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Potential Benefits of Biochar in Agricultural Soils: A Review

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ABSTRACT

Soil degradation, characterized by declines in nutrient status and simultaneous accumulation of pesticide residues, is a major problem affecting agricultural ecosystems. Previous studies indicate that biochar application to soil has promise as a practical method to alleviate these pressures: increasing crop yield and enhancing pesticide degradation. Here, we review the roles of biochar in both chemical and biological promotion of pesticide degradation and the potential benefits of biochar relating to the efficiency of fertilizer use, availability of nutrients, and nutrient exchange. Biochar typically has a high surface area featuring many functional groups, a high cation exchange capacity, and high stability. Major factors that impact on the nutrient retention characteristics of biochar (*e.g.*, feedstock, pyrolysis temperature, and application rate) are also discussed herein. Nonetheless, more studies of the long-term impacts on soil properties from biochar addition are still required before it can be possible to accurately quantify the sustainability of this approach to sequester carbon and restore soil function.

Key Words: biodegradation, chemical degradation, crop yield, feedstock, nutrient, pesticide, pollution, pyrolysis temperature

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INTRODUCTION

The application of fertilizer (e.g., nitrogen (N),phosphorus (P), and potassium (K)) and pesticides in agricultural soil to improve crop yields has become increasingly intensive. This is most apparent in China, where fertilizer consumption accounts for 90% of all fertilizer used worldwide (Pan et al., 2017), and the annual increase in pesticide consumption is estimated to be equivalent to 61%, 55%, and 10% of the total amount applied annually in Cambodia, Laos, and Vietnam, respectively (Schreinemachers et al., 2015). Under intensive applications, leaching losses of nutrients and pesticides tend to increase, which collectively lead to the deterioration of soil fertility and environmental pollution. Moreover, nutrient leaching from agricultural soils increases the cost of farming, accelerates soil acidification, and reduces crop yields (Laird et al., 2010). Pesticides have the tendency for long-range transport and transboundary dispersion, and their capacity to bioaccumulate in the food chain can threaten human health and the health of other organisms present in the environment (Kuranchie-Mensah et al., 2011).

On the one hand, crop yields must be increased to satisfy the growing food demands in many countries, which requires improvements in soil fertility and nutrient availability (Liu Z X et al., 2014); on the other hand, pesticide degradation is an important goal both to safeguard human health and to protect the environment (Zhang et al., 2011). Microbial degradation (e.g., by bacteria and fungi) can remove pesticides from contaminated soils through enzymatic degradation (Verma et al., 2014). Complex pesticides can also be decomposed into simple organic or inorganic substances via molecular approaches (Wood, 2008). Bioremediation is less hazardous, has a lower cost, and is more socially acceptable (Zhang et al., 2011). However, some pesticides (e.g., organochlorine pesticides) cannot be easily biodegraded within a short period of time. Moreover, the required microorganisms can be sensitive to changes in environmental conditions,

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such as heat, desiccation, and ultraviolet radiation (Chakoosari, 2013), and also to competition between different microbial species and other organisms living in the soil (Pietikäinen *et al.*, 2000). Variable conditions can therefore reduce the efficiency of pesticide biodegradation. It follows that effective, low-cost methods with capacity to enhance soil microbial activity and pesticide degradation could help achieve increases in the yield of good quality, pesticide-free crops and improve the sustainability of agriculture.

Biochar is a carbon-rich solid produced by heating biomass with little or no oxygen (Tan *et al.*, 2015, 2016) and exhibits a porous carbonaceous structure, many functional groups, and an aromatic surface. The production process of biochar mainly consists of slow pyrolysis, hydrothermal carbonization, flash carbonization, and gasification (Tan et al., 2015). Biochar produced from biomass pyrolysis can modify the physicochemical properties of the soil (DeLuca et al., 2006), decrease gaseous N emission (Yanai et al., 2007), alter soil nutrient availability (Chan et al., 2008a), reduce nutrient leaching (Zheng et al., 2013a), and increase crop yields (Major et al., 2010). In addition, biochar produced from pig manure can degrade up to 90.6% of carbaryl (Zhang P et al., 2013). Moreover, soil microbial properties, such as microbial abundance and activity, and mycorrhizal associations were improved after the application of biochar (Warnock et al., 2007; Steiner et al., 2008a). Ren et al. (2016) indicated that biochar amendment of pesticide-polluted soil could enhance its bioremediation. These studies demonstrated that biochar exhibits great potential for maintaining soil fertility, inactivating pesticides through abiotic means, and accelerating biodegradation.

In this paper, we critically review the potential benefits of biochar in the improvement of fertilizer use efficiency through its capacity for increasing nutrient retention (*i.e.*, decreasing nutrient leaching and gaseous nutrient emission) while also permitting nutrient release. Its effect on pesticide chemical degradation and biodegradation is also reviewed. We summarize the physicochemical properties of biochar, analyze the factors and mechanisms influencing various biochar functions, and identify future prospects and knowledge gaps.

PROPERTIES OF BIOCHAR

The physicochemical properties of biochar affect its adsorption properties. For instance, the increase of acidic functional groups in biochar can increase the adsorption of NH_4^+ (Spokas *et al.*, 2012). Biochar has a high specific surface area, high amounts of oxygencontaining functional groups, and a high stability (Huang *et al.*, 2016). The properties of biochar mainly depend on the feedstock and the pyrolysis temperature (Cantrell *et al.*, 2012). Many feedstocks, such as woodchips, organic wastes, plant residues, and poultry manure, can be used to produce biochar (Mohan *et al.*, 2014; Ding *et al.*, 2016a). The common pyrolysis temperature ranges from 200 to 800 °C (Hossain *et al.*, 2011; Song *et al.*, 2012). The impacts of feedstocks and pyrolysis temperature on biochar properties are as shown in Table I.

Specific surface area

The specific surface area of biochar is important because it contributes to the adsorption capacity for both metal ions and organic compounds (Inyang et al., 2012; Zhao et al., 2013). Increasing the pyrolysis temperature can increase the specific surface area and the formation of micropores in biochar. Increasing the pyrolysis temperature from 250 to 600 °C caused the specific surface area of sugarcane bagasse biochar to increase from 0.56 to 14.1 m² g⁻¹ (Ding *et al.*, 2014). Similarly, the specific surface area of soybean stover biochar that was produced at 700 °C was 420 m² g⁻¹, which was extremely high compared to that of biochar produced at 300 °C (6 m² g⁻¹) (Ahmad *et al.*, 2012). One possible explanation is that the release of volatiles within the biochar simply increases with the increase in pyrolysis temperature (Liu X et al., 2014). However, the specific surface area of the biochar also depends on the feedstock used. The specific surface area of biochar produced from bagasse and cocopeat was 202 and 13.7 m² g⁻¹, respectively (Lee *et al.*, 2013). The release of volatile matter mainly produced from celluloses and hemicelluloses during pyrolysis can increase the formation of vascular bundle structure in the biochar and thus improve the specific surface area and pore structure of the biochar (Li et al., 2013). For instance, with the decrease of the volatile matter content within corncob biochar, the specific surface area of the corncob biochar increased from 61.8 to 192.9 $m^2 g^{-1}$ (Liu X et al., 2014). Generally, the influences of feedstock and pyrolysis temperature on the biochar specific surface area may be mainly attributed to the release of volatile matter (Ahmad et al., 2012; Li et al., 2013).

Surface functional groups

The surface chemical compositions of biochar, such as oxygen-containing functional groups, are highly dependent on the pyrolysis temperature, which in turn affects the adsorption properties of the biochar (Pintor *et al.*, 2012). Generally, the hydrogen and oxygen

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TABLE I

$\operatorname{Properties}^{\operatorname{a})}$ of biochars as affected by feeds tocks and pyrolysis temperature

	D I I I	D l ·		G	N.	0	ana	DET (37.11
Study	Feedstock	Pyrolysis temperature	рН	С	Ν	0	CEC	BET surface area	Fixed carbon	Volatile matter
		°C			_ %		$\text{cmol}_{c} \text{ kg}^{-1}$	$m^2 g^{-1}$	· ·	%
Baronti et al. (2014)	Orchard pruning	500	9.80	77.81	0.91	_	101	410	_	_
Méndez et al. (2013)	Sewage sludge	400	7.76	_	_	_	30	33	4.64	23.34
		600	8.72	_	_	_	12	37	4.77	16.70
Kim et al. (2013)	Miscanthus	300	8.28	68.48	0.31	25.71	_	0.56	53.84	41.87
	sacchari florus	400	8.68	81.20	0.42	14.29	_	2.41	68.73	25.70
	Geodae-Uksae 1	500	9.49	86.66	0.40	9.74	_	181.0	80.32	14.98
		600	10.05	90.71	0.33	6.70	_	381.5	88.60	7.70
Zhao et al. (2013)	Cow manure	500	10.2	43.7	-	_	149	21.9	14.7	17.2
	Pig manure	500	10.5	42.7	-	_	82.8	47.4	40.2	11.0
	Shrimp hull	500	10.3	52.1	_	-	389	13.3	18.9	26.6
Genesio et al. (2015)	Sewage sludge	400	7.7	22.6	3.5	-	_	0.1	26.7	21.3
		450	8.2	18.6	2.4	-	_	2.9	27.1	17.3
		500	9.0	17.7	2.3	-	_	3.2	28.2	14.2
		550	9.9	19.6	2.1	-	_	13.3	28.5	13.0
Hale <i>et al.</i> (2013)	Cacao shell	350	10.42	70	1.4	-	37	18.6	-	-
	Corn cob	350	8.97	66	0.6	_	34	36.4	_	-
Chen <i>et al.</i> (2014)	Sewage sludge	500	8.81	17.46	1.54	10.449	76.75	25.424	-	
		600	9.54	18.40	1.38	7.353	30.81	20.268	-	-
		700	11.11	16.92	0.95	6.860	50.34	32.167	-	-
		800	12.18	16.20	0.50	3.641	126.62	48.499	_	_
		900	12.15	15.92	0.53	2.439	247.51	67.603	_	-
Zhao et al. (2013)	Bone dreg	500	9.57	24.2	_	_	87.9	113	10.5	11.0
	Wastewater sludge	500	8.82	26.6	_	-	168	71.6	20.6	15.8
	Waste paper	500	9.88	16.4	_	-	516	133	16.4	30.0
Liu et al. (2010)	Pinewood	700	6.60	95.30	0.12	3.76	_	29	57.05	3.20
Méndez et al. (2012)	Sewage sludge	500	9.54	-	_	-	2.36	32.24	5.7	24.1
Liu X et al. (2014)	Corncob	300	8.1	67.21	0.67	27.63	_	61.8	49.1	43.6
		400	9.1	79.65	0.67	15.72	_	180.1	71.7	20.8
		500	9.3	83.27	0.78	12.62	_	212.6	81.6	8.6
		600	10.4	84.31	0.76	12.52	_	192.9	82.4	7.2
	Sawdust	500	9.5	87.61	0.93	9.31	_	243.1	80.6	13.1
	Cornstalk	500	10.1	77.34	2.79	17.23	_	201.3	76.6	8.9
Inyang et al. (2010)	Undigested bagasse	600	7.7	76.45	0.79	19.83	4.19	14.07	-	-
Inyang et al. (2010)	Digested bagasse	600	10.9	73.55	_	24.04	14.30	17.66	-	-
Lee <i>et al.</i> (2013)	Bagasse	500	9.3	85.59	1.11	10.48	_	202	80.97	9.17
	Cocopeat	500	10.3	84.44	1,02	11.67	_	13.7	67.25	14.30
	Paddy straw	500	10.5	86.28	3.25	7.35	_	45.8	39.10	6.46
	Palm kernel shell	500	6.9	87.85	1.11	8.14	_	191	80.85	12.29
	Wood stem	500	9.5	89.31	0.78	7.34	_	316	83.47	12.79
	Wood bark	500	9.6	84.84	1.83	10.20	_	13.6	68.66	18.14
Zhao et al. (2013)	Peanut shell	500	10.5	73.7	_	-	44.5	43.5	72.9	16.0
	Chlorella	500	10.8	39.3	_	-	562	2.78	17.4	29.3
	Waterweed	500	10.3	25.6	_	_	509	3.78	3.84	32.4
Ahmad et al. (2012)	Soybean stover	300	7.27	68.81	1.88	24.99	_	5.61	-	-
. ,		700	11.32	81.98	1.30	15.45	_	420.3	-	-
	Peanut shell	300	7.76	68.27	1.91	25.89	_	3.14	_	_
		700	10.57	83.76	1.14	13.34	_	448.2	-	-
Zhao et al. (2013)	Sawdust	500	10.5	75.8	_	_	41.7	203	72.0	17.5
× /	Grass	500	10.2	62.1	_	_	84.0	3.33	59.2	18.9
	Wheat straw	500	10.2	62.9	-	-	95.5	33.2	63.7	17.6

^{a)}CEC = cation exchange capacity; BET surface area = Brunauer, Emmett, and Teller (BET) surface area.

contents and the molar hydrogen to carbon (H/C) ratio decrease with the increase of pyrolysis temperature, and a significant increase in the removal of polar functional groups (–OH and C–O) also occurs at high temperature (Ahmad *et al.*, 2012). Also with increasing pyrolysis temperature there is a decrease in the molar oxygen to carbon (O/C) ratio, which results in surfaces featuring a higher abundance of aromatic moieties and increased hydrophobicity (Chen *et al.*, 2008). The number of surface functional groups within the biochar

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decreases with the increase of pyrolysis temperature, as shown by Fourier transformed infrared (FTIR) spectra (Kim *et al.*, 2013; Uchimiya *et al.*, 2013).

Cation exchange capacity (CEC) and pH values

The CEC of biochar indicates its ability to adsorb cations such as NH_4^+ and Ca^{2+} , which are essential nutrients for plants (Rhoades, 1982). Thus, biochar with a high CEC can decrease nutrient leaching losses from soils (Steiner et al., 2008b). The CEC of biochar produced from cordgrass appeared to increase from 8.1 to $44.5 \text{ cmol}_{c} \text{ kg}^{-1}$ and then decreased to $32.4 \text{ cmol}_{c} \text{ kg}^{-1}$ when the pyrolysis temperature increased from 200 to 550 °C (Harvey et al., 2011). Harvey et al. (2011) indicated that the CEC of loblolly pine biochar produced at 200, 300, 350, and 650 °C were 16.5, 16.9, 23.8, and 2.1 cmol_{c} kg⁻¹, respectively. Similarly, the CEC of sugarcane bagasse biochar increased from 6.40 (pyrolyzed at 250 °C) to 9.66 cmol_c kg⁻¹ (pyrolyzed at 500 °C) and then decreased to 4.19 cmol_c kg⁻¹ (pvrolyzed at 600 °C) (Ding et al., 2014). These indicate that biochar produced at high pyrolysis temperature $(i.e., > 500 \,^{\circ}\text{C})$ has a low CEC. The decrease in CEC at high pyrolysis temperature has been attributed to the aromatization of biochar as well as the disappearance of functional groups on biochar (Méndez et al., 2013). Therefore, biochar produced at low temperature has more potential to retain fertilizer cations such as NH_4^+ and thereby improve their utilization efficacy.

The application of biochar can increase soil pH due to the pH of the biochar itself and through enhancing the retention of cations within the soil (*e.g.*, Ca^{2+} , Mg^{2+} , and K^+) (Novak *et al.*, 2009; Sohi *et al.*, 2010). Biochar produced at higher temperature has a higher pH, which is due to the release of alkali salts from the organic matrix of the feedstock (Ahmad *et al.*, 2012). For instance, the pH value of biochar produced from corn straw increased from 9.37 to 11.32 when the pyrolysis temperature increased from 300 to 600 °C (Yuan *et al.*, 2011). The pH values of swine manure biochar produced at 400 and 800 °C were 7.60 and 11.54, respectively (Tsai *et al.*, 2012).

Biochar stability

Biochar is increasingly recognized as a valuable tool for long-term soil amendment (*e.g.*, carbon sequestration, nutrient retention, and pesticide-contaminated soil remediation), but its long-term environmental stability is yet to be fully evaluated. As mentioned previously, biochar stability depends mainly on the pyrolysis temperature and the feedstock. Schmidt *et al.* (2011) reported that certain types of biochar can degrade relatively rapidly in some soils, possibly depending on the conditions under which they were produced, which suggested that pyrolysis could be optimized to generate a more stable biochar. Generally, the stability of biochar can be increased by increasing pyrolysis temperature (Rondon et al., 2007; Mašek et al., 2013). For example, the stability of sugarcane bagasse biochar increased significantly with increasing pyrolysis temperature from 350 to 550 °C (Cross and Soli, 2013). In addition, Cross and Soli (2013) indicated that the stability of biochar produced from chicken manure was lower than that from sugarcane. In fact, biochar stability depends on the amount of recalcitrant carbon substrates (Willmann and Fakoussa, 1997; Hofrichter et al., 1999). Overall, biochar produced at low temperature can be easily degraded, but biochar produced at high temperature is recalcitrant to degradation.

POTENTIAL OF BIOCHAR FOR IMPROVING SOIL FERTILITY

Improvement of fertilizer use efficiency through biochar application

Increasing fertilizer use efficiency can be a useful method to improve crop yields. Chan et al. (2008a) carried out a pot trial to investigate the effects of greenwaste biochar on radish (Raphanus sativus var. Long Scarlet) yields: they found that the application of biochar did not increase the radish yield in the absence of N fertilizer. However, the radish yield evidently increased with biochar application in the presence of N fertilizer, indicating that biochar could efficiently improve plant N utilization. In the presence of N fertilizer (100 kg N ha^{-1}), the radish yield increased by 42% and 96% with biochar application at 10 and 50 t ha^{-1} , respectively (Chan *et al.*, 2008b). The increased yields were attributed to the enhanced N availability brought about by the application of biochar. Zhang et al. (2010) indicated that a biochar application at 40 t ha^{-1} increased rice yield by 12.1% in soil with N fertilization, and the agronomic N use efficiency increased from 1.3 kg of increased grain production per kg of N fertilized with no biochar to 5.3 kg of increased grain production per kg of N fertilized with the biochar application at 40 t ha^{-1} . Additionally, biochar has been shown to increase maize grain yields by 28% and calcium (Ca), magnesium (Mg), K, and P availability by 17%-600% in a field amended with biochar (Major et al., 2010). Therefore, biochar is considered to have a great potential for enhancing plant fertilizer use efficiency by increasing nutrient availability in the soil.

Nutrient retention in biochar-amended soils

The heterogeneous composition of biochar means that its surface can exhibit hydrophilic, hydrophobic, acidic, and basic properties, all of which contribute to the ability of the biochar to adsorb solutes from soil solution, thus affecting nutrient retention. On the one hand, biochar can increase nutrient retention by the adsorption process. For instance, biochar produced from bamboo at 900 °C adsorbed about 1.2 mg $NO_3^$ g^{-1} (Mizuta *et al.*, 2004). Peanut hull and pepperwood biochar, produced at 600 °C, effectively reduced the total amounts of NO_3^- , NH_4^+ , and PO_4^{3-} in the leachates by 34.0%, 34.7%, and 20.6% and 34.3%, 14.4%, and 39.1%, respectively (Yao et al., 2012). Biochar produced from wood at 400 °C adsorbed approximately 250-500 mg P g⁻¹ (Morales *et al.*, 2013). In addition, Harvey et al. (2011) indicated that Spartina spartinae biochar produced at 350 °C can adsorb 0.5 mmol $\rm K^+$ g^{-1} . Therefore, biochar can be added to soil to reduce nutrient leaching. On the other hand, biochar can reduce gaseous N losses. Nelissen et al. (2014) indicated that after the application of biochar, NO and N_2O emissions decreased by 43% and 62%, respectively. Yanai et al. (2007) indicated that N_2O emission decreased by 80% after the addition of biochar. In addition to chemical interactions, the enhanced nutrient retention may also be attributed to improved soil physical properties, such as the increase of porosity and water storage capacity and the decrease of bulk density (Lu et al., 2014). Enhanced biological activities or community shifts may also facilitate nutrient cycling (Pietikäinen et al., 2000). As shown in Table II, nutrient retention by biochar depends strongly on the feedstock used, pyrolysis temperature, and application rate. Overall, biochar has a great potential for improving fertilizer use efficiency by reducing nutrient leaching and gaseous N losses.

Factors affecting nutrient adsorption

Nutrient adsorption depends highly on the biochar pyrolysis temperature. The NH_4^+ adsorption rates of pepperwood biochar produced at 450 °C (15.7%) was higher than that of pepperwood biochar produced at 300 °C (3.8%) and 600 °C (11.9%) (Yao *et al.*, 2012). Morales *et al.* (2013) showed that biochar produced from Embaúba (*Cecropia hololeuca* Miq.) at 600 °C adsorbed 47 and 13 times less P than biochar produced at 400 and 500 °C, respectively. Compared to biochar produced at 300 and 600 °C, biochar produced from sugarcane bagasse at 450 °C had a higher phosphate adsorption rate of 3.1% (Yao et al., 2012). Pinus taeda biochar produced at 200, 300, 350, and 650 °C can adsorb approximately 0.19, 0.17, 0.28, and 0.03 mmol g^{-1} of K⁺, respectively (Harvey *et* al., 2011). These indicate that relatively lower pyrolysis temperature (e.g., 400–500 °C) facilitates nutrient adsorption. Additionally, different feedstocks can exhibit different nutrient (e.g., N, P, and K) adsorption. For instance, NO_3^- adsorption rates were 3.7%, 2.5%, 0.2%, and 0.12% for sugarcane bagasse, bamboo, peanut hull, and pepperwood biochar, respectively (Yao et al., 2012). Yao et al. (2012) also indicated that the NH_4^+ adsorption capacities of these biochars ranged from approximately 2% to 12%. Ingá (Inga edulis Mart.) biochar adsorbs less P than Lacre (Vismia quianenses Aubl.) and Embaúba (Cecropia hololeuca Miq.) biochar (Morales et al., 2013). In addition, Harvey et al. (2011) indicated that biochar (350 °C) produced from Prosopis glandulosa Torr., Pinus taeda, and Spartina spartinae can approximately adsorb 0.19, 0.28, and $0.52 \text{ mmol K}^+ \text{ g}^{-1}$ soil, respectively.

Nutrient leaching from biochar-amended soils varies with the pyrolysis temperature and soil type. Biochar produced from sewage sludge at higher pyrolysis temperature has an improved ability to retain PO_4^{3-} over biochar produced at lower temperature (Yuan et al., 2016). For instance, biochar produced at 300, 500, and 700 °C can reduce PO_4^{3-} leaching in soil by 7.9%, 12.1%, and 23.4%, respectively. Zheng et al. (2013a) indicated that the use of biochar had no significant effect on the cumulative amount of NH_4^+ leached from NH₄⁺-fertilized soils, while it approximately decreased NH_4^+ leaching by 75% from NO_3^- -fertilized soils. The possible reason attributed to this effect is that the NH_4^+ content retained by the acid functional groups on the biochar's surface in the NH₄⁺-fertilized soil contributed to the adsorption capacity. In addition, after biochar application, the adsorption of P was 333, 454, and 667 $mg kg^{-1}$ in brown soil, black soil, and fluvo-aquic soil, respectively (Xu et al., 2014). Different pH values of various soils may contribute to different capacities for nutrient adsorption. Higher soil pH increases the precipitation of exchangeable Al, as new, highly active P-adsorbing surfaces in soils, and the co-precipitation with Ca, Al, and Fe oxides (Haynes, 1982; Agbenin, 1995). Besides, with the increase of biochar application rates from 0 to 20 g kg⁻¹, the total N and total dissolved P leaching decreased by 11% and 69%, respectively (Laird et al., 2010). Similarly, P adsorption capacity increased from 454 to 713 mg kg⁻¹ in black soil with increasing biochar application rates from 1%to 10% (Xu et al., 2014). It is clear therefore that nu-

the decrease of Al content in acid soils are all considered to be important factors in this increased nutrient

TABLE II

Effects of biochar feedstock, pyrolysis temperature, and application rate on nutrient retention

Nutrient retention	Nutri- ent	Feedstock	Pyrolysis tempe- rature	Biochar rate	Nutrient adsorption capacity	Nutrient emission reduction	Reference
			°C		$mg kg^{-1}$	%	
Nutrient	$\rm NH_4^+$	Brazilian pepperwood	300	0.1 g per 50 mL	190		Yao et al. (2012)
adsorption	4		450	0.1 g per 50 mL	785		
			600	0.1 g per 50 mL	595		
		Arundo donax L.	300	$0.35 \mathrm{~g~per} ~12 \mathrm{~mL}$	2101.9		Zheng et al. $(2013b)$
			350	$0.35 \mathrm{~g~per} \ 12 \mathrm{~mL}$	1432.6		
			400	$0.8~{\rm g~per}~12~{\rm mL}$	1043.4		
			500	0.8 g per 12 mL	362.8		
			600	0.8 g per 12 mL	371.8		
	NO_3^-	Sugarcane bagasse	600	0.1 g per 50 mL	636.4		Yao et al. (2012)
		Bamboo	600	0.1 g per 50 mL	430		Yao et al. (2012)
		Peanut hull	600	0.1 g per 50 mL	34.4		Yao et al. (2012)
		Arundo donax L.	600	0.1 g per 50 mL	20.6		Yao et al. (2012)
			500 600	0.3 g per 8 mL	533 5		Zheng et al. (2015b)
	DO^{3-}	Wheet street	250 550	U.5 g per 6 mil	000.0		$\mathbf{V}_{tt} = \mathbf{v}_{t} = \mathbf{v}_{t} = \mathbf{v}_{t} = \mathbf{v}_{t}$
	ro_4	Wheat Straw	550-550	1%	222		Au et al. (2014)
				5%	625		
				10%	769		
				For black soil			
				1%	454		
				5%	625		
				10%	713		
		Sugarcane bagasse	450	$0.1 \mathrm{~g~per} 50 \mathrm{~mL}$	477.4		Yao et al. (2012)
	K^+	$Prosopis\ glandulosa\ {\rm Torr}.$	300	$0.30-0.35 \text{ mL min}^{-1a}$	8.6		Harvey et al. (2011)
			350	$0.30-0.35 \text{ mL min}^{-1a)}$	7.4		
			650	$0.30-0.35 \text{ mL min}^{-1a)}$	2.7		
		$Spartina\ spartinae$	300	0.30–0.35 mL min ^{$-1a$})	19.5		Harvey et al. (2011)
			350	$0.30-0.35 \text{ mL min}^{-1a)}$	20.3		
			550	$0.30-0.35 \text{ mL min}^{-1a)}$	12.5		
		Pinus taeda	300	$0.30-0.35 \text{ mL min}^{-1a)}$	6.6		Harvey et al. (2011)
			350	$0.30-0.35 \text{ mL min}^{-1a)}$	10.9		
			650	$0.30-0.35 \text{ mL min}^{-1a}$	1.2		
Gaseous N emission reduction	NO	Willow	< 550	20 t ha^{-1}		79	Nelissen et al. (2014)
			≥ 550	20 t ha^{-1}		92	
		Pine	< 550	20 t ha^{-1}		80	Nelissen et al. (2014)
			≥ 550	20 t ha^{-1}		89	
		Maize	< 550	20 t ha^{-1}		61	Nelissen $et al.$ (2014)
	NO		≥ 550	20 t ha^{-1}		94	
	N_2O	Salix dasyclados	700	1.5%		50 50	Ameloot <i>et al.</i> (2013)
		Arrundo donar I	700 600	1.3%		07 5	$ \begin{array}{l} \text{Americot et al. (2013)} \\ \text{Wang et al. (2013)} \end{array} $
		Oak pellet	550	1%		53.0	Stewart et al. (2013)
		Oak penet	550	5%		72.4	Stewart et al. (2013)
				10%		76.3	
				20%		83.5	
		Willow	< 550	20 t ha^{-1}		34	Nelissen et al. (2014)
			≥ 550	20 t ha^{-1}		41	· /
		Pine	< 550	20 t ha^{-1}		50	Nelissen et al. (2014)
			≥ 550	$20 \text{ t } \text{ha}^{-1}$		91	
		Maize	< 550	20 t ha^{-1}		78	Nelissen et al. (2014)
			≥ 550	20 t ha^{-1}		85	

 $^{\rm a)}{\rm Flow}$ rate of a solution with 50–60 mg of biochar.

retention (Guo *et al.*, 2014). Overall, the major influencing factors, such as pyrolysis temperature, feedstock, biochar application rate, and soil type, should be taken into account before applying biochar into the soil.

Factors reducing gaseous N emission

Biochar pyrolysis conditions, especially pyrolysis temperature, can influence the mitigation of gaseous N losses from the soil. For instance, compared to the control soil, willow biochar decreased NO emission by approximately 59% for low pyrolysis temperature (< 550 °C) and by 45% for high pyrolysis temperature $(\geq 550 \text{ °C})$ (Nelissen *et al.*, 2014). Moreover, Nelissen et al. (2014) indicated that N₂O emission decreased by about 46% for biochar pyrolyzed at low temperature (< 550 °C) and by 53% for biochar pyrolyzed at high temperature (≥ 550 °C). Compared to the addition of wood biochar produced at 350 °C, N₂O emission decreased by 50% after the addition of biochar produced at 700 °C (Ameloot et al., 2013). In addition, the mitigation of gaseous N losses also depends on the feedstock used. Nelissen et al. (2014) indicated that N_2O emission decreased by about 40%, 46%, and 60% with the use of pine, willow, and maize biochar, respectively, all produced at low pyrolysis temperature (< 550 °C). Overall, lower-temperature biochar is more efficient in reducing NO emission, while higher-temperature biochar is more efficient in decreasing N_2O emission.

Nutrient retention mechanisms

To predict the change of nutrient adsorption after application of biochar, it is necessary to identify the underlying mechanisms of the adsorption process. The adsorption behavior of biochar for different nutrients (*e.g.*, NH_4^+ , NO_3^- , and PO_4^{3-}) differs with adsorption mechanisms depending on the biochar properties, including pore structure, surface functional groups, and mineral components (Yao *et al.*, 2012; Zheng *et al.*, 2013b).

The adsorption mechanisms of polar and apolar compounds are mainly attributed to chemisorption, including hydrophobic bonding and π - π electron donor-acceptor interactions, resulting from fused aromatic carbon structures (Swiatkowski *et al.*, 2004; Zhang P *et al.*, 2013). The adsorption mechanisms of NH₄⁺ onto biochar include physical adsorption (van der Waals adsorption), attraction to negatively charged surfaces, reaction with acidic functional groups (*e.g.*, carboxylate (-COOH) and hydroxyl (-OH)) to form amides and amines, binding to cationic species sites on the biochar surface, and π - π electron donor-acceptor interactions (Hale *et al.*, 2013; Zheng *et al.*, 2013b; Zhang *et al.*, 2015). Additionally, the adsorption mechanisms of NO₃⁻ mainly include anion exchange (Thangarajan *et al.*, 2015), non-electrostatic sorption of NO₃⁻ through the micropores of biochar or diffusion of NO₃⁻ onto the biochar surface. Dissimilarly, P adsorption is due to the precipitation of exchangeable Al as new, highly active P adsorption sites or due to co-precipitation with Al and Fe oxides (Haynes, 1982; Agbenin, 1994).

Biochar can reduce N₂O and NO emissions, transformed from NO_3^- and NH_4^+ by denitrifying and nitrifying bacteria, respectively (Van Zwieten et al., 2009; Spokas et al., 2010). The potential explanations and mechanisms for mitigation of N₂O emission are abiotic interactions, which can improve the aerobic conditions in soil, such as through a decrease in bulk density allowing infiltration of water and through an increase in nutrient availability and opening of soil structure (Spokas et al., 2009; Beesley et al., 2011; Taghizadeh-Toosi et al., 2012). In addition, the most likely mechanisms for mitigation of NO emission include: i) stimulated NH₃ volatilization, ii) biotic N immobilization, and iii) non-electrostatic sorption of NH_4^+ (Nelissen et al., 2014). Taghizadeh-Toosi et al. (2011) observed decreased NO_3^- concentrations with the addition of biochar, and attributed this to the reduced availability of inorganic N for nitrification. They suggested that biochar could stimulate NH₃ emission from high-pH micro-sites close to biochar particles and then NH₃ could be adsorbed by the acidic groups on the biochar surface, especially by oxygen-containing functional groups (Zhang X et al., 2013).

Nutrient release from biochar

Various studies have indicated that biochar can affect nutrient availability and that it has a great potential as a slow-release fertilizer in soil (e.g., Dinget al., 2016b). The release of nutrients from biochar (including the adsorbed nutrients) mainly depends on desorption properties of the biochar. Several factors can have significant effects on nutrient desorption from biochar. Zhang et al. (2015) indicated that desorption rates of NH_4^+ from hardwood biochar increased from 18% to 31%, as a result of a decrease in pyrolysis temperature from 600 to 400 °C. In black soil, the average percentage of P desorbed over lower P loads (20 mg L^{-1}) increased from 36% to 41% with increasing biochar application rates from 0 to 10% (Xu et al., 2014). Xu et al. (2014) indicated that more than 60% of the P adsorbed by biochar was released at higher P loadings (100 and 240 mg L^{-1}). These suggest

that the desorption percentage of P could increase with increasing biochar application rates and P loadings. In addition, cacao shell biochar desorbed 1484 mg kg⁻¹ of PO_4^{3-} and corn cob biochar desorbed 172 mg kg⁻¹ of PO_4^{3-} (Hale *et al.*, 2013). Therefore, biochar desorption properties mainly depend on the pyrolysis temperature, type of feedstock, and biochar application rate. Therefore, it is deemed that various types of biochar can be used to manage different soil nutrients in the same soil, or they can be used preferentially in different soils to acquire the desired nutrient supply effects.

POTENTIAL OF BIOCHAR FOR ENHANCING RE-MEDIATION OF PESTICIDE-CONTAMINATED SOILS

Pesticide removal by microorganisms

As shown in Table III, several microorganisms including bacteria and fungi have been studied for pesticide removal. For instance, Brevibacterium aureum DG-12, a new bacterial strain isolated from active sludge, can degrade and utilize cyfluthrin as a growth substrate and has resulted in an 88.6% degradation of cyfluthrin (50 mg L^{-1}) within 5 d (Chen *et al.*, 2013). Singh and Singh (2011) reported that the Achromobacter xylosoxidans strain C8B degraded 94.1%, 84.5%, and 80.1% of α -endosulfan, β -endosulfan and endosulfan sulfate, respectively. Microbial consortia can be applied for degradation of pesticides, but their efficiency in soil mainly depends on the availability of metabolites or pesticides to the microorganisms and the physiological status of the microorganisms (Hai et al., 2012). Therefore, the removal efficiencies of individual microorganisms are not always high for most pesticides in soil. For example, when a single microorganism (Acinetobacter baumannii, Klebsiella oxytoca, and Stenotrophomonas maltophilia) was used, the degradation efficiency of phenanthrene was very low (48.0%, 11.0%, and 9.0%, respectively) after 360 h of cultivation (Kim et al., 2009). Moreover, most studies have reported pesticide biodegradation efficiencies under optimal conditions, including the most suitable pH, organic matter, soil moisture, and pesticide concentration. Chen et al. (2013) indicated that the cyfluthrin degradation efficiency increased from 54.0% to 88.6% when pH values increased from 5.6to 7.8, and then decreased to 67.0% when pH reached 9.6. In addition, a certain level of organic matter is necessary for pesticide (e.g., dufulin) biodegradation because it can ensure the presence of an active autochthonous microbial population; the dufulin degradation efficiency also increased from 43.6% to 91.4%when soil moisture increased from 20% to 80% (Wang H Z et al., 2014), showing that water limitation can also inhibit biodegradation efficiency. Chen *et al.* (2013)suggested that high concentrations of cyfluthrin (300-600 mg L^{-1}) decreased the biodegradation efficiency (by 78.5%-50.2%), which might be attributed to the microbial growth starting slowly and requiring an acclimation period before accelerated degradation could occur at high pesticide concentrations (Chen et al., 2012). These external environmental factors can affect microorganism properties (e.g., microbial abundance)and activity) and thus affect the pesticide degradation efficiency. Therefore, feasible and effective methods are needed to improve the local environment for microorganisms and enhance pesticide degradation.

Potential of biochar for enhancing pesticide degradation in soil

Biochar has been reported to enhance pesticide degradation by microorganisms in soil (Ren et al., 2016). On the one hand, with the amendment of biochar, pesticide biodegradation may depend on both the increased activities of natural microorganisms in soil and the lowered bioavailability of the pesticide. Many studies have demonstrated that biochar can improve the local environment for microbial activity, including changing soil pH (Wang Y et al., 2014), increasing soil organic matter (Lin et al., 2012), enhancing soil water content (Peake et al., 2014), providing habitat (Fig. 1), and reducing the competition from other microbes by isolation (Lehmann et al., 2011), thus improving the microbial potential (e.g., more diverse microbial community composition, abundance, and functional capacities) (Ding et al., 2016b). However, biochar amendment can also enhance pesticide sorption in soil (Yu et al., 2009; Zheng et al., 2010), which can reduce the concentration of pesticide in the soil solution and decrease the bioavailability of pesticide to microorganisms (Saito et al., 2011). On the other hand, chemical degradation and biodegradation are the two main pathways for pesticide removal in soil with biochar amendment (Ren et al., 2016). Generally, chemical hydrolysis is an important path of chemical degradation and abiotic degradation in soil since modern pesticides are designed to be easily degraded (Zhang P et al., 2013). Some studies have found that pesticide hydrolysis in soil could be catalyzed by amendment with biochar, which was attributed to the combined effects of elevated pH, released dissolved metal ions, and the active groups on mineral surface (Zhang P et

BENEFITS OF BIOCHAR IN AGRICULTURAL SOILS

TABLE III

Effects of microorganisms on pesticide degradation

Microorganism	Pesticide	Effect	Reference	
Brevibacterium aure- um DG-12	Cyfluthrin	Cyfluthrin can be utilized as a growth substrate and 88.6% of cyfluthrin (50 mg L^{-1}) can be degraded by <i>Brevibacterium aureum</i> DG-12 within 5 d.	Chen <i>et al.</i> (2013)	
Pandoraea species	γ -hexachlorocyclohexane (HCH) (lindane) and α - hexachlorocyclohexane	After 8 weeks of incubation in liquid culture, 89.9% and 93.3% of the γ - and α -HCH isomers were degraded by <i>Pandoraea</i> species at an initial concentration of 150 mg L ⁻¹ , respectively.	Benedict <i>et al.</i> (2002)	
Streptomyces sp. M7	Lindane	Lindane was degraded by <i>Streptomyces</i> sp. under ae- robic conditions, and between 72 and 96 h, a maximum of about 86% of the Cl ⁻ was released when lindane was added to the medium at 20 h.		
Penicillium lilacinum BP303	Organophosphorus pesticides $(e.g., paraoxon, parathion, and coumaphos)$	Organophosphorus pesticides could be degraded effectively by hydrolase from <i>Penicillium lilacinum</i> BP303, and the enzyme activity was optimal at 45 $^{\circ}$ C and pH 7.5.	Liu <i>et al.</i> (2004)	
A mixed culture of ba- cteria and white-rot fungi (<i>Coriolus versi-</i> <i>color</i> , NBRC 9791)	Aldicarb, atrazine, and alachlor	During an incubation period of 14 d, the mixed bacteria-fungus culture achieved 47%, 98%, and 62% removal of aldicard, atrazine and, alachlor from the liquid phase, respectively.	Hai <i>et al.</i> (2012)	
Achromobacter xylo- soxidans strain C8B	α -endosulfan, β -endosulfan, endosulfan sulfate	During an incubation period of 20 d at 28 °C, 94.1% α - endosulfan, 84.5% β -endosulfan and 80.1% endosulfan sulfate were degraded by <i>Achromobacter xylosoxidans</i> strain C8B.	Singh and Singh (2011)	
Pseudomonas aerugi- nosa	Endosulfan	Endosulfan degradation by <i>Pseudomonas aeruginosa</i> in the medium with coffee bean was 51% after 7 d of incubation at 30 °C with shaking at 100 r min ⁻¹ .	Barragán- Huerta <i>et al.</i> (2007)	
Cyathus bulleri and Phanerochaete sordida	Lindane	Cyathus bulleri was found to be a slightly more efficient lindane biodegrader than Phanerochaete sor- dida. Moreover, the removal efficiency of lindane by Cyathus bulleri was 97% after 28 d of incubation at a concentration of 0.27 μ mol L ⁻¹ under 28 °C.	Singh and Kuhad (2000)	



Fig. 1 Visual observation of colonization of biochar by microorganisms: fresh biochar showing fungal hyphae (Lehmann and Joseph, 2009) (a), ESEM images of inoculated biochar by bacterial strain (Hale *et al.*, 2014) (b), adhesion of *Escherichia coli* (white arrows) on activated carbon (George and Davies, 1988) (c), and fresh corn stover biochar showing microorganisms in pores (white arrows) (Jin, 2010) (d).

al., 2013). Overall, amendment with biochar may be a feasible and effective method to enhance the bioremediation of pesticide residues in soil.

Chemical degradation of pesticides in biochar-amended soils

Biochar can increase abiotic pesticide hydrolysis in soil due to its catalytic effects. For instance, in order to evaluate the effects of hydrolysis on carbaryl degradation, Ren et al. (2016) sterilized experimental soils through autoclaving at 120 °C for 30 min, and they determined that the degradation rate increased from 44.3% in the control soil to 55.0% in the biocharamended soil. However, the catalytic effects of biochar on pesticide hydrolysis depend on many factors, such as feedstock, pyrolysis temperature, and application rate. The hydrolysis rates of carbaryl for pig manure biochar and maize straw biochar produced at $350 \text{ }^{\circ}\text{C}$ were 55.0% and 52.8%, respectively (Ren *et al.*, 2016). In addition, biochar produced from pig manure at 350 and 700 $^{\circ}$ C can hydrolyze 59.1% and 90.6% of carbaryl and 21.2% and 63.4% of atrazine, respectively (Zhang P et al., 2013). However, Ren et al. (2016) reported that with the increase of pyrolysis temperature of rice straw biochar from 350 to 700 °C, the hydrolysis rate of carbaryl decreased from 53.7% to 50.0%. With the application dosage of biochar increasing from 50 to 500 mg, the hydrolysis rates of carbaryl and atrazine increased from 23.5% to 59.1% and from 12.6% to 21.2%, respectively (Zhang P et al., 2013). It is possible that several changes caused by different biochar applications, such as those in pH, dissolved metal ions, active groups, and pesticide sorption, are responsible for the above phenomena.

The pH can significantly affect pesticide chemical hydrolysis. For instance, hydrolysis of the carbamate ester bond in carbaryl is typically base catalyzed (Liu et al., 2000), while atrazine is a moderately persistent herbicide that can be hydrolyzed in strong acidic or alkaline solutions (Mandelbaum et al., 1993). Therefore, an elevation in pH that is induced by biochar could enhance carbaryl hydrolysis, but it cannot always facilitate atrazine hydrolysis. Moreover, the accumulation of nucleophiles on the biochar surface can facilitate pesticide hydrolysis. Furthermore, hydroxyl groups on the biochar surface may act as nucleophiles, and bound metal ions on the biochar surface may coordinate a hydrolysable moiety, forming complexes with pesticides, which can facilitate the nucleophilic attack of water molecules (Schwazenbach et al., 2005). Similarly, released metal ions from biochar can catalyze pesticide hydrolysis by forming metal hydroxyl species (nucleophiles bound to metal centers), and their nucleophilicities are higher than those of bulk water (Schwazenbach *et al.*, 2005). In addition, Jones *et al.* (2011) indicated that pesticide sorption could be increased in biochar-amended soil, thus reducing the concentration of free pesticide in the soil solution, which could hinder the hydrolytic process and lower the degradation rate. In fact, these aspects should be studied together to determine the pesticide hydrolysis rate. Overall, in soils amended with biochar, pesticide hydrolysis could be enhanced through the catalytic effects, but reduced by the increased pesticide sorption. Therefore, the toxicity and mobility of both the original pesticide and the decomposition products should be taken into account when assessing pesticide control strategies.

Pesticide biodegradation in biochar-amended soil

Biochar can not only catalyze pesticide hydrolysis but also affect microorganism activities and pesticide bioavailability, consequently influencing pesticide biodegradation (Fig. 2). Compared to the sterile soil, the degradation efficiency of carbaryl increased from 55.0% to 75.0% in the unsterile soil, with a 0.5% application rate of pig manure biochar that was produced at 350 °C, after 40 d of incubation (Ren et al., 2016), indicating an enhanced pesticide biodegradation. Furthermore, the enhancement of pesticide biodegradation through biochar application varied with the feedstock type, pyrolysis temperature, and application rate. For instance, Ren et al. (2016) reported that with an application rate of 0.5%, the enhancement of carbaryl biodegradation ranged from 19.5% to 27.3% for the biochar pyrolyzed from rice straw, pig manure, and maize straw at 350 °C, respectively, and from 3.1% to 27.3% for maize straw biochar produced at 350 and 700 °C. Moreover, for pig manure biochar produced at 350 °C, the enhancement ranged from 13.8% to 20.0%with the application rates of 0.5% and 5%. The effects of biochar on the microbial properties and pesticide bioavailability are highly dependent on its properties. Biochar containing high contents of amorphous carbon and dissolved organic matter can enhance the microbial activity, since these compounds can constitute a viable carbon source for growth of microorganisms (Lehmann et al., 2011). The change of soil pH caused by biochar application can significantly affect (generally decrease) the activity of native microorganisms (Ding et al., 2016b). In addition, after biochar addition, the enhanced sorption of a pesticide can lead to reduction in its concentration in the soil solution, thus decreasing pesticide bioavailability and biodegradation (Jones et al., 2011). Loganathan et al. (2009) reported



Fig. 2 Possible benefits from biochar application for fertilizer use efficiency and bioremediation of pesticide-polluted soils.

that biochar reduced the microbial atrazine mineralization by influencing the sorption and desorption processes, thus reducing the bioavailability of atrazine. Therefore, biodegradation of a pesticide may be influenced by its sorption in biochar-amended soil.

Pesticide sorption in biochar-amended soils

Pesticide sorption in biochar-amended soils can reduce its mobility, volatilization, leaching, and uptake by plants (Chen *et al.*, 2009). However, as discussed above, the amendment of biochar could enhance pesticide sorption in soil, which may reduce the pesticide concentration in the soil solution and the pesticide bioavailability to the microorganisms, thus decreasing the chemical degradation and biodegradation of the pesticide. Though both biochar and soil can sorb the pesticide, biochar was more effective in pesticide sorption compared to soil (Martin *et al.*, 2012). Saito et al. (2011) demonstrated that the amendment of woodchip biochar decreased the concentration of dieldrin from 0.055 to 0.04 mg kg^{-1} in soil solution. The sorption capacity of pesticide on biochar depends highly on biochar properties, including its organic carbon content, aromaticity, specific surface area, and ash content (Fang et al., 2014). Pesticide sorption mechanisms are generally complex, as shown in Fig. 3, and include hydrogen bonds, electrostatic attractions, pore-filling, partition into uncarbonized fractions, hydrophobic effects, and π - π interactions (Tan *et al.*, 2015). For instance, Zhang P et al. (2013) indicated that both hydrophobic effects and pore-filling and π - π electron donor-acceptor interactions contributed to carbaryl and atrazine adsorption. Besides, some pesticides are weak bases and exist as neutral molecules,



therefore being able to form weak hydrogen bonds to carboxyl groups or the clay surface through their heterocyclic N atoms (Inyang *et al.*, 2014).

FUTURE PROSPECTIVES

For biochar-amended field soils in the long term, the soil properties may differ significantly from those in laboratory-based short-term experiments, such as column and leaching studies. Nutrients released from 'fresh' biochar are responsible for short-term increases in crop growth (Lehmann et al., 2003). However, Liang et al. (2006) hypothesized that the long-term effects of biochar on the soil nutrient availability are due to an increase in surface oxidation and CEC, which intensifies over time (Cheng et al., 2006, 2008) and can result in a greater nutrient retention in the 'aged' biochar compared to the 'fresh' biochar. This mechanism needs to be demonstrated under field conditions over many years. Major et al. (2010) investigated the long-term (four years) effects of a single biochar on soil fertility and crop maize yield, and they hypothesized that the biochar-amended soil could provide more sites for basecation retention in acid tropical soils, thus enhancing nutrient retention and improving soil fertility. Nevertheless, long-term studies quantitatively determining nutrient dynamics in biochar-amended soils are still lacking. Further studies should be focused on the prediction of nutrient dynamics in biochar-amended soils by establishing and improving the available kinetics models under both laboratory and field conditions. To study nutrient dynamics, it is essential to understand the various mechanisms affecting soil nutrient availability and fertility over time.

Biochar has a strong pesticide sorption capacity (Zhang P et al., 2013), and it can accumulate pesticide residues in soil. In most short-term studies, the release of pesticide from biochar, which could act as a new source of pollution, has not been taken into account. Therefore, it is desirable to evaluate the longterm environmental fate of sequestered pesticides. At present, the application of biochar for the remediation of pesticide-polluted soils is mainly based on laboratory, greenhouse, or small-plot short-term experiments. Nevertheless, field conditions are complicated, and biochar properties can be altered with time due to aging, oxidation, or microbial degradation, all affecting both the sorption and hydrolysis capacities of biochar for pesticides. Therefore, large-scale, long-term field trials are necessary in future studies.

CONCLUSIONS

Integrated nutrient and pesticide management strategies are necessary in order to increase sustainable agricultural productivity and to conserve natural resources. In agriculture development, fertilizers and pesticides are important plant nutritional and protective agents for boosting crop production. However, the use efficiency of fertilizers in crop systems is usually very low. Moreover, the indiscriminate use of pesticides can cause severe environmental contamination. Amending soil with biochar to enhance plant nutrient uptake and pesticide degradation may be a suitable way to ameliorate these problems.

Due to the large surface area, high intensity of diverse functional groups, and good stability, biochar can be used to improve the fertilizer use efficiency and soil fertility. Biochar not only enhances adsorption of nutrients $(e.g., NO_3^-, NH_4^+, and PO_4^{3-})$, therefore decreasing nutrient leaching, but can also mitigate gaseous N losses. Moreover, the sorbed nutrients can later be released from biochar into the soil (slow-release fertilizer). In addition, biochar has the potential to facilitate the degradation of pesticides in soil and reduce the plant uptake of pesticides. On the one hand, the amendment of soil with biochar may enhance the removal rates of pesticides through catalyzing the chemical hydrolysis process. On the other hand, microbial activities could be increased after the application of biochar into polluted soil. Notably, the sorption of pesticide onto biochar can decrease the free pesticide concentration in soil solution, hindering the hydrolytic process (chemical degradation) and reducing pesticide bioavailability (biodegradation). Therefore, pesticide sorption in biochar-amended soil may have negative effects on degradation. In practical applications, the function of biochar is dependent on its feedstock, pyrolysis temperature, and application rate. Overall, the amendment of soil with biochar appears to enhance fertilizer use efficiency, soil fertility, and pesticide degradation and shows potential to improve overall soil health and crop yields, thus improving the sustainability of agriculture.

Further studies should be focused on the knowledge gaps described above, and consider the following: 1) The effects of biochar on soil properties over a long period of time in large-scale field trials are more relevant than short-term laboratory studies; 2) biochar characteristics vary with different feedstock and pyrolysis conditions, thus making it necessary to produce biochar specifically designed for soil management according to the properties of the soil and the environmental conditions; 3) the dynamic mechanisms of pesticides between the microorganisms and the biochar should be understood in order to maximize pesticide remediation efficiency; 4) further research is required to comprehensively explore the influential factors for pesticide degradation by microorganisms and biochar; and 5) the synthesis and application of functionalized biochar as a potential material for soil amendment and remediation should be evaluated.

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REFERENCES

- Agbenin J O. 1994. Adsorbed phosphorus partitioning in some benchmark soils from Northeast Brazil. Fert Res. 40: 185– 191.
- Agbenin J O. 1995. Phosphorus sorption by three cultivated savanna Alfisols as influenced by pH. Fert Res. 44: 107–112.
- Ahmad M, Lee S S, Dou X M, Mohan D, Sung J K, Yang J E, Ok Y S. 2012. Effects of pyrolysis temperature on soybean stover- and peanut shell-derived biochar properties and TCE adsorption in water. *Bioresource Technol.* 118: 536–544.
- Ameloot N, De Neve S, Jegajeevagan K, Yildiz G, Buchan D, Funkuin Y N, Prins W, Bouckaert L, Sleutel S. 2013. Shortterm CO₂ and N₂O emissions and microbial properties of biochar amended sandy loam soils. *Soil Biol Biochem.* 57: 401–410.
- Baronti S, Vaccari F P, Miglietta F, Calzolari C, Lugato E, Orlandini S, Pini R, Zulian C, Genesio L. 2014. Impact of biochar application on plant water relations in *Vitis vinifera* (L.). Eur J Agron. 53: 38–44.
- Barragán-Huerta B E, Costa-Pérez C, Peralta-Cruz J, Barrera-Cortés J, Esparza-García F, Rodríguez-Vázquez R. 2007. Biodegradation of organochlorine pesticides by bacteria grown in microniches of the porous structure of green bean coffee. Int Biodeter Biodeg. 59: 239–244.
- Beesley L, Moreno-Jiménez E, Gomez-Eyles J L, Harris E, Robinson B, Sizmur T. 2011. A review of biochars' potential role in the remediation, revegetation and restoration of contaminated soils. *Environ Pollut.* 159: 3269–3282.
- Benedict C O, Tariq S, Marta C A, William T F. 2002. Biodegradation of γ-hexachlorocyclohexane (lindane) and αhexachlorocyclohexane in water and a soil slurry by a Pandoraea species. J Agr Food Chem. 50: 2548–2555.
- Benimeli C S, Castro G R, Chaile A P, Amoroso M J. 2006. Lindane removal induction by *Streptomyces* sp. M7. J Basic Microbiol. 46: 348–357.
- Cantrell K B, Hunt P G, Uchimiya M, Novak J M, Ro K S. 2012. Impact of pyrolysis temperature and manure source on physicochemical characteristics of biochar. *Bioresource Tech*nol. **107**: 419–428.
- Chakoosari M M D. 2013. Efficacy of various biological and microbial insecticides. J Biol Today's World. 2: 249–254.
- Chan K Y, Van Zwieten L, Meszaros I, Downie A, Joseph S. 2008a. Agronomic values of greenwaste biochar as a soil amendment. Aust J Soil Res. 45: 629–634.

- Chan K Y, Van Zwieten L, Meszaros I, Downie A, Joseph S. 2008b. Using poultry litter biochars as soil amendments. *Aust J Soil Res.* 46: 437–444.
- Chen B L, Zhou D D, Zhu L Z. 2008. Transitional adsorption and partition of nonpolar and polar aromatic contaminants by biochars of pine needles with different pyrolytic temperatures. *Environ Sci Technol.* 42: 5137–5143.
- Chen H, He X H, Rong X M, Chen W L, Cai P, Liang W, Li S Q, Huang Q Y. 2009. Adsorption and biodegradation of carbaryl on montmorillonite, kaolinite and goethite. *Appl Clay Sci.* 46: 102–108.
- Chen S H, Luo J J, Hu M J, Geng P, Zhang Y B. 2012. Microbial detoxification of bifenthrin by a novel yeast and its potential for contaminated soils treatment. *PLoS One.* **7**: e30862.
- Chen S H, Dong Y H, Chang C Q, Deng Y Y, Zhang X F, Zhong G H, Song H W, Hu M Y, Zhang L H. 2013. Characterization of a novel cyfluthrin-degrading bacterial strain *Brevibacterium aureum* and its biochemical degradation pathway. *Bioresource Technol.* 132: 16–23.
- Chen T, Zhang Y X, Wang H T, Lu W J, Zhou Z Y, Zhang Y C, Ren L L. 2014. Influence of pyrolysis temperature on characteristics and heavy metal adsorptive performance of biochar derived from municipal sewage sludge. *Bioresource Technol.* 164: 47–54.
- Cheng C H, Lehmann J, Engelhard M H. 2008. Natural oxidation of black carbon in soils: changes in molecular form and surface charge along a climosequence. *Geochim Cosmochim Acta.* **72**: 1598–1610.
- Cheng C H, Lehmann J, Thies J E, Burton S D, Engelhard M H. 2006. Oxidation of black carbon by biotic and abiotic processes. Org Geochem. 37: 1477–1488.
- Cross A, Sohi S P. 2013. A method for screening the relative long-term stability of biochar. *Glob Change Biol Bioenerg.* 5: 215–220.
- DeLuca T, MacKenzie M D, Gundale M J, Holben W E. 2006. Wildfire-produced charcoal directly influences nitrogen cycling in ponderosa pine forests. Soil Sci Soc Am J. 70: 448– 453.
- Ding W C, Dong X L, Ime I M, Gao B, Ma L Q. 2014. Pyrolytic temperatures impact lead sorption mechanisms by bagasse biochars. *Chemosphere.* 105: 68–74.
- Ding Y, Liu Y G, Liu S B, Li Z W, Tan X F, Huang X X, Zeng G M, Zhou Y Y, Zheng B H, Cai X X. 2016a. Competitive removal of Cd(II) and Pb(II) by biochars produced from water hyacinths: performance and mechanism. RSC Adv. 6: 5223–5232.
- Ding Y, Liu Y G, Liu S B, Li Z W, Tan X F, Huang X X, Zeng G M, Zhou L, Zheng B H. 2016b. Biochar to improve soil fertility. A review. Agron Sustain Dev. 36: 36.
- Fang Q L, Chen B L, Lin Y J, Guan Y T. 2014. Aromatic and hydrophobic surfaces of wood-derived biochar enhance perchlorate adsorption via hydrogen bonding to oxygen-containing organic groups. *Environ Sci Technol.* 48: 279–288.
- Genesio L, Miglietta F, Baronti S, Vaccari F P. 2015. Biochar increases vineyard productivity without affecting grape quality: results from a four years field experiment in Tuscany. *Agric Ecosyst Environ.* 201: 20–25.
- George N, Davies J T. 1988. Parameters affecting adsorption of microorganisms on activated charcoal cloth. J Chem Technol Biotechnol. 43: 173–186.
- Guo Y J, Tang H, Li G D, Xie D T. 2014. Effects of cow dung biochar amendment on adsorption and leaching of nutrient from an acid yellow soil irrigated with biogas slurry. *Water Air Soil Pollut.* 225: 1820.

- Hai F I, Modin O, Yamamoto K, Fukushi K, Nakajima F, Nghiem L D. 2012. Pesticide removal by a mixed culture of bacteria and white-rot fungi. J Taiwan Inst Chem Eng. 43: 459–462.
- Hale L, Luth M, Kenney R, Crowley D. 2014. Evaluation of pinewood biochar as a carrier of bacterial strain *Enterobac*ter cloacae UW5 for soil inoculation. Appl Soil Ecol. 84: 192–199.
- Hale S E, Alling V, Martinsen V, Mulder J, Breedveld G D, Cornelissen G. 2013. The sorption and desorption of phosphate-P, ammonium-N and nitrate-N in cacao shell and corn cob biochars. *Chemosphere.* **91**: 1612–1619.
- Harvey O R, Herbert B E, Rhue R D, Kuo L J. 2011. Metal interactions at the biochar-water interface: energetics and structure–sorption relationships elucidated by flow adsorption microcalorimetry. *Environ Sci Technol.* 45: 5550–5556.
- Haynes R. 1982. Effects of liming on phosphate availability in acid soils. *Plant Soil.* 68: 289–308.
- Hofrichter M, Ziegenhagen D, Sorge S, Ullrich R, Bublitz F, Fritsche W. 1999. Degradation of lignite (low-rank coal) by ligninolytic basidiomycetes and their manganese peroxidase system. Appl Microbiol Biotechnol. 52: 78–84.
- Hossain M, Strezov V, Chan K Y, Ziolkowski A, Nelson P F. 2011. Influence of pyrolysis temperature on production and nutrient properties of wastewater sludge biochar. J Environ Manage. 92: 223–228.
- Huang X X, Liu Y G, Liu S B, Tan X F, Ding Y, Zeng G M, Zhou Y Y, Zhang M M, Wang S F, Zheng B H. 2016. Effective removal of Cr(VI) using β-cyclodextrin–chitosan modified biochars with adsorption/reduction bifuctional roles. *RSC Adv.* 6: 94–104.
- Inyang M, Gao B, Pullammanappallil P, Ding W C, Zimmerman A R. 2010. Biochar from anaerobically digested sugarcane bagasse. *Bioresource Technol.* **101**: 8868–8872.
- Inyang M, Gao B, Yao Y, Xue Y W, Zimmerman A R, Pullammanappallil P, Cao X D. 2012. Removal of heavy metals from aqueous solution by biochars derived from anaerobically digested biomass. *Bioresource Technol.* 110: 50–56.
- Inyang M, Gao B, Zimmerman A, Zhang M, Chen H. 2014. Synthesis, characterization, and dye sorption ability of carbon nanotube–biochar nanocomposites. *Chem Eng J.* 236: 39– 46.
- Jin H Y. 2010. Characterization of microbial life colonizing biochar and biochar-amended soils. Ph.D. Dissertation, Cornell University.
- Jones D L, Edwards-Jones G, Murphy D V. 2011. Biochar mediated alterations in herbicide breakdown and leaching in soil. *Soil Biol Biochem.* 43: 804–813.
- Kim W K, Shim T, Kim Y S, Hyun S, Ryu C, Park Y K, Jung J. 2013. Characterization of cadmium removal from aqueous solution by biochar produced from a giant *Miscanthus* at different pyrolytic temperatures. *Bioresource Technol.* 138: 266–270.
- Kim Y M, Ahn C K, Woo S H, Jung G Y, Park J M. 2009. Synergic degradation of phenanthrene by consortia of newly isolated bacterial strains. J Biotechnol. 144: 293–298.
- Kuranchie-Mensah H, Atiemo S M, Palm L M N D, Blankson-Arthur S, Tutu A O, Fosu P. 2011. Determination of organochlorine pesticide residue in sediment and water from the Densu river basin, Ghana. *Chemosphere.* 86: 286–292.
- Laird D, Fleming P, Wang B Q, Horton R, Karlen D. 2010. Biochar impact on nutrient leaching from a Midwestern agricultural soil. *Geoderma.* 158: 436–442.
- Lee Y, Park J, Ryu C, Gang K S, Yang W, Park Y K, Jung J, Hyun S. 2013. Comparison of biochar properties from

biomass residues produced by slow pyrolysis at 500 $^{\circ}\mathrm{C}.$ Bioresource Technol. **148**: 196–201.

- Lehmann J, da Silva J P Jr, Steiner C, Nehls T, Zech W, Glaser B. 2003. Nutrient availability and leaching in an archaeological Anthrosol and a Ferralsol of the Central Amazon basin: fertilizer, manure and charcoal amendments. *Plant Soil.* 249: 343–357.
- Lehmann J, Joseph S. 2009. Biochar for environmental management: an introduction. *In* Lehmann J, Joseph S (eds.) Biochar for Environmental Management: Science and Technology. Earthscan, London. pp. 1–12.
- Lehmann J, Rillig M C, Thies J, Masiello C A, Hockaday W C, Crowley D. 2011. Biochar effects on soil biota – a review. Soil Biol Biochem. 43: 1812–1836.
- Li M, Liu Q, Guo L J, Zhang Y P, Lou Z J, Wang Y, Qian G R. 2013. Cu(II) removal from aqueous solution by *Spartina alterniflora* derived biochar. *Bioresource Technol.* 141: 83–88.
- Liang B, Lehmann J, Solomon D, Kinyangi J, Grossman J, O'Neill B, Skjemstad J O, Thies J, Luizão F J, Petersen J, Neves E G. 2006. Black carbon increases cation exchange capacity in soils. Soil Sci Soc Am J. 70: 1719–1730.
- Lin Y, Munroe P, Joseph S, Henderson R, Ziolkowski A. 2012. Water extractable organic carbon in untreated and chemical treated biochars. *Chemosphere.* 87: 151–157.
- Liu W P, Gan J P, Papiernik S K, Yates S R. 2000. Sorption and catalytic hydrolysis of diethatyl-ethyl on homoionic clays. J Agric Food Chem. 48: 1935–1940.
- Liu X, Zhang Y, Li Z F, Feng R, Zhang Y Z. 2014. Characterization of corncob-derived biochar and pyrolysis kinetics in comparison with corn stalk and sawdust. *Bioresource Tech*nol. **170**: 76–82.
- Liu Y H, Liu Y, Chen Z S, Lian J, Huang X, Chung Y C. 2004. Purification and characterization of a novel organophosphorus pesticide hydrolase from *Penicillium lilacinum BP303*. *Enzyme Microb Technol.* **34**: 297–303.
- Liu Z G, Zhang F S, Wu J Z. 2010. Characterization and application of chars produced from pinewood pyrolysis and hydrothermal treatment. *Fuel.* 89: 510–514.
- Liu Z X, Chen X M, Jing Y, Li Q X, Zhang J B, Huang Q R. 2014. Effects of biochar amendment on rapeseed and sweet potato yields and water stable aggregate in upland red soil. *Catena.* **123**: 45–51.
- Loganathan V A, Feng Y C, Sheng G D, Clement T P. 2009. Crop-residue-derived char influences sorption, desorption and bioavailability of atrazine in soils. Soil Sci Soc Am J. 73: 967–974.
- Lu S G, Sun F F, Zong Y T. 2014. Effect of rice husk biochar and coal fly ash on some physical properties of expansive clayey soil (Vertisol). *Catena.* **114**: 37–44.
- Major J, Rondon M, Molina D, Riha S J, Lehmann J. 2010. Maize yield and nutrition during 4 years after biochar application to a Colombian savanna oxisol. *Plant Soil.* 333: 117–128.
- Mandelbaum R T, Wackete L P, Allan D L. 1993. Rapid hydrolysis of atrazine to hydroxyatrazine by soil bacteria. *Environ Sci Technol.* 27: 1943–1946.
- Martin S M, KooKana R S, Van Zwieten L, Krull E. 2012. Marked changes in herbicide sorption–desorption upon ageing of biochars in soil. J Hazard Mater. 231-232: 70–78.
- Mašek O, Brownsort P, Cross A, Sohi S. 2013. Influence of production conditions on the yield and environmental stability of biochar. *Fuel.* 103: 151–155.
- Méndez A, Gómez A, Paz-Ferreiro J, Gascó G. 2012. Effects of sewage sludge biochar on plant metal availability after ap-

plication to a Mediterranean soil. *Chemosphere*. **89**: 1354–1359.

- Méndez A, Terradillos M, Gascó G. 2013. Physicochemical and agronomic properties of biochar from sewage sludge pyrolysed at different temperatures. J Anal Appl Pyrol. 102: 124–130.
- Mizuta K, Matsumoto T, Hatate Y, Nishihara K, Nakanishi T. 2004. Removal of nitrate-nitrogen from drinking water using bamboo powder charcoal. *Bioresource Technol.* 95: 255–257.
- Mohan D, Sarswat A, Ok Y, Charles U, Pittman J. 2014. Organic and inorganic contaminants removal from water with biochar, a renewable, low cost and sustainable adsorbent – A critical review. *Bioresource Technol.* 160: 191–202.
- Morales M M, Comerford N, Guerrini I A, Falcão N P S, Reeves J B. 2013. Sorption and desorption of phosphate on biochar and biochar–soil mixtures. *Soil Use Manage.* 29: 306–314.
- Nelissen V, Saha B K, Ruysschaert G, Boeckx P. 2014. Effect of different biochar and fertilizer types on N₂O and NO emissions. Soil Biol Biochem. 70: 244–255.
- Novak J M, Busscher W J, Laird D L, Ahmedna M, Watts D W, Niandou M A S. 2009. Impact of biochar amendment on fertility of a southeastern coastal plain soil. *Soil Sci.* 174: 105–112.
- Pan D, Kong F B, Zhang N, Ying R Y. 2017. Knowledge training and the change of fertilizer use intensity: Evidence from wheat farmers in China. J Environ Manage. 197: 130–139.
- Peake L R, Reid B J, Tang X Y. 2014. Quantifying the influence of biochar on the physical and hydrological properties of dissimilar soils. *Geoderma.* 235-236: 182–190.
- Pietikäinen J, Kiikkilä O, Fritze H. 2000. Charcoal as a habitat for microbes and its effect on the microbial community of the underlying humus. *Oikos.* 89: 231–242.
- Pintor A M A, Ferreira C I A, Pereira J C, Correia P, Silva S P, Vilar V J P, Botelho C M S, Boaventura R A R. 2012. Use of cork powder and granules for the adsorption of pollutants: a review. Water Res. 46: 3152–3166.
- Ren X H, Zhang P, Zhao L J, Sun H W. 2016. Sorption and degradation of carbaryl in soils amended with biochars: influence of biochar type and content. *Environ Sci Pollut Res.* 23: 2724–2734.
- Rhoades J D. 1982. Cation exchange capacity. In Page A L, Miller R H, Keeney D R (eds.) Methods of Soil Analysis. Part 2: Chemical and Microbiological Properties. American Society of Agronomy, Madison. pp. 417–435.
- Rondon M A, Lehmann J, Ramírez J, Hurtado M. 2007. Biological nitrogen fixation by common beans (*Phaseolus vulgaris* L.) increases with bio-char additions. *Biol Fertil Soils.* 43: 699–708.
- Saito T, Otani T, Seike N, Murano H, Okazaki M. 2011. Suppressive effect of soil application of carbonaceous adsorbents on dieldrin uptake by cucumber fruits. *Soil Sci Plant Nutr.* 57: 157–166.
- Schmidt M W I, Torn M S, Abiven S, Dittmar T, Guggenberger G, Janssens I A, Kleber M, Kögel-Knabner I, Lehmann J, Manning D A C, Nannipieri P, Rasse D P, Weiner S, Trumbore S E. 2011. Persistence of soil organic matter as an ecosystem property. *Nature.* 478: 49–56.
- Schreinemachers P, Afari-Sefa V, Heng C H, Dung P T M, Praneetvatakul S, Srinivasan R. 2015. Safe and sustainable crop protection in Southeast Asia: status, challenges and policy options. *Environ Sci Policy.* 54: 357–366.
- Schwazenbach R P, Gschwend P M, Imboden D M. 2005. Chemical transformations. I: hydrolysis and reactions involving other nucleophilic species. *In* Schwarzenbach R P, Gschwend P M, Imboden D M (eds.) Environmental Organic Chemistry. John Wiley & Sons, Inc., New York. pp. 489–554.

- Singh B K, Kuhad R C. 2000. Degradation of insecticide lindane (γ-HCH) by white-rot fungi Cyathus bulleri and Phanerochaete sordida. Pest Manag Sci. 56: 142–146.
- Singh N S, Singh D K. 2011. Biodegradation of endosulfan and endosulfan sulfate by Achromobacter xylosoxidans strain C8B in broth medium. Biodegradation. 22: 845–857.
- Sohi S P, Krull E, Lopez-Capel E, Bol R. 2010. A review of biochar and its use and function in soil. Adv Agron. 105: 47–82.
- Song W P, Guo M X. 2012. Quality variations of poultry litter biochar generated at different pyrolysis temperatures. J Anal Appl Pyrol. 94: 138–145.
- Spokas K A, Baker J M, Reicosky D C. 2010. Ethylene: potential key for biochar amendment impacts. *Plant Soil.* 333: 443–452.
- Spokas K A, Koskinen W C, Baker J M, Reicosky D C. 2009. Impacts of woodchip biochar additions on greenhouse gas production and sorption/degradation of two herbicides in a Minnesota soil. *Chemosphere.* **77**: 574–581.
- Spokas K A, Novak J M, Venterea R T. 2012. Biochar's role as an alternative N-fertilizer: ammonia capture. *Plant Soil.* 350: 35–42.
- Steiner C, Das K C, Garcia M, Förster B, Zech W. 2008a. Charcoal and smoke extract stimulate the soil microbial community in a highly weathered xanthic Ferralsol. *Pedobiologia*. 51: 359–366.
- Steiner C, Glaser B, Teixeira W G, Lehmann J, Blum W E H, Zech W. 2008b. Nitrogen retention and plant uptake on a highly weathered central Amazonian Ferralsol amended with compost and charcoal. J Plant Nutr Soil Sci. 171: 893–899.
- Stewart C E, Zheng J Y, Botte J, Cotrufo M F. 2013. Cogenerated fast pyrolysis biochar mitigates green-house gas emissions and increases carbon sequestration in temperate soils. *Glob Change Biol Bioenerg.* 5: 153–164.
- Swiatkowski A, Pakula M, Biniak S, Walczyk M. 2004. Influence of the surface chemistry of modified activated carbon on its electrochemical behaviour in the presence of lead(II) ions. *Carbon.* 42: 3057–3069.
- Taghizadeh-Toosi A, Clough T J, Condron L M, Sherlock R R, Anderson C R, Craigie R A. 2011. Biochar incorporation into pasture soil suppresses in situ nitrous oxide emissions from ruminant urine patches. J Environ Qual. 40: 468–476.
- Taghizadeh-Toosi A, Clough T J, Sherlock R R, Condron L M. 2012. Biochar adsorbed ammonia is bioavailable. *Plant Soil.* 350: 57–69.
- Tan X F, Liu Y G, Gu Y L, Xu Y, Zeng G M, Hu X J, Liu S B, Wang X, Liu S M, Li J. 2016. Biochar-based nanocomposites for the decontamination of wastewater: a review. *Bioresource Technol.* 212: 318–333.
- Tan X F, Liu Y G, Zeng G M, Wang X, Hu X J, Gu Y L, Yang Z Z. 2015. Application of biochar for the removal of pollutants from aqueous solutions. *Chemosphere.* 125: 70–85.
- Thangarajan R, Bolan N, Mandal S, Kunhikrishnan A, Choppala G, Karunanithi R, Qi F J. 2015. Biochar for inorganic contaminant management in soil. *In* Ok Y S, Uchimiya S M, Chang S X, Bolan N (eds.) Biochar: Production, Characterization, and Applications. CRC Press, Boca Raton. pp. 99–138.
- Tsai W T, Liu S C, Chen H R, Chang Y M, Tsai Y L. 2012. Textural and chemical properties of swine-manure-derived biochar pertinent to its potential use as a soil amendment. *Chemosphere.* 89: 198–203.
- Uchimiya M, Orlov A, Ramakrishnan G, Sistani K. 2013. In situ and ex situ spectroscopic monitoring of biochar's surface functional groups. J Anal Appl Pyrol. 102: 53–59.

- Van Zwieten L, Singh B, Joseph S, Kimber S, Cowie A, Chan K Y. 2009. Biochar and emissions of non-CO₂ greenhouse gases from soil. *In* Lehmann J, Joseph S (eds.) Biochar for Environmental Management. Earthscan, London. pp. 227–249.
- Verma J P, Jaiswal D K, Sagar R. 2014. Pesticide relevance and their microbial degradation: a-state-of-art. *Rev Environ Sci Biotechnol.* 13: 429–466.
- Wang H Z, Zuo H G, Ding Y J, Miao S S, Jiang C, Yang H. 2014. Biotic and abiotic degradation of pesticide Dufulin in soils. *Environ Sci Pollut Res.* **21**: 4331–4342.
- Wang Y, Yin R Z, Liu R H. 2014. Characterization of biochar from fast pyrolysis and its effect on chemical properties of the tea garden soil. J Anal Appl Pyrol. 110: 375–381.
- Wang Z Y, Zheng H, Luo Y, Deng X, Herbert S, Xing B S. 2013. Characterization and influence of biochars on nitrous oxide emission from agricultural soil. *Environ Pollut.* 174: 289–296.
- Warnock D D, Lehmann J, Kuyper T W, Rillig M C. 2007. Mycorrhizal responses to biochar in soil—concepts and mechanisms. *Plant Soil.* **300**: 9–20.
- Willmann G, Fakoussa R M. 1997. Extracellular oxidative enzymes of coal-attacking fungi. Fuel Process Technol. 52: 27– 41.
- Wood T K. 2008. Molecular approaches in bioremediation. Curr Opin Biotechnol. 19: 572–578.
- Xu G, Sun J N, Shao H B, Chang S X. 2014. Biochar had effects on phosphorus sorption and desorption in three soils with differing acidity. *Ecol Eng.* 62: 54–60.
- Yanai Y, Toyota K, Okazaki M. 2007. Effects of charcoal addition on N₂O emissions from soil resulting from rewetting air-dried soil in short-term laboratory experiments. *Soil Sci Plant Nutr.* 53: 181–188.
- Yao Y, Gao B, Zhang M, Inyang M, Zimmerman A R. 2012. Effect of biochar amendment on sorption and leaching of nitrate, ammonium, and phosphate in a sandy soil. *Chemo-sphere.* 89: 1467–1471.
- Yu X Y, Ying G G, Kookana R S. 2009. Reduced plant uptake of pesticides with biochar additions to soil. *Chemosphere.* 76: 665–671.
- Yuan H R, Lu T, Wang Y Z, Chen Y, Lei T Z. 2016. Sewage sludge biochar: nutrient composition and its effect on the leaching of soil nutrients. *Geoderma.* 267: 17–23.
- Yuan J H, Xu R K, Zhang H. 2011. The forms of alkalis in the biochar produced from crop residues at different temperatures. *Bioresource Technol.* **102**: 3488–3497.
- Zhang A F, Cui L Q, Pan G X, Li L Q, Hussain Q, Zhang X H, Zheng J W, Crowley D. 2010. Effect of biochar amendment on yield and methane and nitrous oxide emissions from a rice paddy from Tai Lake Plain, China. Agric Ecosyst Environ. 139: 469–475.
- Zhang H, Voroney R P, Price G W. 2015. Effects of temperature and processing conditions on biochar chemical properties and their influence on soil C and N transformations. *Soil Biol Biochem.* 83: 19–28.
- Zhang J Q, Qi S H, Xing X L, Tan L Z, Gong X Y, Zhang Y, Zhang J P. 2011. Organochlorine pesticides (OCPs) in soils and sediments, southeast China: A case study in Xinghua Bay. Mar Pollut Bull. 62: 1270–1275.
- Zhang P, Sun H W, Yu L, Sun T H. 2013. Adsorption and catalytic hydrolysis of carbaryl and atrazine on pig manurederived biochars: impact of structural properties of biochars. J Hazard Mater. 244-245: 217–224.
- Zhang X, Zhang S H, Yang H P, Shi T, Chen Y Q, Chen H P. 2013. Influence of NH₃/CO₂ modification on the characteristic of biochar and the CO₂ capture. *BioEnergy Res.* 6:

1147 - 1153.

- Zhao X C, Ouyang W, Hao F H, Lin C Y, Wang F L, Han S, Geng X J. 2013. Properties comparison of biochars from corn straw with different pretreatment and sorption behaviour of atrazine. *Bioresource Technol.* 147: 338–344.
- Zheng H, Wang Z Y, Deng X, Herbert S, Xing B S. 2013a. Impacts of adding biochar on nitrogen retention and bioavailability in agricultural soil. *Geoderma.* 206: 32–39.
- Zheng H, Wang Z Y, Deng X, Zhao J, Luo Y, Novak J, Herbert S, Xing B S. 2013b. Characteristics and nutrient values of biochars produced from giant reed at different temperatures. *Bioresource Technol.* 130: 463–471.
- Zheng W, Guo M X, Chow T, Bennett D N, Rajagopalan N. 2010. Sorption properties of greenwaste biochar for two triazine pesticides. J Hazard Mater. 181: 121–126.