Complementing compost with biochar for agriculture, soil remediation and climate mitigation

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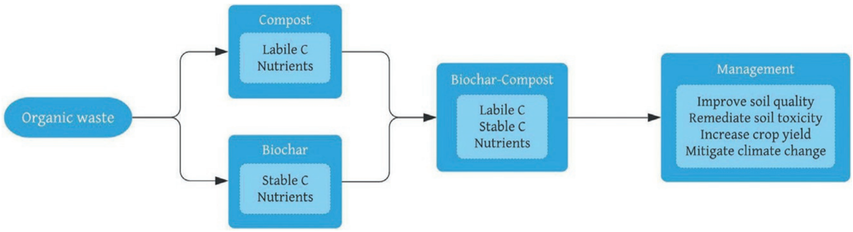
Contents

Abstract

We are racing to manage a phenomenally increasing volume of organic wastes from urban, industrial and agricultural entities. Composting is one of the preferred ways to convert biodegradable wastes into nutrient-rich soil conditioners. The age-old technique of composting process is being improved with innovative scientific means. Biochar, a widely studied soil amendment, is a carbonaceous material that can hold nutrients from endogenic/exogenic sources. Biochar-compost, a biochar-complemented compost, may provide a wide range of benefits expected from both materials. Compost and biochar can improve physicochemical and microbiological attributes of soils by supplying labile and stable carbons, and nutrients. Compost may also supply beneficial microbes. This means biochar-compost is a synergic soil amendment that can improve soil quality, increase crop production, and remediate contaminated soils. Having stable carbon, large reactive surface with nutrient loads, biochar can interact widely with organic biomass and modify physicochemical and-microbial states during a composting process while making biochar-compost. Production and application methods of biochar, compost and biochar-compost are covered for agricultural and contaminated soils. Metal and organic contaminations are also discussed. A case study on making and field-testing a mineral-enhanced biochar and a biochar-compost to improve rice yield, is presented at the end.

Graphic abstract



Abbreviations

AC	activated carbon
CEC	cation exchange capacity
DOM	dissolved organic matter
GHG	greenhouse gas emission
MSW	municipal solid waste
OC	organic carbon
OM	organic matter
PAH	polyaromatic hydrocarbon
PBDE	polybrominated diphenyl ether
PFAS	polyfluoroalkyl substance
POM	particulate organic matter
POP	persistent organic pollutant
PPCP	pharmaceuticals and personal care product
PTE	potentially toxic element
SOC	soil organic carbon
VOC	volatile organic compounds

1. Introduction

Healthy soils are essential for growing healthy and nutritious crops (Lal, 2020). Soil quality is "the capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation" (Karlen et al., 1997). Soil organic carbon (SOC) is a key component of agricultural sustainability and soil quality (Doran and Zeiss, 2000; Fonte et al., 2009). It can assist several soil quality parameters e.g., soil structure (Bailey and Lazarovits, 2003), aggregate stability (Loveland and Webb, 2003), water holding capacity, water drainage (Davey, 1997), plant available water capacity, and plant available essential nutrient (Davey, 1997; Lal, 2020), soil air, and microbial ecology (Davey, 1997). Thus, increasing SOC can increase the soil's capacity to retain water and nutrients, which may reduce the use of chemical fertilizers and irrigation water (Lal, 2020).

A severe loss of SOC may degrade soil functionality (Lal, 2020), thereby requiring restoration and/or increasing SOC to reverse the situation (Banwart et al., 2015). Incorporation of organic residues, for example, manure (Bol et al., 2000; Lal, 2020), cover crops (Lal, 2020), plant residues (Steiner et al., 2007; Tiessen et al., 1994), mulch/compost (Steiner et al., 2007), biochar (Steiner et al., 2007) can increase SOC.

Many agricultural soils are poor in SOC due to various natural conditions (Tiessen et al., 1994) and human activities like intensive agricultural soil management (Tilman et al., 2002). Moreover, organic inputs in soils are commonly labile, and mineralize over time (Steiner et al., 2007) that may be short under high temperature and humid conditions (Tiessen et al., 1994); a large portion (up to 87%) can be lost by a period of five months (Bol et al., 2000). An alternative to labile organic input is a carbonized organic residue (Bolan et al., 2012; Steiner et al., 2007) containing recalcitrant carbon at a high proportion (Krull et al., 2009). Biochar is a carbonized organic material derived by pyrolyzing organic residue or wastes at temperature $<700\,^{\circ}C$ under partial or total absence of O_2 to improve soil carbon storage and productivity (Lehmann and Joseph, 2009). The physicochemical properties of biochar, such as pore diameter, size distribution, total surface area and nutrient contents, are functions of pyrolysis conditions and feedstock (Novak et al., 2009). Biochar particle size may range between <0.5 and $>50\,mm$ (International Biochar Initiative, 2015). It may contain nano ($<0.9\,nm$), micro ($<2\,nm$) and macro pores ($>50\,nm$), and a surface area up to $1360\,m^2\,g^{-1}$ (Downie et al., 2009).

Studies reported that biochar acts as a soil conditioner enhancing plant growth through various mechanisms such as (a) increasing the cation exchange capacity of the soil, (b) enhancing the water holding capacity, (c) neutralizing the acidity of the soil, (d) creating better conditions for the growth of microorganisms (Ahmad et al., 2014; Edenborn et al., 2015; Park et al., 2011; Schmidt et al., 2021). Moreover, biochar is a very stable compound against both chemical and microbial activities, remaining in soils longer than any other form of organic carbon (Kimetu and Lehmann, 2010; Kuzyakov et al., 2014; Wang et al., 2016a). However, one disadvantage of biochar is low content of labile carbon and low availability of some of its adsorbed nutrients, particularly nitrogen. Therefore, biochar alone may not be a complete amendment for degraded soils characterized by low SOC and poor fertility (Córdova et al., 2011; Mulligan et al., 2001).

Composting involves a low-tech process that increases value of organic wastes through conversion into a valuable soil conditioner (Moral et al., 2009). As the composting process breaks down the rigid organic structures, carbon and nutrients become available for plants and microbes. It also considerably reduces odor in organic wastes, making them unattractive to insects (Alexander, 2001). Compost has been reported as having great potential for improving soil fertility through promoting changes in physicochemical and microbiological properties of soils (Bonilla et al., 2012). Therefore, interest is growing in using composts to increase SOC. Some studies show that, having a high decomposition rate (Bolan et al., 2012; Qayyum et al., 2012), positive effects of compost stay for a short period after application in soil. Therefore, compost requires one or more applications annually, depending on crop growing season. Compost may mobilize excessive amount of some elements, particularly metal(loid)s, associated with itself or soil by supplying dissolved organic carbon (DOC) that can bind elements.

Biochar-compost or biochar-complemented compost, a synergic (Fischer and Glaser, 2012) amendment developed by combining biochars and compost, is being explored these days (Agegnehu et al., 2017). This combination can ensure stable and labile carbon (Jindo et al., 2012b; Khan et al., 2014) and an increased amount of nutrients needed for degraded soils (Schulz et al., 2013). Biochar-complemented compost has the potential for sustainable waste and soil management. But there is a lack of integrated knowledge on biochar-compost (Agegnehu et al., 2017). This review aims to present a text that integrates knowledge on production of biochar, compost and biochar-compost and their application in un/contaminated soils for managing agricultural soil quality, remediation of contaminated soils and address climate change.

2. Sources of organic wastes

We are racing to manage the phenomenally increasing volume of organic wastes. Organic waste biomass is biodegradable, and can be used as a potential source to produce organic biofertilizers by composting. Major types of organic wastes include food waste, agricultural residues, animal manure, sewage sludge (biosolids) and solid fractions of anaerobic digestate (Table 1). Based on the characteristics of organic waste, composting and co-composting strategies could be used to mix different organic

Table 1 Global organic waste production.

Type of waste	Quantity	Reference
Food waste	1.3 billion tons/year	Kaza et al. (2018)
Agricultural residues	2 billion tons/year	Duque-Acevedo et al. (2020)
Animal manure	120 million tons/year	Loyon (2018)
Sewage sludge	16.4 million tons/year	Mohajerani et al. (2019)
Anaerobic digestate	180 million tons/year	Bartocci et al. (2020)

wastes streams to obtain optimum conditions such as moisture content of 50–60% and C/N ratio of 25–30 for effective composting (Haug, 1993).

Food waste is available abundantly, which needs further treatment to curb its health and hazardous effects on humans and the environment. Global food waste production is increasing every year due to global economic growth and the changes in modern lifestyles (Kaza et al., 2018). Recycling food waste into value-added products such as compost can be a suitable option to reduce the burden on landfills. Most food wastes are compostable (Kaza et al., 2018; Li et al., 2013). As per the Word bank report, the global food waste production is ∼1.3 billion tonnes per year (Kaza et al., 2018). Most of the food wastes are generated in low- and middle-income countries where proper disposal is still lacking (Kaza et al., 2018). Composting is the most appropriate treatment technology in these countries due to its ease of operation and low maintenance.

Agricultural residues are also available abundantly, typically produced by farm activities such as crop production, cultivation and landscaping, which leads to the production of pruning and grass cuttings. Global agricultural residues generation is increasing every year due to improved efficiency of agricultural activities to fulfill the food requirement. Currently, ∼2 billion tons per year of agricultural waste is generated globally, which is expected to be doubled by 2050 due to the increase in population (Duque-Acevedo et al., 2020). Presence of recalcitrant lignocellulosic components in agricultural residues results in slow decomposition and hence longer duration to produce quality compost.

Animal or livestock manure production is increasing largely due to the demand for meat production globally. About 120 million tons/year of manure is being produced at cattle, swine, sheep and poultry farms which could be used to recover the nutrients in terms of nitrogen, phosphorous and potassium by converting them into organic fertilizer (Loyon, 2018). Composting can also assist in reducing emerging contaminants, such as pharmaceutical residues like antibiotics (Arikan et al., 2016; Dolliver et al., 2008; Hu et al., 2017; Ravindran and Mnkeni, 2017), hormones (Butkovskyi et al., 2016; Derby et al., 2011), non-Steroidal Anti-Inflammatory Drugs (Butkovskyi et al., 2016) present in manure and mu8nicipal sludge.

Sewage sludge or biosolids are the residual solids generated from wastewater treatment plants. Mohajerani et al. (2019) reported ∼16.4 million tons/year of global sewage sludge production, mostly from developed countries such as the USA, EU and Australia due to the efficient wastewater treatment system. In many countries, sewage sludge is used as a co-composting material due to microorganisms, which can enhance decomposition rate during composting.

Anaerobic digestates are byproduct of the anaerobic digestion of organic wastes. Many European countries have adapted anaerobic digestion technology to treat food waste, agricultural residues and animal manure. Results in the digestate production of ~180 million tons per year (Bartocci et al., 2020). The current practice of digestate disposal includes incineration or landfilling. The solid fractions of digestate are rich in nitrogen which can be recovered by composting effectively by adapting advanced in-situ methods such as C/N ratio adjustment and amending with physical (biochar, zeolite), chemical (Mg and P salts) and microbial additives (Manu et al., 2021b).

3. Biochar production

Biochar is a carbon-rich pyrogenic substance derived from biomass resources such as agricultural waste, wood waste, forest residues and food waste through pyrolysis (Fig. 1). The thermochemical process in which the biomass is turned into biochar is known as pyrolysis. Production of the biochar from biomass depends not only on the techniques employed but also the process parameters involved in biochar production (Tripathi et al., 2016). The quality and yield of biochar depend upon various factors such as biomass quality (biomass source, moisture content, particle size) (Atkinson et al., 2010), reaction conditions (temperature, time and heating rate) (Mohanty et al., 2013) and surrounding environments (carrier gas and flow rate) (Tripathi et al., 2016). Apart from yield, biochar properties also have been found to differ with various process parameters like temperature, residence time, heating rate, and particle size among others (Choi et al., 2012; Purakayastha et al., 2019; Zhang et al., 2020b).

Biochar production gets affected by the composition and particle size of the biomass. Presence of both cellulose and lignin in biomass enhances biochar production however, however, higher biochar yield was obtained from biomass having more lignin as compared to cellulose (Gani and Naruse, 2007; Yang et al., 2006). Similar findings were observed in rice and olive husk having more lignin content yielded a higher amount of char (Lv et al., 2010). During pyrolysis, part of the energy is utilized for removing moisture (chemically bound water or free water) content in the biomass. Large amount of moisture in biomass reduces efficiency of biochar production. Biomass with more than 30% moisture content is not considered suitable for biochar production (Bryden and Hagge, 2003). Particle size

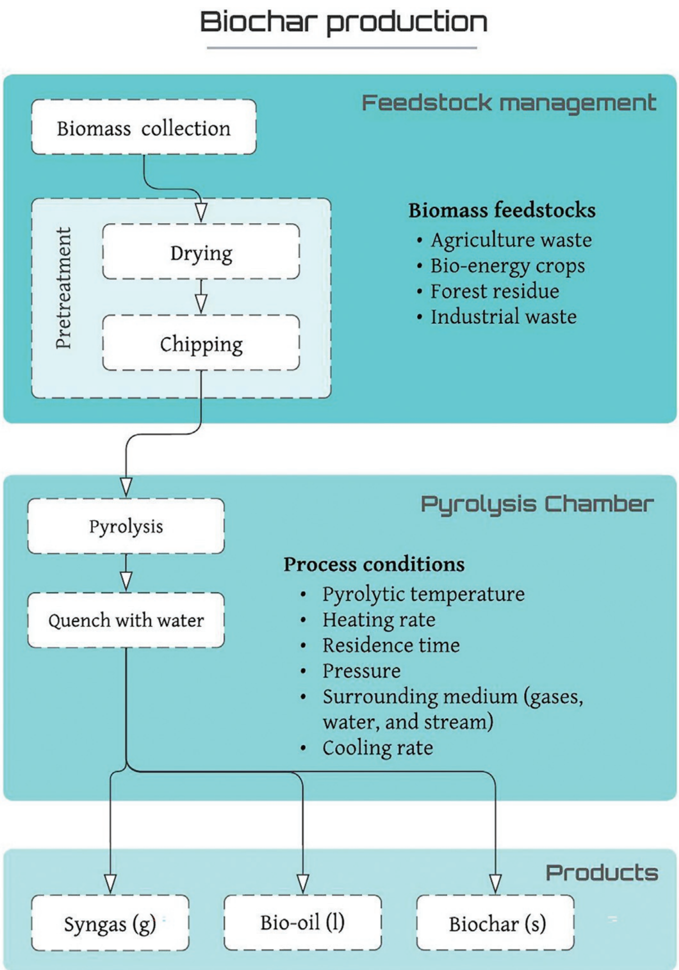


Fig. 1 A diagram of basic production process of biochar. *Modified from Roberts, K.G., Gloy, B.A., Joseph, S., Scott, N.R., Lehmann, J., 2010. Life cycle assessment of biochar systems: estimating the energetic, economic, and climate change potential. Environ. Sci. Technol. 44, 827–833; Nartey, O.D., Zhao, B., 2014. Biochar preparation, characterization, and adsorptive capacity and its effect on bioavailability of contaminants: an overview. Adv. Mater. Sci. Eng. 715398.*

of biomass influences biochar yield as particle size controls heating rate inside a reactor during pyrolysis. Biomass with larger particles yields more biochar than smaller particles due to low heat transfer (Mani et al., 2010).

The standard methods of biochar production are fast pyrolysis, slow pyrolysis, and gasification while slow pyrolysis offers compared to the other two (Igalavithana et al., 2017, reference therein). Pyrolysis temperature and heating are the most determining factors of quantity and quality of biochar. There is a clear trend of decreasing biochar yield on increasing the pyrolysis temperature (Tripathi et al., 2016). At high temperatures biochar produced from primary pyrolysis reaction undergoes secondary reactions, leading to more liquid and gaseous products at the cost of solid char (Choi et al., 2012). Similarly, a high heating rate during pyrolysis of biomass reduces the solid char yield. At a low heating rate, the chances of secondary pyrolysis reaction can be reduced, which results in a more solid char yield (Angın, 2013). However, char produced at low pyrolysis temperature (300 °C) might have more agronomic potential due to the presence of more labile C and available nutrients (Hussain et al., 2017). Plenty of literature is available on the effect of pyrolysis conditions on biochar yield but finding a suitable pyrolysis condition for high biochar yield with desired quality is difficult because optimum pyrolysis condition for higher biochar yield depends upon nature, composition and type of biomass too.

Recent research on biochar production has revealed a tendency toward improved technologies, introducing the concept of customized or tailored biochar (Hussain et al., 2017; Novak et al., 2014). The biochar production method can be customized to have unique features depending on the purpose of its application. Adjusting biomass sources and using sophisticated pyrolysis processes including microwave-assisted pyrolysis, steam-assisted pyrolysis, hydro/wet pyrolysis, co-pyrolysis and catalytic pyrolysis can help to achieve this (Lee et al., 2020; Wang et al., 2020b).

4. Compost production

Compost is a stabilized organic soil conditioner derived through a physicochemical and microbiological process, by which raw organic wastes are decomposed under controlled conditions. Since decomposition of organic matter (OM) is a microbially mediated process, composting process conditions, such as moisture, nutrient, C/N ratio, and temperature, are maintained at levels that favor microbial proliferation. For aerobic composting, it is common to aerate composting piles by passive and/or active means e.g., using bulking materials and a turner/blower machine to maintain O_2 supply at an appropriate level for microbial proliferation.

Selecting an environmentally and economically favorable composting method depends on type, quantity, nature of waste available, and access to space and technology. Composting systems can be classified into two broad groups, (1) open-air i.e., pile and windrow, and (2) in-vessel (Tchobanoglous and Kreith, 2002). The diagram in Fig. 2 shows a compost production process flowchart at an industrial scale. Composting process passes through various phases with initial vigorous decomposition followed by curing (Fig. 3).

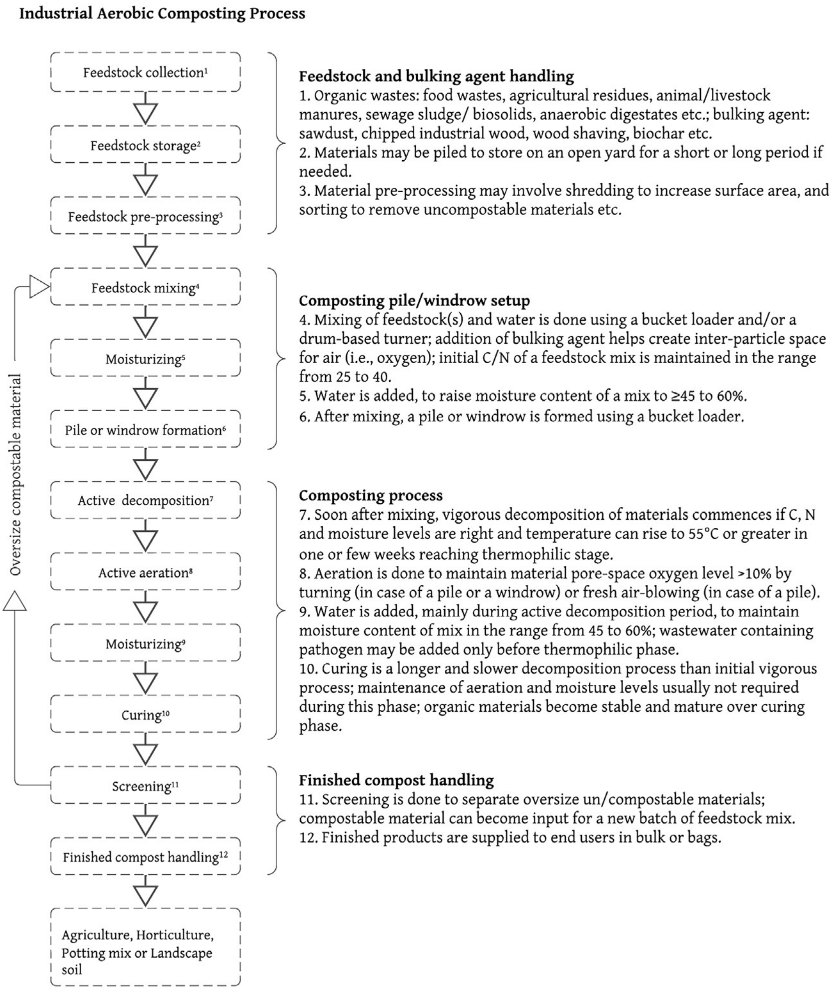


Fig. 2 A flow chart of industrial aerobic composting process and application. *Modified from Rynk, R., Kamp, M.V.D., Willson, G.B., Singley, M.E., Richard, T.L., Kolega, J.J., Gouin, F.R., Lucien Laliberty, J., Kay, D., Murphy, D.W., Hoitink, H.A.J., Brinton, W.F., 1992. On-Farm Composting Handbook (NRAES 54), New York, USA, Plant and Life Sciences Publishing (PALS); Epstein, E., 2011. Industrial Composting: Environmental Engineering and Facilities Management. CRC Press, FL, USA.*

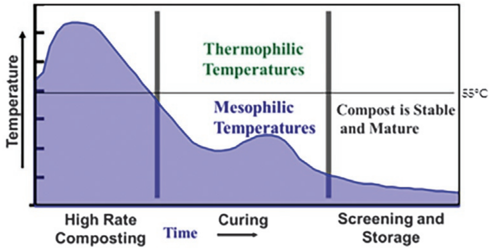


Fig. 3 A typical temperature profile of materials over a composting period. *Adapted from Wong, J.W.C., Wang, X., Selvam, A., 2017. Improving compost quality by controlling nitrogen loss during composting. Curr. Dev. Biotechnol. Bioeng. Elsevier.*

4.1 Windrow composting

Type of windrows can be classified into turned or static piles depends on the aeration mechanism. Turned windrows have a passive aeration mechanism, whereas static piles require forced aeration from external energy sources. Windrow specifications such as windrow size, shape, height, width and number of windrows depend on the quantity of waste to be treated and space availability. Total area calculation should include windrows, maneuvering, and space required for aeration system and other handling equipment (Tchobanoglous and Kreith, 2002).

Windrow construction is done by stacking a compost mixture in an elongated pile formation. A composting mixture consists of raw organic wastes and bulking agents. The ratio of feedstocks is balanced to maintain C:N ratio between 25 and 30. If co-composting is involved where more than one type of waste needs to be composted, such as sewage sludge, food waste, and digestate, preprocessing is required to mix all the feedstock properly before stacking up a pile. Dimension of windrow varies as per requirements. For MSW, windrow height should be maintained within the range of 1.5–2.5 m, and windrow width should be twice the height to ensure pile strength (Vigneswaran et al., 2016).

Turning is essential in windrow composting. Turning involves tearing down a windrow pile and reconstructing a new windrow either at the original position or at an adjacent place depending on the equipment used and available space. By turning the composting pile regularly, the O_2 supply will increase, and microbial nutrient interaction will increase the decomposition rate (Xu et al., 2020). Turning helps uniform decomposition and ensures sanitization of end product by pathogen destruction as temperature rises. Turning once or twice a week is sufficient to match the O_2 uptake rate by the active microbial community. The best turning frequency is once every three days.

On the other hand, (static) an aerated pile, having forced aeration, can avoid expensive turning equipment. Windrow construction starts with installation of evenly spaced perforated pipes at the bottom of windrow area. This pipe is connected to a blower for continuous aeration. To avoid possible short-circuiting of air supply, the pipes are covered by bulking agent like a layer of woodchips to ensure uniform air distribution during composting. The composting mix feedstock and the bulking agents will then be stacked above the bottom aeration system to form a windrow pile. Since regular turning is not provided, a layer of mature compost can be added as a cover layer to absorb odorous emissions and reduce heat loss from the pile (Vigneswaran et al., 2016).

To optimize the composting process of static aeration piles, airflow rate is maintained according to temperature change of pile. Real-time sensors are used to control the blowers to go on or off when the pile temperature reaches the set limits of 55–65 °C. Another way is to use an O_2 sensing device to activate blowers when the minimum O_2 level required drops to a predefined level. In this composting, gaseous emission could be collected by air suction and treated through a biofilter comprising of mature compost and other absorbing materials together with appropriate amounts of bulking agents (Liu et al., 2020b).

4.2 In-vessel composting

In-vessel composting systems were developed to improve composting by providing the best environmental conditions such as aeration, turning, temperature and moisture content in a confined container or vessel. Generally, in-vessel systems use forced aeration strategy for providing O_2 supply whereas decentralized in-vessel composting systems use passive aeration. Several types of composting vessels are available with different aeration and turning configurations. Industrial in-vessel composting systems can treat large quantities of organic wastes and can be broadly classified into two designs, (a) vertical and (b) horizontal reactors (Vigneswaran et al., 2016).

Vertical reactors are generally plug-flow reactors with aeration from bottom. The aeration rate could be adjusted depending on the feedstock type and bulk density. The advantages include uniform air distribution and possibilities of odor control and treatment. Horizontal reactors use rotating drums for composting. Generally, a drum is mounted on large bearings and turned through a gear system. The regular rotation of drum around a central shaft helps in mixing of waste and continuous supply of aeration for effective composting. Typical a rotary drum has 1:10 (diameter:length) configuration to treat organic wastes in a shorter period (Liu et al., 2020b; Manu et al., 2021a). Apart from industrial-scale in-vessel systems, several types of bench-scale reactors (vertical as well as rotary drums) have been used in small communities and laboratories for conducting and developing effective composting strategies for organic waste treatment (Chan et al., 2016; Manu et al., 2021a; Wong et al., 2017).

5. Biochar-compost production

5.1 The concept of mixing biochar and compost

Utilizing organic wastes for production of compost and biochar can resolve the disposal issues and both products can be used as soil amendments. Since their combined application has shown potential enhancement in agronomic values and reduction in nutrient losses, maximum benefits could be obtained by mixing biochar with organic wastes before composting (Xiao et al., 2017).

The interaction of pyrogenic C with OM was observed thousands of years ago is seen in the terra preta soils in the Amazonian region (Fischer and Glaser, 2012). These soils were found to have a high carbon content of up to $150\,g\,C\,kg^{-1}$ soil compared to the surrounding soils with $20–30\,g\,C\,kg^{-1}$ soil. These soils were highly fertile with high P contents, high cation exchange capacity and pH. The charcoal particles in the terra preta are comprised of decomposable organic substances, microbial residues and nutrients that create synergistic environment for microbial-soil-plant interactions (Kammann et al., 2016). Hence, the combination of biochar and compost is logical as it involves the interaction of organics, nutrients, microorganisms and minerals. The continued 'biochar-centered' research combined with composting/compost is gaining potential to understand how biochar surfaces alter during and post-composting (Prost et al., 2013), whether biochar helps in reducing GHG emission and N retention and changes in plant-growth-promoting properties in long terms (Agegnehu et al., 2017).

5.2 Making various types of biochar-compost

There are three modes of production/co-application of biochar-compost: (i) prepare a co-composted biochar-compost by co-composting biochar and organic waste feedstocks (ii) prepare an incubated biochar-compost by incubating biochar and a fresh stabilized/mature compost for a specified period; and (iii) prepare a biochar-compost mix by combining the two materials followed by its instant application to a soil instantly (Blackwell et al., 2009; Khan, 2015) (Fig. 4). Most of the literature (Karami et al., 2011; Steiner et al., 2007) show that, where biochar was mixed with compost, the mixture was applied to soil without any prior incubation as in the mode (iii). However, some reports in the literature suggest that biochar should be incubated with compost to allow the biochar interactions with compost (Khan et al., 2014). Steiner et al. (2004) reported examples of "cultural practices" (i.e., non-experimental) from the Amazonian basin, where biochar was incubated as in the mode (ii). Quality of biochar-compost will depend on biochar rate, feedstock used to prepare biochar and compost, pyrolysis temperature, biochar particle size, and composting process management.

5.2.1 *Co-composting biochar with organic wastes*

Addition of biochar into organic wastes before composting has been practiced in the Asia-Pacific region several years ago (Wiedner and Glaser, 2015). It has been used as a potential amendment in composting as it provides

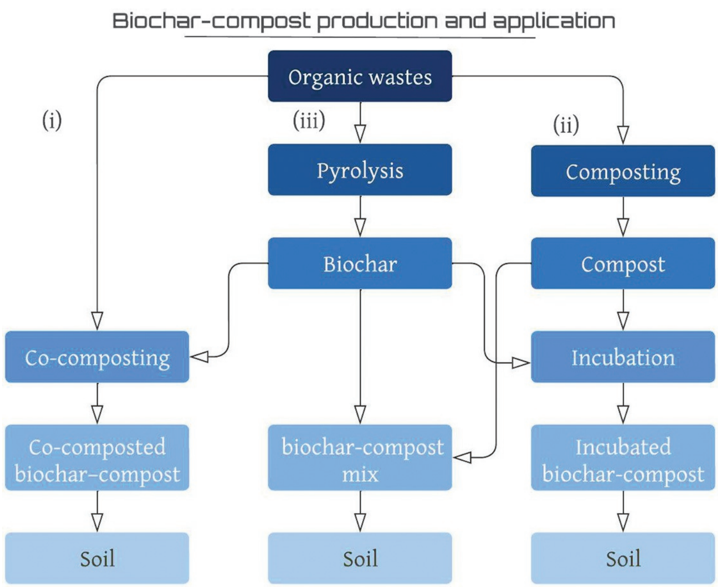


Fig. 4 A diagram showing three modes of making biochar-compost: (i) co-composting of the two materials, (ii) incubation of the two materials, and (iii) mixing the two materials just before application to a soil. *Based on Khan, N., 2015. Synthesis of Biochar Amended Composts and Their Evaluation for Bioremediation of a Sandy Soil Contaminated With Trace Metals Cadmium, Copper and Zinc. PhD thesis, Australia of South Australia.*

several beneficiary effects (Dias et al., 2010). Yoshizawa et al. (2006) demonstrated the benefits of adding bamboo charcoal to enhance OM decomposition and bacterial activities. Later, numerous studies have been published on co-composting various organic wastes with biochar revealing the composting process, microbiological regulations, environmental and agronomical benefits. Biochar can be used as a co-composting amendment (Antonangelo et al., 2021) or bulking agent to improve composting efficiency (Malinowski et al., 2019) for heavy metal remediation (Zhou et al., 2018) or nitrogen retention (Chowdhury et al., 2014). Biochar improves physical property of composting materials by increasing porosity (Guo et al., 2020) and O_2 supply thereby eliminating anaerobic condition (Liu et al., 2017a). Biochar proliferates microbial activities (Xiao et al., 2017) that accelerate OM decomposition rate and shorten composting period. Biochar promotes production of humic-like substances enriched in functional groups during composting process (Dias et al., 2010; Godlewska et al., 2017; Kleber and Johnson, 2010; Wang et al., 2014). In addition, alkaline biochar can increase compost pH (Guo et al., 2020). It may be noted that, there is a debate ongoing about the validity of concept of 'humic substances' as it was recently challenged by (Lehmann and Kleber, 2015; Kleber and Lehmann, 2019), while others found it was useful (Olk et al., 2019). Since, quantity of humic-like substances is an important index of compost maturity, here we have reported all findings in relation to this.

Most of the composting studies used biochar produced from wood with lignocellulose rich feedstock (Wang et al., 2019). Other feedstocks are agricultural wastes, sewage sludge, digestate and food waste (Guo et al., 2020; Xiao et al., 2017). The physicochemical characteristics of biochar vary with the type of feedstock used. The specific surface area of biochar produced from various feedstock was found to be in the range 0–520 $m^2\ g^{-1}$, in which a higher specific area was observed from wood or straw feedstock compared to manure or sludge (Guo et al., 2020). Biochar with higher specific surface area is preferred for effective composting to sorb nutrients as well as to mitigate gaseous emissions. Biochar is used for composting due to the large surface area, large pore volume, and chemical and structural stability. The presence of biochar creates synergistic environment with the composting process wherein the changes in physicochemical properties of composting matrix as well as biochar occurs (Sanchez-Monedero et al., 2018). The elemental composition and surface functional groups significantly affect the composting process (Khan et al., 2016).

Feedstock type and pyrolytic temperature determine the biochar properties such as surface properties, structures, pH, conductivity, CEC and elemental composition (Feng et al., 2021; Ippolito et al., 2020). Biochar C content varies between 30% and 90%, and the higher C content has been observed in wood straw biochar compared to manure and sludge (Ippolito et al., 2020). Biochars produced at lower temperatures have lower aromaticity and more oxygenated functional groups that involve in reactions when in contact with other organic matrix (Iqbal et al., 2015). The aromatic components and recalcitrant nature of biochar increase with the increase in pyrolysis temperature (Zhang et al., 2019). Increase in pyrolysis temperature increases pH, surface area, porosity and C content of biochar, whereas EC, volatile matter, H, N, S and O contents decrease (Feng et al., 2021; Zhang et al., 2019).

Several studies have been conducted with different biochar application rates for effective composting of organic wastes. Biochar application rate of 1–50% showed both positive and negative impacts on composting performance. Biochar application rate of 10% (dry weight basis) is recommended to be an optimal rate in composting (Awasthi et al., 2020a; Liu et al., 2017a). However, low rate of 3–5% also showed a significant beneficial impact on composting process (Liu et al., 2017a). Higher rates of more than 20% mostly slowed down the composting process by affecting microbial activities (Liu et al., 2017a) as N depleted biochars most likely increased C/N ratio to unfavorable values (>30). Besides, the cost of biochar also limits its use for higher rates (Liu et al., 2017a). Wide range of particles (<2 mm up to >16 mm) have been used in composting applications (Chowdhury et al., 2014). Small particles of biochar help in even distribution in composting materials in a pile. The recommended biochar particle size for effective composting is not available due to the lack of studies and nature of composting feedstock. Further research is required in this area.

5.2.2 Mixing biochar with mature compost

Mixing biochar with mature compost is another way of incorporating biochar into soil (Vandecasteele et al., 2016). The nature and mixing ratio of biochar and compost affects the outcome of agricultural applications. Mixing 7–10 t ha^{-1} of biochar from agricultural residues with 20–40 t ha^{-1} of compost has shown a significant increase in plant growth and yield in different soils (Cao et al., 2018; Doan et al., 2015; Khorram et al., 2019). Manolikaki and Diamadopoulos (2019) used 2% rice husk biochar and

2% compost and observed 58–78% ryegrass yield in sandy and loam soil. Similarly, Naeem et al. (2018) observed increased leaf chlorophyll content of maize plants with 0.5% wheat straw biochar and 0.75% compost addition on sandy clay soil. In these studies, improved soil conditions such as increased total N, available P, cation exchange capacity (CEC), SOC and water holding capacity were observed. Teodoro et al. (2020) reported no effect of mixing biochar and compost on yard grass plant development in sandy loam. The above findings show the beneficiary aspects of biochar and mature compost mix in agriculture. However, co-composting organic wastes with biochar (co-compost) may be a better option than mixing biochar with compost biochar can improve composting process in addition to instant capturing (sorbing) of nutrients released from organic material.

5.2.3 Adding biochar at beginning of a composting process

To maximize the effects on the composting process, biochar has to be added at the beginning of the process. In this way, biochar has the opportunity to interact with the diverse microbial communities colonizing the pile, the C forms involved in OM decomposition and transformation, and the N forms engaged in the generation of NH_4^+ and the release of NH_3 and N_2O. Khan et al. (2016) compared the impact of biochar on composting and incubation with mature compost, using the same biochars and composting materials. These authors encouraged co-composting over incubation as the thermophilic stage in the former method increases cation exchange capacity (CEC) of biochars and consequently, their nutrient retention capacity. Hagemann et al. (2017b) also found that adding biochar at the beginning of the process resulted in a compost of higher agronomic quality (superior plant growth promotion in pot trials) compared to the mixing of pristine/fresh biochar (no post-production treatment) into already matured compost. However, several authors found advantages when adding biochar to mature composts. For instance, Vandecasteele et al. (2017) found that in case of soils rich in available P, biochar-complemented mature compost may result in positive effect like reducing readily available P.

Adding biochar at the beginning of the process also allows the modification/weathering of biochar. Composting was found to enhance the functionalization of biochar surfaces, increasing cation exchange capacity and sorption of C, N and plant-available nutrients in and on the biochar. Prost et al. (2013) tested, to which extent composting with farmyard manure increases CEC and nutrients of beech wood biochar prepared at 550 °C. They found an increase in potential CEC (from 20.8 to 39.0 mmol c kg^{-1}), water-extractable OC (from 0.03 to 3.52 $g\,kg^{-1}$), total soluble N (from 3.2 to 377.2 $mg\,kg^{-1}$), available P (from 44 to 190 $mg\,kg^{-1}$) and available K (from 0.6 to 8.5 $g\,kg^{-1}$). Hagemann et al. (2017a) identified that a nutrient-rich organic coating on the surface of composted biochar could be responsible for enhanced nutrient retention.

5.3 Influence of biochar on composting process

Carbonized biomass and ashes have traditionally been used in agriculture as a soil amendment or mixed with manures due to their favorable physico-chemical properties that influence soil nutrient cycles (Ogawa and Okimori, 2010). In particular, there is a growing interest in the interaction of biochar with the composting process due to the enhancement of the final compost product and the creation of a more efficient biodegradable process from the point of eco-friendly and practical views.

Numerous studies were devoted to investigating influence of biochar on organic waste composting. As shown in Table 2, although the application ratio of biochar varied in a wide range (2–50%), biochar application rates of 5–10% were accepted in compost engineering practice (Akdeniz, 2019; Guo et al., 2020). According to current literature, co-composting of organic waste and biochar has many advantages, such as increasing compost substrate porosity; accelerating OM decomposition and humification; increasing compost pile temperature and extending thermophilic phase; enhancing microbial activities; shortening compost production period and advancing product maturity; reducing ammonia and greenhouse gases emission; improving compost agricultural fertility by decreasing N loss; and decreasing potential environmental risk by lowering toxic contaminants mobility and bioavailability (Agegnehu et al., 2017; Guo et al., 2020).

The inherent physicochemical properties of biochars, such as large porosity, surface area, and high cation exchange capacity, can favor microbial growth in a composting pile. Biochar is generally characterized by a recalcitrant structure that is hardly degraded during the process. Biochar in general, is not a significant source of nutrients. Biochars prepared from nutrient-rich feedstocks (e.g., manures) or woody feedstocks at relatively low temperatures can contain C and nutrients (Domingues et al., 2017; Ippolito et al., 2020). Moreover, composting materials can have a large pool of OM, nutrients and microbial biomass. Therefore, in a biochar-complemented composting pile, a synergy is expected due to physico-chemical properties of biochar and composting materials, with biochar

Table 2 Typical studies on biochar-assisted organic waste composting.

Compost substrates	Biochar feedstock	Biochar pyrolysis temperature (°C)	Biochar adding amount	Composting scale and process	Effects on composting	Reference
Poultry manure	Pine chips	400	5% and 20% (d/w[a])	Reactor, 60 L, 42 days	Increased pH and compost temperature Increased CO_2 release and reducing NH_3 emission >52% of total N loss was inhibited	Steiner et al. (2010)
Poultry manure	Wood	300–450	50% (w/w[a])	Turned pile, 500 kg, 210 days	Accelerated organic matter decomposition and compost maturity Decreased NH_3 and odor gases emissions	Dias et al. (2010)
Pig manure mixed with sawdust	Bamboo		3%, 6%, and 9% (w/w)	Reactor, 114 kg, 56 days	Increased temperature and extended thermophilic phase Decreased total N loss by 28%–65% Decreased Cu and Zn bioavailability	Chen et al. (2010)
Cattle manure or poultry manure + apple pomace + rice straw + rice bran	Hardwood tree (Quercus serrate Murray)	400–600	10% (w/w)	Cone-shaped windrows, 200 kg, 12 weeks	Biochar induced specific changes in the microbial community structure, depending on the original organic wastes	Jindo et al. (2012a)

Poultry manure + apple pomace + rice husk + oak bark	Japanese charcoal kiln	400–600	2% (w/w)	Cone-shaped windrows, 3000 kg, 150 days	Biochar addition increased 10% of humic substance extraction and decreased 30% of water-soluble C The urease, phosphatase and polyphenol oxidase activities of the biochar-blended compost were enhanced by 30–40%	Jindo et al. (2012b)
Sewage sludge + rice straw	Wood	500–600	6%, 12%, and 18% (w/w)		Increased compost substrate porosity Facilitated organic matter decomposition and humification	Zhang et al. (2014)
Pig manure + corn stalk powder (1:2; d/w)	Corn stalk	250–300, 450–500, 600–700, and 750–900	2.5% (d/w)	Reactor, 100 L, 90 days	Increased organic matter decomposition 14.8–29.6% Increased peak temperature to 70.9–72.8°C and prolonging thermophilic phase Increased compost pH even up to 9.4 particular for biochar obtained from high pyrolysis temperature of 750–900 °C Prevented NH_3 loss with addition of biochar from low temperatures (250–300 and 450–500 °C) Decreased Cu and Zn bioavailability	Li et al. (2015)

Continued

Table 2 Typical studies on biochar-assisted organic waste composting.—cont'd

Compost substrates	Biochar feedstock	Biochar pyrolysis temperature (°C)	Biochar adding amount	Composting scale and process	Effects on composting	Reference
Cow/poultry manure + apple pomace + rice straw + rice bran	Hard wood	550	10% (volumetric)	Cone-shaped windrow, 200 kg, 12 weeks	Improved compost quality	Jindo et al. (2016)
Pig manure + wheat straw	Wheat straw	500–600	5%, 10% and 15% (w/w)	Reactor, 15 L, 42 days	Enhanced organic matter decomposition and compost maturity	Zhang et al. (2016)
					Increased dissolved organic carbon, ammonium and water-soluble nutrient contents	
Kitchen waste + sawdust + rock phosphate (10%)	NR	NR	NR	Reactor, 12.5 L, 35 days	Increased available P fractions	Wei et al. (2016)
Sewage sludge + wheat straw (1:1; d/w)	Wheat straw	NR	2%, 4%, 6%, 8%, 12% and 18% (d/w)	Reactor, 100 L 56 days	Reduced greenhouse gases emission	Awasthi et al. (2017)
					Reduced NH_3 emission	
Sewage sludge + wheat straw (dry weight, 1:1)	Wheat straw	NR	2%, 4%, 6%, 8% and 12% (d/w)	Reactor, 130 L, 56 days	Reduced volatile fatty acids formation and improved compost humification	Awasthi et al. (2018)
					Reduced odor emission and increased total bacterial abundance	

Pig manure + Wheat straw (10:1; w/w)	Powdered rice straw powdered bamboo, granular rice straw, granular bamboo	NR	10% (w/w)	Reactor, 15 L, 18 days	Granular-biochar improved pore connectivity and benefited methanotrophs activities and reduced CH_4 emission	He et al. (2019b)
					At the same particle size, bamboo biochar had a higher pore volume and more aerobic microenvironment within the compost than rice straw biochar, reducing GHG emissions	
					Bamboo biochar had high aromatic compound and NO_3^- concentrations causing low N_2O emissions and inhibiting denitrifying bacteria	
					Powdered biochar had more exposed reactive functional groups and decreased NH_3 production better than granular biochar	
Pig manure + sawdust (2:1; d/w)	Pine leaf	NR	0%, 2.5%, 5%, 10% and 15% (d/w)	Reactor, 9 L, 50 days	Increased compost bacterial diversity and compost quality in particular with 10% biochar addition	Li et al. (2019a)
Poultry manure + straw (39:11; d/w)	Holm oak	650	0% and 3% (d/w)	Trapezoidal piles, 3000 kg in fresh weight. 20 weeks	Reduced the concentration of volatile organic compounds during the thermophilic phase	Sánchez-Monedero et al. (2019)

Continued

Table 2 Typical studies on biochar-assisted organic waste composting.—cont'd

Compost substrates	Biochar feedstock	Biochar pyrolysis temperature (°C)	Biochar adding amount	Composting scale and process	Effects on composting	Reference
Dewatered sewage sludge + pine wood sawdust	Rice straw	500	5%, 10% and 20% (w/w)	Reactor, 400 L, 28 days	The addition of biochar above 10% inhibited protease activity but promoted the activities of cellulase and peroxidase, which also increased the fluctuation of bacterial diversity during the composting	Du et al. (2019)
					The relationship between the activity of most enzymes and bacterial community was strengthened by the addition of biochar (10% and 20%), which further enhanced the contributions of the functional bacterial communities to composting	
Cow manure + corn stalks (1:0.9; w/w)	NR	NR	10% (d/w)	Reactor, 50 L, 28 days	Biochar addition increased the proportion of inorganic phosphorus	Wei et al. (2021)
Chicken manure + mushroom residues (1:1; w/w)	Maize straw	400	5% (d/w)	Reactor, 72 L, 42 days	Compared to control (92.6%), the abundance of antibiotic resistance genes reduced by 98.7% with biochar addition	Zhou et al. (2021a)
					Addition of biochar mitigate the accumulation and spread of antibiotic resistance genes during composting	

Swine manure + rice straw (8:1; w/w)	NR	NR	10% (w/w)	Reactor, 60 L, 40 days	Improved the physical structure of compost substrate for enhancing humic acid formation	Guo et al. (2021)
					Decreased bulk density and increased free air space of the compost	
					Increased total porosity of compost aggregates by approximately 90%	
Pig manure + sawdust (8:1; d/w)	Pig manure	550	5% (w/w)	Reactor, 50 L, 80 days	Biochar amendment was found to increase the concentrations of monomethylarsonic acid and dimethylarsinic acid during mesophilic (days 0–10) and early thermophilic phases, promoting As volatilization during the maturing phase of composting	Zhai et al. (2021)
					The abundances of As(V) reductase (arsC) and As(III) *S*-adenosyl-L-methionine methyltransferase (arsM) genes were higher in the biochar treatment	
					Biochar amendment influenced the microbial communities by promoting As methylation and volatilization	

[a]w/w: wet weight; d/w: dry weight

providing support for microbial growth favoring OM decomposition, humification, nitrification, denitrification, and methanogenesis (Kammann et al., 2016; Schmidt et al., 2014). At the same time, biochar can undergo intense oxidation that alters its surface chemistry and interaction with nutrients and soluble OM (Hagemann et al., 2017a; Prost et al., 2013).

Biochars can modify various composting process conditions in composting pile. The use of biochar as a bulking agent or compost additive has been reported to have significant impact on aeration and physical structure of composting media (Guo et al., 2021), which governs rate of decomposition of OM. Biochar's porosity enhances micro aeration in composting materials, thereby accelerating composting process and preventing formation of anaerobic spots (Jindo et al., 2012b; Zhang et al., 2014). Biochar with small particle size can be distributed evenly in a composting pile. Biochar with large particle size can be used as a bulking agent. Biochar amendment can increase water retention capacity in a composting pile thereby helping to maintain moisture content that is suitable composting process (López-Cano et al., 2016; Wang et al., 2015). Biochar can influence pH buffering capacity (Zhang et al., 2014) of a composting pile, and reduce availability of in-situ toxic compounds present in compost and/or biochar feedstocks (Borchard et al., 2014c; Schmidt et al., 2014; Spokas et al., 2014).

Modification of physicochemical properties including bulk density, pH, temperature, and C/N of composting piles by biochar amendment, is found to influence compost microbial communities (Jindo et al., 2012b). Biochars generally enhance microbial diversity. Remarkably, it favors bacteria and diazotrophic communities (bacteria + archaea) (Awasthi et al., 2020b; Bello et al., 2020; Cui et al., 2020; Zainudin et al., 2020) rather than fungi community. Proteobacteria and actinobacteria populations can be enhanced with application of biochar to a composting pile (Bello et al., 2020). Overall, some specific bacteria families are probably promoted by biochar amendment due to changes in physicochemical characteristics of the composting materials. Influence of biochar on several composting process parameters including aeration and pH are further detailed below.

5.3.1 Aeration

Aeration is an important factor for composting process, since activities of aerobes are strongly dependent on O_2 level (Bernal et al., 2009). Bulking agents are often added to provide structural support that prevents physical compaction of pile while increasing air voids (Bustamante et al., 2013; Jindo et al., 2012a). Biochar is a carbon-based porous material that can act as a bulking material in a compost pile (Dias et al., 2010; Steiner et al., 2010). Aerobic condition increases decomposition rate that is reflected in a higher pile temperature during thermophilic phase (Dias et al., 2010; Li et al., 2015; Sánchez-Monedero et al., 2019). Although no difference in peak temperature was observed in co-composting of sewage sludge and 9% bamboo biochar (Hua et al., 2009), most studies found that biochar addition increased compost pile temperature, extended thermophilic phase, increased compost substrate porosity, decreased pile density, accelerated OM decomposition and facilitated humification of OM. Zhang et al. (2014) noted that, biochar played a positive role as an O_2 carrier by virtue of its porosity. Biochar addition increased the proportion of small particles (<25 mm) from 36% in control to 55% in the biochar-amended compost pile. Sánchez-García et al. (2015) and Liu et al. (2017a) found that the mixing of biochar with manure decreased the pile density from 0.35 kg L^{-1} (control) to 0.26 kg L^{-1} after addition of 10% biochar. Reduction of particle size and pile density facilitate O_2 diffusion in compost matrix, thereby avoiding anaerobic sites while enhancing effectiveness of composting process (Sánchez-García et al., 2015). Czekała et al. (2016) investigated the O_2 concentration in the outlet air during composting; they observed a higher O_2 content in biochar treatments at the initial stage of composting. He et al. (2019b) compared effects of two types of biochars (bamboo biochar and rice straw biochar) with two particle sizes in each (powder: $\phi < 1$ mm; granular: 10 mm $> \phi >$ 4 mm) on pig manure composting. They noticed that compared to powder-biochar, granular-biochar improved pore connectivity, and bamboo biochar had a higher pore volume and created more aerobic microenvironment within the compost matrix than rice straw biochar. They pointed out that powdered biochar had more exposed reactive functional groups and decreased NH_3 production than granular biochar. Steiner et al. (2010) noticed a significantly higher maximum temperature in 20% biochar treatment (65.1 °C) than in control (58.2 °C); this was attributed to the decreased bulk density and increased aeration. Moreover, the maximum pile temperature reached 54.3 °C, 56.7 °C and 66.3 °C with biochar rates of 0%, 5% and 15% (Zhang et al., 2016). A greater air space is believed to reduce heat loss in biochar-added piles (Dias et al., 2010; Zhang et al., 2016). However, to date, limited research has been conducted to evaluate the variations of O_2 concentration and heat distribution in biochar-complemented compost matrix. The biochar amended compost piles showed a higher OM loss and mineralization rate than control (Sánchez-García et al., 2015). For example, a higher OM decomposition rate of 73.2% was achieved in poultry manure-biochar mixture than those composting with traditional bulking agents like sawdust (65.0%) and coffee husk (84.2%).

5.3.2 pH

pH is a critical parameter in composting as it affects microbial activities during composting. The pH range from 5.5 to 9 is believed to be optimum for composting, and decreased microbial activities would occur when the pH is outside the optimal range (Bernal et al., 2009). The alkaline pH favors NH_3 volatilization, which is one of the sources of odorous gases and averse to the conservation of N (Li et al., 2015). While the presence/formation of organic/inorganic acids during the initial phase of composting would interrupt composting process from the mesophilic to the thermophilic phase (Zhang et al., 2016). Therefore, alkaline materials such as zeolite, fly ash and $CaCO_3$ substrates are usually needed to adjust initial pH for better composting performance (Awasthi et al., 2016; Margaritis et al., 2018; Manyapu et al., 2018). As described before, most biochars are alkaline and usually have a high buffering capacity, making biochar an optimal material for composting pH mitigation. Li et al. (2015) observed that the highest pH value ($\sim$9.4) was noted in the pig manure composting pile that was mixed with corn stalk biochar (pyrolyzed at 750–900 °C) during the whole composting process. And the addition of biochars prepared at $>$700 °C resulted in higher pH and NH_3 emission during the thermophilic phase. Similarly, Wei et al. (2016) reported that higher pH values were found in the biochar treatments, and the decrease of pH in the first three days did not occur in kitchen waste-sawdust-biochar treatment. The feedstock and pyrolysis temperature had a significant effect on biochar alkalinity that in turn influenced the pH. Biochar prepared at low pyrolysis temperature ($<$600 °C) is recommended as a suitable amendment in pig manure composting (Li et al., 2015). The addition of wheat-straw biochar from 0% to 15% did not increase pH in swine manure compost (Zhang et al., 2016).

5.3.3 Compost microbial populations and activities

In addition to aeration and pH, microbial activity is a critical aspect of progression of composting process (Zhang et al., 2016). The communities and populations of predominant microorganisms differ with composting stages (Chang et al., 2021). Large numbers of mesophilic, thermotolerant and thermophilic aerobic microorganisms, and some anaerobic organisms like *Clostridium* participate in the composting process (Chang et al., 2021).

Adding biochar to composting material could further induce changes in microbial communities through multiple pathways (Jindo et al., 2012a). By virtue of high porous structure, large surface area and water holding capacity etc., biochar can serve as a suitable habitat for microorganisms (Tu et al., 2019). A decrease in bulk density, increase in aeration, and maintenance of moisture content at a suitable level in composting pile would positively affect microbial community (Wei et al., 2014). Biochar has large pore sizes that can promote microbial colonization by trapping large number of spores, bacteria, and other microbial communities (Pietikäinen et al., 2000). For these reasons, a proliferation of microorganisms and colonization of the biochar's surfaces in composting piles was observed by many researchers (Pietikäinen et al., 2000; Sun et al., 2016; Tu et al., 2019). The biochar addition showed a significant effect on diversity of microbial community than other porous materials like zeolite (Wei et al., 2014). After addition of biochar, an increase in bacterial number and diversity were observed (He et al., 2019a). Duan et al. (2019), in a 55-days large-scale aerobic co-composting of rice straw biochar and pig/chicken manure, observed that the total number of microbial populations in the biochar-amended treatments, was significantly higher than that of the control. Awasthi et al. (2018) studied co-composting of biosolids-wheat straw with biochar at different rates (2%, 4%, 8% and 12% on a dry weight basis). They found that total bacteria was significantly higher for 8–12% biochar than 2% biochar and control. It is suggested that, both physical and chemical properties of the composting piles affect microbial community structure during composting process (Jindo et al., 2012a; Sun et al., 2016).

Biochar treatment can also affect microbial communities by altering the nutrient supply. Biochar contains labile aliphatic compounds and ash, which can be utilized as a carbon mineral source by microbial communities (Sun et al., 2016). Moreover, microorganisms can also consume the adsorbed dissolved organic matter after proliferation of biochars (Pietikäinen et al., 2000, Sun et al., 2016). Sun et al. (2016) found that during the mesophilic phase, the water-soluble carbon and water-soluble phenols in biochar increased from 2.1 to 26 $mg\,kg^{-1}$ and 5.9 to 101 $\mu g\,kg^{-1}$, respectively; meanwhile, higher number of bacterial communities were found in biochar than the adjacent compost. Under influence of biochar, microbial communities decompose more dissolved organic carbon. Khan et al. (2014) found a high amount of decomposition of dissolved organic carbon in biochar-treated compost pile over 126 days of composting. Wang et al. (2013) found that 3% biochar amendment affected the abundance of denitrifying bacteria:

it decreased the population of N_2O-producing bacteria, but increased the population of N_2O-consumeing bacteria.

5.3.4 Decomposition rate

Biochar addition accelerates the OM decomposition and reduces the composting time by moderating composting conditions (Dias et al., 2010; Jindo et al., 2012b). Accordingly, 20% and 70% of OM decomposition in composting process were achieved by adding biochar at 3% and 50% (Dias et al., 2010; Sánchez-García et al., 2015). Zhang et al. (2016) found that compost was stabilized to a greater degree by day 28, which was shorter than that of the control. Mixing biochar to spent mushroom compost and green waste significantly reduced the composting period from traditional 90–270 days to only 24 days in the two-stage co-composting system (Zhang et al., 2014). A fast OM decomposition attained by adding biochar, would decrease production cost due to reduced production period and freeing up of composting yard (Xiao et al., 2017).

5.3.5 Volatile organic compound

Another practical implication of biochar is the reduction of volatile organic compounds (VOC) in a composting pile, linked to the odor emissions from composting piles (Li et al., 2019b). Volatile fatty acids (VFAs) are typical sources of odor released during composting process. These are generated by a bio-decomposition substrates process (Chowdhury et al., 2014). Duan et al. (2019) found that adding 10% of biochar reduced accumulation of volatile fatty acids in a chicken manure composting pile and reduced the maximum odor index value. Similarly, Sánchez-Monedero et al. (2019) reported that a reduction of N and oxygenated volatile compounds in poultry manure composting pile amended with biochar at a low rate (3%). These authors suggested the importance of sorption capacity of biochar and its impact on the composting progress as the primary drivers for reducing VOC (Sánchez-Monedero et al., 2019).

The release of VFAs and NH_3 were often affected by time, pH, temperature, and substrate in a composting system (Arriaga et al., 2017; Dhamodharan et al., 2019). Owing to its porous property, biochar was proved to be a promising additive for its great potential to control VFAs released from a composting process. Duan et al. (2019) assessed effectiveness of bamboo biochar amendment (0%, 2%, 4%, 6%, 8%, and 10% on dry weight basis) on decomposition and humification of VFAs in chicken manure (CM) composting. They found that the release of VFAs decreased with the increasing rate of biochar; additionally, they found that the biochar extended the thermophilic period, increased the humic acid/fulvic acid ratio and increased the richness of bacterial community. They suggested that, employing 10% bamboo biochar was the optimal and also the feasible rate for manure wastes in view of odor control and compost humification. Similar results were also reported by Awasthi et al. (2018) as observed in co-composting of biosolids-wheat straw with biochar at different rates (2%, 4%, 8% and 12% on a dry weight basis). Sánchez-Monedero et al. (2019) found that, biochar at a low application rate reduced the concentration of VFAs during the thermophilic phase. Biochar significantly reduced the volatile N compounds, the most abundant VFA family, as a result of microbial transformation of the manure-derived N-compounds. The most efficient reduction of VOC was observed in oxygenated volatile compounds (ketones, phenols and organic acids), which are intermediates of OM degradation, whereas there was no effect on other VFA families (aliphatic, aromatic and terpenes). All these findings suggest that biochar, by virtue of its adsorption capacity, can play as a significant controller of VFA emission from composting systems.

5.4 Reduction in ammonia loss from co-composting system

NH_3 is one of the sources of odor in composting process. A study found that, during the composting process about 40% to 80% of initial N was lost as NH_3, which decreased compost nutrient quality while increased air pollution (Li et al., 2020; Shou et al., 2019). Decreasing of NH_3 emission is widely reported. Even though a biochar with high pH favors NH_3 emission, its presence in a compost system decreases volatilization of NH_3, promotes accumulation of NH_4^+ in thermophilic phase, controls NH_4^+—NH_3 equilibrium, and increases nitrification rate (Chen et al., 2010; Li et al., 2015; Steiner et al., 2010). It is believed that, NH_3 and soluble organic N can be retained in biochar pores (Hua et al., 2009, 2011).

Chowdhury et al. (2014) found that biochar addition decreased the cumulative NH_3 losses by about 11–21%. Hua et al. (2009) found that 9% biochar addition can reduce up to 64% NH_3 emission. During composting process, only 4 kg N (=3% N of the initial feedstock) was lost from municipal solid waste amended with 10% biochar in comparison to 35 kg N (=22% N of the initial feedstock) where no was biochar added (Vandecasteele et al., 2016). Steiner et al. (2010) observed a decrease of NH_3 loss at higher biochar rate. It is believed that biochar applied at a high rate can provide a favorable condition for microbial growth and increase immobilization of NH_4^+ into microbial biomass (Awasthi et al., 2017). One of the major benefits of biochar amendment in organic waste composting, is reduction of N loss particularly from N-rich wastes at the early stage of the process.

A recent meta-analysis by Cao et al. (2019) reported that, use of biochar can reduce the emission of NH_3 by 40% during composting. The following mechanisms are believed to be involved in the reduction of NH_3 emission: (1) adsorption of NH_3 on the surface of biochar and retention of NH_4^+ onto negatively charged exchange sites (Cao et al., 2019); (2) modification of the microbial environment (e.g., pH increase, C/N ratio reduction, provision of easily-degradable volatile compound), which favors nitrifying bacteria that convert ammonia to nitrate—the remaining chemical in the compost end-product (Deng et al., 2021; Zhou et al., 2021b).

5.5 Reduction in greenhouse gas emissions from co-composting system

The emission of nitrous oxide (N_2O) and methane (CH_4) during composting (Fukumoto et al., 2003), which account for 0.1% to 10% of the initial feedstock N and 0.8% to 14% of initial feedstock C, made composting as a significant source of greenhouse gases (GHGs). Biochar can reduce greenhouse gas emissions. Addition of biochar can improve physical properties of composting pile, decrease particle size, facilitate gas diffusion, and reduce formation of anaerobic sites in compost matrix. Therefore, methanogenic activity and denitrification process would be declined. Biochar addition has successfully reduced total GHGs emissions by 27–32% (Chowdhury et al., 2014). More specifically, a reduction of CH_4 emission was reported by Vandecasteele et al. (2016) in municipal solid waste and green waste compost amended with 10% biochar. Particle size of biochar strongly affects GHG emissions (CH_4 and N_2O) and microbial community in composting material (He et al., 2018, 2021). Granular biochar (4 mm—1 cm) reduces CH_4 emission more than powder biochar (<1 mm) by changing mcrA/pmoA ratio (He et al., 2018).

Cao et al. (2019) reported that, the use of biochar resulted in reduction of N_2O and CH_4 by 59.8% and 67.5%, respectively. Chowdhury et al. (2014) showed a 27–32% reduction of the GHG emission by hardwood biochar amendment, compared to non-biochar compost material. However, amount of GHG emission varies with feedstock, type and volume of biochar, and composting process conditions. For example, co-composting of chicken manure with hardwood sawdust and rice hull biochar (20%) reduced N_2O emission by 60% (Jia et al., 2016), while, co-composting of pig manure, sawdust and bamboo biochar (5%) reduced N_2O emission by 37% (Wang et al., 2013). The mechanisms of N_2O reduction during the composting can be partly due to the modification in the microbial activity by: (1) reducing inorganic available N pool for nitrifiers and denitrifiers by sorption or microbial immobilization; (2) increasing activity of N_2O^--reducing bacteria and lowering amount of the enzymes linked to N_2O production (Wang et al., 2013); and, (3) facilitating electron transfer to denitrifying microorganisms (Cayuela et al., 2013; Joseph et al., 2010). Change in physico-chemical properties and increased aeration in biochar-complemented composting material reduces CH_4 emission by lowering the activity of methanogens and increasing the activity of methane-oxidizing bacteria (Chen et al., 2020; He et al., 2018; Sonoki et al., 2013).

A biochar applied at a high rate decreases reduction of GHG emission: Jia et al. (2016) showed that among 10%, 20%, 30% of biochar, the 20% treatment significantly reduced the GHG emission. The emission of N_2O was lowered from manure compost process by 31% (Wang et al., 2013, 2016b) after applying a biochar at 3% (that seemed to have no significant impact on N_2O formation (Sánchez-García et al., 2015), though). Therefore, application rate might be a key factor affecting GHG emission from biochar-amended compost. Significantly low CH_4 and N_2O emissions were observed by Awasthi et al. (2017) in the higher-rate biochar treatments: 8%, 12% and 18% in sewage sludge composting. They believed that the formation of anaerobic condition inside large clumps of composting material, as one of the causes of CH_4; and air supplemented by porous biochar would address this problem. Biochar amendment might affect GHG emission by modifying microbial communities. Granular bamboo biochar amendment at the rate of 10% in pig manure/wheat straw composting system, increased pore volume and aerobic microenvironment, and increased methanotrophs activities, thereby reducing CH_4 emission through its oxidation (He et al., 2019b).

5.6 Fate of biochar carbon and humic substances after co-composting

Under environmental conditions of a composting pile, characterized by high temperatures up to 70 °C, abiotic oxidation of biochar particles occurs, representing an initial decomposition step facilitating further microbial decomposition (Cheng et al., 2006). Fragmentation of biochar as a result of biochar decomposition and physical disintegration, may have implications on their (fragments) mobility and C sequestration in soils (Spokas et al., 2014); it can also have similar implications during a composting process.

Release of easily degradable compounds is higher from biochars produced at low temperature or originated from crop or grass-derived feedstocks. Some of the C is adsorbed in to the humic complexes during the composting process and become a part of the humic moiety. Jindo et al. (2016) studied the modification of chemical structure of humic and fulvic acids, extracted and isolated from the composting matrix. They observed enrichment of aromatic-C and carboxylic-C fractions in both humic and fulvic acids, implying a more condensed and chemically stable structure. They observed this effect in a composting pile prepared from poultry manure, where biochar was added at a very low rate of 3% (Jindo et al., 2019). Wei et al. (2014) identified *Actinobacterium* sp. in biochar amended composts, and this bacterium has been reported several times on soils amended by biochar. This bacteria group (e.g., alkaliphilic *Actinobacteria*) is involved in transforming humic acids. It can catalyze the degradation of humic acids, transforming them into soluble forms (Shivlata and Satyanarayana, 2015).

6. Compost quality, regulations, standards and guides

The properties of composts are functions of feedstock properties and composting process. On the other hand, the physical, chemical and biological content of compost can influence soil quality. It is common to have pathogens in compost feedstocks. Some feedstocks may carry trace metal and organic contaminants at excess/toxic levels, and these may end up in compost end-products. Hence, many countries/regions including Australia, Canada, USA, several EU countries established legislation, guidelines, and/or supporting standards to regulate compost quality (CCME, 2005; Standards Australia Limited, 2012; US Composting Council, 2002). Australian AS-4454 is a comprehensive compost standard and test method. "Test Methods for the Examination of Composting and Compost" is one of the most advanced laboratory manuals that provide benchmark methods for compost analysis (US Composting Council, 2002). It may be noted that, a biochar-complimented compost may be considered as compost, and will be regulated accordingly.

7. Biochar influencing compost agricultural property

Biochar is known for its high sorption capacity; therefore, a biochar-compost might retain nutrients. Since compost is used as a nutrient-providing soil conditioner, its quality is a major concern (Li et al., 2020). It is noted above that biochar amendment decreases N loss from composting material by decreasing NH_3 volatilization. Wei et al. (2014) believed that, *Rhizobium* sp., which is capable of N fixing, detected in biochar treatment, may significantly increase N content in biochar-amended compost. Readily available P content was found in high amount after adding biochar at the rate of 10% (Vandecasteele et al., 2016). A combined application of biochar and phosphate-solubilizing bacteria inoculation was found to increase the available P fraction (Wei et al., 2016).

It is desirable that a compost achieves "maturity," a certain quality or biochemical property that ensures it is safe to apply to an agricultural soil. By controlling biochar amendment rate, it is possible to further optimize the product (compost) by maintaining C/N ratio <12 (Bernal et al., 2009; Sánchez-García et al., 2015). "Stability" is another desired quality in a compost. Even though the C/N ratio of biochar-amended compost was high and ranged from 31.5 to 35.7, this product can still be regarded as a stabilized product on the basis of other parameters according to Khan et al. (2014). Awasthi et al. (2017) found that the maximum humic acid (HA) content observed was 18.09% in 12% biochar treatment, and the humic acid:fulvic acid (HA/FA) ratio increased with the increasing biochar application rates.

When applied to soil, biochar and/or compost with supplementary fertilizer can improve soil health and boost productivity of peanuts along with environmental benefits and climate change mitigations (Agegnehu et al., 2015, 2016). A study suggested that, biochar and/or compost application affects bacterial community in soils (Wu et al., 2017). However, there was no additional benefit of co-composting biochar and compost than simply adding them on the field together (Agegnehu et al., 2015). More research should be done to investigate the effect of biochar-compost on agricultural production and soil properties.

Biochar application helps in decreasing the potential risks related to the available toxic metals and residual antibiotic resistance genes (Cui et al., 2016; Thomas et al., 2020; Zhou et al., 2019). Biochar has been known as a potential adsorbent for inorganic contaminant reduction in both soil and water environment. During the composting process, most antibiotics were gradually decomposed (Kui et al., 2020; Zhang et al., 2021). The composting process accelerated the oxidation of biochar surface by creating more oxygen functional groups (Prost et al., 2013). Biochar can bind metal ions by attaching them to its oxygen-containing carboxyl, hydroxyl and phenolic surface functional groups (Uchimiya et al., 2011a). Hua et al. (2009) found that the carboxylic and lactonic groups increased by 2.4 fold and 1.5 fold, respectively, over 42 days of composting. Wiedner et al. (2015) found that biochar surface oxidation can be accelerated through composting, and the calculated O/C ratios increased from 0.13 in fresh biochar to 0.40 in composted biochars. These increased surface functional groups would promote the stabilization of toxic metals. Besides, the increased compost pile pH would also promote stabilization of toxic metals. Li et al. (2015) showed that, the DTPA-extractable Cu reductions were from 11.5% to 24.8% with the increase of pyrolysis temperatures from 300 to 900 °C that resulted in alkalinity in biochar. Several authors reported decrease in bioavailability of toxic metals. Chen et al. (2010) found that, the mobility of Cu and Zn decreased with increased biochar addition, and 9% biochar addition significantly decreased Cu and Zn leachability by 35% and 39%, respectively. Liu et al. (2017b) found that, the adding biochar was beneficial to remediate toxic metals, while adding in did not necessarily led to improved metal passivation. The bamboo charcoal addition effectively enhanced the removal of antibiotic resistance genes (ARGs), and temperature would be the main factor affecting the ARG profiles (Li et al., 2017a). However, it is worth noting that biochar-assisted composting not only stabilizes As and Hg in addition to Cu, Zn, Cd, Pb, and Ni within composting materials, although As and Hg have higher affinities to microbial methylation process than others. Zhai et al. (2021) noticed that biochar amendment increased the concentrations of monomethylarsonic acid (MMA) and dimethylarsinic acid (DMA) during mesophilic (first 10 days) and early thermophilic (days 11–15) phases, and promoted As volatilization during the maturing phase (days 60–80) of composting.

The abundances of As(V) reductase (arsC) and As(III) *S*-adenosyl-L-methionine methyltransferase (arsM) genes were higher in the biochar treatment than that of the control. Moreover, biochar amendment influenced the microbial communities by promoting As methylation and volatilization via Ensifer and Sphingobium carrying arsC genes, and Rhodopseudomonas and Pseudomonas carrying arsM genes. The biochar amended composting facilitated the regulation of As species, helped the methylation and volatilization of As during manure composting, and improved compost product application in agricultural soil.

8. Application methods for biochar in agricultural soils

Methods of biochar application into soils may potentially modify the characteristics and fate of biochar in soil environment (Ding et al., 2016; Joseph et al., 2010). Most of the studies involving biochar application, used techniques like spreading and incorporation in plow layer and deep banding that incorporated a significant amount of biochar (>5 t ha^{-1}) between 60 and 100 cm below surface (Bamminger et al., 2018; Blackwell et al., 2009). However, only a few studies explored how biochar application in different soil layers influences its fate and soil properties. Alternative methods of biochar application include spreading on soil surface, adding to furrow and even alternating layers of biochar in soil (Edenborn et al., 2015; Li et al., 2018). Previous studies have indicated that applying biochar to different soil layers (i.e., topsoil, subsoil and whole plow layer) could change the soil hydraulic properties and leaching process of inorganic nitrogen and DOC due to various structures of soil layers and changes in soil porosity and continuity (Castellini et al., 2015; Li et al., 2018). Incorporation of biochar to plow layer soil (0–20 cm) generally increased the surface soil sensitivity to air temperature (Ding et al., 2019), while adding biochar at depth might nullify this effect (He et al., 2016). The impact of biochar on soil properties can be a function of its application methods (Edenborn et al., 2015, Li et al., 2018). Plowing causes greater soil mechanical disturbance than other methods such as spreading and deep banding (Joseph et al., 2010; Li et al., 2016). Exposure of native OM to microbial attack due to mechanical disturbance of soil aggregate could facilitate faster decomposition of biochar. This suggests that the biochar could manipulate soil microbial health and carbon use efficiency, which are attributed to the biochar itself and its effect on soil environment (Zhou et al., 2017).

9. Application methods for compost and biochar-compost

Since biochar-compost is just a biochar-complemented compost, both can be applied using same the methods. Compost application methods depend on their end use such as agronomic, horticultural, forestry and non-agricultural uses. In addition, compost can be applied to improve organic matter, tilth and fertility of soil, urban landscaping, and cover landfills. On-farm applications are the cost-effective method for compost utilization to reduce the additional marketing management. The available nutrients in compost get released slowly upon application in soil by microbial activities. The annual N mineralization rate of compost varies between 8% and 12% of the total N, whereas P availability is only 25–40% of that of commercial fertilizers. Hence, only a fraction of N, P and K in compost will be utilized in the initial years of application in soil. Measurable benefits have been reported for 8 years or more after the initial application of compost in agricultural soil (Rynk et al., 1992).

9.1 Agriculture

Compost as a soil amendment is used to improve agricultural soils, restore disturbed soils, and establish and maintain landscape plantings. Low-quality compost could be used as a soil amendment, however, the presence of foreign matter should not be more than 5%, and the respiration rate should be less than 400 mg kg^{-1} h^{-1} (HKORC, 2021). Soil should be analyzed for nutrient and pH levels before applying compost. If a soil is acidic, addition of compost will raise soil pH. In contrast, compost contribution is limited in lowering pH of alkaline soils. These upper limits have been formulated to avoid the environmental risks caused by toxic substances in compost (D'Hose et al., 2014; Rynk et al., 1992).

For compost application in a field, rear-delivery or side-delivery manure spreaders can be used, whereas for top-dressing, broadcast cyclone-type applicators or modified rear delivery manure spreaders with brushes are used. In the agricultural fields, compost is commonly applied by incorporation, followed by plowing. However, granulated and pelleted composts (Fig. 5) can be applied at a lower rate in furrows along with seeds. For uniform application in soils, compost should have moisture content less than 40%.

Fig. 5 Compost in different forms. *Courtesy of Peats Group Ltd., South Australia.*

9.2 Horticulture and landscape

Composts can be used in horticulture and landscapes. pH and salt contents of compost may not be critical if its application rate is low. Caution should be taken when both compost and soil have high pH and salt contents. Physical contaminants and heavy metals contents should not exceed the environmental standards. Compost application rate should not exceed $112\,t\,ha^{-1}$ (dry weight basis) (Ozores-Hampton, 2021).

9.3 Containers and pots

In general, container-grown plants are high-value crops; therefore, composts to be used there should be of high quality as outlined in the compost standards. Compost should not contain more than 1% (dry weight basis) of foreign materials (glass, plastic and other particles) (BSI, 2018). In potting mixes, compost is applied at the rate of 20–33%, depending on plant species. A blend of 25% compost, 50% peat moss and 25% perlite (or vermiculite) is used for growing a wide variety of bedding plants. For growing herbaceous and woody ornamental plants in containers, equal volume parts of compost, coarse sand and peat moss are used. Compost usage in potting mixes eliminates requirement of additional trace elements, as these are supplied by compost itself. Regular watering schedule should be maintained in the first 2–3 weeks of planting; liquid fertilizer may be added to meet additional nutrient requirements (Chilosi et al., 2017).

9.4 Home gardens

For home gardening, high quality compost with less/no odor and low soluble salt concentrations should be used. Rate of compost application should be based on soil characteristics. Compost application should be done just before seeding or transplanting for optimum benefits (Rynk et al., 1992).

9.5 Non-food crops

Compost that does not meet the standards for food crop production, can be used to grow forest seedlings, ornamental plants, nursery stocks, and maintain public gardens and landscapes. Typically, harvesting nursery-grown plants along with the roots leads to loss of topsoil, which can be replaced with compost. Use of such compost in maintaining landscapes and public gardens reduces the usage of commercial fertilizers (Rynk et al., 1992).

10. Application of biochar to manage climate change, agricultural soil carbon and nitrogen

Soils are limited resources that deliver a range of ecosystem services supporting a wide range of biodiversity and storage of significant amount of carbon (Adhikari and Hartemink, 2016; FAO and ITPS, 2015; Flores-Rios et al., 2020; Scharlemann et al., 2014). SOC plays a major role in processes governing ecosystem services delivery relevant to agricultural production and climate change mitigation activities (Adhikari and Hartemink, 2016; Flores-Rios et al., 2020). Globally more than 16 million km^2 (~13% of the) land area is occupied by agriculture for cropping (FAO and ITPS, 2015), a land use type that evolved more than 10,000 years BC (Sanderman et al., 2017). Over millennia, increasing demand for agricultural products has been met by expanding cropland at the expense of natural land (e.g., forest, grassland, peatland), which accelerated after 1800 AD (Amelung et al., 2020; Sanderman et al., 2017). The changes in land use type and agricultural land use have affected the soil's capacity to store organic carbon, which induced a net loss of 75 Pg SOC compared to the historic (10,000 BC) SOC stocks (Duarte-Guardia et al., 2020; Sanderman et al., 2017).

In addition to CO_2 (Duarte-Guardia et al., 2020, Sanderman et al., 2017), the agricultural soils are also the major sources of non-CO_2 emissions (Edenhofer et al., 2014). Although most of the non-CO_2 emissions are related to enteric fermentation mainly from ruminants and manure management, agricultural soils (incl. synthetic fertilizers) and paddy soils also contributed 39–49% emissions (Edenhofer et al., 2014). Biochar has been proposed as one of the options to mitigate climate change (Amelung et al., 2020; Edenhofer et al., 2014; Saskia et al., 2019). Evidently, the majority of biochar carbon has potential to (i) remain stable in soil for centuries (Wang et al., 2016a), (ii) stabilize SOC (Abdelrahman et al., 2018; Borchard et al., 2014a), (iii) reduce emissions of non-CO_2 trace gases (CH_4, N_2O) and leaching of N (Borchard et al., 2019; Jeffery et al., 2016; Liu et al., 2018b), (iv) enhance plant productivity, particularly of highly weathered soils (Amelung et al., 2020; Schmidt et al., 2021), and (v) stabilize plant production particularly under water shortage (Kerré et al., 2017). Biochar can increase emission of NH_3 from soils (Liu et al., 2018b), with the potential to accelerate climate (Emmerling et al., 2020) and immobilize soil N (Borchard et al., 2014b; Liu et al., 2018b).

11. Application of biochar or compost to improve soil structure

Soil minerals and OM bind to form "peds" which then aggregate to give rise to soil structure. The macro-and micropores between the peds carry water and air, and hence soil structure is fundamental to providing congenial environments for plant growth. Impeded drainage, compaction and erosion are soil degradations that are directly related to soil structure. Poor soil structure affects crop yield and ecosystem quality as it may easily lose soil materials and nutrients. Soil aggregates are formed by minerals particles by a combination of electrostatic interactions and encrusted OM. Microaggregates can ($<250\,\mu m$) better withstand mechanical and physicochemical stresses than macroaggregates (0.25–2 mm) (Wilpiszeski et al., 2019). Vegetation types, soil fauna and microorganisms, ionic chemicals, OM contents and types and mineral materials are the key factors that control soil structural formation. Biochar and compost would shape soil structure formation by influencing these factors.

Application of biochar or compost for improving soil structure is widely reported in the literature. Biochar has higher porosity and larger surface area than compost, enabling biochar to impart stability to soil aggregates and prevent soil structural degradation (Mataix-Solera et al., 2011). In a 6-year rice-wheat rotation under field conditions, 87–93% increase in 250–2000 μm soil aggregates and 43–48% increase in aggregate stability crop yields after a application of biochar (20 and 40 $t\,ha^{-1}$) to the rice soil (Zhang et al., 2020a). Similarly, successive application of biochar (22.5 $t\,ha^{-1}$) to a red soil over 5 years in wheat-millet rotation under pot cultivation conditions improved the aggregate stability and intra-aggregate pore structure, resulting in an increased water retention capacity of the erosion-prone soil (He et al., 2020). A meta-analysis by Islam et al. (2021) involving 119 published reports indicated that biochar was able to improve soil aggregation by $\sim$17% across various biochar and soil types and experimental conditions, where high application rate gave better soil aggregation, biochar derived from wood at more than 600 °C showed best results, and loamy soils led to better results than sandy soils. The increased OM content of soil directly introduced through biochar and/or enhanced accumulation of OC in soil due to biochar's nucleation and negative priming effect (reduced OC mineralization) and multivalent cation (e.g., Ca^{2+}, Mg^{2+}, $Fe^{2/3+}$) inputs via biochar addition, are the key drivers that bind soil particles together into aggregates, where organic molecules and cations act as the cementing agent (Dungait et al., 2012; Weng et al., 2017; Zhang et al., 2020b).

Biochar is also reported to increase soil pore structure and shear strength, and decrease soil swelling in clayey soils (Lu et al., 2014). Biochar was reported to increase plastic index and reduce tensile strength and cohesion, meaning that it would effectively increase the soil available water content and extend the soil water range needed for growing various crops (Yang and Lu, 2021). In addition, biochar can ameliorate soil compaction by decreasing the bulk density and increasing soil permeability (Burrell et al., 2016; Peake et al., 2014). For example, Menon et al. (2018) demonstrated in a short-term experimental study that, biochar improved dry density, void ratio, and compression index of loamy soils with or without humus. Various mechanisms have been proposed by which biochar might mitigate soil compaction e.g., increased binding forces (cohesion) between particles and within soil aggregates, enhanced elasticity, reduced bulk density, increased growth of roots and fungal hyphae and organic coating on mineral particles reducing friction among particles (Menon et al., 2018; Soane, 1990).

The overall improvement in soil structural properties such as decreased bulk density and increased aggregate stability improve soil's capacity for water and nutrient retention (Ding et al., 2016). However, the surface area of biochar was found to decline during the co-composting process due to the clogging of microspores by adsorption of compost-derived materials, particularly the leachate (Prost et al., 2013). A study under field conditions in a temperate agricultural soil showed that, biochar increased OC storage by 56–62%, 32–47% and 29–32% in the 2–0.25 mm, 0.25–0.053 mm and <0.053 mm particulate organic matter (POM) fractions of soil, respectively, but compost increased OC storage only in the >0.053 mm POM fraction (Cooper et al., 2020). The interaction of biochar with composting substrate might enhance the nutrient retention capacity of the final product whilst it could alter the surface properties of biochar. Consequently, co-composted biochar is likely to influence soil structure property differently than pristine biochar.

12. Application rate of biochar-compost

Application rates of co-composted biochar-compost influence plant growth and yield. Several studies have shown both positive and negative impacts of co-composted biochar-compost on plant growth, soil microorganisms and GHG emission. Wide range of applications rates have been used ranging from 10 t ha^{-1} to 150 t ha^{-1}. The application rate of <20 t ha^{-1} and > 30 t ha^{-1} have shown significant plant growth benefits (Wang et al., 2019). Lashari et al. (2013) showed that, the co-composted biochar-compost application rate of 12 t ha^{-1} in moderately salt-stressed soil increased wheat yield by 38% compared to fertilizer. In another study, 2% of co-composted biochar-compost showed an increase in Chenopodium quinoa plant biomass by 305% due to the slow release of captured NO_3^- from biochar (Kammann et al., 2015). In contrast, von Glisczynski et al. (2016) recommended not to use co-composted biochar-compost in fertile soil as 3 kg co-composted biochar-compost per tree did not improve plant yield in clayey-silty soil. These studies indicate that the application of co-composted biochar-compost depends on various factors such as rate of biochar in co-composted biochar-compost, its application rate, soil and plant type and climatic conditions. Hence, preliminary studies should be carried out before considering application of co-composted biochar-compost in a large land. Majority of these studies have been less than 1 year. Hence, further studies are required to understand the long-term effect of biochar-compost combination in soil and plant growth.

13. Remediation of soils containing toxic elements

13.1 Trace element contamination in soils

The term "trace elements" encompasses the metals and metalloids in soils at levels below 0.1% or plant or animal tissues below 0.01% (Adriano, 2001). The trace elements B, As, Cu, Co, Fe, Mn, Mo, Zn, Cr, F, Ni, Se, Sn and V are essential to organisms, while Cd, Hg and Pb are non-essential (Adriano et al., 2004). Trace elements present in excess may become toxic to biota (Adriano, 2001; Bolan and Duraisamy, 2003). Soil metal and metalloid contamination have become a worldwide environmental issue. Globally, 5 million sites cover 20 million ha of land contaminated by heavy metal(loid)s (Liu et al., 2018a). Prevalence of heavy metals in developed areas has increased due to anthropogenic activities primarily associated with fertilizer use in agriculture, metal mining, and manufacturing by metallurgy, fossil fuel use, and military operations (DalCorso et al., 2019). The presence of excess metals may cause irreversible soil degradation, affecting its physical, chemical and biological properties, limiting plant establishment, and eventually damage human health via the food chain (Jaishankar et al., 2014).

Heavy metals are not biodegradable, hence, persist for a long period in soils and keep causing adverse effects on ecosystem functions when they are present at higher than background level (Adriano et al., 2004; Aponte et al., 2020). Engineering methods involving mechanical approaches have been traditionally used to remediate metal contaminated soils (Saha et al., 2017). In situ environment-friendly techniques are being pursued in recent years (Saha et al., 2017). Biochar and compost are two examples of such an approach (Bolan et al., 2014; Saha et al., 2017).

13.2 Remediation using biochar

Biochar is well-known to play a significant role in carbon and nutrient cycling, but it can also profoundly influence potentially toxic element (PTE) dynamics in soils. Biochar has gained significant importance for its ability in reducing PTE's mobility and availability in soils (Campos and Rosa, 2020; Chen et al., 2018; Xu et al., 2020). Biochar can be an effective adsorbent and repository of PTE due to having a large specific surface area, porous structure and charge density (Nie et al., 2018; Pan et al., 2021; Riedel et al., 2014). Many studies showed decrease of mobility of metal(loid)s such as Zn, Cd, Cr, As and Pb in biochar-amended soils (Lwin et al., 2018; Pan et al., 2021). In contrast, an increase in Cu, Cr and As mobility was observed in soils due to increased dissolved organic matter (DOM) concentration following biochar application (Alaboudi et al., 2019; Liang et al., 2017; Nie et al., 2018).

Many studies demonstrated BC's capability to serve as an environmental friendly sorbent that promotes crops/plantation in metal-polluted soil through (a) reducing metal bioavailability acting as a chelating agent, (b) providing nutrients to plants, (c) increasing cation exchange capacity of soils, (d) enhancing water holding capacity, (e) neutralizing soil acidity, and (f) creating better conditions for growth of soil microorganisms (Ahmad et al., 2014; Edenborn et al., 2015; Meier et al., 2011; Park et al., 2011). Biochar has below remarkable properties that make them suitable for remediation of metal contaminated soils.

1. High internal surface area rich in oxygen-containing functional groups (—OH, —COOH, —C=O and C=N) that enables the creation of surface complexes between cations (e.g., Cd^{+}, Cu^{2+}, Zn^{2+} etc.) and biochar's inner structure (Uchimiya et al., 2011b).
2. A negative charge (negative zeta potential), which allow ion exchange to the internal biochar surface from the soil solution thus promoting metal immobilization into its structure (Mukherjee et al., 2011). This negative charge can also increase soil pH after biochar application due to the

attraction of hydrogen ions from the soil solution (Sizmur et al., 2016). A higher soil pH serves to further increase the sorption of metals from soil because of the deprotonation of pH-dependent cation exchange sites on soil surfaces (Rees et al., 2014).

3. A high resistance to degradation, which allows only a prolonged carbon degradation with predicted C half-lives ranging from 10^2 to 10^3 years (Zimmerman, 2010). Thus, given the timescale (~10 to 100 years) of most soil remediation projects, biochar can be considered an inert material. An advantage of this recalcitrance is the potential to immobilize metals for an extended period (Moore et al., 2018).
4. Minerals such as phosphates, carbonates, sulfides, oxides can cause metal precipitation (Bolan et al., 2014; McGowen et al., 2001; Raskin and Ensley, 2000). Because the precipitates are rather insoluble (metal salts, especially at high pH), this mechanism can considerably contribute to the immobilization of metals (Cao et al., 2009). Moreover, the high ash content of the biochar gives it generally an alkaline pH value promoting a "liming effect" which enhances metals precipitation in soils. Some biochars contain a considerable amount of mineral ash component (e.g., up to 45% in chicken litter and rendering wastes biochars, and even 85% in bone-derived biochars (Amonette and Joseph, 2009)).

Just one type of biochar will not be suitable for remediation of all metal contamination cases (Bolan et al., 2014). Various properties of different BCs will determine their efficacy. Pyrolysis condition and feedstock type play a major role in influencing sorption behavior of BC (Kookana et al., 2011; Joseph et al., 2010; Thomas et al., 2020). Some studies found that metal-availability decreased after BC application; however, other investigations revealed increased mobility of metals especially in multi-metal contaminated soils (Table 3). Therefore, further investigation is needed to determine biochar role in multi-contaminated soils.

13.3 Remediation using compost and organic amendments

The application of compost, mainly through supply of organic acids, improves soil's micronutrient availability (e.g., Cu, Fe, Zn, Mn) through mobilization of elements from insoluble oxide compounds (Deb et al., 2009). However, organic manures and composts made from them, can potentially mobilize or immobilize metal(loid)s in soils depending on the OM and soil types and surrounding chemical conditions (Bolan et al., 2014). Numerous studies presented the role of organic amendments as a

Table 3 Effect of biochar on metal remediation in contaminated soil.

Feedstock	Pyrolysis temperature	Dose applied	Metal	Effect	Reference
Chicken manure	500 °C	5% (w/w[a])	Cu	The biochar addition reduced the Cu exchangeable fraction. Produced and increased pH and CEC. Enhance the soil microbial activities and induced changes in bacterial and fungal communities of soil	Meier et al. (2017), Moore et al. (2018)
Rice straw	500 °C	5% (w/w)	Pb, Cu, Pb and Zn	Biochar increased the pH, EC and CEC, producing a reduction of extractable Cu and Zn of 97% and 62%, respectively	Yang et al. (2016)
Rice straw and sewage sludge biochar	500 °C	0% and 2.5% (w/w)	Cd and As	Biochar amendments reduced concentrations of available Cd, and increased available arsenic concentrations in soils	Wang et al. (2020a)
				Rice straw biochar had a stronger impact on bacterial communities than sewage sludge biochar	
Water hyacinth	450 °C	0%, 2% and 5% (w/w)	Cd, Pb and As	Biochar promoted Cd immobilization	Yin et al. (2016)
				BC application increased soil Pb leachability upon acid exposure	
				Higher KH_2PO_4-extractable As was obtained by addition	
Wheat straw	350–550 °C	0, 10, 20, and 40 $t\,ha^{-1}$	Cd and Pb	Biochar significantly increased soil pH, organic matter and immobilized soil Cd and Pb	Bian et al. (2014)
				Biochar consistently reduced rice Cd and Pb content in three years through metal immobilization due to precipitation and surface adsorption	

Rice straw	500 °C	3% (w/w)	As	Biochar amendment enhanced As release from paddy soil under anaerobic condition, due to enhanced anaerobic dissolution of Fe oxides via the promotion of Fe reducing bacteria and the reduction of AsV to AsIII	Wang et al. (2017b)
Bamboo	750 °C	2% (w/w)	Cd and Pb	The biochar enhanced soil pH and organic carbon contributing to a decrease in the available concentration of metals by reducing metal mobility and bonding metals into more stable fractions	Xu et al. (2016)
Bamboo and rice straw	750 °C	0%, 1% and 5% (w/w)	Cd, Cu, Pb and Zn	Bamboo and rice straw BCs reduced soil extractable metal concentrations	Lu et al. (2017)
				Rice straw biochar was more effective in immobilizing soil heavy metals	
				Metals were mainly bound to organic matter in the biochar-treated soil	
				Rice straw biochar reduced metal bioavailability in the order: Zn > Pb > Cu > Cd	
Peanut shell	350–500 °C	0, 1, 2 and 5% (w/w)	Pb and Zn	Peanut shell biochar application increased soil pH, CEC, and DOC	Chao et al. (2018)
				Biochar treatment reduced the soil bioavailability of Pb and Zn	
				Biochar at a high application rate were more effectively immobilize Pb and Zn in soil	
kiwi pruning branches	550 °C	0, 1, 2 and 4% (w/w)	Cd, Zn and Pb	The addition of biochar increased the soil pH, cation exchange capacity, organic matter, and enzymatic activities (dehydrogenase, urease, and sucrase). The effects were more marked at the highest dose of biochar added	Ren et al. (2020)

Continued

Table 3 Effect of biochar on metal remediation in contaminated soil.—cont'd

Feedstock	Pyrolysis temperature	Dose applied	Metal	Effect	Reference
Chicken manure	Noninformed	0% and 5% (w/w)	Cr and As	Biochar addition increased reduction of Cr(VI) and As(V) in soils	Choppala et al. (2016)
				Availability of reduced As species increased but decreased for Cr	
				Microbial activity decreased in As contaminated soils after biochar application	
Olive mill	400–450 °C	0%, 5%, 10%, and 15% (w/w)	Zn, Cd, Pb	With increasing, biochar application rate and equilibration period, $Ca(NO_3)_2$ exchangeable metals decreased, and growth of plants improved; leaf metal contents reduced, the activities of antioxidative stress enzymes decreased, and soluble protein contents increased. Soil microbial activity, richness, and diversity were augmented	Hmid et al. (2015)
Hardwood	450 °C		As, Cd, Cu, Zn	Reduction in Cd in soil pore water by 10-folds; Zn concentrations reduced 300- and 45-folds, respectively, in column leaching tests	Beesley and Marmiroli (2011)
			As		
Orchard prune	500 °C	30% vol	As	Arsenic was mobilized into soil pore water rapidly following biochar addition	Beesley et al. (2013)
				Transfer of As from soil to roots and aerial plant parts was reduced by biochar addition. Biochar did not affect biomass yield and reduced germination without nutrient supplements	

[a] w/w, wet weight.

source of PTE, while a relatively limited number of studies reported organic amendments' role as a PTE immobilization sink in soils (Liu et al., 2018c; Mahar et al., 2015). For example, treated sewage sludge, also known as "designer sludge," was found to be a promising sink for PTE stabilization in soils (Ren et al., 2017). Adsorption, complexation and redox reactions are involved in PTE stabilization or immobilization due to organic amendment application (Lwin et al., 2018). Contrary to treated sewage sludge, the application of green compost showed As mobility in soils, whereas biosolid compost improved soil As adsorption (Kunhikrishnan et al., 2017). Application of organic amendments could also alter Cr and Se availability through oxidation-reduction reactions in some soils (Li et al., 2017b; Liu et al., 2020a). For instance, reduction of Cr(IV) to less mobile Cr(III) was found due to the addition of cattle manure in soil (Shahid et al., 2017). High fulvic and humic acid contents in organic amendments and composts may contribute to PTE stabilization or immobilization by supplying effective adsorption and/or complexation sites (Kwiatkowska-Malina, 2018; Perelomov et al., 2021). However, the effectiveness of composts and organic amendments for PTE stabilization or immobilization in soils depends on characteristics of PTE, amendments and soil types (Lwin et al., 2018; Mehmood et al., 2017).

13.4 Co-compost of biochar and organics

Co-composting biochar with organics could be very interesting in respect to PTE mobilization or immobilization in soils. During co-composting, the adsorption sites and surface area of biochar (and surface area) are expected to reduce due to clogging of micropores by the compost-derived materials (Prost et al., 2013; Sanchez-Monedero et al., 2018). However, retention of DOM on biochar due to co-composting and biochar-induced pH rise, may lead to increased adsorption of PTE in soils, with the effects more prominently observable in light-textured than heavy-textured soils (Gondek et al., 2018; Riedel et al., 2015). Even municipal solid waste (MSW), which is usually considered a biomass rich in PTE, could be co-composted with biochar for soil application, to reduce PTE availability to plants. Mounissamy et al. (2021) demonstrated that application of a co-composted MSW-biochar product at 10% rate to a sandy loam soil reduced Cu, Cd, Pb, Cr, Ni and Zn uptake in spinach leaves by 20.6%, 29%, 36%, 42%, 41.5% and 41.2%, respectively, compared to MSW treatment. It is thus expected that a decreased mobility (leaching loss) and bioavailability of PTE is caused by enhanced retention of PTE by soil-applied co-composted biochar products, which warrants future studies at pilot and field scales.

14. Remediation of soils containing organic contaminants

14.1 Organic contaminants in biochar and compost

Biochars may potentially contain a wide range of organic contaminants, such as PAHs, PBDEs, PFAS, pesticides, PPCPs, depending on the source of biomass (feedstock) from which biochar is produced, as well as the process and conditions under which it is made. For example, the biochars produced from sewage sludge may be contaminated by the organic (and inorganic) contaminants that were initially present in waste stream, and hence in the feedstock.

Sewage sludge (biosolids) is known to contain industrial organic contaminants, such as petroleum hydrocarbons, flame retardants (PBDEs) as well as common household chemicals (e.g., PPCPs and pesticides), depending on the catchment of the waste stream (Clarke and Smith, 2011, McGrath et al., 2020). Furthermore, in recent years, the persistent and bio-accumulative chemicals, namely per- and polyfluoroalkyl substances (PFAS), which have a range of applications ranging from fire-fighting to industrial use (e.g., in metal plating, paper and textile, photographic and electronic industries), have also been detected in wastewaters and biosolids (Moodie et al., 2021). Therefore, depending on the source of biomass, both the biochar as well as the compost that is used as a soil amendment could potentially serve as a source of contamination. However, such contamination can be easily controlled, and indeed regulatory controls are already in place in many countries/regions (e.g., Australia, EU). For example, in Australia, such contamination is controlled through the compost quality standard AS-4454 (Standards Australia Limited, 2012) and NASAA Certified Organic program (NCO, 2021). Some European Union countries, UK and the US, also have similar controls in place.

Chemical contamination may arise during production of biochar, depending on production process and the conditions associated with it. Therefore, even when a feedstock is uncontaminated (e.g., natural vegetation), there remains a possibility of formation of some toxic organic compounds, such as Polyaromatic hydrocarbons (PAHs) and Dioxins, during the biochar production process itself. PAH is a prominent group of persistent chemicals (with two or more aromatic rings) heavily regulated internationally under the Stockholm Convention of Persistent Organic Pollutants (POPs). PAHs are produced due to incomplete combustion of biomass or fossil fuel. Since production of biochar involves a pyrolytic process (e.g., slow or fast pyrolysis), some PAHs may inevitably be formed during the production process (Hale et al., 2012). However, this aspect has already received considerable attention. The factors affecting the formation of PAHs during biochar productions and their fate, and potential impacts on receiving soils because of biochar applications, have already been investigated (Mayer et al., 2016; Wang et al., 2017a). In a review on PAHs in biochar, Wang et al. (2017a) highlighted that, pyrolysis method, residence time and temperature are important factors that determine the nature of and extent of PAH yields during biochar production. They concluded that the fast pyrolysis with shorter residence times results in higher concentrations of PAHs in biochar, and the formation of low molecular weight PAHs are favored at lower temperature and vice-versa.

14.2 Remediation using biochar

From the environmental impact perspective, biochars may have both positive and negative impacts on the receiving soil environment, depending on the nature of biochar and its use. As mentioned earlier, in some cases, biochar may serve as a source of contamination (Hale et al., 2012; Wang et al., 2017a). Hale et al. (2012) quantified the total load of PAHs and Dioxins on 50 biochars (produced under a range of pyrolysis conditions) and also assessed their bioavailability. They found that, in all of these biochars (except one biochar produced by gasification process), the total and bioavailable PAHs concentrations were below their relevant environmental quality standards. Even when the PAHs are present in biochars, they may not be available as the PAHs formed during the production process may become completely occluded in the biochar structure and thus not bioavailable (Hale et al., 2012).

However, more importantly, biochars may also play a desirable role in mitigating the risk associated with contaminants in the environment (Beesley et al., 2011). In fact, there is now a significant body of work in literature that shows that, due to their carbonaceous nature and high specific surface areas, biochar can strongly bind organic contaminants and thus reducing their bioavailability (see review by Kookana et al. (2011)) and thereby minimizing their ecological and human health risk (Beesley et al., 2011; Macdonald et al., 2015).

Like other carbonaceous materials with high surface areas, such as Activated Carbon (AC), biochars have been shown to have a high affinity for organic contaminants. Due to increased binding affinity for organic pollutants, biochars have a potential to significantly reduce the bioavailability of organic chemicals that their plant uptake from biochar-amended soils than that from the unamended soils. This was observed in the case of uptake of pesticides (Nag et al., 2011; Yu et al., 2006), and other chemicals (Williams et al., 2015). Biochars can also act as catalysts promoting the catalytical transformation of organic compounds containing nitro-aromatic functional groups (such as in explosives), resulting in their rapid degradation (Oh et al., 2013). Such catalytic behaviors may be compound specific.

The effectiveness of biochars in sorbing or removing a range of organic contaminants from water or reducing their bioavailability in biochar amended soils have been summarized in Table 4. For example, Kearns et al. (2014) demonstrated that, the sorption potential of some biochars for a highly soluble herbicide 2,4-D, was comparable to those of AC (Table 4). However, these biochars were produced at high temperatures (600–700 °C) and had high specific surface areas (>400 m^2/g). Similarly, other workers have also reported high sorption for other contaminants, including PPCPs and PFAS (Chen et al., 2008; Jung et al., 2013; Kookana et al., 2011; Teixidó et al., 2011).

The above shows that biochar, when used as a soil amendment, can help bind contaminants such that their bioavailability is reduced. Thus, the ecological and human health impact is minimized. Similarly, biochars can also be used as sorbents to remove contaminants from the water phase, such as runoff water or wastewater.

14.3 Remediation using composting process and compost

A PAH contaminated soil may contain suitable microbes that can degrade this contaminant. PAH bioremediation is performed by soil endogenous/introduced bacteria (Samanta et al., 2002; Watanabe et al., 2002; Yu et al., 2011). Fungal substrate from commercial mushroom production was also found effective in bioremediation of soil PAH (Eggen, 1999). For proliferation, microbes need carbon and nutrients: they get some of these in available (soluble) form in soil and/or organic matrix; and for the rest, they degrade labile/calcitrant organic materials/contaminants available in soils and composting materials. In a contaminated soil-compost feedstock mix, microbes can use/degrade both PAH and composting materials as a source of nutrients. Spent mushroom compost amendment was found to add enzymes, microbes, and nutrients for the microbes involved in degradation in the PAHs contaminated soil (Lau et al., 2003). A soil with a higher level of contamination gets remediated at a higher rate, compared to one

Table 4 The effectiveness of biochars in sorbing or removing a range of organic contaminants from water or reducing their bioavailability in biochar amended soils.

Chemical	Biochar type	Characteristics/ process	Parameter value	Reference material comparison	Reference
PAHs (16 USEPA priority compounds)	49 different biochars produced under different conditions	Total and bioavailable concentrations	Total concentration: 0.07 to 3.27 $\mu g\, g^{-1}$ Bioavailable concentration: 0.17–10.0 $ng\, L^{-1}$	Cattle manure—Total concentration: 87 to 300 $\mu g\, kg^{-1}$ Compost—Total concentration: 0.8 to 2.7 $\mu g\, g^{-1}$	Hale et al. (2012)
Total dioxins	Eight biochars produced under different conditions	Total and bioavailable concentrations	Total concentration: 0.005–1.2 $pg\, g^{-1}$ TEQ		Hale et al. (2012)
2,4-D	Wood chips, Bamboo, Corn cobs, Rice straw (350–700 °C)	Sorption	$K_F = 0.31–54.6\, L\, kg^{-1}$	Various AC $K_F = 8.5–78.8\, L\, kg^{-1}$	Kearns et al. (2014)
Sulfamethoxazole	Eucalyptus (600 °C)	Sorption	$K_F = 81{,}300–1.1 \times 10^6\, L\, kg^{-1}$	–	Teixidó et al. (2011)
17-Ethinylestradiol (sorption)	Wheat straw (400 °C) Poultry litter (400 °C)	Sorption	$K_F = 29.5\, L\, kg^{-1}$ $K_F = 8.3\, L\, kg^{-1}$	–	Sun et al. (2011)

Continued

Table 4 The effectiveness of biochars in sorbing or removing a range of organic contaminants from water or reducing their bioavailability in biochar amended soils.—cont'd

Chemical	Biochar type	Characteristics/ process	Parameter value	Reference material comparison	Reference
Dinitrobenzene	Pine needle (100–700 °C)	Sorption	$K_d = 0–110\,L\,kg^{-1}$	Particulate AC $K_d = 22\,L\,kg^{-1}$	Chen et al. (2008)
Trifluralin (reduction rate)	Poultry litter biosolid	Decomposition/ transformation	95% removal (120 min) 92% removal (120 min)	Granulated AC 90% removal (120 min)	Oh et al. (2013)
Carbofuran chlorpyrifos	*Eucalyptus* spp. wood chips at 850 °C	Bioavailability (plant concentration)	Soil + 1% biochar $1.8\,mg\,kg^{-1}$ $0.8\,mg\,kg^{-1}$	Soil only $14.4\,mg\,kg^{-1}$ $14.1\,mg\,kg^{-1}$	Yu et al. (2009)
Per- and poly-fluroalkyl substances (PFAS) PFOA PFOS	Pine derived biochar produced at 750 °C	Removal from water by biochar amended sandy clay loam	Soil + 5% biochar 18.8% removal 70.6% removal	Sandy clay loam 1.6% removal 0.8% removal	Askeland et al. (2020)

with a lower level of contamination (Jørgensen et al., 2000; Sayara et al., 2010; Wu et al., 2013; Zappi et al., 1996). It happens so, as microbes prefer abundantly present degradable nutrients instead of PAHs present at low concentrations.

A PAH contaminated soil can be bioremediated ex-situ by mixing it with fresh organic wastes or stable compost in a composting system that stimulates microbial populations (Sayara et al., 2011; Zhang et al., 2011). Performance of microbial decomposition of PAHs depends on various composting process factors including nutrients (N, P) (Lukić et al., 2016), organic carbon (Lukić et al., 2016), temperature (Lukić et al., 2016) and water solubility/bioavailability of PAH (Gemmell et al., 2016).

Humic acid, a by-product available in composting materials, can enhance bioavailability of PAHs (Tang et al., 2012), thereby increasing their decomposition rate (Sayara et al., 2011). Composting materials with higher stability (by virtue of its higher amount of humic substances, perform better in bioremediating soil PAH) than that of lesser stable compost or uncomposted organic amendments (Plaza et al., 2009; Sayara et al., 2010, 2011). Humic substances in composts (or composting materials) act as biosurfactants that desorb organic particle-bound PAH by lowering surface tension, developing micelles and integrating PAHs in micelles cores (Janzen et al., 1996; Montoneri et al., 2009; Quagliotto et al., 2006). Microbial decomposition performance of PAH increases with their solubility/bioavailability that usually decreases with their increasing molecular weight or number of aromatic rings that makes it more resistant (Peng et al., 2008; Wischmann and Steinhart, 1997). Han et al. (2017) observed this in the soil amended with organic wastes, particularly for those with a low C:N ratio (or high amount of N); and Lukić et al. (2016) noticed this in the in-vessel composting material containing contaminated soil (soil:feedstocks ratio 5:1). Kobayashi et al. (2009) observed that water-extractable organic matter (WEOM) from cow, chicken and pig manure compost increases apparent solubility of PAHs and their degradation; mainly high molecular weight WEOM (>1000 Da) contributed to the solubility and enhanced decomposition. PAH can be bioremediated in a composting system at both mesophilic and thermophilic phases/temperatures with the former condition being more efficient (Amir et al., 2005; Houot et al., 2012; Purnomo et al., 2010; Sayara et al., 2010; Viamajala et al., 2007).

A drawback of composting approach is the risk of significantly incomplete remediation of organic contaminants, thereby ending up with a larger contaminated pile of compost! Usually, around two-third of an organic contaminant can be degraded rapidly (2 months) in a composting system; the residual (and aged) contaminants may degrade very slowly (Jørgensen et al., 2000) due to being inherently recalcitrant or locked up in soil/organic particle pores (Huesemann, 1997). Therefore, as a strategy, contaminated soil must not be added to a composting pile in excess to avoid residual PAH at a toxic level. Where there is a chance of incomplete degradation, applying a mixture of biochar and compost can be an attractive strategy (see the next section "Remediation using biochar-compost"). Wang et al. (2011) found that feedstock:contaminated soil ratio of 2:1 was suitable where C:N ratio of the mix was higher (15,1) than the ideal rate (30,1).

A PAH contaminated soil can also be remediated in situ by using compost (Wu et al., 2013) by microbes like bacteria (Han et al., 2017). Mushroom compost and spent mushroom compost amendments were found to remediate contaminated soils (Eggen, 1999; Guerin, 1999; Lau et al., 2003; Trejo-Hernandez et al., 2001). Wu et al. (2013) noticed that, high PAH removal was observed at high initial soil PAH concentration. In line with this observation, Sayara et al. (2010) found that the microbial activity was insufficient in municipal compost amended soil with low initial PAH concentration. Type of compost (from green waste and green waste + meat) and rate of application of compost to soil (250 and 750 t ha^{-1}) appeared to have a less significant effect than soil type and compost contact time (Wu et al., 2013).

14.4 Remediation using biochar-compost

Combined application of biochar and compost can immobilize by sorbing soil PAHs, while also facilitate degradation of bioavailable fractions. Sigmund et al. (2018) found that, compost (10%) amended in the contaminated soil increases sorption of PAH by 10 folds, and up to 100 folds for the combination of compost (10%) and biochar (5%). In the soil-compost mixture, the PAH degradation rate also increased by two folds due to compost substrate over 120 days, probably due to introduction compost-originated additional microbial species. In contrast, degradation in the soil-compost-biochar mixture slowed down up to ten folds probably due to the additional sorption by biochar (see section "Remediation using biochar" above), although some degradation occurred. Beesley et al. (2010) suggested that, although not significant, compost reduced total and bioavailable PAH concentrations in the contaminated soil due to microbial proliferation facilitated by compost-derived organic matrix (Kästner and Mahro, 1996).

Like Sigmund et al. (2018), Beesley et al. (2010) also found that biochar and compost, when applied combinedly, is less effective in reducing total PAH or bioavailable PAH, as these amendments reduce PAH concentration by contradictory mechanisms: biochar makes a significant fraction of PAH bio-unavailable by sorption, while compost assists degradation of the bio-available PAH fraction by introducing organic matrix.

15. Future research

- Many studies have demonstrated that biochar addition could facilitate the O_2 transfer and prolong the duration of thermophilic phase of composting substrate, however, up to date, limited research has been conducted to evaluate the variations of O_2 concentration and heat distribution in compost piles with biochar addition. Better understanding of O_2 concentration and heat distribution will be helpful for biochar assisted composting technology development.
- Biochar additions benefit composting aeration, pH adjustment, and microbe populations and activities. While property of biochar is strongly depended on the feedstock, pyrolysis condition, and particle size.
- Many recent findings are based on lab-scale experiment utilizing very small composting reactors (9–100 L), where composting process is under laboratory control. Thus, more large-scale investigations are required under field conditions.
- Although numerous studies have proven that biochar amendment helps the reduction of antibiotics and antibiotic resistance genes in composting, however, the latter were not eliminated completely. The residual antibiotic resistance genes in composting will further spread and transfer in environment after agricultural use. Therefore, we need to find ways to reduce the resistance genes.
- After a remediation process, stabilized toxic metals continues to exist in compost products. Studies are required to address potential bioavailability of these metals in the future.
- Compost feedstocks can contain a wide range of particles (<2 mm up to >16 mm) (Chowdhury et al., 2014). Studies are required to determine optimal particle size of biochar for co-composting.
- Reports on the effect of co-composted biochar on soil structure specially under field conditions are scarcely available in the literature and warrant future investigations.

- The impact of biochar application methods on soil microorganisms and carbon use efficiency is a scientific question worth studying.
- Studies are needed to assess optimum incubation period for a mix of biochar and mature compost.

16. Salient points

- Organic wastes are abundant everywhere.
- Biochar-compost can be made from organic wastes.
- Biochar-compost can be developed by (i) co-composting of biochar with organic waste, (ii) incubation of biochar and a compost product, and (iii) mixing of biochar and a compost product just before application to soil.
- Biochar-compost may provide a wide range of benefits expected from both biochar and compost.
- Biochar-compost can be used as an amendment for improving agricultural soil quality, increasing crop production, and remediating contaminated soils.

Case Study

Making and field testing of a mineral-enhanced biochar, compost and biochar-compost applied at low application rates for increasing rice yield in Thai Nguyen, Viet Nam.

1. Introduction

Rice is a major crop for Vietnam and in the upland areas of Vietnam. Due to the increasing use of chemical fertilizer, increased use of mechanical plowing and harvesting and reduction in the use of dung soils have become more acidic and OM has decreased. Twenty-five percent of crop residues are now being burnt as there is a reduced need for animal feed resulting in an increase in air pollution and reduction in nutrient recycling (SNV, 2012). This study investigates the effect of biochar and biochar-compost on rice yield in paddy fields with a loamy sandy soil, in a suburb of Thai Nguyen City, Vietnam over 2 crop cycles.

Recent social survey and field trials carried out in Dinh Hoa and Ba Thuoc in two poor provinces of Thai Nguyen and Thanh Hoa indicated that households have sufficient residues including manure, straw, green waste, bamboo, and rice husks to produce approximately 1.5 t of biochar per year (Joseph et al., 2015) from available residues including manure, straw, green waste, bamboo, and rice husks. Households in this district soak wood and bamboo in a nutrient-rich pond for up to a year, and after drying, use the biomass for cooking food. Biochar is produced from these pretreated biomasses due to the lack of O_2 in the open fires and simple drum kilns. This biochar has a high content of minerals and macronutrients (Vinh et al., 2014). Field trials in these two districts resulted in increases in rice yields by 20–30% when either 500 kg ha^{-1} of a mixed-feedstock biochar and 9.5 t ha^{-1} of composted manure and NPK, or 2.5 t ha^{-1} of biochar and NPK were applied and compared with a control of only adding NPK (Hien and Vinh, 2012).

To determine the wider broader benefits to society from biomass and grain yield increase, and increased soil carbon storage, a 2-year research project was established near the city of Thai Nguyen to test the following hypotheses.

1. A significant increase in yields can be achieved by adding NPK fertilizer to biochar, made from mixed-feedstock, applied to sandy loam soils at an application rate of 1.5 t ha^{-1}.
2. Mixing 5% biochar with compost and applying at 10 t ha^{-1} along with NPK fertilizer every crop cycle, will result in an increase in yield and an increase in SOC above what was added through the biochar-compost mixture.
3. Addition of a small quantity of biochar during the composting of manure will lower greenhouse gas emissions from the soil.

This study is unique in using a mixed feedstock at application rates that are achievable using simple and low-cost equipment that can be built, operated, and maintained by village people.

2. Materials and methods

2.1 Study site

This study was undertaken in Thai Nguyen city, Thai Nguyen province, Viet Nam (21°34 275' North; 105°46 796' East). Rice has been cultivated in this field for at least 20 years. The climate in this area is humid tropical, the mean annual temperature is 23.6 °C and the mean annual precipitation is 1600 mm with max rainfall occurring between May and September (Year range and source: Thai Nguyen station).

2.2 Soil properties

The initial soil was acidic, loamy sand with a particle size distribution of 79.8%, 11.3%, and 8.9%, respectively, of sand, silt, and clay. Its bulk density was 1.34 g cm^3. The other soil properties are presented in Table 8. The CEC, total P, total N, total K, OC were low compared to the more fertile fluvisols found in Thai Nguyen.

2.3 Production of multi-feedstock biochar

Multi-feedstock biochar was produced from the pre-treated feedstock. Pre-treated bamboo and wood, soaked in a nutrient-rich pond followed by sun drying, was obtained from a farmer in Thai Nguyen city and added in equal proportions to dry rice husk and rice straw. Based on the work of Novak et al. (2009) and Nielsen et al. (2014), we suggest that mineral or nutrient enhanced multi-feedstock biochar would be more effective at increasing yields at low applications than biochar made from a single clean feedstock. Based on the soil analysis, a slurry consisting of 2% lime, 10% clay and 5% of buffalo manure and 5% rice straw ash was added to the feedstock of pre-treated bamboo and wood, rice husk and rice straw. It was expected that this mixture would both meet the soil constraints discussed above as well capable of being pyrolyzed at a temperature of around 450 °C with a biochar yield greater than 35%. Based on the work of Chia et al. (2014) and Joseph et al. (2013), clay was used to help increase the concentration of oxygen functional groups on the surface, capture labile organic molecules emitted during pyrolysis and minimize the loss of N and K compounds. The mineral coated biomass was dried, placed in layers in a top-lit updraft gasifier (TLUD), and then pyrolyzed (Vinh et al., 2014) at about 450 °C over 1.5–2 h (Fig. 6). The temperature of the biochar was controlled by spraying a fine mist of water into the reactor. At the end of pyrolysis, when the liberation of volatile gases had ceased, water was applied until the temperature dropped below 60 °C. After this, the biochar was cooled down by spraying water. Wet biochar was weighed; and its moisture and yield were calculated. Almost 37% of the total biomass was converted into biochar through this process.

2.3.1 Production of mineral-enriched biochar

A biochar-NPK composite was produced for the experiment. The procedure described by Joseph et al. (2013) was used to ensure that N, P and K were adsorbed into the biochar material. Biochar (20% dry weight)

Fig. 6 Production of biochar in the field using a top lit updraft gasifier (TLUD).

was mixed with NPK (ratio 15:10:10) in an airtight container for 30 days before application to the field site.

2.4 Production of compost and biochar-compost

The co-composting method (Vinh et al., 2014), developed to optimize plants yields, was utilized using soft plant residues with or without enhanced biochar. In the case of producing biochar-compost, for every t of manure there was 5% biochar, 1% Siam shrub (*Eupatorium odoratum* L.), and 0.5 kg of Effective Microorganism purchased from Japan (EMRO, www.emrojapan.com). Compost and biochar-compost were made in each season (i.e., trial). The composting was completed after 55–60 days in spring and 40–45 days in summer.

2.5 Experimental design and treatment

Rice seedlings (*Oryza sativa* L. indica, cv Khang Dan 18) from a nursery bed were transplanted to the experimental plot after 40 days in the spring and 20 days in the summer. Transplanting was performed with 2–3 seedlings per hill and 20 cm x 20 cm spacing between hills. Field trial plots (5 m x 6 m) were set up as per Complete Randomized Design with five treatments (Table 5) and three replicates. Conventional chemical fertilizer was

Table 5 Experimental treatments.

Treatment	Amendment per hectare
Control or CT	No amendment
CCF	Conventional chemical fertilizer
BCF	Conventional chemical fertilizer plus 1.5 t biochar
MCF	10 t composted manure plus conventional chemical fertilizer
MBCF	9.5 t composted manure and 0.5 t biochar plus conventional chemical fertilizer

applied at 100 kg N, 90 kg P_2O_5, 60 kg K_2O per ha in spring and 80 kg N, 60 kg P_2O_5, 60 kg K_2O per ha in summer as per the recommendation of Viet Nam Academy of Agricultural Science (VAAS). The amount of CCF applied was the same in all the treatments, except the control where no amendment or fertilizer was added. Water management (flooding and drying) was carried as per farm practice. There were five stages of water management in the spring ((a) irrigation for flooding with 3–5 cm of water; (b) drainage; (c) irrigation for flooding with 2–3 cm of water; (d) flooding at the surface with additional water; (e) drainage), and three stages in the summer (a, c, e).

2.6 Sample analysis

The properties of the biochar were measured as per Rajkovich et al. (2012) and soil analysis as per Van Zwieten et al. (2010). Statistical analyses of all data were performed using R statistics. The effects of different treatments were examined by comparing means using one-way ANOVA. When the differences among treatments were significant ($P < 0.05$), the means were compared using the Duncan ($P = 0.05$) post-hoc test.

3. Results

3.1 Mineral enhanced biochar properties

The mineral-enriched biochar, made from mixed biomass, had a high content of total organic carbon, nitrogen as well as calcium, magnesium, potassium and phosphorus (Table 6). The total carbon and the total nitrogen in biochar increased by 16% and 39.1%, respectively, after composting with buffalo manure for 45 days. The C/N ratio of the compost was reduced from

Table 6 Chemical properties of mineral enriched biochar.

Property	**Value**
Total organic carbon (%)	17
Total carbon (%)	17
Formic acid soluble P ($mg\,kg^{-1}$)	13,000
KCl extractable ammonium-N ($mg\,kg^{-1}$)	1.2
KCl extractable nitrate-N ($mg\,kg^{-1}$)	19
Exchangeable cations	
Calcium (cmol(+)kg^{-1})	6.3
Potassium (cmol(+)kg^{-1})	49
Magnesium (cmol(+)kg^{-1})	4.3
CEC (cmol(+)kg^{-1})	60
Acid extractable elements	
Cu ($mg\,kg^{-1}$)	54
Mg ($mg\,kg^{-1}$)	1.7
Mn ($mg\,kg^{-1}$)	680
Zn ($mg\,kg^{-1}$)	220

95.8 to 80 due to an increase in N in the composting process. The pH of the biochar after composting was increased from 9.87 to 11.15, total C from 47% to 54%, total N from 0.49% to 0.68%.

3.1.1 Effect of compost and biochar amendment on rice yield

The yields from all treatments in spring were higher than the ones in summer due to flooding caused by excessive rainfall. The results in Table 7 show that there was a significant increase in yield by 45% in spring and 74% in summer when 1.5 $t\,ha^{-1}$ of biochar was applied together with CCF fertilizer. In the spring, the yield of the BCF was 1 t greater than the CCF and in the summer, 1.11 $t\,ha^{-1}$ compared to the addition of CCF (Table 7). In the summer, the heavy rains and storms affected the growth of the rice plant during the flowering phase, which in turn lowered the yield (Table 7). However, the trend was the same in the plots with BCF, producing the highest yield in both the seasons. MCF and MBCF had similar yields in the spring, while there was no significant difference between these two treatments

Table 7 Yields over two seasons.

Treatment	Spring yield ($t\,ha^{-1}$)	Summer yield ($t\,ha^{-1}$)
Control	5.1a	2.2a
CCF	6.4b	2.71ab
BCF	7.4c	3.82c
MCF	6.8bc	3ab
MBCF	6.7bc	3.29ab

Note: Values followed by different letters are significantly different at $P < 0.05$ level.

and CCF treatment. The yields performed similarly in the summer. This result is different from similar treatments reported by Vinh et al. (2014) when trials were carried out in Plinthic Acrisol.

3.1.2 Soil property change

Table 8 illustrates the temporal change of soil properties. After the spring the pH.KCl, CEC, OC, total N, available P_2O_5 (P) and available K_2O (K) were significantly increased in BCF, MCF and MBCF treatments, where biochar and/or compost were/was applied, in comparison to the initial soil.

After both the trials, soil OC was higher in BCF, MCF, MBCF than in the control. However, there was no significant difference in OC in both trials between the three treatments: MCF, CCF and BCF. The highest content of OC was measured in the soil amended with MBCF in both seasons (1.56% in spring and 1.61% in summer). Soil total nitrogen increased in BCF, MCF, MBCF, compared to the control. Total nitrogen was lowest in CCF with 0.09% in both seasons. This parameter remained similar for the other treatments with slight changes from spring to summer (from 0.10% to 0.13%). However, there was no difference in yields between applying straight compost with CCF and compost biochar and CCF in both seasons.

The following significant changes in soil properties were observed for the spring.

1. The pH.KCl, CEC, OC, total nitrogen, NH_4^+—N, NO_3^-—N was significantly different between treatments.
2. The pH.KCl in treatment with adding biochar was significantly higher than the other treatments.
3. The CEC was increased significantly (~2–4 $cmol\,kg^{-1}$) in treatments with biochar and manure application.

Table 8 Soil properties after application of biochar and compost in two seasons.

			Treatment				
Analyte	**Season**	**Initial soil**	**Control**	**CCF**	**BCF**	**MCF**	**MBCF**
pH H_2O	Spring	6.27	6.11a	6.46a	6.61a	6.48a	6.5a
	Summer		6.4a	6.4a	6.4a	6.3a	6.7a
pH KCl	Spring	5.3	5.0b	5.4b	5.7a	5.5b	5.9a
	Summer		5.2c	5.3c	5.90ab	5.60b	6.2a
CEC (cmol kg^{-1})	Spring	10.3	10.90c	11.48c	12.15b	13.50ab	14.90a
	Summer		10.50c	11.70c	14.9b	14.1b	16.8a
OC (%)	Spring	1.23	1.15c	1.23c	1.35b	1.37b	1.56a
	Summer		1.09c	1.20b	1.45b	1.45b	1.61a
TN (%)	Spring	0.08	0.07b	0.09b	0.11a	0.11a	0.12a
	Summer		0.05b	0.09b	0.12a	0.10a	0.13a
P_2O_5 (%)	Spring	0.02	0.07a	0.09a	0.10a	0.09a	0.09a
	Summer		0.08a	0.09a	0.12b	0.09a	0.1a
K_2O (%)	Spring	0.03	0.06a	0.08a	0.07a	0.08a	0.09a
	Summer		0.07b	0.08b	0.12a	0.10b	0.13a
NH_4^+—N (mg 10^{-2} g soil)	Spring	125.3	197.5bc	272.8ab	221.3abc	399.5a	69.7c
	Summer		12.8c	41.6bc	101.2a	74.4ab	39.0bc
NO_3^-—N (mg 10^{-2} g soil)	Spring	58.7	82.6ab	116.6ab	55.6b	246.8ab	73.9b
	Summer		86.8a	62.5a	69.3a	44.1a	96.6a
Available P_2O_5 (mg 10^{-2} g soil)	Spring	22.2	7.85b	10.3b	12.2a	11.8a	12.9a
	Summer		7.8c	9.8c	13.5a	12.3b	13.4a
Available K_2O nmg 10^{-2} g soil)	Spring	12.2	7.9c	9.7cb	8.8cb	9.6cb	11.5a
	Summer		6.7c	9.5b	9.4b	9.8b	12.0a

Note: Values followed by different letters are significantly different at $P < 0.05$ level.

4. The organic carbon was highest in MBCF and significantly different with other treatments.
5. The soil ammonia and nitrate were significantly changed between treatments after rice harvest in each season.

The following changes in soil properties were observed for the summer.

1. The pH_{H2O}, total phosphate, total potassium, NO_3^- —N, did not change significantly.
2. The pH.KCl, CEC, OC, total nitrogen, NH_4^+ —N, available P, available K changed significantly between treatments.

1.1.1.1 Spring

The principal component analysis (Fig. 7) summarizes the relationships between the major soil properties such as pH, OC, total P, total N, NH_4^+, NO_3^-, available P, CEC, and all treatments after spring.

- The first two ordination axes explained 52.7% of the variance.
- This analysis shows that rice yield in spring was most closely associated with CEC, but not soil $pH.H_2O$, NH_4^+ or NO_3^-.
- The soil properties were significantly different between the compost and biochar treatments and the Control and CCF treatments.
- The available soil P, total N and total P was also closely associated but did not influence yield.
- It is interesting to note that there was a negative association between NH_4^+, NO_3^- and $pH.H_2O$ with total N which may reflect a more rapid N mineralization at higher soil pH.

3.1.2.1 Summer

The principal component analysis (Fig. 8) of the data for the summer trial indicates the following.

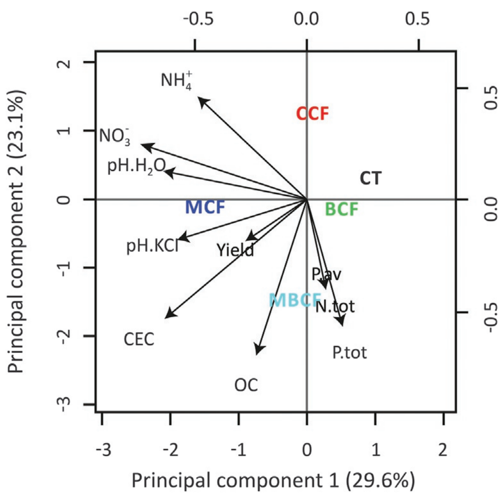


Fig. 7 Bi-plot of the first two principal components calculated from the rice yield and soil parameter after spring.

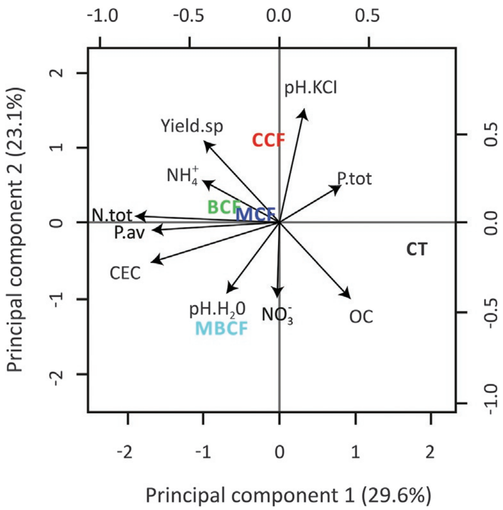


Fig. 8 Bi-plot of the first two principal components calculated from the rice yield and soil parameters after summer.

- The first two ordination axes explained 52.7% of the variance.
- This analysis shows that rice yield in summer was closely associated with NH_4^+ and total N, but not OC, NO_3^- and pH.
- The soil properties were significantly different between CT and other treatments.
- The available soil P, total N and total P were also closely associated but not influence to yield.
- It is interesting to note there was a negative association between total P and available P.
- There was a positive association between total P and pH.KCl (increased P fixing and decreased P efficiency).
- The CEC and available P was closely associated, but not OC.

The largest increase of SOC in the spring was observed in the plots with BCF and MBCF (Fig. 9). In the summer, the highest significant increase in SOC was achieved in the MBCF. There was a significantly higher increase in the SOC over and above that added (from the biochar and the compost) for all of the biochar treatments. This is a similar finding to that of Slavich et al. (2013).

The total C in soil varied significantly across two seasons (Fig. 10). In the spring, total C in soil of treatment MBCF was highest (37.6 t ha^{-1}), and MCF was ranked second (30.2 t ha^{-1}). The total C in control was the lowest. This trend remained similar in the summer.

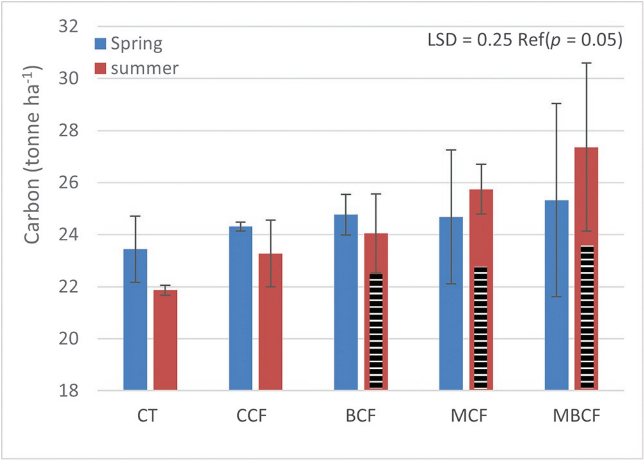


Fig. 9 Soil organic carbon at the end of the second season.

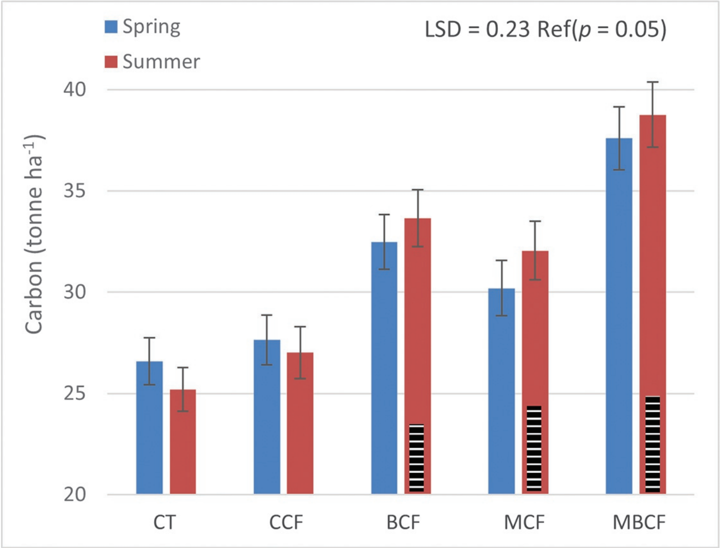


Fig. 10 Soil organic carbon at the end of the 2-year experiment.

4. Discussion

The results from this field experiment show that the greatest yield increase was obtained when biochar was incorporated with CCF and allowed to react in a closed container for a month. Joseph et al. (2013) and Qian et al. (2014) showed that the oxygen functional groups in the biochar made from crop residues at temperatures of approximately 450 °C will react with urea to form carboxyl urea derivatives. This will stabilize the N and allow it to be released slowly. In addition, the soil analysis shows that there is a significant increase in P in the treated soils for only the BCF treatment. Previous research showed that P can precipitate with Ca and/or the Fe in the biochar to form calcium or iron phosphates especially where clay with high Fe content is formed on the surface of biochar (Joseph et al., 2010; Vanek and Lehmann, 2015).

5. Conclusion

Application of all the treatments increased rice yield than the control (26–45%). The greatest increase in yield (45% higher than the control) was with the application of BCF. There was no significant difference ($P > 0.05$) in yield between the other treatments. Compost or biochar-compost increased the amount of SOC compared to the control. The highest SOC and nitrogen content was observed in 9.5 t ha^{-1} compost treatment mixed with 0.5 t biochar (MBCF).

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