

# Progresses in restoration of post-mining landscape in Africa

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**Abstract** Mining alters the natural landscape and discharges large volumes of wastes that pose serious pollution hazards to the environment, to human health and to agriculture. As a result, the recent 2 decades have witnessed a global surge in research on post-mining landscape restoration, yielding a suite of techniques including physical, chemical, biological (also known as phytoremediation) and combinations. Despite the long history of mining in Africa, no systematic review has summarized advances in restoration research and practices after mining disturbance. Thus, the aim of this review was to document the state-of-knowledge and identify gaps in restoration of post-mining landscape in Africa through literature review. We found that: (1) there has been substantial progress in identifying species suitable for phytoremediation; (2) few studies evaluated the feasibility of organic amendments to promote autochthonous colonization of mine wastelands or growth of planted species; and (3) restoration of limestone quarries in Kenya, sand mining tailings in South Africa,

and gold mine wasteland in Ghana are successful cases of large-scale post-mining restoration practices in Africa. However, the pace of post-mining landscape restoration research and practice in Africa is sluggish compared to other parts of the global south. We recommend: (1) mainstreaming the restoration of mine wastelands in national research strategies and increased development planning to make the mining sector “Green”; (2) inventory of the number, area, and current status of abandoned mine lands; (3) expanding the pool of candidate species for phytostabilization; (4) further evaluating the phytostabilization potential of organic amendments, e.g., biochar; (5) assessing the impacts of mining on regional biodiversity.

**Keywords** Phytoremediation · Phytostabilization · Reclamation · Remediation · Tailing dams

## Introduction

Mining is a major economic venture in many parts of the world. In Africa, mining has a long tradition and small-scale mines have been scattered throughout the continent dating back to the African Iron Age of the second century AD up to about 1000 AD. However, large-scale mining began in Africa during the colonial era (Ashton et al. 2001; Miller 2002). Since the beginning of the twenty-first century, there has been increasing development of mining in sub-Saharan Africa through direct foreign investment. Although the land use change associated with mining is relatively small compared to logging or conversion for agriculture, its negative impacts are long lasting (Chen et al. 2015). In addition, the adverse impacts of mining are expected to increase with increasing mined area, mainly in the global south (Limpitlaw and Woldai 2000; Cooke and

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Johnson 2002; Kangwa 2008). The most notable impact of mining is the change in land form caused by clearing of vegetation, removal of topsoil and disposal of large amounts of waste. Mine wastes usually include waste rock, overburden, slag, and tailings on land surfaces, while mine wastelands are comprised of stripped areas, open-pits, loose soil piles, waste rock and overburden surfaces, subsided lands, tailings dams and other lands degraded by mining facilities (Wong 2003; Li 2006; Sikaundi 2013; Venkateswarlu et al. 2016).

Surface mining, which creates tailings dams, has the biggest impact on surrounding areas due to its relatively great volumes of material moved (Lin et al. 2005). For instance, to produce one ton of copper, 350 tons of waste are generated, of which 147 tons are tailings (Kangwa 2008). In Zambia, about 9125 ha of land is estimated to contain 791 million tons of tailings, while 20,646 ha of land are covered by 1899 million tons of overburden, 388 ha are covered by 77 million tons of waste rock, and 279 ha of land are covered by 40 million tons of slag in the Copperbelt Province alone (Sikaundi 2013). At global scale, between 5 and 7 billion tons of tailings dams are created annually (Edraki et al. 2014). In the absence of adequate mining closure management, metalliferous mine tailings and overburden materials pose serious hazards to human health and agricultural productivity through surface or groundwater pollution, offsite contamination via aeolian dispersion and water erosion, and uptake by vegetation and bioaccumulation in food chains (Juwarkar et al. 2009; Chaturvedi et al. 2012; Kuter 2013).

Restoration of mine wastelands has been a subject of much research worldwide for the past 4 decades. For instance, a search of the Scopus database on “phytoremediation” (one form of restoration of mine wasteland) returns 9698 hits. On the global scale, research output in phytoremediation has been increasing at a faster rate than other restoration techniques over the past decade, particularly in Southeast Asia (Koelmel et al. 2015). This large and spatially scattered body of knowledge has been systematically reviewed, covering a range of issues from ecological impacts (Venkateswarlu et al. 2016), concepts and applications (Ali et al. 2013), to restoration challenges (Pietrzykowski 2015; Mahar et al. 2016; Nirola et al. 2016) and potential restoration techniques (Rajkumar et al. 2012; Paz-Ferreiro et al. 2014; Sarwar et al. 2017). National reviews of ecological restoration of mine wasteland are also available, e.g., for China (Li 2006) and Ghana (Mensah 2015). To our knowledge, no systematic review is available for research and practices in post-mining landscape restoration in Africa despite the long history of mining, and the many copper and gold mines in the Democratic Republic of Congo (D.R Congo), and the presence in Zimbabwe and Zambia of historical abandoned

mine sites. Such a review is needed to guide restoration researchers, practitioners, and decision-makers to achieve the “Green Economy” goal set by the United Nations Environment Program (Twerefou 2009). Thus, the aim of this review was to document the state-of-knowledge and identify gaps in ecological restoration of mine wastelands in Africa through literature review. This review is organized in five sections as follows: characteristics of mine wastelands, environmental and social impacts of mine wastes, overview of ecological restoration of mine wastelands, the state-of-knowledge in Africa, and finally conclusions and recommendations.

## Characteristics of mine wastelands

### Types of mine wastelands

There are two types of mining, underground and surface mining (Northey et al. 2013), and the process starts by stripping and/or destroying the vegetation and removing the topsoil and overburden to varying extents (Cooke and Johnson 2002; Gathuru 2011; Mensah 2015). Depending on the quality of the ore, the processing method is either through pyrometallurgical or hydrometallurgical processing (Northey et al. 2013). While pyrometallurgy involves the use of thermal treatment of mineral ores, hydrometallurgy involves the use of aqueous chemistry for the recovery of metals from ores; the latter generates substantial amounts of effluents. Mining generates huge amounts of wastes in the form of coarse rock and very fine grained particles in tailings dams (Lottermoser 2010). These mine wastes have generally poor water holding capacity, low organic matter content, low nutrient content, low microbial activity, and elevated levels of heavy metals (Krzaklewski and Pietrzykowski 2002; O’Dell et al. 2007). They also typically have characteristics as described below.

### Waste rock

Waste rock contains mineral concentrations too small to be of interest for extraction of minerals or metals (Rankin 2011), and the waste dumps are made of heterogeneous, coarse-grained rock that is stored at the mine site (Krzaklewski and Pietrzykowski 2002; Rankin 2011; Broda et al. 2015). The internal structure of the dump has a major impact on the local environment and ground water as a result of acid mine drainage (AMD) from the metal sulphides present in the waste dump (Naicker et al. 2003; Franks et al. 2011; Broda et al. 2015). Waste rock dumps occupy large areas and are of environmental concern because of the AMD (Franks et al. 2011; Likus-Cieřlik

et al. 2017). Some waste rock is stored within tailings dams to prevent AMD (Rankin 2011).

**Overburden material**

Overburden includes soil and rock that are removed to gain access to ore deposits (Rankin 2011; Vela-Almeida et al. 2015). Topsoil can be excavated to depths of 30 m or deeper (Carrick and Krüger 2007), and the removed topsoil is either stored in dumps surrounding the mine operations (Franks et al. 2011), stored at the mine site for use in reclamation, or it is used elsewhere (Sheoran et al. 2010). Storing the soil at the mine site may lead to a loss of organic carbon due to exposure to heat, drying, and, in some cases, freezing–thawing, as well as reduced nutrient cycling (Mensah 2015), and lower inputs of nutrients (Gathuru 2011). Overburden is nutrient poor and deeply excavated soils can be phytotoxic (Table 1), thus these are not suitable for reclamation without amendments (Carrick and Krüger 2007) but are often used for landscape contouring (Rankin 2011).

**Tailings dams**

After the extraction and beneficiation of the minerals, the residuals and mill rejects are combined into a form of slurry and disposed into lagoons, creating tailings dams. Tailings are often materials of very fine grain size but grain size can differ depending on the parent rock, and range in diameter from < 2 mm to 625 µm (Edraki et al. 2014; Kossoff et al. 2014). Fine particle sizes often lead to compaction, low infiltration rates, and high bulk densities (Wong 2003; Titshall et al. 2013; Mensah 2015). As the dams are man-made, the soils are very young and are characterised by their instability and lack of cohesion (Asensio et al. 2013a). Furthermore, the tailings lack

organic matter (Cooke and Johnson 2002; Titshall et al. 2013), have low pH, and are often acidic (Chaturvedi et al. 2012) and toxic due to high concentrations of heavy metals, such as arsenic, cadmium, copper, manganese, lead, and zinc (Table 1).

**Environmental and social impacts of mine wastes**

Apart from altering the natural landscape, metalliferous mine tailings and stockpiled overburden pose serious pollution hazards to the environment, to human health and to agriculture. Groundwater pollution due to AMD and seepage from mine waste disposal are the most common environmental concerns (Sracek et al. 2010; Likus-Ciešlik et al. 2017). AMD is produced when sulphide-bearing material is exposed to oxygen and water (Ashton et al. 2001; Sheoran and Sheoran 2006) and releases acid, sulphate and metals. The oxidation process acidifies water in the dams, which then enters the groundwater, affecting water quality by reducing the pH and increasing contamination by heavy metals (Sracek et al. 2010). The oxidation process can go down to 5-m depth in sand tailings and down to 2 m on slime dumps (Naicker et al. 2003) with marked variations between sites and seasons of the year (Akcil and Koldas 2006; Tutu et al. 2008).

Tailings are not always acidic but can also be neutral to alkaline depending on the parent material (e.g., dolomites and limestones are alkaline). Chemicals used to “enrich” ores can render tailings and tailings waste waters saline (Krzaklewski and Pietrzykowski 2002; Karczewska et al. 2017). Thus, seepage from mine waste dumps and elevated concentrations of heavy metals and other trace metals can pollute groundwater (von der Heyden and New 2004; Krzaklewski et al. 2004; Sheoran and Sheoran 2006; Tutu et al. 2008). At many locations, e.g., the Zambian

**Table 1** Physico-chemical characteristics and concentration of heavy metals in mine wastes in Zambia (Chileshe 2014)

Mine wastes	Physico-chemical characteristics									
	Bulk density (g cm <sup>-3</sup> )	Clay content (%)	Total organic C (%)	Total N (mg kg <sup>-1</sup> )	P (mg kg <sup>-1</sup> g)	K (mg kg <sup>-1</sup> )	Mg (mg kg <sup>-1</sup> )	Ca (mg kg <sup>-1</sup> )	Na (mg kg <sup>-1</sup> )	PH
Tailings dam	1.49	5.85	1.32	500.48	5.05	35.9	853	3309.5	50	7.89
Overburden	1.44	5	1.37	500	5.27	35	111	243	50	5.72
Mine wastes	Heavy metals (mg kg <sup>-1</sup> )									
	Cu	Co	Ba	Zn	V	Pb	Ni	Cr	As	Cd
Tailings dam	12,233.33	337.5	147.08	66.08	20.42	20.17	19.83	14.23	6.21	0.60
Overburden	5708.33	136.67	272.42	49.75	26.75	5.76	21.33	20.92	1.04	0.25

Copperbelt region, tailings dams are often built near human settlements where the contaminated groundwater can pose a threat to farming communities (von der Heyden and New 2004).

Aeolian dispersion and water erosion of mine wastes are major sources of soil contamination by heavy metals around mine wastelands (Krzaklewski et al. 2004). Concentrations of heavy metals are typically greatest at distance up to 100 m from the edge of waste dumps, but can be spread over distances up to 2 km from wastelands (Kuter 2013). This dispersion is often because the dam walls are barren and the impoundments are often on floodplains where the surfaces of the dumps are susceptible to erosion by water during the rainy season (Kuter 2013). In addition, when volatile carbonyl combines with atmospheric water vapour, it is precipitated as acid rain, which exacerbates AMD and further pollutes the soil (Tembo et al. 2006; Twerefou 2009). Soil contamination is typically accompanied by reduced diversity of microbe communities (Wong 2003; Wang et al. 2007; Mensah 2015), or by their reduced abundance, species richness and activity (Liao and Xie 2007). The abiotic stress caused by heavy metal pollution affects the growth, morphology and metabolism of microorganisms (Mummey et al. 2002; Liao and Xie 2007), and reduced microbial activities result in poor soil quality (Wang et al. 2007). Soils contaminated by elevated concentrations of heavy metals also restrict growth for all but the most tolerant plants (Wong 2003), leaving mine wastelands devoid of vegetation for extended periods of time. Furthermore, mining sometimes lead to social conflicts (Venkateswarlu et al. 2016) although many countries have imposed liabilities on mining companies, inter alia, re-vegetation of mine wastelands before their operations are decommissioned. The wind erosion from tailings together with the release of fumes from the mining process is another environmental concern associated with mine wastes (Ashton et al. 2001; Twerefou 2009; Lottermoser 2010). As the metals are non-biodegradable, the impact can be long lasting (Tembo et al. 2006) and hazardous to human health as well as cattle and wildlife.

## Ecological restoration of post-mining landscapes: an overview

### Ecological restoration definition

Ecological restoration, as defined by The Society for Ecological Restoration (2002) is “the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed”. In the literature, the terms restoration, rehabilitation, reclamation and remediation are often used interchangeably (Seabrook et al. 2011). While

rehabilitation is the reparation of ecosystem processes, productivity and services without necessarily achieving a return to pre-disturbance conditions, reclamation is the physical stabilization of the terrain to a non-erosive state, and remediation is the process of correcting a specific problem and thereby reversing or stopping damage to the environment (Table 2). Within the mining context, restoration is synonymous with rehabilitation and is defined as progression toward the recovery of the original ecosystem (Lima et al. 2016), or to a novel ecosystem when the biotic and abiotic changes have been too extreme (Hobbs et al. 2009; Pietrzykowski 2015). In other words, it is a process by which the impacts of mining on the environment are repaired through reconstruction of a stable land surface followed by revegetation or development of an alternative land use on the reconstructed land form. We use the term restoration in this review as it encompasses both reclamation and remediation and its use is more established in the literature.

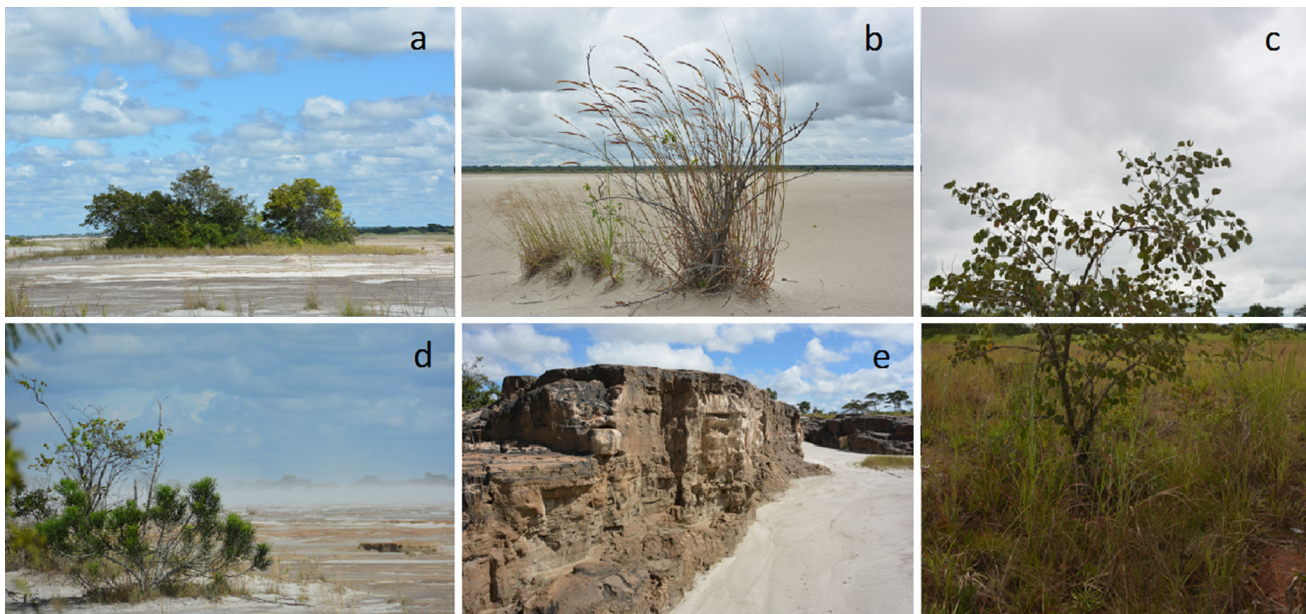
Restoration of forested landscapes after severe mining disturbance poses substantial challenges. These can range from re-creating land-form complexity and redeveloping soil types that develop slowly over long time periods in natural systems, to establishing structurally and functionally complex forest ecosystems (Macdonald et al. 2015). Available evidence has shown that autochthonous colonization (passive restoration) can deliver fully developed and functional ecosystems (Leteinturier et al. 2001; Weiersbye et al. 2006) although the succession process is slow (Bradshaw 1997; Fig. 1). For instance, we recorded 30 woody species on copper mine tailings dams in Zambia that were abandoned in 1950 and 1988, compared to 55 species in the nearby natural forests (Table 3). To achieve the restoration goal within a reasonable timeframe, restoration of mine wasteland often requires human intervention (active restoration), especially for the metal-mined tailings dams where growth conditions are harsh due to soil contamination. However, the choice of restoration approach (passive vs. active) depends on ecosystem resilience, goals for restoration, landscape context, and projected costs of restoration (McIver and Starr 2001; Holl and Aide 2011). Techniques for post-mining restoration can be broadly classified into physical, chemical and biological (Table 4). A brief account of each method is provided below.

### Physical method

The physical method focuses on re-creating the land form by ploughing, grading, smoothing, and placement of topsoil (Seenivasan et al. 2015). Soils on tailings dams are often nutrient poor, acidic/alkaline and of very poor quality (Adriano et al. 2004), hence adding topsoil can be a

**Table 2** Definitions and explanations of different restoration-related concepts. *Source:* Kuter (2013), Bozzano et al. (2014) and Lima et al. (2016)

Concept	Definition
Remediation	To remedy is “to rectify, to make good”. The process of correcting a specific problem, reversing or stopping the damage to the environment
Reclamation	To reclaim is to bring back the land to a proper state, or to provide with a suitable substitute; the physical stabilization of the terrain to bring back the land to proper state; i.e., the site will be hospitable to the original inhabitants, or those similar to the original ones; the pre- and post-disturbance land uses are nearly the same. Similar to restoration, but focuses on one aspect of the ecosystem services
Rehabilitation	To rehabilitate is an act of restoring close to a previous condition or status, not expected to bring the land back to perfection, not as heathy or in an original state as a restored land; the establishment of a stable and self-sustaining ecosystem. Rehabilitated land will prevent continued environmental deterioration and is consistent with the surrounding aesthetic values. More of a managerial term, measuring costs and benefits of maintaining environmental quality and optimizing local land management capacity
Restoration	To restore is to bring back the original state or to a healthy and vigorous state; the process of rebuilding the ecosystem that existed prior to disturbance; or recreating the initial structures and dynamics



**Fig. 1** An overview of copper mine tailings in Mufulira and Kitwe, Zambia decommissioned in early 1950s and 1980, respectively. **a** Spontaneous sprouting of grasses and trees (some *Ficus* sp.) on tailings in Mufulira; **b** arrested autochthonous colonization of tailings in Mufulira; **c** Spontaneous colonization of grasses and trees on

tailings in Kitwe; **d** Wind erosion in progress on tailings in Kitwe; **e** Compacted slurry formed on tailings in Mufulira, and during the rainy season water carves put deeper paths between the formations (Photos by Emma Sandell Festin)

solution to improve the soil quality (Wong 2003; Sheoran et al. 2010). During restoration with topsoil, soil is moved in from nearby areas (Mensah 2015) or topsoil salvaged during mining is used. This approach is costly (Bradshaw 2000) and salvaged topsoil that has been stockpiled for long periods can have low nutrient and biological quality (Grant et al. 2007; Mackenzie and Naeth 2010).

Instead of adding topsoil, mine wastelands can be restored using manufactured technosols that are mine wastes amended with organic materials, such as manure,

sewage sludge, paper mill wastes, or green waste compost (Asensio et al. 2013a; Pietrzykowski et al. 2017). This approach is relatively inexpensive and is non-destructive to surrounding environments compared to the use of topsoil removed and brought in from nearby sites. Amending soils with organic residues is an effective method for increasing soil quality without requiring much treatment before application (Farrell et al. 2010; Beesley et al. 2010; Asensio et al. 2013b). The use of vegetative compost can increase microbial activity and protect against erosion

**Table 3** Autochthonous colonizers of abandoned tailings dams in the Copperbelt region of Zambia together with woody species recorded in the surrounding natural forests. *Source:* Our own vegetation survey

Species	Natural forest	Tailing dam	Species	Natural forest	Tailing dam
<i>Acacia polyacantha</i> Willd.	x	x	<i>Isobertinia angolensis</i> (Benth.) Hoyle and Brenan	x	
<i>Albizia adianthifolia</i> W.Wight	x	x	<i>Julbernardia globiflora</i> Benth	x	
<i>Albizia amara</i> (Roxb.) Boivin	x	x	<i>Julbernardia paniculata</i> Benth	x	
<i>Albizia antunesiana</i> Harms	x	x	<i>Lannea discolor</i> (Sond.) Engl.	x	x
<i>Albizia versicolor</i> Welw. ex Oliv.	x	x	<i>Marquesia macroura</i> Gilg	x	
<i>Anisophyllea boehmii</i> Engl.	x		<i>Mimusops zeyheri</i> Sond	x	
<i>Annona senegalensis</i> Pers.	x	x	<i>Monotes africanus</i> A. DC	x	
<i>Azanza garckeana</i> (F.Hoffm.) Exell and Hillc.	x	x	<i>Ochna pulchra</i> Hook	x	
<i>Baphia bequaertii</i> De Wild	x		<i>Parinari curatellifolia</i> Planch. ex Benth.	x	x
<i>Bauhinia petersiana</i> Bolle		x	<i>Peltophorum africanum</i> Sond		x
<i>Brachystegia boehmii</i> Taubert	x		<i>Pericopsis angolensis</i> (Baker) Meeuwen	x	
<i>Brachystegia floribunda</i> Benth	x		<i>Phyllanthus polyanthus</i> Pax	x	x
<i>Brachystegia longifolia</i> Benth	x	x	<i>Piliostigma thonningii</i> (Schumach.) Milne-Redh	x	x
<i>Brachystegia spiciformis</i> Benth	x		<i>Pseudolachnostylis maprouneifolia</i> Pax	x	
<i>Brachystegia taxifolia</i> Harms	x	x	<i>Psidium guajava</i> L.	x	x
<i>Byrsocarpus orientalis</i> Baill	x		<i>Pterocarpus angolensis</i> DC	x	
<i>Cassia abbreviate</i> Oliv		x	<i>Rhus longipes</i> Engl.	x	x
<i>Combretum collinum</i> Fresen	x		<i>Rothmannia engleriana</i> (K. Schum.) Keay	x	
<i>Combretum molle</i> R.Br. ex G.Don, Engl. and Diels	x	x	<i>Strychnos innocua</i> Delile	x	
<i>Combretum zeyheri</i> Sond	x	x	<i>Strychnos spinosa</i> Lam.	x	
<i>Dalbergia nitidula</i> Welw. ex Baker	x		<i>Swartzia madagascariensis</i> (Desv.) J.H. Kirkbr. and Wiersama	x	
<i>Dichrostachys cinerea</i> Wigth and Arn	x	x	<i>Syzygium guineense</i> (Willd.) DC.	x	
<i>Diospyros batocana</i> Hiern	x		<i>Syzygium cordatum</i> Hochst. ex. C.Krauss	x	x
<i>Diplorhynchus condylocarpon</i> (Müll. Arg.) Pichon	x	x	<i>Terminalia stenostachya</i> Engl. and Diels	x	x
<i>Dombeya rotundifolia</i> Planch		x	<i>Toona ciliata</i> (exotic) M. Roem.		x
<i>Erythrina abyssinica</i> Lam. ex DC	x		<i>Uapaca kirkiana</i> Müll	x	
<i>Erythrophleum africanum</i> (Welw. ex Benth.) Harms	x		<i>Uapaca nitida</i> Müll	x	
<i>Ficus capensis</i> Thunb	x	x	<i>Uapaca sansibarica</i> Pax	x	
<i>Ficus craterostoma</i> Mildbr. and Burret	x	x	<i>Vitex doniana</i> Sweet	x	
<i>Ficus sycomorus</i> L.	x	x	<i>Vitex trifoliata</i> L.		x
<i>Garcinia livingstonei</i> T. Anderson	x				

(Ruttens et al. 2006; Carlson et al. 2015) as the compost decreases the concentration of contamination within the soil and creates a more suitable growing environment for plants. Technosols manufactured with addition of organic materials can increase water holding capacity and nutrient concentrations of technosols and increase the nutrient concentrations of ryegrass grown on technosols (Watkinson et al. 2017). The application of compost and/or limestone on degraded technosols has proven to be most effective in reducing the uptake of heavy metals and their

negative impacts on plant fecundity (Shutchka et al. 2015) as well as in improving crop yields.

However, not all organic materials are equally good for manufacture of technosols. For instance, technosols manufactured with primary paper sludge have higher pH and hence results in lower shoot biomass than do technosols manufactured using woody residuals (Watkinson et al. 2017). Similarly, the application of alkaline sewage sludge can result in dramatic increase of Cu phytotoxicity and enhanced uptake of Cu by red fescue (Cuske et al. 2016).

**Table 4** An overview of restoration methods and practices for mine wastelands and intended purposes

Method	Practices	Purposes
Physical	Ploughing, ripping	Re-creating the desired land form
	Contouring	Reduce erosion, run off
	Top soil addition/Organic amendments	Improving physico-chemical quality of substrate for re-vegetation
Chemical	Addition of lime	Increasing substrate pH
	Addition of fertilizers	Enhancing nutrition and plant growth
	Addition of synthetic chelates	Improving heavy metal solubility and bioavailability
	Nanoparticles	Improve soil physical and chemical properties, enhance soil fertility, stabilize soil contaminants, or reduce soil erosion.
Biological	Uses of microbes	Modifying heavy metal bioavailability in the soil and increase plant growth
	Phytoextraction	Uptake and translocation of heavy metals by hyperaccumulators
	Phytostabilization	Immobilization of heavy metals through soil amendment and planting of fast-growing species

Biochar, a low-density carbon-rich material produced by pyrolysis of plant biomass at temperatures of 300–1000 °C, has gained increasing attention recently as soil amendment (Beesley et al. 2011; Park et al. 2011; Zhang et al. 2012). Besides being a carbon source, biochar also contains high proportions of essential plant nutrients: nitrogen, phosphorus, potassium, calcium, magnesium, iron, and zinc (Forján et al. 2017) that are bioavailable for plant growth (Park et al. 2011; Sovu et al. 2011). Application of biochar can also increase pH and enhance the physical properties of soil by raising its porosity and thereby its water holding capacity (Carlson et al. 2015). Studies suggest that biochar has the potential to affect the behaviour of metals in the soil by altering their availability, solubility, transport and spatial distribution (Barrow 2012; Lebrun et al. 2017), thereby immobilizing heavy metals and reducing uptake by plants (Fellet et al. 2011; Karami et al. 2011; Park et al. 2011; Lomaglio et al. 2017). The overall objective of the physical method is to reduce erosion and soil compaction while improving soil quality, thereby creating conditions suitable for re-vegetation of mine wastelands or their conversion into other productive land uses.

### Chemical method

The chemical method mainly focuses on removing contaminants (heavy metals and metalloids) from the substrate and correcting soil pH (Mensah 2015). The current method for preventing AMD is to raise the soil pH above the threshold for iron-oxidizing bacteria. If the pH value is under 3.5, Fe(III) acts as an oxidising agent of pyrite (Tutu et al. 2008). Soil pH can be raised by adding fertilizers (Mensah 2015) such as dolomite (limestone) and by applying biological amendments such as organic waste

(Juwarkar et al. 2009; Seenivasan et al. 2015). Van der Heyden and New (2004) reported that AMD can be buffered by application of residual lime to impoundment dams.

The solubility and bioavailability of heavy metals can be improved by adding synthetic chelators such as ethylene diamine tetraacetic acid (EDTA), diethylene triamine pentaacetic acid (DTPA), and ethylene glycol tetraacetic acid (AGTA), which enhance uptake by plants (Saifullah et al. 2009; Pereira et al. 2010). The molecules of the chelators bind the metal atom, thereby increasing its concentration in soil aqueous phase and its mobility (Wu et al. 2010). In some cases, strong chelating reagents, such as Sodium-EDTA, can be used to increase the mobility of some metal ions that are strongly bonded to the soil phase and are less bioavailable. For instance, application of calcium salt ( $\text{Ca}(\text{H}_2\text{PO}_4)_2 \cdot \text{H}_2\text{O}$ ) with low solubility aided remediation of Mg-contaminated soils (Wang et al. 2015). The chemical method has, however, several limitations, including high cost for chemical reagents and machines, the need for skilled technicians, and potential for polluting ground water and adversely affecting soil quality in the event of excessive application (Wu et al. 2010). With due attention to environmental impact and cost, chelators can be applied in practice. Combined with biological methods (e.g., re-vegetation of mine wastelands), application of chelates increases metal solubility in soils, overcomes the diffusional limitation of metals in the rhizosphere, and facilitates root-to-shoot translocation of the metal (McGrath et al. 2001). Recently, application of nanoparticles has emerged as a novel technology for restoration of mine wastelands owing to their large specific surface area, reactivity, and deliverability (Liu and Lal 2012). These authors also noted that zeolites, zero-valent iron nanoparticles, iron oxide nanoparticles, phosphate-based

nanoparticles, iron sulphide nanoparticles, and C nanotubes have large potential for mine soil reclamation. For instance, Tafazoli et al. (2017) applied zero-valent iron nanoparticles (nZVI) on sites contaminated by heavy metals and observed increased removal of heavy metals and improved growth of planted tree seedlings.

### Biological method

The biological method, also known as phytoremediation, involves the use of green plants and associated microorganisms to minimize the toxic effects of potential contaminant in the environment (Mendez and Maier 2007). The natural colonization of post-mining soils is slow due to poor soil quality, thus establishing a stable plant cover is the starting point for successful restoration using biological methods (Conesa et al. 2007a, b). Phytoremediation basically includes phytoextraction—uptake and translocation of heavy metals by plants, and phytostabilization—the use of plant species as well as soil amendments to immobilize heavy metals through absorption and accumulation by roots, adsorption onto roots, or precipitation within the rhizosphere (Mendez and Maier 2008; Bolan et al. 2011).

Plants suitable for phytoremediation have two major heavy metal resistance strategies, viz. exclusion and accumulation (Baker 1981). While excluders are plants that restrict the transport of metals to the aboveground part and maintain relatively low heavy metal concentrations in shoots, accumulators translocate and accumulate high levels of metals in their above-ground parts. Within the accumulators group, hyperaccumulators can accumulate more than  $1000 \mu\text{g g}^{-1}$  of copper, cobalt, chromium, nickel and lead or more than  $10,000 \mu\text{g g}^{-1}$  manganese and zinc in their aboveground dry matter (Adriano et al. 2004; Peng et al. 2012). At global scale, more than 400 plant species are known to be hyperaccumulators (Faucon et al. 2007), and about 30 hyperaccumulators have been identified in South Central Africa (Faucon et al. 2007). Most of the research in this area has focused on grasses and shrubs while trees as accumulators are a relatively new concept. However, because trees produce larger biomass and have deeper and bigger root systems, they could be able to decontaminate soils for longer periods of time and accumulate greater amounts of heavy metals (Chaturvedi et al. 2012).

Phytoremediation techniques, notably phytoextraction, have not proven useful for large-scale applications due to their several limitations: (a) slow growth of naturally occurring hyperaccumulator species and their low above-ground biomass production; (b) the long time needed to remediate contaminated soils; (c) limited bioavailability of metals; (d) the risk of recycling of heavy metals back to the ecosystem if proper disposal mechanisms are not in place;

and (e) its limited applicability to sites containing slightly to moderately toxic concentrations of metals (Wong 2003; Ali et al. 2013; Sarwar et al. 2017). Because of these limitations of phytoextraction, phytostabilization has emerged as a sustainable “green technology” for restoration of mine wastelands (Mendez and Maier 2008; Bolan et al. 2011). Ideally, candidate species for phytostabilization restrict the transport of metals to their aboveground parts and maintain relatively low concentration of heavy metals in their shoots. Plants native to metalliferous sites are highly preferred for phytostabilization purposes, owing to their characteristic capacity to survive, grow and reproduce under such environmentally stressful conditions in contrast to other less tolerant plant species that might be introduced from other environments (Weiersbye et al. 2006; Titshall et al. 2013).

For phytostabilization to be successful, soil amendments are a prerequisite not only to improve the growing conditions for plantings but also to immobilize the heavy metals or to decrease their bioavailability so as to prevent them from leaching to ground water or entering into food chains (Erakhrumen 2007). There are several ways by which the mobility of heavy metals can be modified, including synthetic chelators, rhizosphere microbes, and biochar in combination with organic residues (Sarwar et al. 2017). For instance, addition of phosphogypsum alone resulted in more immobilization of heavy metals (zinc, nickel, lead and cadmium) than did a combined soil treatment of phosphogypsum and rice straw composite but the latter treatment significantly improved biomass production of canola (Mahmoud and Abd El-Kader 2015). Soil microorganisms associated with plants, mycorrhizal fungi and plant growth promoting bacteria, have the potential to influence heavy metal availability and uptake by plants in the rhizosphere (Rajkumar et al. 2012; Seth 2012). While fungal associations modify heavy metal bioavailability in soils through changes in the chemical composition of root exudates and soil pH (Chen et al. 2015), bacteria have been shown to alleviate toxicity of heavy metals (Farwell et al. 2007) and hence increase plant growth via reduction in ethylene production under stress, nitrogen fixation and specific enzyme activity (Glick et al. 1998).

The least expensive and most effective method for immobilizing heavy metals while increasing soil quality is the use of biochar and compost (Kumpiene et al. 2008; Farrell et al. 2010; Beesley et al. 2011). The use of vegetative compost increases microbial activity and protects against erosion (Ruttens et al. 2006; Carlson et al. 2015) while biochar immobilizes toxic heavy metals owing to its high pH, large surface area for sorption of metals, alkalinity, and ash content (Namgay et al. 2010; Paz-Ferreiro et al. 2014). A meta-analysis of recent studies on biochar responses of woody plants concluded that addition of



biochar has potential for large tree growth responses, with 41% average increase in biomass, and hence holds promise for forest restoration (Thomas and Gale 2015).

Once a site is amended, planting a mixture of tree species creates long-term vegetation cover (Singh et al. 2004a) and a less homogenic landscape. Other benefits are improved soil conditions as the deep roots of the trees lead to less compacted soils and reductions in soil bulk density (Singh et al. 2004a; Mensah 2015; wa Ilunga et al. 2015). In addition, re-establishment of native trees adapted to local conditions decreases the likelihood of tree mortality caused by native pests and pathogens (Bozzano et al. 2014). Trees also help to create new topsoil layer and increase the mass and concentrations of organic matter and available nutrients (Singh et al. 2004a). Studies of contaminated sites have proven that trees that are either pioneers or legumes show higher survival. These include species of *Albizia* (Singh et al. 2004b; Gathuru 2011), *Acacia* and *Leucaena* (Mensah 2015).

Interest in the use of transgenic plants has emerged as a novel technology for restoration of mined lands (Seth 2012). Plants could be genetically engineered through insertion of transgenes for increased bioaccumulation and degradation of metals. For instance, Farwell et al. (2007) evaluated growth performance of transgenic canola (*Brassica napus* L. cv. Westar), expressing a gene for the enzyme 1-aminocyclopropane-1-carboxylate deaminase under different flooding conditions and elevated soil nickel concentration, and found greater shoot biomass and increased nickel accumulation in transgenic than non-transformed canola under low flood-stress conditions. Introduction of a heavy metal resistance gene, ScYCF1, of yeast into poplar trees enhanced growth, reduced toxicity symptoms, and yielded higher phytoextraction capacity (Shim et al. 2013).

## Restoration research and practice in Africa

### Restoration research

“Land degradation is hindering Africa’s sustainable economic development and its resilience to climate change, but this cycle can be reversed. Africa has the largest restoration opportunity of any continent in the world—more than 700 million ha of degraded land” (World Resources Institute 2016). The contribution of mining to overall land degradation in Africa is largely unknown because numbers and areas of abandoned mine sites are not well documented. However, given the long history of mining in Africa (both legal and illegal Artisanal mining), mine wastelands are predicted to cover large areas. For instance, Venkateswarlu et al. (2016) reported ca. 6150

officially listed abandoned mines in South Africa alone, with areas contaminated by toxic and radioactive mine residues in Gauteng province, a known source of gold ore, covering 321 km<sup>2</sup>. In the Copperbelt Province of Zambia, about 30,438 ha of land are covered by tailings, overburden, waste rock and slag (Sikaundi 2013).

Research on restoration of mine wastelands is limited in Africa compared with the substantial advances elsewhere (Table 5). One of the earliest studies of restoration of mined lands in Africa was identification of plant species which are hyperaccumulators of nickel, copper and cobalt on serpentine soils in Zimbabwe and in metalliferous regions of the D.R Congo (formerly Zaire) in the late 1970s and 1980s (Reeves 2003). In D.R Congo alone, 35 plant species have been reported as cobalt accumulators and 25 species as accumulators of copper above 1000 mg kg<sup>-1</sup> (Reeves and Baker 2000), of which 12 species appeared to be hyperaccumulators for both Co and Cu (Reeves 2003). Families with more than four species include Asteraceae (5 species), Scrophulariaceae (8 species) and Lamiaceae (9 species). Species sequestering the highest concentrations of Cu include *Aeollanthus subacaulis* var. *linearis* (Burkill) Ryding (13,700 mg kg<sup>-1</sup>), *Ipomoea alpine* Rendle (12,300 mg kg<sup>-1</sup>), *Crepidiorhopalon perennis* (P.A. Duvign.) Eb. Fisch. (9322 mg kg<sup>-1</sup>) and *Haumaniastrum katangense* (S. Moore) P.A. Duvign. and Plancke (9222 mg kg<sup>-1</sup>), while *H. robertii* (Robyns) P.A. Duvign. and Plancke (10,232 mg kg<sup>-1</sup>) and *A. subacaulis* var. *linearis* (5176 mg kg<sup>-1</sup>) sequestered the highest concentrations of cobalt in their shoots (Reeves and Baker 2000). Reeves (2003) reported two nickel hyperaccumulator species in Zimbabwe, *Pearsonia metallifera* Wild and *Dicoma niccolifera* Wild that sequestered concentrations of 1000–10,000 mg kg<sup>-1</sup>. Recently, the role of plant functional traits in the ecological restoration of degraded mine sites was investigated in D.R Congo. Annual life cycle, growth phenology in wet season, depth 0–10 cm of underground system, bud bank by seeds, propagule size, and dispersal mode by adhesion were found to be potential indicators for selection of species for revegetation (wa Ilunga et al. 2015).

In South Africa, Weiersbye et al. (2006) recorded 438 species that naturally colonized gold and uranium tailings dams and adjacent polluted soils. In the Central Province of Zambia, Leteinturier et al. (2001) conducted phytogeochemical investigation on lead/zinc mining-generated slag heaps covering an area of over 75 ha, and identified 39 taxa of which *Aristida adscensionis* L., *Cynodon dactylon* (L.) Pers., *Indigofera spicata* Forssk., *Melinis repens* (Willd.) Zizka and *Pennisetum setaceum* (Forssk.) Chiov. were recommended for phytostabilization. Similarly, Kambingaá and Syampungani (2012) conducted vegetation survey on tailings dams at Nkana, east of Kitwe District, Zambia, and

**Table 5** Studies on restoration of mine wastelands in different parts of Africa

Country	No. of studies	Restoration research and practices
D.R Congo	6	Characterization of species naturally colonizing old mine sites for hyperaccumulation of copper and cobalt; evaluation of plant functional traits for identifying species suitable for phytoremediation of copper-mine wasteland; a field trial on the potential of soil amendments for catalysing autochthonous colonization and growth of planted species
South Africa	3	Survey of autochthonous colonizers on gold and uranium tailings dams and the adjacent polluted soils; evaluation of phytoremediation potential of five grasses species with application of fertilizer to restore lead/zinc mine tailings; Large-scale restoration of sand mine tailings using top soil addition and additional measures to assist natural colonization
Kenya	2	Large-scale restoration of exhausted limestone quarries using a mixture of tree species and litter decomposer; evaluating phytostabilization potential of four woody species, <i>Acacia xanthophloea</i> , <i>Schinus molle</i> , <i>Casuarina equisetifolia</i> and <i>Grevillea robusta</i> , for the restoration of limestone quarries
Ghana	3	Restoration of gold mine waste land using a combination of physical, chemical and biological methods; monitoring restoration progress based on soil quality indicators and possible improvements in the future
Zambia	2	Characterization of naturally colonizing species on lead/zinc mining-generated slag heaps and copper mine tailings
Zimbabwe	2	Identifying nickel hyperaccumulators; evaluating early growth performance of three indigenous <i>Acacia</i> sp. established on nickel mine tailings amended with addition of top soil
Rwanda	1	Organic amendments of degraded Technosols on former Tantalum mining sites
Morocco	1	Evaluation of 25 species grown naturally on copper and polymetallic mining sites for their ability to accumulate copper, cadmium, lead and zinc

recorded 21 species, of which *Acacia polyacantha* Willd (33.5%), *Toona ciliata* M. Roem (21.4%), *Acacia sieberana* DC (9.9%), *Bauhinia thonningii* Schumach (9.1%), and *Peltophorum africanum* Sond (8.3%) were the most dominant species in terms of importance values for stems greater than 5 cm dbh with potential for phytoremediation of copper mine wastelands. At copper and polymetallic mining sites in southern Morocco, Boularbah et al. (2006) studied 25 species grown naturally for their ability to accumulate copper, cadmium, lead and zinc, and found that the species were hyper-tolerant but were not hyperaccumulators thus could be used for phytostabilization.

Phytoremediation on manufactured technosols was evaluated at an experimental scale. In Kenya, Gathuru (2011) evaluated growth performance of *Acacia xanthophloea* Benth., *Schinus molle* L., *C. equisetifolia* and *Grevillea robusta* A. Cunn. ex R. Br. in an exhausted limestone quarry that was backfilled with limestone mine waste in a semi-arid area on the Athi River, Kenya, between 2005 and 2008. *C. equisetifolia* had the best growth performance and also had a higher positive influence on soil properties. It was followed by *A. xanthophloea* while *G. robusta* showed poor performance and recorded the lowest growth increments. In the province of Katanga, D.R Congo, the feasibility of using three grass species (*Rendlia altera* (Rendle) Chiov, *Monocymbium cerasiforme* (Nees) Stapf, *C. dactylon* (L.) Pers), and two soil amendments (compost and lime) for the phytostabilization of soils contaminated by Cu was evaluated (Shutchta et al.

2010). Results demonstrated high survival of *R. altera* on un-amended soil, suggesting that this species is a good candidate for phytostabilization, while liming ensured survival of *C. dactylon* and increased plant reproduction and reduced copper accumulation in leaves compared to compost. In a 3-year field experiment, Shutchta et al. (2015) further evaluated the feasibility of two amendments (compost and lime) on spontaneous colonization of bare soil contaminated by copper smelting activities and growth of planted *Microchloa altera* (Rendle) Stapf in Katanga, DR Congo. Results showed that soil amendments, especially compost application, had the greatest positive effect on bare soil conditions (higher pH and nutrients and lower trace metals), which in turn facilitated natural plant establishment. Furthermore, a combined application of lime and compost was most effective in improving fecundity of *M. altera*, and reducing metal uptake and accumulation by its leaves. Boisson et al. (2016) evaluated the phytostabilization potential of seven frequently occurring Poaceae species in copper hill communities in D.R Congo. They identified *Andropogon schirensis* Hochst. ex A. Rich., *Eragrostis racemose* (Thunb) Steud, and *Loudetia simplex* (Nees) C. E. Hubb. as candidate species for phytostabilization.

In Rwanda, the application of 5 t compost ha<sup>-1</sup> (on a dry matter basis) on degraded technosols (former Tantalum mining sites) improved bean (*Phaseolus vulgaris* L.) grain yield by 156% compared to the un-mined sites (Cao Diogo et al. 2017). In South Africa, the revegetation potential of

tailings from a lead/zinc mine was investigated under glasshouse conditions using five perennial grass species (*Cenchrus ciliaris* L., *Cymbopogon plurinodis* Stapf ex Burt Davy, *Digitaria eriantha* Steud., *Eragrostis superba* Peyr. and *Fingeruthia africana* Lehm.) with three rates of inorganic fertiliser—full application rate of NPK (100:150:100 kg ha<sup>-1</sup> for NPK, respectively), half the full rate, and unfertilized pots (Titshall et al. 2013). The results showed an increase in the yield of all grass species with increasing fertiliser application rate, but the yield of *C. ciliaris* at the full fertiliser application rate was significantly higher than the other species tested, followed by *D. eriantha* and *C. plurinodis*. Concentrations of Zn in the foliage tended to be over the reported grass foliage ranges, whereas Pb concentration was within typical norms.

In Zimbabwe, early growth performance was evaluated for three indigenous *Acacia* species (*Acacia gerrardii* Benth., *Acacia karoo* and *A. polyacantha* DC) established on nickel mine tailings amended with topsoil (12,300 kg ha<sup>-1</sup>), sewage sludge (13,000 kg ha<sup>-1</sup>), and compound fertilizer (N:P<sub>2</sub>O<sub>5</sub>:K<sub>2</sub>O:S; 7:14:7:6.5 at a rate of 88.2 kg ha<sup>-1</sup>), and an untreated control (Nyakudya et al. 2011). The trial revealed no comparative advantage of amendments over the untreated control in terms of survival (ranged from 60 to 100%) for all species, except *A. gerrardii* that had lower survival in the sewage sludge-treated plot (71%) than with topsoil addition (100%), fertilizer application (100%) and the unfertilized control (100%). In terms of relative growth rate, fertilizer application was better than other soil amendments, especially sewage sludge. The study concluded that *A. karoo* had the best growth performance compared with both *A. gerrardii* and *A. polyacantha* although all three species exhibited satisfactory performance and good potential for phytostabilization of nickel mine wasteland.

### Restoration practices

There are very few cases of large-scale post-mining restoration practices in Africa. The most notable examples came from Kenya, South Africa and Ghana. In Kenya, large-scale ecosystem restoration on exhausted quarries at Haller Park, Bamburi was started in 1971 by initially planting 26 tree species on 2-km<sup>2</sup> areas of open quarries (Siachoono 2010). Three species, *Casuarina equisetifolia* Forst., *Conocarpus lancifolius* Engl. and Diels and *Cocos nucifera* L. (coconut palm) showed better survival after 6 months, however *C. equisetifolia* was identified as a better pioneer owing to its ability to tolerate salinity and dry conditions, and its nitrogen-fixation, fast growth (reaching 2 m in 6 months), and evergreen habit that enables continuous dropping and renewal of foliage. However the high tannin content of *C. equisetifolia* needles

inhibits their decomposition by micro-organisms. As a result, a millipede (*Epibolus pulchripes* Cook) was introduced to digest the needles and initiate humus formation. By the year 2000, more than 300 indigenous plant species had inhabited the open quarry without any substrate amendments while 30 species of mammals and 180 species of birds had found refugia in the park (Siachoono 2010).

In South Africa, dredge mining for heavy minerals such as rutile, ilmenite, and zircon in coastal dunes has taken place in Zululand since 1977 (Cooke and Johnson 2002). The heavy minerals are separated from the sand using a floating dredger that pumps the sand in slurry to a gravity separator, thereafter the mined sand is pumped back as tailings. The physical restoration process started by re-spreading the salvaged topsoil during sand mining on the non-toxic tailings to a depth of about 10 cm to initiate natural succession and establishment of indigenous dune forest (Cooke and Johnson 2002). To assist the natural colonization process, artificial windbreaks were erected and a mixture of fast-germinating species was directly seeded (e.g., *Helianthus annuus* L., *Sorghum* spp., *Penisetum americanum* (L.) R. Br., *Crotalaria juncea* L.). After the nurse crops died, the succession progressed towards vegetation dominated by the major pioneer species (*Acacia karoo* Hayne) from the reinstated soil seed bank. With this topsoil application method, over 400 ha have been reclaimed since 1978 (Cooke and Johnson 2002).

Unlike the above two cases, a combination of physical, chemical and biological methods was carried out by AngloGold Ashanti at the Iduapriem mine at Tarkwa, Ghana (Tetteh et al. 2015a). The mining company had 110 ha of concession and started mining in 1991. Restoration work had been done on old sites that were 2, 5, 9 and 11 years old. The restoration methods involved: (1) earthwork/slope-battering to create a more visually pleasing blend of the landscape; (2) spreading of oxide material to bind the soil together and enhance soil stability; (3) topsoil amendment and use of manufactured technosols (poultry droppings and cow manure) and fertilizers over the oxide material; (4) creating crest drains to prevent runoff and control erosion; (5) establishment of cover crops such as *Puereria phaseoloides* (Roxb.) Benth. and *Centrosema pubescens* Benth. to further enhance erosion control; (6) planting seedlings of *Acacia mangium* Willd., *Gliricidia sepium* (Jacq.) Kunth ex Walp., *Leucaena leucocephala* (Lam.) de Wit and *Senna siamea* (Lam.) Irwin and Barneby followed by weeding, pruning, and fertilizer application. While the first three species are nitrogen fixers, *S. siamea* forms association with vesicular–arbuscular mycorrhiza (Tetteh et al. 2015a). Revegetation could improve fertility of degraded mined lands but it required longer periods to restore the fertility to approximations of the original levels (Mensah 2015; Tetteh et al. 2015b).

## Conclusion and recommendations

Mining in Africa has a long tradition and has generated large areas of unrestored mined lands. However, the actual numbers and areas of mine wastelands remain poorly documented. Most studies of post-mining landscape restoration in Africa have focused on identifying native species that have potential for restoration of metalliferous sites. Passive restoration (autochthonous colonization) has been reported for most parts of studied sites but the environmental cost can be high due to the slow process of natural revegetation and succession. With the recent knowledge of plant species that can accumulate or exclude heavy metals and the positive role of organic amendments, African wastelands can be restored to functioning ecosystems, as demonstrated by case studies in Kenya, South Africa and Ghana. The success of post-mining landscape restoration, particularly phytostabilization, relies on planting pioneer and nitrogen-fixing native species after site amendments to immobilize the migration of heavy metals and improve the nutrient availability and soil structure.

Generally, the pace of post-mining restoration research and practice in Africa is sluggish compared to other parts of the global south. This highlights the limited attention given to post-mining landscape research and progressive restoration practices claimed by many mining companies and regulatory departments. Thus, mainstreaming restoration of mine wastelands in national research strategies, development planning, and strict implementation of environment policy is needed to make the mining sector “green”. There are several gaps in restoration of post-mining landscapes in Africa that need to be addressed:

1. Number of abandon (dormant) mined lands, their areas and current status must be inventoried to provide a foundation for developing and implementing appropriate restoration plans and monitoring their outcomes;
2. The current pool of species proven suitable for phytostabilization is few and specific to particular metal types (copper and cobalt). Further screening of candidate species should be undertaken. Breeding programs for selecting highly productive clones under the prevailing growth conditions of the abandoned mine sites should be initiated;
3. The potential of site amendment measures, particularly biochar alone or in combination with other organic amendments, for phytostabilization remains poorly evaluated in Africa. As biochar can be produced from readily available and inexpensive bio-resources, its use will be more financially attractive. Thus dose–response trials using both metalliferous and non-metalliferous species should be tested under field conditions;
4. Adverse impacts of mining on regional biodiversity are poorly documented. Pre-mining inventory of species composition should be integrated into the permitting process for mining concessions. In addition, collection and preservation of propagules for future restoration purpose should be considered and evaluated if feasible.

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