

Review

Nutrient recovery in pyrolysis biochars for agricultural applications–Biochemical compositions and nutrient extraction techniques

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ABSTRACT

Biochar has a very wide range of uses in industrial processes, wastewater treatment, agriculture and soil remediation. Scaling up biochar deposits in soils has been proposed for carbon sequestration in the context of climate mitigation strategies. Although certain contaminants in feedstock can be eliminated through pyrolysis, others persist, and the process itself produces additional hazardous substances. Considerable uncertainty exists regarding interactions with soil microorganisms and longer-term dynamics. Large-scale deposits of biochar in soils may pose risks, and geostorage has been suggested as a safer alternative. Both geostorage and biochar deposits in deep soil layers would restrict availability to plants and soil ecosystems of valuable nutrients, such as phosphorus. Against this background, a systematic search was undertaken in the Web of Science Core Collection of publications up to February 2024 to provide an overview of existing studies on methods to retrieve nutrients from biochar. The search produced a total of 17 relevant publications. Findings point to the need for additional studies and work to develop nutrient extraction methods that are both efficient and environmentally safe. None of the reviewed studies tested the extracted phosphorus as fertilizer, and simultaneous extraction of plant nutrients from biochar might be a more advantageous approach.

1. Introduction

The global average of carbon dioxide (CO₂) concentration in the atmosphere has increased from around 280 ppm in the mid-1700s to about 415 ppm in 2021 (Friedlingstein et al., 2022), while it was never above 300 ppm during the interglacial cycles over hundreds of thousands of years ago (Valone, 2021). The large-scale burning of fossil fuels is the most significant driver of the contemporary rise of atmospheric CO₂, which is one of the main drivers of global warming and climate change that threaten life on Earth. It has been estimated that the CO₂ concentration can be more than doubled by the end of the 21st century under the business-as-usual climate pathway (Zemah-Shamir et al., 2021). There is thus an urgency to identify possible carbon sinks to store excess CO₂ (Lal, 2004, 2021; IPCC, 2022).

The Intergovernmental Panel on Climate Change defines carbon sequestration as the “process of storing carbon in a carbon pool” (IPCC, 2021a, p. 2242). Soil is a natural carbon pool, functioning as a medium for climate regulation by storing broken-down carbon containing

organic matter (Ge et al., 2022). Carbon sequestration in vegetation and soils through land use policy or sustainable agricultural practices (Guenet et al., 2021; Hussain et al., 2021; Rajbanshi and Das, 2021) has therefore attracted attention as one of the possible approaches to climate mitigation. According to Lal (2004) carbon sequestration potential of agricultural soils might lie in the range of 0.4–1.2 Gt of carbon per year, while more recent studies on effects of changes in agricultural practices suggest a range of 0.1–0.68 Gt C yr (Lessmann et al., 2022). Uncertainties notably include insufficient knowledge about the impact of microbial processes, the need for more nutrients (nitrogen (N), phosphorus (P), and sulfur (S)), and water for carbon sequestration (Elbasiouny et al., 2022). A warming climate further jeopardizes storage through vegetation and soils, both directly and through disturbances, such as fires and pests (Lorenz and Lal, 2022; Werner et al., 2024), and recent studies show a concerning weakening of the capacity of land to function as a carbon sink (Ke et al., 2024). Lack of soil organic carbon (SOC) can reduce soil moisture capacity and thereby adversely affect soil fertility. Additionally, the vegetation and soil carbon stocks can become

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saturated and reduce sequestration potential, or even become emission sources (Bai and Cotrufo, 2022; Moinet et al., 2023), while application of biochar on soil surfaces could drive climate heating through albedo effects (Kelleway et al., 2017). Considering that factors such as dynamic climate feedback mechanisms and long-term soil carbon storage potentials are not clearly rendered through the IPCC definition of carbon sequestration, Don et al. (2024) therefore recommend more stringent use of terminology in estimating implications of strategies. Estimates on potentials of biochar in climate mitigation further depend on which biochar uses are considered, which phases in the life-cycle are included, and which alternative options the use of biochar is compared with (Lehmann et al., 2021).

Currently, global soil carbon, in both organic and inorganic forms, is estimated slightly over 4000 picograms (pgC), which is more than five times the amount of carbon currently present in the atmosphere (Habib et al., 2023). Therefore, even a small release, of the order of a per cent, of the amount of carbon present in soils - due, for instance, to a rise in ambient temperature and subsequent stimulation of microbial metabolism - could lead to a marked increase in atmospheric carbon and adversely change the climate (Chabbi et al., 2022). A sustainable sequestration strategy cannot thus solely rely on vegetation-based and soil-mediated solutions.

Pyrolysis biochar (charred feedstocks through pyrolysis, i.e., heating in low oxygen or inert atmosphere) is a carbon-rich material that has been considered as a supplementary solution, due to its stability and positive impacts on vegetation. So-called 'hydrochars' are also obtained from biomass, but are slurry materials (a two-phase mixture of solid and liquid) produced through completely different processes, such as hydrothermal carbonization (HTC) in a water medium, whereas pyrolysis biochars are produced as solid by-product materials through a dry carbonization process of pyrolysis (Devnath et al., 2024).

1.1. Soil carbon storage and biochar as a climate mitigation strategy

It has been estimated that by 2050, CO₂ removal from the atmosphere using biochar might be in the range of 0.3–2 Gt of CO₂ per year (Fawzy et al., 2020). Although the focus in the present review is on carbon, other potent greenhouse gases must not be forgotten, notably nitrous oxide (N₂O) and methane (CH₄) (Hassan et al., 2022; IPCC, 2021b; Lugato et al., 2018; Rosentreter et al., 2021). Reducing emissions of CH₄ in particular, has been discussed as a way to gain time for other types of climate action to be implemented and have impacts (Staniaszek et al., 2022). In the context of climate mitigation, the potential of biochar to deliver emission reductions has been estimated to be in the order of 3.4–6.3 PgCO₂e (Lehmann et al., 2021), while Wang et al. (2023) estimated potentials to 2.56×10^9 t CO₂e per year, corresponding to 5.0 % of global greenhouse gas (GHG) emissions. Examining the effects of biochar on greenhouse gas emissions in upland soils, rice paddies/wetlands, and composting environments, Lyu et al. (2022) found an overall suppression effect on the different emissions, in the order N₂O > CH₄ > CO₂, while a study by Atilano-Camino et al. (2022) found an increase in CO₂ and decrease in N₂O emissions, both impacts correlated to microbial activity. Differing effects on the various GHGs under specific conditions are important to consider, since several biochar applications are in agriculture and waste management. Li et al. (2021) found, for instance, that climate mitigation effects of biochar deposited in soil were dependent on depth.

Pyrolysis biochar is mostly used to remove contaminants from water, soil and air, while additional applications are for improving soil properties, waste management, energy production and climate change mitigation (Kumar et al., 2022; Neogi et al., 2022; Twagirayezu et al., 2024). However, different types of biochar are attracting increasing interest, and in recent years there have been a growing number of non-agricultural applications (Marzbali et al., 2024). Certain biochars can, for instance, be incorporated into building materials, or used to improve the durability of road bitumen, simultaneously reducing the

emission of volatile organic compounds (Pahlavan et al., 2023, 2024). Various studies have further reported that the acidic solid residues of biochar from the extraction process can have alternative applications such as adsorbents, catalysts, nanocomposites, and fillers in construction materials (e.g., cement, porous tiles and ceramics, bricks, or asphalt) (Pahlavan et al., 2023; Mousavi et al., 2023).

With respect to agricultural uses of pyrolysis biochars, many studies suggest that soil can be a suitable medium for biochar disposal, mostly considering crop yield (see for instance Spokas et al., 2012; Lorenz and Lal, 2014). Effects on soil fertility (Pandey et al., 2021) have been attributed to the liming, moistening, and aggregating effects of biochar (Basak et al., 2022). The porous surface of biochar also makes it suitable as carrier for microbial inoculants (Bolan et al., 2023). However, soil pH, P-buffering capacity, clay content, and the levels of iron, aluminum oxide, and lime can affect the uptake of nutrients by plants. These factors also influence the availability for plants and beneficial soil microbes of dissolved P introduced into the soil through the use of biochar. Furthermore, biochar chemical and physical characteristics and feedstock type affect the nutrients released in the soil (El-Naggar et al., 2019). Both positive and negative effects of biochar and its environmental interactions are biochar-, plant-, and site-specific. For example, soils with high intrinsic fertility seem to benefit less from biochar (Farhangi-Abriz et al., 2021). Effects and characteristics of biochar are also highly dependent on feedstock and pyrolysis temperature (Ippolito et al., 2020; Kumar et al., 2022).

Due to the wide variety of biochar properties and largely unknown knowledge about the dynamics and duration of biochar stability in a particular soil, recommendations regarding soil carbon management cannot be generalized (Twagirayezu et al., 2024; Xie et al., 2022). The complexity of soil-biochar dynamics and pollution diffusion pathways notably poses significant challenges in establishing standards for environmental regulations (Sobol et al., 2025; Vieira et al., 2024). Contradictory results in terms of agricultural benefits have also been reported. Approximately 50 % of the 44 studies reviewed by Spokas et al. (2012) indicated positive effects of biochar on plant yield and growth. 30 % had negligible effects, and 20 % had negative effects. The negative effects typically concern nutrients' availability, plant growth, environmental pollution, soil water content, salinity and erosion, plant pesticides, population and diversity of soil microorganisms, soil chemical and physical properties, and soil carbon cycle (Brtnicky et al., 2021). Most of the literature in this area is based on short experiments at laboratory or greenhouse scales, usually using high levels of biochar (Rasse et al., 2017), while Jones et al. (2012) observed that many of the short-term effects of biochar on soil behavior and plant growth that were reported in laboratory studies were not observed in the field. Moreno et al. (2022) argued that biochar addition to soil might only cause small and transient changes in agroecosystem function. Insufficient understanding of the mechanisms of biochar's impact on soil health and organisms (Godlewska et al., 2021), has led to concerns about its long-term and large-scale use in soil (Brtnicky et al., 2021). Moreover, while the addition of biochar to the soil is considered an approach to the net long-term removal of atmospheric CO₂ at an order of above 100 years (Gan et al., 2021), it has been suggested that CO₂ should be stored for much longer periods (> 10,000 years) to mitigate climate change (Mackey et al., 2013; Jenkins et al., 2021).

1.2. Soil health, regulatory environments and differing priorities for investors and users

Soil health is crucial for multiple ecosystem services, including food and nutritional security, water purification and renewability, biodiversity, carbon sequestration, air quality, atmospheric chemistry, and elemental cycling (Tomalka et al., 2024; Wood and Blankinship, 2022;). Yearly costs of soil degradation in the EU are in the order of EUR 50 billion, agricultural run-off is contributing to eutrophication and water pollution, while Southern Europe is exposed to drought (European

Commission, 2020). At global levels, investments in the order of USD 2.6 trillion would be required by 2030 to restore degraded land (UNCCD, 2024). Nevertheless, according to estimates prepared by the United Nations Convention to Combat Desertification ahead of the COP16 Conference of the Parties that took place in Riyadh, Saudi Arabia in December 2024, these investments would be more than recovered by reducing the losses caused by land degradation and desertification.

Achieving soil health is one of the European 2030 missions (European Commission, 2020), aiming to ensure food security, reduce pollution and contribute to climate mitigation. Cultivated lands in the European Union (EU) are currently losing carbon at a rate of 0.5 % per year, while the mission aims to achieve a yearly increase in soil carbon storage of 0.1–0.4 %. Issues of soil health in the EU are for instance addressed in the 2021 Zero Pollution Action Plan. Specific EU regulations regarding biochar are comparatively recent (Štrubelj, 2022), and include Reg. 2021/2088 (on modified fertilizing products) that specifies minimum processing conditions, Reg. 2019/ 2164 (on soil conditioners for organic products). However, Catenacci et al. (2022) anticipate that EU climate neutrality targets by 2050 and objectives in connection with the Circular Green Economy will lead to policy incentives for biochar.

Assessing risks of soil contamination and establishing effective and meaningful regulatory frameworks is challenging (Sobol et al., 2025; Vieira et al., 2024). This is not only due to complex, site-specific dynamics and diffusion pathways, but also to the large number of regulatory levels, categorizations and sectors involved (Štrubelj, 2022). Similar issues affect environmental risk assessments and regulations globally (Antoniadis et al., 2019; Puri et al., 2023). In China, for instance, soil health is regulated within the framework of the so-called ‘Soil Ten’ initiative, and the Action Plans linked to it. Action on soil, air and water pollution is integrated, but insufficient data exists on

regulatory efficacy. Some Chinese studies suggest, however, that emissions of regulated pollutants have decreased, but that local authorities experience difficulties in prioritizing conflicting targets, where economic development is often prioritized. At the same time, surveys suggest that citizens would prioritize a clean and healthy environment (Lyu and Yang, 2024). This points to conflicting priorities and interests, both between policy targets, and between rural or urban populations.

In their analysis of the potentials of biochar as a negative emissions technology from an industry perspective, Fawzy et al. (2021) mainly point to benefits and opportunities, and argue that negative emissions trading should be integrated in the carbon economy. This supposes biochar and production process certification, for instance by the European biochar certificate or the US-based international biochar initiative (IBI) (see also Lehmann et al., 2021). However, Buma et al. (2024) assess cropland biochar amendment as a climate mitigation pathway with existing protocols and market activity that still has relatively high uncertainties, although the mitigation potentials are high. Specifically with respect to agricultural user uptake of biochar, Campion et al. (2023) emphasize the importance of how biochar price compares with anticipated benefits and gains for farmers and land managers. Also, repeated biochar applications may not continue to increase crop yields, thus reducing incentives over time. By contrast, external benefits such as carbon sequestration and reduced nitrate leaching would increase with additional application (Campion et al., 2023).

1.3. Ecotoxicity concerns and impacts on soil microbial communities

Biochar can be applied in agricultural soils in a variety of climates and cultivation systems, but the effects are not always absolute and fully beneficial (Fig. 1). Concerns have also been voiced about the long-term

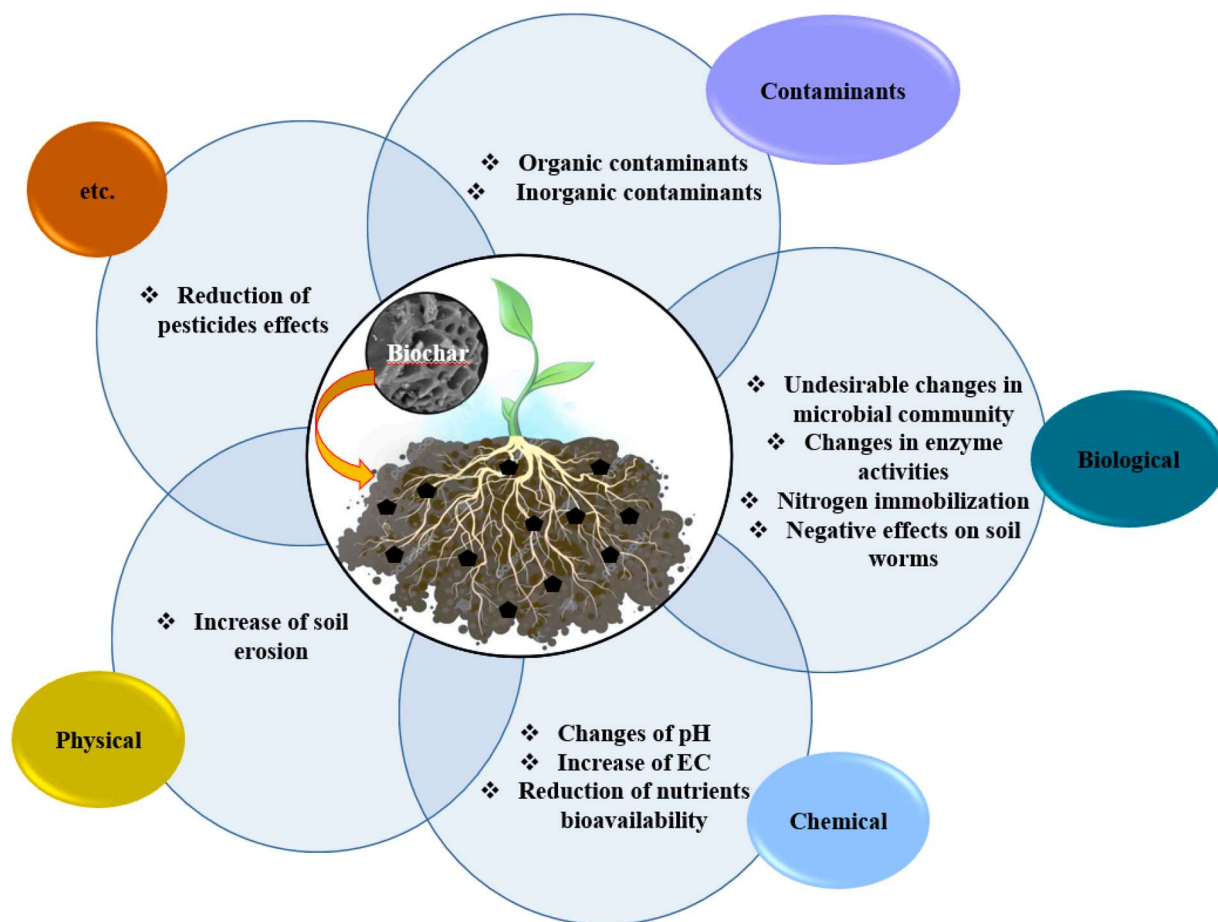


Fig. 1. Different aspects related to effects of biochar in soil.

safety and consequences of large-scale biochar soil application (Abukari et al., 2022; Murtaza et al., 2023; Subedi et al., 2017). Biochar applied and accumulated in the soil (Yang et al., 2021) will be difficult to remove (Lal, 2011), and leaves uncertainties related to known and unknown future negative effects.

Biochar can contain multiple organic and inorganic contaminants that result in toxicity of the environment, affecting health of plants, key invertebrates such as earthworms, microorganisms and arthropods (Godlewska et al., 2021). Hazardous substances can accumulate in soils, affecting livestock and human health (Sobol et al., 2025). In addition to nutrient contents, biochar can, for instance, contain heavy metals, and there are certain risks because the fate of some of the heavy metals is not fully understood (Bridle and Pritchard, 2004). Pyrolysis reduces certain contaminants, while others are generated through the process. These contaminants can exist in the soil for a variable time, indicated by their half-life. The half-life of a compound is influenced by several factors, including the specific compound, soil characteristics, environmental conditions, and microbial activity (Table 1). Frequent application of biochar could thus not only cause excessive accumulations in the soil of toxic contaminants, but intensification of soil erosion could entail a subsequently increased risk that contaminants transfer to water bodies or air, e.g., in the form of fine dust.

It is also crucial to consider possible changes in the diversity and population of microorganisms resulting from biochar application in

soils, as well as the possibility of their mutations or evolutions transferring new traits to the next generations. As a result of toxicity released from various contaminants, biochar can adversely affect soil organisms, for instance by increasing the population of pathogenic microbes and reducing the beneficial soil biota (Mukherjee and Lal, 2014). Furthermore, metals contaminating soils affect the microbial activity of soils amended with biochar (Atilano-Camino et al., 2022). While Xu et al. (2023) observed that biochar significantly promotes the genetic richness and diversity of soil bacteria, Anjum et al. (2014) reported a significant mutagenic potential for cannabis-derived biochar compared to wood and bitumen-derived biochar. Therefore, the specific effects and mechanisms of different types of biochar concerning soil biota must be identified (Godlewska et al., 2021) to avoid irreversible negative outcomes.

Consumption of biochar by soil microbes can lead to changes in soil microbial composition and abundance. These changes can also be caused indirectly due to the physical and chemical changes in the soil after adding biochar (Twagirayezu et al., 2024; Villagra-Mendoza et al., 2021). Changes in microbial biomass may result (Asirifi et al., 2021). The addition of biochar to soil usually increases soil pH and C/N ratio and affects other soil properties, such as water-holding capacity and porosity (Sathe et al., 2021), characteristics that are frequently desired in soil amendment. However, the subsequent changes in available resources (e.g., water, carbon, and nutrients), non-biological agents (e.g., toxic elements), or physical habitat can lead to the dominance of some microbial groups over others and thus alteration of the composition of microbial society (Lehmann et al., 2011). Changes in microbial society and soil properties may in turn affect the biochar decomposition rate. A systemic understanding of such dynamics is largely missing. Although geological records present evidence of decay processes in earlier periods (Denis et al., 2021), as well as indications of permanence (Sanei et al., 2024), we have no historical precedents for many of the conditions existing today, leaving considerable uncertainties regarding aging processes in soils amended with biochar.

Various microbial groups respond differently to biochar. For instance, the addition of biochar to soil increases the abundance of bacterial genes but decreases fungi (Asiloglu et al., 2021; Chen et al., 2021; Twagirayezu et al., 2024). Bacteria, especially gram (+), are known to be important consumers of more resistant soil carbon (Bird et al., 2011) and may consume aromatic carbon in the biochar. It has long been recognized that coal is susceptible to microbial transformation. No study has specifically reported the isolation and characterization of biochar-degrading microorganisms. However, Spokas (2010) reported that microbial communities, especially fungi, are involved in biochar biodegradation, with the primary pathway being through extracellular enzymes. He suggested that due to the different characteristics of biochars, there is a great variation in the estimation of biochar biodegradation. Table 2 represents some of the microorganisms that are capable of dissolving and decomposing coal (see also Singh and Siddique, 2013).

1.4. The rationale for nutrient extraction from biochar for agricultural applications

Against the background of the climate emergency and environmental concerns, a long-lasting carbon sequestration that can avoid many of the concerns regarding biochar soil application, particularly for heavily contaminated feedstock from industries or urban sludge, is carbon geostorage or geosequestration. Dufour (2013) has suggested the concept of biochar geostorage as a true carbon sink and CO₂ mitigation strategy with a higher potential for long-term carbon sequestration than biochar soil application. Dufour (2013) argued that geostorage would address concerns regarding the potential toxicity of biochar and its effects on the microbial community, its erosion and transfer to surface water bodies, and the possibility of its long-term decomposition. Similarly, Osman et al. (2022) proposed disposal of biochar in landfills as a

Table 1
The half-lives and effects of some major organic and inorganic contaminants in biochar.

Biochar contaminants	Organic	Polycyclic aromatic hydrocarbons (PAHs): half-lives in the soil: 1.5 to 4.4 weeks (Roslund et al., 2018). They have carcinogenic, mutagenic, and teratogenic properties and are very harmful to most plants and microbial communities. Dioxins: half-lives in the soil: 60 to 80 years (Nhung et al., 2022). The compounds of dioxins are highly resistant but in small amounts in biochar and formed during the pyrolysis process, particularly in food waste-derived biochar. Volatile organic compounds (VOCs): half-lives in the soil: from hours to years (Jin et al., 1994). VOCs have potentially toxic effects on living organisms, including carcinogens, as well as effects on the respiratory, gastrointestinal, and nervous systems, and also cause phytotoxicity. Perfluorochemicals (PFCs): half-lives in the soil: 1.1 to 2.4 years (Venkatesan and Halden, 2014). These compounds are resistant to chemical and thermal degradation. High concentrations of PFCs are often found in sewage sludge, which is increasingly popular as a feedstock for biochar production (see Catenacci et al., 2022)
	Inorganic	Heavy metals: non-biodegradable and have long half-lives (Thakur et al., 2016). Heavy metals in biochar can be either essential, like iron (Fe), manganese (Mn), zinc (Zn), copper (Cu), nickel (Ni), or non-essential, such as cadmium (Cd), lead (Pb), chromium (Cr), and mercury (Hg). In small amounts, essential heavy metals are essential nutrients, but in large amounts, they can cause toxic effects. Cobalt (Co) can also be found in biochar, and is a heavy metal that is essential for legumes and a beneficial nutrient for other plants. Free radicals (FRs): very short half-lives (10–9 s) (Juan et al., 2021). These compounds produce reactive oxygen species (ROS), which can damage DNA in vitro and cellular structure.

Table 2
Processes involved in coal solubilization and decomposition by microorganisms.

Process	Microorganism	Reference
Solubilization of coal	<i>Streptomyces viridosporus</i>	Strandberg and Lewis (1987)
Biosolubilization of lignite	<i>Streptomyces flavovirens</i>	Moolick et al. (1989)
Biological solubilization of untreated North Dakota lignite	<i>Penicillium</i> sp. <i>Bacillus cereus</i> .	Moolick et al. (1989)
Coal solubilization	<i>Arthrobacter</i> sp..	Torzilli and Isbister (1994)
Depolymerization of low-rank coal by extracellular fungal enzyme systems	<i>Nematoloma frowardii</i>	Hofrichter and Fritsche (1997)
Liquefaction/solubilization of Spanish coals	<i>Aspergillus</i> sp. <i>Penicillium</i> sp. <i>Trichoderma</i> sp.	Laborda et al. (1999)
Decolorization and depolymerization of solubilized low-rank coal	<i>Phanerochaete chrysosporium</i>	Ralph and Catcheside (1994)
Fungal biosolubilization of Rhenish brown coal	<i>Trametes versicolor</i>	Götz and Fakoussa (1999)
Microbial degradation of low-rank coals.	<i>Pseudomonas putida</i>	Machnikowska et al. (2002)
Bioliqefaction/solubilization of low-rank Turkish lignites	<i>Coriolus versicolor</i>	Başaran et al. (2003)
Indian coal solubilization	<i>Pseudomonas stutzeri</i>	Singh and Tripathi (2013)
Fungal biodegradation of hard coal	<i>Neosartorya fischer</i>	Igbinigie et al. (2008), Sekhohola et al. (2014), and Sekhohola and Cowan (2017)
Biogenic oxidation of sub-bituminous coal	Oxidation of sub-bituminous coal by <i>Pseudomonas fluorescens</i>	Hazrin-Chong et al. (2014)
Biosolubilization of Polish brown coal	The solubilization of brown coal by <i>Gordonia alkanivorans</i> and <i>Bacillus mycoides</i>	Romanowska et al. (2015)

long-term storage reservoir for sustainable carbon sequestration purposes. Thakkar et al. (2016) performed a techno-economic assessment of straw-derived biochar and sequestering the carbon through landfilling. They evaluated the cost of production and landfilling of biochar, as well as the greenhouse gases emissions in this pathway, and reported that the cost of carbon sequestration is higher than the biomass-based electricity generation pathway but lower than some estimates of carbon capture and storage technologies for carbon sequestration. There are significant debates concerning potential risks of geostorage more generally, but studies are lacking on risks that might be associated with biochar geostorage in particular. Further research on potential issues is therefore needed.

Biochar contains considerable amounts of plant nutrients useful in agricultural fertilization, such as P, N, potassium (K), and calcium (Ca) (Wise et al., 2012). Pyrolysis can significantly reduce the mobility of inorganic contaminants and reduce or eliminate organic or biological contaminants, while the concentration of nutrients significantly increases relative to feedstock type (Frišták et al., 2018). The natural sources of some of these nutrients, such as P, are scarce, and human activities and agricultural practices contribute to depletion of P in soils (Huang et al., 2024). On the other hand, the production of chemical fertilizers often involves unsustainable practices with high environmental costs, such as extensive reliance on the mining of phosphate rocks (Jupp et al., 2021). Since geostorage of biochars or their deposit in deep soil layers as part of a carbon sequestration strategy would make valuable nutrients such as P unavailable to plants and soil microbiota, methods would need to be developed to avoid wasting these resources. Nutrient recovery from biochar could at the same time eliminate the

risks of heavy metals and other pollutants (Zhu et al., 2022) that would arise if biochar containing hazardous contaminants is deposited directly in soils.

Nutrient recovery usually relies on chemical methods to extract nutrients from biochar for subsequent agricultural use (Tang et al., 2018). It is different from nutrient recycling in the environment, where biochar is directly applied as a soil fertilizer (Kleemann et al., 2017), resulting in a gradual, but non-regulated, release of nutrients in the soil. It should be noted, however, that in many cases such gradual release may nevertheless be preferable (Bindraban et al., 2015; Dimkpa et al., 2023) rather than soil degradation and nutrient leaching resulting from the use of chemical fertilizers, with impacts such as eutrophication of water bodies (Ge et al., 2023).

Studies regarding nutrient recovery from biochar is a relatively new area of research, still in its early stages. The field therefore both necessitates additional studies, and broader reviews for synthesis, to map interconnecting areas, as well as to identify gaps and understudied areas. From our own scanning of the field, it appears that most studies on nutrient recovery from biochar to date have focused on recovery from waste-derived ash, particularly sewage sludge ash, which has been reviewed extensively in literature (Tan and Lagerkvist, 2011; Guedes et al., 2014; Jama-Rodzeńska et al., 2021). For the present systematic review of the literature, we therefore chose to focus on methods to directly recover nutrients from pyrolysis biochar to support further work in this area.

2. Materials and methods

This paper reviews studies on nutrient recovery from biochar, which is of particular interest in cases when the biochar is intended for geostorage. Studies were identified using a hybrid approach (Mourão et al., 2020), combining a database search with supplementary snowballing.

2.1. Search strategy

The initial systematic search for publications was based on the established guidelines, including the Preferred Reporting Items for Systematic Reviews and the Meta-Analyses (PRISMA) method. The primary identification of the studies was based on the Web of Science Core Collection database, where a search phrase was defined to find a comprehensive set of studies related to biochar nutrient recovery. The search phrase comprised three necessary parts representing biochar (as a single word), nutrient (or, at least, one among the names of some macronutrients found in biochar, e.g., N, P, etc.), and recovery (or some synonyms/changed forms for it, e.g., retrieve). Accordingly, our search was set as follows:

TS= (biochar AND (nutrient OR phosph* OR P OR potass* OR K OR nitr* OR N OR calci* OR Ca OR magnes* OR Mg) AND (recover* OR reclaim* OR reclamation OR retriev*))

The use of “biochar” and “nutrient” without any wildcard in them did not avoid results of other slightly different forms, such as “bio-char” and “nutrients” by default.

The last date of searching in Web of Science was 2024/02/11, but no limit was set for the beginning of the search period covered. The search was in the “Topic” field, which includes Title, Abstract, Author Keywords, and the so-called Plus Keywords, considering “words or phrases that frequently appear in the titles of an article’s references but do not appear in the title of the article itself.”.

2.2. Screening and supplemental searches

The results (n = 2153) were exported in a CSV file (including Abstract, Title, etc.) and uploaded to the Rayyan web-based application for screening and systematic review. The studies were first labeled as Included, Maybe, or Excluded, based on reading titles and abstracts. Studies that did not focus on direct recovery of nutrients for agricultural

use were excluded (see Appendix 1). Then, full-text results labeled Included or Maybe were read ($n = 131$), which resulted in excluding a further 72 studies that concerned nutrient recycling through biochar application in soil, 23 studies on biochar functioning as a nutrients carrier by surface adsorption, 2 articles on physicochemical characteristics of biochar and nutrients leaching in biochar treatment soils, 13 articles on hydrochar, and 3 articles on extraction of useful molecules such as 2,3-Butanediol and determination of polycyclic aromatic hydrocarbons. Among the remaining 18 articles, there were 5 review articles that were also excluded. The review articles studied the change of P species during HTC (Chen et al., 2022; Huang et al., 2017), the fate of P and N during HTC (Aragón-Briceño et al., 2021), evaluation of thermal treatment of sewage sludge for P recovery (Zhu et al., 2022), and pyrolysis of biological waste with high P for application in agriculture (Sun et al., 2018). Of these, one systematic review article concerned P-rich biochar recycling (Chen et al., 2022).

Finally, the included studies from the systematic database search were supplemented with additional studies based on backward and forward snowballing (Mourão et al., 2020). The backward snowballing could theoretically find results anywhere, including within or outside the Web of Science databases. The forward snowballing was conducted using Google Scholar to increase the chance of finding additional relevant publications. Snowballing from the 13 studies that had been included after full-text screening led to the inclusion of 3 additional studies, which, together with 1 extra study identified based on the authors' previous knowledge, resulted in four additional studies that were not a direct result of our Web of Science database search. The search process is summarized in Fig. 2.

Notably, none of the 17 finally included studies were review articles. Therefore, the present study appears to be the first comprehensive review that focuses on direct nutrient recovery from biochar. The 17

included articles are discussed in the results below. The methods used for the recovery of nutrients from biochar and their resultant recovery percentage are summarized in Table 3, as well as other details such as the feedstock used in biochar. Due to the higher importance of P recovery compared to the other nutrients in biochar and the availability of more studies on the topic, studies on P recovery and recovery of other nutrients are presented in two separate sections.

3. Results and discussion

3.1. Phosphorus recovery from biochar

Wet chemical extraction (acidic or alkaline extraction) of P from biowastes-derived biochar has been a research focus for P recovery over the past decades (Azuara et al., 2013; Tiwari et al., 2024). This process can even enable the extraction of 100% P from biochar (Salimbeni et al., 2023). In addition, the solid residue after pyrolysis coupling with acid washing can result in a low environmental risk for subsequent resource utilization (the values of the potential ecological risk index < 20) (Li et al., 2023). Topcu et al. (2022) reported that the efficiency of P recovery from biochar was approximately 50 % of P in the feedstock. They attributed the low recovery of P to the formation of soluble P salts in the alkaline precipitation stage. Uchimiya and Hiradate (2014) extracted 92 % of P from biochar with ammonium oxalate. Their results indicated that feedstock type and pyrolysis temperature have a great effect on P species in biochar. The dominant type of P in animal manure biochar at a temperature of more than 650 °C was amorphous calcium phosphate (soluble), while plant residue biochar at a temperature of 350–500 °C resulted in strong pyrophosphate complexes (insoluble).

Some studies confirm these results and the importance of feedstock type and pyrolysis temperature for biochar P recovery. Li et al. (2018)

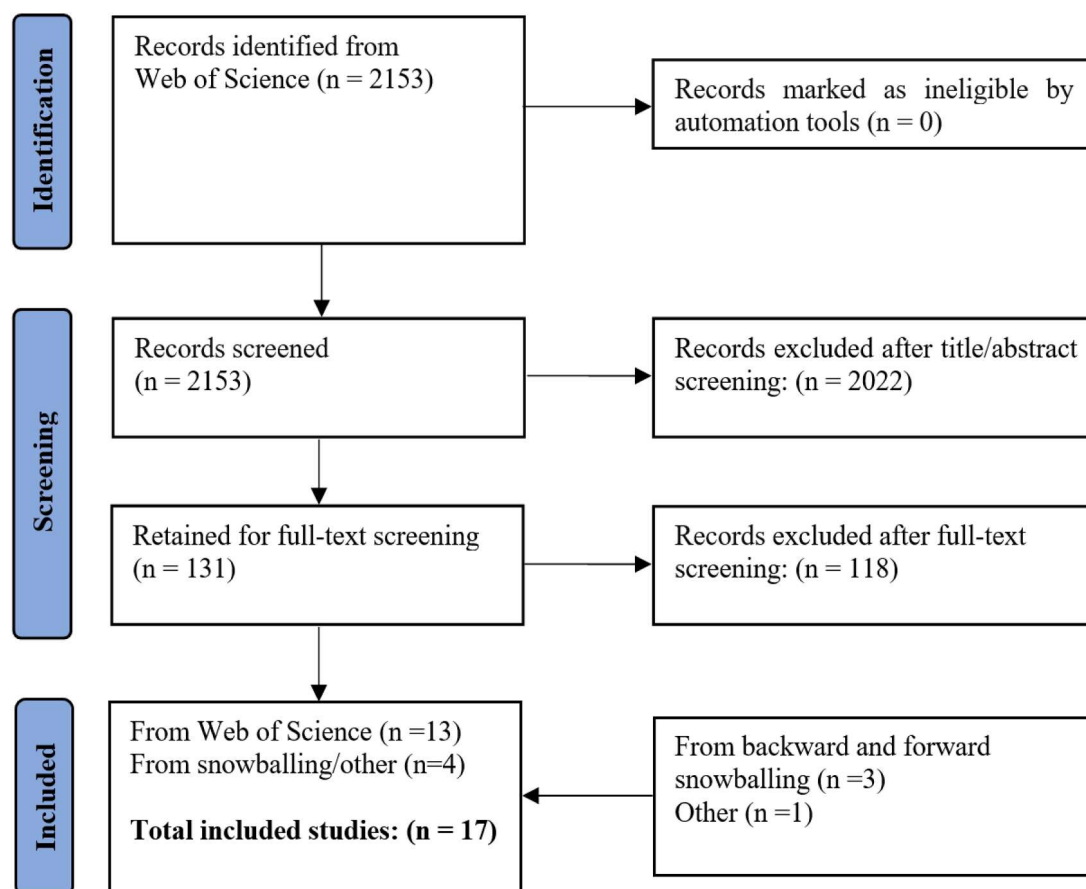


Fig. 2. Prisma diagram summarizing the search process.

Table 3

List of included studies from the systematic review, summarizing key details about the used nutrient recovery methods.

No.	Feedstock	Pyrolysis conditions	(Best) extraction conditions	Precipitation conditions	Production	Nutrient recovery (%)	Reference
1	Broiler litter	Temperature, held, atmosphere: 650 °C, 1 h, N ₂	0.5 g of sample was shaken in 25 mL of 0.2 M ammonium oxalate at pH 3.5 for 4 h.	–	P in the extraction solution	$P = 92$	Uchimiya and Hiradate (2014)*
2	Pig manure	Fast pyrolysis was done at 400, 500, and 600 °C.	P extraction was studied by using 100 mL of 200 mmol L ⁻¹ sulfuric acid after 2 h of stirring at room temperature.	–	P in the extraction solution	$P = 60\text{--}75$	Azuara et al. (2013)**
3	Sewage sludge	Temperature, held: 850 °C, 2 min	H ₂ SO ₄ : 0.8 M liquid/solid ratio of 10 with 30 min contact time	–	P in the extraction solution	$P = 90$	Kleemann et al. (2017)***
4	Sewage sludge	Pyrolysis at three temperatures of 400, 600, and 900 °C	P was extracted from the char using an alkali.	–	P in extraction solutions	The highest alkaline P recovery efficiency was 88 % from the amended sewage sludge char.	Tiwari et al. (2024)***
5	Hardwood <i>Acer pseudoplatanus</i>	Temperature, held: 500 °C, 24 h	Particle size: 0.15–0.60 mm, 0.60–1.18 mm, 1.18–4.00 mm, and >4.00 mm, Deionized water: biochar 20:1 shaken for 4 h at 20 °C	–	P, K, and Mg solution	$P = 30\text{--}103$ $Mg = 6\text{--}27$ $K = 82\text{--}122$	Angst and Sohi (2013)
6	Poultry litter	Temperature, heating rate: 450 °C, 20 °C min ⁻¹	1 M HCl at 1:50 solid/solution ratio under room temperature rotary shaking for 24 h,	–	P in acidic solution	$P = 99.75$	Li et al. (2018)
7	Tobacco stems	Temperature, heating rate, held, atmosphere: 300 °C, 2 h, N ₂	10 g of biochar in the presence of 300 mL deionized water. These were placed on a magnetic stirrer at room temperature, which was adjusted to pH 1, 2, and 3 with 15 % HNO ₃ in the beakers.	Separation of extractable K and metals. 50 mL filtrates were put into new beakers on a magnetic stirrer at room temperature, using a 40 % KOH solution to adjust the pH to 9, 10, 11, 12, 13, and 14, with three replications performed.	P in alkaline solution	$P = 145$	Yang et al. (2019)
78	Manure-wood mixture	Slow pyrolysis in a 5 L gas-fired rotating drum kiln, temperature: 250, 350, 450, and 550 °C. Heating rate: 15–20 °C min ⁻¹	0.35 g of biochars were shaken with 35 mL of 2 % formic acid on an end-over-end shaker at 32 rpm for 30 min at room temperature after dispersing with ultrasonication for 10 min.	–	P in extraction solution	$P = 88.99$	Wang et al. (2012)
	Biosolid-wood mixture	Slow pyrolysis in a 5 L gas-fired rotating drum kiln, temperature: 250, 350, 450, and 550 °C. Heating rate: 15–20 °C min ⁻¹	0.35 g of biochars were shaken with a 35 mL of 1 M neutral ammonium citrate-EDTA solution for 24 h at room temperature.	–	P in extraction solution	$P = 89.86$	
9	Poultry manure	Temperature, heating rate, held, atmosphere: 300/400/500 °C, 10 °C min ⁻¹ , 1 h, N ₂	H ₂ SO ₄ : 0.1, 0.3, and 0.5 M, liquid/solid: 50- and 100 mL g ⁻¹ , contact times: 2 and 4 h	NaOH: 1 M, pH = 8, age: 4 h	Amorphous calcium phosphate	$P = 48.5\text{--}57.3$	Topcu et al. (2022)
10	Activated sludge	N ₂ flow (~1 mL sec ⁻¹) at 250, 350, 450, and 600 °C with a heating and cooling rate of 200 °C h ⁻¹ and a soaking duration of 4 h. The anaerobic sludge was treated at only 450 °C.	0.25 M NaOH and 0.05 M EDTA at a solid-liquid ratio of 0.2:4 g mL ⁻¹ and was shaken at 150 rpm for 16 h at room temperature.	Samples were combusted at 600 °C for 2 h in a furnace, followed by an HCl extraction (1 M) for 16 h.	P in extraction solution	Pyrophosphate = 50–70	Huang and Tang (2015)
11	Sewage sludge	Pyrolysis process at 400 °C	P was sequentially extracted by water, 0.5 M NaHCO ₃ , 0.1 M NaOH, and 1 M HCl solution at the solid-to-liquid volume ratio of 1:100. The shaking speed, reaction temperature, and reaction time were respectively 200 rpm, 25 ± 3 °C, and 16 h.	–	P in extraction solutions	$P = 90$	Qian et al. (2019)
12	Sewage sludge algae shrimp shell bone dreg	500 °C at a heating rate of 15 °C min ⁻¹ ; they were held for 2 h in a flowing Ar atmosphere with a flowing rate of 150 mL min ⁻¹ .	–	–		Conversion of protein-N (70–90 % in biowaste) into char-N (25–35 %), oil-N	Nan et al. (2020)

(continued on next page)

Table 3 (continued)

No.	Feedstock	Pyrolysis conditions	(Best) extraction conditions	Precipitation conditions	Production	Nutrient recovery (%)	Reference
13	Wastewater sludge	2.5 h at 400 °C	Mehlich 3-P	–	P in extraction solutions	(15–30 %), and gas-N (30–60 %) $P = 2-59$	Freitas et al. (2023)
14	Sewage sludge	350 °C, for 2 h under a N ₂ flow of 200–300 mL min ⁻¹ .	P extractable with Olsen extractant	–	P in extraction solutions	$P = 77.7$	Yin et al. (2021)
15	Sorghum, switchgrass, and corn stover	Pyrolysis using a fixed-bed pyrolyzer Pyrolysis using bench-scale fluidized-bed reactor	Sulfuric acid digest	–	–	–	Wise et al. (2012)
16	Sewage sludge	The pyrolysis experiment was conducted in a fixed-bed apparatus under N ₂ flow at different temperatures between 350 and 750 °C.	Optimal washing specifications by H ₃ PO ₄ (acid concentration of 2.47 mol L ⁻¹ , L/S of 9.85 mL.g ⁻¹ , and a washing temperature of 71.18 °C)	–	Elements including Cu, Zn, Cr, and Ni in extractant solution	The total maximum element removal efficiency was 95.05 % under the optimal washing specifications by H ₃ PO ₄ .	Li et al. (2023)
17	Sewage sludge	In a 3 kg h ⁻¹ slow pyrolysis reactor at 450 °C for 30 min	The char was processed by chemical leaching with acid (HCl, HNO ₃) and alkali (KOH) reagents to extract inorganic compounds.	–	The leaching tests enabled us to extract P, Mg, and Ca from the char.	The leaching tests enabled us to extract 100 % of P, Mg, and Ca from the char.	Salimbeni et al. (2023)

* The study was not a result of systematic review, and it was included here based on the authors' previous knowledge.

** The study was a result of backward snowballing.

*** The studies were results of forward snowballing.

reported that the pyrolysis of poultry litter in the range of 400 to 450 °C converts almost all polyphosphates and organic P into orthophosphates; however, P extracted by 1 M HCl is significantly reduced at above 450 °C owing to the recalcitrant of inorganic P and its conversion to hydroxyapatite and oxyapatite. Hence, pyrolysis temperature should be taken notice of when producing biochar to save energy for P extraction from biochar. Wang et al. (2012) produced biochar using two feedstocks (manure-wood mixture and biosolid-wood mixture) at four pyrolysis temperatures (250, 350, 450, and 550 °C). Their results showed that P in the feedstock remained completely in the biochar, and its concentration increased with increasing pyrolysis temperature. With the increase of temperature from 250 to 550 °C, the concentration of total P in biochar derived from the biosolid-wood mixture and manure-wood mixture increased 2.4 and 2.9 times, respectively. Besides, the concentration of total P in biosolid-wood mixture biochar was almost 6 times that of manure-wood mixture biochar. These results indicated that feedstock and pyrolysis temperature should be considered to advance P recovery goals.

Wang et al. (2012) showed that manure-wood mixture biochar (prepared at 450 °C) had the highest content of P (7.09 g kg⁻¹ and 88.99 % of total P) extracted by 2 % formic acid, while biosolid-wood mixture biochar (prepared at a temperature of 550 °C) indicated the highest content of P using 1 M neutral ammonium citrate extractant (45.51 g kg⁻¹ and 89.86 % of total P). These results show that extracted P from different types of biochar can be different based on the extraction method. Huang and Tang (2015) reported that P speciation in raw activated sludge was dominated by orthophosphate and long-chain polyphosphates, while increasing amounts of pyrophosphate and short-chain polyphosphates were observed after pyrolysis at 250–600 °C. In addition to the molecular changes of P compounds, pyrolysis changed the physical state and the extractable speciation of P in biochar, so that with the increase in pyrolysis temperature, both total P (combustion/HCl extraction) and polyphosphates (extractable with NaOH/EDTA) were reduced. The results of this study show that P speciation and availability in sludge-derived biochar can be governed by pyrolysis techniques and conditions.

Qian et al. (2019) reported that increasing the pyrolysis temperature from 400 to 700 °C caused a significant decrease in the extractable P with 1 N HCl, as the remaining P increased by approximately 45 %. Their results showed that the diversity of P compounds in biochar, the

number of iron/aluminum-containing compounds, and the biochar functional groups are influential factors in P recovery. Also, the XPS (X-ray photoelectron spectroscopy) analyses showed that metaphosphate and P associated with C bridged by O (C—O—P) were the main species in biochar (Qian et al., 2019). Therefore, the degree of polymerization (chain length) of metaphosphate, the stability of the metaphosphate structure, as well as the stability of organic matter related to P through the C—O—P state in biochar, play a main role in the stability and release rate of P from biochar.

Through comparative acid leaching experiments and characterization of sewage sludge biochar and sewage sludge ash, it was found that Fe is more easily leached in sewage sludge biochar (Kleemann et al., 2017). This may be due to the lack of oxygen during the pyrolysis process, which did not lead to the formation of acid-insoluble P and ultimately led to the final extraction of more P from sewage sludge biochar. Another reason was the higher amount of whitlockite (an unusual form of calcium phosphate) in this biochar. Qian et al. (2019) report that sludge-derived biochar contains more Ca-P bonds compared to sewage sludge ash. Kleemann et al. (2017) also reported that heavy metals were less soluble in sewage sludge biochar than in sewage sludge ash in an acid-leaching experiment because they were more strongly bound to the particles. Since biochar production also aims at carbon sequestration, it may be desirable to recover P from sewage sludge biochar instead of ash, and methods used to recover P from ash may prove useful to recover P from biochar as well.

3.2. Other nutrients recovery from biochar

The recovery of other nutrients from biochar has received less attention. However, some studies have investigated nutrient species changes, loss upon pyrolysis, and their simultaneous extraction with P. Wise et al. (2012) showed that the retention of P, K, Ca, and magnesium (Mg) nutrients in biochar depends on the reactor design, pyrolysis conditions, and feedstock. Therefore, a higher amount of carbohydrate content in the feedstock results in a higher P concentration in the biochar. In addition, older plants used for biochar production have higher P concentrations in biochar. This is why Ca, Mg, and K did not show a significant change in this regard. Wise et al. (2012) also reported that sorghum-derived biochar contained 90–93 % of feedstock P, while it was almost 50 % in corn stover and rice straw-derived biochar. The content

of K, Ca, and Mg in biochar of these three feedstocks was 4.8–30.2, 40.7–69, and 44–63.3 %, respectively. Their results indicated that the cause of the loss of nutrients during the pyrolysis process is through the gas phase or due to "slagging" in the reactor piping. For example, K losses can result from the evaporation of K into KCl, (KCl)₂, K₂SO₄, and KOH during pyrolysis or ashing. Furthermore, if the flow of nitrogen gas (N₂), argon (Ar), or similar gas is at a high flow rate and the produced gas is removed from the reactor, it will contribute to K loss. Other reasons for the loss of nutrients during the pyrolysis process were the lack of catalyst anchoring sites, such as carboxyl and hydroxyl groups, which are important to prevent the loss of K during pyrolysis and gasification. Moreover, high oxygen inside the reactor, the cooling effect of water vapor on solid by-products, the low Si/K ratio of feedstock, and the high humidity of feedstock have a negative effect on the concentration of nutrients in biochar. Additionally, the physiological maturity of feedstock, general characteristics of feedstock nutrients, growing conditions of feedstock and drying, grinding, and general handling of feedstock before pyrolysis are factors that can affect the concentration of nutrients in biochar.

Angst and Sohi (2013) reported that the preparation of *Acer pseudoplatanus* hardwood-derived biochar made the concentrations of P, K, and Mg four, two, and two times that of the feedstock, respectively. They extracted these three nutrients from biochar at six stages with deionized water continuously. Their results showed that reducing the size of biochar particles increased the extracted K in the first stage. Also, K quickly declined from biochar, so the amount of K in the last stage of extraction was only 6–18 % of the amount of K extracted in the first stage. Angst and Sohi (2013) showed that the dissolution of biochar P in water was slow and continuous, but the quantity was small. The size of biochar particles had a significant effect on the amount of dissolved P, so the release of P from biochar with a particle size of 0.15–0.6 mm was twice that with a size larger than 4 mm. The release of Mg was also continuous like P, but with a small amount. Their results showed that the pyrolysis process may produce slightly soluble Mg salts in water, but P and K will completely dissolve in water.

Nan et al. (2020) reported that 25–35 % of protein-N in biowaste is converted to biochar N (prepared at 500 °C), and the rest is lost in oil or gas. This fraction of N was mainly present as pyrrole-N and pyridine-N forms with the breakdown of protein-N. Freitas et al. (2023) reported that thermal treatment during the conversion of biosolids to biochar caused a decrease and an increase in the N and P content of the biochar, respectively, and the conversion of P into less soluble forms even among biosolids with diverse mineralogical composition. Yin et al. (2021) reported that combined pyrolysis of sewage sludge and organic additives, including reed straw, brewers' spent grain, and sawdust, resulted in a significant increase in biochar carbon compared to sewage sludge biochar alone, but the concentration of total nutrients decreased. However, the concentration of P (extractable with Olsen extractant), N (extractable with KCl extractant), K, and iron increased. They reported that this increase in the number of available elements was due to salt formation. The solution of nutrients was linked with functional groups and the increase of the specific surface of biochar. These results showed that the preparation of biochar with the aim of carbon sequestration and nutrient recovery simultaneously can benefit from such strategies to achieve goals such as increasing the carbon content and the content of extractable nutrients at a lower cost.

Yang et al. (2019) reported that the pH of the extractant solution had a significant effect on the extraction of nutrients from biochar, and the extractant solution with pH = 1 was suitable. They showed that increasing the pH of the extracted solution up to 11 led to the removal of heavy metals such as Fe, Al, Cu, Zn, Cd, and Pb, while there were some amounts of Ca, Mg, and P in the solution. However, the concentration of K in this solution was significantly high. They suggested this method for the preparation of K fertilizer.

None of the articles that were studied in this review used the extracted P for plant growth, and it was limited to the characterization of

precipitates or merely the solubility of P in the extracted solution. Therefore, greenhouse and field studies for different crops and soils are necessary to understand the efficiency of the extracted P. More studies should be done to extract all P from biochar and convert it into solid fertilizers. Perhaps it is better to consider elements such as N, K, Ca, Mg, etc., which can be extracted simultaneously with P, to form P precipitation. This may be economically important. Also, the remaining biochar after element extraction can be used to produce activated carbon (Topcu et al., 2022), building materials (Legan et al., 2022), and fuel (Oliver-Tomas et al., 2019), along with biochar geostorage as a larger scale approach to carbon sequestration.

4. Conclusions and recommendations

Biochar as a carbonaceous material obtained from pyrolysis of biomass could be a significant CO₂ mitigation strategy. While there is also a growing number of studies recommending biochar utilization for soil amendment, remediation, and other agricultural applications, the gaps of knowledge about long-term and large-scale mechanisms of some known negative effects of biochar in soil cannot be ignored. To better understand the mechanisms involved in the direct and indirect toxicity of biochar addition to soil, research efforts should be focused on determining specific parameters related to biochar toxicity, including evaluation of the available fraction of contaminants over time. More ecotoxicological experiments in soil-biochar mixtures need to be conducted, especially at long intervals and in field conditions, while the diversity of test organisms used to evaluate toxic effects of biochar must be enhanced. Further areas that require additional research include determining the erosion potential and transfer of biochar from soils to water and its possible adverse effects on aquatic ecosystems. The potential of biochar to be converted to dust blown with wind also merits closer investigation in view of potential adverse impacts on human health and other living organisms, including plant respiration and photosynthesis. Studies regarding nutrient recovery from pyrolysis biochar are scarce. Also, geostorage efforts have been more extensive in areas such as CH₄, CO₂ and wood. Consequently, there is a significant gap in the literature regarding nutrient recovery from biochar, as well as experimental data, on-site research, or modeling on biochar geostorage and its subsequent implications.

Biochar can contain significant amounts of some plant nutrients, such as P, that are naturally limited and non-renewable. In cases where geostorage of biochar is contemplated as a carbon sequestration option, the question of effective, economically viable, and environmentally safe methods for nutrient recovery requires attention. The present review suggests that several factors affect the recovery of nutrients from pyrolysis biochar, including the feedstock type, pyrolysis temperature, and environmental gases (such as N₂, Ar, and CO₂), which must be considered for optimal recovery of nutrients. Research in the area is under development, but more studies and meta-analysis of their results are necessary to lower uncertainties of the optimum extractable nutrient content calculations for different feedstock types, while enhancing energy use efficiency and environmental measures.

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Mohammad Reza Maghsoodi: Writing – review & editing, Writing – original draft, Visualization, Project administration, Conceptualization. **Hasan Hosseini:** Writing – review & editing, Writing – original draft, Software, Methodology, Data curation, Conceptualization. **Larissa Ghodszad:** Writing – review & editing, Writing – original draft. **Helen Avery:** Writing – review & editing, Writing – original draft,

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Declaration of competing interest

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Appendix I. Exclusions based on screening titles and abstracts

The abstract and title screening involved a quick initial assessment to determine whether a study broadly fits the inclusion criteria or not depending on the mention and/or relevance of the extraction/recovery of one or more nutrients from biochar before its utilization or storage. Almost half of the excluded studies based on the abstract review focused on the potential of biochar and modified biochar applications in synthetic or natural aquatic systems as sorbents for organic and inorganic pollutants, including compounds and/or elements such as lead (Pb), Zinc (Zn), cadmium (Cd), Arsenic (As), chromium (Cr), manganese (Mn), copper (Cu), nickel (Ni), vanadium (V), silver (Ag), Mercury (Hg), uranium (U), thorium (Th), rhenium (Re), technetium (Tc), phosphorus (P), nitrogen (N), methylene blue, tetracycline, naphthalene, dyes, dimethyl sulfide, and phenols. 20 % of the excluded studies examined the effects of biochar on plant growth, as well as the physical, chemical, and biological characteristics of various soils, particularly saline and sodic soils, and heavy metal-contaminated soils. Additionally, 2 % of the excluded studies focused on investigating biochar production methods and the effect of various factors on biochar physical and chemical properties without relevance/mention to nutrient recovery potential. Almost 25 % of the excluded studies addressed various topics, including hydrochar, biochar applications as a catalyst, co-composting, energy recovery, additives in digestion systems, and uses in the electrochemical field. The remaining 3 % of the excluded studies were general review papers about biochar or specialized in some of the above topics. It is noted that while the abstracts were primarily screened by the first author, a sample including 25 % of the abstracts were screened by other authors to confirm a comparable rate of exclusions by different authors.

Data availability

No data was used for the research described in the article.

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