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Biochar

REVIEW **Open Access**

Engineered biochar composite with minerals: organo-mineral interactions, physicochemical changes, and implications for practical application

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Abstract

Engineered biochar materials are under development for a wide range of environmental applications. Many of these engineered materials combine biochar with minerals such as non-clay silicates, clay minerals, oxide minerals, and carbonates, or are manufactured through the co-pyrolysis of mineral additives with the biomass, both forming organo-mineral complexes. In this review, we provide an exploration of the mechanisms of organo-mineral interactions, physicochemical changes, and real-world practical applications. We first provide an overview of organo-mineral interactions between biochar and minerals in the natural environment to offer insights into organo-mineral interactions in engineered biochar composites. Secondly, we propose a classification of biochar composites with minerals. A quantitative analysis of physicochemical changes in engineered biochar composites is presented, revealing an increase in ash content and polarity, with either improved or degraded porous structure. Based on these physicochemical changes, enhancement mechanisms of the mineral components are assessed. These mechanisms primarily involve direct stabilization of biochar carbon (C) and indirect negative priming when applied to soil, nutrient delivery, introduction of functional groups for adsorption/immobilization, and reduced toxicity to support microbial colonization and improve soil health. In addition, evidence for the practical application of biochar-mineral composites is presented, including field studies for soil applications and pilot-scale non-soil applications such as in wastewater and stormwater treatment. Finally, challenges and future research directions are proposed, including examining the molecular binding mechanisms between minerals and the C matrix, investigating the reversibility of mineral attachment and the long-term effectiveness of composites, and exploring emerging non-soil applications of novel biochar composites with minerals.

Highlights

- Organo-mineral interactions between biochar and minerals in the natural environment serve as analogues of engineered composites.
- Types and physicochemical changes of biochar composites with minerals were summarized.

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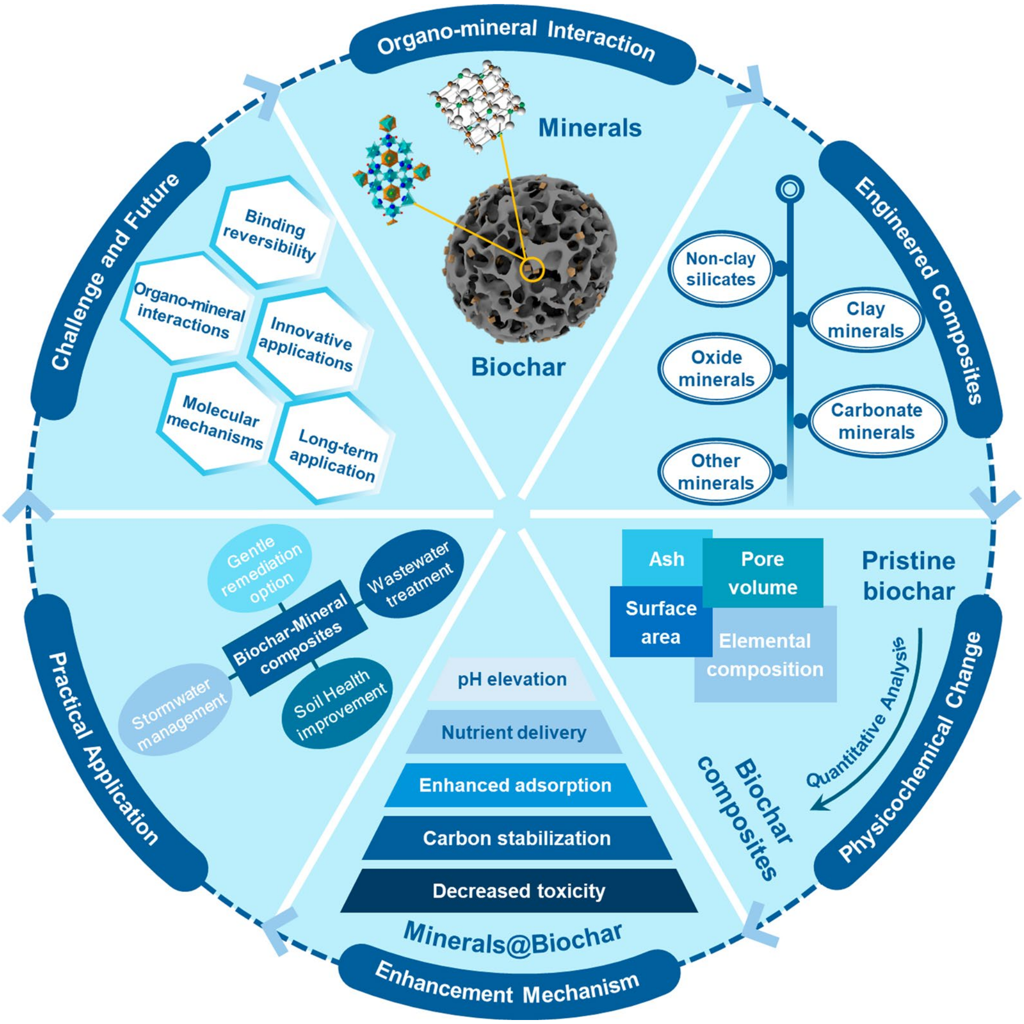
- Real-world practical applications of biochar-mineral composites were reviewed

Keywords Biochar, Composite, Mineral, Mechanism, Environmental application

Graphical Abstract



1 Introduction

Biochar is a carbonaceous material produced through the thermal conversion of biomass under oxygen (O_2)-limited conditions (IBI 2015). Biochar is an "ancient" material, and its early use dates back to Pre-Columbian Amazonians, who amended soils with pyrolyzed biomass material to produce Amazonian Dark Earth (*Terra Preta*), thus improving soil fertility (Glaser and Birk 2012; Lehmann et al. 2003). Sharing similar physicochemical properties with naturally formed pyrogenic carbon (C) (e.g., char from wildfires) (Bird et al. 2015; Knicker 2011; Schmidt et al. 2019), this anthropogenic amendment has gained renewed interest across various sectors. To date, the potential of biochar has been widely explored in areas such as agriculture (Bo et al. 2023; Lehmann and Joseph 2024), environmental remediation (Anae et al. 2021; Nguyen et al. 2023), construction and infrastructure (Murali and Wong 2024; Senadheera et al. 2024), aquaculture (Rai et al. 2025; Wong et al. 2024), and energy storage (Ma et al. 2024; Rawat et al. 2023), with commercialization for large-scale soil amendment successfully demonstrated (O'Connor et al. 2018; Vijay et al. 2021; Xiang et al. 2023).

It is now well established that biochar can be produced from the pyrolysis of a wide variety of feedstocks (e.g., crop residues, wood, animal waste, sludge) under varying conditions (e.g., pyrolysis temperature, heating rate, residence time) and as a result, has a wide range of physicochemical properties including ash content, elemental composition, porous structure, and surface functionality (Feng et al. 2024; Ippolito et al. 2020; Keiluweit et al. 2010; Wang et al. 2020a). However, it has frequently been reported that pristine biochar may fail to meet specific environmental application requirements (Hilber et al. 2017; Joseph et al. 2021; Shen et al. 2016); in other words, it may not be fit for purpose. To design smart, tailored biochars for targeted environmental applications, current research focuses on fabricating engineered biochars and biochar composites to improve performance in areas such as contaminant adsorption/retention, nutrient delivery, C storage, and ecosystem health improvement (Liang et al. 2021; Mandal et al. 2021; Wang et al. 2022c). A variety of biochar composites have been proposed, such as those combined with minerals (Arif et al. 2021; Lin et al. 2013; Su et al. 2023), metal nanoparticles (Joseph et al. 2015a; Liu et al. 2020a; Shaheen et al. 2022), and microorganisms (Bolan et al. 2023b; Saravanan et al. 2023).

Among various types of biochar composites, those combined with minerals stand out due to their environmentally benign nature, ease of synthesis, and cost-effectiveness (Ibrahim et al. 2023b; Mandal et al. 2021; Rawal et al. 2016; Wang et al. 2022c). Different mineral types, such as primary silicate minerals (Ahmad et al. 2019; Diagboya et al. 2022), clay minerals (Rafiq et al. 2017; Zhao et al. 2023a), metal oxides (Dong et al. 2024b; Zhang et al. 2020), phosphate (Luo et al. 2022; Piri and Sepehr 2024), and sulfide/sulfate (Yang et al. 2019; Zhang et al. 2025), have been successfully integrated into biochar's carbonaceous structure using a range of approaches, including co-pyrolysis and post-pyrolysis modifications (Ahmed and Hameed 2020; Mandal et al. 2021; Wang et al. 2022c). Some of these composites have shown promising potential in areas such as soil remediation, soil fertility enhancement, wastewater treatment, and stormwater management, as evidenced by lab-scale and proof-of-concept studies (Biswal et al. 2022; Mandal et al. 2021; Pan et al. 2021).

Despite these encouraging findings for engineered biochar composites, many of the mechanisms, either in the manufacturing or application of the products, remain relatively poorly understood. Firstly, the interactions between biochar and mineral materials are often neglected or only briefly discussed, even though these interactions are essential for determining the durability and stability of such composite materials (Chang et al. 2020; Hilscher and Knicker 2011a; Jing et al. 2022; Joseph et al. 2021; Rawal et al. 2016). Similar to organo-mineral interactions in natural environments which are key to C stabilization, the interactions between biochar and minerals can substantially influence the fate and performance of these materials in environmental applications (Hemingway et al. 2019; Knicker 2011). Secondly, most studies focus on laboratory-scale tests, with a paucity of field-based studies using engineered biochar composites (Mandal et al. 2021; Wang et al. 2022c).

To address these issues, this review first provides an overview of organo-mineral interactions between biochar and minerals in natural environments to shed light on analogous interactions in engineered biochar-mineral composites. We then classify biochar composites with minerals by mineral type and summarize their production methods. We present a quantitative analysis of how mineral incorporation alters the physicochemical properties of biochar and discuss potential enhancement mechanisms. Finally, we examine existing successes and lessons from field-scale or pilot-scale applications of these composites to inform future research, commercialization opportunities and practical applications.

2 Organo-mineral interaction between biochar and minerals in the natural environment

Climate change research has seen a resurgence in interest in organo-mineral interactions with natural soil organic matter (SOM) as a protection mechanism from biotic and abiotic mineralization (Hemingway et al. 2019; Newcomb et al. 2017; Possinger et al. 2020; Zhao et al. 2023c). Biochar, either produced via pyrolysis and applied to soil, or unintentionally formed during wildfires, also forms organo-mineral complexes that further stabilize the pyrolytic C in soil.

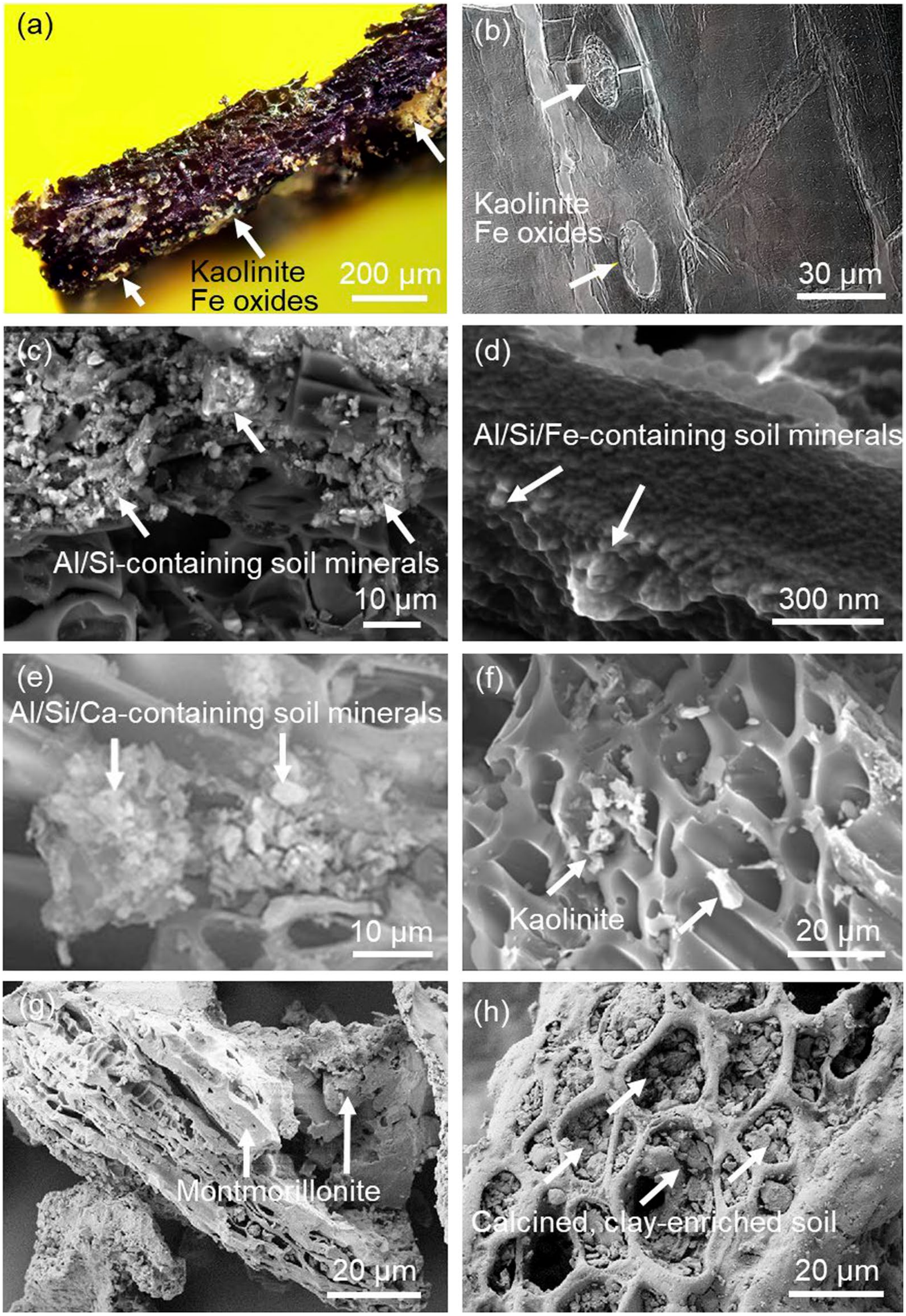
Direct evidence of such organo-mineral interactions has been reported at archaeological sites. For instance, biochar (dated to 600 ~ 1000 years BP) retrieved from *Terra Preta* at the Hatahara site near Manaus, Brazil, showed surface attachment of mineral particles, which was believed to improve the long-term stability of the biochar (Fig. 1a) (Weng et al. 2020). Synchrotron-based characterization suggested that the charosphere was enriched with poorly crystalline iron (Fe) oxides such as ferrihydrite, with Fe–O–C covalent bonds identified (Weng et al. 2020). Oxygen-containing surface functional groups, particularly phenolic groups, played a vital role in inducing organo-mineral interactions (Weng et al. 2020). It was also shown that the vascular structure in plant biomass provided substantial surface sites for mineral attachment in the retrieved biochar, suggesting that a porous structure induced by biomass properties is key to organo-mineral interactions (Fig. 1b) (Weng et al. 2020). It is noteworthy that magnetite (Fe_3O_4) was sometimes detected in *Terra Preta*, suggesting a redox active and superparamagnetic nature of these naturally aged biochars (dos Santos et al. 2023; Joseph et al. 2015b).

In addition to Fe oxides, other minerals in natural soils have been reported to bind with biochar. Surface characterization and density functional theory calculations of Amazonian Dark Earth retrieved from Manacapuru, Brazil, suggested that calcium (Ca) was strongly adsorbed onto oxidized C sites (Archanjo et al. 2014). Results from theoretical calculations further suggested that Ca could bind with oxidized C in different configurations, including Ca adsorption in the middle of the basal graphene plane (Fig. 2a, adsorption energy -121 kcal mol^{-1}), and Ca binding with epoxide (Fig. 2b, -147 kcal mol^{-1}), hydroxyl (Fig. 2c, -125 kcal mol^{-1}), carbonyl (Fig. 2d, -126 kcal mol^{-1}), carboxyl (Fig. 2e, -146 kcal mol^{-1}), and carboxylate (Fig. 2f, -292 kcal mol^{-1}) (Archanjo et al. 2014). Similarly, Ca sourced from soil was shown to coat biochar particles in a dark earth from Australia in a temperate region supporting the notion that mineral coatings/agglomeration protect biochar from further oxidation or mineralization in soil (Downie et al. 2011). Transmission and scanning transmission electron microscopy of microaggregates (<2 mm) recovered from Hatahara and Lago Grande sites near Manaus, Brazil revealed that biochar particles were closely associated with Ca-, Si-, and Al-containing mineral phases (Chia et al. 2012). A field survey at a preindustrial kiln site (approximately 220 years old) suggested that nitrogen-rich compounds derived from root and microbial activities may also promote such interactions (Burgeon et al. 2021). Another field survey at a fire-affected site in Portugal suggested that organo-mineral interactions did not require the opening of aromatic rings, and that both Fe/Al oxides and mixed-layer silicates (formed due to dehydration under fire impact) provided key interfaces for C binding (Filimonova et al. 2014). Furthermore, fine-grained minerals in the clay and silt fractions have been reported to preserve biochar in high-latitude soils. Schiedung et al. (2024) suggested a positive correlation between biochar and clay/silt content, based on mineral soils collected from 11 forest sites in the North Canadian permafrost regions. Clay minerals also protected biochar at a 55-year-old site, as indicated by the observation that the majority (55–90%) of biochar was found in heavy soil fractions enriched with clay minerals, with a density greater than 2 g cm^{-3} (Vasilyeva et al. 2011). Notably, biochar may also bind to soil minerals indirectly, as suggested by Solomon et al. (2012), who found that biochar was associated with soil minerals through interactions with non-pyrogenic C in soil. These mechanisms are complex, involving physical entrapment, ligand exchange, polyvalent cation bridging, and hydrogen bonding.

Beyond archaeological findings, long-term field experiments with biochar soil amendments have also demonstrated the role of organo-mineral interactions in protecting both biochar and rhizodeposits (new C) in soil. Surface mineral coatings have been observed on biochar after several years of aging in soil (Fig. 1c,d), and it has been shown that some mesopores in the C matrix become blocked by minerals (Fig. 1c) (Archanjo et al. 2017). Surface attachment of soil minerals is typically heterogeneous, with fine mineral aggregates attached to localized regions (Fig. 1e) (Lin et al. 2012). Similar to findings from archaeological contexts, Fe oxides such as ferrihydrite have been reported to be important in developing organo-mineral interactions (Giannetta et al. 2024, 2020). Similar to their role in aged biochar in soils (e.g., *Terra Preta*), Ca and Al also contribute to the binding of oxidized biochar and soil minerals in more recently amended soils (Lin et al. 2012; Quan et al.

(See figure on next page.)

Fig. 1 Morphologies of biochar suggesting organo-mineral interactions following aging with natural soils or minerals. Arrows in each panel indicate attached minerals. **a** Morphology of biochar retrieved from Amazonian Dark Earth from the site Hatahara, near Manaus, Brazil, showing attachment of minerals including kaolinite, ferrihydrite, goethite, and hematite. The age of the biochar ranged from 600–1000 years BP. Reproduced with permission from ref. (Weng et al. 2020). Copyright 2020 Elsevier. **b** Plant vascular structure providing substantial surface sites for mineral attachment in retrieved biochar. Reproduced with permission from ref. (Weng et al. 2020). Copyright 2020 Elsevier. **c** Surface coating of soil minerals on a green waste biochar following 9.5 years of natural aging in a pasture soil. Reproduced with permission from ref. (Archanjo et al. 2017). Copyright 2017 Elsevier. **d** Organo-mineral coating layer formed on the pore wall of a mixed plant biomass biochar following 2.5 years of natural aging in a rice paddy. Reproduced with permission from ref. (Archanjo et al. 2017). Copyright 2017 Elsevier. **e** Fine mineral aggregates attached onto localized regions in paper sludge biochar surface following 3 months of natural aging in an agricultural soil growing sweet corn. Reproduced with permission from ref. (Lin et al. 2012). Copyright 2012 Springer Nature. **f** Heterogeneous surface attachment of kaolinite on pine wood biochar following 3 months of incubation with this mineral in laboratory. Reproduced with permission from ref. (Jing et al. 2022). Copyright 2022 Elsevier. **g** Surface coating of montmorillonite on pig manure biochar following 1 month of incubation. Reproduced with permission from ref. (Ren et al. 2018). Copyright 2018 Elsevier. **h** Pore blocking of pig manure biochar following 1 month of incubation with a calcined, clay-enriched soil without organic matter in an incubation study. Reproduced with permission from ref. (Ren et al. 2018). Copyright 2018 Elsevier.



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2020). Biochar soil amendment has also been shown to improve soil aggregation, further enhancing organo-mineral interactions (Soucémarianadin et al. 2014; Sun et al. 2020). Importantly, recently photosynthesized C (i.e., rhizodeposits) can be stabilized through enhanced soil microaggregates, protecting this newly fixed C from mineralization (negative priming effect) (Weng et al. 2017, 2022).

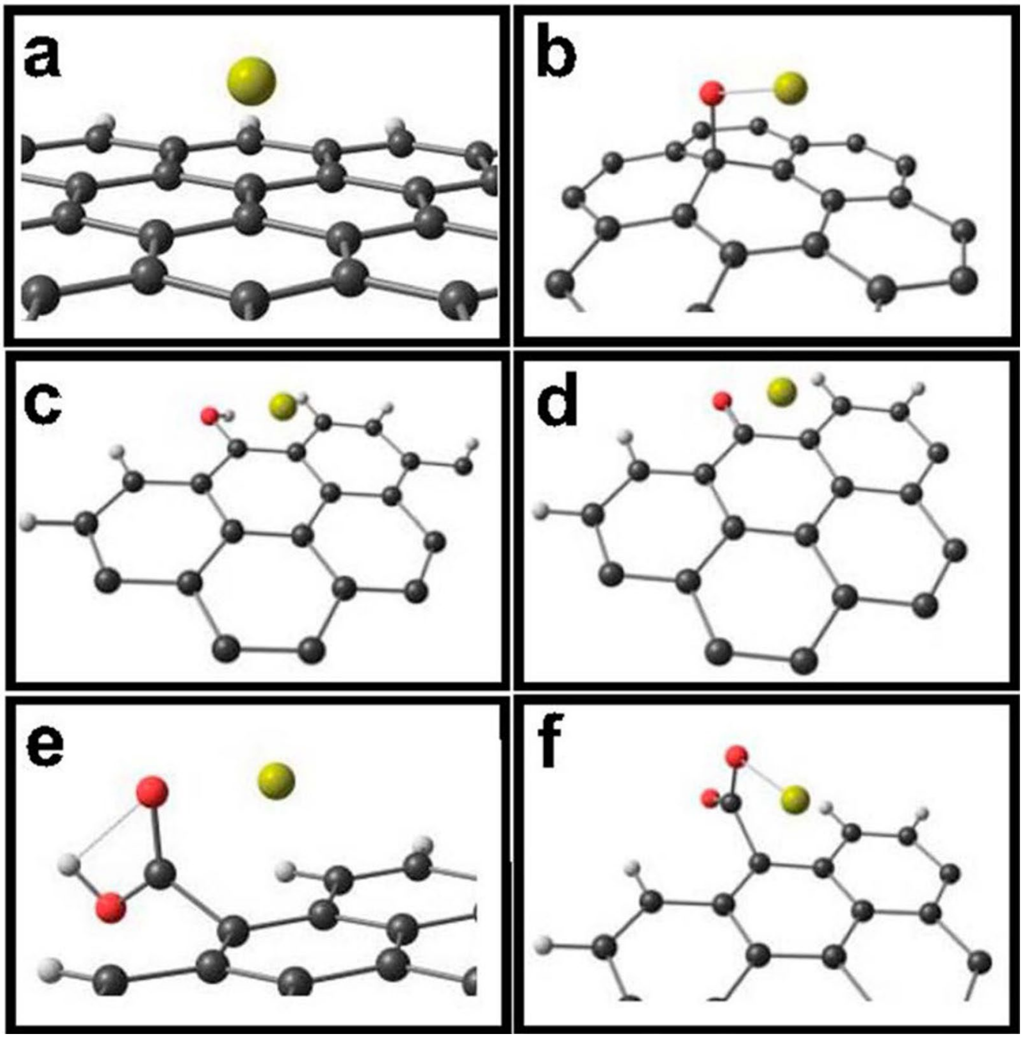

Fig. 2 Configurations of Ca binding to oxidized C. **a** Ca adsorption in the middle of the basal graphene plane; (**b**) binding with epoxide; (**c**) binding with hydroxyl; (**d**) binding with carbonyl; (**e**) binding with carboxyl; (**f**) binding with carboxylate. Black – C; White – H; Red – O; Yellow – Ca. Reproduced with permission from ref. (Archanjo et al. 2014). Copyright 2014 American Chemical Society.

Finally, lab-scale aging experiments with biochar and minerals directly indicate that organo-mineral interactions between biochar and minerals increase C stability, a key factor for longevity. These interactions occur rapidly within several months, suggesting that the initial aging process of biochar after soil amendment already involves mineral attachment and binding (Hilscher and Knicker 2011b; Jing et al. 2022; Ren et al. 2018). Similar morphologies have also been reported, with heterogeneous attachment of aggregated mineral particles and pore blocking as key features (Fig. 1f–h) (Jing et al. 2022; Ren et al. 2018). A 28-month incubation study suggested that aged biochar possessed an aryl backbone that was highly oxidized and substituted with carboxylic functional groups, enabling mineral binding (Hilscher and Knicker 2011a). Another 3-month study demonstrated that kaolinite and montmorillonite clays were attached to biochar surfaces, and that transformation of aromatic functional groups to esters favored the formation of stable C–O–Al complexes (Jing et al. 2022). Incubating biochar, kaolinite, and metal ion salts for 3 months successfully induced organometallic complexes such as Fe–O–C, which promoted biochar C stability, as revealed using the hydrogen peroxide (H_2O_2) oxidation test (Yang et al. 2016a).

It is important to note that several key findings from explorations of organo-mineral interactions between biochar and minerals in natural environments may provide valuable insights into molecular interactions and environmental applications of biochar-mineral composites. First, surface oxygen-containing functional groups are critical to organo-mineral interactions (Archanjo et al. 2014; Hilscher and Knicker 2011a; Joseph et al. 2010; Weng et al. 2020). Therefore, a biochar C matrix rich in such functional groups (e.g., low-temperature biochars pyrolyzed under 450 °C) may lead to more stable organo-mineral interactions. These functional groups are often introduced to biochar surfaces during aging with minerals (Hilscher and Knicker 2011a; Jing et al. 2022; Ren et al. 2018), which may also facilitate environmental applications involving adsorption or retention mechanisms (Chen et al. 2024; Dong et al. 2024a). Second, pore blocking may occur (Archanjo et al. 2017; Jing et al. 2022; Ren et al. 2018), indicating that mineral modification may lower the apparent porosity of biochar. Finally, improved C stability is expected for biochar-mineral composites (Han et al. 2016; Yang et al. 2021, 2016a).

3 Engineered biochar-mineral composites

3.1 Classification

Although several previous studies have summarized the use of biochar-mineral composites, such as biochar-clay and biochar-Fe oxide composites for various applications, discrepancies and inconsistencies still exist regarding the classification of such biochar composites. Here, we attempt to provide a classification of biochar composites with minerals based on mineral type (Fig. 3).

3.1.1 Biochar composites with non-clay silicate minerals

This category mainly includes biochar composites with quartz (Ahmad et al. 2019; Yang et al. 2020), feldspar (Dawodu et al. 2021; Diagboya et al. 2022), zeolite (Mosa et al. 2020; Wu et al. 2025), and tourmaline (Liu et al. 2025b; Zhu et al. 2025). Although less studied and utilized compared to other types, this group is crucial in terms of mineral classification, as non-clay silicates are typically the dominant mineral types in soils and other environmental compartments (Brady and Weil 2017).

3.1.2 Biochar composites with clay minerals

This group primarily includes biochar composites with kaolinite as a 1:1 clay mineral (Du et al. 2023; Olu-Owolabi et al. 2021), and with montmorillonite (Chen et al. 2017; Song et al. 2020), palygorskite (also known as attapulgite) (Quan et al. 2022; Wang et al. 2019), sepiolite (Hou et al. 2024b; Ibrahim et al. 2022), and vermiculite (Liu et al. 2020b; Ma et al. 2025) as 2:1 clay minerals. Note that biochar composites with Layered Double Hydroxides (LDHs) are not included in this paper, as naturally occurring LDHs such as hydrotalcites are rarely reported. For a detailed overview of synthesis and application of biochar-LDHs composites, we refer readers to Fang et al. (2021a).

3.1.3 Biochar composites with oxide minerals

Here, we use "oxides" as an umbrella term for oxides, hydroxides, and hydroxy-oxides (Komárek et al. 2013; Wang et al. 2022b; Zong et al. 2023). Biochar composites can be formed with Fe, Al, magnesium (Mg), and manganese (Mn) oxides, encompassing a variety of minerals such as goethite (Yu et al. 2023; Zhu et al. 2020), hematite (Liu et al. 2025a; Yang et al. 2019), magnetite (Han et al. 2015; Nayak et al. 2021), ferrihydrite (Chen et al. 2023; Zeng et al. 2024), gibbsite (Yu et al. 2019), periclase (Zhu et al. 2021), brucite (Wang et al. 2022b), birnessite (Wang et al. 2023b), and pyrolusite (Yang et al. 2016b).

3.1.4 Biochar composites with carbonates

This group includes composites with calcite (Ramola et al. 2021; Zhao et al. 2022), dolomite (Vamvuka et al.

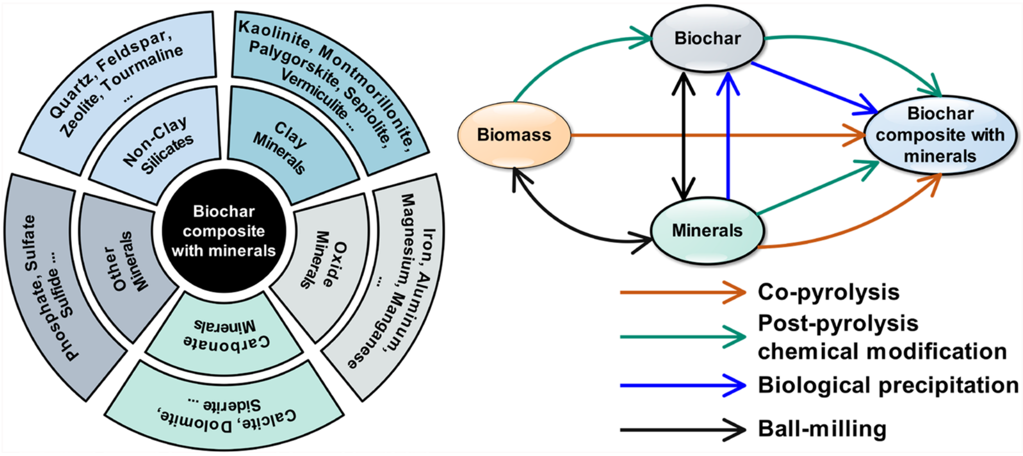

Fig. 3 Various types and fabrication methods of biochar composites with minerals

2024; Xi et al. 2023), and siderite (Lou et al. 2025; Yu et al. 2022).

3.1.5 Biochar composites with other minerals including phosphate, sulfate, and sulfide

This category includes composites with struvite (Li et al. 2020; Piri and Sepehr 2024), gypsum (Natalio et al. 2020; Zhang et al. 2025), pyrite (Tang et al. 2021; Yang et al. 2019), and related minerals.

It is noteworthy that, in some circumstances, a biochar-mineral composite may belong to multiple groups in this classification system if it contains two or more mineral types. For instance, biochar composites doped with both clay minerals and Fe oxides have been synthesized for environmental applications (Liang et al. 2019; Sun et al. 2021). Given that the mineral components in such biochar composites usually provide a synergistic effect of each mineral (Liang et al. 2019; Sun et al. 2021), we do not attempt to provide a separate classification for these composites, as there could theoretically be many possible mineral combinations.

3.2 Fabrication methods

Similar to other engineered biochars and biochar composites, biochar-mineral composites can be fabricated either via co-pyrolysis of biomass and minerals, or post-pyrolysis modification where the minerals are reacted with biochar (Fig. 3). Co-pyrolysis refers to the process in which biomass is pyrolyzed together with mineral materials or their precursors (Ahmed and Hameed 2020; Li et al. 2021b). During co-pyrolysis, clay minerals or silicates are directly pyrolyzed with biomass following the application of minerals to biomass through simple mixing (Gao and Goldfarb 2021), ball-milling (Ahmad et al. 2019), or soaking in a mineral suspension (Lin et al. 2022). It has been noted that changes in clay mineralogy may occur due to dehydration during the pyrolysis process (Filimonova et al. 2014; Rawal et al. 2016). For instance, the formation of non-porous metakaolin was reported when bamboo biomass was co-pyrolyzed with kaolinite at 550 °C, resulting in mesopore blockage (Rawal et al. 2016). Oxide minerals such as Mg, Fe, Al, and Mn oxides have been typically loaded onto biomass soaked in soluble metal salts (particularly chlorides) prior to co-pyrolysis (Dong et al. 2024b; Hou et al. 2024a; Nan et al. 2020; Zhang et al. 2022a). Pyrolysis conditions can lead to the formation of various oxide minerals within the resulting biochar composite. For example, pyrolysis of $FeCl_3$-soaked cottonwood sawdust at 500 °C produced magnetite (Fe_3O_4), whereas pyrolysis at 800 °C yielded goethite (FeOOH) (Zhang et al. 2022a).

For post-pyrolysis modification, minerals are applied onto biochar (Wang et al. 2020a; Xie et al. 2022). Several methods have been employed for this approach, including ball milling mineral particles with biochar (Shan et al. 2016), soaking biochar with minerals in aqueous suspension (Du et al. 2023), or using biological precipitation to deposit minerals onto biochar surfaces (Liu et al. 2021). It is important to note that the potential effects of post-pyrolysis modification methods on organo-mineral interactions should be further explored. For instance, whether chemical binding occurs under physical modification methods such as ball milling is a key question affecting the long-term stability of ball-milled biochars. Current evidence suggests that ball milling mainly promotes physical mixing and downsizing rather than chemical binding (Li et al. 2024b; Yang et al. 2024); therefore, the practical performance of ball-milled biochar-mineral composites should be carefully examined. Similarly, whether aqueous soaking enables the irreversible attachment of suspended mineral particles onto the biochar C matrix remains an underexplored topic, despite numerous attempts to fabricate novel biochar-mineral composites using this facile synthesis approach (Mandal et al. 2021; Wang et al. 2022c). The reversibility of organo-mineral interactions in such composites should be further investigated. For a comprehensive overview of biochar composite fabrication techniques, readers are referred to Wang et al. (2020a).

4 Physicochemical properties, structure, and enhancement mechanisms

4.1 Physicochemical properties of biochar composites

Substantial changes in physicochemical properties have been reported for biochar-mineral composites compared to their matched biochar without mineral modification. An increase in ash content is a notable feature following mineral modification, as has been reported for various minerals including non-clay silicates, 1:1 clay minerals, 2:1 clay minerals, carbonates, and Fe oxides (Fig. 4). The ash content of pristine biochar is highly dependent on feedstock type; sludge- and manure-derived biochars typically possess much higher ash content than those derived from wood and crop residues (Ippolito et al. 2020; Wang et al. 2020a). Considering that ash content mainly consists of mineral matter, it is expected that mineral modification would lead to an increase in ash content. The implications of this increase are discussed in Sect. 4.3.

Elemental composition (ultimate analysis) indicates a substantial increase in the atomic ratios of O/C and H/C following mineral modification (Fig. 5a,b). Modifications with non-clay silicates and Mg oxides resulted in the highest increases in both ratios (Fig. 5a,b). It is well established that higher O/C and H/C values correspond to a greater abundance of oxygen-containing functional

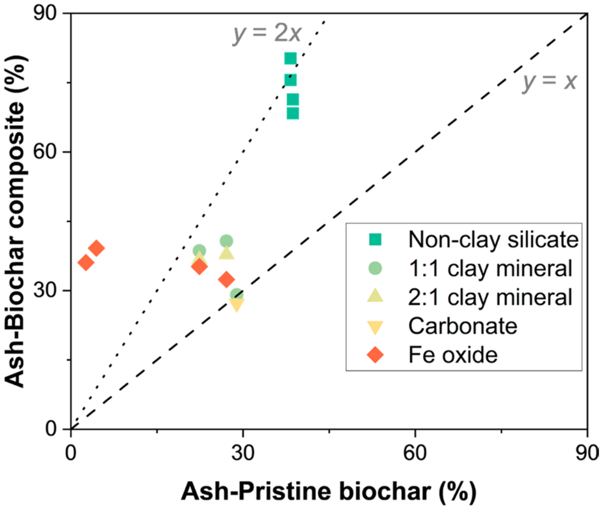

Fig. 4 Ash content of pristine biochar and biochar composites manufactured with minerals. Data sources: (Ahmad et al. 2019; Li et al. 2014; Rawal et al. 2016; Zhang et al. 2022a; Zhao and Zhou 2019)

groups, higher hydrophilicity, and increased polarity (Li et al. 2019; Wang et al. 2020a), which are crucial for environmental applications (Sect. 4.3). Furthermore, a Van Krevelen diagram of atomic O/C and H/C ratios revealed that the modified biochars exhibited higher values and wider ranges than pristine biochars (Fig. 5c). Minerals incorporated during co-pyrolysis alter biomass decomposition pathways by acting as catalysts. For example, Fe_2O_3 and Al_2O_3 accelerated the cleavage of lignocellulosic bonds in corn stalks, forming greater quantities of condensed aromatic structures (Li et al. 2025c). This catalytic effect reduces the activation energy for cellulose decomposition by 15–20%, as metal oxides facilitate free radical reactions that stabilize intermediate products (Zhang et al. 2022b).

It is also important to note that although mineral introduction leads to a bulk increase in the O/C ratio, C derived from the biomass itself may become more condensed (i.e., with higher aromaticity and stability, as well as lower H and O content). For instance, the introduction of metal oxides promotes the loss of H and O relative to C in biomass through multiple mechanisms, including 1) absorbing oxygen-containing gases such as CO_2 and H_2O produced during pyrolysis, effectively removing O from the condensed C phase and lowering the O/C ratio in the resulting biochar (Wang et al. 2020b); 2) catalyzing the decomposition or cracking of oxygenated functional groups (e.g., carboxyl, phenol) in the biomass, which facilitates the release of oxygen as gases and enhances carbonization and aromatization of the solid residue (Chatterjee et al. 2020); and 3) promoting secondary reactions that break down oxygenated compounds and polycyclic aromatic hydrocarbons (PAHs), leading to a more condensed C structure with fewer oxygen functionalities (Norouzmehr et al. 2025).

The porous structure of biochar may be either improved or degraded following mineral modification (Fig. 6). For instance, 60.9% of 62 observations reported an increase in the specific surface area (SSA) of modified biochar, while 39.1% reported a decrease (Fig. 6a). A majority of observations (65.6% of 62) found the SSA of modified biochars to be 50–200% of that of the pristine biochar (between the 1:2 and 2:1 lines) (Fig. 6a). Similarly, 67.3% of 50 observations indicated an increase in total pore volume (PV), whereas 32.7% noted a decrease (Fig. 6b). Again, most observations (67.3%) found the PV of modified biochars to fall between 50–200% of that of the pristine ones (Fig. 6b). Although mineral modification did not alter the mesoporous nature of biochar, 57.1% of 41 observations reported an increase in average pore diameter (PD), while 42.9% reported a decrease

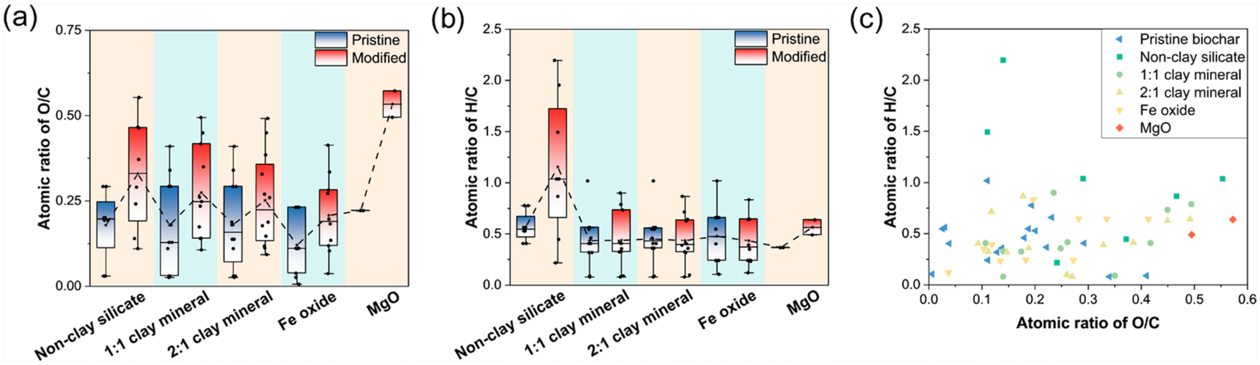

Fig. 5 Elemental composition of pristine biochar and biochar composite with minerals. **a** Comparison of atomic O/C ratio; **b** Comparison of atomic H/C ratio; **c** Van Krevelen diagram of atomic O/C and H/C ratios. Data sources: (Ahmad et al. 2019; Chen et al. 2017; Du et al. 2023; Gao and Goldfarb 2021; Luo et al. 2021; Qiu et al. 2020; Yao et al. 2014; Zhang et al. 2022a; Zhao and Zhou 2019)

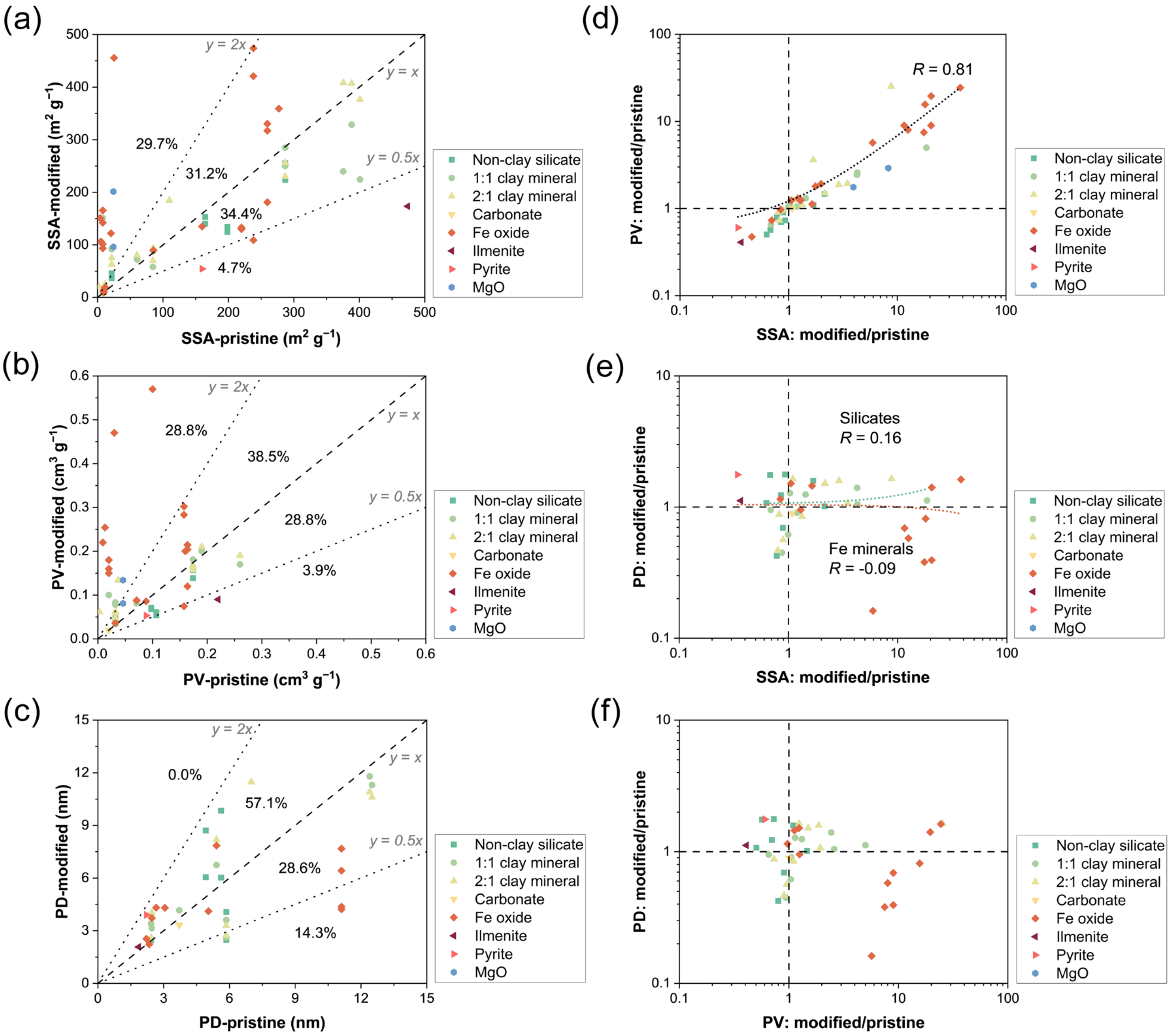

Fig. 6 Porous structure of pristine biochar and biochar composite with minerals. **a** Specific surface area (SSA); **b** Total pore volume (PV); **c** Average pore diameter (PD); **d** Ratio of specific surface area and total pore volume of engineered composite versus pristine biochar; **e** Ratio of specific surface area and average pore diameter of engineered composite versus pristine biochar; **f** Ratio of total pore volume and average pore diameter of engineered composite versus pristine biochar. Data sources: (Ahmad et al. 2019; Chen et al. 2017; Dong et al. 2024b; Du et al. 2023; Gao and Goldfarb 2021; Kim et al. 2024; Li et al. 2014; Lin et al. 2022; Liu et al. 2022; Luo et al. 2021; Qiu et al. 2020; Song et al. 2021; Sun et al. 2022; Yang et al. 2019; Yao et al. 2014; Zhang et al. 2022a; Zhao and Zhou 2019)

(Fig. 6c). Most observations (85.7% of 41) showed PD values between 50–200% of those of the pristine biochars (Fig. 6c). A positive correlation was observed between changes in SSA and PV, suggesting that an increase or decrease in SSA generally corresponded to similar changes in PV (Fig. 6d). However, the impact on SSA and PD varied with mineral type. An increase in SSA generally led to a higher PD for silicates, but a lower PD for Fe-containing minerals such as oxides, ilmenite, and pyrite (Fig. 6e). Fe-containing oxide minerals are typically formed from Fe salts during pyrolysis, while silicates are usually directly introduced to the biomass prior to pyrolysis (Sect. 3.2). This discrepancy may account for the observed differences; though further research is needed for confirmation. Moreover, an increase in PV did not consistently lead to higher PD across different mineral types (Fig. 6f).

A limited number of studies have explored the mechanisms by which minerals affect biochar porous structures. A previous study suggested that 2:1 clay minerals (montmorillonite and bentonite) promote devolatilization during pyrolysis, increasing micropore development by 30–40% compared to pure biomass pyrolysis (Zhang et al. 2022b). These structural changes correlate with enhanced oxygen functional group retention, as minerals inhibit volatile loss at temperatures below 500 °C. In contrast, it was reported that when bamboo biomass was co-pyrolyzed with iron sulfate and kaolinite clay (a 1:1 clay mineral), the porosity of the resulting biochar-mineral composite decreased with increasing pyrolysis temperature (Fig. 7) (Rawal et al. 2016). This phenomenon was unexpected; as elevated pyrolysis temperatures typically improve the porous structure of biochars (Fig. 7) (Leng et al. 2021; Rawal et al. 2016; Wang et al. 2020a). It is possible that mesopore blocking occurred when kaolinite transformed into non-porous metakaolin during pyrolysis at relatively high temperatures (Rawal et al. 2016). Such discrepancies highlight the complexity of porous structure changes associated with different minerals under varying pyrolysis conditions. Engineering and optimizing PV, PD, and SSA are critical for environmental applications, and significant opportunities exist to optimize biochar composites for these purposes (Sect. 4.3).

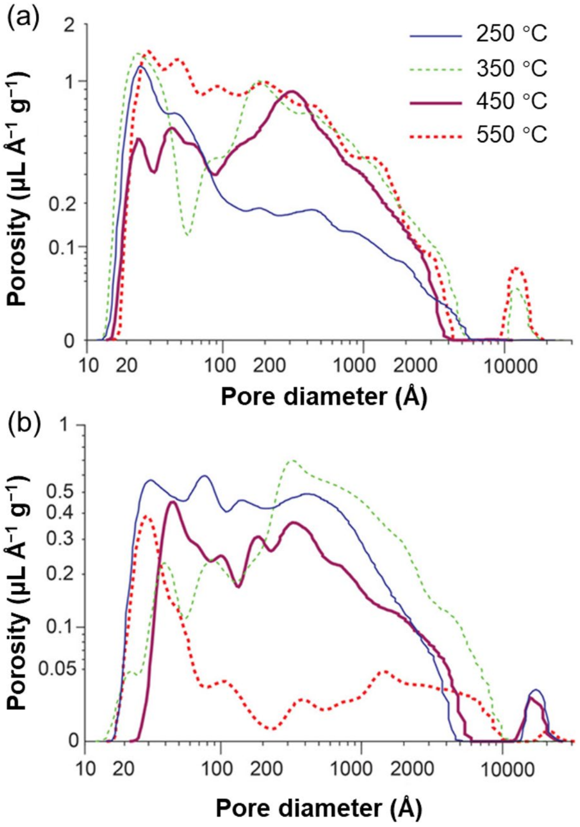

Fig. 7 Porosity of pristine biochar and biochar-Fe-clay composite synthesized under varying pyrolysis temperatures. **a** Pristine bamboo biochar; **b** Biochar composite synthesized from co-pyrolysis of bamboo, $FeSO_4$, and kaolinite. Reproduced with permission from ref. (Rawal et al. 2016). Copyright 2016 American Chemical Society.

4.2 Structure of biochar composite with minerals

The co-pyrolysis of biomass with mineral additives, or the post-pyrolysis modification of biochar using minerals, has emerged as a transformative approach to engineer biochar with tailored surface functional groups, enhancing its environmental applications (Mandal et al. 2021; Wang et al. 2020a). This process modifies oxygen-containing functional groups (e.g., carboxyl, hydroxyl), introduces mineral-derived moieties (e.g., silicates, metal oxides, phosphates, sulfates), and stabilizes aromatic structures through catalytic interactions (Ahmed and Hameed 2020; Li et al. 2021b; Nan et al. 2022). Minerals such as clay minerals, Fe oxides, and phosphate-rich compounds act as catalysts during pyrolysis, promoting the formation of electron-donating phenolic/quinoid groups and metal-organic complexes (Chatterjee et al. 2020; Li et al. 2025c; Zhang et al. 2022b). These structural changes improve biochar's cation exchange capacity, redox activity, and adsorption efficiency towards contaminants (Premarathna et al. 2019b; Su et al. 2023; Wang et al. 2022c). By systematically altering surface chemistry, mineral co-pyrolysis or post-pyrolysis modification enables the design of biochars for targeted pollutant remediation, soil amendment, and C sequestration applications.

Structural changes to biochar have been observed using a range of methodologies, including microscopy, following the engineering of biochar composites (Fig. 8). For instance, quartz grains have been observed to fuse with the biochar C matrix following ball milling of cellulose and quartz powder, followed by co-pyrolysis at 600 °C for 4 h (Yang et al. 2020) (Fig. 8a). Zeolite particles completely covered the biochar C matrix after mixing date palm tree biomass with zeolite in aqueous media, followed by co-pyrolysis at 600 °C for 3 h (Ahmad et al. 2019) (Fig. 8b). Clusters of kaolinite clay nanoparticles were observed when pig bone biochar was mixed with kaolinite in aqueous media at 25 °C for 7 days (Du et al. 2023) (Fig. 8c). Montmorillonite sheets were reported to cover the surface of cellulose-derived biochar following co-pyrolysis at 600 °C for 1 h (Gao and Goldfarb 2021) (Fig. 8d). The fibrous clay mineral palygorskite was successfully introduced to biochar via mixing sesame straw with palygorskite in aqueous media and co-pyrolysis at 500 °C for 2 h (Lin et al. 2022) (Fig. 8e). Calcite particles were incorporated into the C matrix after ball milling cellulose with calcite powder and co-pyrolysis at 600 °C for 4 h (Yang et al. 2020) (Fig. 8f). Magnesium oxide nanoparticles

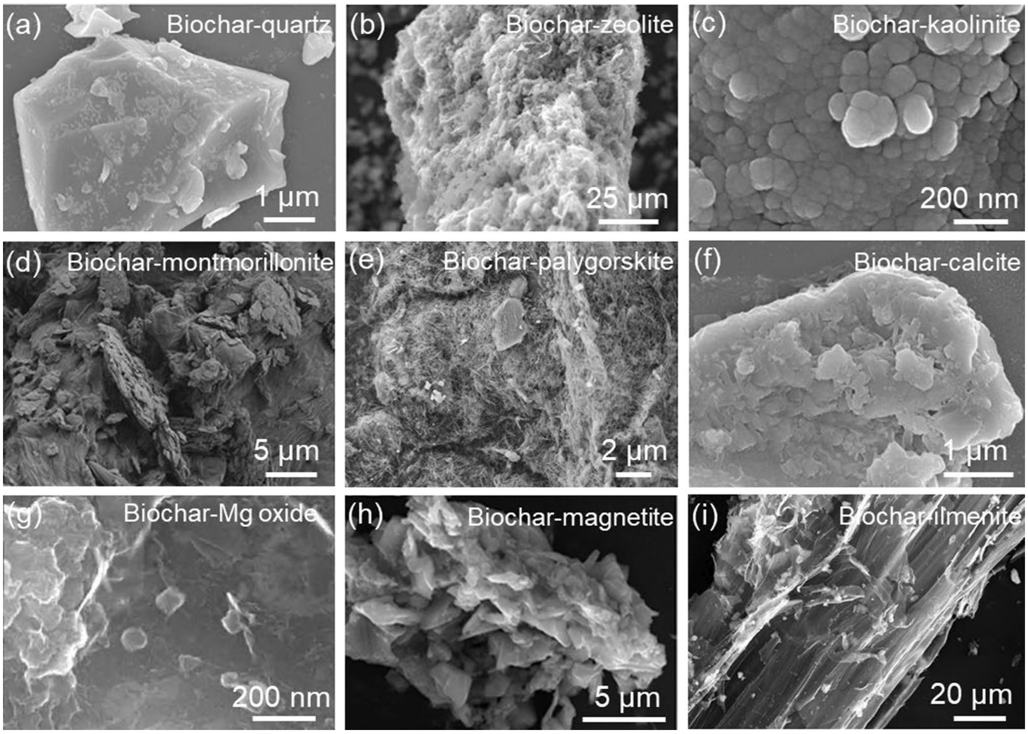

Fig. 8 Morphologies of synthesized biochar composites with minerals. **a** Biochar-quartz composite synthesized via ball milling of cellulose and quartz powder, and co-pyrolyzed at 600 °C for 4 h. Reproduced with permission from ref. (Yang et al. 2020). Copyright 2020 Elsevier. **b** Biochar-zeolite composite synthesized via mixing of date palm tree and zeolite in aqueous media, and co-pyrolyzed at 600 °C for 3 h. Reproduced with permission from ref. (Ahmad et al. 2019). Copyright 2019 Springer Nature. **c** Biochar-kaolinite composite synthesized via post-pyrolysis mixing of pig bone biochar and kaolinite in aqueous media at 25 °C for 7 d. Reproduced with permission from ref. (Du et al. 2023). Copyright 2023 Elsevier. **d** Biochar-montmorillonite composite synthesized via dry mixing of cellulose of montmorillonite, and co-pyrolyzed at 600 °C for 1 h. Reproduced with permission from ref. (Gao and Goldfarb 2021). Copyright 2021 Springer Nature. **e** Biochar-palygorskite composite synthesized via mixing of sesame straw and palygorskite in aqueous media, and co-pyrolyzed at 500 °C for 2 h. Reproduced with permission from ref. (Lin et al. 2022). Copyright 2022 Elsevier. **f** Biochar-calcite composite synthesized via ball milling of cellulose and calcite powder, and co-pyrolyzed at 600 °C for 4 h. Reproduced with permission from ref. (Yang et al. 2020). Copyright 2020 Elsevier. **g** Biochar-magnesium oxide composite synthesized via co-pyrolysis of $Mg(OH)_2$ powder and KOH-soaked tree leaves at 600 °C for 2 h. Reproduced with permission from ref. (Luo et al. 2021). Copyright 2021 Elsevier. **h** Biochar-magnetite composite synthesized via soaking peanut shell with $FeCl_3$, and co-pyrolyzed at 800 °C for 3 h. Reproduced with permission from ref. (Dong et al. 2024b). Copyright 2024 Royal Society of Chemistry. **i** Biochar-ilmenite composite synthesized via mixing of corn stalk and ilmenite in aqueous media, and co-pyrolyzed at 600 °C for 6 h. Reproduced with permission from ref. (Liu et al. 2022). Copyright 2022 Elsevier.

showed heterogeneous attachment on biochar following co-pyrolysis of $Mg(OH)_2$ powder and KOH-soaked tree leaves at 600 °C for 2 h (Luo et al. 2021) (Fig. 8g). Magnetite particles were attached to the C matrix after soaking peanut shells in $FeCl_3$ and co-pyrolyzing at 800 °C for 3 h (Dong et al. 2024b) (Fig. 8h). Finally, ilmenite was sparsely loaded onto biochar by mixing corn stalk with ilmenite in aqueous media, followed by co-pyrolysis at 600 °C for 6 h (Liu et al. 2022) (Fig. 8i).

Mineral additives during pyrolysis introduce new functional groups to biochar through chemical bonding and surface deposition. Spectroscopic techniques such as Fourier Transform Infrared Spectroscopy (FTIR) and X-ray Photoelectron Spectroscopy (XPS) have revealed that biochar-mineral composites exhibit a mixed spectroscopic pattern derived from both biochar and the associated minerals (Arif et al. 2021; Dong et al. 2024b; Song et al. 2020; Wang et al. 2022b; Yang et al. 2020). Notable features include the presence of moieties such as Si–O, Al–O, Fe–O, and Mg–O, as well as an increased intensity of oxygen-containing functional groups including hydroxyl, carbonyl, and ester groups (Chen et al. 2017; Liu et al. 2023; Song et al. 2020; Wang et al. 2022b). Besides, phosphate minerals (e.g., hydroxyapatite) react with biomass-derived volatiles to form C–O–P and P–O–P linkages, detectable via FTIR at 1040–1100 cm^{-1} (Ahmed and Hameed 2020). Sulfur-containing minerals like gypsum promote sulfonic acid ($-SO_3H$) group formation through sulfonation reactions above 450 °C, which persist even after pyrolysis due to mineral stabilization (Li et al. 2021a).

X-ray Diffraction (XRD) analysis has shown that the C matrix of biochar is generally amorphous (Qiu et al. 2020; Song et al. 2020; Wang et al. 2022b). However, biochar-mineral composites modified with carbonates or non-clay silicates exhibit strong diffraction peaks corresponding to crystalline minerals such as quartz, feldspar, and calcite (Ahmad et al. 2019; Ramola et al. 2021). It should be noted that Fe oxides typically produce weaker diffraction peaks due to their amorphous nature (Liu et al. 2023; Zhang et al. 2020), and that clay minerals generally show weaker diffraction signals compared to non-clay silicates (Chen et al. 2017; Song et al. 2020).

It should be noted that although numerous studies have engineered various biochar composites with minerals, the molecular binding mechanisms between the C matrix and minerals remain poorly understood. Relying solely on microscopic approaches and surface characterizations such as FTIR, XPS, and XRD is insufficient to fully elucidate these interactions. To date, only a limited number of studies have addressed this issue. Using quantitative ^{13}C nuclear magnetic resonance (NMR) spectroscopy, Rawal et al. (2016) found that the presence of kaolinite (1:1 clay) or bentonite (2:1 clay) inhibited the degradation of cellulosic components at low pyrolysis temperatures (250 °C). In contrast, at higher temperatures (350–550 °C), these clays promoted biomass degradation into condensed aromatic moieties, likely due to their catalytic role in facilitating condensation and polymerization reactions (Rawal et al. 2016). It was also found that in some cases, minerals were fused into the C matrix (Fig. 8a,f), while in others, mineral particles were merely attached to the surface of the biochar (Fig. 8b,d,h,i). The mechanisms underlying these phenomena warrant further investigation. The reversibility and long-term stability of these mineral attachments require further investigation, particularly in non-soil applications (Sect. 6).

4.3 Enhancement mechanisms

The mechanisms by which introduced mineral components enhance biochar's performance vary and are dependent on specific purposes (Fig. 9). First, biochar has been originally used as a soil amendment to improve soil fertility (Chan et al. 2007; Sohi et al. 2010; Van Zwieten et al. 2010). Biochar composites have been developed with the aim of improving soil mineral nutrition over the pristine biochars; for example, co-pyrolysis with kaolinite

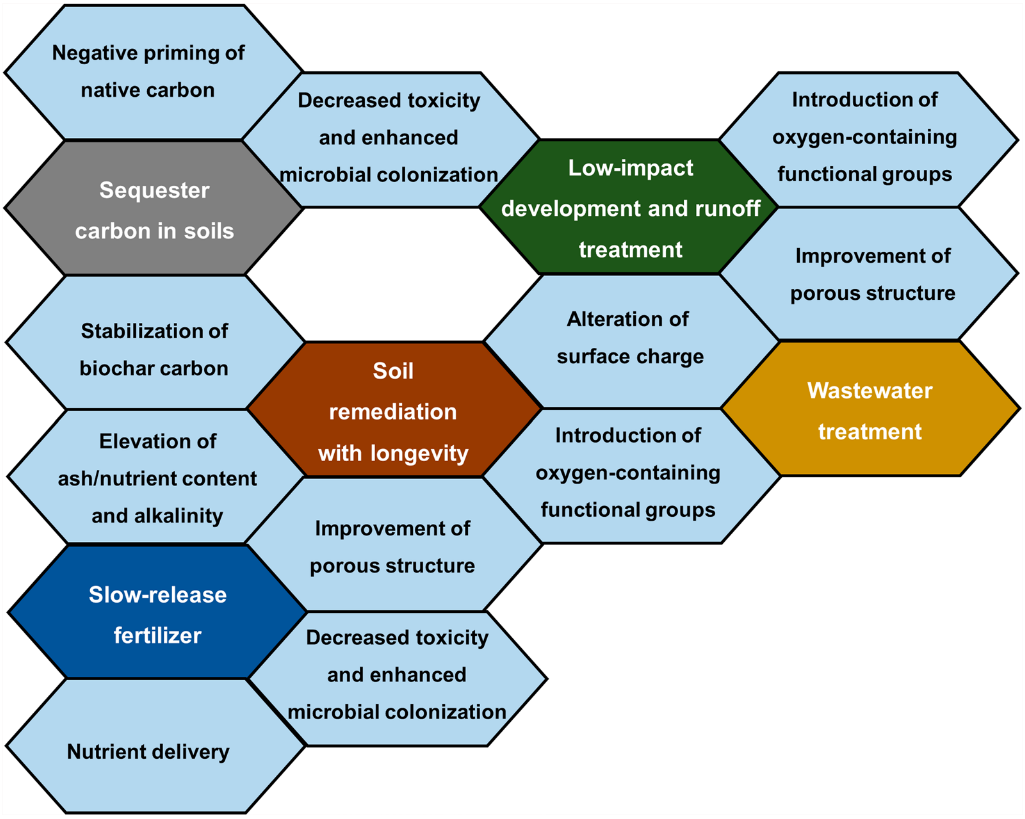

Fig. 9 How biochar doped with minerals improve its performance in real-world applications

(Al-Swadi et al. 2024), montmorillonite (Chen et al. 2017), zeolite (Dong et al. 2025), magnesium oxide (Ibrahim et al. 2023a), calcite (Li et al. 2024d), or struvite (Piri and Sepehr 2024), can directly deliver more nutrients (e.g., Mg, K, Ca, P) to the soil due to the nutrient-rich nature of these minerals. An increase in alkalinity may also reduce soil acidity, promoting plant growth in acidic soils (Bolan et al. 2023a; Brady and Weil 2017; Ding et al. 2016). Furthermore, it has been documented that biochar itself can contain toxic substances such as PAHs (Buss et al. 2022a; Yao et al. 2024). The introduction of minerals to biochar can reduce the toxicity of such substances via suppressed formation or catalytic degradation (Cheng et al. 2025; Zhao et al. 2020) and can enhance colonization by plant growth-promoting bacteria in the rhizosphere (Bolan et al. 2023b; Zhang et al. 2018), thereby improving soil health (Antonangelo et al. 2025; He et al. 2021). The high cation exchange capacity of certain minerals can enhance nutrient retention, enabling biochar to function as a slow-release fertilizer (Vithanage et al. 2023; Yao et al. 2014). In some cases, increased SSA and PV may also favor nutrient retention; however, not all modifications lead to such improvements (Fig. 6).

Biochar has also been introduced as a C-negative tool to store C in the ground beyond its agricultural use (Ayaz et al. 2025; Wang et al. 2023a). In this context, mineral modifications provide mechanisms similar to biochar-mineral interactions in the natural environment (Sect. 2), which may enhance resistance to environmental aging processes such as oxidation (Jing et al. 2022; Yang et al. 2016a, 2018). Following the co-pyrolysis of bamboo biomass with Fe sulfate-clay slurries, Rawal et al. (2016) reported a much higher nonprotonated aromatic C fraction in the resulting biochar, indicating a higher degree of aromatic condensation and, therefore, greater C stability. Similarly, following the co-pyrolysis of tree leaves with silicate minerals, Ahmad et al. (2019) reported higher aromaticity and thermal stability compared to pristine biochar. Furthermore, decreased C loss (i.e., increased biochar yield) was observed when biomass feedstock was co-pyrolyzed with minerals, accompanying an elevation in C stability of the resulting biochar (Buss et al. 2022b; Wang et al. 2022a). Negative priming of native soil organic matter can also be achieved, enhancing the long-term stability of newly photosynthesized C (e.g., rhizodeposits) (Weng et al. 2017).

Biochar has also been widely applied for soil remediation due to its promising capacity for immobilizing potentially toxic elements (PTEs) and organic pollutants (Nguyen et al. 2023; Wang et al. 2021a; Xie et al. 2015). Immobilization/retention mechanisms can be both direct and indirect (He et al. 2019; Wang and Hou 2024). Direct mechanisms primarily involve surface complexation with oxygen-containing functional groups, surface precipitation, hydrogen bonding, ion exchange, and electrostatic interactions (Bandara et al. 2020; He et al. 2019; Palansooriya et al. 2020). Physical adsorption due to biochar's porous structure is also a direct mechanism but is often secondary, as decreased SSA and PV do not always correspond to reduced immobilization efficiency after mineral modification (Liu et al. 2024; Wang et al. 2022b). Indirect mechanisms primarily involve the modification (usually increase) of soil pH, creating conditions favorable for contaminant adsorption onto soil particles (He et al. 2019; Lwin et al. 2018). For remediation, several types of biochar-mineral composites have shown promise. Biochar composites with oxide minerals provide abundant sites for inner-sphere complexation and precipitation of PTEs, as well as electrostatic and hydrogen bonding with organic pollutants (Bandara et al. 2020; Beiyuan et al. 2023; Gong et al. 2022). Biochar composites with silicates such as zeolite (Zhao et al. 2023b), kaolinite (Wang et al. 2022e), montmorillonite (Ahmad et al. 2024), sepiolite (Hou et al. 2024b), and palygorskite (Xia et al. 2021) offer abundant hydroxyl functional groups and high cation exchange capacity. Biochar composites with magnesium oxide exhibit high affinity toward PTEs due to the cementitious behavior of Mg during hydration, enabling surface precipitation and physical entrapment (Li et al. 2022; Shen et al. 2019; Wang et al. 2021b). Biochar composites with phosphate and sulfide minerals can promote the precipitation of cationic metals into insoluble phosphates or sulfides (Li et al. 2024a, 2025b). Accelerated aging studies in laboratory settings have shown that biochar-mineral composites, especially those with oxides, exhibit improved long-term stability, which is essential for sustained remediation (Li et al. 2025a; Wang et al. 2022b). Enhanced C stability from mineral modification may further improve long-term performance (Yang et al. 2016a, 2018).

Biochar's potential to adsorb pollutants such as PTEs, dyes, pesticides, and antibiotics in aqueous media has also been well explored in the past decade, due to its porous nature and tunable surface chemistry (Ahmad et al. 2014; Cheng et al. 2021; Liang et al. 2021). Applications include removal of PTEs and organic pollutants from wastewater (Cheng et al. 2021; Jagadeesh and Sundaram 2023; Qiu et al. 2022) and stormwater runoff (Biswal et al. 2022; Ouedraogo et al. 2023). To further enhance the adsorption performance for PTEs and organic pollutants, a variety of minerals can be introduced into biochar materials. For instance, biochar composites with oxides and clay minerals show promise for PTEs adsorption in aqueous environments (Arif et al. 2021; Shaheen et al. 2022; Zhao et al. 2021). These mechanisms mirror those of direct soil immobilization, including surface complexation, surface precipitation, and electrostatic interactions (Arif et al. 2021; Shaheen et al. 2022; Zhao et al. 2021). Biochar composites with clay minerals have also been applied for the adsorption of organic pollutants such as dyes (Premarathna et al. 2019a; Tcheka et al. 2025; Yang et al. 2025), pesticides (Rallet et al. 2022; Wang et al. 2022d), and antibiotics (Ashiq et al. 2019; Premarathna et al. 2019a). Their enhancement mechanisms include hydrogen bonding, electrostatic interactions, and physical adsorption (Premarathna et al. 2019a; Wang et al. 2022d; Yang et al. 2025). Numerous lab-scale studies have developed such composites; however, the reversibility and strength of mineral attachment are critical considerations, as they affect the reversibility of adsorption (Imran et al. 2021; Murtaza et al. 2022; Premarathna et al. 2019b).

5 Practical utilization of biochar composites

5.1 Field application to lower heavy metal availability from soil

Although biochar-mineral composites are widely used as gentle remediation options (GROs) in lab-scale and pot experiments (Arif et al. 2021; Mandal et al. 2021; Su et al. 2023), only a few field-scale applications have been reported. For instance, to improve cadmium (Cd) immobilization in field conditions, a biochar composite was engineered from sawdust biomass that was soaked with $MnSO_4$ and co-pyrolyzed at 600 °C for 2 h, forming oxides and sulfates including hausmannite (Mn_3O_4), manganosite (MnO), szmikite ($MnSO_4 \cdot H_2O$), and anhydrite ($CaSO_4$) on the biochar surface (Fang et al. 2021b). The resulting biochar composite was applied at a dose of 2.6 t ha^{-1} in a rice paddy, successfully reducing Cd in rice grain by 78–82% (Fang et al. 2021b). Increased Mn availability in the soil led to reduced Cd uptake due to Mn–Cd competition at the OsNramp5 transporter (Fang et al. 2021b).

Pristine biochar has been shown to be ineffective at immobilizing oxyanions like arsenic (As) and antimony (Sb) due to electrostatic repulsion and increased bioavailability at elevated pH. To overcome this, Fe oxides were loaded onto biochar, and the resulting composite was applied to soil in a rice–wheat rotation at a dose of 1.5–3 t ha^{-1} (Tang et al. 2020). Sustained immobilization of As and Cd and increased crop yield were reported over two years (Tang et al. 2020). The proposed mechanisms for lowering metal content involved surface complexation and precipitation (Tang et al. 2020). Another study applied Fe-oxide-modified biochar composites to immobilize As and Sb at a smelting site (Li et al. 2024c). In this case, the biochar composite not only immobilized As and Sb chemically but also served as a long-term C source for inoculated Sb-oxidizing bacteria, offering synergistic chemical-biological remediation (Li et al. 2024c).

5.2 Field application for agriculture and soil health improvement

Field studies have also used biochar-mineral composites for nutrient delivery. An Amazonian Dark Earth-like biochar composite was fabricated by thermally treating a mixture of clay, organic matter, and wood biochar at 220 °C in a steam environment, then applied to wheat and sorghum fields (Blackwell et al. 2015). Similar to findings from *Terra Preta* (Lehmann et al. 2003), the biochar composite improved nutrient uptake (N, P, K, S, Zn) and mycorrhizal colonization (Blackwell et al. 2015). Co-pyrolysis of yak dung and palygorskite formed biochar-clay composites that enhanced pasture growth (Rafiq et al. 2017). These improvements were attributed to surface oxygen-containing groups, increased SSA and PV, and higher concentrations of Fe/O nanoparticles such as magnetite, which enriched the rhizosphere microbiome (Rafiq et al. 2017). Biochar-MgO and biochar-FeOOH composites have also increased phosphate retention in saline-alkaline soils, functioning as slow-release fertilizers through direct phosphate adsorption (Wu et al. 2019, 2020).

In addition to nutrient delivery, biochar-mineral composites offer other environmental benefits. A 3-month field trial showed that applying biochar-MgO composite at 40 t ha^{-1} mitigated carbon dioxide (CO_2) emissions from agricultural soils, possibly via $MgCO_3$ formation (Ibrahim et al. 2022). However, it should be noted that not all effects were positive. The field trial by Ibrahim et al. (2022) found that while biochar-sepiolite composite initially reduced CO_2 emissions in the first 42 days, cumulative emissions increased over time. This may be due to nanoscopic channels filled with molecular water, limiting CO_2 capture despite promising lab results (Cecilia et al. 2018; Ibrahim et al. 2022). Furthermore, future field trials may also explore the potential role of biochar-mineral composites in reducing methane (CH_4) and nitrous oxide (N_2O) emissions from soils. These greenhouse gases have much higher global warming potentials compared with CO_2 and are key contributors to emissions from waterlogged soils such as paddy fields (Liu et al. 2019; Qian et al. 2023). An assessment of published meta-analyses has revealed that biochar can lower N_2O emissions by up to 49% in dryland systems and 22% in paddy systems, although the effect tends to be most pronounced within the first 2 years of application to soil (Van Zwieten et al. 2024). The same analyses revealed that CH_4 emissions can be lowered by 18% (Van Zwieten et al. 2024). Based on these findings, it can be expected that biochar-mineral composites can be optimized to lower soil CH_4 and N_2O emissions, with Van Zwieten et al. (2024) suggesting potential modification mechanisms to optimize greenhouse gas emissions.

5.3 Practical application for wastewater treatment

Compared to soil uses, non-soil applications of biochar-mineral composites remain very limited, with only a few pilot-scale studies. Dalahmeh et al. (2020) used Fe- and Ca-modified biochars as filter media for phosphate adsorption over 148 weeks. These modified biochars outperformed pristine biochar at high hydraulic loading rates due to strong surface complexation and phosphate precipitation, forming akaganeite and dicalcium phosphate (Dalahmeh et al. 2020). Huang et al. (2022) used $AlCl_3$-soaked bamboo biomass to fabricate Al oxide biochar composites, which effectively removed antibiotics such as sulfamethoxazole and sulfapyridine. Furthermore, Jia et al. (2020) found that Fe-loaded biochar improved nitrogen removal in constructed wetlands by enhancing microbial activity and increasing genes associated with nitrogen cycling (e.g., *amx*, *nirS*, *nosZ*) (Jia et al. 2020).

5.4 In-situ application for stormwater management

Urbanization has increased impervious surfaces, generating stormwater runoff rich in contaminants (Lapointe et al. 2022; Lund et al. 2019). Due to its porosity and adsorption capacity, biochar-mineral composites have been proposed for stormwater management. For instance, biochar was uniformly mixed with lateritic red soils as a filter layer in a field-scale bioretention pond (Mai and Huang 2021). Compared to other settings such as biochar stratification with red soils, biochar-red soil composite substantially increased retention of nitrogen and phosphorous from stormwater (Mai and Huang 2021). Another field study by Teixidó et al. (2022) adopted a mixed media of biochar and Mn oxide, which effectively removed both organic and inorganic contaminants from stormwater, such as 2,4-dichlorophenoxy acetic acid, diuron, copper (Cu), and Cd. Mn-oxide-modified biochar composites provided multiple adsorption mechanisms, including pore filling, hydrogen bonding, and electrostatic interactions for enhanced sorption of organic contaminants and precipitation for metals (Teixidó et al. 2022).

6 Challenges and future research directions

In recent years, the fabrication of biochar composites doped with minerals has emerged as a prominent research topic. The focus remains largely centered on the development of novel composites, with an emphasis on their characterization and evaluation of their performance. Despite these scientific and engineering advances, only limited examples of commercial or field scale deployment exist.

A wide variety of biochar composites have been fabricated using diverse methods, primarily through co-pyrolysis and post-pyrolysis modification. Although these fabrication techniques are well documented, an important issue often overlooked is the reversibility of organo-mineral interactions, which directly affects the stability of biochar-mineral composites. Reversibility is a critical factor directly linked to the stability of biochar-mineral composites, especially in non-soil applications such as wastewater or stormwater contaminant adsorption. While natural aging processes of biochar suggest that mineral attachment is typically unidirectional in the environment, it remains uncertain whether organo-mineral binding formed under accelerated synthetic conditions is as stable as that formed naturally. In this regard, co-pyrolysis may offer advantages over post-pyrolysis modification, as the high-temperature treatment may facilitate stronger and more permanent binding between biomass-derived C and minerals. Further research is therefore needed to better understand the stability and changes to biochar composites when used for their specific purposes.

Secondly, the molecular mechanisms governing organo-mineral interactions are still poorly understood. Current studies rely heavily on conventional surface characterization techniques, including Scanning Electron Microscope (SEM), XRD, FTIR, and XPS, to verify mineral incorporation into the C matrix. However, in-depth molecular-level investigations are essential for advancing environmental applications. For example, determining whether minerals are loosely attached to the biochar surface or embedded within the mesopores is vital for predicting changes in key porous features such as SSA and PV. Similarly, understanding the formation pathways and timing of specific oxide minerals during co-pyrolysis of metal salts is crucial for designing biochar composites, such as magnetite-loaded composites for environmental applications.

The commercial utilization of biochar composites remains limited, despite numerous short-term proof-of-concept studies. Only a few investigations have evaluated the practical use of such biochar composites in field or pilot-scale demonstrations, and the long-term effectiveness, an essential consideration in soil applications, has received limited attention. Extended field trials (e.g., longer than 3 years) are needed to assess whether minerals may detach from the C matrix, thereby diminishing or promoting the effects of the composites, while accelerated aging experiments in the laboratory can serve as predictive tools to assess composite longevity before full-scale deployment.

In addition, the production cost and environmental footprint associated with engineered biochar-mineral composites warrant careful consideration. Multiple steps required for biochar-mineral composite production, such as mineral impregnation, high-temperature co-pyrolysis, and surface functionalization, can substantially increase energy consumption and operational complexity compared to conventional biochar production, therefore leading to higher environmental footprints. A life-cycle assessment (LCA) approach is helpful to evaluate the net environmental benefits of such composites compared with pristine biochars. Future research should also focus on optimizing fabrication parameters to achieve a balance between performance enhancement and production sustainability, including the use of low-cost mineral feedstocks from industrial wastes, and renewable energy sources for co-pyrolysis.

Finally, while biochar itself has been applied in a range of emerging fields, such as ruminant animal feed additives, aquaculture, energy storage, and construction materials (e.g., cement additives), current applications of biochar composites remain largely restricted to soil enhancement and pollutant remediation. Given the natural abundance and environmental safety of minerals, future research should pursue innovative applications of biochar-mineral composites to realize multifunctional benefits.

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Author contributions

Liuwei Wang: Conceptualization, Investigation, Methodology, Data curation, Writing-original draft. Jiale Yang: Investigation, Methodology, Data curation, Writing-review & editing. Xuanru Li: Investigation, Methodology, Data curation, Writing-review & editing. Liping Zhang: Investigation, Methodology, Data curation. Lukas Van Zwieten: Investigation, Methodology, Writing-review & editing. Ondřej Mašek: Investigation, Methodology, Writing-review & editing. Stephen Joseph: Investigation, Methodology, Writing-review & editing. Kaikai Zhang: Conceptualization, Supervision, Methodology, Writing-review & editing. Kefu Yu: Conceptualization, Project administration, Supervision, Methodology, Writing-review & editing.

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Data availability

Data will be made available on request.

Declarations

Competing interests

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