Biochar

REVIEW

Nanobiochar for the remediation of contaminated soil and water: challenges and opportunities

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Abstract

Although research on biochar has received increasing attention for environmental and agricultural applications, the significance of nanobiochar for environmental pollutant remediation is poorly understood. In contrast to bulk biochar, nanobiochar has superior physicochemical properties such as high catalytic activity, unique nanostructure, large specific surface area and high mobility in the soil environment. These unique characteristics make nanobiochar an ideal candidate for pollution remediation. Thus far, the research on nanobiochar is still in its infancy and most of the previous studies have only been conducted for exploring its properties and environmental functions. The lack of in-depth summary of nanobiochar's research direction makes it a challenge for scientists and researchers globally. Hence in this review, we established some key fabrication methods for nanobiochar with a focus on its performance for the removal of pollutants from the environment. We also provided up-to-date information on nanobiochar's role in environmental remediation and insights into different mechanisms involved in the pollutant removal. Although, nanobiochar application is increasing, the associated drawbacks to the soil ecosystem have not received enough research attention. Therefore, further research is warranted to evaluate the potential environmental risks of nanobiochar before large scale application.

Highlights

Key synthesis technologies and properties of nanobiochars were discussed.
The interactions pathways of nanobiochar with pollutants were elucidated.
Role of environmental factors on pollutants remediation and life cycle was delineated.
Important research outlooks on nanobiochar's potential in pollutant remediation were elaborated.

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Graphical Abstract

1 Introduction

Environmental pollution threatens the soil and water resources and needs immediate global attention (Hou et al. 2020; Pure Earth/Green Cross 2015; WHO 2019; Zhang et al. 2016). The pollutants may enter the environment through natural processes and anthropogenic activities such as industrial emissions, domestic sewage discharge and agricultural activities (Jaishankar et al. 2014; Liu et al. 2022; Young et al. 2004). Generally, pollutants are classified as biodegradable and non-biodegradable (Gong et al. 2022; Samadi et al. 2021). Heavy metals are non-biodegradable, which makes it difficult to eliminate them from the soil and water environments, resulting in their accumulation by living organisms (Kirpichtchikova et al. 2006). Globally, over 20 million hectares of land have been polluted with heavy metals (Liu et al. 2018b). Prolonged exposure to heavy metals negatively impacts human health and may cause various cancers, hallucinations, insomnia, and other diseases (Zhang et al. 2022b). Although organic pollutants are biodegradable, some persistent organic pollutants are often toxic and cancerogenic to human and animals even

in small amounts (Liu et al. 2020b; Singh et al. 2022). For example, Khan et al. (2021) found that traces of pharmaceutical compound residues in the aquatic environment may pose a threat to the public health and they have even been detected in trace amounts in surface water. Numerous studies have reported that unprecedented amounts of pollutants have accumulated both in aquatic and soil systems (Hou et al. 2020; Gong et al. 2022; Pure Earth/Green Cross 2015). Therefore, it is of great importance to conduct soil and water remediation activities.

In recent years, biochars have been widely used as effective amendments to remediate polluted soil and water (Azadi and Raiesi 2021; Chen et al. 2022; Luo et al. 2022; Wang et al. 2021). Studies have shown that biochar can effectively remove pollutants from wastewater (Li et al. 2020; Lian et al. 2020; Ren et al. 2022; Zhang et al. 2019a) and reduce pollutant bioavailability in soils (Sun et al. 2022). Biochar is pyrolyzed from biomass raw materials under low to high temperatures (300–700 °C) in oxygen limited conditions (Lehmann 2007; Li et al. 2022). Since the raw material sources of producing biochar are usually agricultural and forestry

wastes, it is economical and environmentally friendly and may also become an effective additive for carbon sequestration (Lyu et al. 2020; Tripathi et al. 2011). Biochar with particle sizes in the range of 1–100 nm is considered as nanobiochar (Naghdi et al. 2017a; Sulaiman et al. 2013; Zhang et al. 2022b). Generally, nanobiochar can be obtained by two main ways. One is artificially manufactured and, the other is physical disintegration of biochar by aging process in the environment (Huang et al. 2021; Ramanayaka et al. 2020b). Compared to bulk biochar, nanobiochar a much larger specific surface area (SSA), smaller hydrodynamic radius, more negative zeta potential, and more oxygen-containing functional groups (Lyu et al. 2018a; Naghdi et al. 2017a; Pratap et al. 2022; Ramanayaka et al. 2020b).

Nanobiochar has been studied and applied to different fields including biosensors, soil amendments, photocatalytic materials, and as pollutant adsorbents (Fig. 1a, also see Dong et al. 2018; Xiao et al. 2020; Zhang et al. 2022b). The scientmetric analysis through Web of Science displayed that the number of articles on the term "nanobiochar and nano-biochar" has increased in recent years from 2017 to 2021, and it now appears that research in 2022 will also likely exceed that in 2021 (Fig. 1b). Furthermore, the use of nanobiochar to remediate polluted soils and water was the most

Fig. 1 Summary of research hotspots by using term "nanobiochar" as the keyword (**a**). Proportion of publication by research areas in recent 5 years using the term "nanobiochar, nano-biochar and ball milled biochar" separately, as keywords (**b**). Sourced from web of science, data accessed on 4 August 2022 (**c**)

studied areas (account for 51%, see Fig. 1c). Nanobiochar can be an effective additive for pollutant removal from polluted water (Lyu et al. 2018a; Xiao et al. 2020; Zhang et al. 2021) and can also be used for stabilization of pollutants in polluted soils (Sun et al. 2022; Zhang et al. 2022a). Despite this, the total number of papers searched in Web of Science in the past 5 years was less than 70, which further suggests that this is a new field, and a lot of research needs to be carried out.

As a new research direction in the field of biochar, the research on nanobiochar has not yet been done systematically at present. In addition, long-term research on the environmental functions of nanobiochar is lacking. Although there have been some articles reviewing related research on nanobiochar, these reviews either focus on the properties of nanobiochar or aim at broad overviews of its application for many fields. To our knowledge, there is currently no detailed review that summarizes the use of nanobiochar in environmental pollution remediation. Therefore, this review aimed to summarize the results from studies (1) concerning the effects of nanobiochar on the remediation of water and soil pollution, (2) on the preparation process of nanobiochar, (3) concerning affecting nanobiochar performance in environmental pollutant remediation and the associated mechanisms. This will provide a theoretical basis and practical guidance for the safe and effective use of nanobiochar for environmental pollution remediation.

2 Preparation of nanobiochar

Generally, the raw material of nanobiochar will affect its physiochemical properties. It has been reported that biomass with high hemicellulose usually derives nanobiochar with low carbon content and high oxygen content (Weber and Quicker 2018), while materials with high lignin usually produce nanobiochar with strong aggregation ability (Föhr et al. 2017). Oleszczuk et al. (2016) observed that wicker and wheat straw nanobiochars had relatively open-structured primary small particles while miscanthus nanobiochar possessed large spherical particles and exhibited weaker aggregation effects than the first two nanobiochars. Moreover, the ash content of nanobiochar produced from agricultural waste was directly proportional to the bulk biochar, however, this was not observed in nanobiochar produced from municipal waste (Song et al. 2019). Usually, nanobiochar is produced in the same manner as bulk biochar with a few additional steps (Chausali et al. 2021), which will be discussed in detail below (see Fig. 2).

2.1 Milling method

Ball milling has been a commonly used approach for producing nanobiochar, which can mechanically break
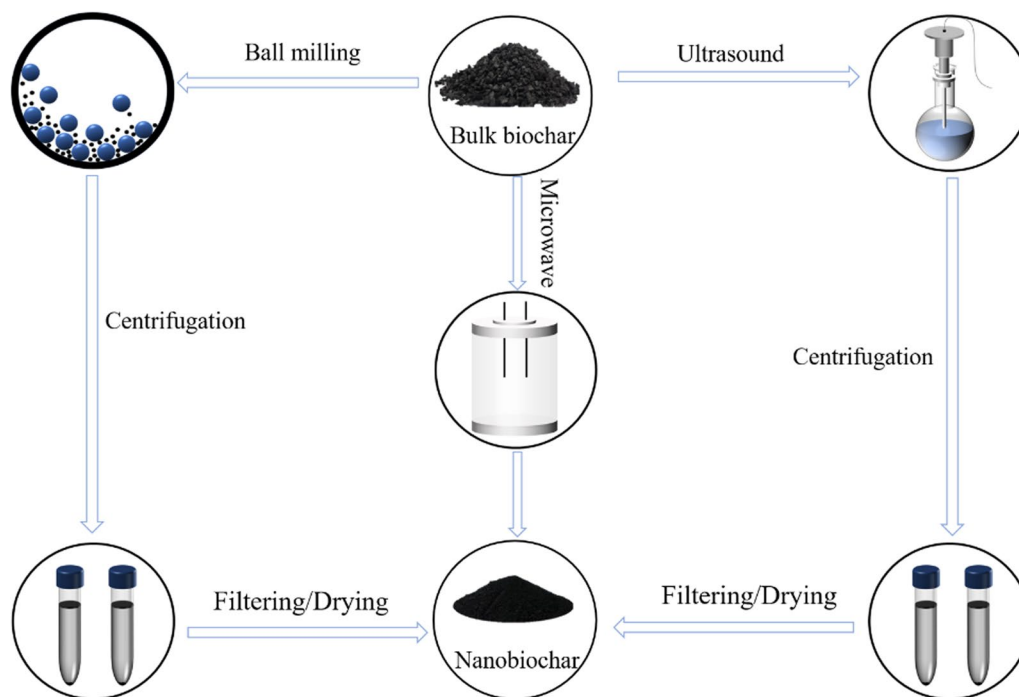

Fig. 2 Methods (ball milling, ultrasound and microwave methods) used for preparing nanobiochar from bulk biochar

the biochar and reduce particles to nanometer size (Anupama and Khare 2021; Chausali et al. 2021; Lyu et al. 2017; Soares et al. 2015). In the milling process, smaller pores with flat pieces are formed from squeezing of longitudinal pores and canals, which leads to the increase of SSA (Ramezanzadeh et al. 2021). This method includes dry milling mode and wet milling mode, which is based on whether the biochar is mixed with water or not during the grinding process. Yuan et al. (2020) showed that wet ball milling for 12 h was more effective in obtaining smaller particle size and more evenly distributed biochar than dry ball milling, and both ball milling methods can obtain larger SSA of biochar than hand grinding.

The ball milling method has been demonstrated to be a green approach with low cost and reproducibility (Kumar et al. 2020; Naghdi et al. 2017b; Wang et al. 2018), however, it is not the only milling method used for nanobiochar production. For example, the double-disc milling method has been used to produce polygonal platelet-shaped particles with the diameter of 50–150 nm (Ramanayaka et al. 2020a). In addition, other studies have found that vibrating disc milling can produce more uniformly shaped nanobiochar and has a greater yield than ball milling (Bayram and Öner 2007; Huang et al. 2022). The selection of a suitable milling method for nanobiochar production usually depends on the purpose of the experiment and raw materials.

2.2 Ultrasonication method
Ultrasonication is another effective method for producing nanobiochar. During the sonication process, a probe tip is immobilized below the surface of the suspension where biochar particles are disintegrated and biochar's graphite-like structure is exfoliated by shockwaves (Li et al. 2013; Oleszczuk et al. 2016). The main mechanism involves the use of ultrasound-induced microjets and shockwaves to create new pores, open blocked pores, and change the biochar structure, which ultimately leads to an increase of microporosity (Sajjadi et al. 2020). Liu et al. (2018a) showed that the fine fragments adhering to the surface or embedded in the pores of the parent biochar are separated by ultrasonic waves and longer sonication time may produce more nanobiochar. Dong et al. (2019) found that the ultrasonically treated regular biochar successfully produced nanobiochar with a particle size of 42 ± 6 nm. Song et al. (2019) carried out ultrasonic work on bulk biochars derived from different raw materials for 30 min and nanobiochar was obtained with a particle size less than 100 nm.

In addition to the above methods, other approaches have also been applied to prepare nanobiochar, for example, microwave pyrolysis reactors have been used to directly produce nanobiochar (Wallace et al. 2019). This method is cost-effective and efficient because only one step is required (Li et al. 2016). Centrifugation technology is another direct method or a step to produce nanobiochar and the resulting biochar nanoparticles are ordered (Lian et al. 2020; Xu and Cölfen 2021). The digestion of the bulk biochar by concentrated nitric and sulfuric acids can also be used to produce nanobiochar (Guo et al. 2020).

2.3 Modified nanobiochar
More recently, many methods have been employed for the modification of bulk biochar and nanobiochar in order to improve their properties (Ndirangu et al. 2019; Wang et al. 2021), which mainly include metal modification (Jenie et al. 2020; Nath et al. 2019; Qiao et al. 2019), ultrasonic modification (Li et al. 2017b), acid-base modification (Mahmoud et al. 2020a), and thiol modification (Lyu et al. 2020) (see Fig. 3). For example, Lyu et al. (2020) modified wood chip nanobiochar with 3-mercaptopropyltrimethoxysilane (3-MPTS) and found that the SSA of the modified nanobiochar was much larger than that of the pristine biochar, the zeta potential was more negative, and it had more surface functional groups. Xu et al. (2019) compared the properties of pristine biochar, ball-milled biochar, and ball-milled N-doped biochar and found that the SSA and pore volume of ball-milled biochar significantly increased after adding ammonium hydroxide. This suggests that ammonium hydroxide may act as an activator to further generate pores on the biochar during ball milling process. The N-doped nanobiochar, however, was easier to agglomerate due to hydrophilic groups present on the biochar particles. Nath et al. (2019) modified rice husk nanobiochar with iron oxides to increase the nanobiochar C–O and Fe–O functional groups and optimize its adsorption capacity. Many studies have shown that the modified nanobiochars has a stronger ability to remove pollutants than the non-modified nanobiochar (Vishnu et al. 2022; Wang and Wang 2018).

3 Adsorption of environmental pollutants by nanobiochar
3.1 Heavy metals
As shown in Table 1, nanobiochar has been demonstrated to be an effective tool for heavy metal immobilization in the environment. For instance, previous studies have shown that nanobiochar could strongly absorb Cd, Pb or Cu in aqueous system, and the heavy metal removal rate increases gradually as the concentration of nanobiochar increases. This increase in heavy metal removal rate may be due to the adsorbent exchangeable sites and the larger SSA of nanobiochar (Li et al. 2017b; Yue et al. 2019a; Vishnu et al. 2022). However, high nanobiochar
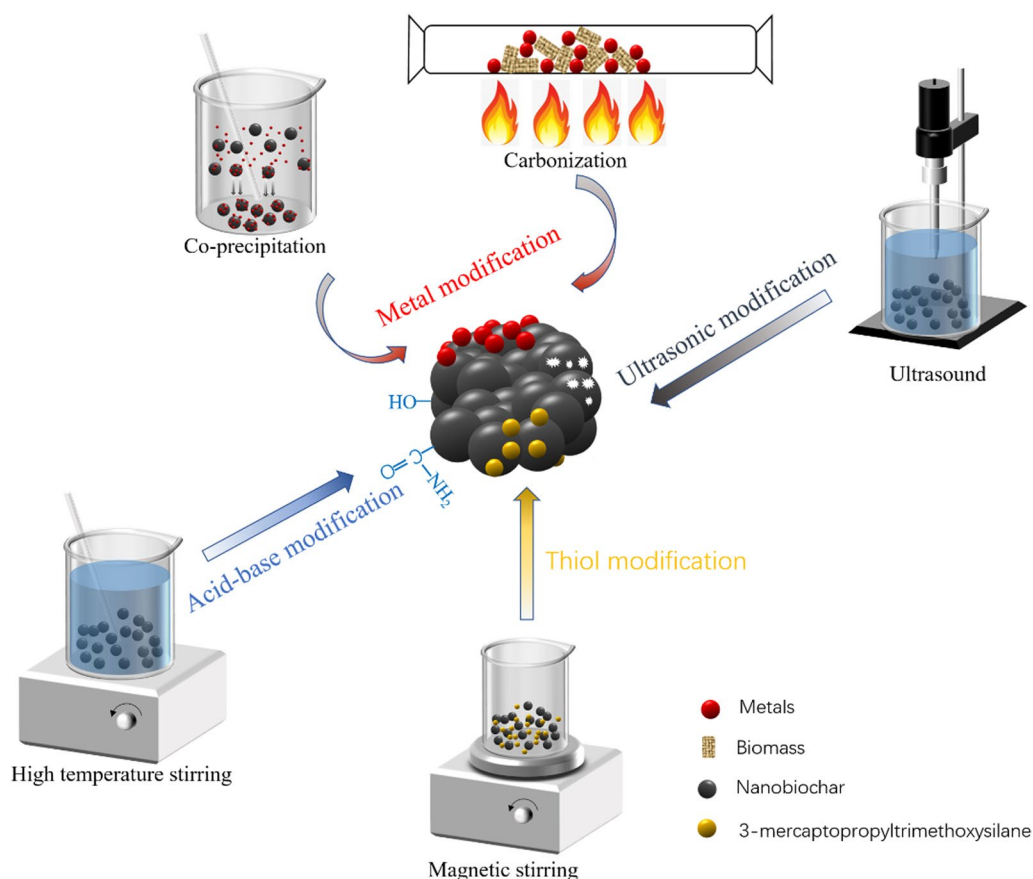
Fig. 3 Methods used for the preparation of modified nanobiochar

concentrations may lead to the occurrence of agglomeration effect especially under acidic conditions. The agglomeration of nanobiochar significantly reduces SSA and nanobiochar ability to immobilize pollutants (Mahmoud et al. 2019; Rasaki et al. 2019). As reported by Lyu et al. (2018a), Nickel (Ni) adsorption rate increased when soils were amended with low rate of nanobiochar but decreased with high rate of nanobiochar applied. In addition, hetero aggregation between nanobiochar and soil minerals may also frequently occur in soil system. This mainly occurs because that the positive charged soil minerals form electrostatic interactions with the negatively charged nanobiochar surfaces, thereby increasing the heteroaggregation (Zhang et al. 2022b). Therefore, there is an urgent need to understand and develop ways to reduce nanobiochar agglomeration and heteroaggregation in the environment to improve its ability to adsorb pollutants.

As mentioned above, studies have used modified nanobiochar to remediate some specific environmental pollutants. It has been reported that pristine nanobiochar had lower adsorption capacity for Chromium [Cr(VI)]

than modified biochar (Wang et al. 2020a). Heavy metals tend to be positively charged, hence nanobiochars with surface functional groups that are not easily protonated or nanobiochars with negative charges introduced on the surface may have better adsorption capability (Yi et al. 2015). Lyu et al. (2020) introduced thiol groups to the surface of nanobiochar using the ball milling method, which increased the surface negative charge and consequently the adsorption capacity of positively charged mercury (Hg) reached 320.1 mg g^{-1}. An iron oxide permeated mesoporous rice husk modified nanobiochar was demonstrated to remove up to 95% of Arsenic (As) in the environment (Nath et al. 2019). However, when the modified nanobiochar was applied in the environment, the modified materials (e.g. metals) may leach from its surface and cause harm to the environment. Therefore, the environmental risk of modified nanobiochar needs to be further studied.

3.2 Organic pollutants
Unlike heavy metals, organic pollutants can be degraded, and the application of nanobiochar can reduce the

Table 1 Effect of nanobiochar application on the mobility of heavy metals

Feedstocks	Heavy metals	Environment	Results	References
Pitch pine	Pb	Aqueous solution	The maximum adsorption capacity of the nanobiochar to Pb is 55.56 mg g^{-1}	Li et al. (2017b)
Bark chips	Cu, Pb, Zinc (Zn)	Soil	The adsorption capacity of the nanobiochar for Cu, Pb, and Zn was 121.5, 336, and 134.6 mg g^{-1}, respectively	Arabyarmohammadi et al. (2018)
Bagasse	Ni	Aqueous solution	The maximum adsorptive removal capacity of nanobiochar was 1949 mmol kg^{-1}	Lyu et al. (2018a)
Wood chips	Cu	Aqueous solution	The adsorption capacity of nanobiochar to Cu is 22 mg g^{-1}	Safari et al. (2019)
Rice-hull	Cd	Soil	Nanobiochar has high adsorption affinity for Cd which greatly reduces the uptake and phytotoxicity of Cd	Yue et al. (2019a)
Poplar wood	Hg	Aqueous solution	The removal efficiency of Hg was 320.1 mg g^{-1}	Lyu et al. (2020)
Sludge	Pb	Aqueous solution	The optimum Pb removal (99.87%) was at 0.5 g of nanobiochar, 5 mg L^{-1} initial concentration and 30 min contact time	Makshut et al. (2020)
Dendro	Cr, Cd	Aqueous solution	The maximum adsorptive removal capacities of dendro nanobiochar to Cd and Cr were 922.27 and 7.46 mg g^{-1}	Ramanayaka et al. (2020a)
Pine wood	Ni	Aqueous solution	The maximum Ni removal was 71%	Sajjadi et al. (2020)
Cornstalk	Cr	Aqueous solution	Removal rates of Cr are 49.6%, 65.8%, and 97.8%, respectively by Fe^0-nanobiochar composite consisting of biochar pyrolyzed at 300, 500 and 700 °C	Wang et al. (2020b)
Wheat straw	Cd	Soil	The application of nanobiochar reduced the available Cd in soil	Liu et al. (2020a)
Cynara scolymus leaves	Cd, Samarium (Sm)	Aqueous solution	The equilibrium uptake capacity values of Cd and Sm were established as 1150 and 650 µmol g^{-1}, respectively	Mahmoud et al. (2021)
Wood	Cd	Soil	The maximum sorption of Cd by nanobiochar reached 1062.4 mg kg^{-1}	Ramezanzadeh et al. (2021)
Cynodon dactylon residues	Cu, Pb	Aqueous solution	The nanobiochar showed the adsorption capacity of 220.4 mg g^{-1} for Cu and 185.4 mg g^{-1} for Pb	Vishnu et al. (2022)
Corn straw	Cd, Pb	Soil	The adsorption capacity of nanobiochar was 18.7 mg g^{-1} for Cd and 126.0 mg g^{-1} for Pb	Zhang et al. (2022a)

migration and bioavailable fraction of organic pollutants in both aquatic and soil systems through a series of mechanisms such as adsorption and catalytic degradation (see Table 2). Due to its special properties, nanobiochar offers excellent sorption capacity for many organic pollutants, such as herbicides, polychlorinated biphenyls (PCBs), and polycyclic aromatic hydrocarbons (PAHs) (Anupama and Khare 2021; Kumar et al. 2018). It has been reported that nanobiochar has a maximum adsorption capacity of 103.4 mg g^{-1} for acetone, which is 16-fold higher than the pristine biochar (Xiang et al. 2020). The development of nanobiochar composites and its modification methods also prove a way to enhance its adsorption capacity for pollutants. The adsorption capacity of the composites with 25% ball-milled nanobiochar (1210 mg g^{-1}) and 75% calcium-alginate for methylene blue was much greater than that of pure ball-milled nanobiochar (184 mg g^{-1}) (Wang et al. 2019). Xu et al. (2019) found that the adsorption capacity of N-doped nanobiochar on reactive red was 18.8 times higher than that of the bulk biochar and over 2 times higher than that of the common nanobiochar. This is due to the increase of the zeta potential and zero-point charge (PZC) of the biochar by N doping, which promotes the adsorption of anionic reactive red (Pereira et al. 2003). The basic N groups on N-doped nanobiochar can be protonated and positively charged, enabling the biochar to adsorb Congo red (a type of dye) through electrostatic adsorption (Tian et al. 2018).

Due to the low surface-active site density and low activation energy of adsorption bonds, nanobiochar can also be coupled with photocatalyst zinc oxide (ZnO) to degrade phenol indirectly (Chen et al. 2016). The addition

Table 2 Effect of nanobiochar application on the removal of organic pollutants

Feedstocks	Organic pollutants	Environment	Results	References
Forest wood	Phenanthrene, pyrene	Soil	The migration and cumulative release of phenanthrene and pyrene were significantly reduced by nanobiochar	Zand (2017)
Sugarcane bagasse	Methylene blue (MB)	Aqueous solution	The maximum MB removal capacity of 450 °C biochar after ball milling can reach 354 mg g^{-1}, which is 20 times higher than that of the pristine 450 °C biochar	Lyu et al. (2018b)
Bamboo	Methylene blue	Aqueous solution	74% MB was removed by nanobiochar	Wang et al. (2019)
Rice husk	Toluene	Aqueous solution	The adsorption capacity for toluene by nanobiochar enriched in silicon was 264 mg g^{-1}	Shen et al. (2019)
Hickory wood	Acetone	Aqueous solution	Nanobiochar exhibited faster adsorption than the pristine ones and the maximum adsorption capacity of acetone was 103.4 mg g^{-1}	Xiang et al. (2020)
Poplar wood	CH$_3$Hg$^+$	Aqueous solution	Modified nanobiochar has higher removal efficiency of CH$_3$Hg$^+$, 104.9 mg g^{-1}	Lyu et al. (2020)
Cellulose nanocrystals	Phenol	Aqueous solution	The degradation rate of nanobiochar to phenol was 99.8% within 90 min	Zhang et al. (2021)
Pine sawdust	MeHg	Aqueous solution	The maximum adsorption capacity of MeHg was 108.16 mg g^{-1}	Zhao et al. (2022)

of nanobiochar may improve the photoresponse and photocatalytic performance of the catalyst during the degradation process and the nanobiochar itself also adsorbs some phenol molecules. Zhang et al. (2021) showed that the degradation efficiency of the composite material to phenol within 90 min was 99.8% higher than that of pure ZnO. Other modification methods (see Sect. 2.2.3) will also strongly affect the properties of nanobiochar, thereby affecting its ability to remove organic pollutants. However, thus far, only few studies on the use of nanobiochar or modified nanobiochar to remove organic pollutants have been done, and therefore further studies are needed to improve our understanding of the role of nanobiochar in this field.

3.3 Emerging pollutants

Emerging pollutants include a range of organic compounds including pharmaceuticals and personal care products (PPCPs), detergents, pesticides, and industrial chemicals (Daughton and Ternes 1999; Matei et al. 2022; Norvill et al. 2016). They are highly polar, hydrophilic and resistant to biodegradation. Many emerging pollutants can enter the biosphere through groundwater and surface water (Gondi et al. 2022; Wanda et al. 2017). Some emerging pollutants may interfere with human enzymatic activities, hormones, or be mutagenic (Basheer 2018). In recent years, due to rapid industrialization, the global emission of emerging pollutants has increased to 10,000 tons per year (Gondi et al. 2022).

Table 3 summaries recent representative studies of nanobiochar application for the removal of emerging pollutants. Due to the abundant surface functional groups and rich pores, nanobiochar's high-efficiency removal ability for mainstream antibiotics in the environment has been demonstrated. Previous studies have shown that the removal rate of tetracycline (TC) in TC contaminated soil amended with ball milled nanobiochar was 96.36%, which was significantly higher than that amended with the bulk biochar (Sun et al. 2022). This result was attributed to the enhanced activity of soil microorganisms through the adsorption and degradation of TC by nanobiochar (Yue et al. 2019a). Moreover, active carbon and metal-organic frameworks (MOFs) were considered as super adsorbents for pollutants in wastewater, however, they are unable to completely remove persistent organic pollutants from wastewater (Hernández-Leal et al. 2011; Westerhoff et al. 2005). However, nanobiochar has exhibited a considerable adsorption capacity (609–2098 mg kg^{-1} for galaxolide) for persistent organic pollutants, which was much higher than active carbon (Zhang et al. 2019b). All these studies further indicated that nanobiochar can be used as a useful amendment for removal of antibiotics in the environment.

Increasing antibiotic exposure in the environment promotes the relative abundance of antibiotic resistance genes (ARG) (Pruden et al. 2006). Lian et al. (2020) found that nanobiochar could adsorb the extracellular DNA (eDNA) of ARGs and the adsorption capacity reached 296 mg g^{-1}. It is worth noting that the adsorbed eDNA is cleaved into short fragments or loses structural integrity over time. At the same time, the comparison found that the binding ability of nanobiochar to ARGs was 50–100 times higher than that of the bulk biochar. They also found that the nanobiochar not only adsorbed and damaged ARGs, but also inhibited their replication (Lian et al. 2020). In contrast, the bulk biochar–ARGs

Table 3 Effect of nanobiochar application on the removal of emerging pollutants

Feedstocks	Emerging pollutants	Environment	Results	References
Corn straw and rice husk	Diethyl phthalate (DEP)	Aqueous solution	The maximum adsorption capacity of DEP by corn straw nanobiochar and rice husk straw nanobiochar was 33.87 and 27.65 mg g^{-1}, respectively	Ma et al. (2019)
Rice husk and wheat straw	Galaxolide	Aqueous solution	The nanobiochar adsorbed 609–2098 mg kg^{-1} of galaxolide, which was about 3 times more than unmilled biochars	Zhang et al. (2019b)
Wheat straw	Tetracycline	Aqueous solution	The adsorption capacity of nanobiochar pyrolyzed at 700 °C for tetracycline was 268.3 mg g^{-1}	Li et al. (2020)
Rice straw	Antibiotic resistance genes (ARG)	Aqueous solution	The maximum adsorption capacity of nanobiochar for eDNA was 296 mg g^{-1}	Lian et al. (2020)
Artichoke leaves	Metformin hydrochloride anti-diabetic drug (MFH)	Aqueous solution	The implementation of modified nano-biochar for removal of MFH anti-diabetic drug (10 mg L^{-1}) from tap water, waste-water and sea water was affirmed by 87.0%, 97.0% and 92.0%, respectively	Mahmoud et al. (2020a)
Poplar woodchips	Enrofloxacin	Aqueous solution	Nanobiochar pyrolyzed at 300 °C exhibited the highest degradation rate (80.2%) and mineralization ability (66.4%) of enrofloxacin	Xiao et al. (2020)
Date-palm	Phosphate and nitrate	Aqueous solution	The maximum monolayer adsorption capacity of phosphate and nitrate by nanobiochar was 177.97 and 28.06 mg g^{-1}	Alagha et al. (2020)
Wheat straw and rice husk	Tetracycline	Soil	The removal rate of tetracycline by wheat straw and rice husk nanobiochar at 500 °C was 96% and 94.9%, respectively	Sun et al. (2022)

interaction only involves adsorption without the damaging and inhibition effects.

Ma et al. (2019) reported that the maximum adsorption capacity of diethyl phthalate by nanobiochar under pH 9 was much greater than that of micro-size biochar, while the adsorption capacity of nanobiochar and micron-scale biochar was insignificant at pH 3, which indicates that the corn stover nanobiochar has high activity under alkaline conditions. Studies have shown that biochar can prevent leaching and increase the dissipation of these biopesticides thereby reducing their potential environmental risk (Anupama and Khare 2021; Gámiz et al. 2016). In terms of pollutant adsorption and degradation, biochar and nanobiochar share similar mechanisms which are discussed in detail below. Shen et al. (2020) found that nanobiochar can weaken the *Imperata cylindrica* allelopathic effect on rice seedlings. Together these studies demonstrated that nanobiochar is an effective absorbent for controlling the release of allelopathic compounds.

According to the above discussion, nanobiochar can significantly stabilize or remove toxic substances from soil, which may offer a healthy environment for plant growth. In addition, nanobiochar combines the pristine biochar and nanomaterial properties to promote plant growth (Garg et al. 2022; Rajput et al. 2022; Shen et al. 2020). Therefore, during phytoremediation process, nanobiochar application to contaminated soil will also further improve the phytoremediation effectiveness.

4 Adsorption mechanism of pollutants using nanobiochar

As a powerful pollutant adsorbent, the adsorption mechanism of nanobiochar is complex and diverse, including ion exchange, complexation, precipitation, electrostatic interaction, and physical adsorption (Zhang et al. 2022b). In the adsorption process, the above reactions or their combined reactions may be involved (Mahmoud et al. 2021). The adsorption mechanisms, however, varied with the properties of nanobiochar, environmental conditions, and the type of adsorbed pollutants (Antonangelo et al. 2019). For example, studies showed that the adsorption of Ni on carbonaceous materials mainly involves electrostatic forces, cationic-π interactions, and surface complexed, while the adsorption mechanism of glyphosate on nanobiochar may not include chemisorption (Meena et al. 2005; Ramanayaka et al. 2020a; Sato et al. 2007). The increase of acidic functional groups will reduce the surface potential and PZC of nanobiochar, thereby

enhancing the electrostatic effect of adsorption (Lyu et al. 2018a). Moreover, high temperature pyrolysis may result in the increase of the SSA and void structure richness of biochar and a decrease in polar surface functional groups, consequently the dominant adsorption mechanism tends to be physical adsorption. On the contrary, the SSA of low-temperature nanobiochar is relatively low and the polar surface functional groups are relatively more, and the dominant adsorption mechanism is biased towards chemical adsorption (Zhuang et al. 2021). Therefore, to explore the nanobiochar with good removal ability for heavy metals and organic pollutants, it is necessary to study the mechanisms that mainly control the adsorption.

Fan et al. (2020) concluded that the main mechanism for the adsorption of heavy metals by nanobiochar was attributed to the interaction between the negatively charge surface carrying oxygen-containing functional groups as electron donors and positively charged metals. Wang et al. (2020b) showed that the complexation of the positively charged $-OH$ groups of nanobiochar with the negatively charged $Cr(VI)$ in aqueous systems was the dominant mechanism involved in the adsorption process. Ramanayaka et al. (2020a) pointed out that no physical adsorption was found in the adsorption of $Cr(VI)$ by nanobiochar, and the surface $-OH$, $-COOH$, $-C=O$ and other oxygen-containing functional groups provide electrons for the reduction of $Cr(VI)$, which was confirmed by Zhang et al. (2013a, b). Cations (Ca^{2+} and Mg^{2+}) were observed to be released from biochar into solution during adsorption process, which suggests that cation exchange is another important adsorption mechanism during $Cr(III)$-biochar reaction (Chen et al. 2015). For anions, nanobiochar showed different adsorption mechanisms. Niazi et al. (2018a) reported that the functional groups were involved in As sequestration by biochar. After As loading, the shift peaks of functional groups consistent with hydroxy, carboxyl, and $C-O$ ester of alcohols were observed in Fourier Transform Infrared Spectroscopy (FTIR) spectra (Samsuri et al. 2013), suggesting the complexation of As with functional groups. In addition to complexation, electrostatic interaction may be another mechanism which largely controls the As adsorption (Li et al. 2017a). This is evidenced by Wang et al. (2016b) who found the surface functional groups with higher degree of protonation at low pH attract As oxyanion through electrostatic interaction. Lyu et al. (2018a) found ball milling increased the internal and external surface area and exposed the graphene structure on the surface of nanobiochar, which enhanced Ni adsorption via strong cation-π interactions. Furthermore, electrostatic interaction and surface complexation were also observed to be the main mechanisms of Ni-nanobiochar reaction.

Different from heavy metals, the removal of organic pollutants by nanobiochar also includes promoting the pollutant degradation (Khan et al. 2022). Xiao et al. (2020) showed that the ball-milled nanobiochar exposed to visible light produced more oxygen-containing functional groups and exhibited a high degradation rate to enrofloxacin. Sun et al. (2022) showed that the adsorption of TC in soil by ball-milled nanobiochar is mainly based on surface adsorption, π–π interactions, hydrophobic interactions, and hydrogen bonding. Tetracycline adsorbed on nanobiochar surface was degraded by microorganisms whose activity were enhanced by nanobiochar (Yue et al. 2019b). It is likely that the removal process was adsorption at initial period and followed by degradation. Another study also reported that the adsorption mechanism of TC in wastewater on magnetic ball-milled nanobiochar was mainly based on the combined effect of electrostatic interaction, hydrogen bonding and $C\pi$–$C\pi$ interaction (Li et al. 2020). Hydrogen bonding may be the key mechanism for TC adsorption because of the TC aromaticity, whose molecular structure contains several components that can act as H^+ acceptors (e.g. $-N(CH_3)_2$ and $-C=O$) or both as H^+ acceptors or donors (e.g. $-OH$, $-CONH_2$, and $-NH_2$). These components can form hydrogen bonds with the surface components of biochar (Jing et al. 2014; Yang et al. 2011). Electrostatic interactions are the most common adsorption mechanism during the binding of nanobiochar to organic pollutants (Abbas et al. 2018; Zheng et al. 2013). The solution pH largely determines the strength of the electrostatic force and whether it is attractive or repulsive (see Sect. 5.3). With the pH value ranging from 1.8 to 9.8, the adsorption capacity of nanobiochar for methyl blue ranged from 216 to 351 mg g^{-1}, which indicates that the electrostatic interaction mechanism plays an important role in the adsorption process (Lyu et al. 2018b). Compared to bulk biochar, nanobiochar has more graphene structure and larger pore volume and pore size. Therefore, pore filling and π–π interactions should occur more frequently during the reaction. Zhang et al. (2019b) found that pore filling and π–π interactions enhanced the adsorption of galaxolide by high-temperature ball-milled nanobiochar. The adsorption mechanism of nanobiochar on pollutants is ultimately determined by its properties, which is discussed in detail below.

5 Factors affecting performance of nanobiochar for environmental remediation

The biogeochemical behavior of nanobiochar during the removal or stabilization of pollutants is affected by various factors, such as the physical and chemical properties

of nanobiochar, characteristics of pollutants and environmental conditions. These factors also have different degrees of influence on nanobiochar's biogeochemical behavior in soil and water environments. For example, when environmental factors (e.g. acid rain) attempt to alter the soil and water pH, the soil pH was more affected than water system due to due to the strong buffering capacity of the soil. In addition, the types of microorganisms in soil and water also vary, which may lead to greater variability in nanobiochar biogeochemical behavior in both systems.

5.1 Properties of nanobiochar

It has been widely accepted that the pyrolysis temperature can significantly affect nanobiochar properties. During pyrolysis, the release of volatile gases leads to an increase in porosity and therefore increases the SSA of biochar/nanobiochar (Niazi et al. 2018b; Swaren et al. 2022; Weber and Quicker 2018). Most nanobiochars have large SSA ranging from 5.58 to 1736 m^2 g^{-1} (Table 4) due to the small particle size. Studies have shown that the SSA of nanobiochar could increase with the pyrolysis temperature (see Table 4 and references therein). However, there are also conflicting opinions which suggest that the increase of pyrolysis temperature had no effect or negatively effect on nanobiochar SSA (Table 4). The reason for these differences may be due to the difference in the raw parent materials of the nanobiochar.

Chen et al. (2011) reported that nanobiochar made from high hemicellulose biomass had a larger surface area than that from high lignin content material. This may be due to the high recalcitrance of lignin pyrolysis below 1000 °C (Neeli and Ramsurn 2018). Under the same preparation conditions, the carbon content of barley grass nanobiochar was only 5.02%, however, the pig manure nanobiochar reached 44.9% (Song et al. 2019). As for the ash content, corn straw nanobiochar had a much lower ash content than that of rice straw nanobiochar, which was mainly due to the rice straw containing more silicon (Si) than corn straw (Ma et al. 2019).

In addition to the raw material, the ash content was usually related to the pyrolysis temperature (Anupama et al. 2021; Nath et al. 2019). As the pyrolysis temperature increases, the base cation and carbonate content of the biochar increases thus leading to an increase in total ash content (Ramanayaka et al. 2020a). For instance, the ash content of 800 °C sugarcane bagasse biochar was 2–3 times higher than low temperature biochar (Hass and Lima 2018). In addition, the ash content of ultrasonic nanobiochar was significantly lower than that of pristine biochar by about 50% because of the separation or dissolution of minerals by the ultrasonic process (Liu et al. 2018a). Similar to the ash content, the carbon content

also showed a strong correlation with temperature and raw material (Zhang et al. 2022c). Under high temperature pyrolysis conditions, the C–H and C–O bonds are broken leading to the release of hydrogen and oxygen and an increase in carbon content (Jindo et al. 2014; Mahinpey et al. 2009; Xu et al. 2020). At low temperatures (< 300 °C), the raw material is difficult to be carbonized and thus retains its original properties, and the carbon content of the biochar is lower (Keiluweit et al. 2010). The effect of raw materials on the carbon content of nanobiochar has been summarized in Table 4. At the molecular scale, the main component of nanobiochar carbon was mainly the aromatic structure (Ma et al. 2019). At higher pyrolysis temperatures (800–900 °C), smaller aromatic rings form larger and more stable aromatic ring structures that were more difficult to depolymerize (Feng et al. 2017). The aromatic clusters were distributed on different graphene-like aggregates (Xiao et al. 2017). Graphene-like multilayer structures have been observed by transmission electron microscopy (TEM) (Ma et al. 2019; Oleszczuk et al. 2016), where these nanosheets are randomly arranged in the turbo-layered state (Xiao et al. 2016). In addition, amorphous carbon was present on the surface of the nanobiochar (Zhao et al. 2022), where intrinsic minerals (mainly the alkali and alkaline earth metals) can act as catalysts (He et al. 2021). The composition and structure of nanobiochar have a great impact on its ability to adsorb pollutants. For example, nanobiochar with higher concentration of cations may enhance the ion exchange ability with heavy metals, thereby stabilizing more metal ions. A more aromatic structure nanobiochar may lead to more π–π interactions between nanobiochar and pollutants (mainly for organic pollutants). Therefore, when nanobiochar is used as an amendment for environmental remediation, its composition and structure should also be taken into consideration.

The aggregation ability, colloidal stability and the electro kinetic properties of nanobiochar may affect the adsorption capacity of nanobiochar on pollutants, which can be assessed by measuring the zeta potential (Filipinas et al. 2021; Safari et al. 2019). Generally, the zeta potential of nanobiochar is lower than that of microbiochar, which is due to the smaller particle size, lower carbon content and higher ash content of nanobiochar (Song et al. 2019). The zeta potential may vary due to differences in chemical composition of the parent material (such as –COOH) (Qian et al. 2016). Song et al. (2019) observed that nanobiochar produced from wood, forage and agricultural waste had a much higher zeta potential than that produced from manure and activated sludge. Surface functional groups of nanobiochar also vary with pyrolysis temperature and parent material. As the temperature increases, the volatile

Table 4 Physicochemical properties of nanobiochar

Feedstocks	Pyrolysis temperature (°C)	Properties					Methods	References
		pH	Surface area (m² g⁻¹)	Ash content (%)	Carbon content (%)	Particle size (nm)		
Rice husk	450	–	–	–	–	45	Milling	Richard et al. (2016)
Pine white wood	525 ± 1	6.61 ± 0.35	47.25	2.0 ± 0.1	83.1 ± 2.5	60 ± 20		Naghdi et al. (2017b)
Bagasse	300	6.4	10.8	–	–	242		Lyu et al. (2018a)
	450	6.8	331	–	–	170		
	600	6.9	364	–	–	140		
Bamboo	300	7.1	8.3	–	–	250		
	450	7.9	299	–	–	165		
	600	8.4	276	–	–	140		
Hickory wood chips	300	6.2	5.6	–	–	333		
	450	7.2	309	–	–	158		
	600	7.8	270	–	–	133		
Poplar wood	500	–	–	–	68.63	10–1000		Liu et al. (2019)
Corn straw	500	7.9 ± 0.2	364	6.27 ± 0.08	77.62 ± 0.28	100–600		Ma et al. (2019)
Rice straw	500	8.7 ± 0.1	298	31.51 ± 0.08	53.05 ± 0.25	100–600		
Bamboo	450	–	298.6	–	–	52.5–159.1		Wang et al. (2019)
Wheat straw	500	–	289	–	62.7	< 100		Zhang et al. (2019b)
Crude biochar from gasification	–	–	9.19	63.1	58.4	70–80		Goswami et al. (2020)
Waste lignin	800	–	83.41	–	84.44	473.6 ± 81.9		Jiang et al. (2020)
Wheat straw	700	–	296.3	–	47.98	< 250		Yin et al. (2020)
Fruit tree branch	350	–	26.06	–	27.75 ± 0.16	44–119		Xu et al. (2020)
	450	–	10.72	–	27.72 ± 0.11	44–120		
	550	–	30.46	–	77.32 ± 0.21	41–112		
Corn straw	350	–	5.58	–	11.65 ± 0.02	51–87		
Peanut straw	350	–	69.6	–	29.67 ± 0.08	60–103		
Poplar wood-chips	300	–	–	–	–	< 100		Xiao et al. (2020)
	500	–	–	–	–	< 100		
	700	–	–	–	–	< 100		
Coconut shell	–	9.6	–	–	–	40		Yang et al. (2020b)
Poplar wood chips	300	–	61.34	–	66.53 ± 1.59	< 1000		Lyu et al. (2020)
Cornstalk	700	–	251.938	–	–	100		Wang et al. (2020b)
Sawdust	600	–	360	2.82 ± 0.13	85.6 ± 4.2	76–230		Yuan et al. (2020)
	600	–	334	16.7 ± 0.14	71.9 ± 3.6	153–307		
Artichoke leaves	350	–	–	–	–	23.54–27.8		Mahmoud et al. (2021)
Sugarcane waste	800	–	–	–	–	150–300		Yao et al. (2021)
Wheat straw	350	6.8	–	–	–	< 50	Sonicator/ Centrifugation	Wang et al. (2013)
	550	6.7	–	–	–	< 110		

Table 4 (continued)

Feedstocks	Pyrolysis temperature (°C)	Properties					Methods	References
		pH	Surface area ($m^2\,g^{-1}$)	Ash content (%)	Carbon content (%)	Particle size (nm)		
Pine needle	350	6.7	–	–	–	< 150		
	550	6.8	–	–	–	< 220		
Wheat straw	350–700	9.1 ± 0.2	29.56 ± 1.1	52.35	27.88 ± 0.09	600		Oleszczuk et al. (2016)
Wicker	350–700	8.9 ± 0.3	18.25 ± 1.5	30.97	51.57 ± 1.7	210		
Miscanthus	350–700	7.8 ± 0.1	36.39 ± 2.9	24.74	56.16 ± 2.7	110		Li et al. (2017b)
Pitch pine	600	–	–	–	79.28	180–400		
Peanut shell	300	6.40 ± 0.04	63.6	6.89 ± 1.14	50.2	17.4 ± 4.2		Liu et al. (2018a)
	400	6.80 ± 0.03	78.6	7.68 ± 0.46	54.8	–		
	500	7.08 ± 0.04	230	8.93 ± 0.34	61.1	–		
	600	7.27 ± 0.03	264	9.16 ± 0.94	61.4	25.3 ± 11.9		
Tobacco stems	500	9.79 ± 0.1	21.108	–	–	42 ± 6		Dong et al. (2019)
Pine wood	500	9.4	–	32.6	44.9	< 100		Song et al. (2019)
Wood chip	500	8.9	–	43.4	17.7	< 100		
Barley grass	500	9.17	–	67.3	5.02	< 100		
Wheat straw	500	9.69	–	65.5	6.57	< 100		
Peanut shell	500	9.82	–	61.3	13.1	< 100		
Rice husk	500	10.2	–	68.2	16	< 100		
Dairy manure	500	9.76	–	56.6	6.58	< 100		
Pig manure	500	9.22	–	68.5	4.21	< 100		
Sewage sludge	500	8.43	–	60.8	11.2	< 100		
Rice hull	300	–	21.7	–	71.34	190		Yue et al. (2019a)
	400	–	80.1	–	71.19	106		
	500	–	90.9	–	81.25	92		
	600	–	123.2	–	80.87	59		
Rice straw	400	–	93.18	–	55	280		Lian et al. (2020)
	700	–	253.9	–	54.8	220		
Sago activated sludge	500	–	78.863	99.9	0.08	45–75	Twice carbonization	Makshut et al. (2020)

organic compounds in the material are devolatilize, resulting in a surface functional group reduction (Alam et al. 2018). Hence, low-temperature pyrolysis nanobiochar was reported to have an abundance of surface functional groups, high absolute zeta potential and strong suspension stability (Xu et al. 2020).

5.2 Characteristics of pollutants affecting sorption by nanobiochar

The adsorption behaviour of nanobiochar for different pollutants may varies greatly due to the large differences in the properties of the pollutants. As for heavy metals, the valence, hydration radius, electronegativity and hydrolysis constant are the main factors that govern the remediation process of nanobiochar (Fig. 4). Zhang et al. (2022a) found that under the same conditions, the adsorption capacity of nanobiochar for Pb was much greater than that for Cd. This could be attributed to the property difference of metals and their affinity for adsorption sites. The hydration radius of Cd (4.26 Å) is larger than that of Pb (4.01 Å) and the pKH (negative log of hydrolysis constant) of Cd (10.1) is larger than that of Pb (7.71). These metallic properties indicate that Pb can be better adsorbed through internal surface complexation or adsorption reactions than Cd. In addition, the electronegativity of Pb (2.33) is greater than that of Cd (1.69), which leads to a better binding tendency of Pb to hard Lewis base surface functional groups (Ni et al. 2019).

The characteristics of organic pollutants such as polarity, hydrophobicity, aromaticity, and molecular size are important factors affecting their interaction with nanobiochar (Fig. 4). Since acetone polarity was higher than that of toluene, the adsorption capacity of nanobiochar
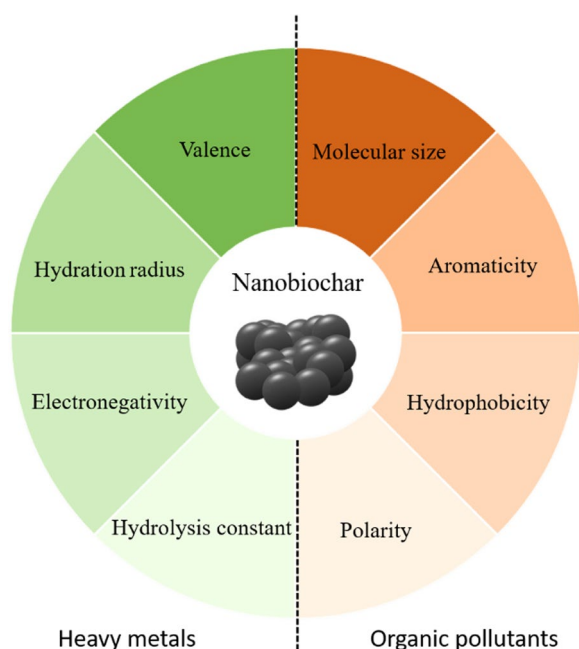
Fig. 4 Effects of heavy metal and organic pollutant chemical characteristics on their adsorption by nanobiochar

(2018) showed that reducing biochar particle size can significantly improve the adsorption rate of hydrophobic organics by biochar. Galaxolide is a hydrophobic organic pollutant and thus can be better adsorbed by ball milled nanobiochar (Zhang et al. 2019b). In addition, the hydrophobic interaction between the polar groups of TC and the nanobiochar enhanced its adsorption (Sun et al. 2022). Aromatic organic compounds can form π–π interactions with biochar and be tightly adsorbed (Teixidó et al. 2011; Zhu et al. 2005). Zeng et al. (2019) reported that TC was adsorbed through the π–π stacking interactions between the four aromatic rings in its molecular structure and the aromatic carbon matrix of nanobiochar. Pollutants with different molecular sizes have different effective contact and interception effects on biochar/nanobiochar, subsequently the adsorption mechanism and adsorption effect are also different (Nguyen et al. 2007). Larger compound molecules are hardly absorbed by biochar due to the size exclusion effect and pore filling mechanism, which lead to organic molecules being largely restricted from entering pores smaller than their size (Han et al. 2014; Zhu et al. 2022). This effect is likely to be amplified on nanobiochar due to its generally small pore diameter.

5.3 Environmental conditions

Environmental factors that strongly affect the performance of nanobiochar in environmental remediation mainly include pH, coexisting ions, dissolved organic
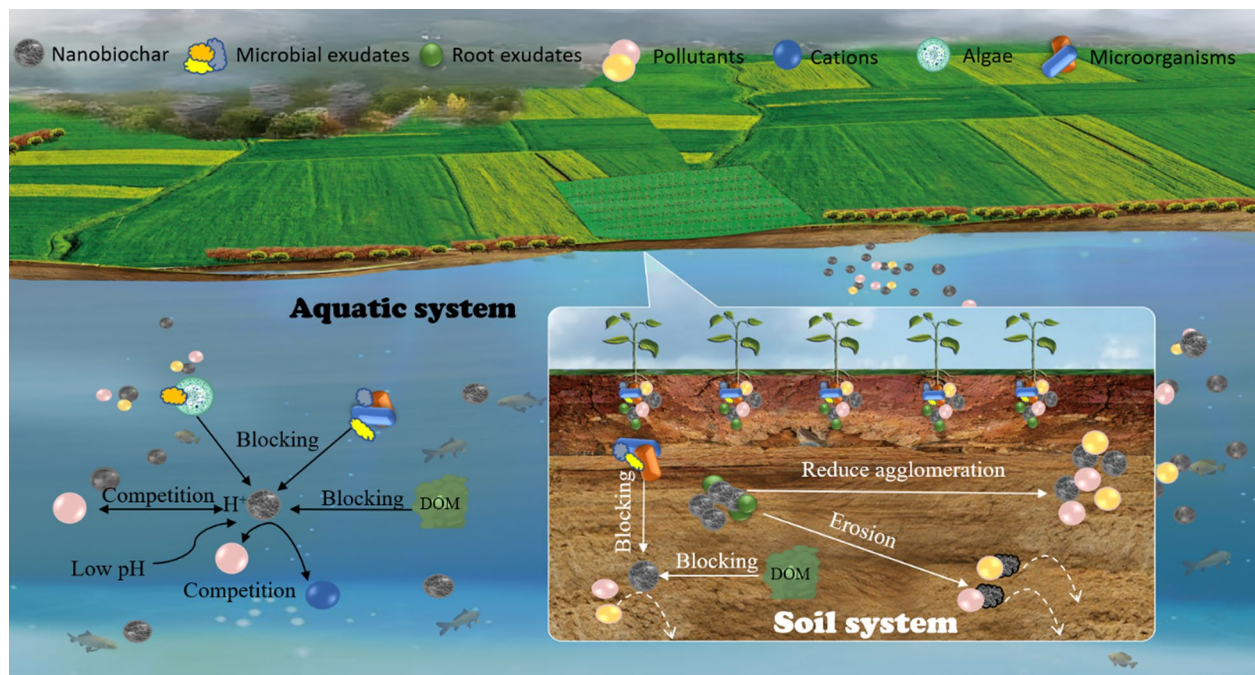
for acetone was greatly improved compared to that of toluene (Zhuang et al. 2021). Generally, the adsorption rate of strongly hydrophobic organics by carbonaceous adsorbents is slow (Choi et al. 2013, 2014). Kang et al.

Fig. 5 Effects of environmental factors on biogeochemical behavior of nanobiochar for remediation of environmental pollution

matter (DOM), organisms and root exudates (soil system) (see details in Fig. 5). Environment pH strongly governs the biogeochemical behavior of nanobiochar for environmental remediation (Gaya et al. 2015). Under low pH conditions, the functional groups of nanobiochar are easily protonated to form H^+, which will cause H^+ and cation pollutant species to compete with each other for cation adsorption sites thus reducing the adsorption capacity of nanobiochar (Mahmoud et al. 2020b; Park et al. 2019). In addition, positively charged surfaces may hinder the adsorption of pollutants due to electrostatic repulsion (Lyu et al. 2018a). Studies have found that the adsorption capacity of Cd by nanobiochar increased with the pH value (ranging from 1 to 7) (Mahmoud et al. 2021; Ramanayaka et al. 2020a). This may be due to the protonation of the nanobiochar surface during the acidic adsorption process, which enables Cd to outcompete H^+ for adsorption sites. However, the negatively charged nanobiochar surface also showed lower affinity for negatively charged or neutral pollutants due to electrostatic repulsion at high pH (Zeng et al. 2019). Previous studies have also shown similar results where the adsorption capacity of nanobiochar increased as pH increased from 2 to 8 and decreased slowly when the pH was greater than 8 (Li et al. 2020; Sun et al. 2022). In summary, it is essential to study pH changes in the environment when using nanobiochar for environmental remediation.

Determination of the PZC is important to accurately investigate how pH affects the adsorption capacity of nanobiochar (Mahmoud et al. 2021). PZC usually occurs when the nanobiochar surface is neutral, and under $pH < pH_{PZC}$ conditions, the nanobiochar surface shows a positive charge, otherwise it is negatively charged (Choudhary et al. 2020). Changes in the adsorbate solution pH can affect not only the surface chemistry of the adsorbent, but also changes in the adsorbate species (Li et al. 2010). For example, Ni^{2+} is the dominant form when the solution pH range is < 9, while the dominant form is $Ni(OH)_2$ (aq) when the pH ranges from 9 to 11, and $Ni(OH)_3$ for pH 12–14 (Lyu et al. 2018a). As for the metal modified nanobiochar, a change in environment pH may affect the properties of nanobiochar itself, thereby affecting its pollutants adsorption abilities. Wang et al. (2020a, b, c) showed that the surface of Fe^0-nanobiochar contained soluble iron oxides at pH lower than PZC under acidic conditions, thus providing more opportunities for electron transfer. Li et al. (2022) found that the low pH condition resulted in structural instability and partial dissolution of the MgAl-layered double hydroxide (LDH) phase of the biochar supported layered double hydroxide composites, which weakened the adsorption capacity of phosphate by the layered double hydroxides.

Previous studies have shown that coexist of multiple ions may affect nanobiochar's behavior for pollutant removal (Yadav et al. 2017; Zhang et al. 2020). Independent adsorption, cooperative adsorption and competitive adsorption are three adsorption relationships which are suitable for substances that coexist in the water environment (Wang et al. 2016a). Wang et al. (2017) showed that the addition of Pb could promote the adsorption of *p*-nitrophenol on biochar. Likewise, the presence of As^{3+} in the aqueous solution increased the adsorption of Cd on the magnetized biochar from 3% to 16% (Wu et al. 2018). The cations and anions in water system may have a much stronger effect on nanobiochar properties than in soil system, which occurs because they can migrate faster in water and are easier to contact nanobiochar. There are large number of organisms in the environment which may produce secretions or excretions, like extracellular polymers by *Escherichia coli* cells and metabolites by algae, and this may block the pores of the adsorbent surface and reduce adsorption sites especially for nanobiochar due to its small pore and particle size (Kenney and Rosenzweig 2018; Zhang et al. 2020).

DOM is abundant in the environment and plays a critical role in the removal and migration of pollutants (Fig. 5, also see details in Ling et al. 2015). As an important fraction, the presence of humic acid will reduce the adsorption of sulfamethazine by biochar (Jia et al. 2018). Yang et al. (2020a) found that DOM would suppress the Bisphenol S adsorption capacity on biochar (should be the same effect on nanobiochar). More importantly, DOM itself can bind with metals or combine with organic pollutants, thereby reducing their bioavailability. It is essential to investigate how different fractions of DOM affect nanobiochar adsorption capacity for pollutants.

In soil systems, specifically, the nanobiochar pores can promote bacteria and fungi coexistence and offer protection from pollutants (Quilliam et al. 2013). The nutrient components of nanobiochar can be absorbed by soil microorganisms or used as carbon sources, thereby enhancing the activity of soil microorganisms, and promoting pollutant degradation (Ameloot et al. 2015). Nanobiochar can affect soil properties such as reducing soil bulk density, increasing soil effective water content and carbon sequestration capacity, which consequently affect microbial community structure (Rajput el al. 2022; Yu et al. 2015). It is important to note that that effect of nanobiochar on soil microorganisms is not unidirectional. Since many microorganisms in the rhizosphere soil produce secondary metabolites such as nonribosomal peptides and terpenes (Tyc et al. 2017) that can easily agglomerate or heterogeneously aggregate nanobiochars, leading to a decrease in SSA. This in turn significantly reduces nanobiochar's pollutant adsorption rate. For soil

fauna, earthworms are the most studied organisms in the interaction between soil animals and biochar (Lehmann et al. 2011). The glandular cells of earthworms are present in the posterior epithelium and release mucus to moisten the epithelium (Wang et al. 2011). Nanobiochar contacting with mucus is likely to lose its adsorption sites due to pore plugging . However, the research on the effects of other soil fauna behavioral processes on the environmental function of nanobiochar is lacked.

In soil systems, plant roots are ubiquitous and release root exudates such as organic acids, amino acids and other simple sugars during plant growth (Ling et al. 2009; Toyama et al. 2011). Root exudates can promote the biodegradation of organic pollutants and control redox sensitive pollutant bioavailability (Martin et al. 2014; Sun et al. 2013). The root exudate components can affect nanobiochar physicochemical properties, which may further influence its adsorption capacity of pollutants. Li et al. (2019) reported that oxalic acid would undergo acid oxidation with biochar to promote biochar polarity and reduce its aromaticity, which directly led to the enhanced adsorption of phenanthrene and the weakened pyrene adsorption by biochar. Interestingly, the high oxalic acid concentration dissolved the ash content and increased the number of micropores in biochar to increase its SSA, and consequently promoted phenanthrene adsorption (Zhang et al. 2013a, b). In addition, root exudates may also diffuse into the pores of nanobiochar to destroy the pore structure, resulting in the release of pollutants adsorbed into the environment (Joseph et al. 2010).

Usually, natural aging process may also strongly affect the adsorption capacity of biochar when using for pollutant removal. This process can partially crack bulk biochar into nanobiochar. Wang et al. (2020c) reported that aging will lead to nanobiochar erosion, oxidation, and disagglomeration. Some studies showed that biochar aging in soil environment will generate more oxygen-containing surface functional groups, which is due to the chemical aging by soil minerals (Chang et al. 2019; Ren et al. 2018). This enhances biochar/nanobiochar ability to immobilize heavy metals. However, Zhang et al. (2016) found that aging process reduced the adsorption capacity of biochar to pollutants due to dissolved organic matter blocking the pores of biochar. Therefore, how aging process affects the performance of nanobiochar on the removal/stabilize pollutants in the environment still need to be further studied.

6 Conclusions and outlooks

Although numerous studies have focused on how to use biochar in environmental remediation, there are relatively few studies on nanobiochar. By summarizing the literatures in this review, we found that nanobiochar has much greater potential for remediation of environmental pollution than biochar, which can be attributed to its special physical and chemical properties and higher mobility in the environment as nanoparticles. However, it is not clear whether nanobiochar will produce nanotoxicity or display negative effects when it enters the environment. Therefore, future studies should be conducted to fill the knowledge gaps which are list as below:

- As discussed in Sect. 2, although various methods can be used for nanobiochar preparation, their yields were still relatively low and this greatly limits the development of related research on nanobiochar. Therefore, further studies are needed to optimize nanobiochar production methods to improve its yield.
- In the remediation process, the biogeochemical behavior of nanobiochar is still unclear, especially the environmental risks. Therefore, studying the biogeochemical behavior of nanobiochar and its environmental risks will provide a guarantee for the safe use of nanobiochar in environmental remediation.
- When nanobiochar is added to the environment, it can stabilize a large amount of pollutants. This will greatly reduce pollutant transport and bioavailability and consequently diminishing their threat to ecosystems and human health. However, it is unknown whether the immobilized pollutants will be re-released into the environment. The issue of how to recover the biochar used for environmental remediation from the environment has not been solved. As nano-level particles, nanobiochar is even more difficult to recycle. Therefore, nanobiochar recovery technology and its environmental risks after adsorption of pollutants need to be further studied.

Author contributions
MJ: conceptualization, data curation, investigation, visualization, writing—original draft. LH: writing—review and editing. NKN: writing—review and editing. HW: writing—review and editing. WG: writing—review and editing. MV: writing—review and editing. KG: writing—review and editing. HS: writing—review and editing. XZ: conceptualization, supervision, writing—review and editing. ZW: writing—review and editing. All authors read and approved the final manuscript.

Availability of data and materials
The datasets generated during and/or analyzed during the current study are available from the corresponding author on reasonable request.

Declarations

Competing interests
The authors have no conflicts of interest to disclose, financial or otherwise.

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