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Role of microorganisms in rehabilitation of mining sites, focus on Sub Saharan African countries

*O Bruneel¹, N Mghazli², L Sbabou², M Héry¹, C Casiot¹, A Filali-Maltouf²

¹ HydroSciences Montpellier, Univ Montpellier, CNRS, IRD, Montpellier, France

² Equipe de Microbiologie et Biologie Moléculaire, Université Mohammed V, Rabat, Maroc

*Corresponding author. Odile BRUNEEL. Laboratoire HydroSciences Montpellier, UMR5569, Université de Montpellier, CC0057 (MSE), 16, rue Auguste Broussonet, 34090 Montpellier, France

Phone: (+33)4 67 14 39 31; Fax: (+33)4 67 14 47 74. Email: odile.bruneel@ird.fr

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ABSTRACT

Pollutions associated with mining activities are environmentally challenging worldwide, particularly in Sub Saharan African countries due to extreme climatic conditions, influencing the dispersion of metallic pollutants in ecosystems. Mine tailings subjected to acid mine drainage or aeolian erosion represent a major source of damages to the environment. Microorganisms play a key role in biogeochemical cycles controlling metals and metalloids fate and behavior in polluted environments. Some microorganisms accelerate the oxidative dissolution of sulfide minerals, leading to the formation of acid mine drainages. In another hand, microorganisms can favor the precipitation of toxic metals and metalloids in solution, contributing to the natural attenuation of polluted waters. These natural capacities can be exploited for the bioremediation of mine impacted sites. Microorganisms also play a key role in phytostabilisation, either indirectly by promoting plant growth or directly through metals and metalloids mobilization or immobilization in soils. Thus, microorganisms are important actors of mining sites remediation. Advances in the field of high throughput sequencing techniques have revolutionized molecular biology and opened a new era in research on microbial ecology, enabling to explore the diversity and activity of microbial communities thriving in polluted environments. In-depth understanding of microbial activity will contribute to improve the mitigation of pollutions. In this work, we review the progress made in using microorganisms to remediate acid mine drainage pollutions, with emphasis on bioremediation case studies in Sub Saharan Africa. The study also highlights the role of microbial communities in the phytoremediation of mine tailings.

Keywords: bioremediation, acid mine drainage, phytoremediation, microorganisms, plant growth promoting microorganisms

1. Introduction

For more than 6000 years, the history of humanity has been linked to the mineral substances that Human has extracted on the surface of the planet. Mining is vital for the global economy, and this is especially true for many countries in Sub Saharan Africa like South Africa, Zimbabwe, Botswana, or Ghana, where mining activities represent often the backbone of their economies (Mathe and Phiri, 2016). This is particularly true in the context of the energy transition towards a low carbon society to limit global warming. Indeed, international agreements on climate change, require important quantities of metals to create clean technologies, such as wind turbines, solar power plants or the production of new batteries to store energy, etc. (Ali et al., 2017).

The extraction of metals and metalloids generates large quantities of liquid and solid wastes. The mine wastes correspond to uneconomic materials including rock, gangue, refuse material, sediment, tailings, roasted ore or processing chemicals, etc (Hudson-Edwards, 2016). Nowadays, the extracted volume of waste is estimated, each year, at several thousand millions of tons, but these quantities are increasing exponentially due to increasing demand and the exploitation of low grade deposits (Hudson-Edwards and Dold, 2015). In the absence of extremely strict management of mining sites, these wastes are a source of significant nuisances, particularly because of the presence of highly toxic compounds such as lead, arsenic, mercury or radioactive elements like uranium. Indeed, due to the carcinogenicity, the cytotoxicity, and the mutagenicity of many metals and metalloids, they constitute a serious threat to human health and environment (Nordberg et al., 2014). This pollution pose alarming threats throughout the world to the local residents via ingestion of polluted water, inhalation of dust or consumption of contaminated crops or cattle, growing on polluted lands (Simate and Ndlovu, 2014; Hudson-Edwards, 2016; Humphries et al., 2017; Festin et al., 2018).

Two major problems facing mine industries are the formation of Acid Mine Drainage (called AMD) and the erosion of tailing wastes by wind or water. AMD, are characterized by low pH and high concentration of sulfate, iron, and toxic metals and metalloids such as lead and arsenic and are generated when the wastes from the sulfide ores come into contact with oxygen and water (Johnson and Hallberg, 2005). AMD can still occur during centuries after mine closure and tens of thousands of kilometers of groundwater, streams, lakes, or estuaries are directly impacted worldwide (Johnson and Hallberg, 2005). Besides being one of the main source of AMD, tailings contain also large quantities of fine particles of toxic metals and metalloids subjected to wind and water erosion, in the absence of vegetation due to the harsh soil conditions that often prevent plants development (Lottermoser, 2010; Karaca et al., 2018). The lack of vegetative cover will have environmental impacts that depend of climate conditions. In arid and semi arid climates, erosion of tailings will be mainly done through a combination of wind and water erosion with strong winds and heavy rainfalls. In contrast, in temperate or tropical and equatorial environments, the tailing will be more subjected to leaching leading to formation of acid mine drainage (Mendez and Mayer, 2008; Nirola et al., 2016). Furthermore, a vegetative cover could limit the infiltration of water inside tailing like see in arid and semi arid region, where evapotranspiration can limit mine drainage formation (Robinson et al., 2006).

The cost of reclamation of mining sites is substantial and extensive research has been done on this subject worldwide (Aznar-Sánchez et al., 2018). As metals and metalloids are not degradable, they can be only transformed to less toxic and/or less soluble form in soils or can be removed from solution by precipitation (Hallberg, 2010; Bolan et al., 2014). Currently, considering the amount of wastes and the need for long term treatments, conventional techniques using chemical or physical processes (e.g. chemical precipitation or filtration for AMD and soil replacement, solidification/stabilization or soil washing, for tailings) are often too expensive or quite ineffective, particularly for abandoned mining sites (Mendez and Maier,

2008; Bolan et al., 2014 ; Kefeni et al., 2017). In this context, the development of cost-effective and sustainable remediation solutions for the treatment of AMD and the remediation of tailings are a challenge and a necessity, particularly for developing countries (Simate and Ndlovu, 2014; Humphries et al., 2017).

Although invisible to the naked eye, microorganisms are essential to life on Earth. In billions of years, microbial communities, able to survive in a wide range of physicochemical stresses, have colonized almost all places on Earth including the most extreme habitats. Because of their vast array of metabolic activities, microorganisms are geoactive and play a central role in the biosphere (Gadd, 2010). They strongly contribute to the natural biogeochemical cycles of nutrients and organic matter, as well as those of metals and minerals (Whitman et al 1998; Madsen, 2011). In particular, microorganisms play central role in mineral formation or dissolution, redox transformations of metals and metalloids, methylation or demethylation, formation of organometallic complexes, etc., which may impact the mobility and toxicity of metallic elements (Gadd, 2010). Prokaryotes and eukaryotes communities have developed diverse resistance and metabolic mechanisms to deal with the presence of metals and metalloids in their environment (Gadd, 2010; Baker and Banfield, 2003). They could also play a key role in the remediation of mining sites. Indeed, different processes based on microbial iron oxidation or on sulfate reduction have been developed to precipitate metals and metalloids in AMD solution (Johnson and Hallberg, 2005). Furthermore, the purpose of phytostabilisation is to stabilize the tailings, by providing a vegetative cover enable to prevent the dispersion of potentially toxic pollutants. However, plant growth is very limited in mine waste dumps due to the very harsh conditions of these environments. Microorganisms have the capacity to promote plant growth on tailings and polluted mined soils or to immobilize pollutants inside soils. The exploitation of microbial communities may thus lead to the development of efficient, sustainable, and eco-friendly technologies and became the subject of extensive researches (Johnson and Hallberg, 2005; Ma et al., 2011, 2016). During the last decades, rapid advances in sequencing technology like Next Generation Sequencing (NGS) have greatly accelerated research in microbiology, enabling the production of large volumes of sequences with an impressive reduction of price, compared to traditional sequencing methods (Knief, 2014). The use of “omics” approaches, such as metagenomic, metatranscriptomic, combined with metaproteomic and metabolomic have also undoubtedly improved the knowledge of the fine processes at work in prokaryotes and eukaryotes communities in mining environments (Knief, 2014; Huang et al., 2016; Garriss et al., 2016; Zuniga et al., 2017). This “omics” approaches combined with more traditional techniques like isolation and physiological study, enable to address some questions that were not conceivable a few years ago, like which microorganisms are present, what are they doing or what is the relation between microorganisms or between microorganisms and plants (Knief, 2014; Mendez Garcia et al. 2015; Huang et al., 2016; Plewniak et al., 2018).

Sub Saharan African countries are characterized by a very wide range of climatic zones going from hot arid and semi arid regions to tropical and equatorial humid areas. These countries can experience extreme climatic conditions such as long droughts, heavy rains, strong winds or long rainy episodes and floods. These differences in climate could impact the formation of AMD but also their remediation and the phytoremediation of tailings. In a context of global change, these extreme climatic conditions are likely to become more pronounced in the near future. This review presents the progress made in using microorganisms to remediate acid mine drainage and to limit the pollution of mine tailings by improving phytoremediation of minewastes with a particular insight for the mining activities in Sub Saharan Africa.

2. Acid mine drainage

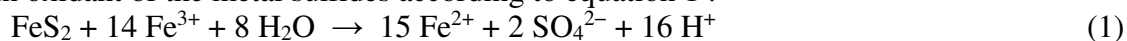
2.1 Implication of microorganisms in formation and remediation of acid mine drainage

2.1.1 Implication of microorganisms in formation of acid mine drainage

Many metals naturally occur as sulfide ores (e.g. Pb in galena, Cu in chalcopyrite, Zn in sphalerite). Acid mine drainage is generated when sulfide minerals (especially pyrite or pyrrhotite) come into contact with oxygen and water (Hallberg 2010). Although AMD occurs naturally, mining operations like excavation and milling accelerate the process by increasing surface area of exposure of sulfide minerals to water, oxygen and microorganisms (Simate and Ndlovu, 2014). AMD is often characterized by pH values below 4 and sometimes as low as -3, as measured at the Richmond mine in California (Nordstrom et al., 2000). Such low pH facilitates the mobilization of metals and metalloids, increasing their dispersion into the water (Humphries et al., 2017). AMD represents one of the most problematic pollution of water in industrial and post-industrial areas throughout the world (Hallberg, 2010).

Numerous works have been done to characterize the autochthonous microbial communities (Bacteria, Archaea, Eucaryota) thriving in these extreme environments (Baker and Bandfield, 2003; Johnson and Hallberg, 2003; Hallberg, 2010; Kuang et al., 2012; Simate and Ndlovu, 2014; Mendez Garcia et al., 2015; Huang et al., 2016; Teng et al., 2017; Plewniak et al., 2018).

The general system of formation of AMD is well known (Schippers et al., 1996; Kuyucak, 2002; Vera et al., 2013). The primary step is the oxidation of the sulfide minerals; as pyrite (FeS_2), the most abundant sulfide minerals present on Earth (Hallberg, 2010). Ferric iron is the main oxidant of the metal sulfides according to equation 1 :



This reaction produces ferrous iron, sulfate and acidity. The regeneration of ferric iron, through the oxidation of Fe(II) in Fe(III) (equation 2), is the rate determining step (Singer and Stumm, 1970).



At pH values below 4, chemical oxidation of ferrous iron is negligible (Stumm and Morgan, 1981) and, therefore, the activity of acidophilic iron oxidizing microorganisms has a pivotal role in the genesis of acid mine drainage by regenerating Fe(III) ions (Johnson and Hallberg, 2003). According to Singer and Stumm, (1970), microorganisms, such as *Acidithiobacillus ferrooxidans* or *Leptospirillum*, are able to increase the iron oxidation rate of one hundred to one million fold. Acid soluble metal sulfides like pyrrhotite can be also dissolved by proton attacks (Vera et al. 2013). Rock dissolution will continue until sulfide ores are fully weathered, which can take hundreds to thousands of years (Nordstrom, 2009).

The recovery of metals using microorganisms, called biomining or biohydrometallurgy, is used worldwide, mainly with lower grade ores. This biotechnological process represents a cost effective and more environmentally friendly approach using low power requirement compared to traditional techniques such as physicochemical processes or pyrometallurgy (Rawlings, et al., 2003; Schippers et al., 2013; Pollmann et al 2016; Sanchez-Andrea et al., 2016; Ijaz et al 2017; Deshpande et al., 2018). Biomining is mainly used to recover copper, cobalt, nickel, zinc or uranium in several countries like South Africa, Brazil, Uganda, Chile, USA or Australia (Coram and Rawlings, 2002; Vera et al., 2013; Deshpande et al., 2018). According to Johnson et al. (2013), around 20% of copper and 5% of gold are recovered through bioprocessing.

2.1.2 Implication of microorganisms in remediation of acid mine drainage

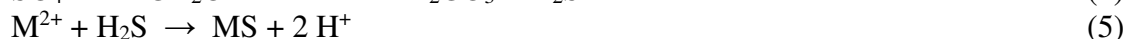
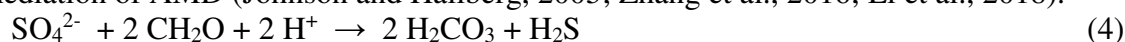
Despite the dramatic effects of these toxic effluents, natural attenuation of AMD has been observed around the world, as for example at the Carnoulès mine site in France (Casiot et al., 2003; Egal et al., 2010), in Spain (Asta et al. 2010), Japan (Fukushi et al., 2003), Australia (Gault et al., 2005), Cuba (Romero et al., 2010), Taiwan (Chen and Jiang, 2012) or Zambia

(Sracek et al., 2011). This phenomenon generally relies on iron oxidizing microorganisms in aerobic conditions or sulfate reducing bacteria (SRB) in anaerobic condition (Johnson and Hallberg, 2005; Hallberg, 2010). Because AMD generally contains high concentrations of soluble Fe(II), and that AMD contain oxygen, natural remediation phenomenon is mainly based on iron oxidation and precipitation (according to equation 3), followed by the adsorption of soluble metals and metalloids to the ferric minerals formed comprising a variety of iron (oxyhydr)oxides and hydroxysulfates such as jarosite, schwertmannite, ferrihydrite or goethite (Ferris et al., 2000; Morin et al., 2003; Johnson and Hallberg, 2005; Rowe and Johnson, 2008; Hallberg, 2010; Klein et al., 2013).



Ferric iron is generally insoluble above pH 2.5 (Johnson and Hallberg, 2005).

Under anaerobic conditions, sulfate reducing bacteria catalyze the reduction of sulfate to hydrogen sulfide (H_2S , equation 4) which can react with dissolved metals and metalloids (noted M in equation 5) to form insoluble metals sulfides. This process can also be exploited for the remediation of AMD (Johnson and Hallberg, 2005; Zhang et al., 2016; Li et al., 2018).



This process generates alkalinity by transformation of a strong acid (sulfuric acid) in a weaker one (hydrogen sulfide), inducing an increase of the pH (Johnson and Hallberg, 2005; Klein et al., 2013).

Other microbially mediated reactions can also be useful in remediation, such as the transformation of highly toxic and soluble metals and metalloids into less toxic and less soluble form, such as the oxidation of As (III) to As(V) (Bruneel et al., 2017) or the reduction of U(VI) to U(IV) (Newsome et al., 2014), or Cr(VI) to Cr(III) (Jobby et al., 2018).

2.2 Biological treatment relying on iron oxidation or sulfate reduction

AMD formation is controlled by a wide range of factors including climatic conditions (temperature, presence of water, etc.), characteristics of the tailings (geology, composition and permeability, size of the particles, presence of neutralizing minerals like carbonate, etc.) or type of microorganisms. Thus treatment of AMD may differ from one mining site to another (Schipper et al., 2010; Simate and Ndovlu, 2014; Anawar, 2015). Another crucial point making their treatment challenging is that AMD must be treated continuously over very long period, from decades to centuries. Currently, most of the AMD are treated by precipitation with hydroxide or limestone due to the simplicity and the quite low cost of the process, particularly in active mines. However, the cost of precipitating agents can be elevated, the large amount of bulky and wet waste materials causes important disposal problems and generally only induce a partial removal of sulphates (Fu and Wang, 2011; Kiran et al., 2017; Kefeni et al., 2017; Fernando et al 2018). Other physical or chemical processes (e.g. reverse osmosis, ion exchange, filtration, or electrodialysis) remain expensive, and are still scarcely used (Fu and Wang, 2011; Anawar, 2015).

Biological treatment, exploiting microbial processes occurring in natural systems, appeared as a more economic and eco-friendly alternative to classical physico-chemical methods (Johnson and Hallberg, 2005; Kieu et al., 2011). Furthermore, the sludges formed by biological activity are more efficient to retain toxic elements than similar precipitates formed abiotically (Gonzalez-Contreras et al., 2010). Therefore, these processes are increasingly being considered as viable long term treatment options, mainly in abandoned mines. Different techniques have been developed to enhance the efficacy of these biological methods, as outlined in several reviews on the subject (Kuyucak, 2002; Kalin, 2004; Johnson and Hallberg, 2005; Cohen, 2006; Mayes et al., 2009; Gadd, 2010; Akcil and Koldas, 2006; Sheoran and Sheoran, 2006; Neculita et al., 2007; Sheoran et al., 2010; Rayu et al., 2012; Klein et al., 2013; Johnson, 2014; Dixit et al., 2015; Panda et al., 2016; Kefeni et al. in 2017; Skousen et al., 2017; Kiran et al., 2017; Thavamani et al., 2017; Li et al., 2018; Fernando et al., 2018; Marques, 2018, etc.). In this section, we will focus on biological treatment of acid mine drainage based on (i) the precipitation of Fe oxides and hydroxides catalyzed by Fe oxidizing microorganisms in aerobic conditions and (ii) the precipitation of metal sulfides catalyzed by sulfate-reducing bacteria in anaerobic conditions.

AMD remediation systems are generally divided into active and passive treatment. Active treatments require constant maintenance and power. On the contrary, passive treatments require less human interventions and are thus generally less expensive and more adapted to remote locations (Johnson and Hallberg, 2005; Kefeni et al., 2017).

Biological treatments applied to AMD have been often implemented in the form of constructed wetlands (Gazea et al., 1996; Sheoran and Sheoran, 2006; Mayes et al., 2009; Zhang et al., 2010; Lizama et al., 2011; Guittonny Philippe et al., 2014). According to Roychowdhury et al. (2015), wetlands represent one of the most widely used passive AMD treatment system around the world. In USA alone, around 400 constructed wetland water treatment systems have been built (Perry and Kleinmann, 1991). Ziemkiewicz et al. (2003) evaluated different passive systems to remediate AMD in U.S. The comparison of 11 anaerobic wetlands and 16 vertical flow wetlands with other physicochemical processes highlighted good results for the wetlands with only one system not efficient to reduce acid load. Natural or constructed wetlands and ponds are open air systems planted with macrophytes (i.e. reeds) that passively treat water flow. Wetlands are complex ecosystems combining physical, chemical and biological processes (Kuyucak, 2002; Akcil and Koldas, 2006). They are classified as aerobic, anaerobic or combined wetlands (Kuyucak, 2002). The removal mechanisms identified in natural and constructed wetlands include oxidation and precipitation of ferric oxides, adsorption and complexation of metals and metalloids by organic matter, sedimentation,

filtration of suspended and colloidal materials, active plant uptake and sulfate reduction followed by precipitation of metals and metalloids as sulfides (Kuyucak, 2002; Mayes et al., 2009). The performance of these systems depends on various parameters such as concentrations of metals and metalloids, hydraulic retention time, temperature, bed media, flow design, sizing aspect and activity of microbial communities (Kuyucak 2002; Weber et al., 2010). These systems are particularly efficient in remediating acid mine drainage and appeared to be a sustainable and low cost systems both in terms of operating and capital costs (Kuyucak, 2002; O'Sullivan et al., 2004; Sheoran and Sheoran, 2006; Neculita et al., 2007; Dean et al., 2013; Humphries et al., 2017). The drawbacks of wetlands are: (i) the important space requirement making them unsuitable in some sites; (ii) the important fluctuations of performances; (iii) the fact that valuable elements are precipitated together with elements with no value.

Contrary to wetlands, bioreactors are designed to optimize treatment process efficiency in order to limit the space required. Their compact design makes them easier to use and to control (Neculita et al., 2007; Kiran et al., 2017). These systems are generally active or semi passive for higher performance or for combining water treatment and selective metal recovery. These biological treatments are very promising but more works are needed to improve them and identify the operational, environmental and biological factors that control their performance (Heinzel et al., 2009a, b; Ahoranta et al., 2016; Fernandez-Rojo et al., 2017, 2018, 2019; Laroche et al., 2018). Furthermore, bioreactors are suitable for the recuperation of element of commercial interest, so AMD may also be considered as a potential reserve of valuable resources (Sanchez-Andrea et al., 2016). The most common elements recovered in AMD include ferric hydroxide, barium sulphate, gypsum, sulphur or sulphuric acid and some rare earth metals (Kefeni et al., 2017). Furthermore, some nanoscale biogenic materials and minerals could also be used for their optical properties, or electronic functionalities (Costa et al., 2013, Le Pape et al., 2017). Recovery of such elements help the development of more sustainable mining and the reduction of environmental pollution in a world where the pression on mineral resources is increasing (Kefeni et al., 2017). The table 1 presents the advantages and drawbacks of using bioremediation processes of acid mine drainage.

Table 1. Advantages and drawbacks of using bioremediation processes.

Biological Treatment	Advantages	Disadvantages	References
Aerobic Wetlands	<ul style="list-style-type: none"> • Cost effective, with very limited maintenance • Adapted to remote locations • Ecofriendly • Currently used worldwide • Can be very efficient in metals removal • Denser, less voluminous and more stable sludges/technical process • Reduce or eliminate the use of hazardous chemicals • Well accepted by social communities with quite natural and aesthetic which support plants and wildlife 	<ul style="list-style-type: none"> • Important space requirement • Fluctuations of performances from one site to another • No recovery of valuable elements • Net increase of acidity, thus need to be combined to systems raising pH downstream 	Kalin, 2004 Kuyucak, 2002 Mayes et al., 2009 Johnson and Hallberg 2005 Roychowdhury et al., 2015 Klein et al., 2013
Aerobic bioreactors	<ul style="list-style-type: none"> • Compact system • Easier to control/ wetlands • Generally semi passive system, with low maintenance • Possibility of recovery of valuable products like ferric hydroxide, sulphur or sulphuric acid, etc. • Can be very efficient in metals removal • Denser, less voluminous and more stable sludges/technical sludges process • Reduce or eliminate the use of hazardous chemicals 	<ul style="list-style-type: none"> • Quite important variations from one site to another probably explained by different operational conditions • Net increase of acidity, thus need to be combined to another system raising pH downstream 	Johnson and Hallberg, 2005 Hedrich and Johnson, 2012 Sheng et al., 2017
Anaerobic Wetlands	<ul style="list-style-type: none"> • Cost effective, with very limited maintenance • Adapted to remote locations • Ecofriendly, well accepted by social communities • Currently used worldwide • Can be very efficient in metals and sulfate removal • Produce alkalinity • Metals sulphides less voluminous and less soluble than biotic metals hydroxides • Reduce or eliminate the use of hazardous chemicals 	<ul style="list-style-type: none"> • Important space requirement • Fluctuations of performances from one site to another • Rapid influx of acidic and/or toxic and/or aerobic waters could dramatically inhibit treatment capacity • No recovery of valuable elements • Generally need systems to increase pH upstream • Need to be avoided in presence of mercury 	Mayes et al., 2009 Sheoran and Sheoran, 2006 Neculita et al., 2007
Anaerobic bioreactors	<ul style="list-style-type: none"> • Compact system • Easier to control/ wetlands • Generally good results when deep reducing conditions are obtained • Possibility of selective recovery of valuable metals and metalloids • More stable sludges compared to iron precipitation process • Metals sulphides less voluminous and less soluble than biotic metals hydroxides • Reduce or eliminate the use of hazardous chemicals 	<ul style="list-style-type: none"> • The cost of construction, operation and maintenance could be important • Generally active or semi passive system with regular supply of organic matter • Quite challenging conception which do not allow accurate assessment of their sizing and performance • Rapid influx of acidic and/or toxic and/or aerobic waters could cause catastrophic system failure • Generally need systems to increase pH upstream • Need to be avoided in presence of Hg 	Neculita et al., 2007 Kiran et al., 2017

2.2.1 Biological treatment process relying on iron oxidation

Aerobic Wetlands. On aerobic wetlands, shallow water allows to maximize oxidation reactions involved in the precipitation of iron as oxides and hydroxides. Like iron oxidation induces a net increase of acidity, this process needs to be combined to another system to raise the pH as for example anoxic limestone drain (Kuyucak, 2002; Mayes et al., 2009). Most aerobic wetlands are planted with macrophytes which act as a support for microbial communities, promote slow water flow, and increase the residence time. Furthermore some plants, like reeds, have the ability to transfer oxygen from aerial tissues to rhizosphere, increasing iron oxidation (Hedin et al., 1994; Batty et al., 2002; Klein et al., 2013). Diffusion of oxygen is an important factor controlling iron oxidation. For this reason, turbulent flow, generated by terraces and waterfalls, show also a remarkable efficiency like see on natural attenuation of AMD (Asta et al., 2010; Chen and Jiang, 2012). Aerobic wetlands are generally used for the removal of Fe, Mn, and Al or As from mine waters (Buddhawong et al., 2005; Macías et al., 2012; Roychowdhury et al., 2015). Such passive systems were set up in the United States, Quebec and Australia (Kuyucak, 2002). However, low temperatures are likely to limit microbial activity and thus to lower the process efficiency as mentionned by Roychowdhury et al. (2015). In Quebec, the wetland, installed since 1997 enabled to meet the regulated standards,

even under the winter conditions prevailing in Canada (Kuyucak, 2002). However, aerobic wetland generally treats AMD with near neutral pH and low metal concentrations (Kalin, 2004; Johnson and Hallberg 2005; Klein et al., 2013). The treatment of highly acidic waters remains challenging (Mayes et al., 2009). Efforts to understand the ecology of the microbial community and their influence on treatment efficiency are still needed, to improve the design and efficacy of aerobic wetlands (Mayes et al., 2009; Weber et al., 2010).

Aerobic bioreactors. Aerobic bioreactors (e.g. column type bioreactor, packed bed bioreactor, fluidized bed reactor, vertical flow reactors, etc.), exploiting biological Fe(II) oxidation, are often dedicated to remediate iron rich water with very acidic pH (Hedrich and Johnson, 2012). Different attempts have been made from the laboratory scale to the pilot scale tested on the field (Elbaz-Poulichet et al., 2006; Heinzl et al., 2009a, b; Hedrich and Johnson, 2012; Kay et al., 2013; Tischler et al., 2013; Ahoranta et al., 2016; Sheng et al., 2017; Fernandez-Rojo et al., 2017, 2018, 2019; Laroche et al., 2018; Blanco et al., 2018, etc.). Iron oxidation rates measured in these studies are promising but also very variable from one study to another. These variations can be explained by the different operational conditions (i.e. pH, concentration of Fe(II), temperature, hydraulic residence times, concentration of oxygen) or by the identity and the activity of the microbial populations, etc. (Sheng et al., 2017).

2.2.2 Biological treatment process relying on sulfate reduction

The use of sulfate-reducing bacteria is a proven technique for the treatment of mine waters and many researches have been done on this process as illustrated in different reviews (Johnson and Hallberg, 2005; Sheoran and Sheoran, 2006; Neculita et al., 2007; Kiran et al., 2017; Nancucheo et al. 2017; Li et al., 2018; Ayangbenro et al., 2018). This process enables the removal of metallic elements such as Cu, Pb, Zn, Cd, Ni or Fe, and sulfate (Kiran et al., 2017; Fernando et al., 2018). Because the microbial catalyzed reactions produce alkalinity, these systems may be used to treat acidic AMD (Johnson and Hallberg, 2005). The biogeochemical processes are based on sulfate reduction activities, mediated or accelerated by a heterotrophic microbial community fueled with organic carbon. The systems (wetland or bioreactor) are filled with organic matter to stimulate/promote the development and the activity of sulfate reducing bacteria (Cohen, 2006). SRB use easily degradable fraction of organic matter such as methanol, ethanol or lactate, however, less expensive and locally available carbon sources are tested such as waste materials originating from food or agricultural processing industry (i.e. sawdust, wood chips, byproduct of sugar processing, olive mill wastewater, winery or cheese waste, cattle manure, sewage sludge, or compost) (Neculita et al., 2007; Costa et al., 2009; Martin et al., 2009; Zhang and Wang, 2014; Carlier et al., 2019). These researches are essential to improve the economic viability of such bioremediation processes. Because SRB cannot generally directly oxidize these complex organic carbon sources, a more complex microbial community must be present to sustain the development of SRB, able to provide them with short chain organic (Kuyucak and St-Germain, 1994; Neculita and Zagury, 2008). The metal sulfides present a low solubility even at acid pH and represent a limited sludge volume (Kiran et al., 2017). They are also less soluble than metal hydroxides, thus improving sludges stability (Gazea et al., 1996; Neculita et al., 2007).

Limitations encountered in the application of SRB for AMD treatment are linked to their quite strict requirements, i.e. an anaerobic environment and a pH in the range of 6 - 8, (most SRB are inhibited by pH less than 5), but acidophilic SRB are also described (Nancucheo and Johnson, 2012, 2014). Thus, rapid influx of acidic and/or aerobic waters can sometimes inhibit sulfate reducing activity and decrease the treatment capacity (Cohen, 2006; Neculita et al. 2007; Ayangbenro et al., 2018). To overcome this problem, alkaline materials like lime, calcium carbonate, caustic soda, or soda ash may be added in some case, to increase the pH and improve

the activity of SRB (Kefeni et al., 2017). Another drawback of SRB remediation process is the toxicity of some metals and metalloids that could inhibit their activity. Moreover, in presence of mercury (Hg), the use of SRB should be avoided. Indeed, methylmercury represents the most toxic species of Hg and SRB like *Desulfovibrio* or *Desulfobulbus* have been shown to play an important role in methylmercury formation in aquatic environments (Li et al., 2018).

Anaerobic Wetlands. In contrast to aerobic wetlands, the key reactions are anaerobic. These treatment systems have been already used for the elimination of metals, radionuclides and sulfate from AMD on a large scale (Kiran et al., 2017). Numerous wetlands are built around the world (Colorado, Pennsylvania, West Virginia, Canada, South Africa and Australia) for AMD treatment but the success of the remediation is quite inconstant (Cohen, 2006; Sheoran and Sheoran, 2006; Ali, 2011). Furthermore, very little is known on the evolution of the microbial communities present in these systems as the ecosystem ages or with evolution of substrate provision (Johnson and Hallberg, 2005). A better knowledge of the intrinsic functioning of such systems with the monitoring of microbial communities could mainly improve these systems.

Anaerobic bioreactor. Different bioreactor designs have been developed e.g. anaerobic packed bed bioreactor, fixed bed bioreactor, filter bioreactor, stirred tank bioreactor, column bioreactors or passive biochemical reactors, etc. (Kiran et al. 2017). These processes achieve generally good results when deep reducing conditions are obtained (Sheoran and Sheoran, 2006; Nancucheo and Johnson, 2012). Acidophiles or acido tolerant SRB have been successfully used for the remediation of AMD at the laboratory and at the field pilot scale (Kimura et al., 2006; Kieu et al., 2011; Nancucheo and Johnson, 2012; Battaglia-Brunet et al., 2012; Kay et al., 2013; Nancucheo and Johnson, 2014; Zhang et al., 2016; Le Pape et al., 2017). The great advantage of this process is the possibility of selective recovery of metals and metalloids, which can be recycled and reused, with predictable and readily controllable performance, even at low initial concentration (Hammack et al., 1993; Huisman et al. 2006; Hedrich and Johnson 2014; Sanchez Andrea et al., 2016; Kiran et al., 2017; Li et al., 2018). However, the cost of construction, operation and maintenance of such systems could be important (Kiran et al., 2017). Furthermore, this treatment generally requires the regular supply of organic material to enhance the microbial activity due to low concentration of organic matter in AMD, thus increasing the operational cost of remediation of this system (Ayangbenro et al., 2018). According to Neculita et al. (2007), some factors are important in the conception of a successful bioreactor such as a source of organic matter and sulfate to support SRB growth, a mostly anaerobic inlet water, a solid matrix enable to provide microenvironments and to retain metal precipitates. Other parameters play also a critical role like the depth of the bioreactor, the redox potential which should be low, or the hydraulic retention time (Neculita et al., 2007; Vasquez et al., 2016). The conception of semi passive treatment that guarantees favorable conditions for SRB on the long term is very challenging. Few reports exist on systems operating on the field for long periods (more than 3 years) to assess the long term efficiency of the system (Neculita et al., 2007). A field treatment over 5 years with a system based on gravity flow is presented by Al et al., (2011). Their design included two upflow bioreactors containing a mixture of limestone, quartz sand and a by-product of the paper industry followed by three horizontal subsurface flow wetlands. This system was able to remove As, Zn and Cd from contaminated water originating from smelter waste and seemed to be controlled by interdependent microbial and abiotic reactions. Although considerable researches have been made on the design of anaerobic bioreactors, the intrinsic biology of this complex process remains poorly understood (Neculita et al., 2007; Baldwin et al., 2015; Rezadehbashi and Baldwin, 2018).

To conclude, there is no universal technology for the treatment of AMD and the choice of the treatment system depend on numerous parameters: (i) the composition and the physico-

chemical characteristic of the water (pH, concentration of metals and metalloids, etc.), (ii) the space available for the remediation, (iii) the climatic conditions and (iv) economic factors (Kefeni et al., 2017). In most cases, one method alone is not sufficient (Kuyucak., 2002). Treatment processes developed on the field generally combine chemical, physical and biological processes, with wetlands, bioreactors, or limestone drains used in parallel. Wetlands, carbonate rock drains and sulfate reducing bacteria reactors are widely used to remediate AMD (Humphries et al., 2017). However, the success of the remediation in combined system requires that all the system components work as predicted (Whitehead et al., 2005). In practice, the performance of such systems is quite variable (Clyde et al., 2016).

Passive or semi passive systems that take advantage of the topography of the site for water circulation are the most desirable combined systems. They have low operational and maintenance requirements, and necessitate little technical expertise for their operation (Cohen, 2006; Akcil and Koldas, 2006; Clyde et al., 2016; Skousen et al., 2017). These systems are particularly well adapted to remote sites but are subjected to failures if they are not well designed and constructed, especially if the systems are not adapted to the discharge flow, the pH or to the metals and metalloids concentrations (Skousen et al., 2017). Their performance is also highly dependent on temperatures and other climatic conditions (Kuyucak, 2002). Combined treatment strategies for the semi passive remediation have been implemented in Canada, United States, or the United Kingdom (Nairn et al., 2010; Clyde et al., 2016). As for other systems, a better knowledge of the microbiology and long term monitoring are necessary to assess their long term effectiveness (Clyde et al., 2016).

2.3 Biological treatment processes of acid mine drainage in Sub Saharan African countries

The formation of AMD and the spatial dispersion by water and wind of metals and metalloids originating from mining wastes are strongly controlled by the climate and the hydrology of the region (Nordstrom, 2009; Anawar, 2013, 2015). Nordstrom (2009) described a gradual increase of acidity and metals and metalloids concentrations in AMD during the long dry periods (corresponding to concentration by evaporation). Conversely, a sudden increase is observed during flushing events following the dry spells (attributed to the dissolution of soluble salts, Doumas et al., 2016). Climatic conditions also impact the weathering of tailings with different weathering profiles reported. During low water or episodic wet dry cycles, weathering profiles in the tailings are generally characterized by the accumulation of secondary iron and sulfate minerals near the surface under arid and semi arid condition whereas iron and sulfate are soluble and leached with AMD under humid climates (Dold and Fontbote, 2001; Navarro et al., 2004). Biological processes implicated in the formation of AMD and their remediation are also sensitive to weather and climatic variables. Indeed, temperature (Volant et al., 2014), rainfall (Edwards et al., 1999), or seasonality (Streten-Joyce et al., 2013) have been shown to influence the structure and the dynamics of microbial communities present in acid mine drainage. Wetland treatment systems could also be less effective in arid and semi arid climatic conditions (Kiran et al., 2017). The long term stability of remediation options is particularly vulnerable to extreme climatic conditions. For example, intense flood encountered in some Sub Saharan African countries could lead to the destruction of remediation devices, as seen elsewhere (Elbaz-Poulichet et al., 2006; Anawar, 2015). The performance of the remediation systems must also be able to handle with very high flow rates during extreme events or with important flow rates fluctuations during long rainy seasons in tropical climate (Kuyucak, 2002).

Few studies reporting remediation systems of acid mine drainage have been done in Sub Saharan African countries (Figure 1) and most of them do not address the microbiological aspects of the treatment (Table 2).

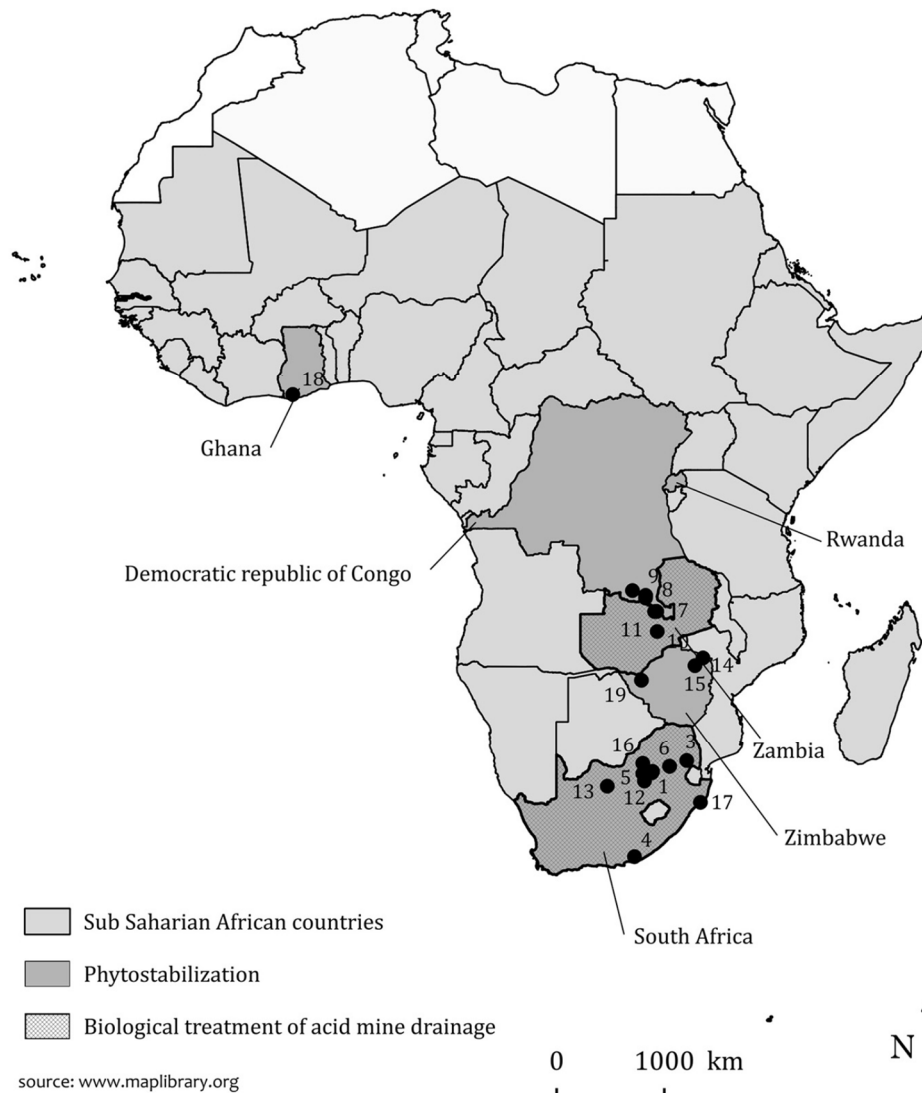


Fig. 1. Map of biological treatment processes of acid mine drainage and phytostabilization in Sub Saharan African countries

In South Africa, the attenuation of pollution arising from acid mine drainage reported for a natural wetland on the Witwatersrand, showed a considerable metal (particularly Co, Ni, Zn, Pb, Cu and U) accumulation within the wetland peats. Metal immobilization occurs through a combination of different mechanisms, including precipitation of gypsum, metal sulfides, Fe-Mn oxyhydroxides and phosphates (Humphries et al., 2017; Abiye et al., 2018).

Novhe et al., (2016) studied a passive treatment integrating aerobic and anaerobic units to treat AMD during six months. This pilot has successfully neutralized the water and substantially removed contaminants, such as Fe, Al, Co, Zn, Ni, As, Pb or V, but only 30% of sulfates. The authors identified also a decrease in treated water quality over time, probably due to clogging or depletion in the treatment materials. Molwantwa et al., (2004) investigated the mechanisms underlying the hydrolysis of a complex organic matter present in a biosulfidogenic recycling sludge bed reactor. In this laboratory experiment, the hydrolysis of this cost effective carbon substances was tested under sulfidogenic and methanogenic conditions and show better results under biosulfidogenic conditions. Maleke et al. (2014) optimized a bioremediation system where bioreduction of soluble U(VI) was facilitated by the biostimulation of an indigenous bacterial community. They succeeded to transform and to precipitate soluble U(VI) in two bioreactors (containing different concentration of uranium). Two bacterial strains belonging to

genera *Desulfovibrio* and *Geobacter*, already known to reduce U(VI), were identified. Van Hille et al. (2016) developed a pilot scale semi passive system for the bioremediation of acid rock drainage. This study demonstrated the feasibility of a novel reactor configuration, using submerged carbon microfibers to increase the surface area for microbial attachment. To limit the cost, electron donor was provided by the anaerobic degradation of complex organic carbon sources (grass, algae, manure) in a separate hydrolysis reactor. Finally, aqueous sulfide was transformed into elemental sulfur, a value adding product. Due to the successful demonstration of the bioreactor at the laboratory scale, a pilot with 3 units of 2000 l was installed at the New Vaal Colliery and functioned with very encouraging results. Marais et al. (2017) tested the effect of different operational parameters (hydraulic residence time, electron donor and reactor size) on the performance of similar semi passive bioreactors.

Table 2. Remediation processes of acid mine drainage in Sub Saharan African countries.

Map reference	Remediation method	Type of mine	Microorganisms	Country	References
1; 2	Natural wetland on the Witwatersrand	Gold mine	Not identified	South Africa	Humphries et al., 2017 ; Abiye et al., 2018
3	Pilot passive treatment plant including anaerobic and aerobic units	Coal mine	Not identified	South Africa	Novhe et al., 2016
4	Study of mechanisms underlying the hydrolysis of complex organic matter in a biosulfidogenic recycling sludge bed reactor	Laboratory experiment	Inoculation of sulphate reducing bacteria and methane producing bacteria	South Africa	Molwantwa et al., 2004
5	Bioremediation system promoting bioreduction of U(VI) based on biostimulation of indigenous bacterial communities	Gold mines containing uranium	<i>Desulfovibrio sp.</i> and <i>Geobacter sp.</i> used for bioreduction of U(VI)	South Africa	Maleke et al., 2014
6	Laboratory and pilot scale semi passive bioprocess using carbon microfibers coupled with anaerobic degradation of complex organic carbon	Acid rock drainage	Inoculation of sulphate reducing bacteria and sulphide oxidizing bacteria	South Africa	Van Hille et al., 2016
6	Semi passive pilot bioprocess using carbon microfibers for simultaneous sulphate reduction and partial sulphide oxidation within a single linear flow channel reactor unit	Acid rock drainage	Inoculation of sulphate reducing bacteria and sulphide oxidizing bacteria	South Africa	Marais et al., 2017
7	Natural remediation of neutral mine drainage coupled with wetlands	Neutral mine drainage of different mines	Not identified	Zambia	Sracek et al., 2011, 2012

In Zambia, Sracek et al., (2011, 2012) reported the natural attenuation of Fe, Cu and CO in a Copperbelt by precipitation of ferruginous ochres composed mostly of amorphous ferric hydroxide that incorporates significant quantities of metals.

The application of biological treatments in Sub Saharan African countries is still limited and has been mainly performed on a laboratory scale. Hence, there is an important need to explore the potential application of these systems for a long-term remediation of AMD with an emphasis on microbial activity to understand and better control the whole process.

3 Phytoremediation of mine tailings

3.1 Role of microbial communities in phytoremediation

Mine tailings represent a major source of multimetallic pollution from mining activity as they contain several toxic elements such as As, Cr, Cd, Mn, Se, Pb, or Tl and are generally composed of very small particles, mainly sand or even silt sized particles (Mendez and Maier, 2008). The tailings pH is generally highly acidic, although it could be alkaline depending on the carbonate content. The tailings lacks also often of soil structure, nutrients (N, P, K), and organic matter (Ernst 2005; Audet and Charest 2007; Doumas et al., 2016). Due to these stressful conditions, tailings are generally devoided of vegetative cover. The absence of vegetation is favorable to the formation of acid mine drainage and to the erosion by wind and water (Mendez and Maier, 2008).

Due to important pressure on metals and raw materials demands, and the amelioration of extraction methods, reuse of tailings could be an option, when it's economically viable (Dino et al., 2018). Treatment methods used for the remediation of soils contaminated with metals and metalloids include the soil replacement, solidification/stabilization and vitrification, soil washing, in situ flushing or electrokinetic remediation, etc. (Bolan et al., 2014; Peng et al., 2018). The main drawbacks of these methods are the use of toxics chemicals, the alteration of physical, chemical and biological capacity of soils, the long treatment time necessary for large area and the prohibitive cost. Furthermore, mine tailings represent considerable quantities of solid wastes to treat, containing very high concentrations of metals and metalloids. All these reasons make these technologies generally not applicable for the remediation of mine tailings (Glick, 2010; Bolan et al., 2014; Karaca et al., 2018; Novo et al., 2018). In this context, phytoremediation emerges as a sustainable, cost effective, environmentally friendly, solar driven technology, aesthetically pleasing and enabling to rehabilitate the landscape. This green technology also enhances soil fertility and improves erosion control and landscape rehabilitation (Novo et al., 2018). Phytoremediation represents furthermore a socially accepted technology to remediate polluted tailings and contributes to CO₂ fixation (Novo et al., 2018; Karaca et al., 2018). Many reviews have been published in phytoremediation of soils contaminated by metals and metalloids (Sheoran et al., 2010; Ali et al., 2013; Bolan et al., 2014; Paz-Ferreiro et al., 2014; Mahar et al., 2016; Sarwar et al., 2017; Muthusaravanan et al., 2018; Yadav et al., 2018; etc.), with some specifically dedicated to mine soils and tailings remediation (Ernst 2005; Pilon-Smits, 2005; Mendez and Maier, 2008; Wenzel, 2009; Pietrzykowski, 2015; Venkateswarlu et al. 2016; Karaca et al., 2018; Novo et al., 2018).

Two major types of phytoremediation are currently applicable for the remediation of mining sites, called, phytoextraction and phytostabilization.

Phytoextraction uses plants to extract metals and metalloids in mine tailings by accumulation, or often hyperaccumulation of metallic elements in the above ground plant biomass (Cunningham et al. 1995; Garbisu and Alkorta, 2001; Ernst, 2005; Novo et al., 2018). Plants need to be harvested and disposed as hazardous waste or ideally can be used for metal recovery. A high biomass production and fast growth rates are required to maximize the accumulation of metals and metalloids by the plants (Zhao et al., 2003; Turnau et al., 2007; Sheoran et al., 2012, 2016). However, phytoextraction is generally not considered as a good remediation strategy for mine tailings (Novo et al., 2018). Indeed, a low biomass production is often observed on mine tailings due to the stressful tailings environment (Keller et al. 2003; Ernst 2005), and the very high metal and metalloid concentrations would require hundreds or thousands of cropping cycles, corresponding generally to an equal number of years. In addition, because hyperaccumulators can generally accumulate only one or two metals, multiple plant species may be required to remediate polymetallic mine tailings (Ernst 2005). Many plants have also a shallow root zone; limiting the depth of metal removal. Furthermore, the extraction of toxic metals and metalloids into plant shoots, may increase the wildlife and cattle exposure to

these pollutants, a major limitation particularly in arid and semi arid areas where food is limited (Wood et al. 1995). Another limitation is the cost of implementation to optimize the process with amendments, irrigation and the costs associated with the harvesting, disposal, and reseeded on very long periods (Mendez and Maier, 2008).

Phytostabilization (also called phytocapping) focuses on the formation of a vegetative cover to contain, immobilize metals and metalloids within the plant rhizosphere (Mendez and Maier, 2008; Karaca et al., 2018). This process greatly limits their bioavailability in the surrounding environment and decreases the exposure of livestock, wildlife, and human (Cunningham et al. 1995; Tordoff et al., 2000; Bolan et al., 2011; Mendez and Maier, 2008; Karaca et al., 2018). The vegetative cover and the plant roots also contribute to the reduction of wind and water erosion. It also enables the sequestration of carbon, and contributes to maintain the food chain (Tordoff et al., 2000; Nirola et al., 2016). Phytostabilization is thus a technically sustainable and economically viable option for large areas affected by metals and metalloids pollution, like tailings (Doumas et al., 2016). However, due to stressful edaphic conditions, the mine tailings are generally not suitable for plant growth.

The necessity to boost the efficiency of phytoremediation processes in heavy polluted environments such as mining sites has prompted the researches on supporting strategies such as the use of plant growth promoting microorganisms. Studies dedicated to plant and microorganisms associations have been the subject of a growing interest due to the potential of microbial communities to increase resistance of plants to metals and metalloids and to stimulate plant growth (Jing et al., 2007; Glick, 2010; Ma et al., 2011; Rajkumar et al., 2012; Ali et al., 2013; Ahemad and Kibret, 2014; Ullah et al., 2015; Ma et al., 2016; Goswami et al., 2016; Thijs et al., 2017; Novo et al., 2018). Indeed, several plant associated microorganisms have been reported to significantly accelerate phytoremediation (Grandlic et al., 2008; Kuffner et al., 2008; Rajkumar et al., 2012; Ullah et al., 2015; Ma et al., 2016; Park et al., 2016). The plant growth promoting bacteria (call PGPB) comprising rhizobacteria and endophytic bacteria deserve particular attention. The endophytic bacteria comprise bacteria able to colonize the internal tissues of plants without inducing detrimental effects on their host. Their beneficial effects are generally more important than those induced by rhizobacteria (Ma et al., 2011, 2016). The mechanisms of plant growth stimulation differ between strains and are dependent from the metabolites released by microorganisms. Phytohormones comprise different types such as IAA (indole-3-acetic acid), ethylene, gibberellins, abscisic acid or cytokinins and are implicated in numerous primordial physiological processes including plant defense system, stimulation of germination, reproduction, protection against biotic and abiotic stresses, tissue differentiation, photosynthesis, pigment formation, etc. (Goswami et al., 2016). Microorganisms also provide nutrients like iron, a fundamental cofactor implicated in various enzymatic reactions, and considered as an essential nutrient for plants. Siderophores produced by bacteria can solubilize insoluble hydroxides and oxihydroxides in soil, and can bind Fe with a very high affinity (Ma et al., 2011; Glick, 2012). Symbiotic nitrogen fixing bacteria like *Rhizobium*, *Bradyrhizobium* or *Mesorhizobium*, etc., improve the growth of legumes by fixing atmospheric nitrogen, a critical element for plant growth (Wani et al., 2008; Ahemad and Kibret, 2014). Phosphorus represents another nutrient vital for plant development and many microorganisms have the capacity to convert insoluble phosphates into available forms through different mechanisms like release of organic acids, chelation or mineralization of organic phosphates by secreting extracellular phosphatases (Ma et al., 2011; Ullah et al., 2015; Goswami et al., 2016). Some plant growth promoting bacteria protect the plants from fungal, bacterial, or viral diseases through mechanisms such as production of antibiotic, siderophores or lytic enzymes, or by induction of systemic resistance in plants (Ma et al., 2016; Goswami et al., 2016).

In addition, rhizospheric microorganisms have the capacity to accelerate mobilization or immobilization of metals and metalloids in soil, playing an essential role in plant metal uptake (Rajkumar et al., 2012). Indeed, metals and metalloids are generally bound to various organic and inorganic constituents in polluted soils and their phytoavailability and toxicity is closely related to their chemical speciation (Gadd, 2010). Microorganisms release inorganic and organic compounds, possessing acidifying, chelating or reductive power and are also able to metabolize metals and metalloids (e.g., based on methylation, demethylation, or redox reactions) modifying their speciation (Gadd, 2010). Thus some PGPB can improve phytoextraction or phytostabilization by altering solubility, availability, and transport of metals and metalloids (Rajkumar et al., 2009; Ma et al., 2011; Rajkumar et al., 2012; Ma et al., 2016). Soil microbial communities are also implicated directly in soil formation and soil structuration (Thavamani et al., 2017).

Mycorrhizal fungi also play an important role in mining sites (for a review, see Deng and Cao, 2017 or Wang, 2017). Around 80% of plants growing on mining sites could be colonized by arbuscular mycorrhizal fungi (Wang, 2017). Many mycorrhizal fungi are metal tolerant, and have the potential to influence metals and metalloids availability and uptake by plants by producing siderophores, organic acids, polymeric substances like glycoprotein or glomaline enable to complex metals and metalloids. They are also able to improve nutritional state of plants by acquiring phosphate, micronutrients and water through the large surface area of their hyphae (Rajkumar et al., 2012).

3.2 Remediation of mine tailings using phytostabilization

The ultimate objective of a successful phytostabilization of mine tailings is to obtain a succession of a diverse plant community on the long term, promoting soil development and restoration of the soil ecosystem functions, in a self sustainability manner (Mendez and Maier, 2008). The most critical step in this is the development of the first vegetative cover (Karaca et al., 2018). The selection of plant species is a crucial step. Ideally, plant candidates for phytostabilization must be hardly native of the metalliferous sites, thus adapted to the area and the climatic conditions. They must be high metal tolerant (metallotolerant or metallophyte) and able to thrive under environmentally stressful conditions of the mining site (Boisson et al., 2015; Festin et al., 2018). They should be fast growing, have dense root and shoot systems to limit metals and metalloids dispersion, with low requirement in term of nutrients, organic matter or water, and should be ideally, able to self propagate and disperse by seeds to allow their implementation on a large scale. They should also limit the transport of metals and metalloids to their aboveground parts or should be unattractive to animals, to limit the dissemination of the pollution (Boisson et al., 2015). The use of native plants avoids the risk of introducing invasive species that could ultimately decrease regional plant diversity (Mendez and Maier, 2008). The initial plant cover will help the development of other species that ultimately will lead to the establishment of a diverse, self sustaining, vegetative cap able to limit wind dispersion, water erosion and leaching processes (Mendez and Maier, 2008). Grasses and legumes are the most commonly plants used for phytostabilization of mine tailings representing often the initial plant cover before the development of other more exigent plants like shrubs and trees (Singh et al., 2002; Mendez and Maier, 2008). They are well adapted to very harsh edaphic conditions encountered in tailings and they grow fast, generating a quick ground cover. Grasses form a dense root mat able to trap soil particles, to retain soil moisture (Singh et al. 2002; Karaca et al., 2018). Shrubs and trees are important to form a dense canopy cover with a deeper root network able to limit erosion over long term. They also favor the development of a new topsoil layer enriched into organic matter and nutrients (Mendez and Maier, 2008).

Addition of organic amendment is generally necessary for the development of the first vegetative cover on mine tailings (Mendez and Maier, 2008). They include woodchips, manure, green waste composted, biochar, etc). Inorganic fertilizers or industrial products such as zeolites or insoluble polymers can also be added to chelate toxic metals and metalloids (de Varennes et al., 2010; Rakotonimaro et al., 2018). Alkaline materials (e.g. lime or carbonaceous wastes) can be added to raise the pH and decrease metal and metalloids solubility (Piha et al. 1995; Ye et al., 1999; Mendez and Maier, 2008). These amendments could also improve the soil structure, the water holding capacity, the organic matter and nutrient contents and could decrease the metal availability, limiting the initial toxicity of the tailings (Piha et al., 1995; Tordoff et al., 2000; Ernst 2005). However, the beneficial effects of such amendments are generally transient and even if these amendments are generally not very expensive, the large area of mine tailings makes it represent one of the major costs associated with phytoremediation. Furthermore, the addition of different chelating agents may increase metals and metalloids bioavailability and their potential uptake by the plants, not suitable for phytostabilization purpose (Karaca et al., 2018). The topsoil from mine sites represents a strategic restoration resource that should be conserved, when is possible, permitting a faster restoration of original ecosystems (Cook and Johnson, 2002; Festin et al., 2018). Thus, many researches aim at reducing the amount of these amendments (Grandlic et al., 2008).

Numerous studies have been made on the use of plant growth promoting microorganisms during the last three decades (Ma et al., 2011; Rajkumar et al., 2012 ; Ma et al., 2016; Gerhardt et al., 2017; Novo et al., 2018). These studies generally focus on the utilization of different strains of microorganisms, mainly isolated from mine tailings and presenting interesting traits like resistance to metal toxicity, production of phytohormones, or capacity to produce

siderophores or solubilize phosphate, etc. The effect of these PGP on plant growth and metal uptake is then correlated with plant parameters such as dry biomass, root and shoot elongation, or metal and metalloids concentration in shoot and root, in pot experiments or in the field (Glick, 2010; Ma et al., 2011, 2016). Studies conducted on mine tailings concluded generally that microorganisms significantly increase metal tolerance and plant growth (Novo et al., 2018). However many trials in mining sites are done on phytoextraction and on plants able to extract metals and metalloids from soil and very few concern plants which do not accumulate metals and metalloids and on microorganisms able to reduce metals and metalloids uptake and their translocation to aerial parts (Ma et al., 2011; Rajkumar et al., 2012; Ma et al., 2016; Novo et al., 2018). More researches are needed on the study of plants and PGP microorganisms implicated in phytostabilization and revegetation of mining sites, more suited for remediation of mine tailings than phytoextraction.

Petrisor et al. (2004) investigated the artificial inoculation in greenhouse condition of mine tailings (phosphogypsum and sulphidic tailings) with two bacterial strains (*Azotobacter chroococcum*, N fixing, and *Bacillus megaterium*, P solubilizing bacteria) on three indigenous herbaceous species (*Elymus repens*, *Bromus tectorum*, and *Cardaria draba*). These experiments of bacterial inoculation were compared with the addition of chemical fertilizers. After six months, high plant growth rates were observed in all inoculated treatments (associated with a significant increase of enzymatic activity), in contrast with poor plant growth and microbial communities from the chemical fertilization treatments alone. Bacterial inoculation improved also plant resistance to heavy metals by reducing the uptake of toxic metals. Grandlic et al. (2008) studied the effect of 20 isolated strains with specific PGP capability (including production of siderophores, phytohormones or solubilization of phosphate) on the development of *Atriplex lentiformis* and buffalo grass in greenhouse studies conducting with bulk tailings samples. This study showed that inoculation of strains improved plant establishment and reduced the need for compost amendment. Rajkumar et al. (2006) showed that the inoculation of strains of *Pseudomonas* sp. and *Bacillus* sp. facilitated phosphate solubilization and production of siderophores and IAA in a pot experiment conducted with soil contaminated with Cr in India. Inoculation of these strains promotes the growth of *Brassica juncea*. Trivedi et al. (2007) evaluated the effect of a strain of *Rhodococcus erythropolis* on the development of pea also in a soil contaminated with Cr in India. This strain was able to reduce Cr(VI) in non toxic Cr(III) and possessed PGP traits (ACC deaminase, indole and siderophore production). Inoculation in pot experiment conducted at 10°C in the presence of Cr(VI), resulted in the increase of plant growth associated with a substantial reduction of Cr(VI) in Cr(III). Vivas et al. (2006) tested the influence of bacterial strains of *Brevibacillus* (isolated from a Zn polluted soil) on the development of *Trifolium repens* and on the symbiotic efficiency of native arbuscular mycorrhizal fungi under Zn toxicity. *Brevibacillus* strain (tolerant to Zn and producing IAA) enhanced plant growth, N and P accumulation, as well as nodule number and mycorrhizal colonization. The amount of Zn in plant was particularly reduced by a mixed bacterium and fungi inocula. Wani et al. (2008) isolated a strain of *Rhizobium* sp. from nodules of pea (*Pisum sativum*) grown in metal contaminated Indian soils. This strain displayed a high tolerance to Ni and Zn and showed PGP activity (nitrogen fixation and production of IAA and siderophores). The inoculated strain provided protection to the plants against the toxic effects of nickel and zinc and reduced their uptake by plant organs. Similar study conducted with *Bradyrhizobium* strain isolated from greengram nodules (*Vigna radiata*), showed similar results with a decrease of the concentration of nickel and zinc in plant organs and plant growth promoting activities. Sbabou et al., 2016 isolated two bacteria from the nodules of the legume *Hedysarum spinosissimum* growing in metal contaminated soils in Morocco and showed that their inoculation promoted host plant growth and suppressed Zn toxicity symptoms in plants, enhancing their phytostabilization potential.

Although interesting results have been reported under laboratory conditions, showing that inoculation of PGP microorganism (bacteria and/or mycorrhizae) may promote metals and metalloids phytostabilization, only few studies have been done on field conditions (Novo et al., 2018). Field experiment was conducted by Dary et al. (2010) to evaluate the metal phytostabilisation potential of *Lupinus luteus* inoculated with *Bradyrhizobium*, *Pseudomonas* sp. and *Ochrobactrum cytisi*. Inoculation with *Bradyrhizobium* alone increased both biomass and nitrogen content, but coinoculation with the three PGPR strains improved plant biomass and limit metal accumulation both in shoots and roots. Burges et al (2016) conducted a field study on an abandoned Pb/Zn mine in Spain characterized by a temperate Atlantic climate. An endophyte consortium of bacteria was set up with a strain of *Microbacterium* sp. and four strains of *Pseudomonas* isolated from five metal tolerant native plant species (one excluder: *Festuca rubra*, and four accumulators: *Noccaea caerulea*, *Rumex acetosa*, *Betula alba* and *Salix atrocinerea*). This consortium was sprayed on vegetation and soil at the Pb/Zn mine. This study showed a lower level of Pb and Zn in native excluder plant as a result of inoculation, but no effect on biomass.

There is no doubt that phytoremediation on mining sites is very challenging. Indeed, major changes are necessary, i.e. physical, chemical and microbiological properties of soils, as well as evolution in drainage and in hydrological properties of these soils. However, as seen in many reviews, there is accumulating evidence that promising outcomes are possible (Mendez and Maier, 2008; Lamb et al., 2015; Nirola et al., 2016; Thavamani et al., 2017; Karaca et al., 2018; Novo et al., 2018). However, more researches are needed on the effect of PGP microorganisms in mine soils and mine tailings in long term in field experiments. Particularly a better knowledge about plant microbiome interactions is required to optimize the rate of phytostabilisation and provide a sustainable plant establishment (Garris et al., 2016; Ma et al., 2016; Thavamani et al., 2017; Valentin-Vargas et al., 2018). The long term monitoring of mining sites (more than 2 years) treated with phytoremediation is also scarce, but this is a necessity to test the effectiveness and robustness of this technique, study the decomposition of amendments and evaluate the evolution of metallic contaminants in terms of mobility and speciation, (Tordoff et al., 2000; Pond et al. 2005; Mains et al. 2006; Nirola et al., 2016; Venkateswarlu et al., 2016). Different parameters must be also monitored on the long term as the capacity of plants to self propagate successfully with no additional inputs or the non toxicity of plants for domestic and wild animals required to prevent further contamination of the ecosystem.

3.3 Phytostabilization of mine tailings in Sub Saharan African countries

The climate is an important factor for phytoremediation. Indeed, according to climate type (equatorial, tropical, versus semi arid, or arid), implementation of phytoremediation will differ with respect to plant types. Different works have been done in arid and semi arid environments (Mendez and Maier, 2008; Padmavathiamma et al., 2014; Doumas et al., 2016; Nirola et al., 2016; Valentin-Vargas et al., 2018). Indeed, arid and semiarid soils often present a higher salinity, due to the evaporation rates which exceed rainfall, besides natural salts originating from saline rainfall (Mendez and Maier, 2008). Because water is rare in arid and semi arid environments, mining operations often use saline groundwater which ultimately becomes hypersaline as it is recycled throughout ore processing. A salt crust may form on the surface of the tailings (Newson and Fahey, 2003). Thus in these environments plants should be ideally also drought and salt tolerant (Tordoff et al., 2000; Mendez and Maier, 2008). In contrast, in temperate or tropical and equatorial regions, tailings can be saturated or waterlogged, so plants adapted to slightly anaerobic conditions can be required (Cardoso et al., 2013). Furthermore, in arid environments, irrigation is often a necessity to increase the chance of success of revegetation; however, if irrigation is not possible, plantation needs to take advantage of the normal rainfall season (Tordoff et al., 2000). The use of native plants adapted to arid conditions contributes to limit the cost and the risk of exhaustion of water reserves in arid areas. Currently, few studies have been published on phytoremediation in Sub Saharan African countries, and again, most of them are dedicated to studies on phytoextraction (Festin et al., 2018, Figure 1, Table 3).

Table 3. Phytostabilization of mine tailings in Sub Saharan African countries.

Map reference	Type of mine	Plant used	Microorganisms	Country	Reference
7	Metal rich bare soils in mining area	Identification of the plant functional traits for phytostabilisation purpose under tropical climate		Democratic Republic of Congo	wa Ilunga et al., 2015
8	Soils artificially contaminated by Cu	Evaluation of 3 grass species (<i>Cynodon dactylon</i> , <i>Monocymbium cerasiiforme</i> , <i>Rendlia altera</i>) and 2 soil amendments		Democratic Republic of Congo	Shutchu et al., 2010a
9	Mining region with copper hills	Evaluation of 7 perennial <i>Poaceae</i> including 3 grasses species (<i>Andropogon schirensis</i> ; <i>Eragrostis racemose</i> ; <i>Loudetia simplex</i>)		Democratic Republic of Congo	Boisson et al., 2015
10	Metalliferous slag heaps (Pb/Zn)	Identification of 5 plant species (<i>Cynodon dactylon</i> ; <i>Indigofera spicata</i> ; <i>Melinis repens</i> ; <i>Aristida adscencionis</i> ; <i>Pennisetum setaceum</i>) for the phytostabilization of slag heaps		Zambia	Leteinturier et al., 2001
11	Tailings dam soils	Evaluation of 5 tree species (<i>Acacia polyacantha</i> , <i>Toona ciliate</i> , <i>Acacia sieberana</i> , <i>Bauhinia thonningii</i> , <i>Peltophorum africanum</i>)		Zambia	Kamingaa and Syampungani, 2012
12	Gold and uranium tailings dam	Identification of 376 plant species of natural colonizers of the families of the <i>Poaceae</i> , <i>Asteraceae</i> , <i>Fabaceae</i> , and <i>Anacardiaceae</i> potentially suitable for phytostabilisation		South Africa	Weiersbye et al., 2006
13	Lead and zinc mine alkaline tailing	Evaluation of 5 grass species (<i>Digitaria eriantha</i> ; <i>Cymbopogon plurinodis</i> ; <i>Cenchrus ciliaris</i> ; <i>Eragrostis superba</i> and <i>Fingeruthia africana</i>) with different inorganic fertilizer		South Africa	Titshall et al., 2013
14	Nickel mine tailings	Evaluation of 3 <i>Acacia</i> species (<i>Acacia gerrardii</i> ; <i>Acacia karroo</i> <i>A. polyacantha</i>) with different amendments		Zimbabwe	Nyakudya et al., 2011
8	Bare soil contaminated by copper smelting	Evaluation of spontaneous and assisted phytostabilisation using limestone and compost as soil amendment. The grass (<i>Microchloa altera</i>) were the only perennial to survive		Democratic Republic of Congo	Shutchu et al., 2015
15	Gold mine tailings dump	Evaluation of growth performance of exotic versus indigenous tree species in the establishment of a cover cap		Zimbabwe	Mulizane et al. 2005

16	Platinum tailings	Evaluation of uncomposted woodchips wastes on grass growth during revegetation trials		South Africa	Van Rensburg and Morgenthal, 2004
17	Ore bearing dune	Evaluation of topsoil using for natural succession and a mixture of fast germinating species (<i>Sorghum</i> spp., <i>Helianthus annuus</i> , <i>Crotalaria juncea</i> or <i>Pennisetum americanum</i>) as a nurse crop followed by the vegetation of <i>Acacia karoo</i>	Use of natural population present in topsoil	South Africa	Cooke and Johnson, 2002
18	Gold mine areas	Evaluation of a combination of physical, chemical and biological methods including the use of plant (<i>Pueraria phaseoloides</i> , <i>Centrosema pubescens</i>) to control erosion and different tree species (<i>Acacia mangium</i> , <i>Senna siamea</i> , <i>Gliricidia sepium</i> and <i>Leucaena leucocephala</i>) for the establishment of a cover cap		Ghana	Tetteh et al., 2015a, b
19	Tin mine tailings	Evaluation of trees, grasses and legumes in presence of different rate of compound fertilizer and addition of <i>Rhizobium</i> .	<i>Rhizobium</i>	Zimbabwe	Piha et al 1995

In Democratic Republic of Congo, wa Ilunga et al. (2015) investigated the role of plant functional traits in the ecological restoration of degraded mine sites in Katanga province, under tropical climate. They highlighted the main plant traits for phytostabilisation purpose of metal rich bare soils in this mining region: an annual life cycle, growth phenology in wet season, 0 - 10 cm depth of underground system, bud bank by seeds, propagule size and a dispersal mode by adhesion. Shutcha et al. (2010), evaluated the feasibility of utilizing three grass species, *Cynodon dactylon*, *Monocymbium cerasiiforme* and *Rendlia altera*, with two soil amendments (compost and lime) for the phytostabilisation of soils artificially contaminated by Cu over two years. *R. altera* seemed to be the best candidate due to its capacity to survive on unamended soil. Lime seems to increase the reproduction of plants and limit the accumulation of Cu in leaves compared to compost. Boisson et al. (2015) evaluated the phytostabilization potential capacity of seven perennial frequently occurring *Poaceae* present in copper hill communities in Congo. They identified three grasses species (*Andropogon schirensis*, *Eragrostis racemosa* and *Loudetia simplex*) that were promising for phytostabilisation purpose since they produce a large quantity of seeds and a high percentage of cover.

In Zambia, 39 taxa were identified on lead/zinc mining-generated slag heaps, of which five were recommended for phytostabilization including *Cynodon dactylon*, *Indigofera spicata*, *Melinis repens*, *Aristida adscencionis* and *Pennisetum setaceum* (Leteinturier et al., 2001). Another study was conducted on a tree survey on tailings dams (Kambing'a and Syampungani, 2012). They recorded 21 species of trees, of which five dominant species could be used for phytostabilisation purpose: *Acacia polyacantha*, *Toona ciliate*, *Acacia sieberana*, *Bauhinia thonningii* and *Peltophorum africanum*.

In South Africa, Weiersbye et al. (2006) identified 376 species, able to naturally colonize gold and uranium tailings dams and polluted soils, which could be potentially suitable for phytoremediation purpose. Titshall et al. (2013) investigated the revegetation potential of a lead/zinc mine tailing under glasshouse conditions using five perennial grass species (*Digitaria eriantha*, *Cymbopogon plurinodis*, *Cenchrus ciliaris*, *Eragrostis superba* and *Fingeruthia africana*) with three rates of inorganic fertilizer (full application, half the full rate, and unfertilized pots). The results showed an increase in the yield of all grass species with increasing fertilizer application rate but also, quite high concentrations of Zn in the foliage of almost all grass species, which is not suited to phytostabilisation purpose.

Field experiments were also conducted in Sub Saharan African countries:

In Zimbabwe, early growth performance was evaluated for three indigenous *Acacia* species (*Acacia gerrardii*, *Acacia karroo* and *A. polyacantha*) on a nickel mine tailings amended with topsoil, sewage sludge, and compound fertilizer and an untreated control (Nyakudya et al., 2011). The field experiment (over six months) revealed no significant advantage of amendments compared to the untreated control in terms of survival for all species.

A. gerrardii had even lower survival in the sewage sludge treated experiment. However, the relative growth rate was improved with fertilizer application compared to other soil amendments; especially the sewage sludge and all three species seemed to have good potential in revegetation of nickel mine tailing dumps. Mulizane et al. (2005) conducted a study on the growth performance of exotic versus indigenous tree species in the establishment of a cover cap in an active gold mine tailings dump. This study showed no significant difference between exotic and indigene plant growth after one year. Experiments on long term should be useful to confirm the stability of such vegetation.

In Democratic Republic of Congo, Shutcha et al. (2015) compared the effect of two amendments (compost and lime) on the natural colonization of bare soil contaminated by copper smelting and the growth of a small grass (*Microchloa altera*) during three years. Results showed that only the perennial *M. altera* was able to survive during the six months dry season. Furthermore, the combination of soil amendments (lime and compost) was the most effective to improve plant fecundity of *M. altera*, and to reduce metal uptake by leaves.

In South Africa, Van Rensburg and Morgenthal (2004) have tested the effect of uncomposted woodchip waste on vegetation establishment during a platinum tailings rehabilitation with seeds originating from a mixture of grasses. The addition of woodchips enhanced vegetation establishment, during early periods but no significant differences in biomass between treated and untreated plots were evident after 3 months.

Two large scale post-mining restoration practices were also conducted on mining sites in South Africa and Ghana (Festin et al., 2018).

In South Africa, dredge mining in coastal ore bearing dune forest has taken place in Zululand since 1977 for the extraction of heavy minerals such as rutile, ilmenite, and zircon (Cooke and Johnson, 2002). Topsoil was stored in an artificial pond. The physical restoration has started by the spreading of around 10 cm of topsoil to initiate natural succession and the establishment of indigenous dune forest. Artificial windbreaks were installed to help the natural colonization process, and a mixture of fast germinating species was seeded (*Sorghum* spp., *Helianthus annuus*, *Crotalaria juncea* or *Pennisetum americanum*). The succession has progressed until the establishment of a vegetation dominated by the major pioneer species, *Acacia karoo*. By this method, around 400 ha have been reclaimed since 1978.

In Ghana, land reclamation, after gold mine activity, was conducted by AngloGold Ashanti Company, using a combination of physical, chemical and biological methods at the Iduapriem mine (Tetteh et al. 2015a, b). Restoration had been done on 4 old sites that were 2, 5, 9 and 11 years old. The restoration involved different steps: (1) a physical restoration with earthwork/slope-battering to create a more visually pleasing landscape; (2) the addition of oxide material, added to bind the soil together and enhance soil stability; (3) the amendment of topsoil, using poultry droppings, cow manure and fertilizers, that were spread over the oxide material; (4) the creation of crest drains to limit the runoff of water and to control erosion; (5) the establishment of a cover cap with *Puereria phaseoloides* and *Centrosema pubescens* to control erosion; (6) the planting of seedlings of *Acacia mangium*, *Senna siamea*, *Gliricidia sepium* and *Leucaena leucocephala* followed by weeding, pruning, and fertilizer application. While the first three species are nitrogen fixers, *S. siamea* forms association with vesicular arbuscular mycorrhiza. Mensah (2015) showed that very long time was necessary to restore the fertility to a level similar to the original ones (Festin et al., 2018).

Few studies were conducted on the benefice of using plant growth promoting microorganisms in phytostabilization of mine soils and tailings in Sub Saharan African countries.

In Zimbabwe, Piha et al. (1995) performed a study on revegetation trials of a tin mine tailings using indigenous and adapted trees, grasses and legumes. Despite hot and dry conditions, around 40% of plant cover was achieved at the end of the first season, with addition

of a moderate rate of compound fertilizer. *Rhizobium* bacteria, selected under high glasshouse temperatures experiment, enabled development of nodules on site. This study showed that nitrogen fixing legume tree species were well adapted to the harsh conditions of the site and that the low input approach (only one initial addition of fertilizer) appeared to have been reasonably successful even after two years.

However, more researches are needed on the use of plant growth promoting bacteria to speed up the processes of phytostabilization in Sub Saharan African countries.

4. Conclusion

Intensive mining and processing activities resulted in the generation of huge amounts of liquid and solid wastes worldwide, and mining pollution represents a growing social concern. In Sub Saharan African countries, mining activities have generated large areas of unrestored mined lands, particularly in South Africa, Zimbabwe, Botswana, or Ghana (Humphries et al. 2017; Festin et al., 2018). Two major problems facing mining industries are the formation of acid mine drainage and the accumulations of millions of tons of tailings and mine wastes devoid of vegetation, thus subjected to eolian and water erosion. The need to protect the environment and to find sustainable and low cost remediation strategies led to an increase of worldwide researches in this field (Aznar Sanchez et al., 2018).

Precipitation of metals and metalloids in solution relying on iron oxidizing microorganisms or sulfate reducing bacteria, implemented in the form of wetlands or bioreactors, appeared to be a technology of choice widely used for AMD remediation (Johnson and Hallberg, 2005; Kiran et al., 2017). The application of such biological treatments in Sub Saharan African countries is still limited and has been mainly performed on a laboratory scale. More works are needed to identify the operational, environmental and biological factors that control the process performances (Neculita et al., 2007; Fernandez-Rojo et al., 2017, 2018). Furthermore, the long term robustness and stability of these treatment systems have to be investigated. The contribution of “omics” studies is also essential for a thorough understanding of the metabolic interactions between microorganisms and pollutants and their fluctuations for improving the design and performance of such processes over long term (Ayangbenro et al., 2018; Rezadehbashi and Baldwin, 2018).

Phytostabilization of mine tailings and mining sites is also very promising and largely used worldwide. Although the establishment of a dense vegetative cover on mine tailings and mine wastes is very challenging, plant growth promoting microorganisms have been the subject of an increasing number of studies, owing their capacity to promote plant tolerance and plant growth under metal stress and to immobilize metals and metalloids in soil, limiting the toxicity and uptake by the plants. However, phytostabilization studies using PGP microorganisms remain limited in mine tailings compared to phytoextraction (Ma et al., 2011, 2016; Novo et al., 2018). In Sub Saharan African countries, restoration was demonstrated to be effective in South Africa or in Ghana on large scale studies (Cooke and Johnson, 2002; Tetteh et al., 2015a, b). However, the use of plant growth promoting microorganisms in field study is always very limited. Long term studies must be also conducted to evaluate the efficacy of phytostabilisation in creating a self sustaining biological cap and promote the formation of soil structure similar to healthy soil plant environment. Role of soil microbial communities need to be also better elucidated in order to speed up the process (Mendez and Maier, 2008; Novo et al., 2018).

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References

- Abiye, T., Bybee, G., Leshomo, J. 2018. Fluoride concentrations in the arid Namaqualand and the Waterberg groundwater, South Africa: Understanding the controls of mobilization through hydrogeochemical and environmental isotopic approaches. *Groundwater for Sustainable Development*, 6, 112–120. <http://dx.doi.org/10.1016/j.gsd.2017.12.004>
- Ahemad, M., Kibret, M. 2014. Mechanisms and applications of plant growth promoting rhizobacteria: current perspective. *Journal of King Saud University-Science*, 26(1), 1-20. <https://doi.org/10.1016/j.jksus.2013.05.001>
- Ahoranta, S.H., Kokko, M.E., Papirio, S., Özkaya, B., Puhakka, J.A. 2016. Arsenic removal from acidic solutions with biogenic ferric precipitates. *Journal of hazardous materials*, 306, 124-132. <https://doi.org/10.1016/j.jhazmat.2015.12.012>
- Akcil, A., Koldas, S. 2006. Acid Mine Drainage (AMD): causes, treatment and case studies. *Journal of Cleaner Production*, 14(12-13), 1139-1145. <https://doi.org/10.1016/j.jclepro.2004.09.006>
- Al, M., Evans, L.J., Gould, W.D., Duncan, W.F.A., Glasauer, S. 2011. The long term operation of a biologically based treatment system that removes As, S and Zn from industrial (smelter operation) landfill seepage. *Applied geochemistry*, 26(11), 1886-1896. <https://doi.org/10.1016/j.apgeochem.2011.06.012>
- Ali, H., Khan, E., Sajad, M.A. 2013. Phytoremediation of heavy metals - Concepts and applications. *Chemosphere*, 91(7), 869–881. <http://dx.doi.org/10.1016/j.chemosphere.2013.01.075>
- Ali, S.H., Giurco, D., Arndt, N., Nickless, E., Brown, G., Demetriades, A., Durrheim, R., Enriquez, M.A., Kinnaird, J., Littleboy, A., Meinert, L.D. 2017. Mineral supply for sustainable development requires resource governance. *Nature*, 543(7645), 367. <https://doi.org/10.1038/nature21359>
- Anawar, H.M. 2013. Impact of climate change on acid mine drainage generation and contaminant transport in water ecosystems of semi-arid and arid mining areas. *Physics and Chemistry of the Earth, Parts A/B/C*, 58, 13-21
- Anawar, H.M., 2015. Sustainable rehabilitation of mining waste and acid mine drainage using geochemistry, mine type, mineralogy, texture, ore extraction and climate knowledge. *Journal of Environmental Management*, 158, 111–121. <http://dx.doi.org/10.1016/j.jenvman.2015.04.045>
- Asta, M.P., Ayora, C., Román-Ross, G., Cama, J., Acero, P., Gault, A.G., Charnock, J.M., Bardelli, F. 2010. Natural attenuation of arsenic in the Tinto Santa Rosa acid stream (Iberian Pyritic Belt, SW Spain): the role of iron precipitates. *Chemical Geology*, 271(1-2), 1-12. <https://doi.org/10.1016/j.chemgeo.2009.12.005>
- Audet, P., Charest, C. 2007. Heavy metal phytoremediation from a meta-analytical perspective. *Environmental Pollution*, 147(1), 231-237. <https://doi.org/10.1016/j.envpol.2006.08.011>
- Ayangbenro, A.S., Olanrewaju, O.S., Babalola, O.O. 2018. Sulfate-reducing bacteria as an effective tool for sustainable acid mine bioremediation. *Frontiers in Microbiology*, 9, 1986. <https://doi.org/10.3389/fmicb.2018.01986>
- Aznar-Sánchez, J., García-Gómez, J., Velasco-Muñoz, J., Carretero-Gómez, A. 2018. Mining waste and its sustainable management: advances in worldwide research. *Minerals*, 8(7), 284. <https://doi.org/10.3390/min8070284>

- Baker, B.J., Banfield, J.F. 2003. Microbial communities in acid mine drainage. *FEMS microbiology ecology*, 44(2), 139-152. [https://doi.org/10.1016/s0168-6496\(03\)00028-x](https://doi.org/10.1016/s0168-6496(03)00028-x)
- Baldwin, S.A., Khoshnoodi, M., Rezadehbashi, M., Taupp, M., Hallam, S., Mattes, A., Sanei, H. 2015. The microbial community of a passive biochemical reactor treating arsenic, zinc and sulfate-rich seepage. *Frontiers in Bioengineering and Biotechnology*, 3:27. <https://doi.org/10.3389/fbioe.2015.00027>
- Battaglia-Brunet, F., Crouzet, C., Burnol, A., Coulon, S., Morin, D., Joulain, C. 2012. Precipitation of arsenic sulphide from acidic water in a fixed-film bioreactor. *Water Research*, 46(12), 3923–3933. <http://dx.doi.org/10.1016/j.watres.2012.04.035>
- Batty, L.C., Younger, P.L. 2002. Critical role of macrophytes in achieving low iron concentrations in mine water treatment wetlands. *Environmental science and technology*, 36(18), 3997-4002. <https://doi.org/10.1021/es020033>
- Blanco, I., Sapsford, D.J., Trumm, D., Pope, J., Kruse, N., Cheong, Y.W., McLauchlan, H., Sinclair, E., Weber, P., Olds, W. 2018. International trials of vertical flow reactors for coal mine water treatment. *Mine Water and the Environment*, 37(1), 4-17. <https://doi.org/10.1007/s10230-017-0491-z>
- Boisson, S., Stradic, S.L., Collignon, J., Séleck, M., Malaisse, F., Shutcha, M.N., Faucon, M.P., Mahy, G. 2015. Potential of copper-tolerant grasses to implement phytostabilisation strategies on polluted soils in South D. R. Congo. *Environmental Science and Pollution Research*, 23(14), 13693–13705. <http://dx.doi.org/10.1007/s11356-015-5442-2>
- Bolan, N.S., Park, J.H., Robinson, B., Naidu, R., Huh, K.Y. 2011. Phytostabilization. *Advances in Agronomy*, 145–204. <http://dx.doi.org/10.1016/b978-0-12-385538-1.00004-4>
- Bolan, N., Kunhikrishnan, A., Thangarajan, R., Kumpiene, J., Park, J., Makino, T., Kirkham, M.B., Scheckel, K. 2014. Remediation of heavy metal (loid) s contaminated soils–to mobilize or to immobilize?. *Journal of Hazardous Materials*, 266, 141-166. <https://doi.org/10.1016/j.jhazmat.2013.12.018>
- Bruneel, O., Mghazli, N., Hakkou, R., Dahmani, I., Maltouf, A.F., Sbabou, L. 2017. In-depth characterization of bacterial and archaeal communities present in the abandoned Kettara pyrrhotite mine tailings (Morocco). *Extremophiles*, 21(4), 671-685. <https://doi.org/10.1007/s00792-017-0933-3>
- Buddhawong, S., Kusch, P., Mattusch, J., Wiessner, A., Stottmeister, U. 2005. Removal of arsenic and zinc using different laboratory model wetland systems. *Engineering in Life Sciences*, 5(3), 247-252. <https://doi.org/10.1002/elsc.200520076>
- Burges, A., Epelde, L., Blanco, F., Becerril, J. M., Garbisu, C. 2017. Ecosystem services and plant physiological status during endophyte-assisted phytoremediation of metal contaminated soil. *Science of the Total Environment*, 584, 329-338. <https://doi.org/10.1016/j.scitotenv.2016.12.146>
- Cardoso, J.A., Rincon, J., Jiménez, J.D.L.C., Noguera, D., and Rao, I.M. 2013. Morpho-anatomical adaptations to waterlogging by germplasm accessions in a tropical forage grass. *AoB Plants*, 5. <https://doi:10.1093/aobpla/plt047>
- Carlier, J.D., Alexandre, L.M., Luís, A.T., and Costa, M.C. 2019. Potential of industrial by-products and wastes from the Iberian Peninsula as carbon sources for sulphate-reducing bacteria. *International Journal of Environmental Science and Technology*, 1-20. <https://doi.org/10.1007/s13762-018-02197-z>
- Casiot, C., Morin C., Juillot F., Bruneel O., Personné J.C., Leblanc M., Duquesne K., Bonnefoy V., Elbaz-Poulichet, F. 2003. Bacterial immobilization and oxidation of arsenic in acid mine drainage (Carnoulès creek, France). *Water Research*, 37(12), 2929-2936. [https://doi.org/10.1016/S0043-1354\(03\)00080-0](https://doi.org/10.1016/S0043-1354(03)00080-0)

- Chen, C.J., Jiang, W.T. 2012. Influence of waterfall aeration and seasonal temperature variation on the iron and arsenic attenuation rates in an acid mine drainage system. *Applied geochemistry*, 27(10), 1966-1978. <https://doi.org/10.1016/j.apgeochem.2012.06.003>
- Clyde, E.J., Champagne, P., Jamieson, H.E., Gorman, C., Sourial, J. 2016. The use of a passive treatment system for the mitigation of acid mine drainage at the Williams Brothers Mine (California): pilot-scale study. *Journal of Cleaner Production*, 130, 116-125. <https://doi.org/10.1016/j.jclepro.2016.03.145>
- Cohen, R.R. 2006. Use of microbes for cost reduction of metal removal from metals and mining industry waste streams. *Journal of Cleaner Production*, 14(12-13), 1146-1157. <https://doi.org/10.1016/j.jclepro.2004.10.009>
- Cooke, J.A., Johnson, M.S. 2002. Ecological restoration of land with particular reference to the mining of metals and industrial minerals: a review of theory and practice. *Environmental Reviews*, 10(1), 41-71. <http://dx.doi.org/10.1139/a01-014>
- Coram, N.J., Rawlings, D.E. 2002. Molecular relationship between two groups of the genus *Leptospirillum* and the finding that *Leptospirillum ferriphilum* sp. nov. dominates South African commercial biooxidation tanks that operate at 40 C. *Applied and environmental microbiology*, 68(2), 838-845. <https://doi.org/10.1128/aem.68.2.838-845.2002>
- Costa, M.C., Santos, E.S., Barros, R.J., Pires, C., Martins, M. 2009. Wine wastes as carbon source for biological treatment of acid mine drainage. *Chemosphere*, 75, 831-836. <http://dx.doi.org/10.1016/j.chemosphere.2008.12.062>
- Costa, J.P., Girao, A.V., Lourenço, J.P., Monteiro, O.C., Trindade, T., Costa, M.C. 2013. "Green synthesis of covellite nanocrystals using biologically generated sulfide: Potential for bioremediation systems". *Journal of Environmental Management*, 128; 226-232. <https://doi.org/10.1016/j.jenvman.2013.05.034>
- Cunningham, S.D., Berti, W.R., Huang, J.W. 1995. Phytoremediation of contaminated soils. *Trends in biotechnology*, 13(9), 393-397. [https://doi.org/10.1016/s0167-7799\(00\)88987-8](https://doi.org/10.1016/s0167-7799(00)88987-8)
- Dary, M., Chamber-Perez, M.A., Palomares, A.J., Pajuelo, E. 2010. "In situ" phytostabilisation of heavy metal polluted soils using *Lupinus luteus* inoculated with metal resistant plant-growth promoting rhizobacteria. *Journal of Hazardous Materials*, 177(1-3), 323-330. <https://doi.org/10.1016/j.jhazmat.2009.12.035>
- De Varennes, A., Cunha-Queda, C., Qu, G. 2010. Amendment of an acid mine soil with compost and polyacrylate polymers enhances enzymatic activities but may change the distribution of plant species. *Water, Air, and Soil Pollution*, 208(1-4), 91-100. <http://dx.doi.org/10.1007/s11270-009-0151-4>
- Dean, A.P., Lynch, S., Rowland, P., Toft, B.D., Pittman, J.K., and White, K.N. 2013. Natural wetlands are efficient at providing long-term metal remediation of freshwater systems polluted by acid mine drainage. *Environmental science and technology*, 47(21), 12029-12036. <http://dx.doi.org/10.1021/es4025904>
- Deng, Z., Cao, L. 2017. Fungal endophytes and their interactions with plants in phytoremediation: a review. *Chemosphere*, 168, 1100-1106. <http://dx.doi.org/10.1016/j.chemosphere.2016.10.097>
- Deshpande, A.S., Kumari, R., Prem Rajan, A. 2018. A delve into the exploration of potential bacterial extremophiles used for metal recovery. *Global Journal of Environmental Science and Management*, 4(3), 373-386. <http://dx.doi.org/10.22034/gjesm.2018.03.010>
- Dino, G.A., Mehta, N., Rossetti, P., Ajmone-Marsan, F., and De Luca, D.A. 2018. Sustainable approach towards extractive waste management: two case studies from Italy. *Resources Policy*, 59, 33-43. <https://doi.org/10.1016/j.resourpol.2018.07.009>
- Dixit, R., Malaviya, D., Pandiyan, K., Singh, U.B., Sahu, A., Shukla, R., Singh, B.P., Rai, J.P., Sharma, P.K., Lade, H., Paul, D. 2015. Bioremediation of heavy metals from soil and aquatic

- environment: an overview of principles and criteria of fundamental processes. *Sustainability*, 7(2), 2189-2212. <https://doi.org/10.3390/su7022189>
- Dold, B., Fontboté, L. 2001. Element cycling and secondary mineralogy in porphyry copper tailings as a function of climate, primary mineralogy, and mineral processing. *Journal of Geochemical Exploration*, 74(1-3), 3–55. [http://dx.doi.org/10.1016/s0375-6742\(01\)00174-1](http://dx.doi.org/10.1016/s0375-6742(01)00174-1)
- Doumas, P., Munoz, M., Banni, M., Becerra, S., Bruneel, O., Casiot, C., Cleyet-Marel, J.C., Gardon, J., Noack, Y., Sappin-Didier, V. 2016. Polymetallic pollution from abandoned mines in Mediterranean regions: a multidisciplinary approach to environmental risks. *Regional Environmental Change*, 18(3), 677-692. <https://doi.org/10.1007/s10113-016-0939-x>
- Edwards, K.J., Gihring, T.M., Banfield, J.F. 1999. Seasonal variations in microbial populations and environmental conditions in an extreme acid mine drainage environment. *Appl. Environ. Microbiol.*, 65(8), 3627-3632.
- Elbaz-Poulichet, F., Bruneel, O. Casiot, C. 2006. The Carnoulès mine. Generation of As-rich acid mine drainage, natural attenuation processes and solutions for passive in-situ remediation. In *Difpolmine (Diffuse Pollution From Mining Activities)*. Dec 2006, Montpellier, France. <https://hal.archives-ouvertes.fr/hal-00184269>
- Egal, M., Casiot, C., Morin, G., Elbaz-Poulichet, F., Cordier, M. A., and Bruneel, O. 2010. An updated insight into the natural attenuation of As concentrations in Reigous Creek (southern France). *Applied geochemistry*, 25(12), 1949-1957. <http://dx.doi.org/10.1016/j.apgeochem.2010.10.012>
- Ernst, W.H.O. 2005. Phytoextraction of mine wastes – options and impossibilities. *Chemie der Erde Geochemistry*, 65, 29–42. <http://dx.doi.org/10.1016/j.chemer.2005.06.001>.
- Fernandez-Rojo, L., Héry, M., Le Pape, P., Braungardt, C., Desoeuvre, A., Torres, E., Tardy, V., Resongles, E., Laroche, E., Delpoux, S., Jouliau, C., Battaglia-Brunet, F., Boisson, J., Grapin, G., Morin, G., Casiot, C. 2017. Biological attenuation of arsenic and iron in a continuous flow bioreactor treating acid mine drainage (AMD). *Water Research*, 123, 594–606. <http://dx.doi.org/10.1016/j.watres.2017.06.059>.
- Fernandez-Rojo, L., Casiot, C., Tardy, V., Laroche, E., Le Pape, P., Morin, G., Jouliau, C., Battaglia-Brunet, F., Braungardt, C., Desoeuvre, A., Delpoux, S., Boisson, J., Marina Héry, M. 2018. Hydraulic retention time affects bacterial community structure in an As-rich acid mine drainage (AMD) biotreatment process. *Applied microbiology and biotechnology*, 102(22), 9803-9813. <https://doi.org/10.1007/s00253-018-9290-0>
- Fernandez-Rojo, L., Casiot, C., Laroche, E., Tardy, V., Bruneel, O., Delpoux, S., Desoeuvre, A., Grapin, G., Saviganc, J., Boisson, J., Morin, G., Battaglia-Brunet F., Jouliau, C., Héry M. 2019. A field-pilot for passive bioremediation of As-rich acid mine drainage. *Journal of environmental management*, 232, 910-918. <https://doi.org/10.1016/j.jenvman.2018.11.116>.
- Fernando, W.A.M., Ilankoon, I.M.S.K., Syed, T.H., and Yellishetty, M. 2018. Challenges and opportunities in the removal of sulphate ions in contaminated mine water: a review. *Minerals Engineering*, 117, 74-90. <https://doi.org/10.1016/j.mineng.2017.12.004>
- Ferris, F.G., Hallberg, R.O., Lyven, B., Pedersen, K. 2000. Retention of strontium, cesium, lead and uranium by bacterial iron oxides from a subterranean environment. *Applied Geochemistry*, 15(7), 1035-1042. [https://doi.org/10.1016/s0883-2927\(99\)00093-1](https://doi.org/10.1016/s0883-2927(99)00093-1)
- Festin, E.S., Tigabu, M., Chileshe, M.N., Syampungani, S., Oden, P.C. 2018. Progresses in restoration of post-mining landscape in Africa. *Journal of Forestry Research*, 1-16. <https://doi.org/10.1007/s11676-018-0621-x>
- Fu, F., Wang, Q. 2011. Removal of heavy metal ions from wastewaters: a review. *Journal of environmental management*, 92(3), 407-418. <https://doi.org/10.1016/j.jenvman.2010.11.011>

- Fukushi, K., Sasaki, M., Sato, T., Yanase, N., Amano, H., Ikeda, H. 2003. A natural attenuation of arsenic in drainage from an abandoned arsenic mine dump. *Applied Geochemistry*, 18(8), 1267–1278. [http://dx.doi.org/10.1016/s0883-2927\(03\)00011-8](http://dx.doi.org/10.1016/s0883-2927(03)00011-8)
- Gadd, G.M. 2010. Metals, minerals and microbes: geomicrobiology and bioremediation. *Microbiology*, 156(3), 609-643. <https://doi.org/10.1099/mic.0.037143-0>
- Garbisu, C., Alkorta, I. 2001. Phytoextraction: a cost-effective plant-based technology for the removal of metals from the environment. *Bioresource technology*, 77(3), 229-236. [https://doi.org/10.1016/s0960-8524\(00\)00108-5](https://doi.org/10.1016/s0960-8524(00)00108-5)
- Garris, H.W., Baldwin, S.A., Van Hamme, J.D., Gardner, W.C., and Fraser, L.H. 2016. Genomics to assist mine reclamation: a review. *Restoration ecology*, 24(2), 165-173. <https://doi.org/10.1111/rec.12322>
- Gault, A.G., Cooke, D.R., Townsend, A.T., Charnock, J.M., and Polya, D.A. 2005. Mechanisms of arsenic attenuation in acid mine drainage from Mount Bischoff, western Tasmania. *Science of the Total Environment*, 345(1-3), 219-228. <https://doi.org/10.1016/j.scitotenv.2004.10.030>
- Gazea, B., Adam, K., Kontopoulou, A. 1996. A review of passive systems for the treatment of acid mine drainage. *Minerals engineering*, 9(1), 23-42
- Gerhardt, K.E., Gerwing, P.D., and Greenberg, B.M. 2017. Opinion: taking phytoremediation from proven technology to accepted practice. *Plant science*, 256, 170-185. <http://dx.doi.org/10.1016/j.plantsci.2016.11.016>
- Glick, B.R. 2010. Using soil bacteria to facilitate phytoremediation. *Biotechnology Advances*, 28(3), 367–374. <http://dx.doi.org/10.1016/j.biotechadv.2010.02.001>
- Glick, B.R. 2012. Plant growth-promoting bacteria: mechanisms and applications. *Scientifica*, 2012. <https://doi.org/10.6064/2012/963401>
- Gonzalez-Contreras, P., Weijma, J., Weijden, R. V. D., and Buisman, C. J. 2010. Biogenic scorodite crystallization by *Acidianus sulfidivorans* for arsenic removal. *Environmental science and technology*, 44(2), 675-680. <https://doi.org/10.1021/es902063t>
- Goswami, D., Thakker, J.N., Dhandhukia, P.C. 2016. Portraying mechanics of plant growth promoting rhizobacteria (PGPR): a review. *Cogent Food and Agriculture*, 2(1), 1127500. <https://doi.org/10.1080/23311932.2015.1127500>
- Grandlic, C.J., Mendez, M.O., Chorover, J., Machado, B., Maier, R.M. 2008. Plant growth-promoting bacteria for phytostabilization of mine tailings. *Environmental Science and Technology*, 42(6), 2079–2084. <http://dx.doi.org/10.1021/es072013j>
- Guittouy Philippe, A., Masotti, V., Höhener, P., Boudenne, J.-L., Viglione, J., Laffont-Schwob, I. 2014. Constructed wetlands to reduce metal pollution from industrial catchments in aquatic Mediterranean ecosystems: a review to overcome obstacles and suggest potential solutions. *Environment International*, 64, 1–16. <http://dx.doi.org/10.1016/j.envint.2013.11.016>
- Hallberg, K.B. 2010. New perspectives in acid mine drainage microbiology. *Hydrometallurgy*, 104(3-4), 448-453. <https://doi.org/10.1016/j.hydromet.2009.12.013>
- Hammack, R.W., Dvorak, D.H., Edenborn, H.M. 1993. The use of biogenic hydrogen sulfide to selectively recover copper and zinc from severely contaminated mine drainage, in: Torma, A.E., Wey, J.E., Lakshmanan, V.L. (Eds.), *Biohydrometallurgical Technologies*. The Minerals, Metals and Materials Society, Warrendale, 1, 631-639. <https://doi.org/10.21000/jasmr94010214>
- Hedin, R.S., Watzlaf, G.R., Nairn, R.W. 1994. Passive Treatment of acid mine drainage with limestone. *Journal of Environment Quality*, 23(6), 1338. <http://dx.doi.org/10.2134/jeq1994.00472425002300060030x>

- Hedrich, S., Johnson, D.B. 2012. A modular continuous flow reactor system for the selective bio-oxidation of iron and precipitation of schwertmannite from mine-impacted waters. *Bioresource technology*, 106, 44-49. <https://doi.org/10.1016/j.biortech.2011.11.130>
- Hedrich, S., Johnson, D.B. 2014. Remediation and selective recovery of metals from acidic mine waters using novel modular bioreactors. *Environmental Science and Technology*, 48(20), 12206–12212. <http://dx.doi.org/10.1021/es5030367>
- Heinzel, E., Hedrich, S., Janneck, E., Glombitza, F., Seifert, J., Schlömann, M. 2009a. Bacterial diversity in a mine water treatment plant. *Applied and environmental microbiology*, 75(3), 858-861. <https://doi.org/10.1128/aem.01045-08>
- Heinzel, E., Janneck, E., Glombitza, F., Schlömann, M., Seifert, J. 2009b. Population dynamics of iron-oxidizing communities in pilot plants for the treatment of acid mine waters. *Environmental science and technology*, 43(16), 6138-6144. <https://doi.org/10.1021/es900067d>
- Huang, L. N., Kuang, J. L., and Shu, W. S. 2016. Microbial ecology and evolution in the acid mine drainage model system. *Trends in microbiology*, 24(7), 581-593. <http://dx.doi.org/10.1016/j.tim.2016.03.004>
- Hudson-Edwards, K. 2016. Tackling mine wastes. *Science*, 352(6283), 288-290. <https://doi.org/10.1126/science.aaf3354>
- Hudson-Edwards, K., Dold, B. 2015. Mine waste characterization, management and remediation. *Minerals*, 5(1), 82–85. <http://dx.doi.org/10.3390/min5010082>
- Huisman, J.L., Schouten, G., Schultz, C. 2006. Biologically produced sulphide for purification of process streams, effluent treatment and recovery of metals in the metal and mining industry. *Hydrometallurgy*, 83(1-4), 106-113. <https://doi.org/10.1016/j.hydromet.2006.03.017>
- Humphries, M.S., McCarthy, T.S., Pillay, L., 2017. Attenuation of pollution arising from acid mine drainage by a natural wetland on the Witwatersrand. *South African Journal of Science*, 113(1-2), 1-9. <https://doi.org/10.17159/sajs.2017/20160237>
- Ijaz, K., Wattoo, J. I., Zeshan, B., Majeed, T., Riaz, T., Khalid, S., and Baig, S. 2017. Potential impact of microbial consortia in biomining and bioleaching of commercial metals. *Advancements in Life Sciences*, 5(1), 13-18.
- Jing, Y., He, Z., Yang, X. 2007. Role of soil rhizobacteria in phytoremediation of heavy metal contaminated soils. *Journal of Zhejiang University Science B*, 8(3), 192–207. <http://dx.doi.org/10.1631/jzus.2007.b0192>
- Jobby, R., Jha, P., Yadav, A. K., and Desai, N. 2018. Biosorption and biotransformation of hexavalent chromium [Cr (VI)]: a comprehensive review. *Chemosphere*, 207, 255-266. <https://doi.org/10.1016/j.chemosphere.2018.05.050>
- Johnson, D.B., Hallberg, K.B. 2003. The microbiology of acidic mine waters. *Research in microbiology*, 154(7), 466-473. [https://doi.org/10.1016/s0923-2508\(03\)00114-1](https://doi.org/10.1016/s0923-2508(03)00114-1)
- Johnson, D.B., Hallberg, K.B. 2005. Acid mine drainage remediation options: a review. *Science of the total environment*, 338(1-2), 3-14. <https://doi.org/10.1016/j.scitotenv.2004.09.002>
- Johnson, D.B., Grail, B.M., Hallberg, K.B. 2013. A new direction for biomining: extraction of metals by reductive dissolution of oxidized ores. *Minerals*, 3(1), 49-58. <https://doi.org/10.3390/min3010049>
- Johnson, D.B. 2014. Recent developments in microbiological approaches for securing mine wastes and for recovering metals from mine waters. *Minerals*, 4(2), 279-292. <https://doi.org/10.3390/min4020279>
- Kalin, M. 2004. Passive mine water treatment: the correct approach? *Ecological Engineering*, 22(4-5), 299-304. <https://doi.org/10.1016/j.ecoleng.2004.06.008>

- Kambing'a, M. K., and Syampungani, S. 2012. Performance of tree species growing on tailings dam soils in Zambia: a basis for selection of species for re-vegetating tailings dams. *Journal of Environmental Science and Engineering. B*, 1(7B), 827-831.
- Karaca, O., Cameselle, C., Reddy, K.R., 2018. Mine tailing disposal sites: contamination problems, remedial options and phytocaps for sustainable remediation. *Reviews in Environmental Science and Bio/Technology*, 1-24. <https://doi.org/10.1007/s11157-017-9453-y>
- Kay, C.M., Hedrich, S., and Johnson, D.B. 2013. Selective metal removal from Scandinavian mine waters using novel biomineralization technologies. In *Advanced Materials Research*. Vol. 825, pp. 479-482. <https://doi:10.4028/www.scientific.net/AMR.825.479>
- Kefeni, K.K., Msagati, T.A.M., Mamba, B.B. 2017. Acid mine drainage: prevention, treatment options, and resource recovery: a review. *Journal of Cleaner Production*, 151, 475–493. <http://dx.doi.org/10.1016/j.jclepro.2017.03.082>
- Keller, C., Hammer, D., Kayser, A., Richner, W., Brodbeck, M., Sennhauser, M. 2003. Root development and heavy metal phytoextraction efficiency: comparison of different plant species in the field. *Plant and Soil*, 249(1), 67-81. <https://doi.org/10.1023/a:1022590609042>
- Kieu, H.T., Müller, E., Horn, H. 2011. Heavy metal removal in anaerobic semi-continuous stirred tank reactors by a consortium of sulfate-reducing bacteria. *Water research*, 45(13), 3863-3870. <https://doi.org/10.1016/j.watres.2011.04.043>
- Kimura, S., Hallberg, K.B., Johnson, D.B. 2006. Sulfidogenesis in low pH (3.8–4.2) media by a mixed population of acidophilic bacteria. *Biodegradation*, 17(2), 57–65. <http://dx.doi.org/10.1007/s10532-005-3050-4>
- Kiran, M.G., Pakshirajan, K., Das, G. 2017. An overview of sulfidogenic biological reactors for the simultaneous treatment of sulfate and heavy metal rich wastewater. *Chemical Engineering Science*, 158, 606-620. <https://doi.org/10.1016/j.ces.2016.11.002>
- Klein, R., Tischler, J.S., Mühling, M., Schlömann, M. 2013. Bioremediation of mine water. In: *Geobiotechnology I* (109-172). Springer, Berlin, Heidelberg. https://doi.org/10.1007/10_2013_265
- Knief, C. 2014. Analysis of plant microbe interactions in the era of next generation sequencing technologies. *Frontiers in plant science*, 5, 216. <https://doi.org/10.3389/fpls.2014.00216>
- Kuang, J.-L., Huang, L.-N., Chen, L.-X., Hua, Z.-S., Li, S.-J., Hu, M., Li, J.-T., Shu, W.-S. 2012. Contemporary environmental variation determines microbial diversity patterns in acid mine drainage. *The ISME Journal*, 7(5), 1038–1050. <http://dx.doi.org/10.1038/ismej.2012.139>
- Kuffner, M., Puschenreiter, M., Wieshammer, G., Gorfer, M., Sessitsch, A. 2008. Rhizosphere bacteria affect growth and metal uptake of heavy metal accumulating willows. *Plant and Soil*, 304(1-2), 35–44. <http://dx.doi.org/10.1007/s11104-007-9517-9>
- Kuyucak, N. 2002. Role of microorganisms in mining: generation of acid rock drainage and its mitigation and treatment. *European Journal of Mineral Processing and Environmental Protection*, 2(3), 179-196
- Kuyucak, N., St-Germain, P. 1994. In situ treatment of acid mine drainage by sulphate reducing bacteria in open pits: scale-up experiences. *Proceedings America Society of Mining and Reclamation*, (2), 303–310. <http://dx.doi.org/10.21000/jasmr94020303>
- Lamb, D., Erskine, P.D., Fletcher, A. 2015. Widening gap between expectations and practice in Australian minesite rehabilitation. *Ecological Management and Restoration*, 16(3), 186–195. <http://dx.doi.org/10.1111/emr.12179>
- Laroche, E., Casiot, C., Fernandez-Rojo, L., Desoeuvre, A., Tardy, V., Bruneel, O., Battaglia-Brunet, F., Joulian C., Héry, M. 2018. Dynamics of bacterial communities mediating the treatment of an as-rich acid mine drainage in a field pilot. *Frontiers in microbiology*, 9. <https://doi.org/10.3389/fmicb.2018.03169>

- Le Pape, P., Battaglia-Brunet, F., Parmentier, M., Joulain, C., Gassaud, C., Fernandez-Rojo, L., Guigner, J.M., Ikogou, M., Stetten, L., Olivi, L., Casiot, C. 2017. Complete removal of arsenic and zinc from a heavily contaminated acid mine drainage via an indigenous SRB consortium. *Journal of hazardous materials*, 321, 764-772. <https://doi.org/10.1016/j.jhazmat.2016.09.060>
- Leteinturier, B., Laroche, J., Matera, J., Malaisse, F. 2001. Reclamation of lead/zinc processing wastes at Kabwe, Zambia: a phytogeochemical approach. *South African journal of science*, 97(11-12), 624-627
- Li, X., Lan, S.M., Zhu, Z.P., Zhang, C., Zeng, G.M., Liu, Y.G., Cao, W.C., Song, B., Yang, H., Wang, S.F., Wu, S.H. 2018. The bioenergetics mechanisms and applications of sulfate-reducing bacteria in remediation of pollutants in drainage: a review. *Ecotoxicology and environmental safety*, 158, 162-170. <https://doi.org/10.1016/j.ecoenv.2018.04.025>
- Lizama, K., Fletcher, T.D., Sun, G. 2011. Removal processes for arsenic in constructed wetlands. *Chemosphere*, 84(8), 1032-1043. <https://doi.org/10.1016/j.chemosphere.2011.04.022>
- Lottermoser, B.G. 2010. Introduction to mine wastes. In *Mine Wastes*. Springer, Berlin, Heidelberg, 1-41. http://dx.doi.org/10.1007/978-3-642-12419-8_1
- Ma, Y., Prasad, M.N.V., Rajkumar, M., Freitas, H. 2011. Plant growth promoting rhizobacteria and endophytes accelerate phytoremediation of metalliferous soils. *Biotechnology Advances*, 29(2), 248-258. <http://dx.doi.org/10.1016/j.biotechadv.2010.12.001>
- Ma, Y., Rajkumar, M., Zhang, C., Freitas, H. 2016. Beneficial role of bacterial endophytes in heavy metal phytoremediation. *Journal of Environmental Management*, 174, 14-25. <https://doi.org/10.1016/j.jenvman.2016.02.047>
- Macías, F., Caraballo, M.A., Nieto, J.M., Rötting, T.S., Ayora, C. 2012. Natural pretreatment and passive remediation of highly polluted acid mine drainage. *Journal of environmental management*, 104, 93-100. <https://doi.org/10.1016/j.jenvman.2012.03.027>
- Madsen, E.L. 2011. Microorganisms and their roles in fundamental biogeochemical cycles. *Current opinion in biotechnology*, 22(3), 456-464. <https://doi.org/10.1016/j.copbio.2011.01.008>
- Mahar, A., Wang, P., Ali, A., Awasthi, M.K., Lahori, A.H., Wang, Q., Li, R., Zhang, Z. 2016. Challenges and opportunities in the phytoremediation of heavy metals contaminated soils: a review. *Ecotoxicology and environmental safety*, 126, 111-121. <https://doi.org/10.1016/j.ecoenv.2015.12.023>
- Mains, D., Craw, D., Rufaut, C.G., Smith, C.M.S. 2006. Phytostabilization of gold mine tailings from New Zealand. Part 2: Experimental evaluation of arsenic mobilization during revegetation. *International Journal of Phytoremediation*, 8(2), 163-183. <https://doi.org/10.1080/15226510600742559>
- Maleke, M., Williams, P., Castillo, J., Botes, E., Ojo, A., DeFlaun, M., and van Heerden, E. 2014. Optimization of a bioremediation system of soluble uranium based on the biostimulation of an indigenous bacterial community. *Environmental Science and Pollution Research*, 22(11), 8442-8450. <https://doi.org/10.1007/s11356-014-3980-7>
- Marais, T.S., Huddy, R.J., van Hille, R.P., and Harrison, S.T.L. Effect of operational parameters on the performance of an integrated semi-passive bioprocess. *Mine Water and Circular Economy. Proceedings of the International Mine Water Association (IMWA 2017) Conference*. 262-268
- Marques, C.R. 2018. Extremophilic microfactories: applications in metal and radionuclide bioremediation. *Frontiers in microbiology*, 9. <https://doi.org/10.3389/fmicb.2018.01191>
- Martins, M., Faleiro, L., Barros, R.J., Verissimo, A.R., Costa, M.C. 2009. Biological sulphate reduction using food industry wastes as carbon sources. *Biodegradation*. 20(4), 559-567. <https://doi.org/10.1007/s10532-008-9245-8>

- Mathe, M. and Phiri, A. 2016. The impact of mining on the environment in Gwanda district Zimbabwe: a case study of Blanket Mine. *Imperial Journal of Interdisciplinary Research*, 2(5).
- Mayes, W.M., Batty, L.C., Younger, P.L., Jarvis, A.P., Kõiv, M., Vohla, C., Mander, U., 2009. Wetland treatment at extremes of pH: a review. *Science of the total environment*, 407(13), 3944-3957. <https://doi.org/10.1016/j.scitotenv.2008.06.045>
- Mendez, M.O., Maier, R.M. 2008. Phytoremediation of mine tailings in temperate and arid environments. *Reviews in Environmental Science and Bio/Technology*, 7(1), 47-59. <https://doi.org/10.1007/s11157-007-9125-4>
- Mendez-García, C., Pelaez, A.I., Mesa, V., Sanchez, J., Golyshina, O.V., Ferrer, M. 2015. Microbial diversity and metabolic networks in acid mine drainage habitats. *Frontiers in Microbiology*, 6. <http://dx.doi.org/10.3389/fmicb.2015.00475>
- Mensah, A. K., 2015. Role of revegetation in restoring fertility of degraded mined soils in Ghana: a review. *International journal of biodiversity and conservation*, 7(2), 57-80. <https://doi.org/10.5897/ijbc2014.0775>
- Molwantwa, J.B., Whittington-Jones, K.J., and Rose, P.D. 2004. An investigation into the mechanism underlying enhanced hydrolysis of complex carbon in a biosulphidogenic recycling sludge bed reactor (RSBR). *Water SA*, 30(5), 150-154
- Morin, G., Juillot, F., Casiot, C., Bruneel, O., Personné, J.C., Elbaz-Poulichet, F., Leblanc, M., Ildfonse, P., Calas, G. 2003. Bacterial formation of tooeleite and mixed arsenic (III) or arsenic (V) - iron (III) gels in the Carnoulès acid mine drainage, France. A XANES, XRD, and SEM study. *Environmental science and technology*, 37(9), 1705-1712. <https://doi.org/10.1021/es025688p>
- Mulizane, M., Katsvanga, C. A. T., Nyakudya, I. W., Mupangwa, J. F., 2005. The growth performance of exotic and indigenous tree species in rehabilitating active gold mine tailings dump at Shamva mine in Zimbabwe. *Journal of Applied Sciences and Environmental Management*, 9(2), 57-59. <https://doi.org/10.4314/jasem.v9i2.17292>
- Muthusaravanan, S., Sivarajasekar, N., Vivek, J.S., Paramasivan, T., Naushad, M., Prakashmaran, J., Gayathri, V., Al-Duaij, O.K. 2018. Phytoremediation of heavy metals: mechanisms, methods and enhancements. *Environmental Chemistry Letters*, 1-21. <https://doi.org/10.1007/s10311-018-0762-3>
- Nairn, R.W., LaBar, J.A., Strevett, K.A., Strosnider, W.H., Morris, D., Neely, C.A., Garrido, A., Santamaria, B., Oxenford, L., Kauk, K., Carter, S., Furneaux B. 2010. A large, multi-cell, ecologically engineered passive treatment system for ferruginous lead-zinc mine waters. In *Proceedings of the International Mine Water Association Annual Meeting (IMWA 2010, Sydney, Nova Scotia, Canada*. pp. 21-24
- Nancucheo, I., Johnson, D.B. 2012. Selective removal of transition metals from acidic mine waters by novel consortia of acidophilic sulfidogenic bacteria. *Microbial biotechnology*, 5(1), 34-44. <https://doi.org/10.1111/j.1751-7915.2011.00285.x>
- Nancucheo, I., and Johnson, D.B. 2014. Removal of sulfate from extremely acidic mine waters using low pH sulfidogenic bioreactors. *Hydrometallurgy*, 150, 222-226 <http://dx.doi.org/10.1016/j.hydromet.2014.04.025>
- Nancucheo, I., Bitencourt, J.A., Sahoo, P.K., Alves, J.O., Siqueira, J.O., Oliveira, G. 2017. Recent developments for remediating acidic mine waters using sulfidogenic bacteria. *BioMed research international*, 2017. <https://doi.org/10.1155/2017/7256582>
- Navarro, A., Collado, D., Carbonell, M., and Sanchez, J. A. 2004. Impact of mining activities on soils in a semi-arid environment: Sierra Almagrera district, SE Spain. *Environmental Geochemistry and Health*, 26(4), 383-393.
- Neculita, C.M., Zagury, G.J. 2008. Biological treatment of highly contaminated acid mine drainage in batch reactors: long-term treatment and reactive mixture

- characterization. *Journal of Hazardous Materials*, 157(2-3), 358-366. <https://doi.org/10.1016/j.jhazmat.2008.01.002>
- Neculita, C.M., Zagury, G.J., Bussi re, B. 2007. Passive treatment of acid mine drainage in bioreactors using sulfate-reducing bacteria. *Journal of Environmental Quality*, 36(1), 1-16. <https://doi.org/10.2134/jeq2006.0066>
- Newsome, L., Morris, K., and Lloyd, J.R. 2014. The biogeochemistry and bioremediation of uranium and other priority radionuclides. *Chemical Geology*, 363, 164-184. <http://dx.doi.org/10.1016/j.chemgeo.2013.10.034>
- Newson, T.A., Fahey, M. 2003. Measurement of evaporation from saline tailings storages. *Engineering Geology*, 70(3-4), 217–233. [http://dx.doi.org/10.1016/s0013-7952\(03\)00091-7](http://dx.doi.org/10.1016/s0013-7952(03)00091-7)
- Nirola, R., Megharaj, M., Beecham, S., Aryal, R., Thavamani, P., Vankateswarlu, K., and Saint, C. 2016. Remediation of metalliferous mines, revegetation challenges and emerging prospects in semi-arid and arid conditions. *Environmental Science and Pollution Research*, 23(20), 20131-20150. <http://dx.doi.org/10.1007/s11356-016-7372-z>
- Nordberg, G.F., Fowler, B.A., and Nordberg, M. (Eds.). 2014. *Handbook on the toxicology of metals*. Academic press. Pp 1381
- Nordstrom, D.K. 2009. Acid rock drainage and climate change. *Journal of Geochemical Exploration*, 100(2-3), 97–104. <http://dx.doi.org/10.1016/j.gexplo.2008.08.002>
- Nordstrom, D.K., Alpers, C.N., Ptacek, C.J., Blowes, D.W. 2000. Negative pH and extremely acidic mine waters from Iron Mountain, California. *Environmental Science and Technology*, 34(2), 254-258. <https://doi.org/10.1021/es990646v>
- Novhe, N. O., Yibas, B., Coetzee, H., Atanasova, M., Netshitungulwana, R., Modiba, M., Mashalane, T. 2016. Long-term remediation of acid mine drainage from abandoned coal mine using integrated (anaerobic and aerobic) passive treatment system in South Africa: a pilot study. *Mining Meets Water-Conflicts and Solutions*, 1, 668-675
- Novo, L.A., Castro, P.M., Alvarenga, P., da Silva, E.F. 2018. Plant Growth–Promoting Rhizobacteria-Assisted Phytoremediation of Mine Soils, in: *Bio-Geotechnologies for Mine Site Rehabilitation* (281-295). <https://doi.org/10.1016/b978-0-12-812986-9.00016-6>
- Nyakudya I.W., Jimu, L., Katsvanga C.A.T. and Dafana M. (2011). Comparative analysis of the early growth performance of indigenous *Acacia* species in revegetating Trojan nickel mine tailings in Zimbabwe. *African Journal of Environmental Science and Technology*, 5(3), 218-227. <http://www.academicjournals.org/AJEST>
- O'sullivan, A.D., Moran, B.M., Otte, M.L. 2004. Accumulation and fate of contaminants (Zn, Pb, Fe and S) in substrates of wetlands constructed for treating mine wastewater. *Water, Air, and Soil Pollution*, 157(1-4), 345-364. <https://doi.org/10.1023/b:wate.0000038882.09628.ab>
- Padmavathiamma, P.K., Ahmed, M., and Rahman, H.A. 2014. Phytoremediation - a sustainable approach for contaminant remediation in arid and semi-arid regions–a review. *Emirates Journal of Food and Agriculture*, 757-772. <https://doi.org/10.9755/ejfa.v26i9.18202>
- Panda, S., Mishra, S., Akcil, A. 2016. Bioremediation of acidic mine effluents and the role of sulfidogenic biosystems: a mini-review. *Euro-Mediterranean Journal for Environmental Integration*, 1(1), 8. <https://doi.org/10.1007/s41207-016-0008-3>
- Park, H., Lee, E.H., Ka, K.H., Eom, A.H. 2016. Community structures of arbuscular mycorrhizal fungi in soils and plant roots inhabiting abandoned mines of Korea. *Mycobiology*, 44(4), 277-282. <http://dx.doi.org/10.5941/myco.2016.44.4.277>
- Paz-Ferreiro, J., Lu, H., Fu, S., Mendez, A., Gasco, G. 2014. Use of phytoremediation and biochar to remediate heavy metal polluted soils: a review. *Solid Earth*, 5(1), 65–75. <http://dx.doi.org/10.5194/se-5-65-2014>
- Peng, W., Li, X., Xiao, S., Fan, W., 2018. Review of remediation technologies for sediments contaminated by heavy metals. *Journal of soils and sediments*, 18(4), 1701-1719. <https://doi.org/10.1007/s11368-018-1921-7>

- Perry A. and Kleinmann RL. 1991. The use of constructed wetlands in the treatment of acid mine drainage. In Natural resources forum (Vol. 15, No. 3, pp. 178-184). Oxford, UK: Blackwell Publishing Ltd.
- Petrisor, I.G., Dobrota, S., Komnitsas, K., Lazar, I., Kuperberg, J.M., Serban, M. 2004. Artificial inoculation - perspectives in tailings phytostabilization. *International Journal of Phytoremediation*, 6(1), 1–15. <http://dx.doi.org/10.1080/16226510490439918>
- Pietrzykowski, M. 2015. Reclamation and reconstruction of terrestrial ecosystems on mine sites - ecological effectiveness assessment, in: Govil JN et al (eds) *Series: energy science and technology, coal energy*, 2nd edn. Studium Press LLC, New Delhi, Houston, 121–151
- Piha, M.I., Vallack, H.W., Reeler, B.M., Michael, N. 1995. A low input approach to vegetation establishment on mine and coal ash wastes in semi-arid regions. I. Tin mine tailings in Zimbabwe. *Journal of Applied Ecology*, 372-381. <https://doi.org/10.2307/2405103>
- Pilon-Smits, E. 2005. Phytoremediation. *Annu. Rev. Plant Biol.*, 56, 15-39.
- Pollmann, K., Kutschke, S., Matys, S., Kostudis, S., Hopfe, S., and Raff, J. 2016. Novel biotechnological approaches for the recovery of metals from primary and secondary resources. *Minerals*, 6(2), 54. <https://doi.org/10.3390/min6020054>
- Plewniak, F., Crognale, S., Rossetti, S., Bertin, P.N. 2018. A genomic outlook on bioremediation: the case of arsenic removal. *Frontiers in microbiology*, 9, 820. <https://doi.org/10.3389/fmicb.2018.00820>
- Pond, A.P., White, S.A., Milczarek, M., Thompson, T.L. 2005. Accelerated weathering of biosolid-amended copper mine tailings. *Journal of environmental quality*, 34(4), 1293-1301. <https://doi.org/10.2134/jeq2004.0405>
- Rajkumar, M., Ae, N., Freitas, H. 2009. Endophytic bacteria and their potential to enhance heavy metal phytoextraction. *Chemosphere*, 77(2), 153-160. <https://doi.org/10.1016/j.chemosphere.2009.06.047>
- Rajkumar, M., Nagendran, R., Lee, K.J., Lee, W.H., Kim, S.Z. 2006. Influence of plant growth promoting bacteria and Cr⁶⁺ on the growth of indian mustard. *Chemosphere*, 62(5), 741–748. <http://dx.doi.org/10.1016/j.chemosphere.2005.04.117>
- Rajkumar, M., Sandhya, S., Prasad, M.N.V., Freitas, H. 2012. Perspectives of plant-associated microbes in heavy metal phytoremediation. *Biotechnology advances*, 30(6), 1562-1574. <https://doi.org/10.1016/j.biotechadv.2012.04.011>
- Rakotonimaro, T.V., Guittonny, M., and Neculita, C.M. 2018. Stabilization of hard rock mines tailings with organic amendments: pore water quality control and revegetation-a review. *Desalination and water treatment*, 112, 53-71.
- Rawlings, D.E., Dew, D., du Plessis, C. 2003. Biomineralization of metal-containing ores and concentrates. *TRENDS in Biotechnology*, 21(1), 38-44. [https://doi.org/10.1016/s0167-7799\(02\)00004-5](https://doi.org/10.1016/s0167-7799(02)00004-5)
- Rayu, S., Karpouzas, D.G., Singh, B.K. 2012. Emerging technologies in bioremediation: constraints and opportunities. *Biodegradation*, 23(6), 917-926. <https://doi.org/10.1007/s10532-012-9576-3>
- Rezadehbashi, M., Baldwin, S. 2018. Core sulphate-reducing microorganisms in metal-removing semi-passive biochemical reactors and the co-occurrence of methanogens. *microorganisms*, 6(1), p.16. <http://dx.doi.org/10.3390/microorganisms6010016>
- Robinson, B., Schulin, R., Nowack, B., Roulier, S., Menon, M., Clothier, B., Green, S., Mills, T. 2006. Phytoremediation for the management of metal flux in contaminated sites. *Forest Snow and Landscape Research*, 80(2), 221-224.
- Romero, F.M., Prol-Ledesma, R.M., Canet, C., Alvares, L.N., Pérez-Vázquez, R. 2010. Acid drainage at the inactive Santa Lucia mine, western Cuba: Natural attenuation of arsenic, barium and lead, and geochemical behavior of rare earth elements. *Appl. Geochemistry*, 25, 716-727. <https://doi.org/10.1016/j.apgeochem.2010.02.004>

- Rowe, O.F., Johnson, D.B. 2008. Comparison of ferric iron generation by different species of acidophilic bacteria immobilized in packed-bed reactors. *Systematic and applied microbiology*, 31(1), 68-77. <https://doi.org/10.1016/j.syapm.2007.09.001>
- Roychowdhury, A., Sarkar, D., and Datta, R. 2015. Remediation of acid mine drainage-impacted water. *Current Pollution Reports*, 1(3), 131-141. <https://doi.org/10.1007/s40726-015-0011-3>
- Sanchez-Andrea, I., Stams, A.J., Weijma, J., Gonzalez-Contreras, P., Dijkman, H., Rozendal, R.A., Johnson, D.B. 2016. A case in support of implementing innovative bio-processes in the metal mining industry. *FEMS microbiology letters*, 363(11). <https://doi.org/10.1093/femsle/fnw106>
- Sarwar, N., Imran, M., Shaheen, M.R., Ishaque, W., Kamran, M.A., Matloob, A., Rehim, A., Hussain, S. 2017. Phytoremediation strategies for soils contaminated with heavy metals: modifications and future perspectives. *Chemosphere*, 171, 710–721. <http://dx.doi.org/10.1016/j.chemosphere.2016.12.116>
- Sbabou, L., Idir, Y., Bruneel, O., Le Quéré, A., Aurag, J., Béna, G., Filali-Maltouf, A. 2016. Characterization of root-nodule bacteria isolated from *Hedysarum spinosissimum* L, growing in mining sites of northeastern region of Morocco. *SOJ Microbiology and Infectious Diseases*, 4(3), 1–8. <http://dx.doi.org/10.15226/sojmid/4/3/00156>
- Schippers, A., Breuker, A., Blazejak, A., Bosecker, K., Kock, D., Wright, T.L. 2010. The biogeochemistry and microbiology of sulfidic mine waste and bioleaching dumps and heaps, and novel Fe (II)-oxidizing bacteria. *Hydrometallurgy*, 104(3-4), 342-350. <https://doi.org/10.1016/j.hydromet.2010.01.012>
- Schippers, A., Hedrich, S., Vasters, J., Drobe, M., Sand, W., Willscher, S. 2013. Biomining: metal recovery from ores with microorganisms. *Geobiotechnology I*, 1–47. http://dx.doi.org/10.1007/10_2013_216
- Schippers, A., Jozsa, P., Sand, W. 1996. Sulfur chemistry in bacterial leaching of pyrite. *Applied and Environmental Microbiology*, 62(9), 3424-3431.
- Sheng, Y., Kaley, B., Bibby, K., Grettenberger, C., Macalady, J.L., Wang, G., Burgos, W.D. 2017. Bioreactors for low-pH iron(ii) oxidation remove considerable amounts of total iron. *RSC Advances*, 7(57), 35962–35972. <http://dx.doi.org/10.1039/c7ra03717a>
- Sheoran, A.S., Sheoran, V. 2006. Heavy metal removal mechanism of acid mine drainage in wetlands: a critical review. *Minerals engineering*, 19(2), 105-116. <https://doi.org/10.1016/j.mineng.2005.08.006>
- Sheoran, A.S., Sheoran, V., Choudhary, R.P. 2010. Bioremediation of acid-rock drainage by sulphate-reducing prokaryotes: a review. *Minerals Engineering*, 23(14), 1073-1100. <https://doi.org/10.1016/j.mineng.2010.07.001>
- Sheoran, V., Sheoran, A. S., Poonia, P. 2012. Phytoremediation of metal contaminated mining sites. *International Journal of Earth Sciences and Engineering*, 5(3), 428-436.
- Sheoran, V., Sheoran, A.S., Poonia, P. 2016. Factors affecting phytoextraction: a review. *Pedosphere*, 26(2), 148-166. [https://doi.org/10.1016/s1002-0160\(15\)60032-7](https://doi.org/10.1016/s1002-0160(15)60032-7)
- Shutcha, M.N., Mpundu, M.M., Faucon, M.-P., Luhembwe, M.N., Visser, M., Colinet, G., Meerts, P. 2010. Phytostabilisation of copper-contaminated soil in Katanga: an experiment with three native grasses and two amendments. *International Journal of Phytoremediation*, 12(6), 616–632. <http://dx.doi.org/10.1080/15226510903390411>.
- Shutcha, M.N., Faucon, M.-P., Kamengwa Kissi, C., Colinet, G., Mahy, G., Ngongo Luhembwe, M., Visser, M., Meerts, P. 2015. Three years of phytostabilisation experiment of bare acidic soil extremely contaminated by copper smelting using plant biodiversity of metal-rich soils in tropical Africa (Katanga, DR Congo). *Ecological Engineering*, 82, 81–90. <http://dx.doi.org/10.1016/j.ecoleng.2015.04.062>

- Simate, G. S., Ndlovu, S., 2014. Acid mine drainage: Challenges and opportunities. *Journal of Environmental Chemical Engineering*, 2(3), 1785-1803. <http://dx.doi.org/10.1016/j.jece.2014.07.021>
- Singer, P.C., Stumm, W. 1970. Acidic mine drainage: the rate-determining step. *Science*, 167(3921), 1121-1123. <https://doi.org/10.1126/science.167.3921.1121>
- Singh, A.N., Raghubanshi, A.S., Singh, J.S. 2002. Plantations as a tool for mine spoil restoration. *Curr Sci* 82:1436–1441
- Skousen, J., Zipper, C.E., Rose, A., Ziemkiewicz, P.F., Nairn, R., McDonald, L.M., Kleinmann, R.L. 2017. Review of passive systems for acid mine drainage treatment. *Mine Water and the Environment*, 36(1), 133-153. doi:[10.1007/s10230-016-0417-1](https://doi.org/10.1007/s10230-016-0417-1)
- Sracek, O., Filip, J., Mihaljevic, M., Kríbek, B., Majer, V., Veselovsky, F. 2011. Attenuation of dissolved metals in neutral mine drainage in the Zambian Copperbelt. *Environmental monitoring and assessment*, 172(1-4), 287-299. <https://doi.org/10.1007/s10661-010-1334-6>
- Sracek O., Kríbek B., Mihaljevic M., Majer V., Veselovsky F., Vencelides Z., and Nyambe I. 2012. Mining-related contamination of surface water and sediments of the Kafue River drainage system in the Copperbelt district, Zambia: an example of a high neutralization capacity system. *Journal of Geochemical Exploration*, 112, 174-188 <https://doi.org/10.1016/j.gexplo.2011.08.007>
- Streten-Joyce, C., Manning, J., Gibb, K.S., Neilan, B.A., Parry, D.L. 2013. The chemical composition and bacteria communities in acid and metalliferous drainage from the wet–dry tropics are dependent on season. *Science of The Total Environment*, 443, 65–79. <http://dx.doi.org/10.1016/j.scitotenv.2012.10.024>
- Stumm, W., Morgan, J.J. 1981. *Aquatic chemistry: an introduction emphasizing chemical equilibria in natural Waters*. Wiley, New York.
- Teng, W., Kuang, J., Luo, Z., and Shu, W. 2017. Microbial diversity and community assembly across environmental gradients in acid mine drainage. *Minerals*, 7(6), 106. <http://dx.doi.org/10.3390/min7060106>
- Tetteh, E., Ampofo, K., Logah, V. 2015a. Adopted practices for mined land reclamation in Ghana: a case study of Anglogold Ashanti Iduapriem mine ltd. *Journal of Science and Technology (Ghana)*, 35(2), 77. <http://dx.doi.org/10.4314/just.v35i2.8>
- Tetteh, E. N., Logah, V., Twum-Ampofo, K., and Partey, S.T. 2015b. Effect of duration of reclamation on soil quality indicators of a surface–mined acid forest oxisol in South–Western Ghana. *West African Journal of Applied Ecology*, 23(2), 63-72.
- Thavamani, P., Samkumar, R.A., Satheesh, V., Subashchandrabose, S.R., Ramadass, K., Naidu, R., Venkateswarlu, K., Megharaj, M. 2017. Microbes from mined sites: harnessing their potential for reclamation of derelict mine sites. *Environmental pollution*, 230, 495-505. <https://doi.org/10.1016/j.envpol.2017.06.056>
- Thijs S, Sillen W, Weyens N, Vangronsveld J. 2017 Phytoremediation: state-of-the-art and a key role for the plant microbiome in future trends and research prospects. *Int J Phytorem* 19(1), 23–38. <https://doi.org/10.1080/15226514.2016.1216076>
- Tischler, J.S., Wiacek, C., Janneck, E., Schlömann, M. 2013. Microbial abundance in the schwertmannite formed in a mine water treatment plant. *Mine Water and the Environment*, 32(4), 258–265. <http://dx.doi.org/10.1007/s10230-013-0250-8>
- Titshall, L.W., Hughes, J.C., Bester, H.C. 2013. Characterisation of alkaline tailings from a lead/zinc mine in South Africa and evaluation of their revegetation potential using five indigenous grass species. *South African Journal of Plant and Soil*, 30(2), 97-105. <https://doi.org/10.1080/02571862.2013.807361>
- Tordoff, G.M., Baker, A.J.M., Willis, A.J. 2000. Current approaches to the revegetation and reclamation of metalliferous mine wastes. *Chemosphere*, 41(1-2), 219-228. [https://doi.org/10.1016/s0045-6535\(99\)00414-2](https://doi.org/10.1016/s0045-6535(99)00414-2)

- Trivedi, P., Pandey, A., Sa, T. 2007. Chromate reducing and plant growth promoting activities of psychrotrophic *Rhodococcus erythropolis* MtCC 7905. *Journal of Basic Microbiology*, 47(6), 513–517. <http://dx.doi.org/10.1002/jobm.200700224>
- Turnau, K., Henriques, F.S., Anielska, T., Renker, C., Buscot, F. 2007. Metal uptake and detoxification mechanisms in *Erica andevalensis* growing in a pyrite mine tailing. *Environmental and experimental botany*, 61(2), 117-123. <https://doi.org/10.1016/j.envexpbot.2007.05.001>
- Ullah, A., Heng, S., Munis, M.F.H., Fahad, S., Yang, X. 2015. Phytoremediation of heavy metals assisted by plant growth promoting (PGP) bacteria: a review. *Environmental and Experimental Botany*, 117, 28-40. <https://doi.org/10.1016/j.envexpbot.2015.05.001>
- Valentin-Vargas, A., Neilson, J.W., Root, R.A., Chorover, J., Maier, R.M. 2018. Treatment impacts on temporal microbial community dynamics during phytostabilization of acid-generating mine tailings in semiarid regions. *Science of The Total Environment*, 618, 357–368. <http://dx.doi.org/10.1016/j.scitotenv.2017.11.010>.
- van Hille, R., Mooruth, N., Marais, T., Naidoo, N., Moss, G., Harrison, S., and Muhlbauer, R. 2016. Development of a pilot-scale semi-passive system for the bioremediation of ARD. *Proceedings of the International Mine Water Association (IMWA 2016) Conference*, 957-964.
- Van Rensburg, L., Morgenthal, T. 2004. The effect of woodchip waste on vegetation establishment during platinum tailings rehabilitation. *South African journal of science*, 100(5-6), 294-300. <https://doi.org/10.1023/b:wate.0000022943.27442.2a>
- Vasquez, Y., Escobar, M. C., Neculita, C. M., Arbeli, Z., and Roldan, F. 2016. Biochemical passive reactors for treatment of acid mine drainage: effect of hydraulic retention time on changes in efficiency, composition of reactive mixture, and microbial activity. *Chemosphere*, 153, 244-253. <http://dx.doi.org/10.1016/j.chemosphere.2016.03.052>
- Venkateswarlu, K., Nirola, R., Kuppusamy, S., Thavamani, P., Naidu, R., Megharaj, M. 2016. Abandoned metalliferous mines: ecological impacts and potential approaches for reclamation. *Reviews in Environmental Science and Bio/Technology*, 15(2), 327-354. <https://doi.org/10.1007/s11157-016-9398-6>
- Vera, M., Schippers, A., Sand, W. 2013. Progress in bioleaching: fundamentals and mechanisms of bacterial metal sulfide oxidation - part A. *Applied microbiology and biotechnology*, 97(17), 7529-7541. <https://doi.org/10.1007/s00253-013-4954-2>
- Vivas, A., Biro, B., Ruiz-Lozano, J.M., Azcon, R. 2006. Two bacterial strains isolated from a Zn-polluted soil enhance plant growth and mycorrhizal efficiency under Zn-toxicity. *Chemosphere*, 62(9), 1523–1533. <http://dx.doi.org/10.1016/j.chemosphere.2005.06.053>
- Volant, A., Bruneel, O., Desoeuvre, A., Héry, M., Casiot, C., Bru, N., Delpoux, S., Fahy, A., Javerliat, F., Bouchez, O., Duran, R., Bertin, P.N., Elbaz-Poulichet, F., Lauga, B. 2014. Diversity and spatiotemporal dynamics of bacterial communities: physicochemical and other drivers along an acid mine drainage. *FEMS Microbiology Ecology*, 90(1), 247–263. <http://dx.doi.org/10.1111/1574-6941.12394>.
- wa Ilunga, E.I., Mahy, G., Piqueray, J., Séleck, M., Shutcha, M.N., Meerts, P., Faucon, M.-P. 2015. Plant functional traits as a promising tool for the ecological restoration of degraded tropical metal-rich habitats and revegetation of metal-rich bare soils: A case study in copper vegetation of Katanga, DRC. *Ecological Engineering*, 82, 214–221. <http://dx.doi.org/10.1016/j.ecoleng.2015.04.084>.
- Wang, F. 2017. Occurrence of arbuscular mycorrhizal fungi in mining-impacted sites and their contribution to ecological restoration: mechanisms and applications. *Critical Reviews in Environmental Science and Technology*, 47(20), 1901–1957. <http://dx.doi.org/10.1080/10643389.2017.1400853>

- Wani, P.A., Khan, M.S., Zaidi, A. 2008. Effect of metal-tolerant plant growth-promoting *Rhizobium* on the performance of pea grown in metal-amended soil. Archives of Environmental Contamination and Toxicology, 55(1), 33–42. <http://dx.doi.org/10.1007/s00244-007-9097-y>
- Weber, K.P., Werker, A., Gehder, M., Senger, T., Legge, R.L. 2010. Influence of the microbial community in the treatment of acidic iron-rich water in aerobic wetland mesocosms. Bioremediation Journal, 14(1), 28–37. <https://doi.org/10.1080/10889860903490219>
- Weiersbye, I.M., Witkowski, E.T.F., Reichardt, M. 2006. Floristic composition of gold and uranium tailings dams, and adjacent polluted areas, on South Africa's deep-level mines. Bothalia, 36(1). <http://dx.doi.org/10.4102/abc.v36i1.349>
- Wenzel, W.W. 2009. Rhizosphere processes and management in plant-assisted bioremediation (phytoremediation) of soils. Plant and Soil, 321(1-2), 385–408. <https://doi.org/10.1007/s11104-008-9686-1>
- Whitehead, P.G., Cosby, B.J., Prior, H. 2005. The Wheal Jane wetlands model for bioremediation of acid mine drainage. Sci. Total Environ. 338, 125–135. <https://doi.org/10.1016/j.scitotenv.2004.09.004>
- Whitman, W.B., Coleman, D.C., Wiebe, W.J. 1998. Prokaryotes: the unseen majority. Proceedings of the National Academy of Sciences, 95(12), 6578–6583. <https://doi.org/10.1073/pnas.95.12.6578>
- Wood, M.K., Buchanan, B.A., Skeet, W. 1995. Shrub preference and utilization by big game on New Mexico reclaimed mine land. Journal of Range Management, 48(5), 431. <http://dx.doi.org/10.2307/4002247>
- Yadav, K.K., Gupta, N., Kumar, A., Reece, L.M., Singh, N., Rezania, S., Khan, S.A. 2018. Mechanistic understanding and holistic approach of phytoremediation: a review on application and future prospects. Ecological Engineering, 120, 274–298. <https://doi.org/10.1016/j.ecoleng.2018.05.039>
- Ye, Z. H., Wong, J. W. C., Wong, M. H., Lan, C. Y., Baker, A. J. M., 1999. Lime and pig manure as ameliorants for revegetating lead/zinc mine tailings: a greenhouse study. Bioresource Technology, 69(1), 35–43. [https://doi.org/10.1016/s0960-8524\(98\)00171-0](https://doi.org/10.1016/s0960-8524(98)00171-0)
- Zhang, L., Wang, M.H., Hu, J., Ho, Y.S. 2010. A review of published wetland research, 1991–2008: ecological engineering and ecosystem restoration. Ecological Engineering, 36(8), 973–980. <https://doi.org/10.1016/j.ecoleng.2010.04.029>
- Zhang, M., Wang, H. 2014. Organic wastes as carbon sources to promote sulphate reducing bacterial activity for biological remediation of acid mine drainage. Miner Eng, 69, 81–90. <https://doi.org/10.1016/j.minereng.2014.07.010>
- Zhang, G., Ouyang, X., Li, H., Fu, Z., Chen, J. 2016. Bioremoval of antimony from contaminated waters by a mixed batch culture of sulfate-reducing bacteria. International Biodeterioration and Biodegradation, 115, 148–155. <https://doi.org/10.1016/j.ibiod.2016.08.007>
- Zhao, F.J., Lombi, E., McGrath, S.P. 2003. Assessing the potential for zinc and cadmium phytoremediation with the hyperaccumulator *Thlaspi caerulescens*. Plant and soil, 249(1), 37–43. <https://doi.org/10.1023/a:1022530217289>
- Ziemkiewicz, P. F., Skousen, J. G., and Simmons, J. 2003. Long-term performance of passive acid mine drainage treatment systems. Mine Water and the Environment, 22(3), 118–129.
- Zuniga, C., Zaramela, L., Zengler, K. 2017. Elucidation of complexity and prediction of interactions in microbial communities. Microbial biotechnology, 10(6), 1500–1522. <https://doi.org/10.1111/1751-7915.12855>