Review

Application of Rice Husk Biochar for Achieving Sustainable Agriculture and Environment

Hossein ASADI1, 3, Mohammad GHORBANI2, Mehran REZAEI-RASHTI3, Sepideh ABRISHAMKESH2, Elnaz AMIRAHMADI4, CHEN Chengrong3, Manouchehr GORJI1

(1Soil Science Department, University of Tehran, Karaj 31587-77871, Iran; 2Soil Science Department, University of Guilan, Rasht 41996-13776, Iran; 3Griffith School of Environment and Science, Australian Rivers Institute, Griffith University, Brisbane, Queensland 4111, Australia; 4Department of Forest Ecology, Sari Agricultural Sciences and Natural Resources University, Sari 48181-68984, Iran)

Abstract: This paper critically reviewed the current knowledge and challenges of rice husk biochar (RHB) production and its effects on soil properties, plant growth, immobilization of heavy metals, reduction of nutrient leaching and mitigation of greenhouse gas emissions. The characteristics of RHBs produced at various pyrolysis temperatures were discussed and compared to biochars derived from other agro-industrial wastes. RHBs produced at higher pyrolysis temperatures show lower hydrogen/carbon ratio, which suggests the presence of higher aromatic carbon compounds. The increase of pyrolysis temperature also results in production of RHBs with higher ash content, lower yield and higher surface area. RHB usually has higher silicon and ash contents and lower carbon content compared to biochars derived from other feedstocks at the same pyrolysis conditions. Although it depends on soil type, RHB application can improve soil organic carbon content, cation exchange capacity, available K concentration, bulk density and microbial activity. The effect of RHB on soil aggregation mainly depends on soil texture. The growth of different crops is also enhanced by application of RHB. RHB addition to soil can immobilize heavy metals and herbicides and reduce their bioavailability. RHB application shows a significant capacity in reduction of nitrate leaching, although its magnitude depends on the biochar application rate and soil biogeochemical characteristics. Use of RHB, especially in paddy fields, shows a promising mitigation effect on greenhouse gas (CH4, CO2 and N2O) emissions. Although RHB characteristics are also related to other factors such as pyrolysis heating rate and residence time, its performance for specific applications (e.g. carbon sequestration, pH amendment) can be manipulated by adjusting the pyrolysis temperature. More research is needed on long-term field applications of RHB to fully understand the advantages and disadvantages of RHB as a soil amendment.

Key words: carbon sequestration; crop production; greenhouse gas; heavy metal; nitrogen leaching; soil amendment; rice husk biochar

Currently, environmental concerns about human induced carbon dioxide (CO2) emission and global warming, high nutrient levels in surface streams and groundwater, and eutrophication have necessitated the identification of efficient strategies to mitigate environmental risks associated with intensive agricultural systems. Production and application of biochar to soil can be an effective option for carbon (C) sequestration, reduction of methane (CH4) and nitrous oxide (N2O) emissions (Pratiwi and Shinogi, 2016; Juriga et al, 2018; Oni et al, 2020), reduction in the bioavailability of heavy metals and organic pollutants (Ahmad et al, 2014; Ajayi and Horn,
2017) and controlling the leaching of nutrients and pollutants from soil (Liu Z Q et al, 2017; Oni et al, 2020). Biochar can boost soil C sequestration when applied to soil, i.e. remains in soil for hundreds of years (Lehmann et al, 2011). Furthermore, improvement of soil properties (Pandian et al, 2016) and plant growth enhancement (Rajkovich et al, 2012) can be achieved as a result of biochar amendment to soil.

Various biomass feedstocks such as crop residues, woody materials, green wastes, animal manures and agricultural wastes including rice husks (RH), can be used for biochar production (Lehmann et al, 2011). Conversion of wastes into biochar through pyrolysis process can result in advantages including energy production, sustainable waste recycling, C sequestration, improvement of soil quality and better plant growth (Dong et al, 2015; Huang et al, 2018). Rice is the main food for more than 50% of the world’s population, particularly in Asia. In some countries like Japan, it is also used for other purposes including flour, livestock feed and biofuel (Shackley et al, 2012). Accordingly, production of edible and forage rice has been increasing rapidly in the past decade (Oladele, 2019), which is in part due to the application of high amounts of nitrogen (N) fertilizers. As a result, excessive N input has led to an inefficient use of N by plants and high N losses to the environment, which is adversely affecting air (greenhouse gas emissions) and water quality, soil biodiversity and human health (Dong et al, 2017).

Extensive research has been done on production of biochar from RH, its characterization and assessment of its effect on soil and crop properties. Although there are some reviews available on the biochar properties and its application to soil (Kajetan et al, 2019; Tomczyk et al, 2020), there is still a need for a specific review on RH biochar (RHB) due to its significant potential for large scale production and application to agricultural lands. This review summarized the research on RHB production and its characteristics in the past decade. In addition, the effects of RHB application on improvement of soil chemical, biological and physico-mechanical properties, C sequestration, plant growth, stabilization of heavy metals in soil, nutrient leaching and greenhouse gas (GHG) emissions are reviewed. A summary of the scope and content of this review is presented in Fig. 1.

**Importance of RH as feedstock for biochar production**

The area of paddy fields is about $1.4 \times 10^8$ ha worldwide (Kögel-Knabner et al, 2010). RH, as the main byproduct of the rice milling process, is produced in huge quantities on a global scale, especially in Asia (Table 1). Considering that RH on average contributes to around 20% of paddy rice weight (Pode, 2016), the annual global production of RH is estimated to be approximately 148.4 and 156.4 million tons in 2014 and 2018, respectively (Table 1).

RH, as an agro-industrial waste, can’t be used efficiently due to its intrinsic properties such as rigid surface, high silicon content, poor feeding value and high resistance to decomposition by soil microorganisms (Zou and Yang, 2019). The majority of produced RH is underutilized or left unused (Fig. 2), except for limited application in agricultural and bioenergy industries. Burning of RH in open fields is a common

**Fig. 1. Summary of scope and content of review focused on production and characterization of rice husk biochar and its application to soil for sustainability of agriculture and environment.**

RHC, Rice husk char; GHG, Greenhouse gas; EC, Electrical conductivity.
land management practice used by farmers. This would result in the loss of nearly all C, around 80% of sulphur (S) and N, and 10%–20% of phosphorus (P) and potassium (K) contents of RH. Furthermore, it is thought to be one of the major contributors to aerosols, greenhouse and hazardous gas emissions including CO₂, CH₄, NOₓ and SO₂ into the atmosphere (Singh et al., 2014). In India, 18–22 million tons of RH are obtained annually, which are mainly used as biofuel (Bharadwaj et al., 2004). The direct incorporation of RH into soil has been suggested in the past four decades (Ponamperuma, 1982) and significantly improves soil quality and productivity (Williams et al., 1972), but it has not been adapted by all farmers. In addition, it is not a sustainable way of C sequestration contributing to GHG emissions (Haefele et al., 2011).

In recent years, biochar production from underutilized wastes is regarded as an efficient approach to achieve sustainable agriculture and environment. In this regard, RH has been recognized as a valuable resource for the production of biochar as a soil amendment (Haefele et al., 2011). An overview of various applications of RHB is displayed in Fig. 2.

**Rice husk biochar production and characterization**

In countries like India and Cambodia, RH is extensively used as a conventional fuel in rural regions due to its abundance (Bharadwaj et al., 2004; Shackley et al., 2012). However, RH combustion is usually an incomplete oxidation process due to its high ash content, which covers the remaining parts. The main component of ash is silicon oxide (Bharadwaj et al., 2004), which protects the organic materials like a shield (Shackley et al., 2012). As a result, burning of RH is practically a pyrolysis process. The efficiency of C conversion in this process is as low as 55% (Bharadwaj et al., 2004). Therefore, the residue is a mixture of ash and a high carbon char (RHC, rice husk char). The yield (ratio of char to feedstock) and C content of the produced char from RH combustion are both around 35% (Shackley et al., 2012).

Table 1. Annual worldwide production of paddy rice and rice husk in 2014 and 2018 (FAOSTAT in September 2020). 

<table>
<thead>
<tr>
<th>Region</th>
<th>Paddy rice 2014</th>
<th>Rice husk 2014</th>
<th>Paddy rice 2018</th>
<th>Rice husk 2018</th>
</tr>
</thead>
<tbody>
<tr>
<td>Global</td>
<td>742.4 ± 22</td>
<td>148.4 ± 6.1</td>
<td>782.1 ± 33.2</td>
<td>156.4 ± 6.6</td>
</tr>
<tr>
<td>Africa</td>
<td>30.7 ± 0.2</td>
<td>51.3 ± 0.8</td>
<td>33.2 ± 0.7</td>
<td>6.6 ± 0.1</td>
</tr>
<tr>
<td>America</td>
<td>37.7 ± 1.0</td>
<td>75.3 ± 7.5</td>
<td>38.8 ± 5.1</td>
<td>7.8 ± 0.1</td>
</tr>
<tr>
<td>Asia</td>
<td>669.2 ± 4.0</td>
<td>133.8 ± 0.8</td>
<td>705.4 ± 4.0</td>
<td>141.1 ± 0.8</td>
</tr>
<tr>
<td>Europe</td>
<td>4.0 ± 0.2</td>
<td>8.0 ± 0.7</td>
<td>4.0 ± 0.7</td>
<td>0.1 ± 0.1</td>
</tr>
<tr>
<td>Oceania</td>
<td>0.8 ± 0.2</td>
<td>1.0 ± 0.7</td>
<td>0.7 ± 0.7</td>
<td>0.1 ± 0.1</td>
</tr>
</tbody>
</table>

Abrishamkesh et al (2015), Pratiwi et al (2016), Wei et al (2017), Ghorbani et al (2019), Huang et al (2019) and Oladele (2019) have studied RHB production and characterization in the past decade. RHB has been produced by pyrolysis process under a wide range of temperatures from 250 °C to 750 °C. The main analyses for characterization of RHB include: structural properties (i.e. surface area, structure and pore sizes) by scanning electron microscopy (SEM); elemental compositions (C, N, etc.) using Elemental Analyzer; pH and electrical conductivity (EC) of a 1:n (RHB: water) suspension; cation exchange capacity (CEC) and bulk density. Table 2 summaries the selected basic characteristics of RH, RHC and RHB produced at different pyrolysis temperatures. Biochar properties are strongly related to feedstock type, pyrolysis conditions including peak temperature, heating rate, pressure, residence time and their particle size (Phuong et al, 2015; Wei et al, 2017). However, pyrolysis peak temperature is the main influencing factor on biochar characteristics, which has been investigated (Uchimiya et al, 2011; Phuong et al, 2015).

Some common properties of RHB are compared to properties of biochars derived from rice straw, corn cob, wood, bagasse, tea waste and grape pomace under the same pyrolysis conditions such as peak temperature (Table 3). The results showed that (i) RHB has higher content of silicon compared to corn cob biochar, (ii) RHB has lower pH value compared to most of other biochars (i.e. biochars produced from rice straw, corn cob, wood and grape pomace), (iii)
ash content of RHB is significantly higher than other biochars, and (iv) C content of RHB is lower than other investigated biochars.

**Chemical properties of rice husk biochar**

**pH**

RH is slightly acidic (pH 6.5–6.8) (Haefele et al., 2008; Shackley et al., 2012; Abrishamkesh et al., 2015). pH value of RHBs produced at pyrolysis temperatures from 250 °C–300 °C to 600 °C–750 °C, ranging from 6.8 to 10.7 (Table 2). There is also a significant positive correlation between pyrolysis temperature and pH of RHBs (Fig. 3). The separation of alkali salts (Shinogi and Kanri, 2003) and the loss of acidic functional groups (Ahmad et al., 2012) would be the reason for the increase in pH value of RHBs. Wei et al. (2017) reported that basic functional groups are higher and acidic functional groups are lower in the RHBs produced at high pyrolysis temperatures. The basic functional groups are mainly associated with the ash content (Suliman et al., 2016), which usually increases with pyrolysis temperature.

![Table 2. Summary of selected properties of rice husk (RH), rice husk char (RHC) and rice husk biochar (RHB) produced at different pyrolysis temperatures.](image-url)
**Elemental composition**

The main elements of RHBs include C, hydrogen (H), oxygen (O), N, P, K and silicon (Si). There are significant positive and negative correlations between pyrolysis temperature and C and H contents in RHB, respectively (Fig. 3). C content of RHBs usually increases by increasing of pyrolysis temperature (Edouah et al., 2019). There are also reports that show the independency of C content from pyrolysis temperature in RHB (Abrishamkesh et al., 2015). However, most importantly, C in RHB is mainly in more stable forms compared to RH. In contrast with C content, H content of RHBs decreases by increasing of pyrolysis temperature and in comparison with RH (Fig. 3 and Table 2).

Ratios of H/C and O/C are practical indicators in determining the level of carbonization and conversion of biomass to biochar (Crombie et al., 2013). By increasing pyrolysis temperature, O and H contents decrease during dehydration and decarboxylation reactions, which result in the decrease of H/C and O/C ratios (Xiao et al., 2016). Higher loss of H and O indicates higher carbonization of the feedstock, greater hydrophobicity of the biochar, formation of more fused aromatic rings and stronger C structure of the biochar (Spokas, 2010). The molar ratios of H/C and O/C are used to estimate the aromaticity and polarity of biochars (Uchimiya et al., 2010). The lower these ratios are, the higher the aromatic C components in RHB. The available information suggests a significant negative correlation between pyrolysis temperature and H/C ratio (Fig. 3).

Reduction of H/C ratio shows the lower hydrophilicity of RHBs produced in higher pyrolysis temperatures (Ahmad et al., 2012). RHBs produced in high temperatures (> 500 ºC) are also extremely carbonized and have high aromaticity (Pratiwi et al., 2016; Wei et al., 2017). Regarding the H/C ratio, a maximum value of 0.7 is suggested for the distinction of biochar and biomass, and a minimum of 0.2 for the differentiation of soot and biochar (Enders et al., 2012).

Table 2 shows that the optimum H/C and O/C ratios for biochar and biomass, and a minimum of 0.2 for the differentiation of soot and biochar (Enders et al., 2012). Table 2 shows that the optimum H/C and O/C ratios for RHB occur at pyrolysis temperatures between 350 ºC and 500 ºC. In other words, this temperature range would be the most suitable range for RHB production.

Molar H/C and O/C ratios of RHBs produced at different pyrolysis temperatures are used to obtain a Van Krevelen diagram (Fig. 3). Spokas (2010) suggested that biochars with O/C ratio of more than

<table>
<thead>
<tr>
<th>PT (ºC)</th>
<th>Feedstock</th>
<th>pH</th>
<th>EC (dS/m)</th>
<th>Ash (%)</th>
<th>C (g/kg)</th>
<th>H (g/kg)</th>
<th>N (g/kg)</th>
<th>Si (g/kg)</th>
<th>H/C</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>350</td>
<td>Rice husk</td>
<td>9.9</td>
<td>35.3</td>
<td>478.0</td>
<td>28.0</td>
<td>3.0</td>
<td>–</td>
<td>–</td>
<td>0.69</td>
<td>Phuong et al, 2015</td>
</tr>
<tr>
<td>450</td>
<td>Rice husk</td>
<td>10.5</td>
<td>23.4</td>
<td>525.0</td>
<td>30.0</td>
<td>12.0</td>
<td>–</td>
<td>–</td>
<td>0.68</td>
<td></td>
</tr>
<tr>
<td>550</td>
<td>Rice husk</td>
<td>10.4</td>
<td>30.8</td>
<td>521.0</td>
<td>23.0</td>
<td>10.0</td>
<td>–</td>
<td>–</td>
<td>0.04</td>
<td></td>
</tr>
<tr>
<td>650</td>
<td>Rice husk</td>
<td>11.7</td>
<td>40.5</td>
<td>512.0</td>
<td>16.0</td>
<td>3.0</td>
<td>–</td>
<td>–</td>
<td>0.03</td>
<td></td>
</tr>
<tr>
<td>300</td>
<td>Rice husk</td>
<td>7.1</td>
<td>–</td>
<td>415.8</td>
<td>–</td>
<td>7.1</td>
<td>138.5</td>
<td>–</td>
<td>–</td>
<td>Edouah et al, 2019</td>
</tr>
<tr>
<td>450</td>
<td>Corn cob</td>
<td>8.9</td>
<td>–</td>
<td>729.8</td>
<td>–</td>
<td>8.9</td>
<td>13.9</td>
<td>–</td>
<td>–</td>
<td></td>
</tr>
<tr>
<td>550</td>
<td>Rice husk</td>
<td>7.4</td>
<td>–</td>
<td>419.8</td>
<td>–</td>
<td>7.4</td>
<td>169.2</td>
<td>–</td>
<td>–</td>
<td></td>
</tr>
<tr>
<td>650</td>
<td>Rice husk</td>
<td>9.2</td>
<td>–</td>
<td>743.1</td>
<td>–</td>
<td>9.2</td>
<td>14.0</td>
<td>–</td>
<td>–</td>
<td></td>
</tr>
<tr>
<td>400</td>
<td>Rice husk</td>
<td>9.5</td>
<td>–</td>
<td>425.5</td>
<td>–</td>
<td>9.5</td>
<td>194.5</td>
<td>–</td>
<td>–</td>
<td></td>
</tr>
<tr>
<td>500</td>
<td>Corn cob</td>
<td>10.3</td>
<td>–</td>
<td>780.7</td>
<td>–</td>
<td>10.3</td>
<td>14.4</td>
<td>–</td>
<td>–</td>
<td></td>
</tr>
<tr>
<td>600</td>
<td>Sugarcane bagasse</td>
<td>10.4</td>
<td>32.8</td>
<td>546.7</td>
<td>–</td>
<td>4.7</td>
<td>–</td>
<td>–</td>
<td>–</td>
<td></td>
</tr>
<tr>
<td>700</td>
<td>Sugarcane bagasse</td>
<td>9.6</td>
<td>12.4</td>
<td>702.2</td>
<td>–</td>
<td>4.7</td>
<td>–</td>
<td>–</td>
<td>–</td>
<td></td>
</tr>
<tr>
<td>450</td>
<td>Rice husk</td>
<td>10.7</td>
<td>33.6</td>
<td>545.0</td>
<td>–</td>
<td>3.6</td>
<td>–</td>
<td>–</td>
<td>–</td>
<td></td>
</tr>
<tr>
<td>500</td>
<td>Sugarcane bagasse</td>
<td>9.7</td>
<td>12.8</td>
<td>696.3</td>
<td>–</td>
<td>3.8</td>
<td>–</td>
<td>–</td>
<td>–</td>
<td></td>
</tr>
<tr>
<td>600</td>
<td>Rice husk</td>
<td>9.7</td>
<td>32.6</td>
<td>454.0</td>
<td>24.0</td>
<td>5.0</td>
<td>–</td>
<td>0.63</td>
<td></td>
<td>Fazeli et al, 2020</td>
</tr>
<tr>
<td>500</td>
<td>Wood</td>
<td>7.8</td>
<td>42.0</td>
<td>379.1</td>
<td>20.6</td>
<td>8.9</td>
<td>–</td>
<td>–</td>
<td>–</td>
<td></td>
</tr>
<tr>
<td>700</td>
<td>Rice husk</td>
<td>7.3</td>
<td>28.3</td>
<td>863.0</td>
<td>7.9</td>
<td>7.3</td>
<td>–</td>
<td>–</td>
<td>–</td>
<td>Kizito et al, 2015</td>
</tr>
<tr>
<td>300</td>
<td>Wood</td>
<td>9.8</td>
<td>5.9</td>
<td>635.5</td>
<td>0.3</td>
<td>8.5</td>
<td>–</td>
<td>–</td>
<td>–</td>
<td>Manolikaki and Diamadopoulos, 2017</td>
</tr>
<tr>
<td>100</td>
<td>Rice husk</td>
<td>10.8</td>
<td>25.7</td>
<td>466.0</td>
<td>16.0</td>
<td>–</td>
<td>–</td>
<td>–</td>
<td>–</td>
<td></td>
</tr>
</tbody>
</table>

PT, Pyrolysis temperature; EC, Electrical conductivity; C, Carbon; H, Hydrogen; N, Nitrogen; Si, Silicon.
0.6 and less than 0.2 have a half-life of < 100 and > 1000 years, respectively. Fig. 3 suggests a half-life of 100±1000 years for RHBs produced under different pyrolysis temperatures (mainly with O/C ratio of 0.2–0.6).

**Chemical functional groups**

The presence and changes in the functional groups of RHBs have been evaluated by Fourier transform infrared spectroscopy analysis. Silica functional groups (i.e. Si–OH, Si–O–Si and Si–H) are dominant in RHBs produced at low pyrolysis temperatures. However, the increase in pyrolysis temperature up to 750 ºC causes the loss of aliphatic C–H stretching vibration and the high-intensity region of C=C ring stretching (Wei et al, 2017). Infrared reflectance spectra of the RHBs shows a strong peak related to aromatic C–H out-of-plane bending vibration (Abrishamkesh et al, 2015). The peak is stronger in RHB produced at the pyrolysis temperature of 500 ºC than that of 300 ºC. The increase in the aromatization of RHB by increasing the pyrolysis temperature, derived from infrared reflectance spectra, supports the reduction of molar ratios of H/C and O/C.

**Physical properties of rice husk biochar**

**Surface area**

As shown in Table 2, the surface areas measured for different RHBs have a wide range from 2.2–2.5 m²/g (Varela Milla et al, 2013; Wei et al, 2017) up to 179 m²/g (Pratiwi et al, 2016). There is a positive and significant relationship between surface area and pyrolysis temperature of RHBs. For example, Wei et al (2017) examined three pyrolysis temperatures of 300 ºC, 500 ºC and 750 ºC, and the surface area measured for the RHBs are 2.57, 18.41 and 53.08 m²/g, respectively. Some studies also reported surface areas as high as > 200 m²/g for RHBs produced at pyrolysis temperatures of 600 ºC–800 ºC (Rostamian et al, 2015). There is a positive relationship between surface area and porosity of RHBs as biochars’ porosity is affected by pyrolysis temperature. In addition, the ash content of biochars increases with the increase in their surface area (Enders et al, 2012; Rajapaksha et al, 2015).

**Physical structure**

The outer layer of rice husk is covered by rectangular skeleton of silica, which remains like a geometric shield after pyrolysis (Shackley et al, 2012). While the primary silica skeleton of RH is remained (Shackley et al, 2012), RHB generates a very porous structure (Abrishamkesh et al, 2015; Singh et al, 2018). The organic components including lignin and cellulose are combusted during the carbonization process and result in the production of geometrically arranged pores and channels (Bharadwaj et al, 2004). SEM images of RHB show well expanded microporous structures after pyrolysis process, while the porosity and surface area are also enhanced significantly (Abrishamkesh et al, 2015). The lower the pyrolysis temperature, the larger
the particle size of RHB and the more similarity of RHB structure to RH. However, the increase in pyrolysis temperature would result in the higher porosity of RHB. While a fast shrinkage occurs at pyrolysis temperatures between 200 °C and 400 °C, further increase of pyrolysis temperature (400 °C–800 °C) would only result in a minimal change in the particle size of RHBs (Bharadwaj et al, 2004).

**Rice husk biochar as soil amendment**

**Improvement of soil chemical property and nutrient balance**

Table 4 summarizes the effects of RHB application on the physicochemical properties of soil. In general, RHB application increases the contents of soil organic C (SOC), soil pH, CEC, available P, available K and total N. However, there are some inconsistencies in effects of RHB application due to the diverse properties of different soils and RHBs produced under different pyrolysis conditions.

**Effect of rice husk biochar on soil pH**

Soil pH is one of the key properties that control many biochemical and physical processes occurring within the soil. Soil pH controls the solubility and bioavailability of the chemicals in the soil, and therefore influences soil quality, crop productivity and environmental pollution (Weil and Brady, 2016). GHG emissions from soil, especially methane emission from paddy fields, are also strongly affected by soil pH (Wassmann et al, 1998). For example, in most of rice paddy fields with acidic soil pH, the bioavailability of micronutrients is higher than that of neutral-alkaline soils (soil pH > 7), which enhances the crop productivity (Lončarić et al, 2008). However, the solubility of heavy metals and some nutrients may pass the critical level of environmental toxicity in these soils. In general, biochars have the potential and capacity to substitute lime for improving the properties of acidic soils and therefore enhance plant growth (Wu et al, 2020).

Paddy soils are usually acidic and the application of alkaline RHB would increase soil pH (Singh et al, 2018; Singh Mavi et al, 2018; Ghorbani et al, 2019; Oladele, 2019). The observed increase in soil pH is usually between 1 to 2 units for the RHB application rates of 3 to 12 t/hm² (Oladele, 2019). An increase in pH of biochar-amended soils has also reported in other long-term incubation studies (Manolikaki and Diamadopoulos, 2017; Ghorbani and Amirahmadi, 2018a). This pH improvement has a significant positive effect on soil productivity and quality in acidic soils. However, Abrishamkesh et al (2015) applied RHB to a calcareous soil with alkaline pH and reported no changes in soil pH mainly due to the high content of calcium carbonate inducing a high buffering effect on soil pH.

**Effects of rice husk biochar on SOC and nutrient contents**

Biochars are generally enriched in nutrients and usually increase soil fertility when applied to soil (Feng et al, 2017). Masulili et al (2010) examined the effect of RHB on the properties of acid sulfate soil, and observed increases in SOC, CEC, soil P, K and Ca contents, and decreases in soil exchangeable Al and soluble Fe. However, Mg and Na contents are remained unaffected by RHB amendment. Singh et al

<table>
<thead>
<tr>
<th>Monitored time (Month)</th>
<th>RHB application rate</th>
<th>Soil tested</th>
<th>Effect on soil chemical property</th>
<th>Effect on soil physical property</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>10 t/hm²</td>
<td>Loamy sand</td>
<td>↑ pH, total OC, available P, CEC, exchangeable K and Ca; ↓ Exchangeable Al, soluble Fe</td>
<td>↑ Porosity, available soil water content; ↓ Bulk density</td>
<td>Masulili et al, 2010</td>
</tr>
<tr>
<td>2</td>
<td>0.4%, 0.8%, 1.6%, 2.4% and 3.3%</td>
<td>Clay loam</td>
<td>↑ Total OC, available K, CEC; ↓ Available P</td>
<td>↓ Bulk density</td>
<td>Abrishamkesh et al, 2015</td>
</tr>
<tr>
<td>3.5</td>
<td>10 t/hm²</td>
<td>–</td>
<td>↑ pH, total C, N, P</td>
<td>↑ Water holding capacity, soil moisture content</td>
<td>Singh et al, 2018</td>
</tr>
<tr>
<td>12</td>
<td>10, 20 and 40 t/hm³</td>
<td>Loamy sand and sandy clay loam</td>
<td>↑ EC, pH, total OC, microbial biomass C, dissolved organic C, available nutrients (NPK)</td>
<td>–</td>
<td>Singh Mavi et al, 2018</td>
</tr>
<tr>
<td>9</td>
<td>1% and 3%</td>
<td>Loamy sand and clay</td>
<td>↑ pH, total OC, CEC</td>
<td>↑ MWD, GMD, WSA; ↓ Bulk density, fractal dimension</td>
<td>Ghorbani et al, 2019</td>
</tr>
<tr>
<td>36</td>
<td>3, 6 and 12 t/hm²</td>
<td>Sandy clay loam</td>
<td>↑ EC, pH; CEC, OC, total N, C/N ratio</td>
<td>↑ WSA, water holding capacity; ↓ Bulk density</td>
<td>Oladele, 2019</td>
</tr>
</tbody>
</table>

OC, Organic carbon; CEC, Cation exchangeable capacity; K, Potassium; Ca, Calcium; Al, Aluminum; Fe, Iron; P, Phosphorus; C, Carbon; N, Nitrogen; EC, Electrical conductivity; C/N, Carbon/nitrogen; MWD, Mean weight diameter; GMD, Geometric mean diameter; WSA, Water stable aggregate.
(2018) reported significant increases in water holding capacity, total C, N and P contents, as well as soil moisture content following RHB application. Ghorbani and Amirahmadi (2018a) also reported that RHB significantly increases the N content and bioavailable K in the amended soils. However, the reported results are contradictory for P. Abrishamkesh et al (2015) reported that P content of the soils amended with 0.4%, 0.8% and 3.3% RHB is significantly lower than that of the unamended control soil. Ghorbani and Amirahmadi (2018a) also observed a decrease in P content by RHB application. Significant improvement in soil bioavailable P content as a result of charcoal application has long been reported in sandy and loamy soils (Tryon, 1948). Oladele (2019) assessed the effect of RHB on Alfisol productivity using the soil quality index. RHB application improves soil quality in a three-year experiment. Although RHB application significantly enhances all of soil quality indicators after one year, its effects on EC, SOC and total N, are declined after three years, while the improvement in soil pH and CEC stays consistent by the end of experiment (Oladele, 2019).

**Effect of rice husk biochar on soil CEC**
Masulili et al (2010), Abrishamkesh et al (2015), Ghorbani et al (2019) and Oladele (2019) have reported an increase in CEC of the soils amended with RHB. However, the rate of this increase is related to the soil type and RHB application rate. For example, Ghorbani et al (2019) reported 20% and 30% increases of CEC in a loamy sand soil for RHB application rates of 1% and 3%, respectively, while reported 9% and 19% increases of CEC in a clayey soil for RHB application rates of 1% and 3%, respectively. Oxidation of RHB and the development of surface negative charges over time may increase its CEC (Mehmood et al, 2018). Any increase in soil CEC provides a potential for nutrient balance, reduction of pollutant leaching and mitigation of GHG emissions.

**Improvement of soil physical properties**
After water storage, soil bulk density (BD) is the most important physical indicator of soil quality, which has been proposed in > 50% of ‘soil quality concepts’ worldwide (Bünemann et al, 2018). Masulili et al (2010) indicated that soil BD decreases from 1.24 to 1.17 g/cm³ after application of 10 t/hm² RHB. Abrishamkesh et al (2015) analyzed soil properties after a 70-day period of lentil growth in an alkaline clay loam soil. Application of RHB reduces soil BD by 14% in average (from 1.4 to 1.2 g/cm³). Ghorbani et al (2019) reported 8% and 22% decreases in BD by application of 3% RHB to clayey soil and loamy sand soil, respectively. Ghorbani and Amirahmadi (2018b) also reported a 21% reduction in BD in a loamy soil under corn plantation after treatment by 4% RHB. In a field trial, Oladele (2019) observed a decrease of 18% in BD by application of 12 t/hm² RHB.

Soil BD improvement is the result of increase in soil porosity which is induced by aggregation. For example, by RHB application, the soil porosity is increased from 40% to more than 50% (Masulili et al, 2010); while from 50% to 54% in a clayey soil and from 39% to 52% in a loamy sand soil (Ghorbani et al, 2019). Ghorbani and Amirahmadi (2018b) observed an increase in soil porosity from 50% to 60% mainly due to the increase of soil macropores (> 10 μm).

While the RHB application to a loamy sand soil has no significant effect on soil aggregation, all of soil aggregate stability indices are significantly improved in RHB-treated clayey soil (Ghorbani et al, 2019). The clay particles apparently play the main role in formation of soil aggregates through binding organic molecules by bi- and three-valent cations (Ca²⁺, Fe³⁺ and Al³⁺) (Jien and Wang, 2013). Percentage of water stable aggregate is increased by 10%, 18% and 23% at soil surface (0–10 cm depth) and by 16%, 20% and 26% at soil depth of 10–20 cm for the RHB application rates of 3, 6 and 12 t/hm², respectively (Oladele, 2019). This aggregation is stable event at the third year of evaluation. Juriga et al (2018) also indicated an improvement in soil aggregation by formation of biochar-soil mineral complexes. The capacity of soil to store and fix organic C, as one of the soil environmental functions, is strongly affected by aggregate stability (Wang et al, 2018). In other words, the formation of stable aggregates will protect soil organic C from microbial decomposition (Jien and Wang, 2013). Soil aggregate stability also enhances soil resistance against erosive agents (Chaplot and Cooper, 2015), and therefore has an important role in reducing soil erosion. Available soil water is another important soil physical property that can be improved by RHB application (Masulili et al, 2010; Ghorbani and Amirahmadi, 2018b; Singh et al, 2018). In addition, soil penetration resistance, as a mechanical soil property affecting plant root growth, will decrease by application of RHB to soil due to the formation of stable soil aggregates, which can consequently increase soil porosity and decrease soil BD (Masulili et al, 2010).
Rice husk biochar as plant growth promoter

Direct application of rice straw to soil has been examined for a long time (Ponamperuma, 1982). This practice has significantly increased soil productivity and crop yield mainly by improvement of soil physico-chemical properties including SOC, total N, available P and K, soil BD and pH (Williams et al, 1972; Mandal et al, 2004; Kumari et al, 2018; Zhao et al, 2019). However, the incorporation of crop residues into the soil usually results in rapid decomposition of organic matters with the sequences of CH4 and CO2 emissions (Haefele et al, 2011) and nutrient release into the soil profile (Naeem et al, 2017). In addition, direct application of RH may temporarily reduce N bioavailability in soil and cause a need for N fertilization (Scheller and Joergensen, 2008; Reichel et al, 2019). Therefore, the use of crop residues in the form of biochar is an alternative approach (Naeem et al, 2017).

Generally, an increase in soil productivity has been reported by application of different biochars (Yamato et al, 2006; Jeffery et al, 2011). Biochar application can improve crop yield by acting as a direct nutrient source and/or enhancing nutrient bioavailability through improvement of soil pH, CEC, surface interactions, soil porosity and soil-water interactions (Laungani et al, 2016; Yuan et al, 2019). There are also several reports of crop yield reduction especially in short timeframes after biochar application (Reibe et al, 2015), possibly because of nutritional imbalance and/or toxicity (Ippolito et al, 2012). Effects of RHB application on plant growth and productivity are summarized in Table 5. In general, crop yield is improved by RHB amendment. However, the magnitude of this effect is related to soil and crop type, as well as RHB characteristics. Haefele et al (2011) reported that the effect of RHB on soil fertility and rice grain yield is site-specific. Abishamkesh et al (2015, 2017) also examined the effects of two RHBs at different rates on the growth of lentil and wheat in a pot experiment using a calcareous alkaline soil. Wheat was planted after lentil harvest in the same pots without reuse of RHB. While the root growth in both crops was significantly increased at the higher RHB application rates, the above-ground biomass was improved just in wheat. Therefore, RHB is a very good root growth promoter due to its effect on soil porosity. The root extension following RHB application may enhance above-ground biomass and yield when the crop is faced with an environmental stress such as drought stress.

Varela Milla et al (2013) compared the effect of different rates of RHB with wood biochar (WB) on water spinach growth in a field experiment. RHB application at the rate of 1.0 kg/m3 results in the highest leaf number and the largest stem size.

Table 5. Summary of selected data on rice husk biochar (RHB) application as a plant growth promoter.

<table>
<thead>
<tr>
<th>Monitored time (Month)</th>
<th>Amendment rate</th>
<th>Application condition</th>
<th>Soil texture</th>
<th>Crop type</th>
<th>Effect on plant growth and biomass</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>10 t/hm²</td>
<td>In comparison with rice straw, rice husk and rice husk ash amendments</td>
<td>Loamy sand</td>
<td>Rice</td>
<td>† Total biomass</td>
<td>Masulili et al, 2010</td>
</tr>
<tr>
<td>Two growth seasons</td>
<td>4–5 kg/m²</td>
<td>Field trial at three locations, in comparison with rice husk, RHB and rice husk- and RHB-fertilizer</td>
<td>Three sites</td>
<td>Rice</td>
<td>† Grain yield in one site</td>
<td>Haefele et al, 2011</td>
</tr>
<tr>
<td>12</td>
<td>0.5, 1.0, 2.0, 3.0 and 4.0 kg/m³</td>
<td>Compared with wood biochar</td>
<td>Clay</td>
<td>Water spinach</td>
<td>† Plant weight, stem size and leaf length</td>
<td>Varela Milla et al, 2013, Zhao et al, 2014</td>
</tr>
<tr>
<td>24</td>
<td>4.5 and 9.0 t/hm²</td>
<td>Compared with 3.75 t/hm² straw for each crop season</td>
<td>Clay</td>
<td>Rice-wheat</td>
<td>† Grain yield</td>
<td>Varela Milla et al, 2013, Zhao et al, 2014</td>
</tr>
<tr>
<td>2</td>
<td>0.4%, 0.8%, 1.6%, 2.4% and 3.3%</td>
<td>Biochar produced under different pyrolysis temperatures (300 °C and 450 °C)</td>
<td>Clay loam</td>
<td>Lentil</td>
<td>† Root biomass</td>
<td>Abrishamkesh et al, 2015</td>
</tr>
<tr>
<td>2</td>
<td>0.4%, 0.8%, 1.6%, 2.4% and 3.3%</td>
<td>Biochar produced under different pyrolysis temperatures (300 °C and 450 °C)</td>
<td>Clay loam</td>
<td>Lentil-wheat</td>
<td>† Shoot and root biomass of wheat</td>
<td>Abrishamkesh et al, 2017</td>
</tr>
<tr>
<td>3</td>
<td>2% and 4%</td>
<td>Biochar produced under 500 °C</td>
<td>Loam</td>
<td>Maize</td>
<td>† Shoot biomass, stem size</td>
<td>Ghorbani and Amirrahmadi, 2018b, Singh Mavi et al, 2018</td>
</tr>
<tr>
<td>12</td>
<td>10, 20 and 40 t/hm²</td>
<td>Pure and combined with N fertilizer (60, 90, 120 and 150 kg/hm²)</td>
<td>Loamy sand and sandy clay loam</td>
<td>Wheat-maize</td>
<td>† Total biomass of maize</td>
<td>Huang et al, 2019</td>
</tr>
<tr>
<td>36</td>
<td>20 t/hm²</td>
<td>Pure and combined with N fertilizer (90 and 150 kg/hm²)</td>
<td>Clay</td>
<td>Rice</td>
<td>† Grain yield, harvest index, number of panicles, total N and N use efficiency</td>
<td>Huang et al, 2019</td>
</tr>
</tbody>
</table>
However, the highest mean leaf width is obtained by application of 2.0 kg/m³ RHB. The leaf lengths in all treatment rates of RHB are significantly higher than those in the unamended control. WB application is less effective on the growth of spinach in comparison to RHB, which can be due to the differences in Si, K, ash content and surface area.

Singh Mavi et al (2018) assessed the effect of RHB on crop growth of wheat-maize sequence in two soils with contrasting texture. Increases in oxidizable organic carbon and bioavailable nutrients in the RHB-treated soils are observed at the end of cropping season, which is in consistence with the increase of maize biomass. This is also confirmed by the positive correlation between maize biomass and soil properties. However, the short effect of RHB on wheat biomass is not significant, similar to the reports on some other biochars (Reibe et al, 2015).

Singh et al (2018) applied 10 t/hm² RHB in a field trail and observed significantly higher rice growth variables including panicle length, tiller number, grain and straw yields. However, the observed effect is more pronounced for the grain yield. Huang et al (2019) studied the effect of RHB on rice yield by continuous application of biochar for six seasons and showed that grain yield tends to decrease with RHB application in the first three seasons, whereas it is significantly increased in the last three seasons of the experiment. This suggests that the positive effects of RHB application on rice yield would depend on the duration and frequency of biochar application. The trend of decrease in rice grain yield with biochar application, during the first three seasons, is mainly attributed to the decrease of rice grain weight.

Ghorbani and Amirahmadi (2018b) evaluated the effect of RHB (2% and 4%) on corn growth and found that plant height in the ninth week of growth period is 85 cm in the soil containing 4% RHB, which is significantly higher than plant height of 75 cm in the unamended control soil. Also, the shoot dry weights are 154.7 and 156.8 g in the RHB rates of 2% and 4%, respectively, which are both significantly higher than that of the control (148.8 g).

Previous investigations have shown that biochar application together with chemical fertilizers would increase crop production, especially in the soils with low fertility, either by acting as a direct source of nutrients or by improving nutrient bioavailability (Lashari et al, 2015). For example, combined application of biochar and N fertilizers can increase N uptake and productivity of crops (Jeffery et al, 2011; Mehmood et al, 2018). Nitrogen immobilization has also been observed by application of C-rich biochars and reduction in bioavailability of essential nutrients through their sorption to biochars’ surface functional groups (de Sousa et al, 2014).

Environmental benefits of RHB application

Biochar can reduce the bioavailability of organic pollutants and heavy metals in contaminated soils (Zhao et al, 2016; Ajayi and Horn, 2017), and can increase plant growth via increasing antioxidant enzymes and improving soil biological and physicochemical properties (Alobwede et al, 2019). Biochar has also been extensively used to remove different types of pollutants (e.g. heavy metals, dye, pharmaceuticals, pesticides/herbicides and phenols) from wastewaters (Dai et al, 2019; Wang et al, 2019). The high efficiency of biochar to adsorb heavy metals and organic pollutants is related to its high porosity, high number of functional groups and high pH (Zhang et al, 2013; Yuan et al, 2019). In addition, the mineral components of biochar including phosphates and carbonates can precipitate with heavy metals and reduce their bioavailability (Cao et al, 2009; Kumar et al, 2018). Moreover, Park et al (2011) and Chen et al (2018) reported that biochar application can reduce the pollution of organic pollutants, decrease the bioavailability of heavy metals and improve the condition of plant and soil in terms of contamination due to its high surface area and CEC, as well as the presence of carboxylic, phenolic, hydroxylic and other oxygen containing functional groups. Summary of the sorption effect of RHB is presented in Table 6.

Immobilization of soil heavy metals

Chen et al (2018) indicated that biochar application to soils significantly reduces the accumulation of cadmium (Cd), plumbum (Pb), copper (Cu) and zinc (Zn) in plant tissues. Alaboudi et al (2019) also observed reductions of Pb and Cd toxicities in soil after biochar application. Jiang T Y et al (2012) applied rice straw biochar (RSB) to three acidic soils and observed a significant increase in Pb adsorption by the treated soils. The main adsorption mechanism is the formation of surface complexes between biochar functional groups and Pb. The results also show a significant potential of RSB for Cu, Pb and Cd immobilization in acidic soils.

Xu et al (2013) compared the effect of RHB and
dairy manure biochar (DMB) in removing Pb, Cu, Zn, and Cd from aqueous solutions. While RHB is less effective than DMB in removing heavy metals, RHB displays higher competition for metal removal than DMB in the presence of all metals. They concluded that DMB is rich in carbonate and/or phosphate minerals which precipitate the metals, but the removal of metals by RHB is only the result of complexation with surface functional groups. Zheng et al (2013) tested the effects of RHB, RSB and rice bran biochar amendment on the uptake and accumulation of arsenic (As), Cd, Pb and Zn by wheat. Addition of all biochars to soil significantly reduces the bioavailable concentrations of Cd, Pb and Zn, which results in a significant decrease in their concentrations in plant shoots. However, the results are invers in the case of As. The efficiencies of the biochars in immobilization of heavy metals in soil and reduction of their accumulation in plant tissues are increased by decreasing particle size of biochar.

The adsorption capacity of RHB for As$^{3+}$ and As$^{5+}$ in comparison with a commercially produced empty fruit bunch biochar (EFBB) was studied by Samsuri et al (2013). The maximum adsorption capacities of RHB and EFBB are 18.9 and 19.3 mg/g, respectively for As$^{3+}$, and 5.5 and 7.1 mg/g, respectively for As$^{5+}$. Activation with Fe$^{3+}$ increases the adsorption capacity of the biochars by around 60% for As$^{3+}$ and 150% for As$^{5+}$. Samsuri et al (2013) concluded that not only surface area, but also the characteristics such as the abundance of functional groups, zeta potentials and the polarity of the biochars are effective in their adsorption capacities for heavy metals. Prapagdee et al (2016) observed the highest efficiency of Cd removal for RHB produced by pyrolysis at 300 ºC and activated by KOH, compared with RHBs produced at 400 ºC and 500 ºC.

Amirahmadi et al (2020) investigated the effect of RHB application to an artificially polluted soil on Cd bioavailability and encouraging oak seedlings. They reported significant increases in seedling height, diameter and biomass of oak seedlings, and a decrease in Cd concentration in plant tissues. Plant tolerance index for the highest Cd rate of 50 mg/kg is increased by 40.9%, 56.0% and 60.6% in RHB application rates of 1%, 3% and 5%, respectively.

Sohi et al (2010) proposed three mechanisms for the adsorption of heavy metals by biochar, including electrostatic interactions, ion exchange and sorptive interaction. Electrostatic interactions and ionic exchange between metal cations and biochars have long been proposed as important mechanisms for adsorption of heavy metals (Park et al, 2011; Rodríguez-Vila et al, 2015). Delocalized π electrons of carbon can induce the sorptive interaction between biochars and heavy metals (Sohi et al, 2010). Another mechanism for heavy metal adsorption by biochar is related to the conversion of the basic cations (i.e. K, Ca, Mg and Na), present in biochar ash, into oxides, hydroxides, phosphates and carbonates (Ramzani et al, 2017). Therefore, the formation of precipitates such as phosphates and carbonates can be considered as the fourth mechanism. Ahmad et al (2014) proposed five mechanisms including complexation, electrostatic interaction, ion exchange, physical adsorption and precipitation, for remediation of polluted soils by biochar. Li et al (2017) proposed six mechanisms for immobilization of heavy metals by biochars, adding reduction to the mentioned list. However, they mentioned that the dominant mechanism depends on the heavy metal and biochar type. Following biochar application to soil, further electrostatic interactions occur between metal cations and the activated functional groups of the soil caused by an increase in

<table>
<thead>
<tr>
<th>Experimental duration</th>
<th>Amendment rate</th>
<th>Pyrolysis temperature (ºC)</th>
<th>Soil texture</th>
<th>Effect of toxic element concentration in soil</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>1 month</td>
<td>3% and 5%</td>
<td>500</td>
<td>Clay</td>
<td>Soluble Cu$^{2+}$, Pb$^{2+}$ and Cd$^{2+}$</td>
<td>Jiang J et al, 2012</td>
</tr>
<tr>
<td>1 month</td>
<td>3% and 5%</td>
<td>550</td>
<td>Sandy loam</td>
<td>Oxidative stress as well as Cd content in soil and plant;</td>
<td>Abbas et al, 2018</td>
</tr>
<tr>
<td>1 d</td>
<td>0.2 g/kg</td>
<td>Not clear</td>
<td>–</td>
<td>Adsorption of As$^{3+}$, As$^{5+}$ from aqueous solution</td>
<td>Samsuri et al, 2013</td>
</tr>
<tr>
<td>1 d</td>
<td>0.1 g/kg</td>
<td>300, 400 and 500</td>
<td>–</td>
<td>Cd removal from aqueous solution</td>
<td>Prapagdee et al, 2016</td>
</tr>
<tr>
<td>8 months</td>
<td>1%, 3% and 5%</td>
<td>600</td>
<td>Leamy</td>
<td>Soil Cd bioavailability, Cd concentration in plant tissues;</td>
<td>Amirahmadi et al, 2020</td>
</tr>
<tr>
<td>6 months</td>
<td>0.01%, 0.02%, 0.05%, 0.1%, and 0.5%</td>
<td>Not clear</td>
<td>Sandy</td>
<td>Tolerance index and removal efficiency</td>
<td>Yang and Sheng, 2003</td>
</tr>
<tr>
<td>1 month</td>
<td>5%</td>
<td>550</td>
<td>Sandy loam</td>
<td>Toxicity of herbicide fenoxaprop-ethyl</td>
<td>Jing et al, 2018</td>
</tr>
</tbody>
</table>
soil pH (Cao et al., 2009; Jiang et al., 2012; Uchimiya et al., 2012). Accordingly, in addition to direct interactions with biochar, changes in soil properties caused by biochar application can also indirectly increase the capacity of soil for reducing the bioavailability of heavy metals. The immobilization mechanisms of heavy metals and organic pollutants by biochar are illustrated in Fig. 4.

**Reduction in bioavailability and toxicity of organic pollutants**

Biochars have extensively tested for removal of organic pollutants (herbicides, pesticides, dye and pharmaceuticals) from soil and water (Cao et al., 2016; Zhong et al., 2018). Khorram et al. (2016) reviewed the functions of different biochars regarding sorption, release and leaching of pesticides in soil. They concluded that the pesticides’ bioavailability to soil organisms decreased by biochar application due to the enhancement of soil capacity to adsorb pesticides and improvement of soil pH and CEC. The results also confirmed a restriction in the movement of pesticides in soil and their leaching from soil profile.

Persistence and bioavailability of atrazine and trifluralin were assessed in two contrasting soils treated by wheat straw biochar (Nag et al., 2011). The biochar application to soil decreases the efficiency of both herbicides and increases their precedence in the soil, but trifluralin is affected much less than atrazine. Treating soil with biochar can increase the input of herbicides. Zhelzezova et al. (2017) evaluated the effect of a wood biochar on the adsorption and degradation of glyphosate and diuron in two soils with contrasting texture. Biochar application increases diuron adsorption in both soils but decreases the adsorption of glyphosate in the sandy soil. The degradation rates of the herbicides inconsistently changed by the biochar application. In the context of herbicides, complicated results have been reported. Biochar application to soil can reduce the pollution risk (Khorram et al., 2016) as well as the efficiency of herbicides especially when applied biochars have high surface area (Nag et al., 2011; Graber et al., 2012).

Jing et al. (2018) studied the effects of RHB on the fate and toxicity of fenoxaprop-ethyl in soil. The elimination of the herbicide in soil is very faster in RHB-treated soil than in the unamended control soil. RHB application also reduces the herbicide risk in the soil profile, which is verified by earthworms’ evaluation. Yang and Sheng (2003) reported that biochars produced from wheat and rice residues are 400–2 500 times more effective than soil in adsorption of the herbicide diuron. The mechanism of organic pollutants’ adsorption onto biochar depends on the properties of both biochars and organic pollutants. In general, the main mechanisms include electrostatic attraction, hydrogen bonds, pore-filling and hydrophobic effect (Tan et al., 2015).

**Reduction of nutrient leaching**

In recent years, several strategies have been used to reduce nitrate leaching from the soils and prevent groundwater contamination. The evaluated techniques include the use of drip irrigation (Farneselli et al., 2015), different fertilization frequencies (Kumar et al., 2016), application of slow-release fertilizers (Zareabyaneh and Bayatvarkeshi, 2015) and zeolite addition to soil.
(Nakhli et al., 2017). Reduction of N leaching from soil has been observed by application of biochars produced from bamboo (Ding et al., 2010), pecan shells (Chaplot and Cooper, 2015) and forest residues (Manolikaki and Diamadopoulos, 2017). A meta-analysis performed by Borchard et al. (2019) on 608 observations, indicated an overall 13% reduction in nitrate leaching from soil by biochar application. However, in some cases, biochar application to soil results in the increase of nitrate leaching depending on soil types (Liu Z Q et al., 2017) and the biochar application methods (Li et al., 2018). Yao et al (2012) reported that the sorption capacity and therefore the efficiency of biochars to reduce nutrient leaching depend on nutrient and biochar types. On the other hand, due to high N-retention capacity of biochars, Schmidt et al (2015) has demonstrated the ability of biochar to act as a slow release N resource for providing N to plants during a long period of growth. Therefore, retention of nitrate and other forms of nitrogen by biochars is beneficial for agricultural production and environmental protection, as well as the reduction in the management costs of farming systems by reducing the demand for N fertilizers (Borchard et al., 2019).

A summary of the effect of RHB application on nutrient leaching and retention in soil is presented in Table 8. Pratiwi et al (2016) evaluated the sorption capacity of RHB and its effect on leaching of N and P. The investigated RHB had the sorption capacity of 4.7 and 2.1 mg/g N for ammonium and nitrate ($C_0 = 200$ mg/L), respectively, and almost no phosphate adsorption capacity. RHB addition to a loamy sand soil reduces nitrate and ammonium leaching by 23% and 11%, respectively, and increases phosphate leaching by 72%. Ghorbani et al (2016) investigated the effect of RHB on nitrate leaching during a five-month experiment. RHB applied to soil in two particle sizes (original and finer than 1 mm) and in two application rates (1% and 3%) in the presence and absence of 1% compost, as N source. The highest and lowest rates of nitrate leaching are observed in the compost-treated no-RHB soil and RHB-treated no-compost soil, respectively. RHB addition to soil at both rates and particle sizes significantly reduces nitrate leaching compared to the control. The particle size of RHB showed no significant effect on nitrate leaching. Ghorbani et al (2019) also evaluated the effect of RHB on nitrate leaching in two contrasting soil types (a loamy sand and a clayey soil) for a nine-month period of wetting (field capacity + 20% water holding capacity) and drying (permanent wilting point) cycles. RHB application significantly decreases nitrate leaching in both soils. Nitrate leaching from the clayey soil is generally less than the loamy sand soil, but the difference is higher in the first couple of months of the experiment. RHB application can increase nitrogen adsorption and retention in soil by increasing the CEC and water holding capacity of soil and enhancing the microbial immobilization of N (Liu S N et al., 2017).

Bu et al (2017) studied the effect of RHB on N and P retention in soil and their leaching in a riparian soil sample by a column test in a 24-week incubation. Leaching of nitrate, ammonium and dissolved organic N is significantly lower, and leaching of phosphate is significantly higher in the soils treated by RHB compared with the unamended control soil. Total N, available P and microbial biomass N are significantly higher in the soil treated by RHB. Bu et al (2019) applied two RHBs produced at pyrolysis temperatures of 450 °C and 650 °C, to a calcareous soil. Application of both RHBs significantly reduces the total cumulative amounts of nitrate and ammonium leached from the soil profile, while significantly increases the total cumulative amounts of phosphate and potassium ions leached from the soil during 28 weeks of measurement. On the other hand, the soils treated with RHB show a significantly higher bioavailable N, microbial biomass N, available P and K than the non-treated soil. RHB produced at 650 °C is more effective in reduction of N leaching, while RHB produced at 450 °C has higher efficiency in enhancing of the microbial biomass and activity in soil.

**Reduction of GHG emissions**

Increase in GHG emission is the driver of global warming. GHG emissions from agricultural soils have a considerable role in the total global emissions. In 2005, agricultural industry accounts for around 10%–12%, 60% and 50% of global CO$_2$, N$_2$O and CH$_4$ emissions, respectively (Smith et al., 2007). The annual net CO$_2$-equivalent GHG emission from soil is estimated to be ≥ 350 Pg compared to 33.4 Pg CO$_2$ emission from fossil fuel combustion and cement industry (Oertel et al., 2016). Monaco et al (2012) estimated that the contribution of agriculture to total NH$_3$ emissions to be more than 80%. About 30%–40% of this emission is due to manure application to soil (Hutchings et al, 2009), which is greatly affected by manure and soil properties, environmental conditions and application methods (Taghizadeh-Toosi et al,
Soil amendment by biochar can reduce N$_2$O emission and increases C sequestration in the investigated soil. Pratiwi and Shinogi (2016) evaluated the effect of two application rates of RHB on CO$_2$ and CH$_4$ production in two soil types. Cui et al. (2017) found that RHB application significantly enhances rates of 2% and 4%, respectively. Cui et al (2017) noted that RHB application significantly reduces N$_2$O emissions by an average of 38% in the first year of application (Borchard et al, 2019).

Haefele et al (2008) compared the effect of RH and RHB on CO$_2$ and CH$_4$ production in two soil types. While RH application significantly increases the emission of both gases, RHB addition does not significantly change CO$_2$ and CH$_4$ emissions. Pratiwi and Shinogi (2016) evaluated the effect of two application rates of RHB on CH$_4$ emission from a paddy soil during a 100-day measurement. They observed decreases of 45.2% and CH$_4$ content in soil (Cui et al, 2017). Cui et al (2017) noted that RHB application significantly reduces N$_2$O emissions by an average of 38% in the first year of application (Borchard et al, 2019).

Haefele et al (2008) compared the effect of RH and RHB on CO$_2$ and CH$_4$ production in two soil types. While RH application significantly increases the emission of both gases, RHB addition does not significantly change CO$_2$ and CH$_4$ emissions. Pratiwi and Shinogi (2016) evaluated the effect of two application rates of RHB on CH$_4$ emission from a paddy soil during a 100-day measurement. They observed decreases of 45.2% and 54.9% in total CH$_4$ emission for RHB application rates of 2% and 4%, respectively. Cui et al (2017) found that RHB application significantly enhances total C and N contents, decreases N$_2$O and CH$_4$ emissions, and increases C sequestration in the investigated soil. Soil amendment by biochar can reduce N$_2$O emission in rice paddy fields by 40% (Borchard et al, 2019).

**CONCLUSIONS**

This critical review indicated that application of RHB, as a soil amendment, can not only enhance soil quality and productivity, but also has the potential to significantly mitigate climate change. In addition, it has benefits for remediation of polluted soils and wastewater. RHB usually maintains the primary skeleton of RH but with a very porous structure and therefore a high surface area. Production of RHB in pyrolysis temperatures higher than 500 ºC can result in an extremely carbonized biochar with a high aromaticity. While the silica functional groups (Si–OH, Si–O–Si, and Si–H) are dominant in RHBs produced at low pyrolysis temperatures, aliphatic and aromatic C–H functional groups can significantly increase with increasing pyrolysis temperatures. RHB usually has an alkaline pH ranging from 7.1 to 10.8, which depends on the pyrolysis temperature. This is a very important characteristic for a soil amendment to be used in acidic soils like rice paddy fields.

Biochar addition to soil can increase soil organic C, CEC, available K and total N. Soil bulk density, as the most important physical indicator of soil quality, is improved by RHB application. Indeed, RHB induces soil aggregation which results in increasing soil porosity and water holding capacity, while decreasing soil bulk density and soil penetration resistance. As a result of soil improvement and due to the presence of nutrients in RHB, plant growth and yield are enhanced when the soil is amended with RHB, but the magnitude of the effect depends on the crop types, soil properties and the characteristics and application rates of RHB.

RHB application can reduce the bioavailability of herbicides and heavy metals in contaminated soils, reduce GHG emissions and decrease nutrient leaching from soil. The dominant mechanism for immobilization of pollutants in the soil by biochar depends on the pollutant type, soil type, RHB type and application rate. Biochar application, the pollution risk of herbicides is reduced, which is beneficial for environment, but the efficiency of herbicides is also reduced, which is an economical concern. RHB has an important adsorption capacity for ammonium and nitrate and therefore can reduce their leaching when applied to soil.

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