REVIEWS

Co-pyrolysis of sewage sludge and biomass for stabilizing heavy metals and reducing biochar toxicity: A review

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Abstract

The huge amounts of sewage sludge produced by municipal wastewater treatment plants induce major environmental and economical issues, calling for advanced disposal methods. Traditional methods for sewage sludge disposal increase greenhouse gas emissions and pollution. Moreover, biochar created from sewage sludge often cannot be used directly in soil applications due to elevated levels of heavy metals and other toxic compounds, which alter soil biota and earthworms. This has limited the application of sewage sludge-derived biochar as a fertilizer. Here, we review biomass and sewage sludge co-pyrolysis with a focus on the stabilization of heavy metals and toxicity reduction of the sludge-derived biochar. We observed that co-pyrolyzing sewage sludge with biomass materials reduced heavy metal concentrations and decreased the environmental risk of sludge-derived biochar by up to 93%. Biochar produced from sewage sludge and biomass co-pyrolysis could enhance the reproduction stimulation of soil biota by 20−98%. Heavy metals immobilization and transformation are controlled by the co-feed material mixing ratio, pyrolysis temperature, and pyrolysis atmosphere.

Keywords Sewage sludge · Co-pyrolysis · Synergistic effects · Heavy metals · Environmental risk assessment · Soil biota

Introduction

There is a global increase in the amount of sewage sludge produced from municipal wastewater treatment plants because of the rapid growth in global population (Mohamed and Li 2022). Existing conventional management methods for the disposal and reuse of sewage sludge include composting, landfilling, and soil application (Buonocore et al. 2018; Alvarado et al. 2020; Morin-Crini et al. 2022). The conventional methods, however, have several environmental and safety concerns and thus contribute remarkably to secondary contamination and

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increase greenhouse gas emissions (Buonocore et al. 2018; Liu et al. 2021b; Huang et al. 2022). Furthermore, sewage sludge derived from wastewater contains elevated amounts of contaminants such as heavy metals (Zhang et al. 2023), microplastics (Khan et al. 2022; Stang et al. 2022), and emerging contaminants (Mohamed et al. 2022a), which cannot be completely removed by conventional management methods (Crini and Lichtfouse 2019; Padervand et al. 2020; Crini et al. 2022; Morin-Crini et al. 2022). Thus, researchers have embarked on the investigation of alternative methods that can alleviate the limitations and disadvantages of conventional sewage sludge management methods. Pyrolysis has been proposed as a potentially efficient and feasible treatment and disposal alternative for municipal sewage sludge (Leng et al. 2020; Huang et al. 2022; Mohamed et al. 2022b). During the pyrolysis process, non-condensable gases, bio-oil and biochar, are generated (Leng et al. 2022), thus avoiding the creation of secondary toxic contaminants (Mohamed et al. 2020, 2021a; Abou Rjeily et al. 2021; Suresh et al. 2021; Zhao et al. 2022). The transformation of biomass waste into biochar has several agronomic and environmental advantages.

Several studies found that biochar application in soil would neutralize soil acidity, improve water and nutrient retention and soil cation exchange capacity, and immobilize heavy metals and other organic contaminants (Bolan et al. 2014; Mohamed et al. 2016, 2017, 2021b, 2022b). Sludge-derived biochar, however, should not be used directly in soil applications because of the high heavy metal content that may pose high environmental risks as a result of soil and groundwater contamination (Wang et al. 2019b, 2020). Therefore, alternate technologies that could reduce heavy metals in sludge-derived biochar should be developed and investigated. Co-pyrolysis is a novel environmental technology, which can generate syngas that is rich in hydrogen and reduce the concentrations of heavy metals in the produced biochar (Ahmed et al. 2020; Zhang et al. 2021; Su et al. 2022). Several studies have focused on the role of sewage sludge co-pyrolysis in reducing the environmental risk of the sludge-derived biochar, and the transformation and migration behavior of heavy metals during the process (Liu et al. 2017; Leng et al. 2018; Min et al. 2022). Nevertheless, no review articles discussing the advances of sewage sludge co-pyrolysis and the role of the process in reducing biochar toxicity have been published. We review the co-pyrolysis of sewage sludge with waste biomass to produce high-quality biochar with reduced contaminant levels and understand the mechanisms of different approaches for reducing toxicity of sludge-derived biochar. Overall, our review contributes toward improving our understanding of sewage sludge co-pyrolysis technologies and provide important information for selecting the appropriate co-feed material for co-pyrolysis process.

Factors affecting heavy metals stabilization in biochar

Heavy metals in sewage sludge can be partitioned during pyrolysis into syngas, bio-oil, and biochar, and heavy metal transformation is mostly determined by the pyrolysis working parameters and metal forms. Recent research has shown that when the temperature higher than 600 °C, for instance, cadmium (Cd), can be volatilized into syngas and bio-oil. However, following the pyrolysis of sewage sludge at a moderate temperature, over 90% of other heavy metals will be remained in the biochar (Jin et al. 2017; Yang et al. 2021; Wang et al. 2022). Therefore, pyrolysis conditions, including temperature and atmosphere, play an important role in determining the distribution of heavy metals. Overall, heavy metals distribution can be affected by the co-feed material mixing ratio, pyrolysis temperature, and pyrolysis atmosphere.

Effect of the biomass mixing ratio

Several studies reported that the mixing ratio of the co-feed material plays a key role in reducing heavy metal concentrations and the toxicity of the produced biochars compared with biochar produced from sewage sludge singular pyrolysis (Kończak et al. 2019; Wang et al. 2019b). An adequate ratio of willow with sewage sludge during co-pyrolysis is necessary to reduce heavy metals content since a small amount of biomass will minimize the 'dilution effect' for heavy metals in the produced biochars (Kończak and Oleszczuk 2020). Note that at low willow to sewage sludge ratios, the effect of the reduced bioavailable proportion of heavy metals was also substantial (Kończak and Oleszczuk 2020). Wang et al. (2019a, b) found that, in addition to the effect of reducing heavy metal contents, the amount of metal leaching from biochar was also influenced by biomass quantity (Wang et al. 2019b). Lowering heavy metals leaching rate was further enhanced as willow mixing ratios increase (Kończak and Oleszczuk 2020). Increasing the amount of biomass during sewage sludge co-pyrolysis results in improved metal transformations from mobile forms to more stable fractions, indicating possible decreases in possible environmental risks resulting from heavy metals elevated concentrations in sewage sludge biochars.

Biomass incorporation with sewage sludge has synergistic effects on heavy metal immobilization during sewage sludge co-pyrolysis with biomass (Table 1). Wang and co-workers found that pyrolyzing sewage sludge with food waste digestate reduced the ecological risk assessment of the heavy metals in biochars obtained from sewage sludge

Table 1 Environmental risk assessment of heavy metals in raw sewage sludge and derived biochars

References	Feedstock	Pyrolysis temperature (°C)	Heavy metals (mg/kg)					
			Zn	Cd	Ni	Cr	Cu	Pb
Leng et al. (2018)	Sewage sludge	300	0.45	18.38	12.81	1.36	1.13	5.72
	Rice straw	300	25.9	14.19	17.22	0.27	12.53	2.37
	Wood sawdust	300	13.78	13.79	14.36	0.63	15.02	13.16
	50% Rice straw	300	1.56	8.94	8.17	1.62	1.61	5.56
	50% Wood sawdust	300	1.07	5.0	7.95	1.89	1.03	6.4
Li et al. (2020)	Hydrothermally treated Sewage sludge	Raw feedstock	2.50/LR	13.04/MR	–	0.04/NR	2.66/LR	0.50/NR
	Pig manure	Raw feedstock	14.86/MR	3.37/LR	–	0.11/NR	1.56/LR	1.21/LR
	10% Pig manure	600	3.30/LR	7.70/LR	–	0.03/NR	2.39.LR	0.27/NR
	30% Pig manure	600	3.27/LR	3.17/LR	–	0.01/NR	0.71/LR	0.17/NR
	50% Pig manure	600	4.68/LR	4.26/LR	–	0.01/NR	0.04/LR	0.17/NR
Wang et al. (2021a)	Sewage sludge	Raw feedstock	50.92/VHR	33.19/HR	30.83/HR	4.77/LR	1.79/LR	0.39/NR
	Sewage sludge	350	23.53/MR	21.76/MR	10.49/MR	0.98/NR	0.46/NR	0.14/NR
		550	20.68/MR	11.93/MR	8.80/LR	0.52/NR	0.09/NR	0.20/NR
		750	2.92/LR	1.36/LR	3.83/LR	0.38/NR	0.08/NR	0.18/NR
	20% Bamboo sawdust	550	16.23/MR	3.46/LR	10.48/MR	0.30/NR	0.12/NR	0.15/NR
	20% Wood sawdust	550	16.17/MR	3.78/LR	7.85/LR	0.43/NR	0.12/NR	0.16/NR
	20% Rice husk	550	13.17/MR	3.95/LR	4.87/LR	0.24/NR	0.09/NR	0.10/NR
	20% Exhausted tea	550	14.09/MR	2.41/LR	9.42/LR	0.37/NR	0.13/NR	0.29/NR
	20% Kitchen waste	550	15.67/MR	0.96/LR	7.59/LR	0.32/NR	0.09/NR	0.07/NR
	20% Polyvinyl chloride	550	28.74/MR	16.15/MR	3.44/LR	0.76/NR	1.33/LR	0.37/NR
Li et al. (2021)	Hydrothermally treated sewage sludge	300	96.34	89.10	86.10	93.43	96.78	94.10
		500	93.80	70.85	83.46	84.37	94.01	83.18
		700	95.72	71.78	90.16	89.56	96.41	83.53
	10% Antibiotic mycelial residue	300	97.98	68.44	94.07	94.22	97.84	75.47
		500	96.96	82.12	85.75	92.2	98.37	97.44
		700	93.18	61.78	81.16	88.13	94.76	92.45
	25% Antibiotic mycelial residue	300	94.11	52.44	83.57	85.30	91.97	79.48
		500	97.18	49.81	96.14	95.97	96.89	77.83
		700	88.03	14.43	78.69	86.12	90.04	98.40
	50% Antibiotic mycelial residue	300	90.54	14.02	79.35	84.12	92.34	95.97
		500	99.48	14.59	85.51	87.81	99.78	90.04
		700	96.34	89.10	86.10	93.43	96.78	94.10

RAI: Risk assessment index; NR: No risk (RAI < 1%); LR: Low risk (1% ≤ RAI < 10%); MR: Medium risk (10% ≤ RAI < 30%); HR: High risk (30% ≤ RAI < 50%); VHR: Very high risk (RAI ≥ 50%)

co-pyrolysis compared with the biochar produced from single pyrolysis (Wang et al. 2022). The reduced ecological risk was due to the increase in the formation of more stable chemical fraction, including oxidizable fraction (F3), and residual fraction (F4), and the formations of residual fractions and oxidizable fractions increase with the increase in the mixing ratio of food waste digestate (Wang et al. 2022). Similarly, co-pyrolyzing sewage sludge with a hazelnut shell has been proposed as an effective and feasible method for reducing environmental toxicity and producing porous building materials (Zhao et al. 2017, 2018). Yang et al. (2020) found that the co-pyrolyzing rice husk and sewage sludge decreased the relative Pb and Cd contents in the produced biochar and decreased the environmental risk levels by 1–2 grades. Additionally, adding cotton stalk to sewage sludge was found to facilitate the transformation and migration of heavy metals from bioavailable to stable components, which reduces the possible toxic environmental impacts of heavy metals, achieving recycling substantially of sewage sludge (Wang et al. 2020). Various metals, Cu, Cr, Zn, Pb, Mn and Ni, are converted into relatively stable states following pyrolysis, resulting in substantial reductions in the immediate phytotoxicity and bioavailability of the biochar.

In contrast, high leaching rates of several heavy metals, including Zn, Cr, and Cu, were found in certain sewage sludge-derived biochars, which was attributable to mixing rice straw and sawdust during sewage sludge co-pyrolysis lowered the overall content of heavy metals in biochars (Huang et al. 2017). Despite reported lowered heavy metals content in the produced biochars, no clear decrease in the amounts of leachable leaching heavy metals was observed (Huang et al. 2017). In brief, sewage sludge and cotton stalk co-pyrolysis are found to be a viable approach for mitigating the possible environmental impacts of heavy metals in sewage sludge biochar-treated soils (Wang et al. 2020). In contrast to non-biomass, mixing sewage sludge and metal-loaded polyvinyl chloride microplastic was shown to have a good influence on heavy metal immobilization during the co-pyrolysis process, and the stable fraction of heavy metals increased with increasing polyvinyl chloride mixing ratio (Li et al. 2022b). Overall, sewage sludge and biomass co-pyrolysis are considered a practical treatment solution for disposing metal polluted waste sludge safely by producing biochar with reduced toxicity. The use of nontoxic catalyst can further enhance heavy metals immobilization and transformation into stable forms.

Effect of pyrolysis temperature

Several researchers studied the effects of different pyrolysis temperatures on heavy metals stabilization in biochars produced from sewage sludge co-pyrolysis. Generally, increasing pyrolysis temperature reduces the amount of extractable heavy metals due to partial fractionation and volatility of these metals at high temperatures, leading to reduced biochar toxicity (Table 1) (Gondek et al. 2014; Kończak and Oleszczuk 2020). Pyrolyzing sewage sludge at 400 °C is widely known to stabilize metals. According to the findings of He et al. (2010), pyrolysis temperatures higher than 350 °C result in better stability of trace heavy metals such as Cd, Pb, Zn, and Cu (He et al. 2010). The environmental damage and toxicity of metals in the co-pyrolyzed biochar decreased further, and co-pyrolysis at 600 °C was found to be suitable for sewage sludge nutrient recycling (Jin et al. 2017; Yang et al. 2021). High pyrolysis temperatures promote the creation of more permanent aromatic bonds and the elimination of transitory aliphatic bonds (Gondek et al. 2014).

Similarly, pyrolysis can stabilize heavy metals by transforming active extractable and exchangeable (F1) and reducible fractions (F2) into stable fractions including oxidizable (F3): metals bound to sulfides and organic matter, and residual fractions (F4): non-silicate bound metals, at low temperatures (300 °C) (Wang et al. 2020). Wang and co-workers evaluated multiple heavy metals in sediments, water, and soils, using the environmental risk index (Table 1). High

pyrolysis temperatures were found to reduce the potential environmental concern of heavy metals, particularly copper (Wang et al. 2020).

Gondek and co-workers investigated the effects of low temperature, 200 °C, on the ecotoxicity of heavy metals and metals mobile forms in biochars produced by co-pyrolyzing sewage sludge with various biomass materials such as bark, sawdust, rape straw, and wheat straw. The concentrations of heavy metals in water-soluble form were substantially reduced in all of the tested co-feed materials as a result of thermal transformation (Gondek et al. 2014). The study also found low toxicity of heavy metal extracts from biomass feedstocks for *Lepidium sativum* and *Vibrio fischeri*, irrespective of the transformation mechanism of heavy metals. As a result, *L. sativum* was more sensitive to heavy metals in the analyzed extraction from biomass feedstocks than *Vibrio fischeri*, as confirmed by the strong correlations between metal concentration and *L. sativum* root development inhibition (Gondek et al. 2014).

Kończak and Oleszczuk (2020) studied the effects of different temperatures (500−700 °C) on the toxicity of biochars produced from sewage sludge and willow co-pyrolysis, and the increased temperature, particularly to 700 °C, reduced the quantity of extractable metals for Al, Cu, Fe, Mn, and Zn. Similarly, Huang and co-workers studied the effects of different temperatures, 300 − 700 °C, on the leachable and total heavy metals from biochars and the physicochemical properties of biochars from sewage sludge co-pyrolysis with rice straw or sawdust. However, the increased pyrolysis temperature was found to concentrate heavy metals in the biochars, as shown in Fig. 1. The increased heavy metals in biochars might be related to heavy metals' higher stability than other sewage sludge compositions (Huang et al. 2017). Further, when temperature increases, metals in the biochar become more concentrated, due to the increased mass loss rate during co-pyrolysis (Huang et al. 2017; Kończak and Oleszczuk 2020). Huang and co-workers also found that pyrolysis temperature has limited effects on heavy metals leachability, while Cd and Pb leachable levels in biochars were above the threshold limits. Several biochars are also have high levels of Zn (Huang et al. 2017).

Another important factor affecting biochar quality is the sample residence time; long residence periods during co-pyrolysis of sewage sludge usually have a positive impact on heavy metal immobilization, resulting in a sufficient reduction of the possible damage to the environment (Wang et al. 2019a). In the meantime, co-pyrolysis will change the bioavailability of heavy metals into stable compounds, leading to a substantial reduction in direct toxicity of biochar (Wang et al. 2019a; Yang et al. 2021). However, very long residence times will reduce the surface area because of the pore structure destruction (Wang et al. 2019a). Overall, pyrolysis temperatures and residence time can have significant effects
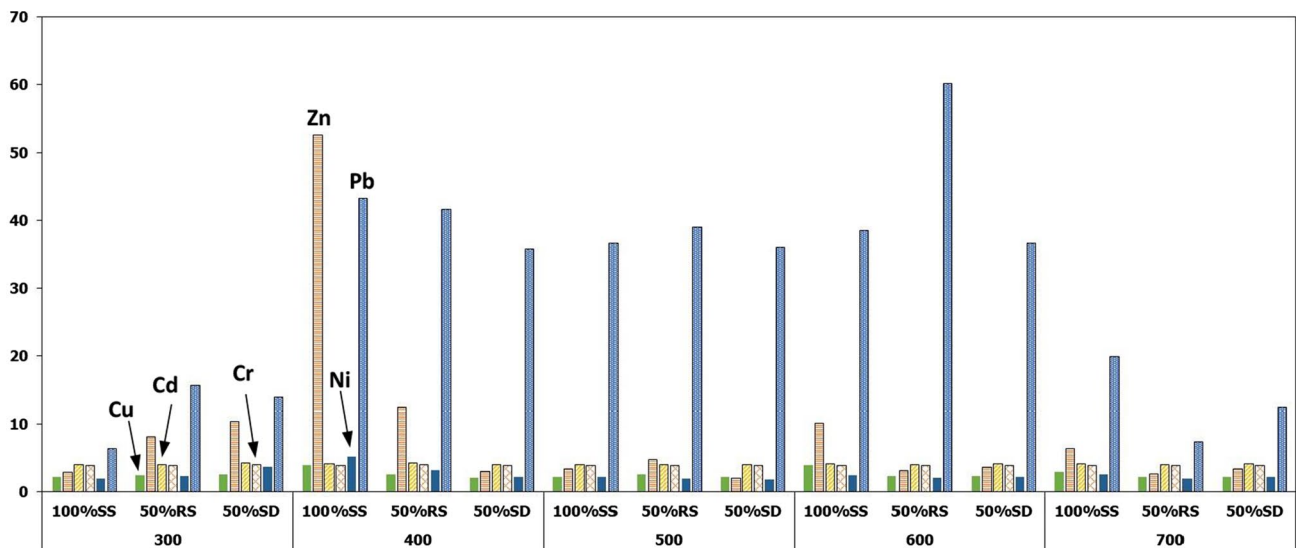
Heavy metals (mg/kg)

Fig. 1 Effects of pyrolysis temperatures, from 300 to 700 ºC, on the total content of heavy metals in biochar produced from sewage sludge (SS) single and co-pyrolysis with rice straw (RS) or sawdust (SD) (Data source: (Huang et al. 2017)). Rice straw (RS) or sawdust (SD) was mixed with sewage sludge (SS) at 50% mass ratio. Among different metals, lead (Pb) is the most affected metal with changing pyrolysis temperatures. When the pyrolysis temperature is increased, heavy metals are concentrated more in biochar because of organic devolatilization, leading to reduced biochar yield and increased ash content in biochar

on the stabilization of heavy metals, which should be further investigated.

Kończak et al. (2019) studied the effects of sewage sludge and willow co-pyrolysis at different mixing ratios of 8:2 and 6:4, wt:wt, different inert gases of N_2 and CO_2, and different temperatures of 500−700 °C, on the physical and chemical biochar properties. Notably, replacing N_2 with CO_2 was found to affect the physicochemical properties of the produced biochars, as shown in Table 2. The biochars exhibited more favorable surface properties, which were reflected by the increased oxygen content by up to 300% increase, and specific surface area by up to 47%. Carbonization and aromatization degree was high, and the salinity and pH were lower than the biochars created with N_2 environment (Kończak et al. 2019). Sewage sludge and willow co-pyrolysis using CO_2 as a carrier gas were also found to affect the total concentration of heavy metals and reduce the leachable metals in biochars (Kończak and Oleszczuk 2020). In another study, Kończak and co-workers found that using CO_2 as a carrier gas not only reduced the ecotoxicity effects of the produced biochar from sewage sludge and willow co-pyrolysis on bacteria: *Vibrio fischeri*, arthropods: *Folsomia candida*, and plants: *Lepidium sativum*, but also stimulated the growth of the tested plant (Fig. 2) and decreased luminescence inhibition of the bacteria by 75−95% (Kończak et al. 2020).

The findings showed that the produced biochars from the sewage sludge and willow in a CO_2 environment demonstrate the characteristics most needed for materials which could be applied in agriculture and in the treatment of contaminated soil and water. However, limited studies have studied the effects of using different atmospheres on stabilizing heavy metals for the biochars derived from sewage sludge and biomass co-pyrolysis. Overall, more future studies might study the effect of using different atmospheres, CO_2, N_2, and different mixtures, with different biomass materials for sewage sludge co-pyrolysis. The effect of gas atmospheres might be dependent on co-feed selection, which should be further explored.

Table 3 summarizes the effects of mixing different biomass materials with sewage sludge on immobilizing heavy metals and reducing the ecological risks of the sludge-derived biochars in addition to the impacts on biochar phytochemical properties. As shown in Table 3, most of the biomass materials immobilized heavy metals and reduced the potential ecological risks. Some materials enhanced the physical and chemical properties of the derived biochar, which could also enhance biochar sorption efficiency. Among various feedstocks, cotton stalk, hazelnut shell, and coconut shell have more positive effects on reducing the toxicity of heavy metals and enhancing biochar characteristics. Thus, future studies could further explore the optimization of the co-pyrolysis process using different carrier gases in addition to the use of proper nontoxic catalysts.ww

Table 2 Heavy metal contents in raw materials and in leachables from biochars produced from co-pyrolysis of sewage sludge and other feedstock materials

Feedstock	Pyrolysis temperature (°C)	Heavy metals (mg/kg)												References
		Cu	Zn	Cd	Cr	Ni	Pb	Co	Mn	Fe	Ba	B	Al	
Sewage sludge	Raw feedstock	301.1	340.3	21.4	104.6	143	451.3	–	–	–	–	–	–	Huang et al. (2017)
Rice Straw	Raw feedstock	9.5	9.0	<2.0	62.9	3.1	3.5	–	–	–	–	–	–	
Sawdust	Raw feedstock	3.1	7.2	<2.0	12.1	2.7	<2.0	–	–	–	–	–	–	
Sewage sludge	300	2.2	2.9	4.0	3.8	1.9	6.3	–	–	–	–	–	–	
	400	3.9	52.7	4.1	3.9	5.2	43.2	–	–	–	–	–	–	
	500	2.1	3.3	4.0	3.8	2.1	36.6	–	–	–	–	–	–	
	600	3.9	10.1	4.1	3.8	2.4	38.5	–	–	–	–	–	–	
	700	2.9	6.4	4.1	3.8	2.6	19.9	–	–	–	–	–	–	
50% Rice Straw	300	2.4	8.1	4.0	3.8	2.3	15.6	–	–	–	–	–	–	
	400	2.5	12.3	4.2	4.0	3.2	41.6	–	–	–	–	–	–	
	500	2.5	4.7	4.0	3.8	1.9	39.0	–	–	–	–	–	–	
	600	2.3	3.1	4.0	3.8	2.0	60.1	–	–	–	–	–	–	
	700	2.1	2.6	4.0	3.8	1.9	7.3	–	–	–	–	–	–	
50% Sawdust	300	2.5	10.3	4.2	4.0	3.7	13.8	–	–	–	–	–	–	
	400	2.0	3.0	4.0	3.8	2.2	35.7	–	–	–	–	–	–	
	500	2.1	2.0	4.0	3.8	1.8	36.0	–	–	–	–	–	–	
	600	2.3	3.6	4.1	3.9	2.2	36.6	–	–	–	–	–	–	
	700	2.2	3.3	4.1	3.8	2.2	12.3	–	–	–	–	–	–	
Sewage sludge	600	8.46	83.76	0.4	3.53	1.15	18.15	–	–	–	–	–	–	Dong et al. (2020)
30% Rice Straw	600	5.63	46.23	0.26	1.99	0.79	13.38	–	–	–	–	–	–	
50% Rice Straw	600	3.94	24.16	0.19	1.57	0.5	7.73	–	–	–	–	–	–	
70% Rice Straw	600	2.52	11.87	0.1	0.92	0.31	4.86	–	–	–	–	–	–	

Table 2 (continued)

Feedstock	Pyrolysis temperature (°C)	Heavy metals (mg/kg)												References
		Cu	Zn	Cd	Cr	Ni	Pb	Co	Mn	Fe	Ba	B	Al	
Sewage sludge	Raw feedstock	0.73	10.97	0.26	0.12	2.39	0.31	0.48	8.47	22.3	1.02	22.3	1196	Kończak and Oleszczuk (2020)
Willow	Raw feedstock	0.27	5.24	0.01	0.04	0.23	0.1	0.01	1.96	3.35	0.41	0.41	134	
20% Willow	Raw feedstock	0.53	8.61	0.19	0.09	1.79	0.27	0.32	7.07	12.7	0.73	15.8	822	
40% Willow	Raw feedstock	0.42	5.86	0.09	0.05	1.23	0.15	0.19	2.17	4.91	0.54	8.25	138	
Biochar (sewage sludge)	500	0.3	6.36	0.05	0.01	0.03	0.01	0.03	0.32	0.23	0.37	19.8	765	
Biochar (sewage sludge)	600	0.26	5.14	0.06	0.01	0.04	0.02	0.01	0.24	0.25	0.98	5.63	482	
Biochar (sewage sludge)	700	0.28	4.97	0.07	0.01	0.03	0.01	0.02	0.15	0.21	0.87	3.75	106	
20% Willow	500	0.21	3.75	0.02	0.01	0.02	0.02	0.04	0.25	0.14	0.38	18.1	810	
20% Willow	600	0.24	2.47	0.01	0.01	0.03	0.02	0.02	0.13	0.05	0.71	5.54	448	
20% Willow	700	0.05	2.31	0.01	0.01	0.01	0.02	0.01	0.03	0.02	0.51	3.69	115	
40% Willow	500	0.11	3.12	<DL	0.02	0.02	<DL	0.02	0.21	0.11	0.3	12.5	777	
40% Willow	600	0.24	1.03	<DL	0.01	0.03	<DL	0.01	0.12	0.02	0.44	4.75	462	
40% Willow	700	0.24	0.3	<DL	0.01	0.01	<DL	0.01	0.01	0.01	0.75	2.12	102	
Biochar (sewage sludge)/CO_2	500	0.22	4.61	<DL	0.01	0.06	<DL	<DL	0.07	0.18	0.42	7.83	2183	
Biochar (sewage sludge)/CO_2	600	0.12	3.21	<DL	0.01	0.04	<DL	<DL	0.05	0.21	0.42	6.25	2295	
Biochar (sewage sludge)/CO_2	700	0.01	0.95	<DL	0.02	0.02	<DL	<DL	0.02	0.03	0.73	4.59	1183	
40% Willow/CO_2	500	0.1	2.01	<DL	0.01	0.01	<DL	<DL	0.08	0.01	0.17	5.6	4274	
40% Willow/CO_2	600	0.02	1.23	<DL	0.01	0.01	<DL	<DL	0.02	0.01	0.28	4.77	2464	
40% Willow/CO_2	700	0.03	0.59	<DL	0.01	0.01	<DL	<DL	0.01	0.01	0.22	4.91	1189	
Sewage sludge	Raw feedstock	571.05	2650	–	–	–	–	–	–	–	–	–	–	Chen et al. (2020)
Excavated waste	Raw feedstock	374.97	3260	–	–	–	–	–	–	–	–	–	–	
Biochar (sewage sludge)	600	950.38	3823	–	673.31	176.09	45.68	–	–	–	–	–	–	
25% Excavated waste	600	921.82	38,833	–	632.47	145.43	55.84	–	–	–	–	–	–	
50% Excavated waste (hydrothermal pretreatment)	600	1069	3102	–	525.86	148.87	84.73	–	–	–	–	–	–	

Table 2 (continued)

Feedstock	Pyrolysis temperature (°C)	Heavy metals (mg/kg)												References
		Cu	Zn	Cd	Cr	Ni	Pb	Co	Mn	Fe	Ba	B	Al	
Hydrothermally treated sewage sludge	Raw feedstock	5736	6192	0.26	5440	–	155.38	–	–	–	–	–	–	Li et al. (2020)
Pig manure	Raw feedstock	5737	6192	0.29	112.79	–	13.24	–	–	–	–	–	–	
10% Pig manure	600	1390	4656	0.34	5104	–	144.01	–	–	–	–	–	–	
30% Pig manure	600	5401	5911	0.3	4390	–	129.55	–	–	–	–	–	–	
50% Pig manure	600	4981	5709	0.3	3630	–	106.2	–	–	–	–	–	–	
Hydrothermally treated sewage sludge	300	2424	1781	3.18	3976	320.67	122.95	–	–	–	–	–	–	Li et al. (2021)
	500	2318	1713	2.53	3543	306.98	111.57	–	–	–	–	–	–	
	700	2195	1650	2.33	3329	293.97	106.76	–	–	–	–	–	–	
10% Antibiotic mycelial residue	300	2056	1574	2.06	3015	265.36	100.22	–	–	–	–	–	–	
	500	2572	1871	3.06	4096	333.43	132.91	–	–	–	–	–	–	
	700	2447	1782	2.31	3875	312.61	129.86	–	–	–	–	–	–	
25% Antibiotic mycelial residue	300	2207	1683	1.88	3501	300.92	112.19	–	–	–	–	–	–	
	500	2125	1591	1.53	3134	276.75	105.46	–	–	–	–	–	–	
	700	2627	1895	0.6	4269	341.39	149.76	–	–	–	–	–	–	
50% Antibiotic mycelial residue	300	2593	1882	0.57	4022	332.33	146.57	–	–	–	–	–	–	
	500	2517	1871	0.55	3790	323.78	133.63	–	–	–	–	–	–	
	700	1164	1707	0.55	3239	291.35	111.21	–	–	–	–	–	–	
Sewage sludge	350	134.67	2060	1.53	110.59	37.38	62.39	–	–	–	–	–	–	Wang et al. (2021a)
	550	169.93	2603	2.01	142.9	49.31	78.54	–	–	–	–	–	–	
	750	178.91	2763	0.45	150.93	50.18	81.65	–	–	–	–	–	–	
20% Bamboo sawdust	550	150.9	2365	1.54	124.37	38.98	67.61	–	–	–	–	–	–	
20% Wood sawdust	550	149.38	2308	1.71	130.95	42.91	70.82	–	–	–	–	–	–	
20% Rice husk	550	155.06	2355	1.72	201.16	74.27	68.82	–	–	–	–	–	–	
20% Exhausted tea	550	153.41	2352	1.68	131.51	41.5	71.6	–	–	–	–	–	–	
20% Kitchen waste	550	153.07	2308	1.79	141.35	42.76	70.67	–	–	–	–	–	–	
20% Polyvinyl chloride	550	153.93	2264	1.64	132.45	40.65	69.38	–	–	–	–	–	–	

<DL, below detection limit

Fig. 2 Effect of N_2 or CO_2 on the ecotoxicity of biochar produced from single pyrolysis of sewage sludge (BCSSL) and co-pyrolysis of sewage sludge with willow (BCSSLW) at 500−700 °C on *Lepidium sativum* **a** and *Vibrio fischeri* **b**. Note that the minus values for Fig. 2a refer to the growth stimulation, meaning no inhibition effects. Using CO_2 as a carrier gas can produce biochar with lower toxicity, which could further enhance crop production. Redrawn from reference (Kończak et al. 2020) with permission from Elsevier

a Root growth inhibition (%)
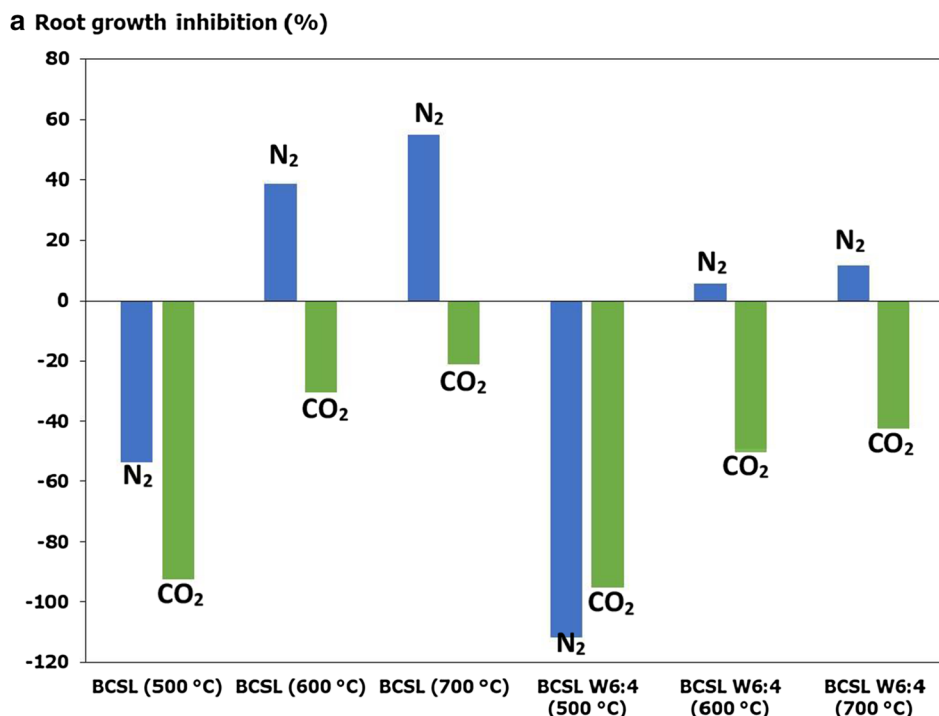
b Inhabitation of *V. fischeri* luminescence (%)
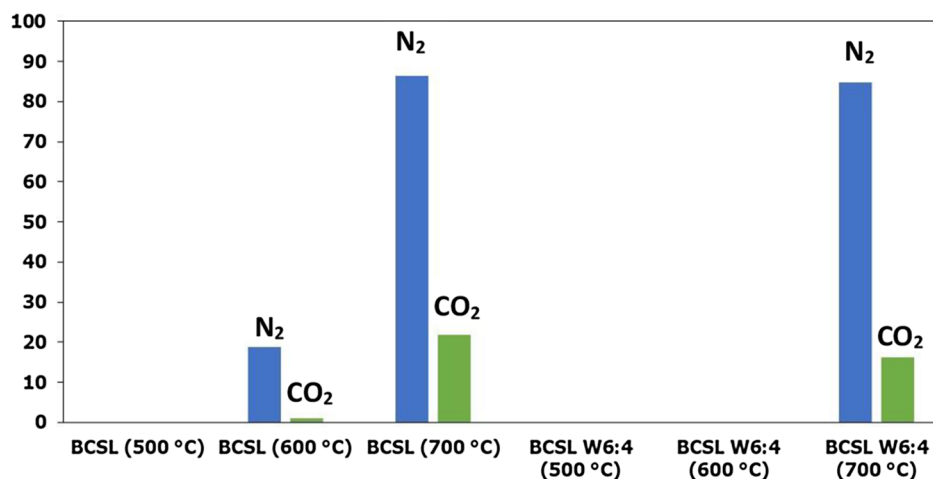
Effect of co-pyrolysis on biochar yield and physiochemical properties

Table 4 shows the incorporation of several biomass materials into sewage sludge during the co-pyrolysis process markedly affects the element contents: C, H, N, S, and O, ash content, fixed carbon, and volatile organic matter in comparison with sewage sludge single pyrolysis. The incorporation of wheat straw with sewage sludge raises the carbon content of biochar and improves the carbon porous structures of the sewage sludge biochar, increasing biochar reactivity markedly (Deng et al. 2017). Elements include

N, C, H, S, and O are devolatilized at different temperatures, whereas inorganic metals are not volatilized at temperatures above 500 °C, which is the primary explanation for the biochar's increased ash content. For wood-based feedstocks, carbon starts to volatilize at ~ 100 °C, nitrogen above 200 °C, S above 375 °C, K and P between 700 °C and 800 °C, and Mg, Ca, and Mn volatilize over 1000 °C during pyrolysis (Lehmann and Joseph 2009).

The biochar yield from co-pyrolysis of microalgae and sewage sludge ranged from 23.6% to 57.6% at 500 °C, and the biochar yield is highly correlated to H/C in biomass materials by the linear equation of $y = -1147.1x + 198.6$.

Table 3 Synergetic effect of co-pyrolyzing sewage sludge with other biomass materials

Sewage sludge co-pyrolysis with different biomasses	Synergetic effects	References
Cotton stalk	Reduced potential environment risk of heavy metals in biochar	Wang et al. (2019b, 2020)
	Optimized biochar pore structures	
	Increased biochar yield	
	Increased biochar aromaticity and reduced the functional groups	
Hazelnut shell (catalytic co-pyrolysis with ZnCl$_2$)	Reduced potential environment risk of heavy metals in biochar	Zhao et al. (2017, 2018)
	Reduced potential ecological risk index	
	Optimized biochar pore structures	
	Increased biochar specific surface area	
Willow	Reduced potential ecological risk index	Kończak and Oleszczuk (2020), Kończak et al. (2019)
	Optimized biochar pore structures	
	Increased biochar yield	
	Increased biochar aromaticity and reduce the functional groups	
Sawdust	Reduced potential environment risk of heavy metals in biochar	Huang et al. (2017), Leng et al. (2018)
	Reduced potential ecological risk index	
	Increased biochar specific surface area	
Rice straw	Reduced potential environment risk of heavy metals in biochar	Dong et al. (2020)
	Optimized biochar pore structures	
Pig manure	Reduced potential environment risk of heavy metals in biochar	Li et al. (2020)
	Reduced potential ecological risk index	
Rice husk	Reduced potential environment risk of heavy metals in biochar	Wang et al. (2021a)
	Reduced potential ecological risk index	
	Increased biochar yield	

Fitting accuracy (R^2) is found to be as high as 0.998, which is in line with pine sawdust and sewage sludge co-pyrolysis. The same liner relationship has also been found for lignite and energy grass co-pyrolysis (Guan et al. 2015; Wang et al. 2016). For future research studies, the linear relationship may be utilized to forecast the pyrolysis char generation for microalgae and sewage sludge co-pyrolysis.

Additionally, mixing sewage sludge with cotton stalk increased the pH, carbon content, and adsorption efficiency, while the biochar yields, specific surface areas, ash contents, and molar H/C ratios were decreased in compression with the biochars obtained from sewage sludge single pyrolysis (Wang et al. 2020). Chen et al. (2019) found that the sewage sludge inherent moisture will have a gasification effect on the biochar structure produced from wood waste and sewage sludge co-pyrolysis, which in turn will increase biochar-specific surface area. The intrinsic moisture was beneficial for forming the biochar surface functional groups, mainly carbonyl, which was confirmed by Raman spectrum and X-ray diffraction (XRD) tests (Chen et al. 2019). The increased wood waste content contributed to the increased specific surface area of biochars from 68.49 to 119.5 cm^2/g, which is because of the high specific surface area of wood waste biochar of 150.6 cm^2/g. However, biochars from sewage sludge and wood waste co-pyrolysis have average pore sizes being less than 10 nm, suggesting that biochar produced from sewage sludge blocked the pores of wood waste-biochar (Chen et al. 2019). Kończak and co-workers found that mixing willow with sewage sludge increased the specific surface area of the produced biochars (Kończak et al. 2019). Similarly, adding rice straw to sewage sludge enhanced the formation of micropore structure with an average pore size values of 28.7 nm for sewage sludge-biochar and 2.51 nm for rice straw-biochar (Dong et al. 2020).

Biochar porosity is one of the important characteristics that affect biochar sorption affinity, where the adsorption happens mainly in micropores, whereas meso- and macropores play a crucial role in the adsorption mechanism, in which both serve as channels to micropores (Pastor-Villegas et al. 2006; Zhang and You 2013). Moreover, since biomass feedstocks are generally pyrolyzed in the biomass particles pores, the synergistic reactions are responsible for major changes in pore structures in biochars. Consequently, the pore structure changes influence the reactions level. Overall, biochar pore diameter and specific surface area may play a key role in the immobilization of heavy metal, which need to be investigated further.

Table 4 Ultimate and proximate analysis, and specific surface area and volume for biochars produced from co-pyrolysis of sewage sludge and other feedstock materials

References	Feedstock	Yield (%)	Ultimate analysis (wt.%)					Proximate analysis (wt.%)			pH	Specific surface area and volume				High heating value (MJ/kg)
			C	H	N	O	S	Ash	Fixed carbon	Volatile matter		Surface area (m²/g)	Average pore size (nm)	Micropore volume (cm³/g)	Total volume (cm³/g)	
Huang et al. (2017)	Sewage sludge	–	26.0	3.9	4.5	–	–	49.1	3.8	47.1	5.7	7.1	20.0	0.0029	–	–
	Rice straw	–	37.3	1.2	5.4	–	–	14.5	1.3	84.2	–	–	–	–	–	–
	50% Rice Straw	54	29.6	1.5	2.7	–	0	55.7	2.6	41.7	7	6.6	17.8	0.0026	0.0566	–
	50% Sawdust	53.2	45.6	2.5	2.6	–	0	35.5	4.4	60.1	7.4	3.9	14.1	0.0013	0.0333	–
Alvarez et al. (2015)	Sewage sludge	–	18.5	1.0	2.7	7.4	1.2	69.3	11.7	19.0	–	42.9	–	–	–	–
	Pinewood	–	82.7	2.9	0.1	11.4	–	2.9	73.6	23.5	–	16.1	–	–	–	–
	50% Pinewood	33	37	1.4	1.5	8.0	0.90	50.7	29.6	19.7	–	30.7	–	–	–	12.6
Jin et al. (2017)	50% Bamboo sawdust	44.5	30.4	1.63	1.76	–	0.56	58.3	–	–	10.1	4.5	–	–	–	–
Wang et al. (2019b)	10% Cotton stalks	57.3	22.0	1.57	2.02	–	–	68.6	8.83	22.6	9.9	23.8	–	–	0.0329	–
	50% Cotton stalks	44.7	37.7	1.9	1.59	–	–	50.4	11.7	37.8	9.6	3.12	–	–	0.058	–
	90% Cotton stalks	39.2	54.9	2.2	1.12	–	–	31.0	15.9	53.1	9	33.6	–	–	0.0709	–
Wang et al. (2016)	50% Microalgae	34.3	–	–	–	–	–	–	–	–	–	–	–	–	–	8.251
Chen et al. (2019)	50% wooden waste	46.1	38.2	0.06	–	0.17	0.01	61.6	37.9	0.53	–	–	9.66	–	–	–
Deng et al. (2017)	20% Wheat straw	22.8	–	–	–	–	–	–	–	–	–	75.3	15.5	–	0.13	–
	60% Wheat straw	14.3	–	–	–	–	–	–	–	–	–	224.1	6.9	–	0.26	–
Dong et al. (2020)	30% Rice straw	52.9	17.8	1.39	1.85	6.39	0.26	72.3	19.6	8.11	9.3	27.7	13.01	0.0028	0.0901	5.66
	50% Rice straw	46.6	25.6	1.5	1.7	6.27	0.17	64.8	26.5	8.79	9.6	58.5	5.16	0.0032	0.0754	8.68
	70% Rice straw	40.2	37.2	1.74	1.53	6.13	0.10	53.4	38.3	8.37	10	65.3	4.56	0.0176	0.0744	13.25
Ruiz–Gómez et al. (2017)	50% Manure	40.0	30.0	1.18	2.38	–	0.90	62.2	18.2	19.5	–	–	–	–	–	11.0
Wang et al. (2020)	50%Cotton stalks	45.0	33.6	1.83	2.32	–	–	65.0	–	–	10.5	10.8	14.3	–	0.039	–

Mechanism of heavy metals immobilization in biochar

Kończak and Oleszczuk studied the mechanism of stabilizing biochars heavy metals created from sewage sludge co-pyrolyzed with willow under different atmospheres of N_2 and CO_2. Note that several mechanisms could affect heavy metals stabilization, including the occlusion in minerals, the entrapment of heavy metals into biochar pores, and the formation of π-bonding with electron-rich domains on biochar aromatic groups (Kończak and Oleszczuk 2020).

Yang and co-workers conducted a Pearson correlation analysis to elucidate the potential mechanism of heavy metals transformation to evaluate the inherent association between the characteristics of biochar and the leachable heavy metal fractions. The pH levels, N concentrations, and chemical forms of Pb all had significant (p = 0.05) effects on the percentages of leachable Pb during pyrolysis. Whereas

the quantities of leachable Cd correlated significantly with the contents of H and N, the N/C and H/C molar ratios, Cd chemical forms and the pH values at $p = 0.01$ (Yang et al. 2020). Heavy metal availability changes were mostly influenced by Cd and Pb chemical forms in the biochars, alkalinity of biochar, and aromatic structure formation (Yang et al. 2020).

For the catalytic effects, during $CaSO_4$ and sewage sludge co-pyrolysis, various mechanisms of speciation change and immobilization of Cu, Cr, Ni, Zn, and Pb were hypothesized (Fig. 3). In general, crystals with chemical stability and high strength are generated because of the solid and liquid phase interactions during $CaSO_4$ and sewage sludge co-pyrolysis (Fig. 3), resulting in heavy metal immobilization (Liu et al. 2021a). In addition, the sewage sludge organics are decomposed to produce carbon, gases and the formation of new organics, as well as numerous pores to aid in heavy metal immobilization (Sun et al. 2018). Simultaneously, inorganics in sewage sludge including silicates, phosphates, and
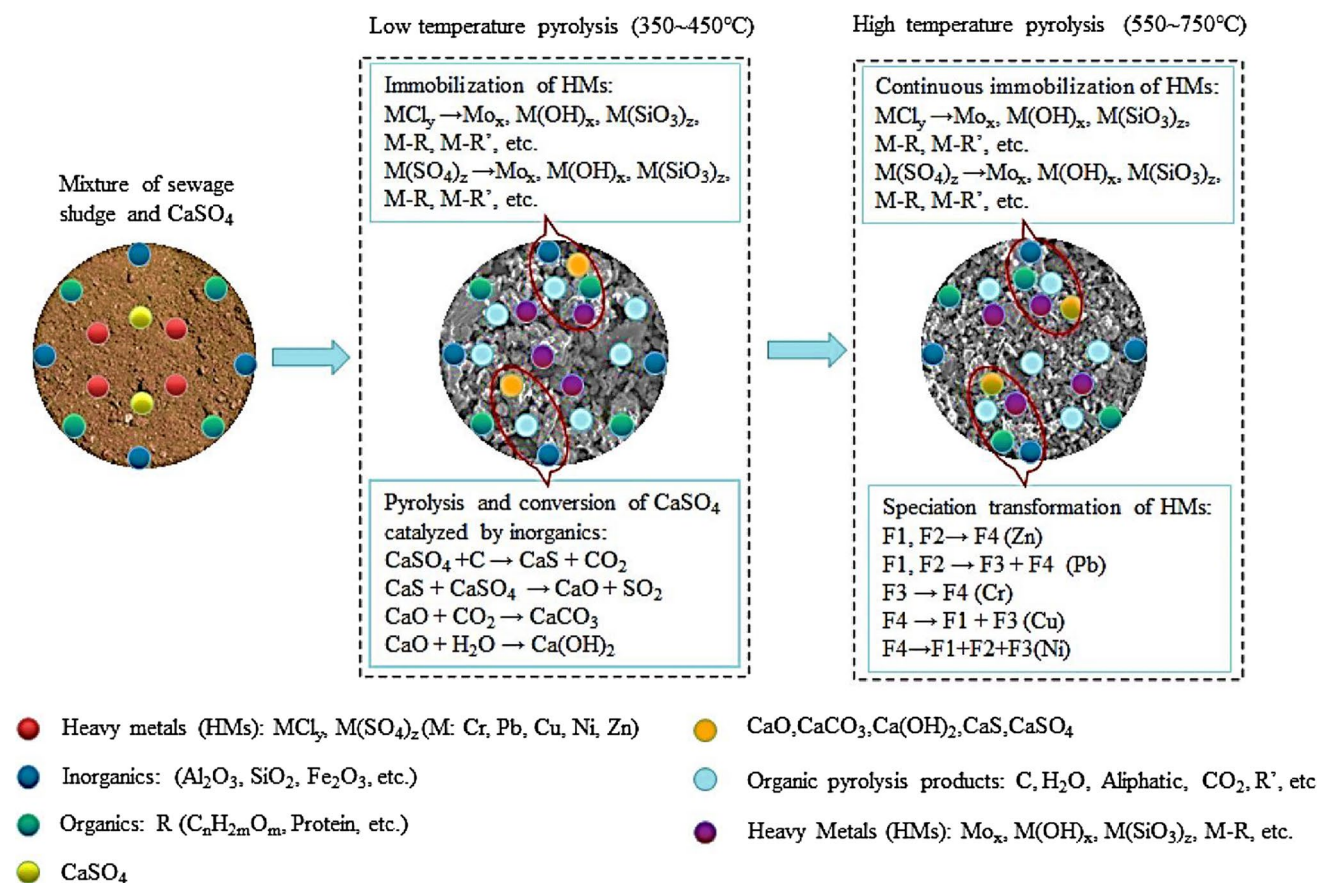
Fig. 3 Heavy metals immobilization during sewage sludge and calcium sulfate ($CaSO_4$) co-pyrolysis. As shown in the figure, pyrolysis temperature affects the mechanism of immobilization of heavy metals. Exch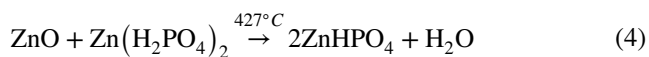angeable fraction (F1), reducible fraction (F2), oxidizable fraction (F3), and residual fraction (F4). High pyrolysis temperatures with the use of $CaSO_4$ promote the transformation of extractable

heavy metals into more stable fractions, such as oxidizable fraction (F3) and residual fraction (F4). $CaSO_4$ can react with organic intermediates and form carbonates, oxides, and hydroxides, which convert heavy metals into stable forms. Reprinted from reference (Liu et al. 2021a) with permission from Elsevier

iron oxides, in addition to other organic pyrolysis products (water, new organics, and carbon), would then react with heavy metals and convert them into other stable forms, including silicate minerals, carbonates, organometallics, and alkali (Liang et al. 2014; Liu et al. 2021a). In addition to the catalytic effects of inorganics in sewage sludge, $CaSO_4$ can combine with other organic intermediate products and form $CaCO_3$, $Ca(OH)_2$, CaO, and CaS (Fig. 3), which are efficient in promoting heavy metals immobilization (Huang et al. 2017; Liu et al. 2021a). Additionally, high pyrolysis temperatures would increase the transformation of Cr, Pb, and Zn into silicates, iron and manganese oxides (Liu et al. 2017, 2021a). In contrast, Gu and co-workers studied the effects of co-pyrolyzing sewage sludge with different mixtures including 0, 15, and 30 wt.% of $Ca(H_2PO_4)_2$ on immobilizing heavy metals and reducing metals leachability. The optimal blending ratio of $Ca(H_2PO_4)_2$ is 15 wt%, which produced biochar with more stable heavy metals and

reduced the environmental risks from 65.73 to 4.39 (Gu et al. 2022). The potential mechanisms for heavy metals stabilization include the formation of heavy metal chelating agents, e.g., metaphophates and polyphosphates, via condensation dehydration reactions of $Ca(H_2PO_4)_2$, the formation of highly aromatic metallic compounds, and physical adsorption (Fig. 4). Equations 1, 2, 3 ,4, 5 show the possible reaction pathways for the formation of zinc pyrophosphate $(Zn_2P_2O_7)$, which is a stable form of zinc and insoluble in water (Gu et al. 2022). Phosphates can also help the formation of other stables forms of lead $(Pb_2P_2O_7)$ and pyromorphite-like minerals (Mohamed et al. 2017; Gu et al. 2022). X-ray diffraction (XRD) analysis confirmed the formation of $Pb_2P_2O_7$, $Fe_2Pb_3(PO_4)_4$, $Cd(PO_3)_2$, $(Cu, Zn)PbVO_4(OH)$ (Mottramite), and $Cu(OH)_3PO_4$ after co-pyrolyzing sewage sludge with $Ca(H_2PO_4)_2$ (Gu et al. 2022).
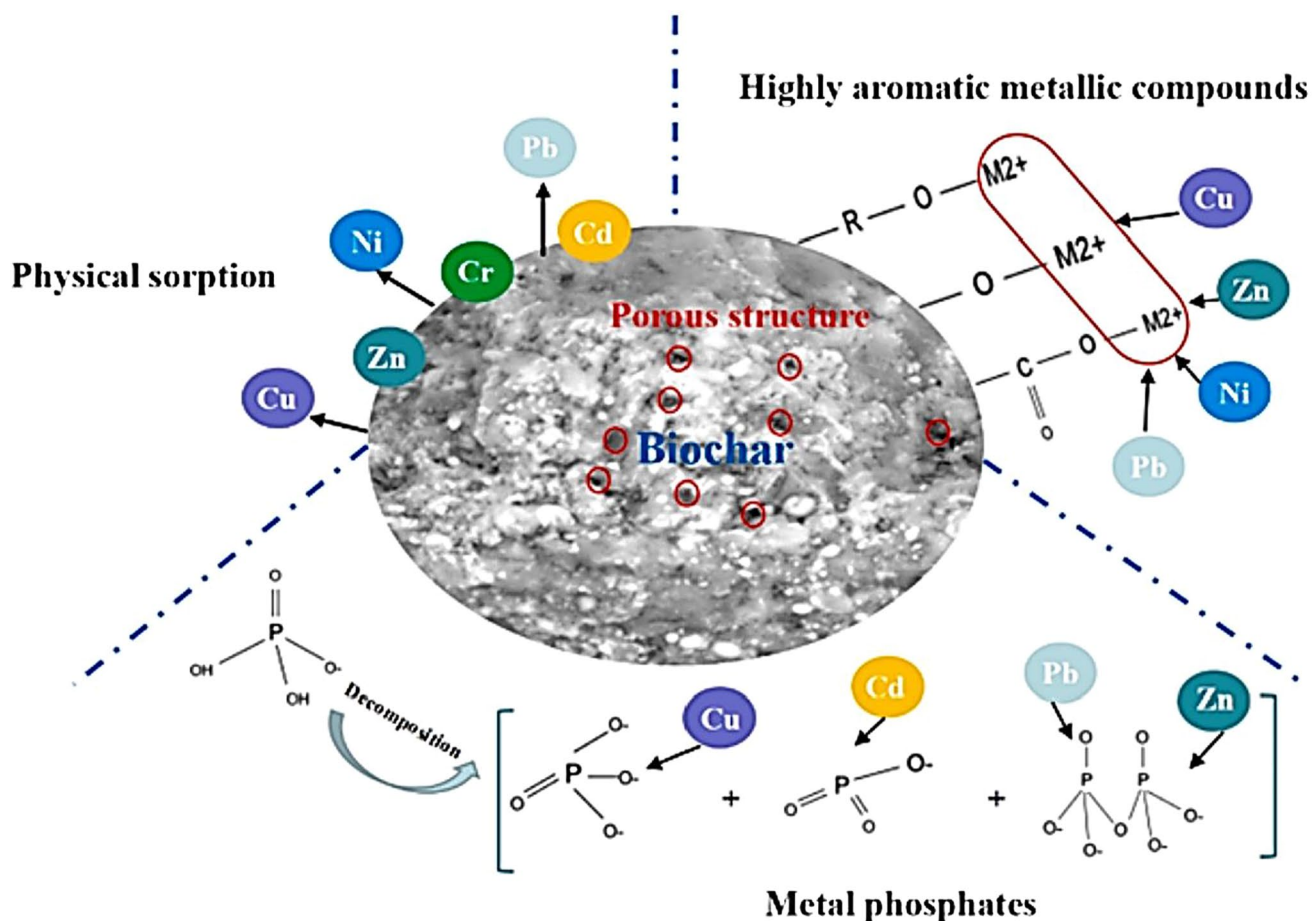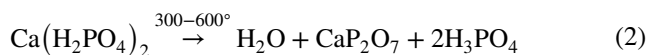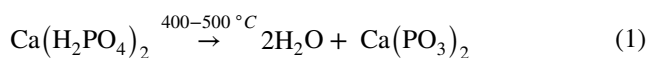

Fig. 4 Heavy metals immobilization during sewage sludge and monocalcium phosphate "$Ca(H_2PO_4)_2$" co-pyrolysis. As shown in the figure, several mechanisms could affect the immobilization of heavy metals in sewage sludge. $Ca(H_2PO_4)_2$ promoted the formation of metal phosphates, which react with heavy metals and form stable forms of phosphates, e.g., pyromorphite-like minerals and mottramite. $Ca(H_2PO_4)_2$ also increased the surface porosity of biochar, which can immobilize heavy metals inside the internal pores of biochar via physical adsorption. Reprinted from reference (Gu et al. 2022) with permission from Elsevier

$$Ca(H_2PO_4)_2 \xrightarrow{400-500\,°C} 2H_2O + Ca(PO_3)_2 \qquad (1)$$

$$Ca(H_2PO_4)_2 \xrightarrow{300-600°} H_2O + CaP_2O_7 + 2H_3PO_4 \qquad (2)$$

$$4ZnO + 4H_3PO_4 \rightarrow Zn_3(PO_4)_2 + 4H_2O + Zn(H_2PO_4)_2 \qquad (3)$$

$$ZnO + Zn(H_2PO_4)_2 \xrightarrow{427°C} 2ZnHPO_4 + H_2O \qquad (4)$$

$$2ZnHPO_4 \xrightarrow{427°C} Zn_2P_2O_7 + H_2O \qquad (5)$$
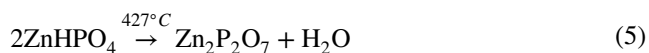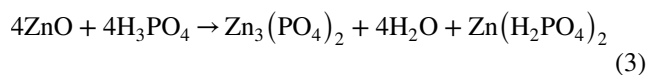
Min and co-workers studied the effect of using $ZnCl_2$ for heavy metals stabilization in biochars obtained from camellia oleifera shells and sewage sludge co-pyrolysis. $ZnCl_2$ impregnation reduced the overall heavy metal content through chlorination and formation of $CdCl_2$ and $PbCl_2$, whereas the heavy metals oxidizable fraction (F3) increased remarkably, with the most noticeable effects found in nickel (Ni) and copper (Cu). The findings of the ecological risk assessment revealed that the impregnation of $ZnCl_2$ with sewage sludge and camellia oleifera co-pyrolysis had a high ability to diminish the sludge-derived biochars heavy metals environmental risk, which reduced the risk index from 407 for sewage sludge biochar to 224 after combining $ZnCl_2$ impregnation (Min et al. 2022). However, zinc is also a heavy metal, and the authors neither reported the content of zinc in the produced biochars nor included the content of zinc in calculating the risk index assessment, which could be remarkably higher than the reported risk index value.

For the effect of carbonates on heavy metals stabilization, the activation with K_2CO_3 during the co-pyrolysis of sewage sludge and cotton stalks converted heavy metals further into stable compounds, including oxidizable (F3) and residual (F4) fractions. The leachability of heavy metals and the associated environmental concerns were significantly decreased by K_2CO_3 activation as well. Using K_2CO_3 raised the biochar's alkalinity and promoted the generation of CaO, $CaCO_3$, and aluminosilicates, all of which helped to immobilize the heavy metals (Wang et al. 2021b). Mixing kaolin or zeolite with sewage sludge can stabilize heavy metals during sewage sludge co-pyrolysis via reacting oxygen ions of aluminosilicate additives and creating silicates ($PbSiO_3$) and aluminates ($CuAl_2O_4$), converting heavy metals into stable fractions (Li et al. 2022a). Note that kaolin might dehydroxylate at high temperatures, making kaolin ideal for heavy metal ion reactions. Kaolin dehydroxylation might be one of the reasons why kaolin was more successful than zeolite in heavy metal stability (Li et al. 2022a).

In contrast, the overall mechanism should be related to the quantity of organic components in municipal solid waste, which might result in the creation of diverse organometallic complexes for varied heavy metals (Wang et al. 2021a). However, the actual pathways of heavy metal immobilization caused by sewage sludge co-pyrolysis have not been substantiated and may require more investigation. Based on the aforementioned information, many mechanisms could play a key role in affecting heavy metal stabilization, which could be affected by the co-feed mixing ratios, use of catalysts, and pyrolysis temperatures. Overall, using a nontoxic metal catalyst could be a promising approach to further reduce the toxicity of the sludge-derived biochars. Future studies should investigate the effect of using different catalysts at different mixing ratios on heavy metals stabilization.

Factors affecting biochar performance and toxicity in soil

Many interconnected factors play a role in determining biochar performance in soil. The type of soil minerals, parent material, temperature, soil texture, aeration, mycorrhizal growth, and soil organic matter (SOM) and water content are the most important aspects. Additionally, biochar's porous structure and huge specific surface area make biochar an ideal substrate for the adsorption of nutrients because of biochar high surface charge density and abundance of polar and nonpolar surface sites (Jin et al. 2017; Zhao et al. 2017; Yang et al. 2021). Increased cation exchange capacity and decreased leaching of nutrients have been linked to biochar application in soil. Plants can absorb nutrients from biochar that are released over time (Wang et al. 2019a; Yang et al. 2021).

Effect of biochar pH

Soil processes depend heavily on pH, stability, nutrient release, anion and cation solubility, nutrient leaching, and crop yields are all affected by biochar stability in the soil and the stability of soil organic matter (SOM). Slightly alkaline or neutral in pH, biochar is the norm (Wang et al. 2019a; Yang et al. 2021). With increasing pyrolysis temperature, biochar's basicity increases and varies with the feedstock. The liming effect of alkaline biochar has been proven to reduce the concentration and toxicity of Al^{3+} in acidic soils, while biochar has little impact or causes a reduction in pH when given to acidic soils. On the other hand, alkaline biochar may harm soils with low buffering capacity. Adding alkaline biochar derived from poultry litter to soil, which raises the pH above 8, has a detrimental effect on fertility. Plant nutrients may precipitate at alkaline pH, reducing nutrients availability for plant growth (Wang et al. 2019a;

Yang et al. 2021), which emphasize the importance of determining biochar pH before using on potentially vulnerable soil types.

When dealing with metal-contaminated agricultural soil, special circumstances may occur. Increased soil pH may help reduce the mobility of potentially hazardous elements (Jin et al. 2017; Wang et al. 2019a; Yang et al. 2021). However, as prefers to bind in acidic soils to iron-oxide surfaces and can be liberated from the soil as the pH rises. The addition of biochar has resulted in mobilization in several experiments (Wang et al. 2019a; Yang et al. 2021). As concentrations in the porewater of alkaline (pH 10) biochar increased five to ninefold within one week, the absorption of *Solanum Lycopersicum L.* was inhibited. According to yet another study, using rice plant biochar with pH 8.1–11.3 increased bioaccumulation of heavy metals in rice shoots by up to 327% (Godlewska et al. 2021). Overall, further studies are still needed.

Effect of biochar nutrient content

Biochar may also contain nutrients, e.g., N, P, K, Mg, S, and Ca, and various micronutrients depending on feedstock type and the production method. However, some biochars are described as holding less nutrients to be utilized as fertilizers or even limiting the availability of nutrients for plants. Despite these conflicting results, biochar significantly impacts nutrient (bio)availability (Jin et al. 2017; Zhao et al. 2017; Yang et al. 2021).

In addition, the amount of total N in soil rises, but bioavailable N does not. Biochar can also impede the availability of N to plants, resulting in lower yields if fertilizer is not used in conjunction with biochar. Furthermore, using biochar and fertilizer together may increase the amount of biomass absorbed (Wang et al. 2019a; Yang et al. 2021). Impacts of biochar on soil phosphorus availability varies from one experiment to the next. Phosphorus adsorption on biochar is primarily responsible for controlling P availability, not biochar's ability to raise or decrease P availability. Even though biochar has a favorable impact on P values in severely weathered acidic soils, the response in alkaline soils is more complicated (Zhao et al. 2017; Wang et al. 2019a; Yang et al. 2021). For instance, the addition of alkaline biochar to calcareous soil improves P sorption while decreasing the availability of adsorbed P substantially.

When wheat straw biochar and fertilizer are combined and generate phosphate precipitation and sorption, biochar negatively impacts plant P availability in saline-sodic soil. According to the current data, biochar appears to have a major impact on soil nitrogen cycles (Jin et al. 2017; Zhao et al. 2017; Yang et al. 2021). However, biochar's strong sorption capacity may limit nutrients' bioavailability, even if the overall nutrient content increases, particularly when biochar and fertilizer are applied together. Biochar has so much promise, which have been studied as a technique for removing nitrogen and phosphorus contamination from the environment (Wang et al. 2019a; Yang et al. 2021). Biochar-soil systems must be evaluated for the overall nutrient content and the efficient bioavailability of these nutrients, which necessitates a series of bioassays.

An interesting correlation was found between soil ecosystem C/N and other nutrients ratios and biological ties and consistency in stoichiometry. A stoichiometric unbalance between C and N, according to the C health threshold model, could result in soil deterioration or a reduction in soil health (Wang et al. 2019a; Yang et al. 2021). As a result, the primary productivity of an ecosystem is affected by the coupling of C and N interactions. Overall, the use of biochar as a climate mitigation method may be undermined by the improper usage of biochar that reduces N bioavailability.

Effect of agrochemicals in soil

Because of the porous nature and large specific surface area of biochar, which can impact the efficacy of these agrochemicals, insecticides, herbicides, fungicides, and nematicides are examples of agrochemicals containing many pesticides. In addition to synthetic fertilizers, hormones and other chemical growth agents can affect soil biota. Reducing pesticide levels in non-target organisms and groundwater contamination by fixing pesticides on pesticide-affected plants is thus essential. Insecticides applied to soil directly may also harm biochar effectiveness, particularly for the pre-emergent pesticides that contact biochar in the soil directly (Jin et al. 2017; Zhao et al. 2017; Wang et al. 2019a; Yang et al. 2021). As a result of pesticides' strong interactions with biochar, bioavailability of pesticides may be lowered, requiring higher treatment rates to maintain adequate control. For example, to obtain the same weed control efficacy as non-treated soil, greater rates of the herbicide diuron, beginning at 0.1% of biochar in soil, must be administered. Even though when the dose of diuron is doubled, weeds cannot be effectively controlled in soils containing 0.5% biochar (Wang et al. 2019a; Yang et al. 2021).

Herbicide efficacy has been found to decline with increasing soil biochars doses, raising questions about such increased herbicide application's economic and environmental ramifications. Increased sorption also limits bioavailability for microbe degraders, reducing pesticide or herbicide decomposition while increasing persistency, which has several implications. Atrazine leaching, bioavailability, and mineralization are all reduced to a larger extent when fine biochar (2 mm) is applied to the soil rather than large-size biochar higher than 2 mm. As a result, a decrease in pesticide effectiveness following the use of fine high-temperature biochar is to be predicted. Furthermore, porosity, hydrophobic

binding sites, and biochar surface area are crucial for sorption often rise with increasing temperature. Organic contaminants such as pesticides are more likely to associate with biochar produced over 500 °C (Yang et al. 2021).

The potential negative effects of biochars application in agricultural soils and the possible mitigation measures are presented in Table 5. Optimization of the co-pyrolysis process in addition to the proper selection of the co-feed material are essential to mitigate most of the negative impacts of biochar application in agricultural soils (Brtnicky et al. 2021). Overall, for pesticide sorption investigations in the laboratory and more sophisticated ones like microcosms and field trials, various biochars with well-defined sorption potential can be tested to see whether biochars characteristics affect pesticide sorption.

Effect of biochar application on soil biota and environmental risk assessment

Biochar can be applied to soil to mitigate the detrimental effects that could occur as a result of removing crop residues from soils and that would possibly ameliorate soil quality and plant productivity (Mohamed et al. 2017, 2021b, 2022c; Fakayode et al. 2020). However, several studies found that biochar has a negative effect on earthworms in most studies of soil macro- and microfauna because of the contents of polycyclic aromatic hydrocarbons and heavy metals (Li et al. 2011; Conti et al. 2018; Godlewska et al. 2021). To evaluate the environmental impact of the suitability and optimum rate of biochar application in soils, the bioavailability and phytotoxicity of metals in the sludge-derived biochar should be estimated (Devi and Saroha 2014; Zhao et al. 2017). The environmental ecological risk index is widely used to determine potential exposure for various metals in the sewage sludge and the sludge-derived biochars, by

utilizing Hakanson's (1980) model (Jin et al. 2017; Yang et al. 2021). Extraction studies using Community Bureau of Reference recommended extraction procedures demonstrated that heavy metals have been immobilized in biochars after heavy metals are converted into stable chemical forms (Wang et al. 2019a; Xie et al. 2020; Godlewska et al. 2021). The Community Bureau of Reference classifies the extracted metals as exchangeable fraction (F1), reducible fraction (F2), oxidizable fraction (F3), and residual fraction (F4) (Xie et al. 2020; Godlewska et al. 2021).

Biochar's impacts on the survival and behavior of earthworm, reproduction and development were found to be inconsistent, which shows that our knowledge of the interactions of biochar organism is still limited. As with other groups of creatures, there have been a variety of findings. There was a significant drop in the density of both cocoons and juvenile earthworms, as well as a reduction in casting activity, when biochar was applied to *Pontoscolex corethrurus* compared to pure soil (Paz-Ferreiro et al. 2015). Many studies have reported that exposure to biochar has a negative impact on earthworm reproduction rates, toxicity, and growth and weight (Godlewska et al. 2021).

According to other researchers, *E. fetida's* degradation within five days of exposure to biochar may have been caused by a fast adjustment in soil pH following biochar addition (Conti et al. 2018). Using pre-moisturized biochar, the authors theorized that earthworm mortality was caused by biochar particles adhering to earthworm bodies and inflicting substantial physical injury. To avoid consuming biochar particles, earthworms often lose weight (Conti et al. 2018), which might be explained by earthworms cutting back on feeding activities to conserve energy. As a result of the decrease in earthworm numbers caused by biochar, the ecosystem services provided by earthworms may be compromised. For springtails, biochar has also been shown to have detrimental consequences.

Table 5 Potential negative effects of biochars and possible mitigation measures

Biochar negative effects	Possible solutions
Reduced efficacy of agrochemicals	Selecting the appropriate timing of agrochemicals
	Use appropriate dosing of biochar
Formation of toxic PAHs	Optimize pyrolysis conditions (temperature, atmosphere, and residence time)
	Higher pyrolysis temperature (more than 500 ºC)
Formation of toxic dioxins	Optimize pyrolysis conditions (temperature, atmosphere, and residence time)
	Higher pyrolysis temperature (more than 500 ºC)
Excessive increase in soil pH	Optimize pyrolysis conditions (temperature, atmosphere, and residence time)
	Use appropriate dosing of biochar
Increase soil salinity	Optimize pyrolysis conditions (temperature, atmosphere, and residence time)
	Pre-treatment for mitigation of reduce plant growth
Adverse effects on soil invertebrates	Use appropriate dosing of biochar
	Use of biochar with low content of toxic chemicals

Various forms of biochar increased mortality while decreasing reproduction in *Folsomia candida* (Conti et al. 2018; Godlewska et al. 2021). When biochar is applied at a $1-5\%$ concentration, springtail populations are protected from poisoning, but when biochar concentration is increased to 10%, these positive effects are completely reversed. Soil microfauna can be stressed by alkaline biochar with pH 9 derived from wheat straw, according to this study (Conti et al. 2018). Some insect species are unable to survive in direct contact with biochar. For a variety of reasons, adding biochar can pose negative impacts on soil quality, including altering soil structure, decreasing mineral and vitamin availability due to sorption in biochar, and increasing the saltiness of biochar. Various types of hydrochar and biochar can affect soil invertebrates if the samples contain soluble organic compounds, while sewage sludge application increases the mortality of soil invertebrates (Li et al. 2011; Oleszczuk and Hollert 2011). As a result of soil communities' decreased potential resilience to environmental changes, the effects of intensive biochar application to agricultural soils should be investigated.

According to the findings, soil invertebrates may have detrimental effects and various inhibitory on their development, survival and reproduction, if larger dosages of biochar, higher than 0.5%, depending on biochar type, are applied. As a result of adding biochar to the soil, these negative impacts are partly caused by the soil's increased pH level but also by the presence of polycyclic aromatic hydrocarbons, dioxins, and other potential toxic elements (Kong et al. 2019; Brtnicky et al. 2021). However, biochar produced from sewage sludge and willow co-pyrolysis was found to reduce the ecotoxicity for arthropods, *Lepidium sativum* and *Folsomia candida* by up to 56% and enhanced reproduction stimulation by $20-98\%$ (Kończak et al. 2020). Producing biochars at high pyrolysis temperatures more than 500 °C could reduce the formation of dioxins (Brtnicky et al. 2021). If nematodes and springtails are abundant, biochar may have less of an effect on the microbial community's composition and, as a result, soil quality. Overall, because earthworms play such an important role in the formation of soil, any negative effects on them may be significantly more serious. Because of this, the earthworm population may be negatively affected by biochar.

Conclusion

This review discusses the latest advances in research experiments about the mechanisms and synergistic effects of co-pyrolysis as well as sewage sludge catalytic co-pyrolysis with a focus on reducing the environmental risk of the sludge-derived biochar. Co-pyrolyzing sewage sludge with biomass feedstocks was found to enhance heavy metals stabilization and reduce the toxicity of the sludge-derived biochar. Furthermore, the use of catalysts during co-pyrolysis significantly affects the pathways and mechanisms of heavy metals stabilization through the formation of stable compounds, including oxidizable and residual fractions, which reduced the environmental risk of sludge-derived biochar compared with sewage sludge singular pyrolysis. Pyrolysis temperatures and residence time have significant effects on the stabilization of heavy metals. Using CO_2 as a carrier gas decreased the leachable metals in biochars and affect the total concentration of heavy metals. Sewage sludge-derived biochars could negatively affect soil biota, including earthworms. As a result, the toxicity of the sludge-derived biochar should be eliminated or reduced before applying biochar to the soil. Selection of the co-feed material and optimization of the co-pyrolysis process are key parameters to mitigate the negative impacts of biochar application in soil, which should be further investigated. As a result, future research might look at the impact of employing different atmospheres including CO_2, N_2, pyrolysis gas, and other mixes, with different biomasses for sewage sludge co-pyrolysis. Catalytic pyrolysis has the potential to decrease the operating costs of biorefineries that use biomass and sewage and to stabilize heavy metals in the sludge-derived biochars. Thus, more future studies are needed to explore the effects of nontoxic catalysts on the stabilization of heavy metals. Furthermore, the long-term interactions between soil biota and sludge-derived biochar applied in agricultural soils should be monitored. The long-term environmental implications of various co-pyrolysis models is still needed to demonstrate that co-pyrolysis should be considered as a viable solution to current technologies.

Declarations

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