CHAPTER TWO

Utilization of Biowaste for Mine Spoil Rehabilitation

H. Wijesekara[*], N.S. Bolan[*,], M. Vithanage[†,1], Y. Xu[‡], S. Mandal[‡], S.L. Brown[§], G.M. Hettiarachchi[¶], G.M. Pierzynski[¶], L. Huang[††], Y.S. Ok[‡‡], M.B. Kirkham[¶], C.P. Saint[§§], A. Surapaneni[¶¶]**

[*]Global Centre for Environmental Remediation (GCER), Advanced Technology Centre, Faculty of Science and Information Technology, The University of Newcastle, Callaghan, New South Wales, Australia
[**]Cooperative Research Centre for Contamination Assessment and Remediation of the Environment (CRC CARE), Advanced Technology Centre, Faculty of Science and Information Technology, The University of Newcastle, Callaghan, New South Wales, Australia
[†]National Institute of Fundamental Studies, Kandy, Sri Lanka
[‡]Future Industries Institute (FII), University of South Australia, Mawson Lakes, Australia
[§]School of Environmental and Forest Sciences, University of Washington, Seattle, WA, United States
[¶]Department of Agronomy, Throckmorton Plant Sciences Center, Kansas State University, Manhattan, KS, United States
[††]Centre for Mined Land Rehabilitation, Sustainable Minerals Institute, The University of Queensland, Brisbane, Queensland, Australia
[‡‡]Korea Biochar Research Center, Environmental Remediation and Restoration Laboratory, Kangwon National University, Chuncheon, Gangwon Province, Korea
[§§]Natural & Built Environments Research Centre, University of South Australia, Building H, Minerals Lane, Mawson Lakes, South Australia, Australia
[¶¶]South East Water, WatersEdge, Frankston, Victoria, Australia
[1]Corresponding author. E-mail address: meththikavithanage@gmail.com

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Abstract

Globally, around 0.4×10^6 km^2 area of land is estimated to be disturbed by mining activities, thereby contributing to severe environmental consequences including the generation of large amounts of mine spoils. The shortfall in topsoil due to poor striping practices and low levels of organic matter have been identified as common problems in rehabilitation of mining spoil. High heavy metal concentrations in mine spoil can adversely impact microbial activity and subsequent revegetation succession. The release of acids associated with mine spoils (ie, acid mine drainage through oxidation of pyrite) can also create adverse effects on the surrounding vegetation.

Large quantities of biowaste, such as manure compost, biosolids, and municipal solid waste (MSW) that are low in contaminants [including metal(loid)s] can be used to rehabilitate mine spoils. These biowastes provide a source of nutrients and improve the fertility of spoils. These biowastes also act as a sink for metal(loid)s in mine tailings reducing their bioavailability through adsorption, complexation, reduction, and volatilization of metal(loid)s.

This review provides an overview of the sources of biowastes and the current regulations for utilization; describes their benefits in terms of improving the physical, chemical, and biological properties of mine spoils; and elaborates on the role of the utilization of biowastes on mine spoil rehabilitation through several case studies. Finally, future research needs and strategies are identified in terms of sustainable biowaste utilization in mine spoil rehabilitation.

LIST OF ABBREVIATIONS

C	Control
CCE	Calcium carbonate equivalent
CEC	Cation exchange capacity
CEPI	Confederation of European Paper Industries
CLBAR	Contaminant limited biosolids application rate
CWA	Clean Water Act
DAT	Days after treatment
EC	Electrical conductivity
HC	High compost at 269 tons/ha
HSD	Honest significant difference
INT	2 (*p*-iodophenyl)-3-(*p*-nitrophenyl)5-phenyl tetrazolium chloride
INTF	Iodonitrote-trazolium formazan
LC	Low compost at 45 tons/ha
LCA	Life cycle assessment

LOAEC	Lowest observed adverse effects concentration
MBC	Microbial biomass carbon
MBR	Microbial basal respiration
MINAS	Mineral Accounting System
MQ	Metabolic quotient
MSW	Municipal solid waste
MSWC	Municipal solid waste compost
ND	Not detected
NEPC	National Environment Protection Council
NLBAR	Nitrogen limited biosolids application rate
NSWEPA	New South Wales Environment Protection Authority
NWQMS	National Water Quality Management Strategy
OC	Organic carbon
OM	Organic matter
ON	Organic nitrogen
OSMRE	Office of Surface Mining Reclamation and Enforcement
OTU's	Operational taxonomic units
PDER	Pennsylvania Department of Environmental Resources
PLFA	Phospholipid fatty acid
PNP	ρ-Nitrophenol
POP's	Persistent organic pollutants
PPCP's	Pharmaceuticals and personal care products
RCRA	Resource Conservation and Recovery Act
SOM	Soil organic matter
SR	Sugarcane residue
SRC	Sugarcane residue compost
TAH	Total anaerobic herterophs
TAM	Total aerobic microorganisms
TC	Total carbon
TF's	Transfer factors
TN	Total nitrogen
TOC	Total organic carbon
TON	Total organic nitrogen
TSP	Triple super phosphate
USEPA	United States Environmental Protection Agency
WHC	Water-holding capacity

1. INTRODUCTION

Mining generates large amounts of valuable natural resources such as minerals, coal, and energy to advance global economic prosperity. However, mining inevitably disturbs the land. Globally, around 0.4×10^6 km^2 area of land is estimated to be disturbed by mining activities and a significant percentage of this area has never been reclaimed, thereby contributing to severe environmental consequences (Hooke and Martin-Duque, 2012). Mine spoils are waste materials from underground mining, quarries, or open-cast excavations. Therefore, mine spoils are often considered as drastically disturbed, nutritionally, and microbiologically reduced habitats that need urgent restoration (Singh et al., 2004). For instance, gold mine spoils include tailings, subsoils, and oxidized waste, while coal mine spoils include fireclay, subsoils, and mudstone (O'Reilly, 1997). Mine tailings represent a major component of mine spoils; "mine tailings" is a generic term which refers to a range of matrices such as mixtures of crushed rock, processing fluids from mills, washeries, or concentrated materials that remain after extracting the mine resource (Kossoff et al., 2014). There is ample evidence that mine tailings contain hazardous contaminants, which may enter groundwater and food chains (Fig. 1). In excavation process of mining, upper segments of mountain, ridge, or valley have to be removed. This "take away" fraction is known as "overburden" which results in unsightly and unproductive soil (Diamond, 1999; Ghose, 2001; Johnson, 2003; Sopper, 1992).

Low levels of organic matter (OM) have been identified as the common problem in mine spoils, which lead to poor soil health for plant growth and soil microbial life (Castillejo and Castello, 2010; Larney and Angers, 2012; Pulford, 1991). Mine spoils with low plant diversity are likely to cause a shortage of OM accumulation. The shortage of OM at mine sites results in poor soil texture and structure (Castillejo and Castello, 2010). Poor water-holding capacity (WHC), erosion of tailings by wind (ie, air pollution) and water, crusting, and cracking of soils are some of the adverse effects that result from poor spoil structure (Hossner and Hons, 1992). Elevated heavy metal concentrations are another common feature of most mine spoils. For instance, Zn and Pb ranging from 6,000 to 14,700 and 2,100 to 27,000 mg/kg, respectively, have been reported in soils contaminated by mining and smelting (Brown et al., 2003a). Soils contaminated with tailings of an old Spanish Pb-Zn mine have been reported, and the mean concentrations of Pb, Zn, and Cu were 28,453, 7,000, and 308 mg/kg, respectively (Rodriguez et al., 2009). Elevated heavy metal concentrations adversely impact the microbial activity and decomposition of OM in terms of formation of humus (O'Reilly, 1997). Production of acids associated with mine spoils (ie, oxidation of pyrite) can also lead to adverse effects on the surrounding vegetation, groundwater, and flows into rivers from rainfall events (Blechschmidt et al., 1999; Brown et al., 2002; Evangelou and Zhang, 1995; Lindsay et al., 2015; Moncur et al., 2015). Negative pH value of as low as −3.6 has been reported in mine waters in California

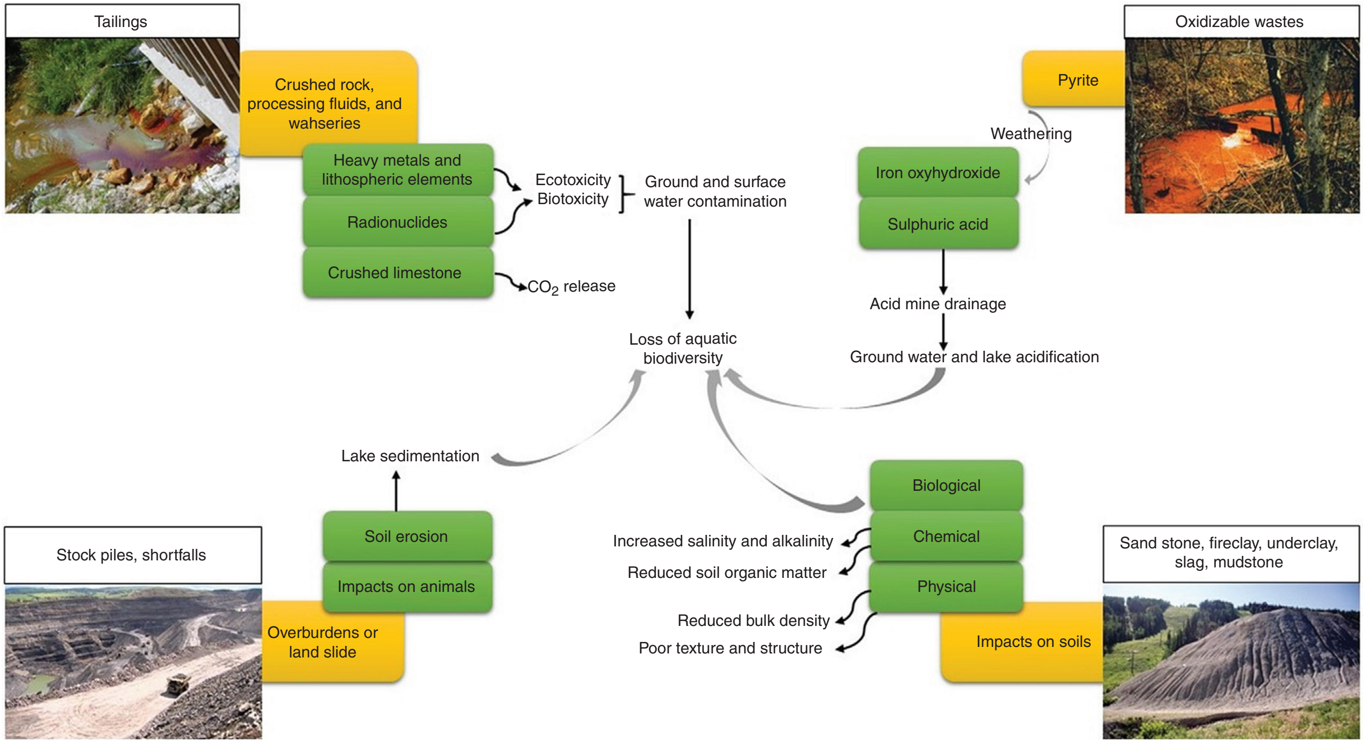

Figure 1 Schematic diagram illustrating the environmental impacts of various components of mine spoils.

(Nordstrom et al., 2000). Further, an extremely low pH of 1 has been reported in groundwater contaminated by acid mine drainage from overburden spoil piles at an open-pit lignite mine in Germany (Gerke et al., 1998). Heavy metal solubility increases at acidic pH, and this may lead to increased metal toxicity to plants and animals, thereby exacerbating serious phytotoxicity or health impact issues (Bolan et al., 2003; Lindsay et al., 2015).

Transfer of radioactive material and substances from uranium mining has been identified as a serious issue. Ambient radiation doses and radionuclides such as ^{238}U (^{234}Th), ^{226}Ra, ^{210}Pb associated with mine tailings from uranium mining sites or their milling facilities have been reported and identified as risks to health and the environment (Carvalho et al., 2007). Enhanced plant uptake of lithophile elements such as Ce, Cr, La, Lu, Rb, Sc, Th, U, V, Y, and Yb has been reported due to mine spoils from uranium mining in South Australia (Lottermoser and Ashley, 2006).

Soil stockpiling at mining areas induces anaerobic conditions in subsurface soils of up to 1-m depth, which prohibits the existence of earthworms (Boyer et al., 2011). Moreover, ground and surface waters can be polluted by enhanced transportation of metals through sediments and acid drainage. If mine spoils are not rehabilitated, harsh conditions with infertile soils result. In 1966, due to poor mine spoil rehabilitation, the colliery spoil tip in the village of Aberfan, South Wales (United Kingdom), collapsed, killing 144 people, 116 of them children (Johnes, 2000). In Spain, the mine tailing accident in Aznalcollar is the largest environmental accident (Grimalt et al., 1999). In this Spanish disaster, approximately 4 million m^3 of acidic water and 2 million m^3 of heavy metal rich mud were released into the Agrio and Guadiamar rivers. The sudden flooding killed vegetation along the banks of the rivers. The mud accumulated at a distance of 40 km along the rivers and affected freshwater fauna including fish, shellfish, and crabs. In addition to aquatic organisms, terrestrial organisms were killed by this sudden release of spoils into the environment. These examples show that mine spoils need to be rehabilitated to restore ecological health to the land.

Depending on the level of disturbance, pollution, and cost, different methods are used to rehabilitate mine spoils. For instance, revegetation of sites containing heavy metal rich tailings can be achieved by using various approaches, such as: immobilization of heavy metals through addition of organic or inorganic amendments (ie, sewage sludge, compost, fly ash, or biochar); application of a cap or liner over the toxic materials; and initiation of growth of vegetation by using uncontaminated soils and heavy metal tolerant genotypes (Lamb et al., 2013; O'Reilly, 1997). Storing mine tailings in isolated impoundments under water or behind dams has been used to avoid contact of mine tailings with the atmosphere (Kossoff et al., 2014). However, storing tailings in isolated impoundments has been reported to fail frequently causing a tremendous risk to the surrounding environment. Recently, nanoenhanced materials such as iron oxides, phosphate-based nanoparticles, iron sulfide nanoparticles, and carbon nanotubes have been used as amendments to reclaim mine sites. They improve mine spoil quality, enhance usage efficiency of amendments, mitigate soil contamination, and control soil erosion (Liu and Lal, 2012). In addition, forest floor and peat materials have been used as surface organic amendments for mine site rehabilitation (Beasse, 2012).

Large quantities of biowastes such as biosolids, municipal solid waste (MSW), animal and poultry manure, papermill sludge, and plant residues are generated as the consequences of human population increment and, subsequently, expansion of livestock or poultry industries (Table 1). Large amounts of biowaste must be treated or utilized following environmental regulations for their safe disposal onto agricultural or disturbed lands such as mine spoils and landfills. However, biowastes are a good source of OM and essential nutrients in soils and, therefore, they have been extensively used in agriculture. For example, biosolids are an excellent source of OM (Torri et al., 2014). Biosolids are nutrient rich because they contain N, P, and K in many instances (Kajitvichyanukul et al., 2008; Lamb et al., 2012; Sopper, 1992). Biowastes play a crucial role in enhancing the physical, chemical, and biological properties of degraded lands, thereby improving soil health. Therefore, biowastes are extensively used as an ideal soil amendment for land reclamation and revegetation (Tian et al., 2006). Larney and Angers (2012) summarize the ecosystem services that addition of biowastes provide in restoring ecosystems, as follows: provisioning services (eg, food and energy production); regulating services (eg, nutrient cycling and carbon sequestration); supporting services (eg, water purification and pest and disease control); and cultural services (eg, recreational experiences and scientific discovery). However, concerns about soil contamination from these organic wastes, when used as amendments for remediation and revegetation of contaminated sites, have arisen. For instance, mobilization of the leachable heavy metals (ie, Ni with relatively poor retention in sewage sludge and Pb complexes with water-soluble OM that are leached by cattle manure) has been reported at some mine spoil rehabilitation sites due to biowaste application (Andres and Francisco, 2008; Schwab et al., 2007). Therefore, a number of countries and organizations around the world have developed

Table 1 Potential quantity of biowastes produced in selected countries and their carbon, nutrient (N and P), and heavy metal (Cu and Zn) contents.

Country	Type of biowastes	Potential quantity produced ($\times10^3$) (tons/year)[a]	Potential carbon, nutrient, and heavy metal contents ($\times10^3$) (tons/year)				
			C[b]	N[c]	P[d]	Cu[e]	Zn[f]
Australia	Biosolids	415	166	12	9	0.21	0.42
	Animal and poultry manures	34,514	10,194	796	126	30	78
	Municipal solid waste (MSW)	10,436	2,400	417	42	1.04	3.65
	Plant residues	163,164	69,522	780	325	—	—
United States	Biosolids	5,865	2,346	176	130	2.93	5.86
	Animal and poultry manures	111,856	32,646	2,769	823	95.64	252.90
	MSW	140,757	32,374	5630	563	14.07	49.27
	Plant residues	667,681	275,567	3,976	1,997	—	—
United Kingdom	Biosolids	1,170	468	35	26	0.58	1.17
	Animal and poultry manures	15,987	4,656	389	95	13.67	36.15
	MSW	28,071	6,456	1,123	112	2.81	9.82
	Plant residues	27,545	11,858	134	54	—	—
New Zealand	Biosolids	81	32	2	2	0.04	0.08
	Animal and poultry manures	13,694	4,042	316	47	11.70	30.96
	MSW	1,944	447	78	8	0.19	0.68
	Plant residues	1,360	591	8	3	—	—

China	Biosolids	24,848	9,939	745	547	12.42	24.85
	Animal and poultry manures	225,087	68,827	6,366	2,040	192.44	508.92
	MSW	596,342	137,159	23,854	2,385	59.63	208.72
	Plant residues	1,571,622	504,467	9,918	9,039	—	—
Bangladesh	Biosolids	3,083	1,234	93	68	1.54	3.08
	Animal and poultry manures	21,684	6,335	494	106	18.54	49.03
	MSW	74,003	17,021	2,960	296	7.4	25.90
	Plant residues	361,966	90,651	2,416	2,867	—	—
Japan	Biosolids	2,316	927	70	51	1.16	2.32
	Animal and poultry manures	8,369	2,393	230	98	7.16	18.92
	MSW	55,591	12,786	2,224	223	5.56	19.46
	Plant residues	52,073	13,347	344	402	—	—
South Africa	Biosolids	881	353	26	19	0.44	0.88
	Animal and poultry manures	17,330	5,047	407	90	14.12	39.18
	MSW	21,149	4,864	846	85	2.12	7.40
	Plant residues	43,451	18,518	274	120	—	—
India	Biosolids	22,843	9,137	685	503	11.42	22.84
	Animal and poultry manures	194,857	57,953	4,337	639	166.60	440.57
	MSW	548,243	126,096	21,930	2,193	54.82	191.89
	Plant residues	1,632,904	462,575	10,267	11122	—	—

(Continued)

Table 1 Potential quantity of biowastes produced in selected countries and their carbon, nutrient (N and P), and heavy metal (Cu and Zn) contents.—cont'd.

Country	Type of biowastes	Potential quantity produced ($\times 10^3$) (tons/year)[a]	Potential carbon, nutrient, and heavy metal contents ($\times 10^3$) (tons/year)				
			C[b]	N[c]	P[d]	Cu[e]	Zn[f]
Germany	Biosolids	1,476	590	44	33	0.74	1.48
	Animal and poultry manures	16,679	5,219	423	78	14.26	37.71
	MSW	35,414	8,145	1,417	142	3.54	12.39
	Plant residues	60,543	25,184	319	114	—	—

	Biosolids	Animal and poultry manures	MSW	Plant residues
[a]Biosolids and MSW (g/person per day) or animal and poultry manures (kg/animals per year) or plant residues (tons/plants per ha)	50	1,170	1,200	75
[b]Biosolids and MSW C(%) or animal and poultry manures C (kg/animals per year) or plant residues C (tons/plants per ha)	40	360	23	29
[c]Biosolids and MSW N (%) or animal and poultry manures N (kg/animals per year) or plant residues N (tons/plants per ha)	3	28	4	0.5
[d]Biosolids and MSW P (%) animal and poultry manures P (kg/animals per year) or plant residues P (tons/plants per ha)	2.23	4	0.4	0.3
[e]Cu (mg/kg)	500	855	100	—
[f]Zn (mg/kg)	1,000	2,261	350	—

Note: Except Bangladesh, total animal and poultry manures generation is calculated based on chicken, cattle, pig, and sheep manures. For Bangladesh chicken, cattle, and sheep have been considered as the total animal and poultry manures generation; total plant residue generation is calculated for Australia, United States, China, and South Africa based on barley, maize, rice, rye, and wheat residues. Barley, rye, and wheat; barley, maize, and wheat; barley, maize, rice, and wheat; barley, maize, rye, and wheat for United Kingdom, New Zealand, Bangladesh, India, Japan, and Germany, respectively

Adapted from Thangarajan et al. (2013).

regulations for the application of biowastes on agricultural lands or its use in land rehabilitation. Most of these regulations have been focused on avoidance or minimization of heavy metal contamination. However, there has been no comprehensive review recently concerning the governing regulations for sludge application and the effects of biowaste addition on rehabilitation of mine spoils.

Therefore, this review provides a general overview of the sources of biowastes; current regulations for their utilization; a description of the physical, chemical, and biological effects of biowaste addition on mine spoils by presenting several case studies; and an elaboration of the role that biowastes have for rehabilitating mine spoils. Finally, future research needs and strategies are identified for sustainable biowaste utilization in mine spoil rehabilitation.

2. SOURCES OF BIOWASTE

Large quantities of organic biowastes, including MSW, biosolids, animal and poultry manure, and papermill sludge, are produced (Table 1). Municipalities generate two major sources of organic wastes that can be used as soil amendments. These organically based by-products include MSW and biosolids (often referred to as sewage sludge). Only some of the MSW is eventually recycled. Global MSW production rate was 1.3 billion tons in 1994 and it rose by 31% to 1.7 billion tons in 2008 (Foo and Hameed, 2009). It is estimated that this number will increase to 2.2 billion tons by 2025, when there will be a rate of 1.42 kg/person per day (Hoornweg and Bhada-Tata, 2012). The per capita generation of MSW is higher in wealthier countries of the world. For instance in the USA, due to prosperity and urbanization, MSW generated is approximately 251 million tons/year with a rate of 1.98 kg/person per day (USEPA, 2014). However, irrespective of the per capita generation, China and India, two developing countries, have become two of the top 10 MSW generating countries due to the size of their urban populations (Hoornweg and Bhada-Tata, 2012).

Mainly, MSW is derived from the disposal of general waste streams that include green waste, food wastes, and miscellaneous products (ie, leather, textile, metal scraps), which can be separated as noncompostable materials (Hargreaves et al., 2008; Menikpura and Basnayake, 2009). Municipal authorities are responsible for MSW management and these authorities typically face a number of challenges including safe disposal of organic wastes, which is a challenge because the landfill area available for disposal of organic wastes has declined over the last few decades (Sims and Pierzynski, 2000). Due to its hazardous nature, landfill leachate can aggravate the complexity of MSW management on most occasions (Mor et al., 2006; Wijesekara et al., 2014). Significant fluxes of greenhouse gas emission from landfills has also been reported (Chiemchaisri and Visvanathan, 2008). Even though most MSW ends up in unsightly landfill sites, a significant quantity of it has been used to produce different value-added products such as compost, feedstuffs, and biogas (Garcia et al., 2005; Vithanage et al., 2014). Municipal Solid Waste Compost (MSWC) is increasingly being used in agriculture by a number of countries as a soil conditioner and as a fertilizer (Hargreaves et al., 2008). The MSWC has a high content of OM, which is a good soil conditioner. The high content of nitrogen and phosphorus in MSWC increases its nutritional value (Table 1) (Garcia-Gil et al., 2000; Hargreaves et al., 2008). However, elevated levels of heavy metals and a high salt concentration in MSWC negatively affect soil health (Garcia-Gil et al., 2000; Hargreaves et al., 2008).

An increased number of high capacity wastewater treatment facilities generate large quantities of biosolids. Global production of biosolids is estimated as 10×10^7 tons/year (Thangarajan et al., 2013). The amount of biosolids produced in the United States and Australia have been estimated at 5645×10^3 tons/year and 407×10^3 tons/year, respectively (Thangarajan et al., 2013). Composition of biosolids differs due to many factors, such as the process of generation, age of biosolids, and environmental conditions (Wang et al., 2008). Alkaline materials, such as lime, kiln dust, Portland cement, and fly ash, are used by many biosolids producers to reduce pathogen content and to immobilize heavy metals (Kajitvichyanukul et al., 2008; Pichtel and Hayes, 1990; Wang et al., 2008). Biosolids typically contain 40–70% OM with organic carbon content ranging from 20–50% (Torri et al., 2014). Usually, biosolids contain around 3.2% N with low nitrate-N (ie, less than 0.05%) content; most of the N is present as organic-N in complex molecular forms such as proteins, nucleic acids, amines, and other cellular materials (Kajitvichyanukul et al., 2008; Wang et al., 2008). Biosolids contain around 2.3 and 0.3% for P and K, respectively, which add nutritional value (Kajitvichyanukul et al., 2008). Some heavy metals in biosolids are present in high concentrations. For instance, Zn, Cu, Ni, and Pb have been observed in significant concentrations in biosolids (Table 1). Even though there are several end uses of biosolids in agriculture, for example, landscaping, forestry, and land rehabilitation, large quantities of biosolids end up in landfills and stockpiles due to public opposition for their use on land. Potential health risks associated with the pathogens and contaminants in biosolids, plus attraction of disease causing vectors, are some of the major factors for local public opposition (Wang et al., 2008). However, production of quality controlled biosolids or stabilization methods for land application can be used to avoid these issues related to contaminants, pathogens, and vectors. For example, biosolids are often alkaline stabilized, which is the most common method for generation of Class A biosolids. The stabilization minimizes danger from pathogens and contaminants (Edwards and Someshwar, 2000; McBride, 1998). Alkaline stabilization of biosolids not only helps to eliminate pathogens but also immobilizes heavy metals, thereby decreasing their bioavailability. It also enhances carbon stabilization (Palumbo et al., 2004). Cocomposting of biosolids with inorganic waste products is increasingly being developed to suit the end-use.

With the continuous decline in the availability of land area for crop production, the increase in food demand for human consumption is likely to be met mainly through intensive poultry and livestock production (Thangarajan et al., 2013). As a result of this, a large volume of manure by-products from these industries are produced and have to be dealt with to abide by environmental regulations, including safe disposal onto land (Power et al., 2000). Confined animal production, including beef and dairy cattle, poultry, and swine, are the major source of manure by-products in most countries. For example, the USA produced over 8.6 billion broilers in 2010 (NCC, 2015), resulting in ~23 billion kg of poultry waste (Nachman et al., 2005). Even though, this huge content of manure is being utilized in environmentally benign practices such as biogas generation and agriculture in the USA, the United States Environmental Protection Agency (USEPA) has estimated methane emissions from livestock manure management at 3 million tons in the year 1997, which accounted for 10% of total methane emissions of the USA in 1997 (USEPA, 1999). Further, the USEPA expects the methane emission resulting from livestock manure to grow by over 25% from 3.2 to 4.6 million tons during 2000–2020. In Australia, it is estimated that about 34 million tons of livestock manure are collected from farm buildings and yards (Thangarajan et al., 2013).

Manure is considered as a by-product that contains almost all the essential elements necessary for plants. The nutritional value of livestock and poultry manures for crop and pasture growth is well understood (Bolan et al., 2010). In most cases, livestock and poultry manures are directly used as a soil conditioner and fertilizer (Petersen et al., 2007). In spite of the direct use of manure, composting has been performed to reduce the quantity of animal manure needing disposal, to stabilize the organic material, and to eliminate or reduce the risk of spreading of pathogens, parasites, and weed seeds (Bernal et al., 2009). High-quality compost should be produced to overcome the cost of composting (Bernal et al., 2009). Here, high-quality compost refers to compost produced from animal manures that have added agricultural value specifically on the contents of nutrient, OM humification, and their degree of maturity. The usefulness and properties of animal manure depend on factors such as animal type, animal attributes (age, size), water use, manure collection (floor type), season, bedding type, and storage and handling of manure (Miller et al., 2003). Safe disposal practices of manure by-products should be followed to avoid negative impacts on the environment, such as soil and water pollution and odor and gaseous emissions (Bernal et al., 2009). Metal contamination in manure occurs in many ways. Feed additives that are rich in metals serve as a direct source (Bolan et al., 2004). Soil ingestion by the livestock (ie, soil ingestion is an important source of Cd contamination by grazing sheep and cattle) and metal contamination during manure collection (ie, usage of shovel made of crushed coal ash results in higher concentration of Cu and Zn) contribute indirectly to metal accumulation in manure (Bolan et al., 2004; Loganathan et al., 1999; Sweeten et al., 1995). A number of feed additives such as As, Co, Cu, Fe, Mn, and Se are used in intensive livestock management to prevent or reduce diseases, improve weight gains, and improve egg production in poultry (Bolan et al., 2004; Jackson and Bertsch, 2001). After metabolism, a major portion of these metals enter into the livestock and poultry manure, and they become environmental contaminants. Based on As concentrations in poultry waste, it has been estimated that this waste contributes to between 250 and 350 tons of As annually in the USA (Nachman et al., 2005). Chemical properties, such as the complexation form, speciation based on environmental conditions, and partitioning of individual metals present in manure, are important in terms of their distribution as the contaminants. For example, more than 90% of As in poultry manure have been observed in the water-soluble form, which indicates possible contamination through leaching or runoff (Jackson and Bertsch, 2001). Various approaches have been used to overcome environmental problems associated with contaminants in animal manure. These include production of an animal diet with reduced metal contents and developing alternative growth promoters that are low in metal(loid)s, both of which should reduce metal loadings to soils (Bolan et al., 2004).

Papermill and pulp industries produce enormous quantities of paper and pulp products each year. Total production of paper and boards in 18 European countries belonging to the Confederation of European Paper Industries (CEPI) is around 91 million tons in 2014 (CEPI, 2014). About 48% of these produced paper and boards are used for packaging, whereas about 32, 8, and 7.6% are coated or uncoated graphics, newsprint, and hygienic paper, respectively (CEPI, 2014). Around 160 pulp mills produced over 38 million tons of pulp in 2012, which has been used mainly for mechanical and semichemical pulp (ie, sulfite and sulfate pulp), chemical pulp, and woodpulp for paper-making industries (CEPI, 2012). The production of paper and pulp generates tons of wastes, including organic solid wastes and sludge (Bajpai, 2015). In 2005, 11 million tons of waste, which is 11% of the total production of paper, was generated in paper industries under the CEPI (Marko and Polonca, 2012). Apart from Europe, the USA alone produced about 4.1 and 5.6 million tons of dry weight of sludge by paper and pulp mills in years ~1990 and 1995, respectively (Park et al., 2011; Scott et al., 1995). Meanwhile, Canada produced 1.5 million tons of sludge in the same way in 2003 (Marche et al., 2003).

Sludge rich with OM is generated in high content in the paper and pulp industries. Determination of sludge composition based on paper and pulp is problematic because it depends on several factors such as process, grade of the paper product manufactured, and method used in production of fibers in the pulp manufactured (Monte et al., 2009). About 60–94% organic content is available in papermill sludge, which shows its potential for use as a soil amendment on disturbed lands (Marko and Polonca, 2012). Although paper and pulp mill sludge are rich in OM, they contain less N and P than biosolids and composts (Park et al., 2011). Hence, papermill sludge often needs additional nutrient input to be used in mine spoil rehabilitation due to its low N and P levels (Park et al., 2011). Papermill sludge is frequently incorporated with other amendments, such as waste wood fibers, fly ash, waste lime, and kaolin to enhance performance as a soil amendment (Li and Daniels, 1997). It is the sixth largest polluting industry after the oil, cement, leather, textile, and steel industries, and many environmental contaminants are associated with discharge of paper and pulp mill sludge (Ali and Sreekrishnan, 2001). It contains polychlorinated dibenzo-*p*-dioxins, dibenzofurans, chlorinated lignins, resin acids, phenols, and furans (Ali and Sreekrishnan, 2001; Kuehl et al., 1987). Some of these contaminants, including dioxins and furans, are known as persistent organic pollutants

(POPs) due to their resistant nature for degradation in the environment. Paper and pulp mill sludge is managed by different methods, such as landfilling and use as landfill-capping materials, in landspreading, composting, land reclamation, and in utilization for brick, light aggregate, and cement production (Marko and Polonca, 2012). Due to legislation and increased taxes, incineration with energy recovery is becoming the main sludge treatment method in European paper mill industries (Monte et al., 2009).

A range of other organic waste products, such as digestates and yard and wood wastes, are generated in significant quantities, and they have been used as soil amendments in mine spoil rehabilitation (Allen et al., 2007; Nada et al., 2012). Information about the quantity of digestates is rare, which may be due to the fact that they are produced in specific locations. These organic wastes are derived from the biological treatment or anaerobic digestion of organic wastes such as biosolids, MSW, animal manure, or some other industrial wastes (ie, sugar and ethanol production, wastewater from food and agroallied industries). Anaerobic digestion of organic waste has been practiced to reduce waste amounts and to fulfil requirements for land disposal; to destroy pathogen and vectors; and to produce methane for energy generation. But this digestion process ends up with sludge as a by-product known as the digestates (Abdullahi et al., 2008; Smith et al., 2005). These digestates are often rich in OM as well as nutrient content, including N and P (Moller and Muller, 2012). Although in pot experiments increased benefits for N availability to plants by digestate application, compared to untreated animal manure, have been reported, only small or inconsistent benefits have been reported under field conditions, perhaps due to increased gaseous N loss under the field conditions (Moller and Muller, 2012). Phytotoxicity, viscosity, odor, and difficulty of handling when applying to soils have to be considered when using digestates on land (Abdullahi et al., 2008).

Yard wastes include plant leaves, shrub or tree trimmings and lawn clippings. Municipalities collect yard waste and produce compost or wood chips (ie, to produce fuel, mulch, or compost). These practices are cost effective and easy to manage (Richard et al., 1990). Wood wastes are mainly derived from wood processing facilities (WPF) that include tree chips and bark chips. These wood waste materials have been found to vary greatly in size, composition, and relative decomposition rates (Allen et al., 2007). Although wood wastes serve as a significant and beneficial source for an OM amendment or mulching material, they are being increasingly utilized as fuel in industrial boilers, and, therefore, they are not as readily available for other purposes (Venner et al., 2011).

3. REGULATIONS OF BIOWASTE UTILIZATION

Land application of biowastes has been practiced by many countries for centuries. Biowastes, in particular biosolids, manure, and MSW composts, have received much attention with respect to contaminants, which have negative effects on soil microorganisms and cause phytotoxicity and toxicity to animals. Among these biowastes, biosolids play an important role in land application because of their elevated production with increased numbers and capacities of wastewater treatment plants. Therefore, most regulations have focused on biosolids utilization. Even though the history of guidelines for sludge application to land is scanty, the first guidelines for utilization of sludge to soils were formulated around 1970 (Tjell, 1986). They recognized that mobile metals in sludge resulted in phytotoxicity, and were formulated to avoid metal toxicity to plants (Tjell, 1986). Most of the early guidelines for sludge utilization focused on avoiding soil contamination by metals on agricultural land due to long-term utilization of sewage sludge as a fertilizing material or soil conditioner. At present, a number of countries and organizations around the world have developed regulations for application of biowastes to agricultural land to enhance soil fertility or to disturbed land for rehabilitation. Most of these regulations focus on avoidance or minimization of heavy metal contamination in soils (USEPA, 1994a). Some of the regulations are for pathogens present in waste sludge (NWQMS, 2004). Few regulations consider emerging contaminants and reactive nanomaterials associated with sewage sludge (Jones-Lepp and Stevens, 2007). Land application of manure is traditionally based on N and/or P loadings without considering the metal contents in manure (Bolan et al., 2004). Metal accumulation in soils is due to repeated application of manure and/or biosolids. Table 2 shows the regulatory authorities for the application of biowastes to land, nature of the regulations, and the constituents included in these regulations.

3.1 Regulations in the USA

The USA regulates land application of sludge at the federal, state, and local level. The National Environmental Policy Act of 1969 regulated municipal wastewater sludge utilization and disposal systems to protect the environment (USEPA, 1979). The USEPA is the main federal regulative authority for promulgating and enforcing regulations on land application of sludge

Table 2 Authorities regulating application of biowastes to land, nature of the regulations, and the constituents of the biowastes included in the regulations.

Authority or country	Nature of regulations	Biowastes type	Biowastes constituents included	References
United States Environmental Protection Agency (USEPA)	Organic amendments for landfill and mine soil rehabilitation by improving soil properties (chemical, physical, and biological)	Recycled municipal sewage sludge	Cd, Cu, Ni, Pb, Zn	USEPA (1994a)
Department of Environment and Conservation (DEC), NSW, Australia	Recycled organics in mine site rehabilitation	Biosolids, flyash, mulch, compost	Zn, Pb, Cd, N, P	DEC (2006)
USEPA	Surface disposal of biosolids on mine site for improvement of plant growth	Biosolids, sewage sludge from municipal waste materials	Zn, Pb, Cd, N, P	USEPA (1994b)
Concentrated Animal Feeding Operation (CAFO), USA	Application of manure and wastewater on soils	Manures (cattle, turkey, chicken)	N, P, K, Cd, As	CAFO (2005)
Surface Mining Control and Reclamation Act (SMCRA), USA	Application of coal combustion products on soils	Surface mined coal, coal mined underground, lignite	N, P, TOC, Cd, Pb, Zn	SMCRA (2008)
USEPA	Application of biosolids on mine site for soil quality management	Two types of municipal biosolids, Blue plains lime-stabilized biosolids and processed biosolids from the Passaic Valley Sewerage Commissioners	TOC, ON, Zn, Cd, Pb	USEPA (1995)

USEPA	Application of N and P containing organics on mine soil	Biosolids, manure, compost, wood waste, lime, wood ash, coal combustion products, gypsum	Cd, Pb, Zn, Cu, N, P, K, Ni, As	USEPA (2007)
Department of Environment and Climate Change NSW, Australia	All recycled organics amendments with the exception of mulch were broadcast spread and ripped into the soil	Biosolids, mulch, municipal solid waste compost (MSWC), fertilizer, soil conditioner from compostable organic materials (garden organics, food organics, and agricultural organics)	N, P, K, Pb, Cd, Ca, Cu, Fe, Zn, As, Ni	DECC (2008)
Division of Agricultural Sciences and Natural Resources, Oklahoma State University, USA	Application of organic amendments on mine site	Sewage sludge from wastewater treatment plants with partially decomposed organic matter and significant amounts of plant nutrients	N, P, K, Cd, Zn, Pb, Cu, Ni	DASNR (1995)
USEPA	Land application of organic amendments on mined area for revegetation	Pathogen reduced sewage sludge, biosolids, municipal waste products	Cd, Zn, Ni, Cu, Pb, As, N, P, K	USEPA (2003)
Natural Water Quality Management Strategy (NWQMS), Canberra, Australia	Application of organics for mine site management, landfill surface rehabilitation, and environmental protection practice	Stabilized biosolids for two key components, reduction in pathogen loads, and controls to avoid attraction of vectors	Cd, Pb, Zn, Cu, N, P, K	NWQMS (2004)

(*Continued*)

Table 2 Authorities regulating application of biowastes to land, nature of the regulations, and the constituents of the biowastes included in the regulations.—cont'd.

Authority or country	Nature of regulations	Biowastes type	Biowastes constituents included	References
Environmental Protection Agency (EPA), South Australia	Application of organic amendments for mine site rehabilitation and landfill management	Sludge for landfill site management includes spreadable solid waste and biosolids for mine site management	Cd, Zn, N, P, K, Cu, Pb	SAEPA (2009)
Department of Primary Industries, Water and Environment: Tasmania, Australia	Application of compost organic materials for mine site rehabilitation and revegetation	Sewage sludge/biosolids compost for maintaining C: N ratio, pathogen control, pH, moisture control, temperature, particle size	Cd, Zn, Ni, Cu, Hg, Pb	Dettrick and McPhee (1999)
Canadian Council of Ministers of the Environment (CCME), Canada	Application of municipal organics for mine site rehabilitation by improving vegetation	Municipal biosolids serve as a source of nutrients and organic matter to promote soil development and establishment of vegetation on degraded sites	Cd, Pb, Hg, N, P, K, Zn	CCME (2012)
European Commission (EC), DG Environment under Study Contract DG ENV	Organic amendments for forest, landfill site, and mine soil rehabilitation	Sewage sludge for recycling of plant nutrients like N, P, K, and effective replacement of chemical fertilizers	Cd, Pb, Zn, Hg, Ni, Cu, N, P, K	EC (2008)
Western Australian guidelines for biosolids management (WAGBM)	Application of organics for landcare and mine site rehabilitation program	Biosolids used as an organic humus and as a fertilizer substitute	Cd, Pb, Zn, Ni, Cu, N, P, K	WAGBM (2012)

The Pennsylvania Department of Environmental Resources (PDER), USA	Municipal sludge application for mined land reclamation	Sewage sludge for revegetation of inactive mines or active coal refuse piles	Cd, Cu, Cr, Pb, Hg, Ni, Zn	PDER (1977)
NSW EPA, Environmental guidelines, use, and disposal of biosolid products, Australia	Application of biosolids on washery reject materials, tailing dams or spoil materials from metalliferous mines or other land rehabilitation projects where sites have been disturbed or degraded	Two types of biosolids, liquid-defined as any biosolids which have the capacity to flow and be conveyed via pump, and solid-defined as nonflowable	Cd, Zn, Pb, Hg, Cu, Ni, N, P, K	Ang and Sparkes (1997)

(USEPA, 1983). The EPA is responsible for regulating sludge management defined by the Resource Conservation and Recovery Act of 1976 (RCRA) and the Clean Water Act of 1977 (CWA). In 1979, the USEPA highlighted the utilization of sludge based on three end-use groups (USEPA, 1979): as a soil amendment, as a source of heat, and as a source of other useful products. Sludge has been used for reclamation of disturbed lands such as strip-mined lands and gravel pits. The importance of sludge to stabilize bankspoils and moving sand dunes, and as a nutrient supplier to cover crops for reclaimed lands, was emphasized in the USEPA's 1979 guidelines (USEPA, 1979). The USEPA guidelines in 1983 identified four options for safe disposal of sludge: agricultural utilization, land reclamation, forest utilization, and dedicated land utilization (USEPA, 1983). Strip mine lands, mine tailings, and other disturbed or marginal lands have been considered for the purpose of land revegetation and reclamation (USEPA, 1983). The EPA has indicated that these sludge disposal options are not mutually exclusive. For instance, dedicated land disposal sites for sludge can be used for agriculture. However, production of agricultural crops and forests, or improvement of soil characteristics, are considered of secondary importance for sludge utilization, compared to use on dedicated lands, due to the higher annual application rates of lower quality sludge that can be placed on dedicated land, which are not possible with the other three options. The use of sludge on dedicated lands leads to the risk of phytotoxicity with excess metals and salts (Qasim, 1998; USEPA, 1983). Because at dedicated land disposal sites transportation of pollutants elsewhere through erosion, leaching, and volatilization occurs, it has been suggested that physical removal of contaminants in soils by plants (ie, phytoremediation) be used, and this allows the conversion of dedicated land disposal sites into potential lands for agriculture (Mclaughlin et al., 2000). The USEPA's 1983 guidelines give typical requirements for sludge that can be used for land rehabilitation, as follows (USEPA, 1983):

- Apply sludge in a one-time application (ie, sludge is not applied again to the same land area in the future). However, if sludge is used again in a postsludge application, guidelines related to the Federal Surface Mining Control and Reclamation Act of 1977, as well as other federal and state guidelines of land application of sludge, should be considered. Usually, if additional applications occur they are likely to be at intervals of 5–10 years apart. This one-time application reduces potential harmful impacts on vegetation (phytotoxicity).
- Reduce or eliminate commercial fertilizer use.

- Allow soil to support vegetation to retard erosion.
- Alleviate existing degradation on disturbed land by pertinent sludge application and management.

The USEPA's part 503 regulation provides comprehensive requirements for 10 heavy metals (As, Cd, Cr, Cu, Pb, Hg, Mo, Ni, Se, Zn) and pathogen or vector attraction reduction requirements for the utilization of biosolids on land including sites to be reclaimed (USEPA, 1994a). Pollutant concentration limits in the sludge have been described for these 10 metals with their respective ceiling concentration limits in the soil. If a limit for any metal is exceeded, the biosolids cannot be applied to land until such time that the ceiling limits are no longer exceeded. However, if metal concentrations in biosolids are above the concentration limits but below the ceiling limits, biosolids can be land applied until the cumulative metal loadings allowed are reached. However, the USEPA's part 503 standards do not describe the effect on each metal of soil type, plant species, bioaccumulation potential, and retention (ie, effect of precipitation) in soils (Ritter, 2000). The USEPA part 503 regulations give limited information on sludge application on highly acidic ($pH_c < 5.0$) soils, and, therefore, using these guidelines for acidic soils is not recommended (Whatmuff, 2002). The United States National Research Council (NRC) has independently reviewed the USEPA's part 503 regulations and recommends that occurrence of pharmaceuticals and personal care products (PPCPs) in biosolids should be investigated (Jones-Lepp and Stevens, 2007).

In addition to the EPA's regulations at the federal level, the Office of Surface Mining Reclamation and Enforcement (OSMRE) has been established under the Federal Surface Mining Control and Reclamation Act of 1977 (OSMRE, 2015). The OSMRE has strict regulations concerning revegetation for rehabilitation of disturbed lands from mining (Sopper and Kerr, 1980a). Although there are recommended performance standards for meeting the revegetation requirements under this Act, no information or guidelines are given for the use of biowastes in mine spoils reclamation.

The Pennsylvania Department of Environmental Resources (PDER) is an example of a state level body that gives guidelines specifically related to sludge utilization for mine spoil rehabilitation. They were published in 1977 with the title, "Interim Guidelines for Sewage Sludge Use for Land Reclamation" (PDER, 1977). The PDER issued these specific guidelines because by that time around 97,000 ha had been disturbed from uncontrolled mining or inadequately rehabilitated land due to strip mining of coal in Pennsylvania (Sopper and Kerr, 1980a). The PDER's guidelines and requirements give information for sludge application rates. For instance, the PDER recommends a maximum sludge application rate of 134 dry tons/ha should be used to provide nitrogen to degraded lands during the reclamation process. Trace elements loading rates to soil were recommended separately for land reclamation and for farming (Sopper et al., 1980b). The PDER guidelines include soil pH adjustment, such as the utilization of lime to immobilize trace metals thereby reducing their phytotoxicity. Some of the requirements attached to the PDER guidelines emphasize proper sludge application procedures, including the use of suitable lands, climatic conditions for sludge application, and how to incorporate sludge into different types of soils. They also give environmental risk factors (ie, distance to the rivers, land slope, and dairy cattle management).

At the local governmental level (ie, municipalities), some of the deed restrictions and ordinances are more stringent than the federal level restrictions (USEPA, 1979). For example, when sludge is used on land as a fertilizer for food crop cultivation, some of the local rules require sludge to be analyzed more frequently than required by the federal regulations (USEPA, 1979).

3.2 Regulations in Australia and Europe

Regulations for utilization of biowastes in Australia vary within the States and Territories. The National Water Quality Management Strategy (NWQMS) is the national approach to achieve sustainable use of Australia's and New Zealand's water resources including municipal sludge (NWQMS, 2004). In 2004, the NWQMS issued guidelines for the management of biosolids from sewage systems (NWQMS, 2004). According to these guidelines, biosolids belong to the chemical contaminant grade C2 and pathogen grade P3, when they are used for mine, quarry, or degraded land rehabilitation. Here, chemical contaminant levels for grade C2 have been based collectively from values derived from different sources. Example sources included the National Environmental Protection (Assessment of Site Contamination) Measure issued by the National Environment Protection Council (NEPC) in 1999 and Environmental Guidelines: Use and Disposal of Biosolids Products by the New South Wales Environment Protection Authority (NSWEPA) in 1997 (Ang and Sparkes, 1997). The NWQMS emphasizes that its guidelines may be considered if State or Territory guidelines are unavailable (NWQMS, 2004).

Among the Australian State's guidelines on utilization of biosolids, there are those of the New South Wales government, which has strict guidelines attached to the NSWEPA for use of biosolids for mine spoil rehabilitation. Based on usage, the NSWEPA has grouped biosolids into three groups: as unrestricted use products, restricted use one products, and restricted use two products (Ang and Sparkes, 1997). Mine spoils include the spoil materials from open-cut coal mine operations, washery reject materials, tailing dams, spoil materials from metalliferous mines, or other sites that have been disturbed or degraded due to mining (Ang and Sparkes, 1997). Maximum allowable application rates of biosolids have been introduced by the NSWEPA to avoid groundwater and surface water pollution from rehabilitated lands that are to be used for agriculture. Definitions of restricted use one and two products are needed to follow the prescribed application rates, and application rate limits do not apply to the unrestricted biosolids products unless specifically required by the owner or controlling authority of the land. The allowable biosolids application rate is established by determining the contaminant limited biosolids application rate (CLBAR) and the nitrogen limited biosolids application rate (NLBAR). The CLBAR is the biosolids application rate that does not exceed the maximum allowable concentration of contaminants in the soil. The NLBAR gives the biosolids application rate that can be applied to soils without exceeding the annual nitrogen requirements for the crop or vegetation grown on the land. The criteria for pathogen reduction and vector attraction reduction, the NSWEPA follows the widely accepted USEPA's Code of Federal Regulations Part 503 rule for land application of sewage sludge.

The regulations governing land application of biowastes including animal manure have focused mostly on efficient management of nitrogen and phosphorus. For instance, recognizing that there are unavoidable losses of nitrogen and phosphorus, the Netherlands introduced the Mineral Accounting System (MINAS) legislation to reduce mineral surpluses in soils from large scale farms (Jongbloed and Lenis, 1998). The MINAS considers only phosphate and nitrogen to achieve equilibrium in fertilization application. Farmers have to record inputs (ie, feed, animal manure) and outputs (ie, discharge animal manure) on their farms to account for nitrogen and phosphate. The MINAS prescribes a maximum allowable loss or standard losses because of the practical difficulty to balance exactly the input and output from farms. Farmers whose nitrogen and phosphate exceed the maximum are required to pay prescribed levies. Even though the MINAS is based only on nitrogen and phosphate, this kind of accounting system may well prove to be a model in safeguarding the nutrient input-out relationship, and it might be used for the application of biowastes to soils.

Notwithstanding to the USEPA approach based on risk assessment that involves analyzing risk to flora and fauna from exposure to contaminants by different pathways, most European countries have adopted the lowest observed adverse effects concentration (LOAEC) approach to set guidelines for land application of biosolids (McGrath et al., 1995). The LOAEC approach involves setting soil limits for contaminants, and they are derived from laboratory trials involving dosing soils with high bioavailability metals rather than from field trials involving land application of biosolids containing metals (CEC, 1986; NZWWA, 2003). Therefore, the recommended values from the LOAEC are low compared to the USEPA values, which are based on field trials for land application of biosolids (McGrath et al., 1995, USEPA, 1994a). Consequently, it has been reported that the LOAEC approach is inherently precautionary (NZWWA, 2003) while the USEPA values have been questioned (McBride, 1995, 2003; Schmidt, 1997).

4. EFFECTS OF BIOWASTE ADDITION ON MINE SPOILS

For decades, application of biowastes for mine spoil rehabilitation has been well understood and reported (Bendfeldt et al., 2001; Gardner et al., 2010; Sopper, 1992; Stolt et al., 2001). Many field-scale studies have demonstrated the direct and indirect benefits of biowastes for enhancing the receiving spoil environment (Larney and Angers, 2012). Direct effects of biowastes are the changes in the intrinsic properties of soils (eg, lower bulk density), whereas the indirect enhancement is by improving chemical, physical, and biological properties in the receiving spoil soils. In addition, behavioral, nutritional, or growth responses of flora and fauna and groundwater quality changes in the surrounding environment can result from application of biowastes (Sopper, 1992). Extensive work during the past four decades has been carried out on utilization of biowastes for rehabilitation of mine spoils, and it aims at reestablishing ecosystems services (Arocena et al., 2012; Brofas et al., 2000; Brown et al., 2014; Ussiri and Lal, 2005; Zanuzzi et al., 2009). Table 3 lists field and laboratory investigations on mine spoil rehabilitation in which biowastes have been used.

4.1 Physical Characteristics

The high OM of biowastes is the main cause for improvement of physical properties in mine spoil soils. Improvements of these physical properties

Table 3 Selected references on rehabilitation of mine spoils using biowastes.

Cause of land disturbance	Location	Materials used for rehabilitation (organic amendment type)	Parameters assessed	References
Metal mining	Tui mine, Te Aroha, New Zealand	Composted sewage sludge, lime, various metal tolerant grass varieties	Germination rates, dry matter yields, soil pH	Morrell et al. (1995)
Abandoned sandpit	Saint-Lambert-de-Levis, Quebec, Canada	Paper deinking sludge, mineral N and P	Water retention capacity, CEC, pH and bulk density	Fierro et al. (1999)
Mining site for mineral ores such as, sphalerite, galena, and pyrite	Cartagena–La Union Mining District, Murcia, Spain	Pig manure and sewage sludge with a blanket application of calcium carbonate from marble industry	pH, TOC, total N, total porosity, and aggregate stability of mine waste materials	Zanuzzi et al. (2009)
Mine tailing pond for metal waste	El Gorguel and El Lirio tailing ponds at Cartagena-La Union Mining District in Murcia, Spain	Marble waste and pig slurry	Total metal contain, pH, total N, total C, and CEC	Kabas et al. (2014)
Mine soil for flotation of sulfides during copper processing	Touro mine, North Western Spain	Sludge from a bleach plant, urban solid waste and material from the area around the mine (schist and natural soil), and sewage sludge	pH, EC, total C and N, and CEC	Forjan et al. (2014)
Open-cast gypsum mine	Sorbas, Almeria, South Eastern Spain	Soil forming material and municipal solid waste compost (MSWC)	TOC, total N, pH, available phosphorus, and exchangeable potassium	Castillejo and Castello (2010)

(*Continued*)

Table 3 Selected references on rehabilitation of mine spoils using biowastes.—cont'd.

Cause of land disturbance	Location	Materials used for rehabilitation (organic amendment type)	Parameters assessed	References
Open-cast lignite mine site	Meirama, North Western Spain	Compost, limestone, and NPK	pH, fertilizer type, CEC, total N, OM, exchangeable cation, and vegetation establishment	Pedrol et al. (2010)
Reclaimed coal mining site	Eastern Ohio (Harriettsville, New Athens and Barnesville), USA	NPK, cow manure, mulching with oat straw, and chiselling	Vegetation establishment, soil properties (soil organic carbon, N pool, and pH), bulk density, above ground biomass production	Shrestha et al. (2009)
Calcareous strip-mined site	Fulton county, Western Illinois, southwest of Chicago, USA	Biosolids and sewage sludge	SOM, C sequestration, pH	Tian et al. (2009)
Coal mining site	Lake Wabamun, Edmonton, Alberta, Canada	Cattle manure	Soil pH, soluble Ca and Mg, exchangeable Na and K, exchangeable Ca:Na ratio, and loss of ignition	Bateman and Chanasyk (2001)
Spoil bank clay of a surface mine	Chomutov, Northern Bohemia, Czech Republic	Compost and lignocellose papermill waste	Plant growth and mycorrhizal development	Gryndler et al. (2008)
Coal mine spoil	Yam pa coal, North-Western, Colorado, USA	Sewage sludge	Biomass production, canopy cover, total N and P concentration	Topper and Sabey (1986)
Lead/zinc mining site	Guangdong Province, People's Republic of China	Lime and pig manure	pH, EC, seed germination, and plant growth	Ye et al. (1999)

Coil mine site	Beringen, North Eastern Belgium	Steel shots (iron bearing material) and OM	Changes in arsenate concentration, pH, and plant growth	Bleeker et al. (2002)
Mineral ores such as sphalerite, galena and pyrite, pyrrhotite, and marcasite	Murcia, South Eastern Spain	Pig manure, sewage sludge and marble waste	Accumulation of OM, pH, soil microorganism growth, presence of aggregation (granular structure of soil)	Arocena et al. (2012)
Mine sites with Fe-oxyhydroxides, sulfates, and heavy metals.	Cartagena-La Union Mining District, Murcia, South Eastern Spain	Pig manure, sewage sludge, and marble waste	Plant cover and richness, soil chemical, biochemical and biological parameters	Zornoza et al. (2012)
Copper mine tailing	South-West of Kamloops, Vancouver, British Columbia, Canada	Sewage sludge and biosolids	Vegetation establishment, OM, nutrients for plant growth	Gardner et al. (2012)
Calcareous spoil from bauxite mine	Giona mountain, central Greece	Sewage sludge	Plant biomass, plant density and foliar cover, total N and P, WHC and pH	Brofas et al. (2000)
Copper mine tailing	Salt Lake City, Utah, USA	Municipal sewage sludge termed as biosolids	Plant biomass, metal concentration in the planted species	McNearny (1997)
Copper-nickel mine tailings	Sudbury, Ontario, Canada	MSWC, crushed limestone, fertilizer, and hay mulch	pH, OM, moisture retention, water soluble Cu and Ni	Bagatto and Shorthouse (2000)

(Continued)

Table 3 Selected references on rehabilitation of mine spoils using biowastes.—cont'd.

Cause of land disturbance	Location	Materials used for rehabilitation (organic amendment type)	Parameters assessed	References
Copper mine tailing	Gaspe copper smelter at Murdochville in the province of Quebec, Canada	Peat moss-shrimp waste compost, ammonia nitrate, superphosphate, and potassium muriate as NPK fertilizers, respectively	Plant (maize) biomass, Cu concentration, exchangeable and soluble Cu, pH	De Coninck and Karam (2008)
Copper mining plant	Coquimbo Region, North-Central Chile	Biosolids, goat manure, grape, and olive residues, sediments from irrigation canal	Plant yield, pH, microbial inoculation for nutrition management and C:N	Santibanez et al. (2012)
Copper mine tailing	Touro, Spain	Phytoremediation by using *Brassica juncea*, compost, and technosol	Phytoremediation techniques. Soil-covering ability, vigorous seedling growth, tolerance to metal toxicity and potential immobilization of excluded trace metals in the rhizospher	Novo et al. (2013)
Copper mining	Touro, Galicia, Spain	Compost, biochar, and phytoremediation by *B. juncea* L	Total C and N content, concentration of metals, and plant growth	Rodriguez-Vila et al. (2014)
Copper mine tailing	Sixth region of Chile	Sewage sludge	Mobility of nutrients, content of pathogens, and leaching of metals and nutrients, pH	Garrido et al. (2012)

CEC, cation exchange capacity; TOC, total organic carbon; N, nitrogen; EC, electrical conductivity; OM, organic matter; SOM, soil organic matter.

include decreasing bulk density and temperature, increasing porosity and aggregation, increasing hydraulic conductivity and WHC, maintaining soil texture, and reducing erosion and sedimentation (Sopper, 1992). Generally, the bulk density of biowastes is lower (less dense nature) than the spoil soils due to the fact that spoil soils tend to be poorly aggregated with ill-defined soil texture (Allen et al., 2007). Application of biowastes causes a reduction in bulk density in spoil soils by increasing pore space and developing soil texture. Biowastes enhance macroporosity, mesoporosity, and microporosity in soils, thereby increasing total pore space. For example, a laboratory incubation study in which green waste compost was applied to gypsum-treated bauxite sands showed an increased mesoporosity and microporosity from 13.2 to 21.3% and 16.4 to 19.3%, respectively (Jones et al., 2012). In soils, physical properties such as bulk density, porosity, infiltration, and hydraulic conductivity are interrelated. Therefore, addition of biowastes simultaneously enhances most of these physical properties. For example, an opencast mine with poor soil physical properties was rehabilitated successfully with biowastes (Nada et al., 2012). The WHC of this mine, which had tertiary and quaternary substrate contaminated coal spoil, was increased, and the bulk density and dry particle density were decreased, with increased application rate of biowastes. Table 4 lists changes of physical characteristics in mine spoil soils after application of biowastes.

4.2 Chemical Characteristics

Biowastes enhance many chemical properties in mine spoil soils, as shown in Table 5. These chemical properties include pH, EC, CEC, and content of nutrients, heavy metals, and OM. Acid spoils have extremely low pH, which is raised by application of biowastes (Nada et al., 2012; Sopper, 1992). Generally, biowastes have a neutral to alkaline pH, and lime is sometimes added with biowastes during their stabilization, which results in a significantly high pH (Amuda et al., 2008). This causes a decrease in soluble metals due to the liming effect (Feagley et al., 1994). However, even though pH may be raised initially by application of biowastes, the pH eventually may decline with time due to the spoil activity. For instance, mine spoils containing sulfur bearing minerals may be oxidized over time, which decreases pH (Sopper, 1992). Nitrification and organic acid production are also reported to decrease the soil pH slightly (Sopper, 1992). Interestingly, at a copper mine tailing site in Canada, the pH of the soil was not affected after application of biosolids because the pH of the biosolids was slightly lower

Table 4 Selected soil physical property changes after application of biowastes on mine spoils.

Site/country	Biowastes type	Site description	Evaluated physical property	Observations	References
Vancouver, Canada	Biosolids	Mine tailing	Bulk density	Decreased from 1.3 to 0.7 Mg/m^3 Decreased from 1.5 to 0.9 Mg/m^3	Gardner et al. (2010)
Virginia, USA	Hardwood sawdust	Mining soils	Bulk density Soil course fragment Soil clay content Soil sand content Soil silt content	Decreased from 1.24 to 1.12 Mg/m^3 Increased from 61 to 66% Decreased from 10 to 9% Decreased from 64 to 63% Increased from 26 to 28%	Bendfeldt et al. (2001)
Virginia, USA	Municipal sewage sludge	Mining soils	Bulk density Porosity Soil clay content Soil sand content	Decreased from 1.24 to 1.09 Mg/m^3 Increased from 53 to 59% Decreased from 10 to 9% Increased from 64 to 65%	Bendfeldt et al. (2001)
Murcia, Spain	Pig manure	Acidic tailings	Porosity Aggregate stability	Increased from 53 to 80% Increased from 19 to 30%	Zanuzzi et al. (2009)
Murcia, Spain	Sewage sludge	Acidic tailings	Porosity Aggregate stability	Increased from 53 to 82% Increased from 19 to 28%	Zanuzzi et al. (2009)
Almería, Spain	Urban municipal solid waste (MSW)	Gypsum spoil quarry	Soil clay content Soil sand content Soil silt content Available water capacity Permanent wilting point	Decreased from 27.6 to 12.3% Increased from 33.6 to 59.8% Decreased from 37.8 to 27.8% Increased from 13.7 to 17.1% Decreased from 14.1 to 11.8%	Castillejo and Castello (2010)

Alberta, Canada	Cow manure	Coal strip mined soils	Bulk density Soil clay content Soil sand content Aggregate geometric mean diameter	Decreased from 1.12 to 0.90 Mg/m^3 Decreased from 39.1 to 37.5% Increased from 22.1 to 24.2% Increased from 1.9 to 2.3 mm	Bateman and Chanasyk (2001)
Ontario, Canada	MSW	Deposited mine tailing soils	Moisture content	Increased from 8.48 to 26.25%	Bagatto and Shorthouse (2000)
Queensland, Australia	Compost produced from shredded municipal green waste	Bauxite residue sand soils	Macropores distribution Mesopores distribution Micropores distribution Field capacity	Decreased from 77.8 to 59.4% Increased from 10.0 to 21.3% Increased from 12.2 to 19.3% Increased from 102.5 to 194.7 kg/m^3	Jones et al. (2012)
Queensland, Australia	Biosolids	Bauxite residue sand soils	Macropores distribution Mesopores distribution Micropores distribution Field capacity	Decreased from 76.0 to 63.0% Increased from 11.0 to 14.0% Increased from 13.0 to 22.0% Increased from 109 to 177 kg/m^3	Jones et al. (2010)
Queensland, Australia	Green waste compost	Bauxite residue sand soils	Macropores distribution Mesopores distribution Micropores distribution Field capacity	Decreased from 76.0 to 66.0% Increased from 11.0 to 16.0% Increased from 13.0 to 17.0% Increased from 109 to 174 kg/m^3	Jones et al. (2010)

(*Continued*)

Table 4 Selected soil physical property changes after application of biowastes on mine spoils.—cont'd.

Site/country	Biowastes type	Site description	Evaluated physical property	Observations	References
Ginoa mountain, Greece	Sewage sludge	Calcareous soils from bauxite mining	Available water capacity	Increased from 2.32 to 6.87%	Brofas et al. (2000)
Queensland, Australia	Poultry manure	Bauxite residue sand soils	Bulk density Total porosity Macropores distribution Field capacity	Decreased from 1.67 to 1.58 Mg/m^3 Increased from 0.47 to 0.49 m^3/m^3 Decreased from 60 to 53% Increased from 185 to 230 kg/m^3	Jones et al. (2011)
Queensland, Australia	Green waste derived biochar	Bauxite residue sand soils	Bulk density Total porosity Macropores distribution Field capacity	Decreased from 1.65 to 1.55 Mg/m^3 Increased from 0.46 to 0.51 m^3/m^3 Decreased from 76 to 61% Increased from 109 to 199 kg/m^3	Jones et al. (2010)
Atlantic coastal plain, USA	Yard waste compost	Mineral sand tailings	Bulk density Porosity	Decreased from 1.43 to 1.38 g/cm^3 Increased from 45 to 48%	Stolt et al. (2001)

Table 5 Selected soil chemical property changes after application of biowastes on mine spoils.

Site/country	Biowastes type	Site description	Evaluated chemical property	Observations	References
Vancouver, Canada	Biosolids	Mine tailing	EC TC Total CEC	Increased from 1.44 to 3.26 dS/m Increased from 6.7 to 50.1 g/kg Increased from 0.383 to 1.317 mmol	Gardner et al. (2010)
Virginia, USA	Hardwood sawdust	Mining soils	pH CEC Total Kjeldahl N	Decreased from 6.8 to 6.1 Increased from 7 to 10.1 cmol/kg Increased from 0.5 to 1.12 mg/kg	Bendfeldt et al. (2001)
Virginia, USA	Municipal sewage sludge	Mining soils	pH CEC Total Kjeldahl N	Increased from 6.8 to 7.4 Increased from 7 to 10.1 cmol/kg Increased from 0.62 to 1.85 g/kg	Bendfeldt et al. (2001)
Murcia, Spain	Pig manure	Acidic tailings	pH TOC	Increased from 2.9 to 7.3 Increased from 0.86 to 2.46 g/kg	Zanuzzi et al. (2009)
Almería, Spain	Municipal solid waste (MSW)	Gypsum spoil quarry	TON TOC Available P Water extractable Na	Increased from 348 to 1,393 mg/kg Increased from 2,400 to 13,940 mg/kg Increased from 31,667 to 94,167 mg/kg Increased from 37 to 222 mg/kg	Castillejo and Castello (2010)

(Continued)

Table 5 Selected soil chemical property changes after application of biowastes on mine spoils.—cont'd.

Site/country	Biowastes type	Site description	Evaluated chemical property	Observations	References
Alberta, Canada	Cow manure	Coal strip mined soils	pH Soluble Mg Soluble Ca Exchangeable K	Increased from 6.1 to 6.5 Increased from 12.0 to 22.8 meq/L Increased from 30.4 to 49.3 meq/L Increased from 0.9 to 3.9 cmol (+)/kg	Bateman and Chanasyk (2001)
Oklahoma, USA	Lime-stabilized biosolids	Soil contaminated with Zn and Pb	pH	Increased from 6.4 to 7.6	Basta et al. (2001)
Murcia, Spain	Pig manure	Acidic tailings	pH EC TN TC Total S	Increased from 2.8 to 5.9 Decreased from 2.8 to 2.2 dS/m Increased from 0.1 to 0.2 g/kg Increased from 1.4 to 8.3 g/kg Decreased from 25.7 to 18.7 g/kg	Arocena et al. (2012)
Murcia, Spain	Sewage sludge	Acidic tailings	pH EC TC Total S	Increased from 2.8 to 5.8 Decreased from 2.8 to 2.3 dS/m Increased from 1.4 to 5.6 g/kg Decreased from 25.7 to 16.0 g/kg	Arocena et al. (2012)
Ginoa mountain, Greece	Sewage sludge	Calcareous soils from bauxite mining	pH CEC TN OM Olsen P	Decreased from 8.3 to 7.4 Increased from 3.62 to 13.4 cmol/kg Increased from 0.1 to 5.1 g/kg Increased from 3.0 to 63.4 g/kg Increased from 24.7 to 255.3 mg/kg	Brofas et al. (2000)

Huelva, Spain	Sewage sludge	Mine waste soil	pH EC OC	Increased from 7.14 to 7.69 Increased from 2.25 to 4.11 dS/m Increased from 1.47 to 3.10%	Mingorance et al. (2014)
Queensland, Australia	Compost from municipal green waste	Bauxite residue sand soil	pH Water extractable Mg Water extractable Ca Water extractable Na	Increased from 8.2 to 8.4 Increased from 0.04 to 2.09 mmol/kg Decreased from 40.9 to 14.6 mmol/kg Increased from 16.9 to 41.3 mmol/kg	Jones et al. (2012)
Queensland, Australia	Biosolids	Bauxite residue sand soils	pH EC Extractable P NH_4–N NO_3–N	Decreased from 9.7 to 9.1 Increased from 10.1 to 11.7 dS/m Increased from 13.0 to 508.8 mg/kg Increased from 4.6 to 423.6 mg/kg Increased from 5.8 to 1.6 mg/kg	Jones et al. (2011)
Queensland, Australia	Mushroom compost	Bauxite residue sand soils	CEC Exchangeable Ca Exchangeable K Exchangeable Na	Increased from 10.4 to 15.6 mmol/kg Increased from 96.5 to 403.9 mg/L Increased from 9 to 798 mg/L Decreased from 19.2 to 15.4%	Jones et al. (2010)

(*Continued*)

Table 5 Selected soil chemical property changes after application of biowastes on mine spoils.—cont'd.

Site/country	Biowastes type	Site description	Evaluated chemical property	Observations	References
Spain	Mixed MSW	Mine soils	pH	Increased from 4.8 to 6.7	de Varennes et al. (2010)
			Water extractable P	Increased from 6.6 to 29.6 mg/kg	
			Water extractable As	Increased from 3.4 to 14.0 mg/kg	
			Water extractable Fe	Increased from 2.8 to 5.4 mg/kg	

than that of the unamended tailings (Gardner et al., 2010). Application rate of biowastes and their depth of incorporation into the spoils are crucial factors affecting mine spoil rehabilitation. Application of biowastes on mine spoils increases soluble salts (ie, chloride, sulfate, and Na), thereby increasing the EC (Feagley et al., 1994; Wilden et al., 1999). This may result in increasing metal–inorganic complexes, which increase phytoavailability of metals during phytoremediation (Smolders and McLaughlin, 1996; Smolders et al., 1998). Maintaining biowastes at field capacity is important to prevent inhibition of plant growth by the high concentration of soluble salts (Rodgers and Anderson, 1995). However, reduction in soil EC with biowastes addition has also been reported. This is mainly due to the immobilization of metal ions by the OM fraction of biowastes and leaching of soluble salts (Clapp et al., 1986). The CEC of biowastes varies widely depending upon the presence of clay, mineral particles, and organic colloids that are the result of decomposition of the OM in the biowastes (Fierro et al., 1999; Larney and Angers, 2012). Even though addition of biowastes usually increases organic matter induced CEC of soils, this effect varies among soil types (Gardner et al., 2010).

Most biowastes contain nitrogen in organic forms (ie, protein, nucleic acids) rather than in the inorganic forms (ie, nitrate, nitrite, ammonium), which are readily available to plants but also have the tendency to leach or runoff from soils (Daniels et al., 2001; Larney and Angers, 2012; Stehouwer et al., 2006). Biosolids have a high content of nitrogen, and, as is well reported, their application to mine spoil soils increases fertility (Jones et al., 2010). Sometimes the fertility is enhanced by addition of other amendments with the biowastes. For instance, application of biosolids combined with chemical fertilizers increased available N, P, and K in open-cast mining areas in China (Li et al., 2013a). With the application of biowastes to mine spoils, not only are the main nutrients increased, but other important nutrients, such as Ca, Mg, and S and trace elements, such as B, Cu, Fe, Mo, Mn, and Zn are increased. Increase in Ca, Mg, and Fe occurred after application of compost derived from MSW to pyrite mine spoils in Sao Domingos mines in Portugal (de Varennes et al., 2010).

4.3 Biological Characteristics

Generally, application of biowastes raises the content of the soil organic matter (SOM), which is used by soil microorganisms as a readily available main energy source. Increased microbial biomass carbon (MBC) and microbial enzymatic activities can be detected following the addition of biowastes to soils. A long-term study in the USA at a copper mine tailing site that had received class A biosolids revealed an increased microbial population and sustained microbial activities such as nitrification, sulfur oxidation, and activity of dehydrogenase throughout the 10-year study period (Pepper et al., 2012). Sewage sludge combined with chemical fertilizers increased the populations of fungi, bacteria, actinomycetes, and total microorganisms along with soil urease activity in abandoned open-cast mining areas that had been heavily degraded with coal gobs in Shanxi, China (Li et al., 2005, 2013a).

Earthworms also are affected by biowastes. Application of biowastes (ie, biosolids, compost, and coal ash) resulted in a higher population density of earthworms in open-cast coal mining spoil soils compared to mineral fertilizers (Emmerling and Paulsch, 2001). Endemic earthworm survival was studied in tailings of an active open-cast coal mine located in New Zealand. Biosolids had been applied for rehabilitation, which resulted in a significant increment in earthworm mortality. The results suggested the need for ecological tests before application of biowastes (Waterhouse et al., 2014). Ecological factors such as diet of earthworms and seasonal influences on them need to be understood to evaluate the effect of a heavy metal burden on their activities (Ireland and Wooton, 1976; Ireland, 1983). For instance, some earthworms prefer leaves that are rich in metals, which affects metal accumulation in soils through degradation (Ireland, 1983). Biological changes to mine spoils after application of biowastes are presented in Table 6.

5. CASE STUDIES OF BIOWASTE UTILIZATION

5.1 Biosolids in Combination With Calcium Carbonate for Metal Contaminated Hard Rock Mining Sites in the United States

Biosolids in combination with high calcium carbonate equivalent (CCE) materials have been used to restore a self-sustaining vegetative cover on a number of metal contaminated hard rock mining sites in the United States. These include sites on the U.S. Environmental Protection Agency's National Priorities list, also known as Superfund sites (Allen et al., 2007). With the restoration of a self-sustaining cover, the potential for the restored sites to function as an attractive nuisance has been raised. In this case, restored vegetation can provide a habitat for a range of wildlife that could recolonize the sites. The wildlife potentially could be exposed to metals on the site.

Table 6 Selected soil biological property changes after application of biowastes on mine spoils.

Site/country	Biowastes type	Site description	Evaluated biological property	Observations	References
Vancouver, Canada	Biosolids	Mine tailing	TAM TAH Iron reducers Sulfate reducers Denitrifiers	Increased from 97×10^5 to 416×10^5 g^{-1} soils Increased from 1×10^4 to 146×10^4 g^{-1} soils Increased from 5×10^2 to 255×10^2 g^{-1} soils Increased from 3×10^2 to 20×10^2 g^{-1} soils Increased from 3×10^3 to 441×10^3 g^{-1} soils	Gardner et al. (2010)
Huelva, Spain	Stablized sewage sludge	Mine waste soils	Dehydrogenase β-Glucosidase Alkyl phosphatase Protease Arylsulfatase Total enzyme activity Soil MBC Soil-induced respiration	Increased from 0.1 to 3.7 μg substrate g^{-1} h^{-1} Increased from 10.2 to 173.0 μg substrate g^{-1} h^{-1} Increased from 2.6 to 530.0 μg substrate g^{-1} h^{-1} Increased from 0.3 to 181.0 μg substrate g^{-1} h^{-1} Increased from 0.2 to 14.9 μg substrate g^{-1} h^{-1} Increased from 0.61 to 61.9 μg substrate g^{-1} h^{-1} Increased from 67.4 to 77.7 mg C 100 g^{-1} soils Increased from 2.9 to 3.5 mg CO_2 100 g^{-1} h^{-1} Increased from 0.0 to 16.5 mg C kg^{-1}	Mingorance et al. (2014)

(Continued)

Table 6 Selected soil biological property changes after application of biowastes on mine spoils.—cont'd.

Site/country	Biowastes type	Site description	Evaluated biological property	Observations	References
Queensland, Australia	Compost derived from municipal green waste	Bauxite residue sand soils	MBC MBR MQ	Increased from 3.0 to 22.5 μg CO_2-C g^{-1} day^{-1} Increased from 0.0 to 140 μg CO_2-C mg^{-1} day^{-1}	Jones et al. (2012)
Queensland, Australia	Poultry manure	Bauxite residue sand soils	L-Asparaginase Phosphatase MBC MBR MQ	Increased from nd to 0.01 μmol product cm^{-3} h^{-1} Increased from 0.1 to 0.6 μmol product cm^{-3} h^{-1} Increased from 0.0 to 55.0 mg C kg^{-1} Increased from 15.0 to 37.5 μg CO_2-C g^{-1} day^{-1} Increased from 0.0 to 62.5 μg CO_2-C mg^{-1} day^{-1}	Jones et al. (2011)
Portugal	Sewage sludge	Pyrite mine soils	β-Glucosidase Acid phosphatase Cellulase Urease Protease	Increased from 0.2 to 2.1 μmol PNP g^{-1} h^{-1} Increased from 0.9 to 1.9 μmol PNP g^{-1} h^{-1} Increased from 0.0 to 0.1 μmol glucose g^{-1} h^{-1} Increased from 2.5 to 38.0 μmol NH_4-N g^{-1} h^{-1} Increased from 0.0 to 5.5 mmol tyrosine g^{-1} h^{-1}	Alvarenga et al. (2008)
Spain	Mixed municipal	Mine soils	Under *Erica australis*	Increased from 1.2 to 7.5 μg TPF g^{-1} 16 h^{-1} Increased from 49.2 to 100.0 nmol PNP g^{-1} h^{-1}	de Varennes et al. (2010)

	solid waste (MSW)		Dehydrogenase θ-Glucosidase Cellulase Urease Protease Phosphatase	Increased from 250.0 to 500.0 nmol glucose g^{-1} 16 h^{-1} Increased from 1100.0 to 5000.0 nmol NH_4^+-N g^{-1} 2 h^{-1} Increased from 1.2 to 20.0 nmol tyrosine g^{-1} 2 h^{-1} Increased from 125.0 to 250.0 nmol PNP g^{-1} h^{-1}	
Spain	Mixed MSW	Mine soils	Under *Dactylis glomerata* Dehydrogenase θ-Glucosidase Cellulase Urease Protease Phosphatase	Increased from 2.5 to 25.0 μg TPF g^{-1} 16 h^{-1} Increased from 380.0 to 620.0 nmol PNP g^{-1} h^{-1} Increased from 800.0 to 1550.0 nmol glucose g^{-1} 16 h^{-1} Increased from 4200.0 to 6000.0 nmol NH_4^+-N g^{-1} 2 h^{-1} Increased from 3.8 to 21.3 nmol tyrosine g^{-1} 2 h^{-1} Increased from 1250.0 to 2000.0 nmol PNP g^{-1} h^{-1}	de Varennes et al. (2010)
Southern Spain	Municipal waste compost	Mine soils	MBC Arylsulfatase β-Glucosidase Dehydrogenase	Increased from 93.6 to 564.9 mg C kg^{-1} soils Increased from 80.7 to 767.9 mg PNP kg^{-1} soils Increased from 157.4 to 900.0 mg PNP kg^{-1} soils Increased from 3.5 to 53.5 mg INT kg^{-1} 20h^{-1}	Mora et al. (2005)

(*Continued*)

Table 6 Selected soil biological property changes after application of biowastes on mine spoils.—cont'd.

Site/country	Biowastes type	Site description	Evaluated biological property	Observations	References
Southern Spain	Biosolids	Mine soils	MBC Arylsulfatase β-Glucosidase Dehydrogenase	Increased from 93.6 to 244.2 mg C kg^{-1} soils Increased from 80.7 to 400.0 mg PNP kg^{-1} soils Increased from 157.4 to 376.8 mg PNP kg^{-1} soils Increased from 3.5 to 49.5 mg INT kg^{-1} 20 h^{-1}	Mora et al. (2005)
Southern Spain	Deciduous forest litter	Mine soils	MBC Arylsulfatase β-Glucosidase Dehydrogenase	Increased from 93.6 to 416.8 mg C kg^{-1} soils Increased from 80.7 to 319.4 mg PNP kg^{-1} soils Increased from 157.4 to 1106.6 mg PNP kg^{-1} soils Decreased from 3.5 to 2.2 mg INT kg^{-1} 20h^{-1}	Mora et al. (2005)
Southern Spain	Leonardite—a low rank coal between peat and subbituminous rich in humic acids	Mine soils	MBC Arylsulfatase β-Glucosidase Dehydrogenase	Increased from 93.6 to 218.1 mg C kg^{-1} soils Increased from 80.7 to 277.4 mg PNP kg^{-1} soils Increased from 157.4 to 465.5 mg PNP kg^{-1} soils Increased from 3.5 to 4.4 mg INT kg^{-1} 20h^{-1}	Mora et al. (2005)

Southern Spain	Leonardite	Mine soils	MBC Dehydrogenase Arylsulfatase β-Glucosidase Acid phosphatase	Increased from 19.6 to 339.4 mg biomass C kg^{-1} Increased from 0.1 to 1.1 mg INTF $kg^{-1}h^{-1}$ Increased from 118.5 to 237.0 mg PNP $kg^{-1}h^{-1}$ Increased from 89.2 to 330.9 mg PNP $kg^{-1}h^{-1}$ Increased from 1733.8 to 4061.9 mg PNP $kg^{-1}h^{-1}$	Mora et al. (2006)
Southern Spain	Deciduous forest litter	Mine accident soils	MBC Dehydrogenase Arylsulfatase β-Glucosidase Acid phosphatase	Increased from 19.6 to 164.9 mg biomass C kg^{-1} Increased from 0.1 to 3.5 mg INTF $kg^{-1}h^{-1}$ Increased from 118.5 to 330.9 mg PNP $kg^{-1}h^{-1}$ Increased from 89.2 to 1588.2 mg PNP $kg^{-1}h^{-1}$ Increased from 1733.8 to 7430.3 mg PNP $kg^{-1}h^{-1}$	Mora et al. (2006)
Southern Spain	Municipal waste compost	Mine soils	MBC Dehydrogenase Arylsulfatase β-Glucosidase Acid phosphatase	Increased from 19.6 to 444.2 mg biomass C kg^{-1} Increased from 0.1 to 11.0 mg INTF $kg^{-1}h^{-1}$ Increased from 118.5 to 1362.9 mg PNP $kg^{-1}h^{-1}$ Increased from 89.2 to 663.2 mg PNP $kg^{-1}h^{-1}$ Increased from 1733.8 to 3405.7 mg PNP $kg^{-1}h^{-1}$	Mora et al. (2006)

(*Continued*)

Table 6 Selected soil biological property changes after application of biowastes on mine spoils.—cont'd.

Site/country	Biowastes type	Site description	Evaluated biological property	Observations	References
Southern Spain	Biosolids compost	Mine soils	MBC Dehydrogenase Arylsulfatase β-Glucosidase Acid phosphatase	Increased from 19.6 to 339.9 mg biomass C kg^{-1} Increased from 0.1 to 6.9 mg INTF $kg^{-1}h^{-1}$ Increased from 118.5 to 760.5 mg PNP $kg^{-1}h^{-1}$ Increased from 89.2 to 456.9 mg PNP $kg^{-1}h^{-1}$ Increased from 1733.8 to 2650.2 mg PNP $kg^{-1}h^{-1}$	Mora et al. (2006)

This may directly harm the wildlife that recolonizes the site as well as have an indirect effect through food chain transfer of contaminants because of predator/prey relationships.

The potential for restored sites to function as an attractive nuisance was investigated at two sites where a vegetative cover had been restored using biosolids and limestone: alluvial tailings in Leadville, Colorado (CO), and mine tailings in Jasper County, Missouri (MO) (Fig. 2). In each case the

Figure 2 Mine waste in Jasper County, Missouri. (A) Tailings amended with biosolids and lime 12–14 years prior. (B) Treated tailings amended with biosolids and lime; the area was not seeded postamendment addition. (C–E) Deposits of alluvial tailings along the Arkansas River in Leadville, Colorado, pre- and 10-years postamendment addition. Areas closest to the river were amended with biosolids compost and lime. Areas more than 3-m from the river were amended with biosolids and limestone (Brown et al., 2005, 2007, 2009).

primary contaminants were Cd, Pb, and Zn. Both sites represented cases of historic contamination and cover large areas.

Historic alluvial tailing deposits along the Arkansas River in Leadville, Colorado, had large areas with high metal (Cd, Pb, and Zn) contamination (Table 7). Oxidation of pyrites in the tailing deposits had resulted in acidification of the tailings. These historic deposits had been barren for many decades. Proximity to the river resulted in re-entrainment of the deposits during high flow events. This in turn resulted in acidification and metal contamination of the river waters with resulting destruction of aquatic habitat. Municipal biosolids and lime (each at 224 tons/ha) were applied to the tailing deposits in 1998, as both a full-scale remediation effort and as small-scale replicated field plots (Brown et al., 2005). The replicated field plots included evaluations of different residuals/lime mixtures (Brown et al., 2005, 2007, 2009). The EPA carried out a full-scale ecosystem evaluation on the large-scale restored areas. Soil function was evaluated using population counts of different soil organisms. Microbial function (CO_2 respiration and ammonia oxidation) was also evaluated. Plant diversity and metal uptake were measured. An earthworm (*Eisenia fetida*) toxicity test was also conducted. Small mammal trapping and total body burden were measured.

Mine wastes at the Jasper County site consisted of a mixture of overburden and tailings. Overburden material was coarse with high percentages >2 mm. This resulted in poor physical properties but lower levels of contamination. Biosolids and lime were added at varied rates, 110–336 tons/ha biosolids and 24–48 tons/ha $CaCO_3$, to large tracts of the site starting in 1998 with operations completed in 2001 (Brown et al., 2014). Materials were mixed, surface applied, and then incorporated into the waste materials using a large plow. Site evaluations were carried out periodically after amendment addition with the final evaluation conducted in 2012 (Brown et al., 2014). A similar range of variables, as was used in Leadville, were measured. Small mammal trapping was done in 2002 with kidney pathology and total Cd, Pb, and Zn measured to assess body burden.

Both sites showed little to no potential for the restored sites to function as attractive nuisances. Modeling in Leadville showed no potential for ecosystem transfer of contaminants (Table 7). Results from Jasper showed high rates of recolonization of the restored sites by a range of small mammals. Kidney examination showed no indication of pathology for 61 animals with 22 showing some evidence of Cd exposure. However, only four of these had damage that may have been extensive enough to compromise function. The

Table 7 Range of metals in control and amended tailings, earthworm Cd for depurated worms, Cd concentration in deer mice (*Peromyscus maniculatus*) for whole body (Leadville, CO) and kidney (Jasper County, MO), and the ratio of NH_4:NO_3 for amended tailings.

	Total soil metals					
Site	**Cd mg/kg**	**Pb**	**Zn**	**Earthworm Cd mg/kg**	**Deer mouse Cd**	**NH_4:NO_3**
Leadville, CO	9.5–24.3	1560–3170	1400–2520			
Control				ND		100
Treated				18.1	0.53 ± 0.06	0.08
Jasper, MO	7.4–57	300–4900	1044–10000			
Control				104–111		2.4
Treated				15–49	13.5 ± 10	0.2

ND, not detected.

final sampling at Jasper also showed that the restored tailings were beginning to function as a soil (Fig. 2). Nutrient analysis showed that the biosolids amended tailings had similar or higher nutrients than topsoil that had been excavated from home gardens. Total C and N were also similar or greater than the topsoil. WHC for the biosolids amended soils was also much greater than the unamended chat.

The results from these two long-term sites indicated that restoration of mine impacted, metal contaminated sites using municipal biosolids and lime, can result in restored ecosystem function over time with minimal potential for the sites to function as an attractive nuisance. In the future, as the restored tailings began to function as soils, many of the ecosystem benefits associated with functional soils will be realized.

5.2 Crop Residues as Biowastes for Metal Immobilization in Rice Paddies Affected by Mining Activities in Korea

One of the most serious problems for rice production in Korea is the contamination of rice paddies by heavy metals from mining activities. Abandoned or closed mines are the primary source of Cd and Pb contamination in rice paddy soils in Korea (Ok et al., 2011a; Yang et al., 2006, 2007).

Biowastes transform metals into less available fractions and are effective in alleviating heavy metal toxicity to plants. Biowastes, such as crop residues, can be incorporated into metal-contaminated soils to maintain SOM, improve soil physicochemical and biological properties, and increase crop production.

The importance of energy crops that are used to produce biofuels or electricity is now being recognized around the world, and, in Korea, the rapeseed plant is being cultivated as a biodiesel resource under a double cropping system in rice paddies. Typically, the rapeseed plant can be cultivated in rice paddies as a winter crop after the rice plant is harvested. Previous studies evaluated the effects of adding rapeseed residue as a green manure in a rice–rapeseed double cropping system in order to reduce N fertilizer (Choi et al., 2014; Ok et al., 2011b).

Ok et al. (2011b) showed that rapeseed residue as green manure resulted in an increase in SOM and enhanced the microbial populations in the soil (Ok et al., 2011b). Sequential extraction also revealed that the addition of rapeseed residue decreased the easily accessible fractions of Cd by 5–14% and that of Pb by 30–39% by transforming them into less accessible fractions, thereby reducing metal availability to the rice plant (Fig. 3). Choi et al. (2014)

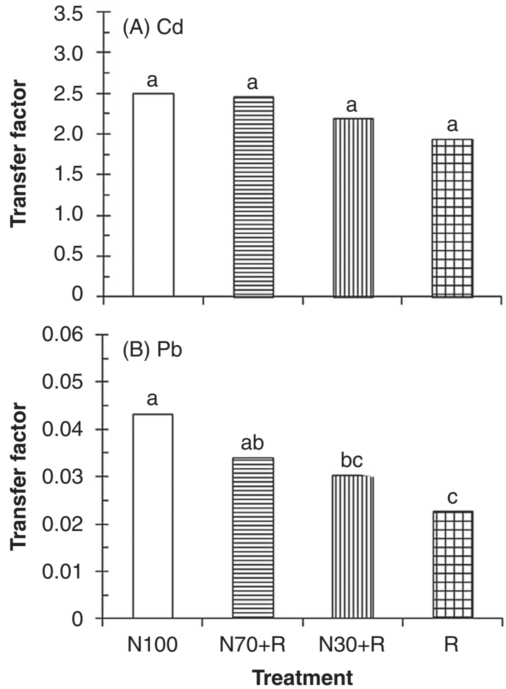

Figure 3 Effect of the rapeseed residue alone or in combination with mineral N fertilizer (N100, 100% mineral N fertilizer; N70+R, 70% mineral N fertilizer + rapeseed residue; N30+R, 30% mineral N fertilizer + rapeseed residue; R, rapeseed residue alone) on transfer factors (TF's) of (A) Cd and (B) Pb [TF, mg metal content in rice tissues (leaves + stems + hulls)/mg metal content in contaminated paddy soil]. The same letters above each bar indicate no difference at a 0.05 significance level, as determined by Tukey's HSD test (n = 3). *Adapted from Ok et al. (2011b).*

found greater SOM and exchangeable cations in paddy soils treated with rapeseed residue relative to the case where conventional mineral N fertilizer alone was used (Choi et al., 2014).

Lee et al. (2013) further evaluated the efficacy of crop residue, in combination with powdered eggshell waste, in immobilizing Cd and Pb in soils (Lee et al., 2013). They concluded that the combination of eggshell waste and rapeseed waste from biofuel manufacturing could mitigate soil acidification and could immobilize heavy metals in soils.

Overall, the incorporation of rapeseed residue into metal-contaminated rice paddy soils may sustain SOM, improve the chemical and biological properties of soils, and decrease heavy metal phytoavailability.

5.3 Crop Residues as Biowastes for Sulfidic Tailing Soil Rehabilitation in Australia

Mine wastes such as bauxite residues and sulfidic tailings from base and precious metal mines are extreme cases of mine spoils in mined landscapes, which pose the greatest challenges in the closure and rehabilitation of mined lands (Huang et al., 2012a,b). Many native plant species at mine sites possess ecological traits such as tolerance to drought, salinity and nutrient deficiencies, particularly those in the ancient landscapes of Australia (Grigg et al., 2008; He et al., 2012; Laliberte et al., 2013; Lambers et al., 2010). Despite these tolerances of harsh environmental conditions, physiological stresses induced by extreme physical (eg, mechanical compaction) and chemical (eg, elevated metal/metalloid and salt concentrations and strongly acidic conditions) factors in the tailings far exceed any plant physiological tolerance boundaries of these species (Huang et al., 2012b; Lottermoser, 2010; Mendez and Maier, 2008a, 2008b). As a result, stimulating soil formation and functional root zone development in the tailing landscapes is a prerequisite for plant germination, growth, and survival, let alone sustainable plant community development (Uzarowicz and Skiba, 2011). For this purpose, amendments with OM (particularly those of plant biomass origin) may be useful in both stimulating weathering and "soil" (technosols) development in the tailings and improving plant survival and growth. The preference of OM of plant origin over other OM rich in N such as biosolids and sewage sludge in remediating sulfidic tailings and stimulating soil formation is based on two key considerations: (1) the volume of biosolids (sewage sludge) is often small at remote mines and (2) biosolids are rich in N and P, and Australian native plant species that represent main species in target plants communities used in rehabilitation in tailing landscapes do not require large amounts of N and P. Observations in a long-term field trial on weathered Cu/Pb-Zn tailings (decommissioned for 20–30 years before the trial) showed that the amendments of nutrient-rich sewage sludge caused competitive colonization of buffel grass (*Cenchrus ciliaris*) and severely suppressed the survival of native woody species.

5.3.1 Physical Improvement for Root Penetration in Neutral Base Metal Mine Tailings

In sulfidic tailings with adequate neutralization capacity and which are relatively low in geochemical reactivity, amendments of plant biomass-based OM (such as aged hay) directly improved physical properties by significantly improving root penetration and water infiltration, leading to successful colonization of native grass species and a high plant cover (Table 8)

Table 8 A visual estimation of vegetation cover and observed plant growth status of the grass species (*Cenchrus ciliaris*) in aged hay treatments.

Tailings treatment	Surface coverage (%)	Growth status
Intact tailings	1–2	Some germination, but little survival and further growth
Tailings (loose)	3–5	Some germination, low survival, very limited vegetative growth, but ceased shortly after
10% Hay	35–45	Low productivity and biomass with green shoots, developed to flowering stage
20% Hay	80–90	High biomass and dense coverage, developed to flowering and seeding, able to complete life cycle

(Huang et al., 2011). Neutral tailings (pH 7.5–8.2) generated from Cu (chalcopyrite)-Au ores at Ernest Henry Mine, Cloncurry, Northwest Queensland, Australia were studied (Huang et al., 2011). In the hay-amended tailings (with about 20% v/v), roots of the native grass reached a depth of >60 cm below the surface, in contrast to the rooting depth of 5–10 cm in the compacted tailings without any OM amendment (Fig. 4). The poor root penetration into the tailings was the result of the fine particle size and high bulk density of the tailings (Fig. 5). This improvement is highly critical for the survival of revegetated plants during the prolonged dry season under semiarid climatic conditions, such as at the mine at Cloncurry.

5.3.2 Organic Matter Amendments in Soil Formation

The development of functional technosols from mine tailings is possible after an initial rapid weathering and addition of OM amendments (Li and Huang, 2014; Uzarowicz and Skiba, 2011). Subsequently in weathered tailings with a much reduced sulfide content (eg, <5%) and neutral pH conditions, the development of physical structure (ie, aggregates and pores) and of soil-like heterotrophic microbial communties is fundamental for the formation of technosols with soil-like biogeochemcial processes (Li and Huang, 2014).

Improving aggregation in base metal mine tailings is an important step in the engineered pedogenesis to rehabilitate functional root zones for direct revegetation with native plant communities. The organomineral association critical for microaggregation can be stimulated in OM amended tailings through the interaction between functional organic ligands in the

Figure 4 The emergence and growth of mixed grass species in (A) intact tailings (B) loose tailings (C) tailings amended with 10% hay (v/v) and (D) tailings amended with 20% hay (v/v). The surface of the tailings was covered with a layer of black soil (about 10 cm), into which mixed grass species were sown on Jan. 18, 2011. The photos were taken on Apr. 10, 2011 (Huang et al., 2011).

OM and charged surfaces of minerals in the tailings. In a pot experiment that lasted 40 days, neutral Cu/Pb-Zn tailings were amended with sugarcane residue (SR) or its compost (SRC), which were planted with or without a pioneer plant species, red flinders (*Iseilema vaginiflorum*) (Yuan, 2014). It was found that the organic amendments (particularly the SRC) with functional organic ligands (eg, amine, carboxylic, hydroxylic, alcoholic, and phenolic hydroxyls) mainly stimulated microaggregation in the Cu/Pb-Zn tailings, while the role of plant roots in the tailings was mainly related to the formation of macroaggregates. Therefore, both organic amendments rich in functional groups and pioneer plant growth may be adopted to improve physical structure and stimulate the development of technosols in Cu/Pb-Zn tailings.

In weathered sulfidic tailings of base metal mines such as Pb-Zn-Ag mines, OM amendments or organic carbon sources from plant roots were useful

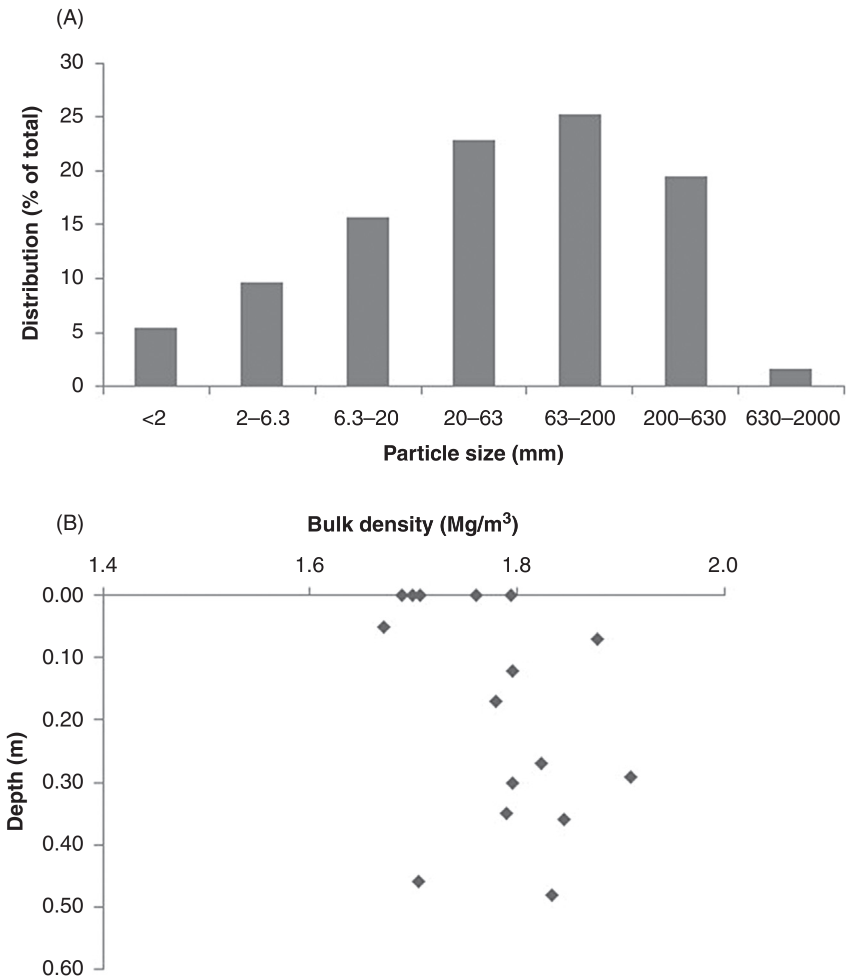

Figure 5 (A) The distribution of particle size in Cu-Au-Mo mine tailings (Ernest Henry Mine) bulk-sampled from the top layer (0–0.6 m) (B) Bulk density of tailings at different depths from the surface (0–0.6 m) below the surface. The data were pooled from five pits accessible in the area across the tailings (Huang et al., 2011).

to stimulate further weathering of unstable minerals and induce the shift of microbial communities toward heterotrophic ones in the root zones (Li et al., 2013b, 2014; Li and Huang, 2014). Community composition, which was compared in Pb-Zn-Cu tailings with and without revegetation under subtropical and semiarid climatic conditions, was characterized by using 16S rRNA gene based pyrosequencing with universal primers (Li et al., 2014).

Bacterial diversity, as indicated by both the operational taxonomic units (OTU's) number and the Shannon index of the revegetated samples, was significantly higher than that of the sample from the pure tailings. At the phylum level, *Proteobacteria* and *Bacteroidetes* were remarkably higher in the revegetated samples compared to the pure tailings; this is possibly related to the change in the organic carbon pool. Phylotypes belonging to *Thiobacillus* were found thriving in the revegetated tailings (Li et al., 2013b).

5.4 Revegetation of Mine-Impacted Areas Using Organic Waste Amendments: A Case Study From the Tri-State Mining Region, USA

The Tri-State Mining Region is comprised of portions of the states of Kansas (KS), Missouri, and Oklahoma in the central United States. Lead and Zn were mined extensively from the mid 1800s until the early 1950s, and smelted until the early 1970s. A variety of environmental issues remain including Pb-, Zn-, and Cd-contaminated soils in a number of communities, highly contaminated abandoned smelter sites, large quantities of mine wastes, and thousands of hectares of land with little or no vegetation. Surface water and shallow groundwater are already negatively impacted by metals. A lack of vegetative cover can be attributed to poor soil chemical, physical, and biological properties that adversely impact the establishment or maintenance of vegetation. One particularly challenging aspect of phytostabilization is long-term viability of the vegetation. Soil amendments allow rapid growth of vegetation shortly after application, often in areas completely void of vegetation previously, but the vegetation slowly declines and can be completely gone within 3–5 years (Pierzynski et al., 2002).

The use of P to reduce Pb bioavailability has been proposed as an effective in situ stabilization option for Pb-contaminated soils and mine waste materials (Hettiarachchi and Pierzynski, 2004). Early work on contaminated materials, including mine waste materials from southeast KS, has demonstrated the treatments are effective in reducing Pb uptake by rats with relative bioavailability (as compared to Pb acetate) reduced from 0.34 mg/kg in unamended soil to 0.24 mg/kg in the same soil amended with rock phosphate (Hettiarachchi et al., 2003). Moreover research focused on the nature of the mineralogical changes in Pb and Zn induced by addition of various P sources to mine waste materials and mine-impacted soils from the region and how they may change over time have been investigated (Baker et al., 2012, 2014; Hettiarachchi et al., 2001).

Organic amendments seem to have mixed effects on bioavailability of Pb in soils (Attanayake et al., 2014; Brown et al., 2003a, 2012; Defoe et al., 2014; Sauve et al., 1998; Vega et al., 2009). Differential responses to organic amendments are not surprising because speciation and bioavailability of Pb in soils amended with OM depend on the composition and maturity of the OM and site-specific soil chemistry. Researchers have credited reduced bioavailability and/or phytoavailability of Pb due to compost addition to high Fe and Mn concentrations in some compost materials such as biosolids (Brown et al., 2003a, 2012); increased SOM content and subsequent increase in CEC (Vega et al., 2009); dilution (Attanayake et al., 2014; Defoe et al., 2014); and increased soil-available P concentration (Attanayake et al., 2014). In contrast, Sauve et al. (1998) showed that application of leaf compost increased soil Pb solubility at a soil pH range of 6.5–8 by promoting formation of dissolved organic Pb complexes, suggesting that the overall impact of compost addition is decided by magnitude of number of different favorable or unfavorable reactions that occur simultaneously in the soil upon compost addition (Sauve et al., 1998).

Two field studies were established in areas highly contaminated in Pb and Zn in the Tri-State Mining area near Galena, KS (37° 9'16" N; 94° 50' 2" W). Of particular interest here are the contaminated control (C), low compost (LC, 45 tons/ha), and high compost (HC, 269 tons/ha) treatments of composted beef manure. Soil samples were taken approximately 1 week after application to establish a Time 0 and switch grass (*Panicum virgatum*) was sown into each plot. Soil samples were again taken after 157, 371, 553, 729, 1082, 1274, 1468, 1636, and 2885 days after treatment (DAT). Soil enzyme activities (β-glucosidase, alkaline, and arylsulfatase) were assayed following methods described by Tabatabai (1994) within 2 weeks after each sampling event on field-moist samples and are reported on a dry-weight basis (Tabatabai, 1994). Changes to soil microbial communities were assessed at 729 DAT by measuring the soil phospholipid fatty acid (PLFA) content (White and Ringelberg, 1998). Immediately after sampling/sieving, approximately 20 g of moist material was frozen at -20^{o}C and lyophilized. The total lipids were extracted from the lyophilized material, the total lipid extract was separated into PLFA and waste lipids using silicic acid chromatography, the fatty acids were cleaved from the glycerol backbone by KOH saponification, and the harvested fatty acids were methylated to form fatty acid methyl esters. The nomenclature used to designate the identified fatty acids is the total number of carbon atoms and number of double bonds, followed by the position of the double bond from the methyl end of the molecule. For additional information see Baker et al. (2011).

Experimental results gathered over a period of nearly 8 years showed that the HC treatment had significantly higher microbial biomass, enzyme activities, total organic N, total organic C, and extractable P and K than the control, while, in general, all LC treatments did not differ from the control. Alkaline phosphatase activity is presented as a representative enzyme. Extractable P and phosphatase activities were significantly increased by the HC treatment at all sample times (Fig. 6) and remained relatively constant with time. Extractable Zn, Cd, and Pb were also significantly lower for the HC treatments compared to all other treatments. PLFA analysis and subsequent fractionations indicated that gram (+) and gram (−) bacteria, fungi, and total microbial biomass were significantly increased by compost

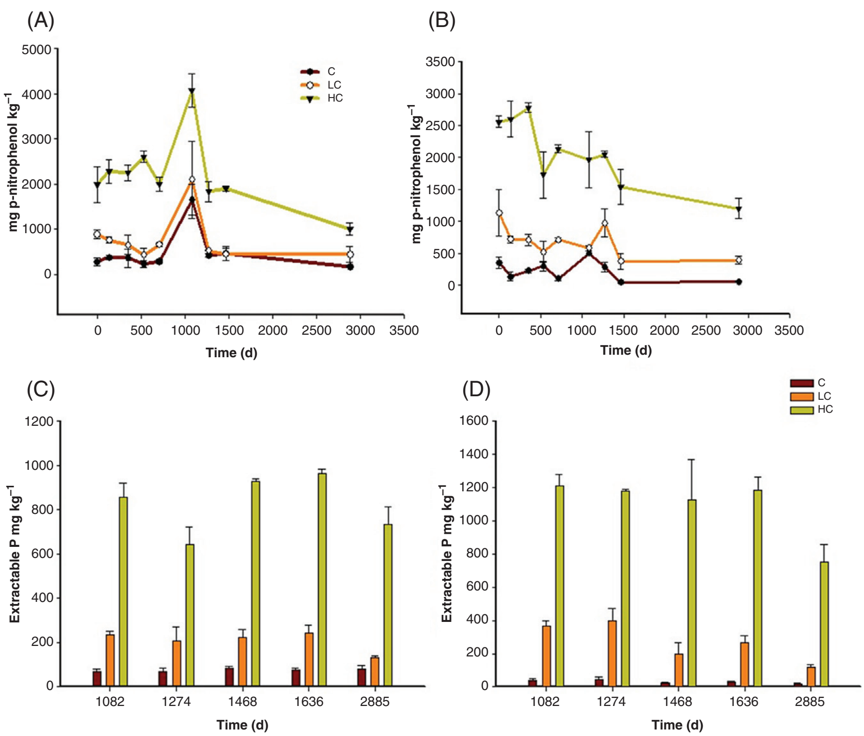

Figure 6 Alkaline phosphatase activity at (A) Site A and (B) Site B. Mehlich-III extractable P concentration at (C) Site A and (D) Site B. Site A and Site B were two different sites with experimental plots at Galena, Kansas, where a Pb-Zn mine was located. C, control; LC, low compost at 45 tons/ha; and HC, high compost at 269 tons/ha (Baker et al., 2011).

Figure 7 (A) A significant portion of the Tri-State Mining Region is heavily impacted by years of Pb- and Zn-mining activities. (B) Contrasting differences between control or LC-treated plots versus HC plots.

additions at 729 DAT, but actinomycetes were unchanged relative to the control (Baker et al., 2011). A decrease in plant biomass and enzyme activities seen in the HC treatments after nearly 8 years suggests that long-term sustainability of these efforts may require repeated addition of soil amendments every 4–5 years. Application of high rates compost has been shown to provide efficient vegetative growth, reduce metal exposure, and improve chemical and biochemical properties of the soil (Fig. 7). This long-term research study provides evidence that high rates of organic waste amendments can be used for mine-impacted area reclamation.

6. EFFICACY OF BIOWASTES ON MINE SPOIL REHABILITATION

Success in rehabilitation of mine spoils with biowastes is more complex than simply adding biowastes to them. Even so, utilization of biowastes is a first step in increasing OM in spoils, thereby accelerating mine spoil rehabilitation (Castillejo and Castello, 2010; Larney and Angers, 2012). Several factors influence utilization of biowastes for mine spoil rehabilitation. These factors include the availability of biowastes and their transportation cost, public acceptance and political ramifications, suitability of biowastes for the targeted spoil, and application rate (Castillejo and Castello, 2010; Larney and Angers, 2012). The immediate goal of the rehabilitation has to be decided at the very beginning. For instance, quickly establishing a vegetative cover to avoid soil erosion was the immediate goal at the Antaibao surface mine area in Shanxi, China (Li et al., 2013a). Here, sewage sludge combined with nitrogen fertilizer was used to establish four indigenous grass species present in the mining area. Significant increments in their biomasses occurred compared to the control. High salt and nutrient content of biowastes has a great influence on plant establishment at mine spoils. For example, a high rate of MSW compost application resulted in overfertilizing of plants due to the excess salt and nutrients content (Castillejo and Castello, 2010). Salts that leached out quickly from MSW into soil aggravated plant growth (Albaladejo et al., 1994).

Most of mine spoil rehabilitations depend on a one-time application of large amounts of biowastes rather than multiple applications of lower amounts. Castillejo and Castello (2010) reported no significant effect to spoil soils by a low rate of MSW compost. However, relatively low rates of biosolids, but applied over a long term, had significant long-term effects on soil chemistry including the soil microbial community and plant community in semiarid rangelands of northern Colorado (Sullivan et al., 2006).

Surface application of biosolids in combination with other biowastes (ie, woody debris, wood ash, pulp and paper sludge, and compost) at a Superfund site resulted in reduced concentrations of extractable metal ions in plant species (Brown et al., 2003b). Neutralized red mud from bauxite refining mixed with biosolids was reported to be an efficient liming agent under extremely acidic conditions, and it allowed grass and trees to grow (Maddocks et al., 2004). Different combinations and amounts of iron by-products derived from production of TiO_2, triple superphosphate (TSP), phosphoric acids, and biosolids compost were used to immobilize Pb, Zn, and Cd from smelter-contaminated soils (Brown et al., 2004).

Biosolids compost alone is an effective treatment to immobilize Pb, Zn, and Cd in soils. The nature of metal adsorption behavior to the soils has been identified to differ among types of biowastes. For instance, metal adsorption in manure is largely administered by OM, however the corresponding adsorption with biosolids is largely governed by the certain mineral phases such as phosphates, aluminum compounds, and iron compounds (Bolan et al., 2004). The decomposition or mineralization of organic fraction of biowastes and mobilization of contaminants over time leads to serious concerns in mine spoil rehabilitation. For example, an incubation study performed with biowastes (ie, manure and compost) and heavy metal contaminated soils has observed a significant degradation of OM from biowastes thereby increased Cu bioavailability through metal-OM chelation (Clemente et al., 2006). On the other hand, the heavy metal chelation with OM derivatives has been identified to contaminate groundwater through leaching, runoff, and erosion (Schwab et al., 2007). However, some mineral phases associated with clay minerals in metal-contaminated soils have been reported to enhance heavy metal leaching due to the soil aging and weathering (Lamb et al., 2009). Despite of the metals immobilization or reduction in metals uptake to the plants by the mineral phases associated with biowastes (ie, phosphates, aluminum, and iron compounds associated with biosolids), these mineral phases can also effect to leach heavy metals from biowastes applied spoil soils. In addition, inorganic nutrients such as phosphorus and nitrate leaching have reported further, indicating issues (ie, groundwater pollution) related to the land application of biowastes. However, high affinity of metal–inorganic associations of biowastes has provided evidence for increased retention of Cd (Li et al., 2001). Iron and Mn compounds have also been shown to contribute to low mobility of the trace metals in biowastes (Hettiarachchi et al., 2006). Therefore, the stabilization of biowastes thereby minimizing decomposition is an important factor to be considered. In this particular challenge, the rate of biowaste degradation has been reported to depend on the source and pretreatment of biowastes prior to application to the lands. For instance, composting is used to convert the easily degradable organic fractions into a more recalcitrant fraction through humification, thereby increasing the maturity of the organic fractions of biowastes (Bernai et al., 1998). The applicability of biowastes in terms of long-term storage and stabilization of carbon in degraded mine sites has been identified elsewhere (Shrestha and Lal, 2006).

According to Table 3, which summarizes application information, biosolids have been used in most studies to enhance the health of the spoil material. Quickly establishing vegetative cover is important (Li et al., 2013a). Many projects fail, but, nevertheless, knowledge is gained from them. For example, sewage sludge was used to counteract effects of sulfide oxidation of copper mine tailings in northern Sweden (Forsberg and Ledin, 2006). However, the sludge-treated tailings decreased the pH from 6.4 to 4.8 due to sulfide oxidation, and increased the levels of soluble Zn. In addition, soluble As and Cd were lower in the unaltered tailings compared to the sludge-treated ones.

7. CONCLUSIONS AND FUTURE RESEARCH NEEDS

Mining advances global economic prosperity. However, mining activities cause mine spoils that are identified as waste materials, tailings, open-cast excavations, drainage ditch, or other cut-in engineering work, and shortfalls or overburden in topsoil and subsoil associated with mining. Elevated heavy metal concentrations associated with mine spoils cause serious contamination to surrounding soils, groundwater, and the food chain leading to phytotoxicity or health impacts. Low levels of OM are a common problem in mine spoils, resulting in poor soil health for plant growth and soil microbial life. Inherent poor soil texture and structure further lead to poor WHC. In addition, unsightly and unproductive surroundings are common on lands that have been disturbed by mining activities.

Even though different methods are available for mine spoil rehabilitation, utilization of biowastes has been identified as a potential, useful, remedy that has been used on many occasions. Biowastes such as biosolids, MSW, animal and poultry manure, papermill sludge, and plant residues are generated in large quantities as a consequence of increased human population and subsequent expansion of livestock or poultry industries. Biowastes have been identified as a good source of OM and essential nutrients, which enhance physical, chemical, and biological properties of degraded lands. Therefore, biowastes are extensively used as ideal soil amendments for land reclamation and revegetation. Advantages associated with the use of biowastes in the remediation of mine spoils are as follows: restoration of ecosystem services (eg, food and energy production), regulating services (eg, nutrients cycling and carbon sequestration), supporting services (eg, water purification, pest and disease control), and cultural services (eg, recreational and scientific discovery) (Fig. 8). Case studies presented in this review have illustrated the following:

- Biosolids in combination with lime have been used to restore a self-sustaining vegetative cover on Zn-Pb-Cd mine tailings (Brown et al., 2005, 2007, 2009, 2014). The restored tailings have started to function successfully as a soil through establishing nutrients similar or higher than topsoils excavated from home gardens and improvement of WHC. High rates of recolonization of the restored sites by a range of small mammals have been observed confirming no potential for ecosystem transfer of contaminants.
- Reduction of heavy metal phytoavailability in metal-contaminated rice paddy soils, sustained SOM, and improved chemical and biological properties of soils have been observed by application of rapeseed plant residues with mineral nitrogen fertilizer (Ok et al., 2011b).
- Improvement of root penetration and water infiltration, along with successful colonization of native grass species and high plant cover, has been observed by application of aged hay to Cu-Au-Mo mine tailings (Huang et al., 2011). In addition, development of soil microaggregates in

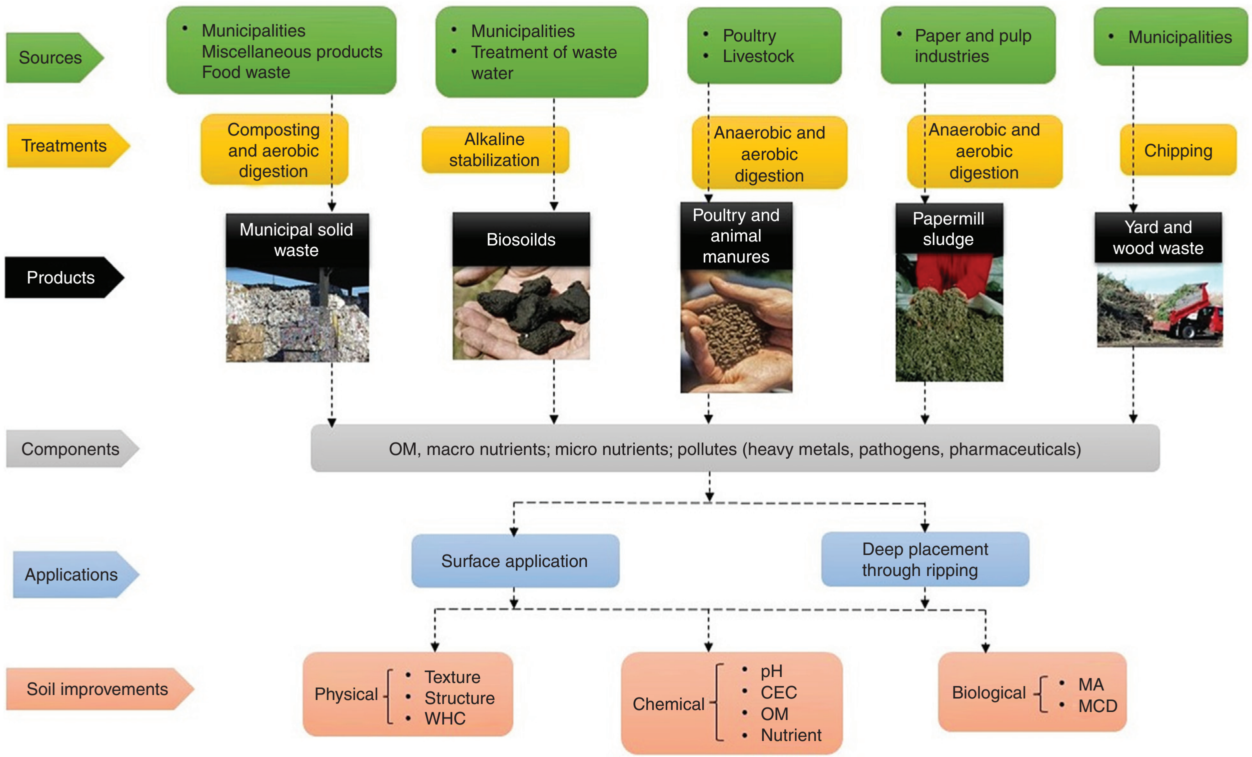

Figure 8 Various approaches to the utilization of different biowastes for mine spoil rehabilitation. *CEC*, cation exchange capacity; *MA*, microbial activity; *MCD*, microbial community diversity; *OM*, organic matter; *WHC*, water holding capacity.

Cu-Pb-Zn mine tailings was observed by application of sugarcane residue or its compost, and this study highlighted the importance of functional organic ligands for microaggregation (Yuan, 2014). Biowastes or organic carbon sources from plant roots have been identified as stimulating further weathering of unstable minerals and inducing a shift of microbial communities toward heterotrophic ones in the root zone in weathered sulfidic mine tailings. Bacterial diversity, characterized by using 16S rRNA gene based pyrosequencing with universal primers, was higher in the revegetated samples than in the pure tailings (Li et al., 2014).

- Soil enzyme activities and soil microbial communities in Pb-Zn mine spoils treated with low and high levels of composted beef manure were compared. The results showed that the HC treatment had a higher microbial biomass, soil enzyme activities, total organic N, total organic C, and extractable P, K, Zn, Cd, and Pb (Baker et al., 2011).

Heavy metals in some biowastes (eg, manure and biosolids) can leach and result in soil contamination. Accelerated transport of heavy metals in tailings can occur due to chelation of the metals with OM complexes. Nutrients such as nitrate and phosphate can also leach during mine spoil rehabilitation. Decomposition of the organic fraction of biowastes is the main cause for leaching of metals in spoils. To minimize leaching of metals in biowastes, a number of options have been suggested, such as pretreatment of the biowastes (ie, composting of manure and MSW) and chemical stabilization (ie, alkaline treatment of biosolids).

Due to constraints on utilization of biowastes and public opposition related to their use, a number of countries and organizations have developed regulations for their application to agricultural land or land for rehabilitation. Most regulations focus on avoidance or minimization of contamination from heavy metals and major inorganic nutrients.

This review shows that biowastes can be used for mine spoil rehabilitation, but challenges remain, and they need to be addressed carefully to minimize negative environmental effects. Based on the current knowledge of biowastes application to mine spoils, the following research could be pursued for sustainable rehabilitation:

- Detailed characterization of biowastes and mine spoil environment are needed to enhance the rehabilitation process.
- Utilization of stabilized or pretreated biowastes should be considered to minimize contamination of the environment.
- Continued maintenance and scientific investigation of spoils treated with biowastes should be carried out. Long-term studies are crucial to identify

the merits and demerits of biowastes on spoils. In particular, heavy metal availability in soils treated with various sources of biowastes should be investigated.

- Environmental risks (ie, soil and ground water pollution, soil salinization, fauna and flora toxicity, ultimately contamination of foods through bioaccumulation) associated with the heavy metals and nutrients leached from the biowastes should be considered. With increased detection of emerging contaminants in some biowastes (ie, manure and biosolids), their fate and transport need careful consideration. Air pollution associated with biowastes used on spoils should be investigated further, to overcome nuisance from odors in the surrounding environment.
- Enhancement of stocks of carbon using biowastes on the low carbon containing mine spoils needs to be studied. In addition, stabilization of the spoils by biowastes and altered emissions of CO_2 and CH_4 associated with the biowastes need to be considered. With biowastes, mine spoils may be considered as carbon storage sites in the future.
- Regulations for mine spoil rehabilitation using biowastes need to be formulated and updated. They need to focus on the type of biowastes and their application rates, as well as their cumulative effects on different soil types, plants species, microbes, and animal species.
- Life cycle assessment (LCA) methodology needs to be applied to biowastes for recognizing the validity of storing carbon as a means of direct action climate change mitigation strategy, and dependence as a bioenergy source. Based on the outcomes from LCA, regulations and protocols need to be developed.

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