



Role of Biochar on Greenhouse Gas Emissions and Carbon Sequestration in Soil: Opportunities for Mitigating Climate Change

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Abstract

Biochar, a pyrolyzed product of biomass, is richer in aromatic carbon (C) and poorer in oxygen which provides structural recalcitrance to it against microbial decomposition in soil. Biochar, being a stable source of C when applied to soil, remains there for longer period of time imparting long-term soil C sequestration. This sequestering effect of biochar has another advantage to mitigate climate change by reducing emission of greenhouse gases (GHGs) from soil. Both the interconnected processes imparted by biochar have its prominent role in climate resilience and environmental sustainability. Researchers around the world have been focusing on this aspect; thus revealing new facts and findings on managing biochar in agriculture. In this chapter, an attempt has been made to describe the biochar-governed mechanisms on emission of GHGs from soil, how the structural and functional properties of biochar regulates that, and the other associated factors like feedstock type and pyrolysis temperature during biochar preparation and soil inherent properties controlling various processes. Similarly, highlights of C sequestration potential of biochar made up of different crop/animal residues and other regulating factors have been described. Increase in pyrolysis temperature and switching over from manure to wood as a feedstock for biochar production increase the stability of biochar and reduce emission of GHGs from soil. The

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soils low in organic matter trigger C mineralization than that with high organic matter content. Biochar in presence of N fertilizer is reported to enhance CH₄ sink/decrease source strength of soil. The strongest effect of biochar on enhancing C sequestration and reducing GHGs emission is evident when it is applied in acid soils than alkaline soils. Both the concurrent processes of C sequestration and GHGs emission bring sanity to soil by physically more stable, enriching soil fertility, biologically more active and resulting to enhanced soil quality and lowering the C-footprint in agroecosystems.

Keywords

Crop residues · Bio(active)-char · Pyrolysis · Soil-biochar interactions · Feedstock type · GHGs emission · Stability of biochar

11.1 Introduction

Anthropogenic greenhouse gas (GHG) emissions have increased since the pre-industrial era, driven largely by economic and population growth and are now higher than ever. This has led to atmospheric concentrations of carbon dioxide (CO₂), methane (CH₄) and nitrous oxide (N₂O) that are unprecedented in at least the last 800,000 years (Mastrandrea et al. 2010). The concentration of CO₂, CH₄ and N₂O in the atmosphere since industrial revolution increased by 41.2%, 152–170% and 20–20.7%, respectively due to anthropogenic activities (Blasing 2013). Total CO₂ emissions from fossil fuels and industry rose by 1.6% in 2018 to 37.1 Gt CO₂ (Kelly 2018). Climate change will amplify existing risks and create new risks for natural and human systems.

Agricultural lands occupy about 40–50% of the Earth's land surface which accounted for an estimated emission of 51 to 61 Gt CO₂-eq yr⁻¹ (10–12% of total global anthropogenic emissions of GHG). The world population is expected to approach 10 billion people by 2050. With this projected increase in population and shifts to higher-meat diets, agriculture alone could account for the majority of the emissions budget for limiting global warming below 2 °C (3.6 °F) (Waite and Vennard 2018). This level of agricultural emissions would render the goal of keeping warming below 1.5 °C (2.7 °F) impossible.

Of global anthropogenic emissions, agriculture accounts for about 60% of N₂O and about 50% of CH₄. N₂O emissions from soils and CH₄ from enteric fermentation constitute the largest sources, 38% and 32% of total non-CO₂ emissions from agriculture in 2005, respectively (US-EPA: 2006). Biomass burning (12%), rice production (11%) and manure management (7%) account for the rest. Human-induced warming reached approximately 1 °C (likely between 0.8 °C and 1.2 °C) above pre-industrial levels in 2017, increasing at 0.2 °C (likely between 0.1 °C and 0.3 °C) per decade (Allen et al. 2018). Limiting warming to 1.5 °C implies reaching net zero CO₂ emissions globally around 2050 and concurrent deep reductions in emissions of non-CO₂ forcers, particularly CH₄ (Rogelj et al. 2018).

Adaptation and mitigation are complementary strategies for reducing and managing the risks of climate change. Substantial emissions reductions over the next few decades can reduce climate risks in the twenty-first century and beyond, increase prospects for effective adaptation, reduce the costs and challenges of mitigation in the longer term and contribute to climate-resilient pathways for sustainable development. In order to achieve large reductions in GHG emissions, sequestering carbon (C) in the terrestrial sink is needed (Paustian et al. 2016). The major challenges before the agricultural scientists is how to mitigate climate change by employing various methods to reduce emissions of GHGs into atmosphere and capturing CO₂ from atmosphere to securely store in the above ground and below ground.

11.2 Climate Change Mitigation Options

Among the principal components of radiative forcing of climate change, CO₂ has the highest positive forcing leading to warming of climate. Carbon dioxide has the least global warming potential among the major GHGs (viz. N₂O-298, CH₄-25 and CO₂-1), due to its much higher concentration in the atmosphere; it is the major contributor towards global warming and climate change. There are a number of improved and innovative agricultural practices available for reducing GHGs emissions from agroecosystems (Fig. 11.1) (Lal 2011). The agricultural practices are broadly divided into reducing emissions and sequestering emissions. Under reducing emissions, soil management, water management and crop management are the options. The soil management includes conservation tillage, high soil biodiversity and higher aggregation; the water management includes reduce runoff losses, soil amendments, aerobic rice, etc.; the crop management includes genetically improved varieties, high crop biomass production with deep root system, recalcitrant residues, etc. Land use, farming systems and soil, water and crop management are the pathways under sequestering emissions. Conservation of soil, water and nutrient, increase in ecosystem C pool, multiple ecosystem are the important avenues; agroforestry, lay farming, cover cropping are the important options under farming system approach; under soil, water and crop management, conservation tillage, integrated nutrient management, fertigation, bio-film and soil amendments with biochar are important pathways under land use. Biochar is considered as one of the important strategies under sequestering emissions option.

11.3 What Is Biochar?

Biochar is made by heating any organic material, such as wood, straw or manure, in an oxygen limited or zero oxygen environment, which releases gases (called syngas) and liquids (called bio-oils) and yields a solid product, which if intended for use as a soil amendment, is named biochar (Fig. 11.2) (Lehmann et al. 2006; Shackley and Sohi 2010). There are many ways to prepare biochar and most widely used method is electrically operated biochar maker in presence of continuous purging of nitrogen

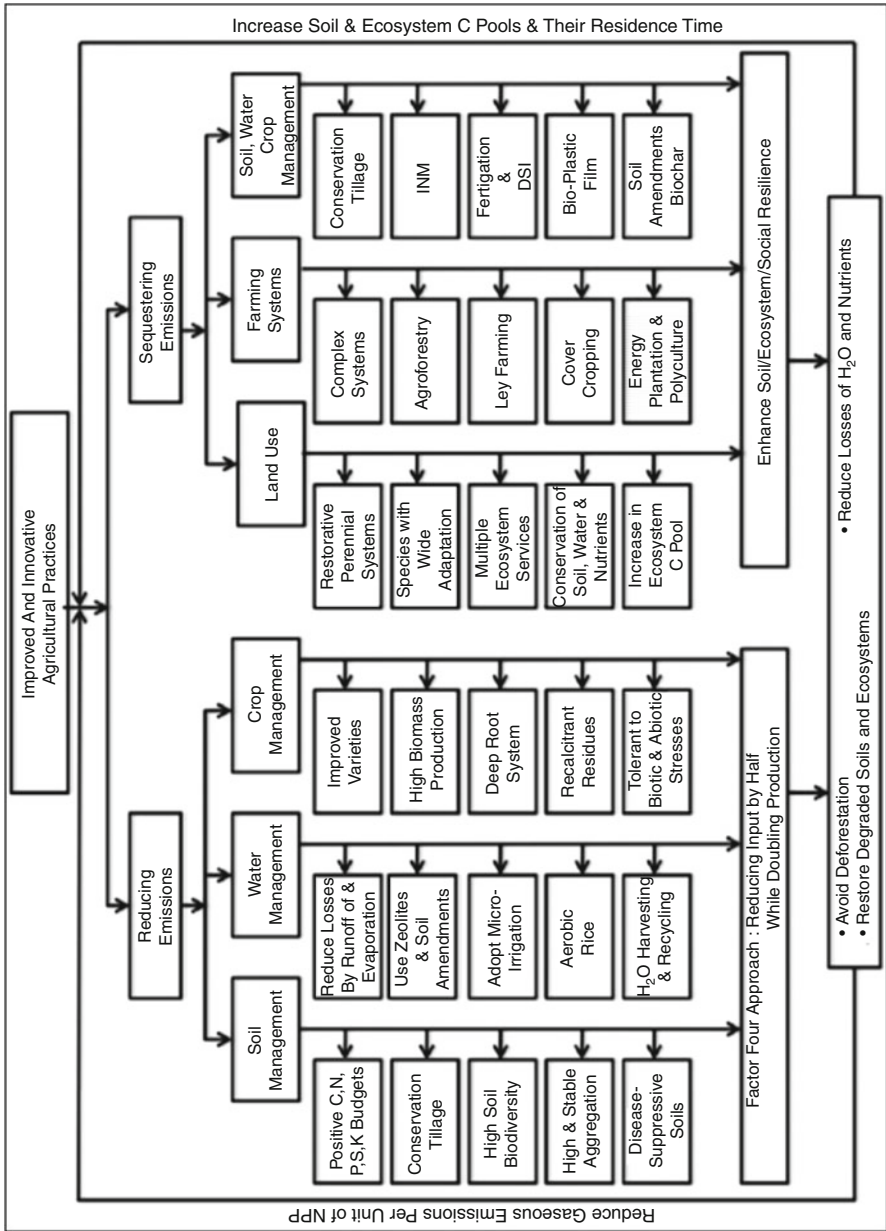


Fig. 11.1 Technological options to reducing emissions and sequestering emissions from agricultural ecosystems. *Source:* Lal (2011)

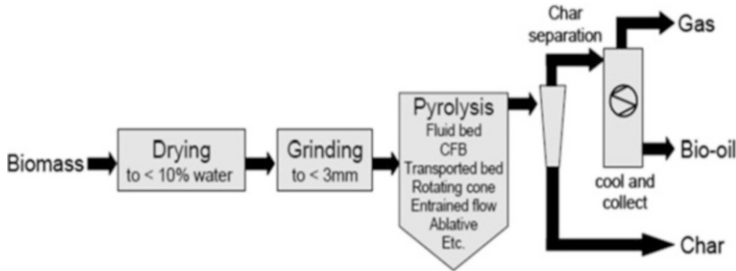


Fig. 11.2 Schematic diagram showing biochar production from biomass. *Source:* Sohi et al. (2009)



Fig. 11.3 Electrically operated temperature controlled biochar maker. *Source:* Purakayastha et al. (2016a)

gas (Fig. 11.3) (Purakayastha et al. 2016a). In contrast to the organic C-rich biochar, burning biomass in a fire creates ash, which mainly contains minerals such as calcium (Ca) or magnesium (Mg) and inorganic carbonates (Lehmann and Joseph 2009). The defining property is that the organic portion of biochar has a high C content, which mainly comprises the so-called aromatic compounds characterized by rings of six C atoms linked together without O or hydrogen (H), the otherwise more abundant atoms in living organic matter (Fig. 11.4). If these aromatic rings were

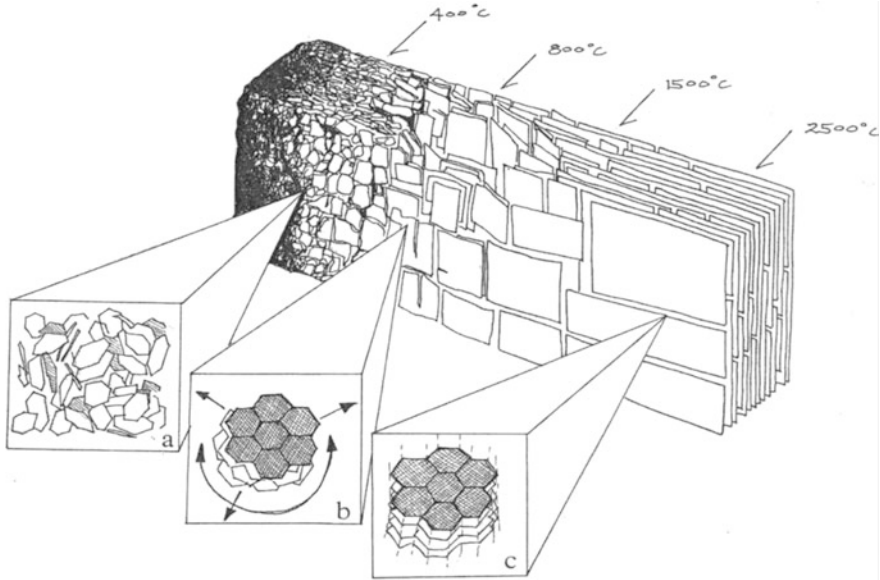


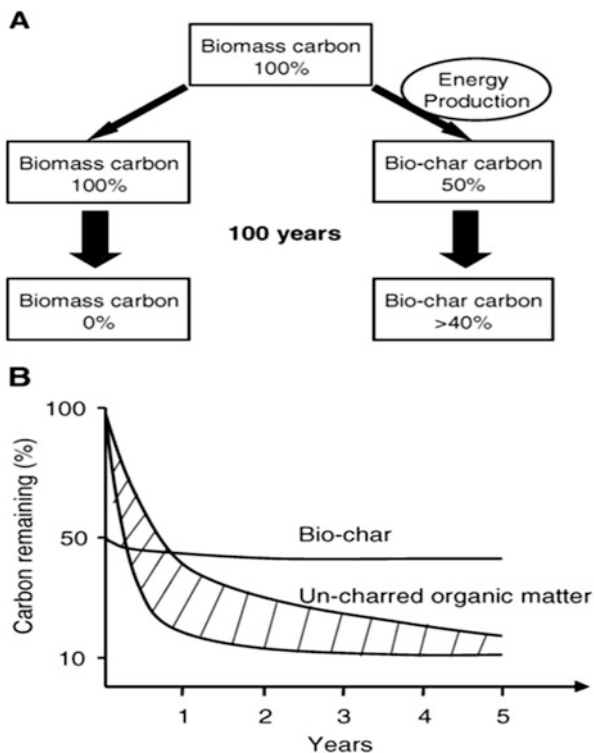
Fig. 11.4 Changes in structure of biochar with increase in pyrolysis temperature, (a) increased proportion of aromatic C, highly disordered in amorphous mass, (b) growing sheets of conjugated aromatic carbon, turbostratically arranged, (c) structure becomes graphitic with order in the third dimension. *Source:* Downie et al. (2012)

arranged in perfectly stacked and aligned sheets, this substance would be called as graphite. Under temperatures that are used for making biochar ($<700\text{ }^{\circ}\text{C}$), graphite does not form to any significant extent.

11.4 Biochar to Mitigate Climate Change: Complex Mechanisms

The production and application of biochar—a C-rich material produced during the pyrolysis of biomass—to soil has been proposed as a means for mitigating anthropogenic GHG emissions (Lehmann et al. 2006). The Pyrolysis-Biochar Bioenergy Platform (PBBP) has the potential to mitigate GHG emissions through three principal pathways. First, bioenergy produced by PBBP will offset GHG emissions from the burning of fossil fuels and by converting photosynthetic biomass C into recalcitrant biochar C. Indeed, pyrolysis converts 10–50% of the original biomass C into biochar C, which persists in soils for hundreds to thousands of years (Lehmann et al. 2006; Lehmann 2007; Laird 2008; Roberts et al. 2010). Second, biochar amendments increase soil quality, potentially increasing net primary productivity and thereby reducing economic pressure to convert native lands to agricultural production (Kauffman et al. 2014). Third, soil biochar applications may directly reduce GHG emissions from soils.

Fig. 11.5 Schematics for biomass or biochar remaining after charring and decomposition in soil. (a) C remaining from biomass decomposition after 100 years from IPCC (1996); (b) range of biomass C remaining after decomposition of crop residues from Jenkinson and Ayanaba (1977). Source: Lehmann et al. (2006)



Biochar found in high proportions in the so-called *Terra Preta* soils of the Amazon region (Liang et al. 2008) has been radiocarbon dated and found to originate from 500 up to 7000 years BC (Neves et al. 2004). Because of higher half-life, biochar is considered suitable for long-term C sequestration in soil. It was estimated the global C sequestration potential of C 0.16 Gt yr⁻¹ as forest residues, mill residues, field crop residues and urban wastes is used for biochar production (Lehmann et al. 2006). Thus, biochar allows more C input as compared to the C output and this is the basis behind biochar's possible C negativity and hence its potential for climate change mitigation. It is possible to increase 25% of soil C as the biochar storage capacity of temperate grassland and cropland is about 400 Gt (Lehmann et al. 2006). The charred material releases 50% of the labile C into the atmosphere during its formation and remaining non-labile C remains into soil while non-biochar material application into soil releases C into the atmosphere (Lehmann et al. 2006) (Fig. 11.5).

Biochar being a pyrolyzed product is highly stable and resistant to decay by microorganisms. Thus there is considerable interest in the concept of applying biochar in to soil as a long-term sink for C, thereby mitigating climate change (Prayogo et al. 2014). In this connection, the application of biochar to soils has been shown to achieve the net C gain in soils while also serving for increased plant biomass production by enhancing the nutrient supply to plants and increasing

nutrient and water use efficiencies (NUE and WUE) by plants (Kookana et al. 2011; Lehmann et al. 2006; Lehmann et al. 2015; Minasny et al. 2017; Purakayastha et al. 2015, 2016b, 2019) and decreased N₂O and CH₄ emission from soils (Rondon et al. 2005). Besides direct effects of biochar on nitrifying organisms, it is possible that biochar could induce strong N immobilization and could decrease ammonification and nitrification in the short term (Lehmann et al. 2006; Warnock et al. 2007). Mukherjee and Lal (2013) described the probable mechanism governing GHG flux of biochar-amended soils following 2-phase complex formation hypothesis. The initial flux of CO₂ from biochar-added soil is a result of microbial interaction of labile-C (volatile and short-duration compounds) of biochars in a weak complexation (non-specific EDA type interaction/H-bonding) with soil mineral surface. The second phase of GHG emission is not instant but gradually happened over a longer time and often slower in rate, as a consequence of relatively stable complex formation (cyclic aromatic compounds) within the inner core of biochar in interaction with soil mineral and microbial biomass.

Methane flux measured at the soil–atmosphere interface is the net effect of two processes: methane production by methanogens and methane uptake by methanotrophs (Dunfield et al. 1993). Biochar applications are expected to make soil conditions favourable for methanotrophs and unfavourable for methanogens, thereby increasing the CH₄ sink capacity of soil. The mechanisms by which biochar may affect soil CH₄ fluxes include sorption of CH₄ to biochar’s surfaces (Yaghoubi et al. 2014) and soil aeration by biochar addition, which may increase diffusive CH₄ uptake (Van Zwieten et al. 2010; Karhu et al. 2011), as microbial CH₄ oxidation in upland soils is mostly substrate-limited (Castro et al. 1994).

Thus, biochar application to soils has been recommended as an important component of the pathway to “climate-smart” soil management practices in modern global agriculture (Paustian et al. 2016; Purakayastha et al. 2019). Therefore, biochar addition is a *win–win* strategy for climate change mitigation and enhancing crop production.

11.5 Biochar Stability: A Prerequisite for Carbon Sequestration in Soil

The composition changes through a complete destruction of cellulose and lignin and the appearance of aromatic structures (Paris et al. 2005) with furan-like (five-membered aromatic ring with four C atoms and one oxygen) compounds (Baldock and Smernik 2002) during pyrolysis have a significant effect on the stability of biochar. The following properties of biochar make it more stable in soil system.

11.6 Aromaticity

Biochar is commonly considered to be highly aromatic and containing random stacks of graphitic layers (Schmidt and Noack 2000). Purakayastha et al. (2015) conducted FTIR analysis and confirmed the functional groups present in maize stover biochar contributed significantly to the cation exchange properties (Fig. 11.6). In general, H/C and O/C ratios in experimentally produced biochars decrease with increasing temperature (Shindo 1991; Baldock and Smernik 2002; Purakayastha et al. 2016b) and increased with time of heating (Almendros et al. 2003).

11.7 Presence of Amorphous Structures and Turbostratic Crystallites

Biochar is mainly characterized by amorphous structures and turbostratic crystallites that may contain defect structures in the graphene sheets with oxygen (O) groups and free radicals (Bourke et al. 2007). Ordered graphene sheets were found to increase only at a carbonization temperature above 600 °C (Kercher and Nagle 2003). Because of their unordered structure, amorphous and turbostratic crystallites have a high stability (Paris et al. 2005), which could be one reason for the stability of biochar produced at relatively low temperatures of <600 °C.

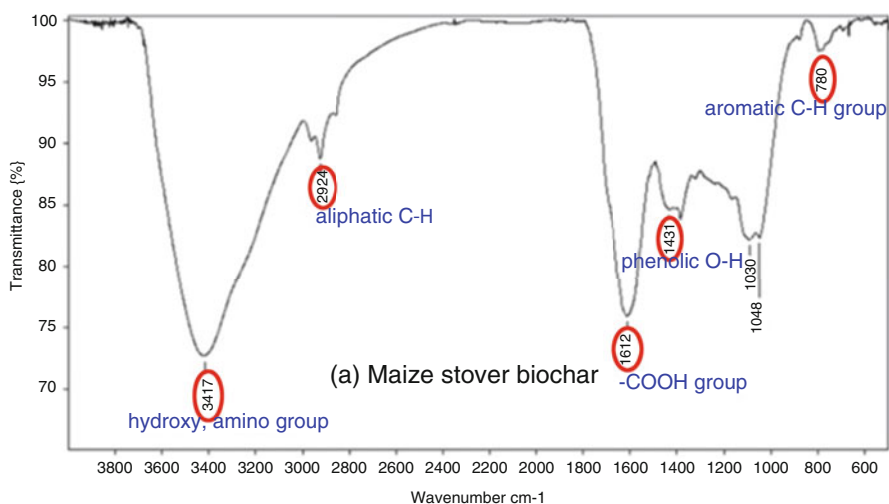


Fig. 11.6 Infra-red spectrogram of maize stover biochar. *Source:* Purakayastha et al. (2015)

11.8 Presence of Rounded Structures

Rounded structures may be even more stable than turbostratic structures in biochar (Cohen-Ofri et al. 2007). For cedar wood pyrolyzed at 700 °C, onion-like graphitic particles have been observed that are probably formed from lignin (Hata et al. 2000), but it is not clear whether these are a common feature in biochar (Shibuya et al. 1999). The round structures are actually fullerenes, molecular-scale spherical structures that include both hexagonal and pentagonal rings that have great stability (Harris 2005). Rounded features were also reported in biochars from German Chernozems with ages of 1160–5040 years using high-resolution transmission electron microscopy (Schmidt et al. 2002).

11.9 Reduced Accessibility to Decomposers

Biochar has been preferentially found in fractions of SOM that reside in aggregates rather than as free organic matter (Brodowski et al. 2006; Liang et al. 2008), which is considered to reduce its accessibility to decomposers. Biochar particles are, indeed, abundant within stable micro-aggregates. Moreover, microorganisms can be spatially associated with biochar in soils as porous structure of biochar invites microbial colonization. Reducing accessibility by aggregation is, therefore, proposed to be significant in controlling biochar decomposition, but of less importance than chemical recalcitrance.

11.10 Particulate Nature

The particulate form may have an important role in decreasing decomposition rates of biochar and increasing recalcitrance of biochar. Oxidation of biochar particles starts at its surfaces (Cheng et al. 2006) and typically remains restricted to the near-surface regions even for several millennia (Lehmann et al. 2005; Liang et al. 2006; Cohen-Ofri et al. 2007). Therefore, due to particulate nature, outer regions of a biochar particle protect the inner regions from access by microorganisms and their enzymes.

11.10.1 Interactions with Mineral Surfaces

A significant portion of biochar is found in the organo-mineral fraction of soil (Brodowski et al. 2006; Laird et al. 2010), suggesting that biochar forms interactions with minerals. Rapid association of biochar surfaces with Al and Si and, to a lesser extent, with Fe was found during the first decade after addition of biochar to soil (Nguyen et al. 2008). Coating of biochar particles with mineral domains is frequently visible in soils (Lehmann 2007) and suggests interactions between negatively charged biochar surfaces and either positive charge of variable-charge oxides

by ligand exchange and anion exchange, or positive charges of phyllosilicates by cation bridging. Similarly, Ca was shown to increase biochar stability, most likely by enhancing interactions with mineral surfaces (Czimczik and Masiello 2007).

11.11 Role of Biochar on Soil C Sequestration

Soil C sequestration refers to capture of CO₂ from atmosphere and securely store into soil so that it is not immediately emitted into atmosphere. Plant biomass decomposes in a relatively short period of time, whereas biochar is orders of magnitudes more stable. So, given a certain amount of C that cycles annually through plants, half of it can be taken out of its natural cycle and sequestered in a much slower biochar cycle. By withdrawing organic C from the cycle of photosynthesis and decomposition, biochar sequestration directly removes carbon dioxide from the atmosphere and stores it in a much more durable form in soil. So, locking C up in soil makes more sense than storing it in plants and trees that eventually decompose (Lehmann 2007). The biochar C sequestration is influenced by various factors, e.g., feedstock type, pyrolysis temperature, soil properties, etc., which are described below.

11.11.1 Feedstock Type and Pyrolysis Temperature

The type of feedstock influences the efficiency of C conversion into the resultant biochar provided that the pyrolysis temperature for production is in the range of 350–500 °C (Lehmann et al. 2006). Any biomass material can be converted in to biochar but its yield and other physico-chemical properties vary (Verheijen et al. 2010). Baldock and Smernik (2002) showed that 20% of the added organic C from unaltered *Pinus resinosa* wood (heated at 70 °C) was mineralized, but, this value was <2% for samples heated at temperatures ≥200 °C indicating much higher stability of thermally altered woods. The greater stability of biochar prepared at higher temperature mainly due to the differences in proportion of alkyl and aromatic groups that increases with rise in temperature (Mcbeath and Smernik 2009).

Biochar prepared from wood pellets made from a mixture of Black Spruce (*Picea mariana*) and Jack Pine (*Pinus banksiana*), the solid fraction of pig manure and switchgrass (*Panicum virgatum* L.) at the highest pyrolysis temperature with low O/C_{org} and H/C_{org} ratios resulted in the lowest increase in CO₂ emissions, which could indicate a higher biochar C stability (Brassard et al. 2018). Wood biochar was most stable and pig manure biochar was least stable in silty loam and loamy sand soil; biochar prepared from switch grass was medium in stability (Brassard et al. 2018). Bruun et al. (2010) reported that mineralization of ¹⁴C labelled biochar decreased considerably as production temperature increased from 400 °C to 500 °C, but reduced at 600 °C. The increased CO₂ evolution, in the early stages of experiment is derived from the carbonates of biochar, whereas at 600 °C the carbonate content is more in biochar showing less-induced mineralization.

Purakayastha et al. (2016b) reported that corn stover biochar prepared at 600 °C was more stable in Mollisol and Ultisol.

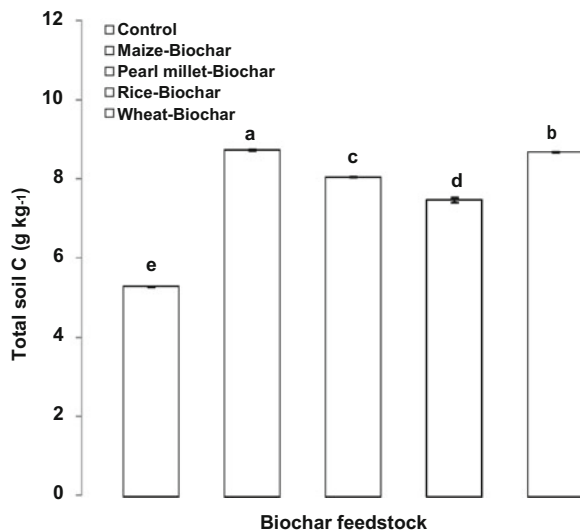
11.11.2 Application Rate of Biochar

The dose of biochar into soil is an important aspect to acquire C stabilization in soil. As Butnan et al. (2017) reported that application of biochar at 2% dose in soil helps in better stabilization over the 1% or 4% doses. In other study, application of rice husk biochar at a dose of 41.3 Mg ha⁻¹ in Gleysol, Nitosols, Acrisol could increase 12.9, 12.4 and 0.51 kg of soil C with respect to control (Haefele et al. 2011). Similarly, the application of maize stalk and pinewood biochar at the rate of 10 Mg ha⁻¹ and 5 Mg ha⁻¹ in Nitosols could increase soil C by 0.77% and 0.71% in comparison to control (Nigussie et al. 2012). Purakayastha et al. (2015) reported that application of maize stover, pearl millet stalk, rice straw and wheat straw biochar at the rate of 20 Mg ha⁻¹ enhanced total soil C by 65%, 52%, 41% and 64%, respectively, in an Inceptisol from Delhi (Fig. 11.7).

11.11.3 Soil pH

In general soil pH tends to increase on application of biochar. It was reported that on an average application of biochar at a dose of 20 or 40 Mg ha⁻¹ tends to increase the soil pH by 0.2 or 0.4 units in a loam acidic soils with pH 6.0 (Liua et al. 2019). It was reported that poultry litter biochar is highly alkaline in nature, hence significantly affect the pH of the acidic soils (Purakayastha et al. 2019). In the alkaline soils

Fig. 11.7 Effect of different biochar on total soil carbon at the end of 1 year of carbon mineralization, the bars with different lowercase letters are significant according to Duncan's multiple range test at $P = 0.05$. Source: Purakayastha et al. (2015)



(pH = 8.1), addition of biochar increased C sequestration as native soil organic carbon (SOC) mineralization was minimal (Singh and Cowie 2014). The application of biochar in acidic soil emits more CO₂ in comparisons to alkaline soils. It was reported that addition of olive biochar in acidic soils increased two-fold CO₂ emissions and decreased N₂O emissions by 68% (Wu et al. 2018).

11.11.4 Soil Texture

The role of soil texture has its significance in achieving SOC stability through addition of biochar. The addition of biochar had a significant impact on the SOC stabilization in coarse-structured Al-rich Ultisol as compared to fine textured Mn-rich oxisols (Butnan et al. 2017). The higher clay content in soil reported to enhance SOC stabilization (Bationo et al. 2007). Gleysols had higher C sequestration potential than Nitisols and Acrisols (Haefele et al. 2011) on application of biochar at a fixed dose of 41 Mg ha⁻¹. Biochar-C stabilization was found to be more in oxisols than the soils dominated by permanent charged minerals (Vertisol and Entisol) or sand (Inceptisol) (Fang et al. 2014).

11.11.5 Interaction of Biochar with Native Soil Organic Matter

As biochar is porous in nature, it has higher affinity for natural organic matter (Kasozi et al. 2010). Alternatively, biochar containing labile-C may have a stimulatory effect on native soil C mineralization. The positive priming could occur if biochar acts as a metabolic C source, nitrogen, phosphorus and micronutrients (Chan and Xu 2009) or even a habitat favouring increased microbial heterotrophic activity (Thies and Rillig 2009). The presence of biochar in soils also enhanced the degradation of more labile-C sources such as ryegrass residue (Hilscher et al. 2009). Another study using 16 chars and two soil types, about a third decreased and a third had no effect on SOC respiration (Spokas and Reicosky 2009). Clearly, overall priming direction and magnitude varied greatly with soil and biochar type. One apparent trend, however, is that, for a given biochar biomass type, priming effect on total C oxidation generally decreased with increasing combustion temperature. For 250, 400, 525 and 650 °C biochar, the average priming effect over 1 year was 16, 9, 5 and 12, respectively (Zimmerman et al. 2011). In addition, negative priming was more prevalent in the two soils with the lowest SOC and least potentially mineralizable SOC. The native SOC is an important parameter that decides the C sequestration potential of soils. It was reported that soil with low SOC on application of biochar simulates mineralization of labile C (Singh and Cowie 2014). It was reported that Oxisols with higher native SOC (4.39%) mineralized less CO₂ than the Inceptisol with low SOC content (0.95%) (Fang et al. 2014). Purakayastha et al. (2015) studied stability (C efflux study) of rice, wheat, maize and pearl millet biochars at 400 °C and reported that maize biochar was found to be the most stable showing reduced C mineralization by protecting the native soil organic C (Fig. 11.8).

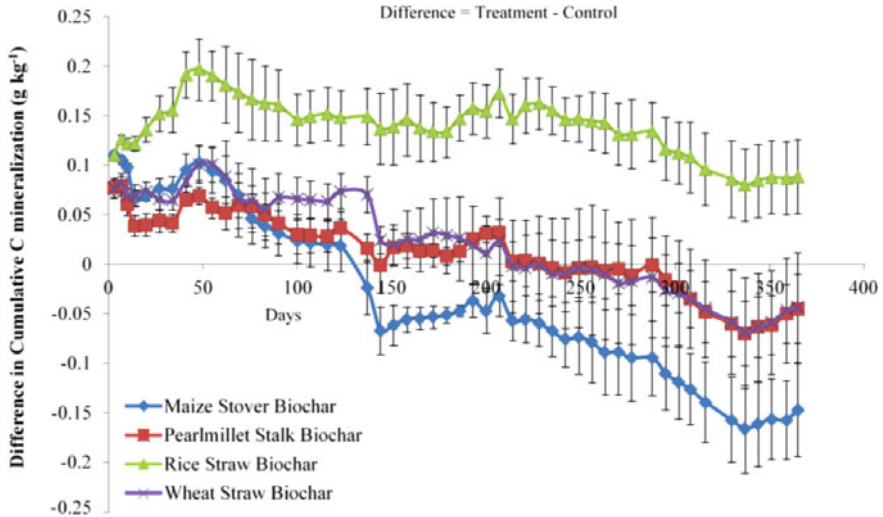


Fig. 11.8 Changes in carbon mineralization (CO₂ efflux) from soil with BC compared to the respective control treatments without BC addition. Error bars show standard errors ($n = 4$). *Source:* Purakayastha et al. (2016b)

Contrarily, rice biochar exhibited higher C mineralization. It is evident that the benefits of C sequestration through biochar are more visible in soils which are lower in C than soils relatively higher in C (Yadav et al. 2017). The interaction of soil and biochar showed that same biochar behaved differently when applied in soils with different organic matter content (Purakayastha et al. 2016b). It was reported that wheat straw biochar at 600 °C showed positive priming effect when applied in a soil (Ultisol) with lower organic matter but showed negative priming effect in a soil (Mollisol) with higher organic matter (Purakayastha et al. 2016b).

11.12 Effect of Biochar on Greenhouse Gas (GHG) Emissions

Many instances evidenced that biochar application to soil has a very good response over the transformation and retention of C and N in soil, which over the time regulates the mechanisms and finally improve the sink capacity of GHG and reducing the emissions. The recalcitrance nature of stable aggregates can increase the shelf-life of biochar-amended soil C over time and reduce the emissions of GHGs (Spokas et al. 2009; Spokas and Reicosky 2009). Contrarily, there are also reports showing increased GHG emissions due to biochar applications in soil (Lin et al. 2017; Liu et al. 2014; Shen et al. 2014; Yanai et al. 2007). There is an obvious chance while multiple factors like feedstock type, pyrolysis temperature, nitrogen fertilizer rate and soil internal factors can significantly affect soil CO₂, CH₄ and N₂O fluxes after biochar amendment (He et al. 2017).

11.12.1 Biochar Feedstock on GHG Emissions

Quite a good number of researches undertaken in last two decades have given a clear-cut indication that the rate of GHG emissions from biochar-amended soil largely depends on two factors: feedstock of biochar and soil types. Over two cropping cycles in a paddy field, China, wheat straw biochar application significantly reduced N_2O emission but CO_2 emission remained unchanged throughout the two cycles; while biochar showed its positive effect in reduced CH_4 emission in the second crop cycle while simultaneous improvement in soil quality. In acidic soils contrasting effects of olive biochar and corn biochar were observed owing to biochar's liming effect and soil pH played a crucial role here, without any visible effect at alkaline clay soil. The corn biochar addition decreased CO_2 and N_2O emissions by 11.8% and 26.9% in the acidic sandy soil, respectively, whereas addition of olive biochar in the same soil triggered two-fold higher CO_2 emission rate and N_2O emission decreased by 68.4% (Wu et al. 2018). Rittle et al. (2018) reported that biochar produced from agricultural residues promotes GHG emissions from soil over a short-term period and that happened more in wet condition in Brazilian soil. Across the nine biochars studied, they reported that swine manure-origin biochar (of lowest C:N ratio) resulted in the highest GHG emissions, while eucalyptus origin biochar (of highest C:N ratio) had resulted in lowest GHG emissions. In another laboratory study, woodchip biochar could result in reduction of CO_2 , N_2O and CH_4 emissions from the soil, while the significant suppression was obtained only at biochar amendment levels $>20\%$ w/w (Spokas et al. 2009).

Muñoz et al. (2019) reported that the cow manure biochar decreased CO_2 and CH_4 emissions across volcanic and non-volcanic soils. On the other hand, in boreal Scots pine forests soil, wood-derived biochar amendment (applied at a rate of $5\text{--}10 \text{ Mg ha}^{-1}$) did not show any pronounced effect on soil CO_2 effluxes (Palviainen et al. 2018).

Using biochar as a bulking agent for composting has been proposed as a novel approach to solve the environmental trade-offs of compost (Sancez-Garcia et al. 2015; Steiner et al. 2010). Biochar-chicken manure co-compost could substantially reduce soil N_2O emissions compared to chicken manure compost (Yuan et al. 2017).

Criscuoli et al. (2019) tested woodchip biochar in this regard and found that variation in temperature (ranging $10\text{--}30^\circ\text{C}$) did not affect soil N_2O emission but marginally affected CO_2 emission whereas showed negative impact on soil CH_4 uptake in a wide range of soil temperatures conducted in a pot experiment at growth chamber. In terms of interactions with feedstock source, biochar produced from biosolids led to a statistically significant increase in sink strength/reduction in source strength. When produced from lignocellulosic waste, biochar significantly decreased the CH_4 sink strength/increased the source strength. No other feedstock showed statistically significant effects on CH_4 fluxes (Jeffery et al. 2016).

Contrarily, the high N_2O emissions from the low-temperature green-waste biochar treatment indicate that the decline in $\text{NO}_3\text{--N}$ observed in this treatment was probably a result of enhanced activity of denitrifiers causing rapid conversion and loss of $\text{NO}_3\text{--N}$ in soil through N_2O emissions rather than an inhibition of

nitrification (Yanai et al. 2007). Biochar amendment of upland soil has been generally accepted to mitigate nitrous oxide (N_2O) emissions. However, this is not always the case in rice paddy soil. In this connection, Lin et al. (2017) reported that wheat straw-derived biochar amendment of paddy soils increased soil pH, which in turn increased the abundance and diversity of ammonia oxidizing bacteria and N_2O emissions. Previous study suggested that increased N_2O emission under biochar application was due to additional N input within the biochar (Shen et al. 2014) or increased denitrification resulting from biochar-derived labile organic C in paddy soils (Liu et al. 2014). However, biochar application has also been determined in increase of soil pH (Wang et al. 2012; Purakayastha et al. 2016b) and improved soil aeration (Zhang et al. 2010); such factors are associated with the abundance and community structure of ammonia oxidizing bacteria (AOB) and ammonia oxidizing archaea (AOA) (Chen et al. 2011; French et al. 2012; Li et al. 2018).

11.12.2 Pyrolysis Temperature on GHG Emission

Pyrolysis temperature of biochar preparation is crucial for GHG emissions from soil. High temperature biochar (willow, pine, maize, wood mixture) was reported to reduce N_2O emissions more than low-temperature biochar (Nelissen et al. 2014) and they reported that biochar application decreased both cumulative N_2O (52–84%) and NO (47–67%) emissions compared to a corresponding treatment without biochar. The application of municipal waste biochar, produced at 700 °C at the rate 10% (w/w) suppressed N_2O emission by 89% in a clay loam soil (Yanai et al. 2007). Soil amended with biochars produced from oak and hickory, pyrolyzed at 450–500 °C, showed a reduction of N_2O flux but increment in CO_2 flux in a long-term incubation experiment (Jones et al. 2011). Singh et al. (2010) demonstrated that after an initial spike of N_2O emission accounted, due to higher labile N content of biochar and microbial activity, the rate of emission decreased over time. Reduced H:C_{org} ratios in high temperature biochars indicate increased aromaticity, which is associated with the reducing effect of biochar on N_2O emissions (Cayuela et al. 2015). Stewart et al. (2013) reported that fast pyrolysis (with lower biochar yield) produced a highly recalcitrant biochar, derived from oak pellets (550 °C) that better sequestered C and reduced GHG emissions, where CO_2 was the primary GHG emitted, followed by N_2O .

Biochar has been shown to increase (Zhang et al. 2010; Spokas and Bogner 2011), decrease (Feng et al. 2012; Dong et al. 2013; Reddy et al. 2014), or have no significant effect (Kammann et al. 2012) on CH_4 emissions from soils. Some contrasting reports suggested that biochar-amended soils may enhance CO_2 and CH_4 emissions. Once a paddy soil was amended with biochar derived from bamboo and rice straw both pyrolyzed at 600 °C, the emissions of CH_4 and CO_2 were reduced by 51 and 91%, respectively (Liu et al. 2011). Another field study carried out in Australia applying cattle waste biochar produced at 550 °C indicated there was no significant difference in GHG fluxes (Scheer et al. 2011).

Rittle et al. (2018) showed that biochar production at higher pyrolysis temperature (600°C) with high C:N biochars (Eucalyptus origin) proved best to minimize GHG emissions. Biochars produced at high temperatures caused a statistically significant increase in CH₄ sink strength/reduction in source strength following application to soils. Mid-temperature biochars (450–600 °C) led to significant reductions in CH₄ sink strength/increased source strength when applied to soil.

11.12.3 Soil Type and Nitrogen Fertilizer Rate

Biochar application to acidic soils (i.e. with a pH <6) resulted in the strongest effect size, causing an increase in CH₄ sink strength/decrease in source strength following biochar application (Fig. 11.9) (Jeffery et al. 2016). Conversely, addition of biochar to soils within the neutral pH range (i.e. 6–8) showed a decrease in CH₄ sink strength/increase in source strength. Application of biochar to soils with a pH > 8.0 did not show any response to biochar application. Biochar effects on CH₄ flux interact with N fertilizer rate (Fig. 11.9). Application of N fertilizers caused a strong increase in CH₄ sink strength/decrease in source strength in the presence of biochar at rates <120 kg ha⁻¹ but no response at higher rate. Biochar increased potential nitrification rates when soil ammonium concentrations were high following fertilizer application, thus enhancing N₂O emissions in the Biochar + Nitrogen treatment early in the season which were likely nitrification associated (Edwards et al. 2018). However, it was reported that over the full growing season, biochar application reduced cumulative N₂O emissions in Biochar + Nitrogen plots to levels similar to the unamended control (Fig. 11.10). The study demonstrates that biochar can have dynamic effects on soil N₂O emissions and the underlying microbial processes that depend on changing edaphic conditions, such as soil inorganic nitrogen availability and moisture, over the growing season.

11.13 Epilogue

Biochar being a highly carbonized product with higher stability in soil emerged as one of the residue management strategies for long-term C sequestration in soil for mitigating climate change. This approach is a win-win strategy while transforming huge amount of residues generated into useful products like bioenergy, bio-oil, syngas and biochar. Biochar prepared from feedstock having higher lignocellulosic material, e.g. wood biomass at higher pyrolysis temperature be having higher C sequestration potential than that prepared from low lignocellulosic material, e.g. straw biomass or manure. Biochar interacts with soil organic matter in a complex way to show either positive, negative or no priming effect, the magnitude varies with soil and biochar type.

Biochar when acts as a source of labile C and nutrients could cause positive priming effect on native soil organic matter, while biochar when adsorbs the refractory pools of soil organic matter in its porous structure might cause negative

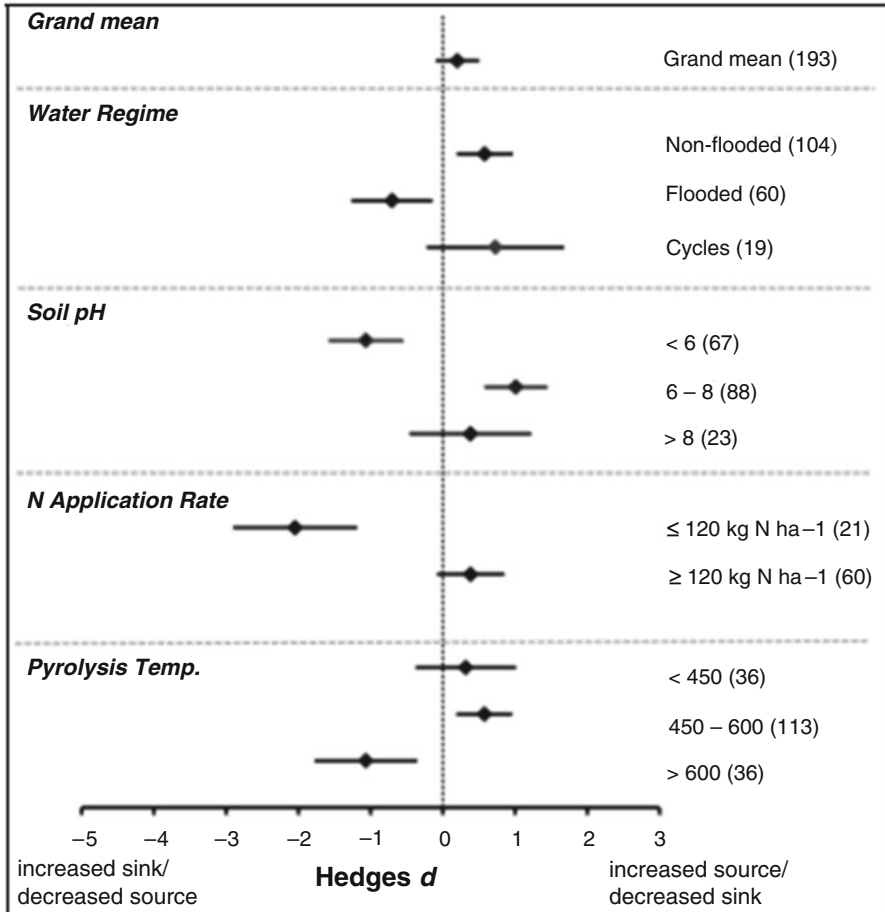
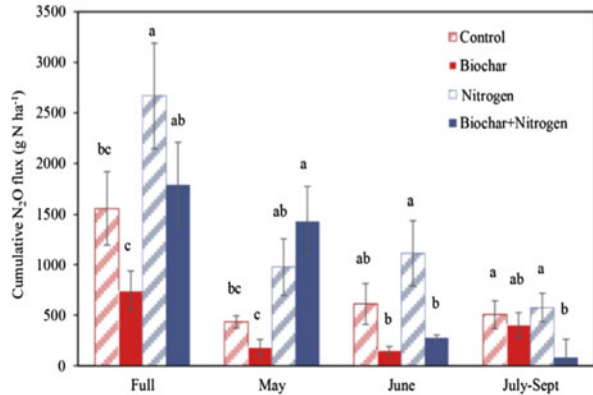


Fig. 11.9 A forest plot of Hedge’s d calculated from published literature grouped by experimental water regime, soil pH pre-biochar amendment, N fertilizer application rate and biochar pyrolysis temperature. Points show means, bars show 95% confidence intervals. The numbers in parentheses indicate the number of pairwise comparisons on which the statistic is based. (For an explanation of the Hedge’s d metric see text). *Source:* Jeffery et al. (2016)

priming. Carbon sequestration by biochar is likely to be less in soils relatively higher in native-C than in soils relatively lower in native-C due to stimulation of native C loss by biochar application. Besides C sequestration, biochar addition can be effective for reducing CH₄, N₂O and NO emissions from soils. However, the effect of biochar is highly dependent on its physical and chemical composition, feedstock from which it is prepared, pyrolysis temperature and soil type. The established literatures indicate that soil and biochar properties, as well as management conditions, must be considered to exploit biochar’s full potential to mitigate GHGs emissions and minimize trade-offs. Low temperature, slow pyrolysis maximize

Fig. 11.10 Effects of biochar and N fertilizer addition on cumulative N_2O flux for the full growing season (Full), early season (May), mid-season (June) and late season (July–September). Error bars represent one standard error of the mean ($n = 4$) and letters indicate statistically significant differences among treatments for a given time period.
 Source: Edwards et al. (2018)



biochar production and thereby also C sequestration potential. However, research on biochar suggests that biochar prepared at higher pyrolysis temperature is more effective at mitigating CH_4 and N_2O emissions. Which one has the greatest potential to mitigate climate change thus remains to be established by employing life cycle assessment approaches. It is an established fact that the pH and ash contents of biochar increased with pyrolysis temperature while CEC of biochar decreased. Therefore high temperature biochar warrants its application to either neutral or alkaline pH soils but this biochar could be suitable for acid soils owing to derive extra benefits of biochar as a liming material. For making the biochar technology be more popular among the farmers, its production cost need to be lowered down and this is possible if the biochar originates from the bioenergy platform as an industrial by-products. Thus the biochar technology could be a win–win strategy which provided an opportunity to transform huge residues to transform into bio-oil, bioenergy, syngases and mitigating climate change by reducing GHGs emissions and enhancing C sequestration potential of soils.

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