen compounds, which reduces lime in the litter, thereby reducing the pH of the litter and manure, which in turn reduces ammonia emissions (Gerlach 2014). Some studies have also shown how biochar changes water retention in soil. The increase in carbon content achieved by adding biochar to the soil contributes to stimulating the humification and carbon sequestration processes as well as the improvement of soil density and water retention (Nair et al. 2017). Cybulak et al. (2016) reported that the application of biochar increases the hygroscopic moisture content of soil (by about 1.5–3.0%), which would be very beneficial to dry and degraded soils. Smaller biochar particle sizes can also increase water retention but may reduce saturated flow (Blanco-Canqui 2017). However, Glaser et al. (2002) reported that Amazonian charcoal-rich anthrosols had a field water retention capacity 18.0% higher than that of surrounding soil without charcoal. It can be expected that charcoal addition may cause an increase or decrease in the water retention of soils, but the direction of the effect will depend on the original characteristics of the soil (Woolf 2008). Tryon (1948) showed that charcoal increased the available moisture in sandy soil but had no effect in loamy soil and decreased the available moisture in clay soil. This suggests that charcoal addition may be ill-suited to clay soils. Conversely, an increase of available moisture observed in sandy soils may make biochar a useful tool in the reversal of desertification (Woolf 2008). Usowicz et al. (2016) described that biochar amendment to fallow land caused decreases in bulk density, particle density, thermal conductivity and thermal diffusivity. However, no significant effect of surface-applied biochar was observed on the soil thermal conductivity and thermal diffusivity under grassland. Moreover, biochar application to agricultural soils can change the surface albedo, which can counteract the climate-mitigating potential of biochar (Usowicz et al. 2016). If there is greater albedo, then less UV will be absorbed by the soil (Kuppusamy et al. 2016). Biochar amendments caused reductions of albedo under both grassland and fallow land (Usowicz et al. 2016).

4.2 Inorganic and organic contaminants in soil

Numerous studies have proven that biochar increases the ability of soil to adsorb heavy metals and other contaminants, which is important for environmental protection and management (Tomczyk et al. 2019; Wei et al. 2018). Biochars have a carbonized and a non-carbonized fraction, which may interact with soil contaminants through oxygen-containing carboxyl, phenolic, hydroxyl and lactonic surface functional groups (Ahmad et al. 2014a). These fractions can play different roles in the adsorption process (Cao et al. 2009). The carbonized fraction is similar to the “glassy” fraction (analogous to glassy polymers), and the non-carbonized fraction is similar to the “soft” fraction (analogous to rubbery polymers) (Xia and Ball 1999). Biochar can be considered as a soil amendment that reduces the biotoxicity of pollutants (Cha et al. 2016; Li et al. 2017; Wang et al. 2019).

The effectiveness of contamination removal depends on the biochars specific surface area and cation exchange capacity (Ahmad et al. 2014b; Kołodyńska et al. 2012; Touray et al. 2014), as well as on the kind of interfering ions (Bogusz et al. 2015), pH of the solution, the dosage of biochar applied (Kılıç et al. 2013; Tomczyk et al. 2019), soil type, soil pH, contact time, metal concentration and temperature (Bradl 2004; Dube et al. 2001; Srivastava et al. 2005) as well as biochar type. Many studies have investigated heavy metal adsorption by biochars, including the adsorption of copper ions from water by a soil:biochar mixture at different pH values (Tomczyk et al. 2019), adsorption of heavy metal ions on kaolin (Srivastava et al. 2005), adsorption of zinc, copper and lead on zeolite (Perić et al. 2004), adsorption of platinum (IV) ions in loess soil (Bojanowska and Jackowska 2005), and the adsorption of heavy metal ions on biochars (Kołodyńska et al. 2012). All of the above studies were aimed at understanding how heavy metal ions are adsorbed from the liquid phase to the surface of the adsorbent and how the presence of biochar influences this process in soil (Liu et al. 2014). Colloids and organic ligands (Kerndorf and Schintzer 1980; Lion et al. 1982) as well as inorganic ones (Srivastava et al. 2005) have an influence on the adsorption process. Cao et al. (2009) investigated the sorption capacities of dairy manure biochar produced at low temperatures (200 °C and 350 °C) and found that the biochar was six times more effective in removing lead (Pb) from wastewater than a commercial activated carbon. Tong et al. (2011) investigated the copper adsorption capacity of three different biochars obtained by the pyrolysis of nut straw, soy and rapeseed. They found that the highest adsorptive
capacity was exhibited by the biochar obtained from nut straw, then from soybeans, and then from rapeseed. They proved that the copper adsorption occurs through the formation of complexes with surface groups, specifically –COOH and phenolic groups (Tong et al. 2011). However, Karami et al. (2011) reported that the addition of oak biochar (in the amount of 20.0% (v/v)) to soil reduced Cu sorption by approx. 69.0%. Xu and Zhao (2013) showed that biochar from straws and peanut (in the amounts of 3.0 and 5.0% (v/v)) increased the Cu sorption capacities of Oxisol, Utisol and Ultimate soils, whereas biochar from rice reduced Cu in Ultisol derived from Quaternary red earth at the same doses (Jiang et al. 2012). However, woody biochar increased Cu adsorption onto Haplic Podzol at pH 3.0 and 5.0, with the most considerable changes observed at pH 5.0 (increase in adsorption of 23.5%) (Tomczyk et al. 2019).

Biochar may also adsorb other contaminants, such as PAH, antibiotics, and pesticides (Mui et al. 2010; Xu et al. 2011). For example, Chen and Yuan (2011) investigated the effect of biochars (obtained from pine needles at different temperatures) on the sorption of the PAHs naphtalene and phenantrene in soil. Biochar produced at high pyrolysis temperatures (400 and 700 °C) demonstrated higher efficiency in the sorption affinity of biochar-soil than biochar obtained at low pyrolysis temperatures (100 and 300 °C). Phenanthrene sorption of above 99.0% was controlled by biochar produced at 300 and 400 °C when present at 5.0%. When the biochar content was only 0.5% for biochar 300 °C and 0.1% for biochar 400 °C, the relative contributions of biochar and soil to total phenanthrene sorption were similar, but when the biochar content was increased to 5.0%, phenanthrene sorption of 90.0% and 98.0% was contributed by biochar produced at 300 and 400 °C, respectively (Chen and Yuan 2011). With the addition of 5.0% biochar at 100 °C, 5.0% biochar at 300 °C, 5% biochar at 400 °C, and 2.0% biochar at 700 °C, sorption of phenanthrene was enhanced by 1.8–3.4 times, 5.1–10.4 times, 23.0–70.6 times, and 43.7–85.5 times in comparison with unamended soil (Chen and Yuan 2011). A similar phenomenon was observed for naphtalene sorption; the enhanced intensities were 1.7–3.2 times, 28.3–113.0 times, 58.6–314.0 times, and 138.0–1170.0 times the sorption of amended soil with a 5.0% series of biochar produced at 100, 300, 400, and 700 °C, respectively. The biochar dominated the overall sorption of naphtalene when added to the soil at a proportion of at least 0.5% for the biochar produced at 300 °C content and 0.1% for the biochar produced at 400 °C (Chen and Yuan 2011). Overall, for biochar-soils with the same biochar content, the saturated adsorption capacity of a sorbent estimated from the high concentration data values for PAHs increased with the biochar pyrolysis temperature in the order of 100 < 300 < 400 < 700 °C (Chen and Chen 2009; Chen and Yuan 2011). Li et al. (2019) investigated the sorption of sulfadiazine and tetracycline on wood biochar obtained at 600–800 °C. The authors analysed the effects of porosity of biochar, size of antibiotic molecules and pH on the sorption process. It was found that the production of biochar at a higher temperature improved its mesoporosity (from 34.3 to 124.0 m²/g) and its affinity for antibiotic sorption from the aquatic environment (from 6.42 to 163.0 mg/g for tetracycline sorption and from 2.20 to 261.0 mg/g for sulfadiazine sorption). The sorption efficiency of mesoporous biochar was associated with a higher internal porosity, which promoted the adsorption of smaller-size antibiotic molecules (Li et al. 2019). Additionally, desorption studies indicate irreversible adsorption of antibiotics, i.e., biochar adsorbs them permanently and they will probably not leach out in the soil (Li et al. 2019). Furthermore, Garcia-Perez (2008) found that biochars produced at temperatures above 700 °C are typically related to the production of PAHs, which are hazardous because of their carcinogenic and mutagenic properties, and low-temperature biochars (pyrolysed in the temperature range of 350–600 °C) appear to carry fewer toxic inferences. Therefore, biochar that is produced as a soil fertility amendment needs to be specifically aimed at carbonizing the biomass material under moist conditions and at low temperatures (Novak et al. 2009).

4.3 Biological properties of soils

The pore system of biochar provides a safe habitat for soil microorganisms (i.e., mycorrhizal fungi, actinomycetes bacteria) (Compant et al. 2010). These microorganisms are food for protozoa, mites, nematodes and other soil biota (Briones 2014). Biochar helps maintain microbiological populations at a higher level and simultaneously reduces the greenhouse gas emissions of soil (Compant et al. 2010). Weyers and Spokas (2011) reported short-term negative effects...
and long-term null effects of biochar amendment on earthworm activity in soil. Biochar derived from rice residues was observed to have a negative effect on the earthworm population that was related to the increase in soil pH induced by the biochar (Haefele et al. 2011). Li et al. (2011) recommended that a wet biochar application to soil could help mitigate negative effects on earthworms by preventing desiccation. Also, a positive effect of biochar amendment has been observed in soil enzymatic activity, which is an indicator of higher soil quality (Mierzwa-Hersztek et al. 2016; Ouyang et al. 2014). Ameloot et al. (2013) and Mierzwa-Hersztek et al. (2016) showed that biochar amendment caused an increase in the activity of dehydrogenase and urease in soil. Dehydrogenase activity increased by 19.0% and urease activity by 44.0% (Mierzwa-Hersztek et al. 2016). Moreover, Mierzwa-Hersztek et al. (2016) reported that the addition of biochar to soil reduced soil acidity and increased the content of nitrogen and organic carbon.

5 Conclusions

This paper provides a review of the effects of temperature and type of feedstock on the physicochemical properties of biochar and its potential use as a soil conditioner.

The physicochemical properties (pH, specific surface area, pore size, CEC, volatile matter, ash and carbon content) of biochar change with pyrolysis temperature and feedstock kind. Studies show that CEC and volatile matter decreased with increasing pyrolysis temperature, whereas pH, specific surface area, ash and carbon content, pore volume increased with the increase in pyrolysis temperature. Increasing temperature also decreased the number of acidic functional groups, especially carboxylic functional groups, and caused appearance of basic functional groups. Biochar produced at high temperatures (600–700 °C) exhibits a highly aromatic nature with well-organized C layers. This is most likely due to the degree of organic matter decomposition and the formation of micropores. In addition, this set of physicochemical properties is also a consequence of destruction of aliphatic alkyls and ester groups as well as the exposure of the aromatic lignin cores to high pyrolysis temperatures.

The moisture, as well as lignin and cellulose content in biomass have considerable influence on biochar formation. These can be attributed to the presence of different organic constituents in feedstock. Lignin is amorphous and hydrophobic polymer with high molecular weight and numerous functional groups of aromatic substructure. Cellulose and hemicelluloses are consisting of simple sugar monomer, which decompose at a temperature lower than 450 °C. Cellulose present in the biomass helps in the formation of tar, while high lignin content is favourable for char production. Biochars produced from animal litter and solid waste feedstocks exhibit lower surface areas, carbon content, volatile matter and high CEC as compared to biochars produced from crop residue and wood biomass, even at higher pyrolysis temperatures. Biochar derived from manures, sewage sludge or crop residues show higher potential as nutrient source or inorganic sorbent. Wood-biochar possesses considerable merit for the improvement: (1) organic pollutants removal (antibiotics, pesticides), (2) carbon sequestration and (3) soil pH, due to the high carbon and ash content, as well as high specific surface area, volatile matter and alkaline pH.

Biochar reveals the potential to contribute to resolving economic, public health and environmental problems that are widespread and need to be overcome. In summary, more research is needed regarding biochar technology, suitability and sustainability as a soil fertilizer or decontaminator.

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