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Review

An Overview of Soil Pollution and Remediation Strategies in Coal Mining Regions

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Abstract: Coal remains a very important source of energy for the global economy. Surface and underground coal mining are the two major methods of coal extraction, and both have benefits and drawbacks. Surface coal mining can have a variety of environmental impacts including ecosystem losses, landscape alteration, soil destruction, and changes to surface and groundwater quality and quantity. In addition, toxic compounds such as heavy metals, radioactive elements, polycyclic aromatic hydrocarbons (PAHs), and other organic contaminants are released in the environment, ultimately affecting the health of ecosystems and the general population. Underground mining has large impacts on underground water supplies and water quality, but generally has less visual surface impacts such as leaving waste and tailings on the surface and subsidence problems. In response to the concern about these environmental issues, many strategies have been developed by scientists and practitioners to minimize land degradation and soil pollution due to mining. Reclamation laws passed in numerous countries during the past 50 years have instituted practices to reduce the impacts of soil pollution including burying toxic materials, saving and replacing topsoil, and vegetating the land surface. While modern mining practices have decreased the environmental impacts, many sites are inadequately reclaimed and present long-lasting soil pollution problems. The current review summarizes progress in comprehending (1) coal mining impacts on soil pollution, (2) the potential risks of soil pollution associated with coal mining, and (3) different types of strategies for remediating these contaminated soils. Research and prospective directions of soil pollution in coal mining regions include refinements in assessing pollutant levels, the use of biochars and other amendments, phytoremediation of contaminated soils, and the release of toxic elements such as mercury and thallium.

Keywords: coal mining; soil pollution; heavy metals; PAHs; remediation methods

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1. Introduction

Coal is the second-most-significant global energy source, contributing 27.1% to the main energy generation [1]. At the beginning of the 18th Century, coal became the main energy source, eventually leading to the Industrial Revolution. From the late Eighteenth Century onwards, the techniques of underground mining were developed, mainly in Britain [2]. Despite its positive impact on human development, coal mining has had adverse effects on the environment. Its activities have resulted in significant waste generation (tailings) and long-term environmental destruction, such as polluting the water systems, soil erosion, mine subsidence, ecosystem and biodiversity destruction, and land surface disruption [3]. Furthermore, mining has resulted in significant societal costs, including fatal accidents, health risks, and community displacement [4].

Historic mining practices have caused various hazards to the environment and humans. Some of those traditional practices are continued today in many parts of the world. In industrialized cultures, mining and reclamation laws have mitigated the degradation by stronger environmental requirements and requiring reclamation once mining has ended. Current mining and reclamation standards regulate site preparation, blasting, road building, overburden removal and placement, water control and maintenance, topsoil replacement, and vegetation establishment. Together, these practices greatly reduce the potential for land and water degradation during and after mining [5]. The largest problem in many countries is reclaiming sites that were abandoned and thus remain exposed to continued erosion and degradation of the surrounding environment. This problem often occurs where funding is unavailable for reclaiming abandoned lands or where laws have not been passed to deal with these abandoned sites. This paper deals with pollution in areas caused by wastes and tailings from surface and underground mining that have been left on the surface.

Coal mining can be classified into two types: surface and underground mining. Large quantities of waste are generated in either case due to the coalification process and other impacts associated with mining and processing technologies [6]. Based on the geological and mining circumstances, a total of 0.4–0.7 t of waste is produced for every tonne of coal that is mined [7]. Compared to underground mines, opencast (surface) coal mines have higher environmental impacts on the soil, water, and air, whereas underground mining entails the risk of land subsidence and groundwater impacts below the land [8]. Heavy metals from dumps of spoil or refuse can be mobilized by surface runoff, which can leach into the subsurface soil or into nearby water resources [9]. The disturbed areas in coal mining regions can, therefore, be environmentally compromised, exposing organisms in the food chain to pollutants by direct contact, along with the inhalation of toxic metals, which can translate into human health effects [10].

Soil contamination studies conducted in coal mining areas have shown that mining and the associated operations release toxic elements into the soil, which then affect nearby riparian environments, surface water, and farmland [11,12]. The elevated levels of heavy metals in materials left on the surface pose a potential risk to the environment and human health, because of their bioaccumulation, toxicity, and release to organisms in the environment [13,14]. This elevated metal pollution may be absorbed by plants and crops, which leads to inferior growth and plant tissue contamination [15]. Polycyclic aromatic hydrocarbons (PAHs) are the primary organic toxic components found in coal mining areas [16]. The emissions from the combustion of small coal particles and coal gangue (also called coal refuse or coal waste) are the primary sources of PAHs in coal mine districts. Depositional environments and coal rank are two primary factors that determine the PAH content in coal gangues [17]. Excessive deposition of coal gangue on the surface can lead to spontaneous combustion under the right conditions, resulting in atmospheric pollution. At the same time, leaching due to precipitation events will result in heavy metals and PAHs contained in the gauge being released into the soil and surface water [18].

Coal has been used for millennia as a fuel source. Archaeologists discovered in 2005 what appeared to be a 120,000-year-old Stone Age opencast coal mine in Germany. According to the China Coal Information Institute, coal in China was utilized as a fuel source during the Neolithic Era, approximately 10,000 years ago. In the Americas, the Aztecs were the first people to use coal. For Aztecs, coal was employed not just as a heating source, but also as a decorative element. In the 18th Century, colonists in eastern North America discovered coal reserves. Worldwide coal reserves are estimated at over 1.1 trillion t. Coal deposits can be found in more than 70 countries across all continents, with the largest reserves located in the United States, Russia, China, and India [19]. In the modern world, the global demand for low-cost energy, iron, steel, and cement has resulted in the continued growth of coal extraction. Coal reserves are projected to last longer than conventional oil and gas reserves by 115 years, given current extraction rates. Ninety percent of the world’s coal production originates from ten countries [20].

Studies regarding soil pollution around coal mining areas have significantly increased in recent decades. The source, distribution, and content of organic and inorganic pollutants in soils around abandoned or active coal mine areas have been the subject of many investigations so far. Currently, the only existing reviews are limited to the broad evaluation of worldwide metal pollution in soils near coal mines and their health risk assessment [21,22] and also changes in soil characteristics caused by surface coal mining and subsequent reclamation [23,24]. The objectives of the study were: (1) to evaluate the contamination levels of organic and inorganic pollutants in soils around both surface and underground coal mining areas and (2) to identify the most-common remediation strategies used for the removal of pollutants from coal mining areas.

2. Methodology

We searched for articles from around the world that described soil pollution in mining areas and tried to sort them into categories and summarize them. Based on the aim of this study, we performed a search on the Scopus and Web of Science databases with the terms (coal mining) AND (soil) AND (PAHs) AND (heavy metal) AND (Remediation Strategies). Finally, in accordance with studies published from 2000 to 2023, a global analysis was undertaken in this study.

3. Contents and Chemical Forms of Trace Metals and Metalloids in Coal

Coal is a highly complex geological material that contains over 200 minerals and nearly all naturally occurring elements. It also has an intricate chemical structure [25]. According to Finkelman [26], the modes of occurrence of metals in coal refer to their chemical bonding and distribution within the coal. Ultimately, this knowledge is extremely important in foreseeing their potential impact on human health and the environment [25]. Many scientific publications have addressed the detrimental environmental effects of the metals contained in coal [27–29]. The majority of these studies were concerned with the volume of elements released into the environment due to coal mining, disposal, and combustion [25]. In the following sections, the occurrence and chemical forms of selected trace elements in coal are briefly detailed.

3.1. Cr

According to Ketris and Yudovich [30], common low-rank coals typically have 15 mg/kg and hard coals contain 17 mg/kg of Cr on average. Considering the toxicity and carcinogenicity of, in particular, the Cr(VI) form, the chemical form of occurrence is very important. Goodarzi et al. [31] found that Cr is associated with illite and occurs in coal primarily as Cr(III) rather than the hazardous Cr(VI) in the corresponding fly ashes. Several studies have indicated that Cr in coal is predominantly linked to organic matter, especially in low-rank coals [32], clay minerals, spinel-group minerals [26,33], and other mineral phases including carbonates [34]. In summary, the main association of Cr in coal is with organic matter and aluminosilicates [25].

3.2. Ni

The common low-rank coals have a mean Ni concentration of 9 mg/kg, which is approximately half of the worldwide concentration in hard coals at 17 mg/kg [30]. Dai et al. [35] revealed that the mean content of Ni in Chinese coals is 13.7 mg/kg, while Orem and Finkelman [33] discovered that the mean content of Ni in U.S. coals is 14 mg/kg. In coal, Ni is linked to clays, sulfides (mostly pyrite and other minor sulfides), and organic matter [26,32,36]. During a density fraction study, Kolker et al. [37] discovered that Ni is enriched in both the heaviest- and lightest-density fractions, suggesting that Ni in coal is bound with both organic and pyritic components. In conclusion, Ni in coal is mostly connected to clays, organic matter, and sulfides [25].

3.3. Cu

The mean contents of Cu in low-rank and hard coals around the world are similar, at 15 mg/kg and 16 mg/kg, respectively [30]. These values were comparable to those reported for U.S. and Chinese coals, which were 16 mg/kg and 17.5 mg/kg, respectively [33,35]. In most coals, Cu is often found as chalcopyrite and pyrite and can be organically bound, particularly in low-rank coals [26,32]. Finkelman et al. [38] found that Cu can be found in low-rank coals mainly in organic fractions (20%), clays (20%), chalcopyrite (30%), and pyrite (30%). Similar forms of occurrence can be found in bituminous coals, but in different percentages, with 5% organic, 20% clays, 30% chalcopyrite, and 45% pyrite. In conclusion, Cu in coal is mainly found in pyrite and chalcopyrite. In some low-rank coals, it could be found in the organic fractions [25].

3.4. Zn and Cd

Zinc and Cd have a similar chemical behavior. They are both commonly associated with sphalerite and usually behave in the same way in coal [39]. Zinc in coal was found at 28 mg/kg in hard coal and 18 mg/kg in common low-rank coal across the world [30]. According to Orem and Finkelman [33] and Dai et al. [35], the mean Zn concentration in U.S. and China coals is 53 mg/kg and 41.4 mg/kg, respectively. These values are significantly higher than the mean values in coals from other parts of the world. However, the concentrations of Cd are much lower in common low-rank coal with 0.24 mg/kg and hard coal with 0.20 mg/kg globally [30]. Mean Cd contents in U.S. and Chinese common coals are 0.47 mg/kg and 0.25 mg/kg, respectively [33,35].

Many studies have shown that sphalerite is the primary host for Zn in coal [40,41]. Although sphalerite is the most-common mineral containing Cd [42], other minerals including silicate and pyrite may also be associated with Cd [43]. Zinc in coal is often bound with sulfide and aluminosilicate, as well as organic matter [44,45]. In conclusion, sphalerite is the primary host of Zn and Cd in coal. Zn in low-rank coal is bound with the organic fraction [25].

3.5. As

Ketris and Yudovich [30] reported that the contents of As in low-rank and hard coals worldwide are 7.6 mg/kg and 9.0 mg/kg, respectively. However, Dai et al. [35] found that a lower mean concentration in Chinese coals of 3.8 mg/kg. According to Swaine [42], As is most commonly associated with pyrite; however, it can also be bound with phosphates, clay minerals, and organic matter. By employing numerous approaches, some researchers demonstrated that As has an organic association in some sub-bituminous and lignite coals [43,46]. In summary, As in coal is mainly found in pyrite and has an association with organic matter. Some As-bearing minerals, such as Ti-As sulfide, zeunerite, orpiment, realgar, getchellite, and arsenopyrite, have been found in coal; however, these minerals are neither common, nor unique hosts for As in coal [25].

3.6. Pb

On a worldwide basis, the mean Pb contents in hard and low-rank coals are 9.0 mg/kg and 6.6 mg/kg, respectively [30]. The mean Pb contents in U.S. and Chinese coals are higher than those found in common coals across the world, at 11 mg/kg and 15.1 mg/kg, respectively [33,35]. In general, Pb in low-rank coals is bound with organic matter [32,42]. Using selective leaching, Finkelman et al. [43] found that Pb in bituminous and low-rank coals is associated with various phases, including silicate, pyrite, monosulfides (galena), and other unknown phases. In conclusion, pyrite is the most-common mineral form of Pb in coal, while it can also be found in organic matter and trace minerals (including, clausthalite and galena) [25].

4. Levels of Soil Pollution in Coal Mining Areas

4.1. Heavy Metals and Metalloids

Around the mine area, a significant quantity of heavy metal dust and polluted wastewater may be produced during the many phases of the coal mining process, including transportation, wastewater treatment, tailings' placement and slurry pumping, topsoil removal, mineral extraction, and ore concentration [47]. These heavy metals potentially can migrate to ground water and aquatic environments via surface runoff. The local natural ecosystem can be severely disrupted by the long-term effects of mining disturbances, which include groundwater quality impacts, soil erosion, and land subsidence [48]. Coal gangue and fly ash are sources of toxic heavy metals, which can be emitted into the environment during coal transport, smoke and dust emissions from coal-fired power plants, stormwater discharges from mining areas, and other operations associated with the coal industry [49].

The primary source of pollution in coal mines is attributed to the process of leaching acidic mine tailings or the creation and release of acid mine drainage (AMD) [50]. AMD is formed because of the presence of pyrite and sulfide minerals within the coal seam and associated rocks. Exposure to water and air causes these sulfide minerals to be oxidized. Free sulfuric acid is one of the oxidation products [51,52]. Metal elements from sulfide oxidation, including Zn, Al, and Ni, as well as toxic materials containing Cr, Pb, and Cu, are released after the pH is decreased [53]. Metal pollutants resulting from mining activities can be transported over long distances by water and wind erosion or remain in tailings for later release with continued weathering [54]. In mining areas, these pollutants in soil may have adverse impacts on nutrient availability and microbial activity [55].

Currently, research on heavy metals in soil has primarily been concentrated on analyzing the distribution pattern, total content, and ecological and human health risks [56]. The source of heavy metals in the soil from the mining area, however, is more complicated given that it may have arisen from not only emissions during coal extraction, but also from atmospheric deposition from coal-fired power plants and the dispersion during the process of transporting coal as well [57]. Studies regarding the concentration of trace metals and metalloids in coal mining areas worldwide are provided in Table 1, and some of those studies are shown below.

According to the literature reviews, during the last ten years, China has conducted numerous studies on soil heavy metal pollution caused by coal mining activities. For example, Yan et al. [58] conducted a study in a coal mining district in Tai'an City and found that Hg and Cd were the primary pollutants in the area. Additionally, Cd exhibited the highest rate of contamination. This result was associated with roads that were employed for transporting coal from the mines. Sun et al. [59] reported similar results in the Tangshan Coal Industrial City, where a majority of soil samples exhibited slight to moderate contamination by Hg and Cd, mainly due to coal-related industrial activities, vehicle emissions, agrochemical application, and sewage irrigation. Zhang et al. [60] discovered that the primary sources of As and Hg in soils from major coal mining regions of Xinjiang were atmospheric dust emissions resulting from coal combustion. Additionally, the high volume of traffic and frequent transportation associated with coal delivery in the study area were the main source of soil Pb pollution. According to Li et al. [61], the increased concentrations of Cu and Cd in paddy soils in the vicinity of the Zhangji coal mine in Huainan City were associated with mining activities, i.e., from rainfall leaching. According to Dong et al. [62], the main cause of soil pollution in the Liuxin mining area of Xuzhou was attributed to coal mining activities. However, there were also other contributing factors such as industrial pollution, sludge addition, and untreated water irrigation.

Kou et al. [63] revealed that the soils in Shengli and Baorixile coal mines located in Inner Mongolia showed elevated levels of pH and heavy metals such as Hg, As, Ni, Mn, Zn, Pb, and Cu, surpassing the mean soil background values, as a result of high coal mining and burning volume. In Mongolia's main coal mining town (Sharyn Gol), Pecina et al. [64] found high levels of Zn, Pb, Cu, and Cd in the soil, resulting from the deposition of mining waste heaps. Due to AMD, moderate soil pollution by Ni, Zn, and Cu was found by Hossen et al. [65] in the area surrounding the Barapukuria coal mine in Bangladesh. Siddiqui et al. [11] provided a comprehensive analysis of the effect of elements in soils from the aforementioned mining area. Zinc exceeded the world normal limit [66] among the evaluated elements. The study revealed that Ag showed the highest degree of enrichment, followed by As. In this study, Zn and As were potentially derived from minerals such as mispickel, melnikovite, and sphalerite, which are commonly found in association with coal deposits.

A study conducted In Neturia, West Bengal, showed that opencast mining has led to an elevated level of toxic heavy metals in the soil, leading to heavy metal contamination and associated non-carcinogenic health risks to humans. Moreover, this study found that topsoil was exposed at the surface, which made it more susceptible to heavy metal pollution [10]. According to a study conducted by Siddiqui et al. [67], soils collected from the Jharia coalfield in India exhibited a moderate level of pollution risk in terms of trace elements such as Cu, Pb, Zn, Cr, Ni, Co, and Cd, released from mining spoils. Raj et al. [68] in their investigation of soils near an opencast coal mine in Eastern India found that coal mining activities including coal dust deposition and transport activities were the predominant source of metal pollution in the area. Reza et al. [69] indicated that agricultural soils in the Ledo coal mining area of India were significantly contaminated with Cd and Pb due to mine drainage. Ahmad et al. [70] found that the levels of Zn, Cu, Ni, and Cr in the soil of the Dukki and Sharigh coal mines in Pakistan exceeded the acceptable limits.

Marove et al. [71] found that the type of leaching solution strongly impacted the hazardous elements from soils around open-pit coal mines in Mozambique. They also discovered that some of these elements were highly bio-accessible, which could lead to health and ecological risks in the area. Zerizghi et al. [72] conducted a study on the soil in the surrounding area of the Greenside coal mine in South Africa. They found that the levels of heavy metals examined in the soil surpassed the local background value. Moreover, the predominant health and ecological hazards associated with Cr were discovered. Ameh and Aina [73] found that the Cr, Cd, and As concentrations in soil samples obtained from the polluted coal mine in Okaba, Nigeria, were within the standard limits for unpolluted soils. However, the Cu concentration in the soil samples exceeded the permissible limits. Galunin et al. [74] reported findings from a coal mining region in Brazil showing that the contents of Mn, Ca, Al, Mg, and pH separately had a considerable impact on the Cd sorption behavior, and similarly, these factors also impacted the desorption behavior.

Alekseenko et al. [75] found that coal mining waste disposal sites in Rostov Oblast, Russia, had a direct impact on the soil, which resulted in increased levels of Ba, Mo, Pb, Zn, Cr, V, Cu, and Mn. In Poland, Pietrzykowski et al. [76] found that areas impacted by the Smolnica hard coal mine that had been reclaimed and reforested did not pose any risk of hazardous levels or the bioavailability of metals. The study conducted by Fiket et al. [77] in Croatia demonstrated that the soils in the Labin City area exhibited distinct metal fingerprints as a result of coal mining activities, particularly coal combustion or transport infrastructure. Consequently, soils showed varying degrees of enrichment in Zn, Sr, Sb, Pb, Mo, Cu, and Co, ranging from moderate to extremely high. Boahen et al. [78] found that the levels of Zn, Cd, Be, and As in the soil samples collected from the North Bohemian Brown Coal Basin (Czech Republic) surpassed the recommended thresholds. The agricultural soils from the Northern Czech Republic were found to be contaminated by anthropogenic material associated with past mining activities, such as lignite mining in North Bohemia and hard coal mining and heavy industry in North Moravia [79]. Potential environmental implications from AMD and associated dissolved metals discharging in Portugal's Douro coalfield have been reported by Ribeiro et al. [80]. In Spain, Boente et al. [81] revealed that coal mining and processing history in the municipality of Langreo had a strong association with the increased levels of Cu in the soil.

Table 1. Concentration of trace metals and metalloids in coal mining areas worldwide (mg/kg).

Country		Mine Type	Status	HMs (mg/kg)								References
				Cd	**Hg**	**Cu**	**Ni**	**Pb**	**Zn**	**As**	**Cr**	
South Korea: Gangreung coalfield		unknown	abandoned	1.10	n.a.	41.0	42.6	32.9	87.3	n.a.	35.8	[82]
China: Linhuan coal mining area		underground	active	0.17	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	[12]
China: Coal mining region of Tai'an		underground	active	0.20	0.03	26.5	29.61	27.6	66.7	9	56.4	[58]
China: Zhundong mining		surface	active	n.a.	0.01	n.a.	n.a.	16.3	0.0067	9.0	53.0	[60]
China: Kaili City		unknown	abandoned	0.52	0.11	13.3	n.a.	30.7	56.2	28.2	75.6	[83]
China: Yulin		underground	active	n.a.	n.a.	15.0	17.9	21.5	57.6	5.3	45.1	[84]
China: Xilinguole League		surface	active	0.11	n.a.	15.0	n.a.	20.0	49.3	9.1	52.9	[85]
China: Xuzhou coal mining region		unknown	active	0.88	n.a.	22.9	n.a.	19.0	128.3	17.1	58.1	[86]
China: Tangshan City		underground	active	0.15	0.06	22.4	16.8	22.9	70.3	5.9	37	[59]
China: Weibei coalfield		underground	active	0.28	n.a.	58.1	88.1	11.49	1139.8	n.a.	143.1	[87]
China: Zhangji coal mine		underground	active	0.54	n.a.	64.4	27.7	20.3	118.0	23.4	85.3	[61]
China: Yanzhou		underground	active	0.14	n.a.	23.1	29.5	23.7	66.3	n.a.	72.1	[88]
China: Eastern Junggar coal mine		surface	active	n.a.	n.a.	19.3	n.a.	16.7	47.8	33.5	68.5	[89]
China: Xinzhuangzi		unknown	unknown	0.09	n.a.	36.8	28.4	25.4	62.0	3.7	n.a.	[48]
China: Datong coal mine		underground	abandoned	n.a.	n.a.	26.4	34.3	50.2	76.1	n.a.	188.6	[90]
China: Xingren coal mine		unknown	active	1.09	0.71	134.1	64. 9	240.6	n.a.	477.5	173.6	[91]
China: Liuxin mining area		unknown	abandoned	0.40	0.03	31.8	n.a.	12.9	91.3	24.3	44.5	[62]
Mongolia: Sharyn Gol		surface	active	1.46	n.a.	22.1	n.a.	69.0	87.6	n.a.	n.a.	[64]
Mongolia	Shengli	surface	active	n.a.	0.03	19.6	25.0	16.8	60.7	9.0	n.a.	[63]
	Baorixile	surface	active	n.a.	0.08	19.7	20.8	26.3	64.2	8.6	n.a.	
Bangladesh: Barapukuria coal mine		underground	active	n.a.	n.a.	31.6	56.5	n.a.	101.9	n.a.	82.3	[65]
				0.25	n.a.	27.4	48.0	11.2	128.0	5.1	17.5	[11]
				n.a.	n.a.	n.a.	n.a.	433.0	296.0	17.5	n.a.	[66]
				n.a.	n.a.	29.9	98.5	188.6	160.0	22.4	107.3	[92]

Table 1. *Cont.*

Country		Mine Type	Status	HMs (mg/kg)								References
				Cd	Hg	Cu	Ni	Pb	Zn	As	Cr	
Bangladesh: Farmland around Barapukuria coal mine		underground	active	0.09	n.a.	2.1	n.a.	0.9	8.2	n.a.	n.a.	[93]
India: Neturia block		surface	active	n.a.	n.a.	677.0	811.5	265.2	893.7	33.7	851.7	[10]
India: Jharia coalfield		underground and surface	active	0.80	n.a.	11.3	11.3	11.4	19.9	n.a.	23.4	[67]
				0.40	n.a.	66.3	64.1	27.8	127.0	n.a.	43.0	[94]
India: Rohini		surface	active	1.11	0.58	n.a.	n.a.	16.9	n.a.	3.1	17.5	[68]
India: Korba coalfield		surface	active	n.a.	n.a.	218.0	n.a.	311.0	426.0	n.a.	567.0	[95]
India: Raniganj coalfield		underground and surface	active	n.a.	n.a.	256.6	347.3	88.2	369.1	15.4	306.3	[96]
India: Eastern part of Jharia coalfield		underground and surface	active	2.49	1.19	26.0	n.a.	15.7	55	2.7	55.0	[97]
India: Ledo coal mining area		surface	active	2.60	n.a.	n.a.	87.5	183.1	n.a.	n.a.	112.3	[69]
India: Surat district		unknown	active	0.77	n.a.	57.0	16.0	9.0	60.0	n.a.	8.0	[98]
Pakistan	Dukki coal mines	underground	active	n.a.	n.a.	421.6	125.1	n.a.	131.2	n.a.	149.1	[70]
	Sharigh coal mines	underground	active	n.a.	n.a.	53.7	115.4	n.a.	68.5	n.a.	95.7	
Iran: Aghdarband coal mine		underground	active	n.a.	n.a.	33.0	14.0	52.0	89.0	33.3	39.0	[99]
Iran: Tazareh coal mine area		underground	active	0.31	n.a.	27.1	43.1	19.2	78.8	0.2	91.5	[100]
Turkey: Oltu coal mine district		underground	active	0.05	1.15	28.0	97.0	49.4	36.8	n.a.	180.2	[101]
Turkey: Ovacik–Yaprakli		underground and surface	abandoned	n.a.	n.a.	58.0	296.8	n.a.	101.1	n.a.	348.0	[102]
Australia	Glenbawn	surface	active	n.a.	n.a.	59.0	n.a.	12.8	137.0	14.2	n.a.	[103]
	Traralgon	surface	active	n.a.	n.a.	22.6	n.a.	22.1	132.8	5.6	n.a.	
Mozambique: Moatize district		surface	active	n.a.	n.a.	34.0	31.0	26.0	78.0	3.0	95.0	[71]
South Africa: Witbank coalfield		underground	active	0.40	n.a.	33.1	98.3	30.7	110.6	19.4	653.3	[72]
South Africa: Emalahleni		underground	active	n.a.	n.a.	22.0	20.0	19.0	36.0	0.5	419.0	[104]

Table 1. *Cont.*

Country	Mine Type	Status	HMs (mg/kg)								References
			Cd	**Hg**	**Cu**	**Ni**	**Pb**	**Zn**	**As**	**Cr**	
Nigeria: Okaba	surface	active	1.05	n.a.	5.9	n.a.	n.a.	n.a.	1.4	5.2	[73]
Morocco: Jerada coal mine	underground	abandoned	n.a.	n.a.	32.6	n.a.	60.6	144.3	24.2	n.a.	[105]
Botswana: Morupule coal mine area	underground	active	n.a.	n.a.	40.3	60.5	36.6	304.5	13.2	155.6	[106]
Russia: Kizel Coal Basin	unknown	abandoned	n.a.	n.a.	n.a.	69.0	n.a.	80.0	10.0	178.0	[107]
			0.93	2.59	33.0	41.9	36.5	61.4	2.9	1653.0	[108]
Russia: Lipovtsy coalfield mine	surface	abandoned	0.03	n.a.	20.7	34.2	12. 8	187.8	n.a.	8.2	[109]
Russia: Rostov Oblast	underground	active	n.a.	n.a.	57.0	41.0	29.0	65.0	n.a.	90.0	[75]
Russia: Vorkuta coal mining area	underground	active	1.0	n.a.	27.0	31.0	38.0	380.0	5.6	27.0	[110]
Russia: Tula coal mining region	unknown	abandoned	n.a.	n.a.	n.a.	n.a.	n.a.	164.5	17.4	n.a.	[111]
Greece: Ptolemais lignite basin	surface	abandoned	n.a.	n.a.	n.a.	10.1	n.a.	n.a.	12.3	17.5	[112]
			0.4	n.a.	44.4	65	62.1	110.0	11.7	876.0	[113]
Poland: Upper Silesian Coal Basin	underground	active	2.55	n.a.	16.5	7.1	106.0	273.9	n.a.	67.1	[114]
			0.80	0.07	8.5	6.3	39.8	62.5	8.8	34.4	[115]
Poland: Smolnica hard coal mine	underground	abandoned	n.a.	n.a.	39.5	n.a.	61.5	80.0	n.a.	n.a.	[76]
Czechia: Litvínov City area	surface and	active	0.46	n.a.	31.1	26.0	48.9	175.0	41.7	45.6	[116]
	underground	abandoned									
Czechia: Sokolov Coal Basin	surface	active	0.98	n.a.	36.1	19.1	40.7	118.0	33.7	40.0	[117]
Czechia: Ostrava City	underground	active	0.21	0.19	21.1	n.a.	37.7	204.5	n.a.	17.4	[118]
Czechia: North Bohemian Region	surface	active	0.33	0.17	35.7	32.6	51.5	107.9	33.8	n.a.	[79]
Croatia: Raša coal mine	underground	abandoned	1.00	0.09	55.0	74.0	45.0	169.0	17.0	123.0	[119]
Croatia: Labin City area	underground	abandoned	2.01	n.a.	2226.0	176.0	484.4	2778.0	21.9	684.6	[77]
Spain: Langreo	underground	abandoned	0.60	0.40	39.0	18.3	91.6	136.2	21.8	18.9	[81]
Portugal: São Pedro da Cova	underground	abandoned	0.11	n.a.	50.2	24.3	50.2	97.0	22.6	74.1	[120]
Portugal: Douro coalfield	underground	abandoned	0.20	n.a.	36.5	21.4	30.8	57.0	38.3	92.3	[80]
Norway: Svalbard	underground	abandoned	n.a.	n.a.	n.a.	5.1	2.0	29.3	0.3	5.6	[52]
Wales (U.K.): Varteg mine	surface	abandoned	1.16	n.a.	63.9	n.a.	224.0	192.8	n.a.	n.a.	[121]

Note: n.a. = not analyzed.

This review revealed that the majority of publications in the analyzed areas originate from Asian countries, specifically China (17 articles) and India (9 articles). There is a research gap in this field of study from countries having high amounts of coal reserves, especially the U.S., Germany, Indonesia, Ukraine, and Kazakhstan. However, there are some limited studies from Australia, Poland, and Russia.

As shown in Figure 1, the most-analyzed element in coal mining areas is Zn followed by Cu and Pb, while Hg received much less attention compared to other metals. Coal mining activities often result in elevated levels of Hg in the surrounding environment [122]. Furthermore, as the extraction of coal increases, the concentration of Hg in regional soil also tends to rise [122]. Coal gangue particles emit Hg into the environment upon spontaneous combustion and leaching. Then, it migrates, transforms, and accumulates in ecological systems, ultimately entering the human body via multiple pathways and impacting human health [123]. Coal processing generates soot, dust, and wastewater, which can lead to the release of Hg into the environment. Additionally, the combustion of coal-preparation byproducts can lead to secondary emissions, which can contribute to local environmental contamination of Hg [124].

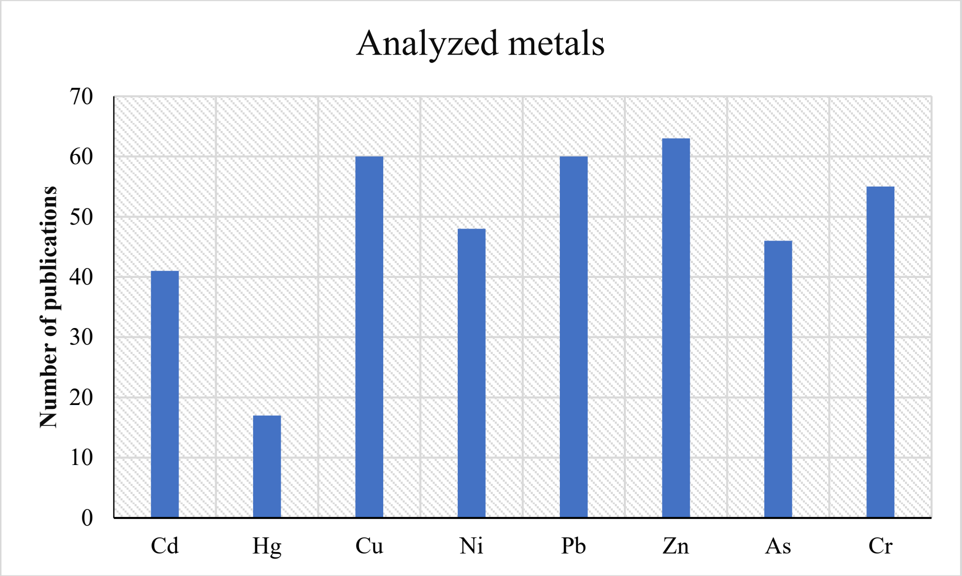

Figure 1. Most-analyzed metals around coal mining areas (mg/kg).

Thallium (Tl) is another element that is ignored in studies conducted around coal mining areas. This metal is classified as 1 of the 13 priority pollutants, which are considered to be more hazardous than Hg and Cd [125]. This metal is heavy, volatile, and highly toxic by exposure to contaminated soils and inhalation. While it is not commonly found in natural systems, it is significant in anthropogenic systems owing to its toxicity. Even at low concentrations in the environment, it has the potential to cause severe ecological risks [126]. Coal mining activities can lead to the release of Tl and its compounds into the environment, resulting in elevated levels of Tl in crops, soils, and water. This can pose a direct threat to human health through food chains and contact [127]. Zhang et al. [127] found that surface soils in the surrounding area of coal mines in Henan Province (China) had higher contents (0.77 mg/kg) of Tl than background values (0.42 mg/kg).

4.2. PAHs

Polycyclic aromatic hydrocarbons are a group of hydrophobic substances that contain at least two annelated aromatic rings and are categorized into subgroups based on the number of rings they contain [128]. They occur in coal and are generated through the process of the incomplete combustion of carbon-based fuels, including wood or coal. Additionally, they can also originate from natural sources such as petroleum input in oil seeps and volcanic eruptions [129]. These pollutants can also be generated through anthropogenic processes such as biomass burning, car exhausts, waste incineration, industrial emissions, and energy production [130]. The coal industry is another major contributor to the release of PAHs into the environment. This is due to the consumption of coal as a source of energy, as well as the accumulation of coal waste and the processing of coal, which have all led to the contamination of soil with PAHs [131]. In general, areas with heavy industrial activities in which coal facilities and metal industries are found are subject to higher levels of PAH emissions [132]. The European Union and the U.S. Environmental Protection Agency (USEPA) have both assigned PAHs as priority contaminants [133]. Wang et al. [134] conducted a comparative analysis of PAH concentrations in coals from different studies. Their findings revealed that the 16 priority PAHs as listed by the USEPA ranged from <0.1 to 260 mg/kg. Based on the coal rank and provenance, the total extractable PAHs in coals can range from a few tens to hundreds, in some cases thousands, of milligrams/kilogram [134,135].

In a study conducted by Xu et al. [136], it was found that the average polycyclic aromatic compounds (PACs) in the coal gangue, coal, and topsoil of typical coal mining regions in the Huaibei area of China were 10,908, 274,815, and 1528 mg/kg, respectively. In this study, alkyl PAHs were the main source of PACs, while coal and coal gangue exhibited notably higher levels of oxygenated PAHs compared to topsoil. Furthermore, the highest levels of PAH pollution were primarily identified in the vicinity of the coal mine region and nearby the road leading to the coal gangue landfill site. Liang et al. [137] conducted a study on soil in the Wuda coalfield area of China, wherein they observed the occurrence of coal fire sponges, which are polluted soil protrusions with a sponge-like appearance, in the Suhaitu mining area. This study showed a high level of contamination at 1000 mg/kg. Additionally, it was observed that low-molecular-weight PAHs were the predominant compounds, constituting over 50% of the total PAHs. In the topsoils from an extensive coal mine in Huainan, China, Zhang et al. [138] evaluated the presence, hazards, and influencing variables of PAHs. The concentration of PAHs in the soil was found to be comparatively higher than that of industrial, urban, and agricultural soils, while being lower than in some regions associated with coal mines and coal-fired power plants across the globe. Earlier work on this site by Wang et al. [139] revealed that the pollution level of PAHs in the soil profiles was higher in the upper layers of soil due to the gob pile and coal preparation plant. In this study, the mean PAH concentration measured in soil samples collected from coal mine areas was 0.84 mg/kg. This concentration is significantly higher than the natural PAH content in soil and also exceeds the Dutch standards. These findings suggest that coal mine areas have a higher carcinogenic burden.

According to a study conducted by Fan et al. [17], the levels of PAHs were found to increase as the distance from the coal gangue dump in the Gequan coal mine (China) decreased. These findings suggest that the organic matter present in the samples originates from coal particles found in the coal gangue dump. In a study from the Tiefa coal mine (Northeast China), Liu et al. [140] found that the surface soil had a total content of 16 PAHs within the range of 5.1 to 5642 mg/kg, with an average of 1118.3 mg/kg. The Tiefa coal mine's activities including coal gangue, unburned coal particulates, and coal combustion have resulted in soil pollution not only within the mining area, but also in the surrounding agricultural and lake bank soils. Moreover, the results indicate that the soils in the study area are heavily contaminated with PAHs and pose a significant health risk. According to Sun et al. [141], the coal gangue dump located at Jiulong coal mine is rich in deleterious organic compounds. Over the course of 15 years, wind and rain have contributed to the transportation of total organic carbon, sulfur, and aromatic compounds from the coal gangue dump into the surrounding soil and riverbed. Mizwar et al. [142] investigated surface soils of three distinct sites in South Kalimantan Province (Indonesia), namely coal stockpile, agricultural, and residential areas. Compared to agricultural and residential soil, they found that coal stockpile locations exhibited higher concentrations of PAHs in their soil. In this study, PAHs were a combination of pyrolytic and pyrogenic sources. Masto et al. [143] reported higher PAH contents in underground mine soils compared to surface coal mining in the Raniganj coalfield area (India), which may be attributed to natural coal burning at these sites. Yakovleva et al. [144] found that light polyarenes constituted a significant proportion (82–91%) of the total soil PAHs in both control and polluted sites in a study on soils affected by coal mining in Russia.

Jakovljević et al. [145] found that the soil in the Labin City area (West Croatia) had significantly increased levels of PAHs due to pollution from the Raša coal mining area, which has been ongoing for centuries. The findings showed the existence of pyrolytic PAHs formed by the burning of Raša coal at high temperatures in power plants, as well as unburned coal-derived PAHs resulting from Raša coal carbonization. In Spain, Boente et al. [146] found an elevated level of high-molecular-weight PAHs in an urban area that has been affected by coal mining activity such as coal combustion. They also noticed that benzo[a]pyrene levels exceeded the threshold in almost all of the assessed soil samples. In Germany, Hindersmann and Achten [147] found high levels of PAHs in urban soils affected by coal mining tailings. These PAHs were typically associated with non-point pyrogenic carbon sources such as particulate matter and soot. In all soil samples taken from the Saar River and the Mosel River downstream (Germany), Pies et al. [148] observed higher PAH concentrations as a result of past coal mining activities in this area. According to a report by Ugwu and Ukoha [149], the plant and soil samples taken from the Nigerian Okobo coal mine area showed low levels of toxic PAHs.

Similar to the heavy metals' and metalloids' section, the majority of conducted studies on PAHs' analysis around coal mining areas are from China. Furthermore, this field of study has been ignored or has limited attention in other countries having high levels of coal reserves.

5. Applied Soil Remediation Strategies in Coal Mining Areas

5.1. Bioaugmentation

The acidic environment and high levels of heavy metals at coal mine sites lead to a decrease in microbial abundance and diversity. In addition, coexisting abiotic parameters including dissolved oxygen, organic carbon, temperature, and ionic composition also contribute to the composition of microorganisms that survive in such an environment [150]. AMD-tolerant microorganisms include archaea, bacteria, and even certain eukaryotes, such as algae and fungi [151]. The adverse environmental conditions in soil and water at coal mine sites restrict the survival of bacteria to a few phylogenetic groups such as *Thiobacillus* spp. and *Methanogen* spp. [152]. The first microorganism that has been successfully separated from the acidic environment is *Thiobacillus ferrooxidans* [153]. The acid and metal tolerance of *Rhodococcus* spp., *Acinetobacter* spp., *Enterobacter* spp., *Staphylococcus* spp., *Klebsiella* spp., and *Bacillus* spp., which were obtained from coal mines and eventually cultured, have been reported [154,155]. According to Shylla et al. [156], indigenous bacteria present in soils with elevated levels of heavy metals could potentially be considered viable options for the bioremediation of heavy metals in polluted areas. Due to the fact that these bacteria are native to the system, it is possible that their ongoing interaction with the conditions of low pH and AMD enabled them to develop resistance. The differences in their ability to tolerate metals may be due to variations of in vitro mechanisms or cell wall composition [157].

Indigenous strains of *Thiobacillus* and *Bacillus* that are tolerant to metals have been discovered in mining environments. These strains have been proven to assist in the bioreduction of toxic metals, including vanadium. This suggests that they could be a favorable method for metal bio-remediation [158]. For example, metal-resistant *Bacillus* spp. as described by Gupta et al. [159] can thrive in soils with Fe concentrations of 400–550 mg/kg. Furthermore, Upadhyay et al. [160] found that *Bacillus* spp. taken from polluted soils in coal mine areas exhibited tolerance to Cr (VI) and showed an efficient reduction of Cr (VI) to Cr (III). Several explanations have been provided regarding the mechanism underlying this conversion. It may happen when chromate functions as the final electron acceptor to obtain energy, or when bacteria produce waste products, or when multiple enzymes interact with Cr(VI) to mitigate its toxicity by turning it into Cr(III). Syed and Chinthala [161] found that *B. subtilis, B. cereus,* and *B. licheniformis* were effective at reducing the content of Pb in soils by 86%, 87%, and 78%, respectively. A study on bacteria in mine soils polluted with metals found that *Bacillus* was the most-predominant species identified. These organisms demonstrated notable levels of tolerance, as evidenced by their ability to endure the highest levels of 207–414 mg/kg Pb and 65–196 mg/kg Zn [162]. In this study, producing metal-chelating substances, including siderophores and organic acids, has been crucial in facilitating metal detoxification and enhancing metal tolerance in bacteria. Singh and Tiwary [163] conducted a study at the Chirimiri coal mines in India and discovered that the *Pseudomonas stutzeri* bacteria collected from the mines exhibited the ability to degrade high concentrations of pyrene and phenanthrene in polluted soil. In this study, several dioxygenase enzymes, namely protocatechuate, catechol, and gentisic acid, were produced by Pseudomonas stutzeri bacteria in order to degrade PAHs.

Despite all the reported advantages of this remediation strategy, Kurniawan et al. [164] claim that, since heavy metals are not effectively separated from the treated medium using bioaugmentation alone, this method cannot be used to treat heavy-metal-polluted soil in real-world settings at large scales. The majority of studies that have documented the effective removal of heavy metals from contaminated soil were primarily conducted in laboratory settings, where environmental conditions are carefully regulated.

5.2. Phytoremediation

The cultivation of hyperaccumulator plants, which are either naturally occurring or created as a result of genetic modification, is one of phytoremediation strategies for removing metals from the soil and accumulating them in plant biomass [165]. It has the potential to not only strengthen the ecological environment of the mining region, but also facilitate the recovery of vegetation [166]. Phytoremediation is a comprehensive remediation strategy that encompasses a range of techniques, including rhizodegradation, rhizofiltration, phytoevaporation, phytostabilization, and phytoextraction, all of which are mediated by microbial interactions with roots and soil [13]. In addition, it has economic benefits and aesthetic values, both of which can result in a wide range of social advantages [167,168]. As an illustration, Pandey and Bajpai [169] suggested in their study that abandoned mining sites have the potential to be converted into public spaces, such as parks, in order to offer ecosystem services and products. Additionally, this finding demonstrates the various advantages of phytoremediation for promoting sustainable ecosystem development [170]. Some concern has been expressed about the disposal of the biomass of hyperaccumulator plants, but there is evidence that metals bound into plant biomass can be stabilized and less available for release over time.

In coal gangue sites that experienced a decade of phytoremediation with single/mixed elm/poplar, Bai et al. [171] found that the soil contents of Th, U, Bi, Co, Ni, Pb, and Cu were considerably decreased. In their study, Feng et al. [172] applied herbaceous plants for performing vegetation restoration efforts in coal mines located in Hulunbuir (Inner Mongolia). The findings of this study indicated that the restoration of vegetation has had a positive impact on the overall restoration of the ecosystem within the studied area. According to a report by Ameh and Aina [73], the only plants that exhibited high potential as phytostabilizers of Cd in a coal mine soil in Nigeria contaminated with toxic metals were *Fuirena umbrellata* and *Selaginella myosurus*. Furthermore, *Hyptis Suaveolens* exhibited hyperaccumulation potential for Cu (>1000 mg/kg accumulation). Furthermore, they concluded that it is possible for native plants to naturally eliminate toxic elements from soils that have been contaminated with metal pollutants.

In Australia, successful Se extraction (48% and 28%) was achieved from phytoremediated post-mining coal wastes area by *Brassica juncea* [173]. Further Se extraction may have been accomplished in this study; however, *B. juncea* crops should be harvested immediately after they reach maturity. If the biomass of *B. juncea* is not harvested and dried, it has a tendency to break into chip-like particles that become easily dispersed by wind, potentially leading to the dispersal of accumulated heavy metals. Matanzas et al. [174] found that herbaceous species including *Lotus hispidus* and *Medicago lupulina* have the ability to translocate As and Pb in the polluted soils of a brownfield site from Spain. Consequently, these two plants demonstrate their capacity to serve as bioindicators for the presence of As and/or Pb pollution. Moreover, they could potentially function as phytoextractors or accumulators under varying conditions, such as when there is a higher concentration of potentially toxic elements in the soil compared to the study site. In a more-recent study at this site, Fernández-Braña et al. [175] found that *Buddleja davidii, Betula celtiberica*, and *Acer pseudoplatanus* were effective for phytostabilization in regions with elevated pollutant concentrations. However, these plant species were only suitable for phytoextraction in soils with low-to-moderate pollution levels.

Desai et al. [121] found that using *Alnus glutinosa* and *Betula pendula* for forestation was successful in remediating metals (Mn, Pb, Zn, Cu, and Cd) on moderately polluted lands resulting from opencast coal mining in South Wales. Additionally, this study examined soil measurements taken from various points along a 14-year forestation chronosequence. The results consistently showed that, as the age of the tree plantation increased, the level of soil metal contaminants decreased. This finding provides further evidence that forestation has a positive impact on reducing soil metal loadings. After 25 years of phytoremediation by the *Eucalyptus* hybrid tree in the Jharia coalfield (India), Bandyopadhyay et al. [176] found a considerable reduction in the soil content of Cu, Zn, and Pb. They suggested the application of metal-tolerant woody trees, specifically the *Eucalyptus* hybrid, with high biomass enhanced phytoremediation of coal mine overburden dumps contaminated with metals. These trees have the ability to accumulate considerable quantities of metals in their tissues and decrease metal contents in the soil. Niu et al. [90] reported that particular plant species, such as *Weigela hortensis* and *Ligustrum lucidum*, were found to have a significant potential for reducing the content of Pb, Ni, and Cr in soil from a reclaimed coal mining area in China. According to Mellem et al. [177], *Amaranthus dubius* is capable of hyperaccumulating As in regions associated with coal mining, but has limited capacity for the bioaccumulation of Ni, Cu, Pb, Hg, and Cr. This plant can extract and transport As to its aerial parts and is able to tolerate high levels of this metal.

Phytoremediation is a strategy that has various advantages and disadvantages. It is important to take these into account when considering the application of this strategy. Although cost-effectiveness is a favorable aspect, the duration required to observe the outcomes may be prolonged. It is imperative that the concentration of pollutants and the presence of other toxins should not exceed the tolerance thresholds of the plant species under consideration for utilization. The process of identifying plants with the capability to remediate multiple contaminants concurrently is a challenging operation. It is necessary to consider the limitations and potential for these plants to become part of the food web when implementing this strategy [178].

5.3. Biochar

Biochar is an appropriate substance for environmental applications, particularly in the remediation of contaminated soils, due to its low production cost and availability [179]. It is typically alkaline and possesses a significant surface area along with various active functional carbon groups capable of binding numerous cations [180]. Thus, biochar has the ability to reduce the availability, leachability, and mobility of toxic elements in polluted soils. Additionally, it can also decrease the absorption of these elements by plants [181]. Furthermore, biochar exhibited significant efficacy in mitigating the levels of PAHs in soil. Biochar generated at temperatures exceeding 400 °C indicated a notable propensity for PAHs owing to surface oxidation–reduction reactions. In addition, they show a higher level of aromatic clusters in a condensed form and possess a high degree of porosity characterized by the presence of well-developed nano- and micro-pores, which enable the sorption of low-molecular-weight PAHs [182].

Heavy metal remediation mechanisms in biochar involve precipitation, cation exchange, electrostatic attraction, reduction–oxidation, and physical adsorption. Biochar has been noticed to possess an interesting characteristic, whereby it can have an impact on the behavior of metals. The porous structures found in biochar have the capability to facilitate the transformation of metals into stable forms [183]. The -COOR and -OH functional groups in biochar surfaces contribute to the sorption of many heavy metals, making them unavailable [184]. According to studies, a competitive condition arises among metal ions in their interaction with functional groups on biochar surfaces due to the cationic nature of most metals. This leads to an enhanced immobilization of potentially hazardous elements in polluted soil. In general, the application of biochar treatment results in a significant increase in the levels of reducible and oxidizable heavy metals [185]. Biochar contains graphene moieties that serve as sites for both redox and adsorption reactions, resulting in a high affinity for metal ions and the capacity to transfer electrons to adsorbed reactants [184].

Reclamation principles are generally similar across different mine environments, despite differences in pollution type and level. The general acceptance of biochar for coal mine reclamation has been limited by lacking knowledge regarding its advantages, its accessibility at reclamation areas, the techniques required for its application, its cost, and the long-term effects of its field implementation. In addition, the study of biochar for the purpose of coal mine reclamation is a novel field of study that needs patience to assess its long-term effects when used in degraded areas [186].

The impact of applying chemical fertilizer and *Eucalyptus* wood biochar simultaneously to remediate Co-, Ni-, Zn-, and Cr-polluted soils from an operational coal mine dump in India was evaluated by Chandra et al. [187]. This study revealed that enhancing the mixing ratio of biochar from 0.5 to 5% (w/w) resulted in a considerable reduction of extractable heavy metal concentrations in the soil. The study conducted by Mujtaba Munir et al. [188] assessed the potential synergistic impacts of hydrothermally treated coal (HTC), raw coal (RC), and biochar (BC) on the accumulation, transformation, speciation, and immobilization of Pb, Cr, and Cd in soil polluted by the Huainan coalfield in Anhui, China. The findings showed that the co-application of BC-2% and BC-HTC amendments proved to be more efficient in mitigating Cd, Cr, and Pb contents in comparison to the singular application of RC or HTC amendments. This was achieved by increasing the organic carbon content and pH in the soil. Additionally, the application of BC-2% and BC-HTC amendments resulted in a respective increase of 1.5 and 2.5 units in soil pH. This led to the reduction of Pb, Cr, and Cd to more-stabilized forms in the soil.

Using soil from farms in the Huainan coal mine district, Dai et al. [189] conducted a laboratory experiment to investigate the effects of biochar amendments on the bioavailability and speciation of heavy metals. In this study, the concentrations of Cd, As, Zn, and Cu in polluted soil decreased by 42%, 7%, 51%, and 57%, respectively, when rice–straw-derived biochar was added to the soil. In a study conducted by Jain et al. [190], *Lemongrass* (*Cymbopogon ciatratus*) -derived biochar was applied as a soil amendment for spoil samples taken from coal deposits characterized by elevated levels of sulfur. According to the results, the application of biochar had a positive impact on the Palmarosa plant's metal tolerance index, increasing it by 54%. Additionally, it led to a reduction in acid generation from acidic mine waste. In an earlier study, Jain et al. [191] assessed the impact of Lemongrass-derived biochar application on the heavy metal contents of *Bacopa monnieri* plants growing in acidic coal mine spoil. In this study, using biochar in acidic mine spoil led to reduced levels of heavy metals; Pb decreased by 93%, Fe by 50%, Cu by 42%, Cr by 65%, and Al by 60%. After the application of algal-derived biochar on coal mine stockpiles, Roberts et al. [192] found that the concentrations of Cr, Ni, and Zn were reduced by 49%, 2%, and 55%, respectively. In this study, remediated soils with this biochar had lower or, in some cases, equal contents of metals compared to soil without biochar remediation.

6. Conclusions and Future Perspectives

Heavy metal soil pollution in coal mining areas has been known for many years. These pollutants are related not directly to the excavation of coal to the surface, but mainly to the deposition of dust from coal combustion in power plants located close to coal mines, coal transport activities, and the deposition of mining waste or coal combustion waste (ash coal). As long as post-mining areas remain unremediated and coal is used as a fuel or raw material for chemicals and other beneficial products, pollutants such as heavy metals can be released into the soil from coal consumption products and can be potentially hazardous. The release of these pollutants can be controlled by pre-mine planning and implementing mining and reclamation techniques to control, minimize, or eliminate the problems. The use of soil replacement on the surface can help mitigate pollutant releases by sorbing and stabilizing, depending on the pH, the organic matter content, the surface properties, and the crystalline structure. Soil remediation technologies have been developed and have shown effective mitigation of pollution levels and release. Methods to identify and analyze geologic materials that can cause problems, such as determining the sulfide contents of rocks, can help to determine the extent and minimize soil and water pollution. Reclamation activities using topsoil, amendments such as fertilizer and organic matter, and revegetation are important practices that need to be more widely adopted worldwide. Such techniques improve microbial interactions and processes that can decrease the availability of heavy metals. Phytoremediation is another process that has demonstrated good success in mitigating pollution. Biochar is capable of improving soil properties in an environmentally sustainable and cost-effective way, making it an excellent strategy for reclamation purposes.

Potential areas for future research include the following:

(1) Countries should identify and categorize the extent of mining-related disturbances, evaluate the pollution type and extent, and follow established guidelines for assessments and ecological remediation in mining regions.

(2) Coal consumption and transportation activities within the mining area can result in the release of other elements of concern into the environment. Two elements of concern, Hg and Tl, can have detrimental effects on ecosystems and pose a potential risk to human health through bioaccumulation in the food web. However, these hazardous metals, as well as others have not received attention in studies conducted around coal mining areas. Therefore, it is recommended to consider these elements in particular in future studies.

(3) Only a few studies focused on the remediation of coal-mine-degraded lands using the application of biochar. Further knowledge is needed to increase its use for reclamation and soil pollution remediation.

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