CHAPTER

BIOSOLIDS ENHANCE MINE SITE REHABILITATION AND REVEGETATION*

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CHAPTER OUTLINE

*This chapter is a condensed version of the following review paper: H. Wijesekara, N.S. Bolan, M. Vithanage, Y. Xu, S. Mandal, S.L. Brown, G.M. Hettiarachchi, G.M. Pierzynski, L. Huang, Y.S. Ok, M.B. Kirkham, C. Saint, A. Surapaneni. UTILIZATION OF BIOWASTE FOR MINE SPOIL REHABILITATION. Advances in Agronomy. (Accepted full manuscript for volume 138)

Environmental Materials and Waste. http://dx.doi.org/10.1016/B978-0-12-803837-6.00003-2

3.1 INTRODUCTION

Modern wastewater and sewage treatment facilities generate biosolids in large quantities. Globally, around 10×10^7 tons year^{-1} of biosolids is generated (Thangarajan et al., 2013). In 2050, the world's population is projected to be 9.6 billion and will potentially generate 17.5×10^7 tons year^{-1} of biosolids at the rate of 50 g persons^{-1} day^{-1}. Furthermore, 66% of the world's population (7.4 billion) is forecasted to be urban by 2050 (UN, 2015). This urban population will generate 13.5×10^7 tons year^{-1} of biosolids. To manage this large quantity, the most appropriate strategy would be reuse and recycling. Biosolids are reused in different ways. A significant quantity of biosolids is used as a fertilizer or soil conditioner. For example, around 59% of generated biosolids is used in agricultural lands in Australia (ANZBP, 2013). Furthermore, 6% and 4% are used in landscaping (composting) and land rehabilitation, respectively. Public opposition has been identified as a major challenge to be overcome for the use of biosolids in agriculture by many countries; therefore, biosolids have been stockpiled at sewage treatment plants indefinitely. These stockpiles cause environmental issues such as the direct emissions of greenhouse gases (GHGs): methane (CH_4), nitrous oxide (N_2O), and carbon dioxide (CO_2), which contribute significantly to global warming (Majumder et al., 2014). Globally, the overall contribution of the waste industry to the emission of GHGs is as small as $<3\%$, with total emissions of approximately 1446 $MtCO_{2\text{-eq}}$ in 2010. However, around 54% and 43% of this emission is contributed by wastewater handling (eg, stockpiling of biosolids) and land disposal of solid waste, respectively (Blanco et al., 2014). Therefore, biosolids must be used in an eco-friendly manner to avoid environmental pollution and ensure their sustainable reuse. Among sustainable uses, biosolids-associated mine site rehabilitation is an important strategy and has a significant role according to the perspective of environmental remediation.

The excavation of economically important resources from terrestrial landmasses, known as "mining," is an integral part of the current development trajectory of the world. Mining advances global economic prosperity by generating a large quantity of valuable natural resources. For example, Australia is the world's leading producer of a variety of minerals. The income from sales and services in the mining industry has been reported to be $217.8 billion between 2012 and 2014 for the Australian economy (ABS, 2014; AM, 2015). However, mining severely disturbs the land. Several thousands of historic and currently operating mines are scattered throughout the earth, with further estimates of around 0.4×10^6 km^2 land area disturbed by mining activities (Hooke and Martin-Duque, 2012). Unfortunately, most of these areas have never been properly reclaimed, thereby causing extensive damage to the environment (Hooke and Martin-Duque, 2012). Mining causes disturbance to the land in many ways. Generation and land disposal of waste materials such as mine tailings, subsoils, oxidized wastes, fireclay, and mudstone are the main causes of land disturbance. These waste materials contain potentially hazardous substances such as heavy metals in elevated concentrations, causing groundwater and soil contamination (Evangelou and Zhang, 1995). Production of acids associated with these waste materials can increase heavy metal pollution through enhanced metal solubility at an acidic pH (Bolan et al., 2003; Lindsay et al., 2015). The excavation of land or other cut-in engineering works causes shortfalls and overburdens topsoil and subsoil. As a result of poor soil characteristics such as low-level organic matter and poor soil texture and structure, these shortfalls and overburdens are identified as drastically disturbed unproductive landmasses (Diamond, 1999; Ghose, 2001; Johnson, 2003; Sopper, 1992). Ultimately, infertile soils and harsh conditions associated with these disturbed lands can adversely affect the establishment of soil microbial life, earthworms, other soil fauna, and plant growth (Boyer et al., 2011; Castillejo and Castello, 2010; Larney and Angers, 2012). Therefore, disturbed lands need to be rehabilitated to avoid potential environmental consequences and to restore the lost ecological wealth.

Depending on the level of land disturbance, environmental pollution, and cost, different methods are used to rehabilitate mine sites. Additions of organic or inorganic amendments (ie, biosolids, compost, fly ash, biochar), nano-enhanced materials (ie, iron oxides and iron sulfide nanoparticles), uncontaminated soils, heavy metal—tolerant genotypes, and forest floor and peat materials are current methods in mine site rehabilitation (Beasse, 2012; Liu and Lal, 2012; O'Reilly, 1997). Application of liner material over mine wastes and the isolation of mine wastes from the environment by storage in impoundments under water or behind dams are also used (Kossoff et al., 2014; Lamb et al., 2013). Among these methods, mine site rehabilitation by incorporating biowaste, in particular the application of biosolids, has been identified as a sustainable strategy, and therefore has been rewarded with greater attention.

This chapter provides a wider outlook at the compositions of different types of biosolids, their current regulations for use, and benefits of mine site rehabilitation. Special attention has been given to linking biosolids with afforestation of reclaimed mine sites. In addition, environmental challenges and pollution prevention strategies related to biosolids land application are discussed. Finally, future research needs and strategies are identified in terms of sustainable biosolids use in mine site rehabilitation.

3.2 GENERATION AND COMPOSITION OF BIOSOLIDS

3.2.1 GENERATION OF BIOSOLIDS

Sewage and wastewater treatment facilities generate biosolids. Biosolids are often referred to as treated sewage sludge. Sewage sludge is defined as any solid, semisolid, or liquid residue generated during the municipal wastewater and sewage treatment process. Biosolids can be defined as stabilized organic solids derived from sewage treatment processes (mostly resulting from the biological treatment of wastewater) which can be managed safely to be used beneficially for their nutrient, soil conditioning, energy, or other values (Shammas and Wang, 2008). An increased number of high-capacity wastewater treatment facilities generate large quantities of biosolids (Fig. 3.1). As a brief overview about the biosolids generation process, wastewater or sewage treatment plants receive wastewater and sewage from domestic, industrial, and agricultural sources. First, large objects, grit, or scum (naturally floating materials) are removed by screening at a grit chamber and the rest is forwarded to a sedimentary tank where primary sludge is produced. Primary sludge contains mainly fecal solids and further treatments are performed to meet the criteria for biosolids production. For this, different wastewater and sewage treatment plants use different techniques such as an activated sludge method, thickening (ie, flotation, gravity, centrifugation), digestion (ie, anaerobic or aerobic digestion), dewatering (ie, vacuum filters, centrifuges, belt press), alkaline treatment (ie, at high pH using lime), conditioning (ie, using coagulants and separation of water), drying or heat techniques (ie, at high temperature), and beta- or gamma-ray irradiation processes (ie, pathogen control) (Kajitvichyanukul et al., 2008; Wang et al., 2008). Some of these methods have their own advantages and disadvantages. For instance, biological stabilization leads to CO_2 emission via the decomposition of organic matter in

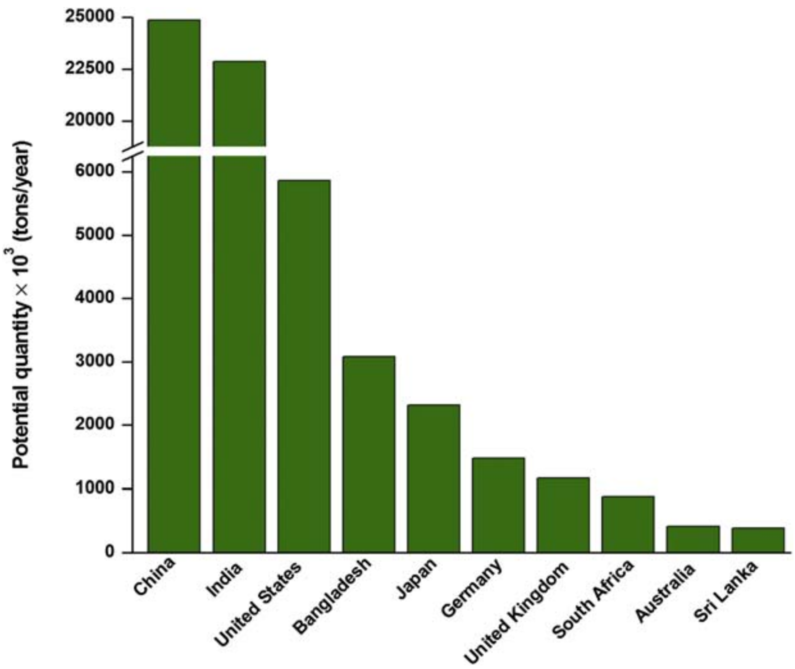

FIGURE 3.1

Potential quantity of biosolids produced in selected countries; values were calculated based on biosolids generation assuming 50 g person^{-1} day^{-1}.

sewage sludge and reduces odors (Wang et al., 2008). Most of these treatment techniques control pathogens (ie, some types of bacteria, viruses, protozoa, parasitic worms) that cause diseases and reduce the potential to attract flies, mosquitoes, or other disease-carrying vectors (Wang et al., 2008). In addition, in many countries it is mandatory to store biosolids for several months before their beneficial use to ensure elimination or inactivation of pathogens. For instance, in some sewage treatment plants in the United States, anaerobically digested sludge is stored for a minimum of 18 months for further stabilization before use (Tian et al., 2009).

Biosolids are defined in different groups by different countries. Table 3.2 shows selected criteria for grouping biosolids in the United States. Three types of biosolids have been defined by the US Environmental Protection Agency's (EPA's) sludge management program in terms of pathogen requirements and site restrictions (USEPA, 1994): "exceptional quality" (EQ) biosolids, Class "A" biosolids, and Class "B" biosolids. For the EQ biosolids, no restrictions are associated with the application of these biosolids to land, which can also be sold even as compost or soil amendments. Strict limits must be met on pollutants and pathogens to become eligible in this biosolids category. Class A biosolids (no detectable pathogens and used like commercial fertilizer) and Class B biosolids (a reduced level of pathogens) are considered as safe; however, certain requirements such as limiting public access to site application of biosolids, livestock grazing limitations, and crop harvesting schedule restrictions are attached with Class B biosolids application. Australia and New Zealand national guidelines consider recycling or disposal of biosolids and are grouped into seven categories (NWQMS, 2004). At the state level in Australia, the New South Wales (NSW) EPA has grouped biosolids into three groups: unrestricted use products, restricted use 1 products, and restricted use 2 products (Ang and Sparkes, 1997). The NSW EPA has strict regulations including application rates, contaminant limits for use of biosolids in agricultural land, and mine site rehabilitation. The European Union has implemented directives to regulate biosolids since 1986 (Evans, 2012): landfill, sludge, and waste incineration.

3.2.2 COMPOSITION OF BIOSOLIDS

Biosolids can be identified as a complex heterogeneous matrix. The composition and properties of biosolids vary with many factors such as the process of generation, age of biosolids, environmental conditions such as temperature and humidity (Wang et al., 2008). For example, alkaline material such as lime, kiln dust, portland cement, and fly ash are used by many biosolids producers to reduce pathogen content and immobilize heavy metals (Kajitvichyanukul et al., 2008; Pichtel and Hayes, 1990; Wang et al., 2008). However, the alkaline stabilization of biosolids with lime is reported to cause the loss of nitrogen (N) because of the volatilization of ammonia (NH_3), thereby reducing the N fertilizer value from biosolids (Wang et al., 2008). Nevertheless, the alkaline stabilization enhances carbon (C) stabilization (Chowdhury et al., 2015; Palumbo et al., 2004).

Total solids (TS) and volatile solids (VS) are some important parameters describing the organic matter and solid contents of biosolids. TS include suspended solids and dissolved solids. Liquid, dewatered, dried, or compost biosolids contain 2–12%, 12–30%, and 50% TS, respectively (Wang et al., 2008). VS indicates the availability of readily decomposable organic matter in biosolids (Wang et al., 2008). The VS fraction has a strong correlation with odor emissions in biosolids, and hence VS is a critical determinant when applying biosolids to land (Wang et al., 2008). Biosolids is composed of a high amount of organic matter ranging from 50% to 70% (Li, 2012). It has been reported that fatty acids constitute the predominant polar fraction, thereby representing 51% of the organic compounds in biosolids, whereas steroids and aliphatic compounds contribute to 13% and 14%, respectively (Torri and Alberti, 2012). Furthermore, biosolids' organic matter significantly differs in chemistry with soil organic matter, because biosolids are composed of more alkyl carbon than soil (Smernik et al., 2003). A study based on ^{13}C-nuclear magnetic resonance spectroscopy revealed the increased alkyl C and decreased aromatic C content in soil humic acids with the application of biosolids (Chiu and Tian, 2011). Another, comparable study revealed the addition of high doses of biosolids resulting in fatty acids, amino acids, and/or paraffinic structures in soils (Antilén et al., 2014). Also, biosolids are rich in carbon-based compounds, mainly microbial biopolymers (Torrecillas et al., 2013). For instance, polysaccharide and proteins can be found in biosolids as their pure forms or in association with other compounds such as glycoproteins and lipopolysaccharides. Lipids, nucleic acids, and humic substances are other major organic constituents in biosolids (García Becerra et al., 2010).

Biosolids are a significant source of inorganic nutrients such as N and phosphorus (P). The ammonium-N $\left(NH_4{}^{+} - N\right)$ content in biosolids depends on sewage treatment, biosolids stabilization, and storage process, because $NH_4{}^{+}$ is readily volatilized into NH_3 gas (Wang et al., 2008). Biosolids contain around 3% N with low nitrate-N (NO_3-N) (ie, <0.05%) content, but most of N exists as organic-N in complex molecules such as proteins, nucleic acids, amines, and other cellular materials (Kajitvichyanukul et al., 2008; Wang et al., 2008). To become an N source for crops, these complex molecules must be converted into inorganic ammonium and NO_3-N through degradation (García Becerra et al., 2010). In addition, biosolids contain P, indicating its additional nutritional value, and thereby exhibiting the importance of P recovery and conservation (Evans, 2012).

The metal fraction of biosolids is well understood. Heavy metals such as zinc (Zn), copper (Cu), nickel (Ni), and lead (Pb) have been reported to be found in significant concentrations in biosolids. Table 3.1 shows the trace element concentrations of biosolids and sewage sludge from different sewage treatment facilities. Heavy metals in biosolids are thought to be sorbed by both organic matter (McBride, 1995) and inorganic constituents (Hettiarachchi et al., 2003). The role of sulfide minerals in binding and precipitating heavy metals (ie, Zn and Cu) in freshly produced and stockpiled biosolids has been reported elsewhere (Donner et al., 2011). A metal speciation approach involving synchrotron techniques concluded the importance of a heavy metal—iron oxides association, but not as the dominant mechanism governing metal binding in biosolids as previously suggested. Zn sorption with iron oxides and Cu sorption with organic matter has been reported, which highlights the need for future studies on metal association with the mineralizable biosolids fraction (Donner et al., 2012). In the Australian context, Victorian water authorities produce a variety of biosolid types and have reported different physicochemical parameters. Depending on the process of generation and the stockpiled duration or age of biosolids, diverse characteristics of individual elements (ie, C, N, P, sulfur [S], silicon [Si], and aluminum [A]), organic matter, clay, silt, and sand have been reported (Albuquerque et al., 2014). Furthermore, the sludge-drying process by clay-lined drying pans contributes to the clay mineral matter content and obviously increases in Al and Si fractions (Albuquerque et al., 2014). The same study revealed that biosolids more than 3 years old had less organic C than those less than 3 years old, which suggests C loss owing to mineralization with time and dilution of biosolids with clay impurity that influences the sludge-drying process (Albuquerque et al., 2014). In a study in India, a higher percentage of heavy metals (ie, iron [Fe], Zn, and cadmium [Cd]) in residual fraction and a lower percentage of exchangeable and carbonate-bound fraction were reported in biosolids (Bai et al., 2012). That particular study revealed that the concentrations of heavy metals in the biosolids were lower than those established by the European legislations.

3.3 LAND APPLICATION OF BIOSOLIDS AND POLLUTION PREVENTION

3.3.1 HEAVY METALS

The occurrence of heavy metals in biosolids can be a major cause of soil and water pollution. The repeated application of biosolids to Genesee silt loam soils has increased the lability of chromium (Cr), Pb, cobalt (Co), Zn, Cu, Ni, and arsenic (As) by 20%, 11%, 9%, 9%, 8%, 6% and 4%,

Table 3.1 Selected References for Trace Element Concentrations in Municipal Sewage Sludge and Biosolids

Sample	Concentration ($mg\ kg^{-1}$ or %)														References
	Al	As	Cd	Co	Cr	Cu	Hg	Mn	Fe	Mo	Ni	Pb	Se	Zn	
Athens sewage sludge		4.97	11.2		75.1	54.7		1248			53.4	2.47	1.24	294	Jackson and Miller (2000)
Urban compost			0.48		71	119		214			15	324		328	de Abreu et al. (1996)
Urban compost			0.45		65	89		350			4304	85.3		354	de Abreu et al. (1996)
Sewage sludge			11.4		645	870		497			479	226		1788	de Abreu et al. (1996)
Denver sewage sludge		8.1	26	7.1	280	816	7.8	220		84.9		950	4.57	1672	Capar et al. (1978)
City sewage sludge		14.3	104	9.6	1441	1346	8.6	194		14.3		1832	3.1	2132	Capar et al. (1978)
Austin sewage sludge (Autinite)		9.4	3.3	4.10	106	300	1.5	430			36.7	86.9	2.57	563	Raven and Loeppert (1997)
Milwaukee sewage sludge (Milorganite)			7.2	4.07	2940	—	1.1	142			31.2	130	1.04	450	Raven and Loeppert (1997)
Anaerobically digested, dried by centrifugation, air-dried in wind rows, and finally stockpiled	5660								32,200						Bolan et al. (2013) Donner et al. (2012)
Anaerobically digested, dried by centrifugation, air-dried in wind rows, and finally stockpiled	6.3%					438		246	2.8%					134	
Digested sewage in an Imhoff tank, dewatered on sand bed, and aged in storage area	1.9%					1121		298	2.1%					1157	Donner et al. (2012)
Anaerobic digested and lagoon-dried sewage	1.4%					913		340	1.7%					1068	Donner et al. (2012)
Activated sludge-derived biosolids		5.6	2.3	2.1	29	463	1.7	245	9792		23	59	5.0	482	Galvín et al. (2009)
Biosolids derived through biodisks as biological process		18	1.4	6.5	70	649	1.2	687	16,717		47	172	12	1117	Galvín et al. (2009)

respectively (Islam et al., 2013). This study revealed that the extractable fractions of Pb, As, Zn, and Cu concentrations were significantly higher at 0–15 cm soil depth (Islam et al., 2013). Consequently, the accumulated heavy metals may mobilize from the soils to groundwater and surface water bodies. Organic matter in biosolids, hydrous oxides (ie, iron [Fe] and manganese [Mn] oxides in clays), and mineral phases (ie, phosphates and aluminum compounds) immobilize heavy metals (Hettiarachchi et al., 2006). Nevertheless, the rapid decomposition of organic matter releases organically bound heavy metals and increases the bioavailability or toxicity (Obrador et al., 2001). The increased bioavailability of heavy metals causes their excessive uptake by plants or leaching down to the soil profile. The plant uptake of heavy metals is correlated with extractable forms of metals rather than the total metal contents in soils. Interestingly, some plant species can protect the food chain by providing an effective barrier against the uptake of most heavy metals (Lu et al., 2012). The influence of biosolids on the availability of heavy metals and their effects on seed germination has been also reported (Walter et al., 2006). The results revealed that pelletization of sludges increased the availability of Ni, whereas dewatering sludge led to the greater availability of Cr and Mn. Furthermore, biosolids affected seed germination and most serious adverse effects were caused by dewatered sludge.

The protection of soil systems would be better guaranteed if the input heavy metal concentration were evaluated systematically in biosolids before land application. Alkaline stabilization of biosolids is identified as a better strategy to immobilize heavy metals. Fly ash–stabilized sludge remarkably reduced the availability of heavy metals by chemical modification into less available forms (Su and Wong, 2004). The study revealed that loamy acid soil amended with fly ash–stabilized sludge was able to reduce the availability of Cu, Zn, Ni, and Cd in the sludge. With increasing fly ash amendment rates, the percentages of Cu, Zn, and Ni in the residual fraction increased. Furthermore, concentrations of Zn and Cu in shoot tissues of corn were reduced remarkably after the application of ash amendments (Su and Wong, 2004).

3.3.2 NUTRIENTS

The land application of biosolids has the potential to contaminate surface and groundwater bodies with excess nutrients such as N and P (Haney et al., 2015). Application of biosolids followed by irrigation resulted in the excessive leaching of nutrients in agricultural fields (Price et al., 2015). Biosolids are rich in nutrients; therefore, excess nutrients can be released into surface and groundwater bodies, contributing to eutrophication (He et al., 2015). The potential for nutrient leaching resulting from the application of biosolids depends on several factors: topography, rainfall intensity, soil texture, soil pH, cation exchange capacity (CEC), redox conditions, organic matter content, cropping systems, and time and methods of biosolids application (Nkoa, 2014; Zinati et al., 2004). Nitrogen leaching has gained significant attention in studies, possibly owing to the high concentration of N in biosolids (Oladeji et al., 2013; Yager and McMahon, 2012). On the contrary, high amounts of N leaching can be attributed to the low efficiency of animal N use. It has been reported that only 20–30% of N consumed by dairy cows is converted into meat and milk; the rest is excreted as feces and urine (Nkoa, 2014). When applying biosolids to lands NH_4^+ can be adsorbed by negatively charged soil particles or uptake by plant root cells or oxidized to nitrate (NO_3^-) by nitrifying microorganisms. It is not possible to adsorb NO_3^- by negatively charged soil particles since the repulsion of ions (Nkoa, 2014). Therefore, NO_3^- is highly mobile along the soil profile, causing increased surface and groundwater contamination. A biosolids-applied farmland has been identified as the main source in which to observe fairly high concentrations of nitrate in shallow groundwater monitoring wells in Colorado, United States (Yager and McMahon, 2012). Groundwater nitrate concentrations were reported in high concentrations in biosolids-amended strip-mined lands compared with reference fields, but they were below the regulatory limit of 10 mg L^{-1} (Oladeji et al., 2013).

P accumulates as different fractions in soils, such as water-soluble, exchangeable, carbonate, Fe-Mn oxides, organic, and residual (Zinati et al., 2004). P concentrations in these fractions have different immobilization behaviors under different environmental conditions. The temperature and moisture content of a biosolids-amended soil may determine the forms and availability of phosphorus (O'Connor et al., 2004). Although P is immobile in most soils, coarse-textured soils enhance the P transport through soil profile. Most P in biosolids is available as inorganic P (Elliott et al., 2002). Several studies have reported P release in soils amended with biosolids. A kinetics study revealed that biosolids-amended soil released eight times more P compared with bare soil (Islas-Espinoza et al., 2014). The degree of P saturation (DPC) is a major indicator to determine potential P loss from a particular soil (Alleoni et al., 2014). A long-term field experiment was conducted to investigate the DPC in Oxisol amended with biosolids; it revealed critical levels of DPC in the top soil layer (Alleoni et al., 2014). Proper management practices included cultivation against slopes, maintaining sufficient gaps between biosolids applications and sowing, consideration of climatic conditions before applying biosolids to the land, and systematic soil testing. In addition, decision making should be taken into account to minimize nutrient leaching from biosolids.

3.3.3 PATHOGENS

Land application of biosolids may be responsible for spreading human pathogens. Biosolids primarily originate from human feces, hospitals, abattoirs, and wastewater treatment plants, and therefore consist of a wide range of pathogens (Lasobars et al., 1999; Sidhu and Toze, 2009). These pathogens include viruses (eg, hepatitis A, rotavirus), bacteria (eg, *Escherichia coli*, *Salmonella*), parasites (eg, *Cryptosporidium*, *Giardia*), and helminths (eg, *Ascaris*) (Brewster et al., 2003; Kato et al., 2003; Monpoeho et al., 2004; Pepper et al., 2012; Zaleski et al., 2005). Human exposure to pathogens can be through direct exposure to biosolids or indirectly via biosolids-based soil amendments. Sewage treatment plant workers and the general public using biosolids-amendment products may be directly exposed to pathogens through accidental ingestion owing to the contamination of hands and clothes. Food crops grown in biosolids-amended soil and water contaminated with biosolids are indirect pathways to spreading pathogens to human beings who reside in remote areas. In other words, pathogens in biosolids may create a transmission route to spread pathogenic diseases between urban and rural areas (Sidhu and Toze, 2009). Some vector animals such as birds, mammals, and insects might propagate infections to humans in remote areas by transporting pathogens over long distances. Human pathogens can survive for a long time, possibly owing to the resistance to inactivation. Different physicochemical and biological parameters such as temperature, moisture content, oxygen, pH, sunlight, soil type, texture, and predation may influence the inactivation of pathogens in biosolids (Sidhu et al., 2001). Nevertheless, the influence of those factors acts differently from one pathogen to another. A number of studies have shown the numbers and survival patterns of pathogens in biosolids, whereas few studies have reported the detection limits of the method used (Mocé-Llivina et al., 2003; Payment et al., 2001; Pourcher et al., 2005).

Viruses are infectious agents and are more concentrated in biosolids during the wastewater treatment process. Different types of enteric viruses, such as norovirus, hepatitis A virus, rotavirus, and enterovirus, are commonly reported in biosolids (Bofill-Mas et al., 2000). Noroviruses have gained worldwide significance owing to the most common cause of acute gastroenteritis (Lodder et al., 1999). Hepatitis A virus has worldwide distribution and is more prominent in developing countries. Rotaviruses are responsible for causing gastroenteritis in young children and in immune-compromised individuals worldwide. Human enteroviruses are commonly detected in all types of biosolids (Sidhu and Toze, 2009).

Bacterial pathogens are a major cause of gastroenteritis all over the world. In general, enteric viruses, parasites, and protozoa are considered to be obligatory parasites; hence, they are unable to multiply in biosolids (Gibbs et al., 1997). Nevertheless, bacteria are able to multiply under favorable conditions in biosolids. For instance, a study with soil amendments and biosolids storage trials determined the density of *E. coli* and *Salmonella* (Gibbs et al., 1997). This study revealed the indicator organisms at undetectable concentrations in a hot, dry summer period whereas their repopulation occurred during the rainfall at the beginning of winter in both trials. *Salmonella* sp. is the most prominent bacteria in raw and treated biosolids and one of the major causes of diarrhea, fever, and abdominal cramps, which can last for 4 to 7 days (Bicudo and Goyal, 2003). *Salmonella* sp. can survive up to 3 months in stored conditions and can regrow under favorable conditions (Bicudo and Goyal, 2003). In the case of *E. coli* O157:H7, they survive in pastures for more than 11 years and more than 6 weeks under winter conditions (Bicudo and Goyal, 2003). In favorable conditions, *E. coli* O157:H7 is able to regrow in biosolids. The symptoms of *E. coli* infection are similar to *Salmonella*, but in approximately 10–20% of patients it causes serious diseases such as hemorrhagic colitis or hemolytic uremic syndrome (Bicudo and Goyal, 2003). *Shigella* sp. is a prominent cause of gastric illness globally and is commonly found in sewage worldwide. Another prominent bacterial pathogen spread by biosolids is *Helicobacter pylori*, which is responsible for gastric ulcers and is linked to gastric cancer (Peng et al., 2002).

Among protozoans of concern, genera *Cryptosporidium* and *Giardia* are highly resistant to environmental stress. They are a significant cause of gastroenteritis, possibly owing to low infectious doses (Caccio et al., 2003). These protozoans need a host for reproduction and are impossible to inactivate as a result of their oocysts, which gain a thick spore coat that protects them from many disinfectants (Bicudo and Goyal, 2003). *Cryptosporidium* oocysts are able to survive in soil for up to 3 months (Kato et al., 2003). *Cyclospora cayetanensis* is an oocyst-forming intracellular protozoan reported to be a significant cause of gastroenteritis. Human pathogenic microsporidia such as *Encephalitozoon intestinalis* and *Enterocytozoon bieneusi* are responsible for diarrheal illness worldwide (Sidhu and Toze, 2009).

Helminth infections are leading concerns, especially in developing countries, possibly as a result of the lack of sanitation facilities. As a result of high settling velocities, most helminth eggs concentrate rapidly in biosolids (Nelson, 2003). *Ascaris lumbricoides* is a major parasite worldwide, and almost a quarter of the world's population is infected by *Ascaris* (Johnson et al., 1998). Available information about the density and fate of helminths in biosolids is reported mainly in *Ascaris* (Johnson et al., 1998), which may be due to the higher prevalence of *Ascaris* than other helminths. *Ascaris* eggs can tolerate environmental conditions (ie, storage lagoons in temperate locations) and hence remain infective for long time, possibly several years. *Ascaris* eggs have been found after 29 weeks in stored biosolids (Johnson et al., 1998). Moreover, hookworms such as *Ancylostoma duodenale* also infect a quarter of human population at any one time (Johnson et al., 1998). Overall, improper treatment practices of sewage, utilization, and management practices of biosolids can produce an exposure pathway causing pathogenic diseases in humans. Consequently, biosolids handling processes need to be designed to ensure safety from numbers of pathogens and to safeguard public health.

3.3.4 ODOR EMISSIONS

The land application of biosolids releases odorous volatile organic compounds (VOCs) (eg, terpenes, alcohols, ketones, furans, sulfur-containing compounds, and amines) and ammonia (Komilis et al., 2004; Maulini-Duran et al., 2013). These unpleasant odors are released during biodegradation and cause discomfort. Exposure to higher concentrations of these odors leads to toxicological effects such as sensory irritation and psychohygenic effects to humans (Lleo et al., 2013). The initial substrate chemical composition, pH, moisture content, redox potential, temperature, microbial activity, and physical and chemical properties of VOCs in biosolids affect the extent of odor generation when applied to land (Rosenfeld et al., 2001). In one study, terpenes and ketones were identified as the most abundant VOCs in overall emissions (Rosenfeld et al., 2001). Odor emissions from two anaerobically digested biosolids from Washington, the United States, consisted of ammonia, dimethyl disulfide, carbon disulfide, trimethyl amine, acetone, and methyl ethyl ketone, whereas dry biosolids produced volatile compounds including methyl ethyl disulfide, methane thiol, acetic acid, propionic acid, and butyric acid (Rosenfeld et al., 2001). Identification and semiquantitative determination of volatile and semivolatile organic compounds were assessed by Kotowska et al. (2012), and approximately 170 VOCs were detected, including aliphatic and aromatic hydrocarbons, alcohols, esters, carbonyls, as well as sulfur-, nitrogen-, and chlorine-containing compounds. Ethyl ether, *n*-hexane, *p*-xylene, *o*-xylene, mesitylene, *m*-ethylbenzene, limonene, *n*-decane, *n*-undecane, and *n*-dodecane were detected as more prominent substances (Kotowska et al., 2012).

3.3.5 GREENHOUSE GAS EMISSIONS

Biosolids contribute to GHG emissions in different stages, mainly through stockpiles and their land application. The direct emission of GHG generated from biosolid stockpiles in Melbourne, Australia was reported (Majumder et al., 2014). According to this study, the youngest biosolids (<1 years) released higher amounts of CH_4 and N_2O emissions of 60.2 kg of CO_2-e per Mg of biosolids per year. In comparison, stockpiles aged between 1 and 3 years emitted higher overall GHGs (about 29 kg CO_2-e Mg^{-1} $year^{-1}$) compared with the oldest stockpiles (about 10 kg CO_2-e Mg^{-1} $year^{-1}$). Moreover, the youngest stockpiles released two-thirds of the GHG as N_2O, whereas equal amount of N_2O and CO_2 were emitted from the 1- to 3-year-old stockpiles and CO_2 emissions were dominant in old stockpiles. This particular study also assessed GHG flux from a large biosolids stockpiles and two shallow stockpiles, one of which was planted with willow (*Salix reichardtii*) trees. Results revealed that all stockpiles emitted large amounts of GHGs ranging from 38 to 65 kg CO_2-e Mg^{-1} $year^{-1}$. Furthermore, GHG emissions were dominated by N_2O and CO_2, whereas CH_4 emissions were negligible and accounted for less than 2% (Majumder et al., 2014). Consequently, this particular study concluded that the biosolids stockpiles were significant sources for GHG emissions, but planting willow trees did not reduce GHG emissions. Charcoal amending onto biosolids may reduce the emissions of GHGs. Aguilar-Chavez et al. (2012) monitored the effect of charcoal application on GHG emissions from biosolids-amended arable land cultivated with wheat. Results revealed lower cumulative GHG emissions over 45 days as 2%, 34%, and 39% in the presence of 1.5%, 3.0%, and 4.5% charcoal, respectively, than for the control soils.

The emission of N_2O could have resulted from biochemical (ie, nitrification and denitrification) and chemical (ie, chemodenitrification) pathways in soil; these processes are controlled by the availability of O_2 and soil pH (Nkoa, 2014). Nitrifying bacteria such as *Nitrosomas*, *Nitrosococcus*, *Nitrobacter*, and *Nitrococcus* are able to convert nitrite $\left(NO_2^-\right)$ to N_2O. Denitrifiers such as *Paracoccus*, *Pseudomonas*, and *Thiobacillus* are able to release N_2O when reducing nitrate $\left(NO_3^-\right)$ to dinitrogen (N_2) during denitrification. The global warming potential of N_2O is significantly high (approximately 300 times higher than that of CO_2) and therefore it is considered to be one of the major GHGs and ozone-depleting substances released from the earth (Borjesson and Berglund, 2007).

Methane is produced as a result of microbial respiration under anaerobic conditions when carbon is available as the only electron acceptor (Maulini-Duran et al., 2013). When energetically favorable electron acceptors such as oxygen, iron, nitrogen, sulfur, and manganese are exhausted, carbon is used as an electron acceptor (Brown et al., 2008). In the case of NH_3 emission, ammonium, urea, and other organic nitrogen compounds in biosolids are possible sources. Temperature, pH, and aeration may govern the release of NH_3 into the environment (Nkoa, 2014). Once NH_3 gas is emitted, it reacts with atmospheric gases such as H_2SO_4, HCl, and HNO_3 and produces soluble aerosol salts such as NH_4Cl, NH_4NO_3, and NH_4SO_4 (McMurry et al., 1983). The atmospheric deposition of NH_3 through acid rain results in the acidification of terrestrial and aquatic ecosystems. Eutrophication of freshwater bodies could occur from atmospheric NH_3 deposition (Chowdhury et al., 2014).

Some actions such as selecting remote sites, minimizing the length of time for storage of biosolids, and avoiding land application when wind conditions favor the transport of odors should be taken into account to minimize odor and GHG emissions from biosolids applications.

3.3.6 EMERGING CONTAMINANTS

Over the past couple of decades, significant attention has been given to selected groups of persistent organic pollutants in biosolids, including chlorinated dioxins/furans, polychlorinated biphenyls, and polycyclic hydrocarbons (Clarke and Smith, 2011). Most of these compounds do not affect human health when biosolids are recycled to farmland, possibly because of effective source control (Hundal et al., 2008; Wilson et al., 1997). Nevertheless, a number of emerging organic contaminants were identified in biosolids based on environmental persistence, human toxicity, and evidence of bioaccumulation in humans and the environment. For instance, perfluorinated chemicals, polychlorinated naphthalenes, triclocarban, benzothiazoles, synthetic musks, steroids, and reactive nanomaterials were recognized for priority attention because they can enter into living organisms via biosolids-amended soil (Brunetti et al., 2015). Thereby, precautionary actions such as quality control during biosolids generation and continuous monitoring (long-term) after biosolids land application need to be taken to ensure long-term sustainability and environmental security. The production of quality-controlled biosolids and their stabilization methods during land application could be used to avoid issues with respect to nutrients, pathogens, emerging contaminants, and vectors. Fig. 3.2 shows possible pathways of chemical and biological contaminants released by biosolids into the environment.

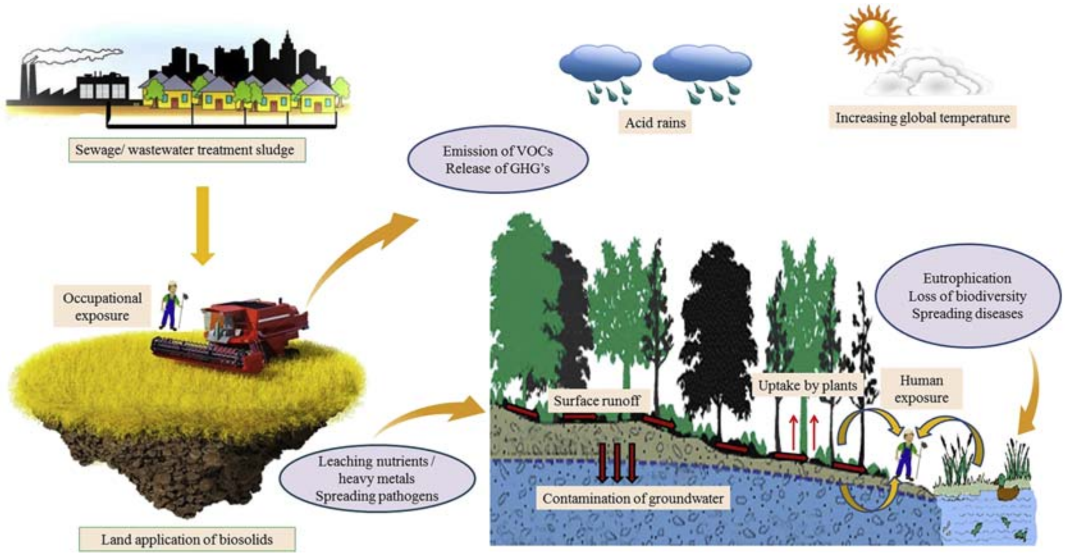

FIGURE 3.2

Possible pathways of chemical and biological contaminants released by biosolids in the environment.

3.4 REGULATIONS OF BIOSOLIDS USE

A number of countries and organizations around the world have developed regulations for the application of biosolids to agricultural lands or disturbed lands for rehabilitation. Mainly, three reasons can be identified for the development of these regulations regarding biosolids. The elevated production and fertilization or soil conditioning value of biosolids has increased their land application; therefore, regulations are required for their sustainable management (Tjell, 1986; USEPA, 1994). Potential risks (ie, spread of pathogens) associated with the land application of biosolids has led to the formulation of regulations to ensure minimal harm to the environment and to minimize public health risks (NWQMS, 2004).

Among various regulations regarding biosolids in different countries, regulations in the United States draw significant attention and their guidelines are frequently used as model regulations by some countries. For example, the pathogen and vector attraction reduction criteria of the US EPA's Code of Federal Regulations Part 503 rule for land application of sewage sludge are used by the NSW EPA in Australia (Ang and Sparkes, 1997). However, before formulating the regulations for biosolids use in the United States, the background was not environmentally responsive. For example, tons of partially treated wastewater from many wastewater treatment plants were discharged into the rivers, lakes, and bays in the United States before the 1950s (Lu et al., 2012). On the contrary, sludge disposal into the ocean from wastewater treatment plants continued until the Ocean Dumping Ban Act was ratified in 1988. The National Environmental Policy Act of 1969 regulated wastewater sludge use to protect the environment (USEPA, 1979). The Federal Water Pollution Control Act was enacted in 1972, further contributing deed restrictions against environmental pollution from wastewater treatment plants in the United States. Under provisions of the Resource Conservation and

Table 3.2 Ceiling Concentrations for 10 Heavy Metals (Defined as Pollutants) That Define Different Options on Biosolids Under US Environmental Protection Agency's Sludge Management Program (USEPA, 1994)

Pollutant	Ceiling Concentration Limits for All Biosolids Applied to Land (mg kg^{-1})[a]	Pollutant Concentration Limits for EQ and PC Biosolids (mg kg^{-1})[a]	Cumulative Pollutant Loading Rate Limits for CPLR Biosolids (kg ha^{-1})	Annual Pollutant Loading Rate Limits for APLR Biosolids (kg ha^{-1} $year^{-1}$)
As	75	41	41	2
Cd	85	39	39	1.9
Cr	3000	1200	3000	150
Cu	4300	1500	1500	75
Pb	840	300	300	15
Hg	57	17	17	0.85
Mo[b]	75	—	—	—
Ni	420	420	420	21
Se	100	36	100	5
Zn	7500	2800	2800	140

EQ, *exceptional-quality biosolids;* PC, *pollutant concentration biosolids*; CPLR, *cumulative pollutant loading rate biosolids;* APLR, *annual pollutant loading rate biosolids.*
[a]*Dry-weight basis.*
[b]*Pending EPA consideration.*

Recovery Act of 1976 and the Clean Water Act of 1977, the US EPA was established. The US EPA is the federal regulative authority responsible for regulating sludge management in the United States. Currently, the United States regulates land application of sludge at the federal, state (ie, the Pennsylvania Department of Environmental Resources guidelines), and local levels (PDER, 1997). The US EPA focuses on the use of sludge based on three end-use groups: as a soil amendment, a source of heat, and a source of other useful products including reclamation of disturbed lands from mining (USEPA, 1979). In 1983, the US EPA identified four related options for the safe disposal of sludge: agricultural use, land reclamation (ie, strip-mined lands, mine tailings, and other disturbed or marginal lands), forest use, and dedicated land use (USEPA, 1983). Furthermore, benefits (ie, reduction in commercial fertilizer use, retardation of soil erosion), environmental consequences (ie, risk of phytotoxicity from excess metals and salts), and remediation strategies (ie, sludge application rates, phytoremediation) have been identified by the US EPA's 1983 regulations (USEPA, 1983). Based on comprehensive requirements for the reduction of selected metals and pathogens in biosolids for land reclamation/or application, the US EPA's Part 503 regulation was developed (Table 3.2) (USEPA, 1994).

3.5 EFFECTS OF ADDITION OF BIOSOLIDS IN MINE SITE REHABILITATION

To date, extensive work has been carried out on the use of biosolids for the rehabilitation of mine sites. During the process of mine site rehabilitation using biosolids, direct (eg, decreasing bulk density,

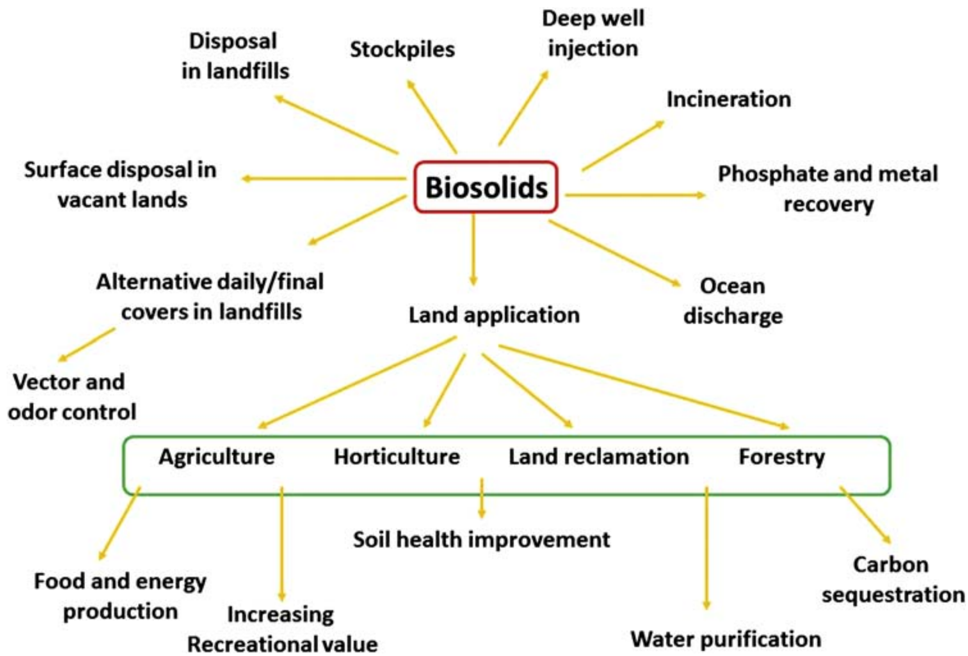

FIGURE 3.3

Various approaches to the utilization of biosolids.

avoiding soil erosion) and indirect (ie, improving chemical, physical, and biological properties) effects of receiving degraded soils could be achieved to restore a degraded environment. Fig. 3.3 illustrates various approaches to using biosolids, including mine spoil rehabilitation.

3.5.1 PHYSICAL CHARACTERISTICS

The addition of biosolids enhances most soil physical properties in degraded lands from mining. The high organic matter of biosolids is the main cause for improvement in physical properties in mine spoil soils. Improvements in physical properties including decreased bulk density and temperature, increased porosity and aggregation, increased hydraulic conductivity and water holding capacity, increased infiltration, maintained soil texture, and reduced erosion and sedimentation have been reported in mine spoil rehabilitation with biosolids (Brofas et al., 2000; Gardner et al., 2010; Sopper, 1992). The application of biosolids causes a reduction in bulk density in degraded soils by increasing pore space by enhancing macropores, mesopores, and micropores and developing soil texture, thereby increasing the field capacity (Jones et al., 2010, 2012). A remarkable increase in porosity and aggregate stability has been reported in acidic tailing rehabilitation with sewage sludge in Spain (Zanuzzi et al., 2009). A long-term study revealed that the application of biosolids caused short-term improvements but no lasting improvement in the physical properties of degraded soils (Bendfeldt et al., 2001).

3.5.2 CHEMICAL CHARACTERISTICS

The application of biosolids raises chemical properties such as pH, electrical conductivity (EC), CEC, nutrient contents, and organic matter in soils. Mostly, the low pH of degraded mine soils (ie, acidic tailing-rich soils) is raised with biosolids or sewage sludge application (Basta et al., 2001; Bendfeldt et al., 2001; Sopper, 1992). The high pH of alkaline mine spoils experiences decreased pH with biosolids application (Brofas et al., 2000; Jones et al., 2011). However, degraded land characteristics such as the oxidation of minerals over time can decrease in pH (Brofas et al., 2000; Sopper, 1992). As a result of an increase in soluble salts, the EC increases (Gardner et al., 2010; Jones et al., 2011; Mingorance et al., 2014). However, a reduction in EC after biosolids application has also been reported (Arocena et al., 2012). This may be the result of the leaching of soluble salts and the immobilization of metal ions. The presence of clay, mineral particles, and organic colloids in biosolids increases the CEC in the receiving degraded sites (Bendfeldt et al., 2001; Brofas et al., 2000; Gardner et al., 2010). The application of biosolids to mine spoils increases key plant nutrients such as N, P, Ca, and S, thereby increasing fertility (Bendfeldt et al., 2001; Brofas et al., 2000; Castillejo and Castello, 2010; Larney and Angers, 2012). To increase the supply of plant nutrients, biosolids combined with chemical fertilizers are also used in mine site rehabilitation (Li et al., 2013). Biosolids increase organic matter in spoil soils, accelerating mine site rehabilitation (Brofas et al., 2000; Castillejo and Castello, 2010; Larney and Angers, 2012). An increase in the immobilized or reduced concentrations of extractable metal ions in plant species has been reported from biosolids application with woody debris, paper sludge, and iron by-products at some superfund sites (Brown et al., 2003, 2004).

3.5.3 BIOLOGICAL CHARACTERISTICS

Biosolids application increases microbial biomass carbon and microbial enzymatic activities in receiving mine spoils (Gardner et al., 2010; Mingorance et al., 2014; Mora et al., 2005). Biosolids creates an energy-rich soil environment favorable to soil microorganisms by increasing soil organic matter, which is the main energy source of microbes. An increased amount of soil microorganisms such as denitrifiers, sulfate reducers, iron reducers, total aerobic microorganisms, and total anaerobic heterotrophs was observed in a mine site rehabilitated with biosolids in Canada (Gardner et al., 2010). Microbial enzyme concentrations of dehydrogenase, β-glucosidase, alkyl phosphatase, protease, and arylsulfatase increased after the addition of stabilized sewage sludge in a mine site (Mingorance et al., 2014). Furthermore, populations of fungi, bacteria, actinomycetes, and total microorganisms along with soil enzymatic activity have been reported with the application of sewage sludge combined with chemical fertilizers (Li et al., 2013). The addition of fresh sewage sludge was reported to result in a higher population density of earthworms in open-cast coal mine site rehabilitation (Emmerling and Paulsch, 2001). The mortality of earthworms increased owing to biosolids application in a mine site rehabilitation in New Zealand, which further suggests the need for ecological tests before implementing remediation strategies (Waterhouse et al., 2014). In a comprehensive case study, microbial functions (ie, CO_2 respiration and ammonia oxidation), plant diversity and metal uptake, earthworm (*Eisenia fetida*) toxicity, and deer mice (*Peromyscus maniculatus*) total body burden by selected heavy metals were reported with biosolids application to overburden and tailings (Brown et al., 2014). This study indicated no potential for ecosystem transfer of contaminants and showed high rates of recolonization of the restored sites by a range of small mammals owing to biosolids-enhanced rehabilitation.

Considering all of these physical, chemical, and biological changes with the use of biosolids, beneficial effects such as food and energy production by enhancing vegetation, nutrient cycling and carbon sequestration, water purification and pest/or disease control, and enhancement of recreational value are also achieved (Larney and Angers, 2012; Sopper, 1992). Compared with the beneficial effects, above- and below-ground carbon sequestration is identified as a major opportunity. Table 3.3 shows biosolids used in various vegetation sites in terms of carbon sequestration.

Large-scale mine sites are highly altered from their historical levels of biotic and abiotic integration, which make them unsuitable for the natural succession of historical species or direct planting of tree species. The concept of novel ecosystems, which is a new mixture of physical and biological components, including both native and nonnative species, is proposed as a feasible option for the rehabilitation of mine sites. As discussed in the previous section, after improving the physical, chemical, and biological properties of soil, such mine sites suitable for establishing novel ecosystems (Doley and Audet, 2013; Hobbs et al., 2009). The role of biosolids is important to reduce the time taken for restoration by improving soil biological and physicochemical properties. The application of biosolids provides levels of stability and functionality of an ecosystem (Doley and Audet, 2013; Li et al., 2013). In this section, we propose the land application of biosolids for soil rehabilitation and subsequent establishment of novel ecosystems as a feasible option for mine site rehabilitation and increasing carbon sequestration.

The early establishment of photosynthetic organisms in a bare land, such as lichens proceeded by bryophytes and annual grasses in a mine site, gradually enhances the thickness of the fine soil. As the natural succession proceeds, such lands will be suitable for tree establishment. For instance, in Estonia, long-term monitoring of the natural colonization of a pine stand on a reclaimed mine site showed that nearly 25 years is required to develop 6.2 cm thickness of fine soil with 4.38% of organic carbon (Laarmann et al., 2015). Moreover, 7-year-old stands of black alder, silver birch, and Scots pine with various planting densities sequestered 2563, 161, and 1899 kg ha^{-1} of aboveground biomass, respectively (Kuznetsova et al., 2011). That study further suggested black alder as a promising tree species for the reclamation of oil shale postmining areas in Estonia. Similarly, ecological assessments focusing on changes in the floristic compositions, impact from wildfires on the persistence of restored species, and factors affecting the plant growth in abandoned mine sites are available for major mining areas across the world, especially in Australia (Herath et al., 2009; Martínez-Ruiz et al., 2007; Ohsowski et al., 2012; Pallavicini et al., 2013; Vickers et al., 2012; Wiegleb and Felinks, 2001). However, integrated studies focusing on both land application of biosolids and restoration of the ecosystem as a whole are rare. Pallavicini et al. (2013) studied factors affecting herbaceous richness and biomass accumulation in reclaimed coal mines and found that pH, soil erosion severity, and soil nutrient availability are the key determinants. In tropical environments in which tree growth rates are faster, Tripathi et al. (2014) studied a mine spoil in India and showed that total carbon accumulation in total plant biomass, mine soil, and soil microbial biomass is 44.5, 22.9, and 1.8 Mg ha^{-1}, respectively. Therefore, afforestation followed by the land application of biosolids is a promising alternative because afforestation increases the net ecosystem carbon sequestration in reclaimed mine sites and enhance opportunities for other potential uses such as recreation. Therefore, we recommend the integration of land application of biosolids and establishment of novel ecosystems in mine sites as a feasible and more economical strategy for rehabilitation.

Table 3.3 Selected References for Biosolids Application to Various Vegetation Sites in Terms of Carbon Sequestration

Country or Region	Application Rate (Mg ha^{-1})	Tested Plants	Carbon Sequestration (Mg C^{-1} ha^{-1} year^{-1})	Remarks	References
United States	455–1654	Corn, wheat, sorghum, soybean, grass	1.73	• 8–23 years in rotation application of biosolids controlled the SOC (−0.07–0.17) • World's largest reclamation project with biosolids	Tian et al. (2009)
Australia	25 or 50	Indian mustard, sunflower	3.11–7.54 for mustard; 2.48–4.79 for sunflower	• Effect of biosolids application rate on carbon sequestration performed • Effect of plant species to SOC sequestration studied	Bolan et al. (2013)
United States	10 and 18		Biomass yields 35–40%	Degradation rate model applications identify 2 SOC phases in soil	Zhai et al. (2014)
United States	2.5, 5, 10, 21, and 30	Semiarid grassland	Soil total C% increased from 1.63 to 3.58 for short term biosolids application rate at 30 Mg year^{-1}	Long-term (13–14 years) and short-term (2–3 years) effects of biosolids application to soils 0–8 cm deep	Ippolito et al. (2010)
Spain	50		Mean increase 1.7 g kg^{-1}, peak increases 3.8 g kg^{-1}	• Sampled from ploughed layer (0–30 cm) • Short-term SOC pool in 60 contrasting agricultural soils	Soriano-Disla et al. (2010)
	4.2 kg m^{-2}		Increased crop residue C from 11.8% to 32.5%	• 13-year study • Crop residue C analyzed by ^{13}C technique	Tian et al. (2015)
Europe	1.0		SOC accumulation rate 0.49% year^{-1}	Maximum yearly carbon mitigation potential 2.65 Tg year^{-1}	Smith et al. (2000)
England	33		0.18	Organic C in biosolids, when applied to soil, is less decomposable than that in farm manures	Powlson et al. (2012)

3.6 CONCLUSIONS, CHALLENGES, AND FUTURE RESEARCH NEEDS

A large quantity of biosolids is generated as a consequence of increased human population and the subsequent expansion of wastewater treatment industries. Because of the presence of high amounts of organic matter and nutrients, biosolids are used extensively as a soil amendment or fertilizer for agricultural, land reclamation, and revegetation purposes. Biosolids enhance physical, chemical, and biological properties of degraded lands. Advantages associated with the use of biosolids in the remediation of mine sites are food and energy production through enhancing vegetation or afforestation, nutrients cycling and carbon sequestration, water purification and pest/or disease control, and enhancement of recreational and scientific discovery. However, a number of challenges are linked with biosolids land application, in particular mine site rehabilitation: groundwater and soil contamination from enhanced leaching of inorganic (ie, N and P) and organic nutrients, heavy metals, pathogens, and emerging contaminants (ie, pharmaceuticals and reactive nanomaterials). GHGs and odorous gaseous emission from biosolids stockpiles and during their handling (ie, land application) have been identified as contributing to air pollution and global warming. Therefore, a number of countries have developed regulations for the production and land application of biosolids. Most regulations focus on minimizing pathogens and environmental contamination from heavy metals and major inorganic nutrients. For example, chemical stabilization (ie, alkaline treatment) and pretreatment (ie, composting) are used to minimize the contaminants (ie, immobilization of heavy metals, disinfection of pathogen) during the production of biosolids. These options also ensure the quality of biosolids (ie, composting of biosolids with biodegradable wastes) and enhance their efficacy.

This chapter recognizes the challenges and future research needs that have to be addressed carefully to minimize negative environmental effects associated with biosolids:

- To minimize GHG emission and loss of carbon value from biosolids, minimizing decomposition of biosolids is an important factor to be considered. Options include minimizing the duration of biosolids stockpiles, applying efficient stabilizing agents, and/or using methods to convert easily degradable organic fractions into a more recalcitrant fraction, thereby minimizing microbial decomposition.
- To convert degraded mine sites as "future carbon storage sites," unique characteristics (ie, extremely low pH and diminished microorganisms in soils) associated with degraded lands and incorporation of biosolids in terms of increasing residual carbon and carbon sequestration from afforestation need to be studied further. Also, a scientific approach is needed to estimate the carbon sequestration resulting from biosolids use.
- To encourage biosolids use, issues such as the availability of insufficient quantities for remote areas or sites, the high cost of transportation, and the loss of nutrients or organic matter at field conditions need to be resolved.
- To recover the nutrients and their conservation from biosolids (ie, P), future research should be carried out.

ACKNOWLEDGMENTS

This research was partly supported by Australian Research Council Discovery Projects (DP140100323). This chapter is one of the outcomes of the research project on "Carbon sequestration from land application of biosolids." We thank South East Water (Dr. Aravind Surapaneni), Western Water (William Rajendran), Gippsland Water (Mark Heffernan), City West Water (Sean Hanrahan), Yarra Valley Water (Andrew Schunke), and Trans-pacific Industries (Chris Hetherington) for supporting this research project.

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