“Soil in the City: Papers,” *Journal of Environmental Quality*
Summary: Researchers at the MWRD have published studies in the Jan. 11, 2016 *Journal of Environmental Quality* which focus on biosolids. “Soil in the City: Sustainably Improving Urban Soils,” “Greening a Steel Mill Slag Brownfield with Biosolids and Sediments: A Case Study” and “Restoring Ecosystem Function in Degraded Urban Soil Using Biosolids, Biosolids Blend, and Compost” are among the papers in a special section “Soil in the City” in the recent publication.

Published January 11, 2016

Soil in the City: Sustainably Improving Urban Soils

Kuldip Kumar* and Lakhwinder S. Hundal

Abstract

Large tracts of abandoned urban land, resulting from the deindustrialization of metropolitan areas, are generating a renewed interest among city planners and community organizations envisioning the productive use of this land not only to produce fresh food but to effectively manage stormwater and mitigate the impact of urban heat islands. Healthy and productive soils are paramount to meet these objectives. However, these urban lands are often severely degraded due to anthropogenic activities and are generally contaminated with priority pollutants, especially heavy metals and polycyclic aromatic hydrocarbons. Characterizing these degraded and contaminated soils and making them productive again to restore the required ecosystem services was the theme of the "Soil in the City— 2014" conference organized by W-2170 Committee (USDA's Sponsored Multi-State Research Project: Soil-Based Use of Residuals, Wastewater, & Reclaimed Water). This special section of *Journal of Environmental Quality* comprises 12 targeted papers authored by conference participants to make available much needed information about the characteristics of urban soils. Innovative ways to mitigate the risks from pollutants and to improve the soil quality using local resources are discussed. Such practices include the use of composts and biosolids to grow healthy foods, reclaim brownfields, manage stormwater, and improve the overall ecosystem functioning of urban soils. These papers provide a needed resource for educating policymakers, practitioners, and the general public about using locally available resources to restore fertility, productivity, and ecosystem functioning of degraded urban land to revitalize metropolitan areas for improving the overall quality of life for a large segment of a rapidly growing urban population.

Core Ideas

- Urban soils are contaminated by priority pollutants due to anthropogenic activities.
- Urban soils quality can be improved by using local resources such as composts and biosolids.
- Improving soils is key to improving the overall ecosystem functioning in urban areas.
- These papers are a resource for policymakers, practitioners, and the general public.

J. Environ. Qual. 45:2–8 (2016)
doi:10.2134/jeq2015.11.0589
Received 30 Nov. 2015.
Accepted 17 Dec. 2015.
*Corresponding author (Kuldip.Kumar@mwrd.org).

Deindustrialization of metropolitan areas has resulted in large tracts of abandoned urban land. Renewed interest among city planners and community organizers has arisen to use this land productively for multiple purposes. Examples include, but are not limited to, growing plants for fresh food (urban farming), stormwater management, and mitigation of the urban heat island effect. There are several additional advantages and opportunities to improve the environment and ecology of cities by improving microclimate, restoring urban soils to provide ecosystem services, maximizing beneficial utilization of the waste generated in the cities, improving stormwater management, and enhancing biodiversity.

Furthermore, locavorism—the focus on eating locally grown foods—is gaining popularity in the United States and worldwide. This idea, however, seems counterintuitive to most people living in the urban setting who considered farming as an exclusively rural endeavor. Many countries in the world have already adopted urban agriculture as a result of the explosion of the urban population, especially in many Asian cities, mainly due to economic and political changes that have undermined the food distribution systems (Brown and Jameton, 2000). In addition to the value of urban agriculture leading to improved nutritional health, local economy, and food security in the United States, the value of green spaces in the urban setting is also being recognized by policymakers, health professionals, urban planners, environmental advocates, and the local community for improved personal wellness, environmental health benefits, and community betterment.

The renewed interest in urban farming in many metropolitan cities across the United States has resulted in greater attention to improving the quality of urban soils. Understanding the problems inherent to degraded urban soils is essential for bringing more urban soils into productive use for improving the quality of life for a large segment of a rapidly growing urban population.

Urban regions are centers of resource consumption and waste production, where raw materials are consumed directly or transformed into other products that either become part of the urban infrastructure or are discarded through one of many waste streams leaving urban regions for disposal. Material flows (e.g., food, raw materials, and water) in and out of metropolitan

Monitoring and Research Dep., Metropolitan Water Reclamation District of Greater Chicago, 6001 W. Pershing Rd., Cicero, IL 60804. Assigned to Technical Editor Tsutomu Ohno.

Abbreviations: GI, green infrastructure; PSI, P saturation index; WWTP, wastewater treatment plant.

regions, and transformation processes within, are complex pathways and are generally managed by different city agencies in isolation. As major metropolitan regions have adopted urban sustainability programs and policies, there has been an increasing awareness among city planners and local governmental agencies to the importance of addressing the sustainability linkages among these different resource domains simultaneously for more efficient resource use and recovery, as depicted in Fig. 1 (Brose et al., 2014). Recovered resources, such as municipal solid waste compost, biosolids, and harvested nutrients from wastewater, are available in urban areas and could be utilized for improving the fertility and ecosystem function of urban soils. Household composting and municipal-level solid waste composting are notable examples, where city waste is diverted from landfills and turned into compost, which is then used to amend soil to improve productivity as well as other ecosystem services. Another example is the supply of nutrients brought in the form of food to the cities not only from neighboring rural areas but also from around the world. A significant proportion of these nutrients are released in human feces and urine and sent to wastewater treatment plants (WWTPs) (Brose et al., 2014).

The main objective of the Soil in The City conference organized by the W-2170 Committee (USDA's Sponsored Multi-State Research Project Soil-Based Use of Residuals, Wastewater, & Reclaimed Water) was to educate practitioners and the general public on how to mitigate the risks from priority pollutants often found in urban soils and to reclaim and improve urban soils using local resources to grow health food and provide stormwater management benefits. This special section of *Journal of Environmental Quality* comprises 12 targeted papers authored by conference participants to make available much needed information about the characteristics of urban soils.

Soil Quality Assessment in Urban Areas

Contamination of urban soils by trace metals is a major concern because of the risk these elements pose to the environment and to human health. Due to the long residence time of heavy metals in soils, urban soils may act as both a sink and a source for these pollutants. Many of these trace metals may be present in parent material from where soils have developed and may be inherently high in some of these metals; however, in many cities, anthropogenic activities have resulted in the substantial contamination of urban soils (Mielke et al., 1983; Mitchell et al., 2014). The study conducted by Delbecque and Verdoodt (2016) shows that the concentration of trace metals in an urban environment was highly variable due to both diffuse and point sources of these contaminants. The main objective of the study was to reveal spatial patterns of anthropogenic heavy metal enrichment using an urban pollution index in the medium-sized city of Ghent, Belgium. The study focuses on eight heavy metals (As, Cd, Cr, Cu, Hg, Ni, Pb, and Zn), and the urban pollution index was developed based on a database of 2194 observations. The relationship, if any, between enrichment of these metals with land use and time since urbanization is also evaluated.

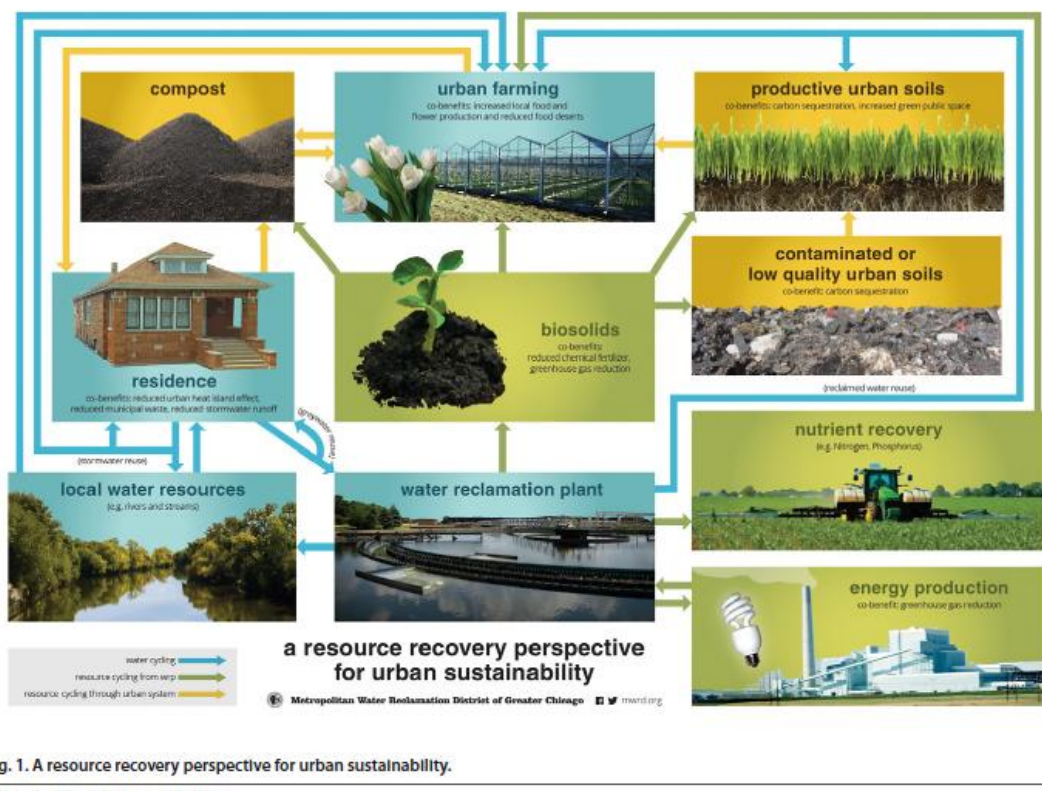

Fig. 1. A resource recovery perspective for urban sustainability.

Montgomery et al. (2016) present the results of a preliminary study conducted by a team of undergraduate students and community members to assess the soil quality of four abandoned residential lots located on the south side of Chicago with the aim of evaluating these vacant lots for appropriate land use, such as rain gardens, vegetable gardens, or playlots for children. All abandoned lots evaluated in this study exhibited the typical characteristics of an urban area in an industrial city and were littered with demolition debris, including glass, metal, bricks, concrete rubble, and other anthropogenic refuse. Results shows the soil quality in these lots was low, total lead concentrations were high with three out of four lots showing total lead concentrations exceeding the USEPA's threshold limit of 400 mg kg^{-1} for children's bare soil play areas, and maximum concentrations encountered in two out of four lots exceeded the USEPA's threshold limit of 1200 mg kg^{-1} in nonplay areas (USEPA, 2001). The results from this study further show that remediation may be necessary, depending on the intended use of vacant space to protect public health.

The urban lots study of Montgomery et al. (2016) and the citywide study of Delbecque and Verdoodt (2016) show that lead contamination of urban soils has been the result of anthropogenic activities, raising a major human health concern that may become a major obstacle for the adoption of urban agriculture in these contaminated soils. However, a critical review of both the direct (ingestion of contaminated soil) and indirect (consumption of food grown on lead contaminated soils) exposure pathways of lead conducted by Brown et al. (2016a) shows that although high concentrations of total lead may be present in urban soils, it is highly unlikely that urban agriculture would result in elevated blood lead levels in children dwelling in the urban areas. These authors argue that best management practices adopted in urban agriculture not only reduce both direct and indirect exposure of lead but also provide health, social, and environmental benefits.

The literature presented in Brown et al. (2016a) demonstrates that the overall potential for lead uptake by plants is relatively low, resulting in very low concentrations of lead in vegetables harvested from urban sites. Recent studies have also shown that despite the presence of very high concentrations of lead in urban soil (as high as 2000 mg kg^{-1}), a very small amount may be bioavailable because the majority of lead is often present in carbonate and phosphate fractions complexed with organic matter or adsorbed to iron oxides in soil and is thus not bioavailable. In addition, Brown et al. (2016a) show that only a small amount of lead that was ingested via food may be absorbed into the blood; for example, ingestion of 1 μg lead in food for a healthy child or a healthy adult will cause only 0.16 μg dL^{-1} and 0.04 μg dL^{-1} increases in blood lead level, respectively. They conclude that the benefits urban communities may realize from urban agriculture far exceed any risks posed by elevated lead in urban soils.

In general, it is believed that the addition of organic amendments and amendments containing soluble P reduces the bioaccessibility of lead by forming lead-phosphate minerals with very low solubility in soils; however, quantifying this reduction in bioaccessibility has been a challenge. Obrycki et al. (2016) tested three modified versions of USEPA Method 1340 to assess in vitro bioaccessibility of lead in two highly contaminated soils from Ohio that were treated with six phosphate amendments. Modifications to USEPA Method 1340 include varied pH (1.5 or 2.5) of extracting solution with and without glycine. Obrycki et al.'s (2016) results show that a modified USEPA Method 1340 without glycine at pH 2.5 had the potential to predict reductions in lead in vitro bioaccessibility resulting from the addition of various P amendments to the lead contaminated urban soils.

Recovering Resources and Restoring Ecosystem Functions of Urban Soils

The environmental impact of both the urbanization and the deindustrialization of urban areas is often noticed either in a reduction in soil ecosystem services, often due to compaction or stripping of surface soil (as in urban and suburban housing or commercial development on newly acquired rural land), or in the complete absence of soil ecosystem services due to degradation caused by industrial activity. Soil ecosystem services can be restored, depending on the ultimate intended use, for example, by (i) using organic amendments available in urban areas, such as composts and biosolids, to improve the spoil material or impacted soils that are unable to support any vegetation or any functional ecosystem; (ii) recovering nutrients, especially P from wastewater as a fertilizer, which may be used in urban agriculture; or (iii) using native prairie garden plants instead of traditional turf in urban landscape to improve soils and their ecosystem functioning related to greater stormwater infiltration and sequestration of carbon deeper in the profile.

Native Prairie Gardens to Improve Urban Soils

As Johnston et al. (2016) note, restoration of tallgrass prairie biome once dominant in the midwestern United States may help in ecologically engineering urban soils to improve their functioning. They hypothesized that residential prairie gardens would have better soil physical properties compared with turfgrass lawn and that well-structured soil under prairie gardens would therefore promote infiltration and mitigate stormwater runoff. Johnston et al. (2016) tested these effects comparing soil physical properties under paired prairie gardens and turfgrass lawn by taking advantage of a "natural" experiment in which homeowners introduced prairie gardens into typical residential landscapes in Madison, WI. Results showed the surface soil beneath prairie vegetation had 10% lower bulk density, 15% lower penetration resistance, 25% greater soil organic matter, and 33% greater saturated hydraulic conductivity compared with the adjacent lawns. Overall, the results from this study show that prairie gardens improve urban soils in the long-run when converted from the typical turfgrass type of landscape.

Beneficial Use of Biosolids, Dredged Sediments, and Recovered Nutrients from Wastewater Biosolids and Dredge Materials

The water reclamation process captures wastewater from domestic, industrial, and stormwater sources from across a metropolitan region and conveys it to a central treatment plant where it is processed to meet federal and state regulatory standards for discharge, usually, to a local waterway. The main products have traditionally been reclaimed water, which is discharged from the plant, and sewage sludge, which is generally land applied as biosolids, incinerated, or sent to landfills. Biosolids consist

of sewage sludge that has undergone further treatment to meet the USEPA's 40 CFR Part 503 regulatory criteria that permit its application to land as a nutrient rich soil amendment (USEPA, 1993).

According to a national survey, approximately 6.5 million dry tons of biosolids were produced in the United States, and about 60% of that amount was land applied (NEBRA, 2007); a majority of that amount was applied to agricultural fields outside of metropolitan regions. However, utilizing biosolids to amend contaminated or low-quality urban soils would provide nutrients and organic matter for urban agriculture and produce compost for use as a soil amendment, thus capturing and retaining nutrients from the metropolitan region. Urban soils are often compacted, unnatural soils that are low in nutrients and can be significantly improved by the addition of biosolids as a soil amendment. Biosolids improve the productivity of urban soils by increasing water infiltration and retention, reducing bulk density, improving structure, and increasing the total carbon stock of the soil (Brown et al., 2011; McIvor et al., 2012).

Due to these properties and their availability in metropolitan areas, biosolids are also considered an important resource to economically revegetate brownfields as Brose et al. (2016) demonstrate. The former US Steel Corporation's South Works site in Chicago, IL, is a 230-ha brownfield situated along the southern lakefront that needed to be reclaimed to support and sustain vegetation before development. The site consisted mostly of a deep heterogeneous fill of steel mill slag materials resulting from the former iron and steel operations. It was unable to support any vegetation and thus has been vacant since the 1970s; however, parks, residential, and commercial development are now planned for the site. The slag will need to be capped with topsoil for establishing turfgrass. The Chicago Park District estimated that up to 380,000 m^3 of topsoil will be needed for new parkland planned for the site, which would be cost-prohibitive. Thus, a more cost-effective alternative was needed. Many approaches were evaluated; one of the proposed approaches to defray the revegetation cost was to substitute topsoil with locally available dredged sediments from navigable waterways (Hundal et al., 2005). Dredged sediments were shown to support crops comparable to fertile farmland soil (Darmody et al., 2004); however, low organic matter content and poor structure resulted in surface crusting and sealing on drying when sediments were used alone for establishing vegetation. The addition of biosolids was shown to mitigate this problem and had a significant positive effect on soil organic carbon, total Kjeldahl nitrogen, total phosphorous, and microbial biomass and activity 1 yr after blending with sediments (Kelly et al., 2007). The Chicago Park District and the Metropolitan Water Reclamation District of Greater Chicago considered the use of biosolids blended with sediments for capping a portion of the former US Steel Corporation's site as an option for establishing parkland vegetation, and a case study was conducted to evaluate the suitability of biosolids and dredged sediments for capping the steel mill slag to establish good quality turfgrass vegetation (Brose et al., 2016). Overall, the results from this case study demonstrate that blends of biosolids and dredged sediments could be successfully used for capping steel mill slag brownfield sites to establish good quality turfgrass vegetation. This case study provides a qualitative assessment of using exceptional quality (EQ) biosolids blended into dredged sediments as an effective cap for establishing turfgrass on steel mill slag brownfields and, thus, potentially other marginal soils. Amending sediments with biosolids provided sufficient nutrients leading to the improved performance of turfgrass. The authors recommend a conservative rate of biosolids application (50% biosolids) in the sediment blend if there is groundwater within 3 m of the underlying soil profile or surface water in close proximity to the reclamation site to ensure that water quality is not affected by excess nutrients, but they suggest that higher rates could be used when groundwater is not shallow and surface water is not in close proximity to the reclamation site.

Dredged material, either alone or in combination with biosolids, has been beneficially used in many remediation projects, and many impacted landscapes have been reclaimed and brought under various productive agricultural or nonagricultural uses. Koropchak et al. (2016) also show that sediments may be beneficially used in agriculture and urban soil reconstruction. They note that more than 200 million m^3 of dredged material is available annually from the 40,000 km of waterways the Army Corps of Engineers maintains in the United States. Only 30% the dredged material is beneficially used for habitat development, aquaculture, beach nourishment, recreation, agriculture, mine reclamation, shoreline stabilization, and industrial use in construction (Brandon and Price, 2007). The traditional approach has been to ignore the fundamental agronomic parameters and to look at heavy metals and polycyclic aromatic hydrocarbons concentrations in the dredged sediments as a screening tool for making decisions on beneficial use (Koropchak et al., 2016). Based on extensive monitoring and research in the past 15 yr utilizing fresh water and saline dredged sediments Koropchak et al. (2016) propose that the most important primary and mandatory screening parameter should be acid-base accounting and that an acceptable secondary screening should be based on a combination of federal and state residual waste and soil screening standards. In addition, basic agronomic principles should be considered. Their proposed screening system separates the beneficial use of sediments for agriculture and urban soil reconstruction into three soil quality management categories of unsuitable, suitable, and clean fill, with different monitoring requirements.

In many deindustrialized cities, vacant urban land has been degraded by the loss of topsoil, contamination, and/or soil conditions such as salinity, acidity, or compaction or in short urban soils that may have lost their ecosystem function to support plant or microbial life. In general, the addition of organic soil amendments, such as biosolids, manure, and composts, restores soil ecosystem functioning. The work of Brose et al. (2016) clearly shows that by using biosolids and dredged sediments, it is possible to grow plants even on steel mill slag and that with time, fertility of the soil improves. Basta et al. (2016) address the use of biosolids and composts used successfully to improve soil ecosystem functioning of the Lake Calumet Cluster Site in Calumet, IL, a Superfund site impacted by heavy industry in the region. The authors evaluated four treatments: (i) biosolids at 202 Mg ha^{-1}, (ii) biosolids at 404 Mg ha^{-1}, (iii) compost at 137 Mg ha^{-1}, and (iv) a blend consisting of biosolids applied at 202 Mg ha^{-1}, drinking water treatment residual, and biochar. The amended soils were planted with a native mix of plants containing grasses, legumes, and forbs. Results of the study show that all soil amendments improved soil quality and nutrient pools, established a

dense and high quality vegetation cover, improved earthworm reproductive measures, and increased soil enzymatic activities that support soil function. Basta et al. (2016) note that overall, biosolids outperformed compost. Although several microconstituents (i.e., pharmaceuticals and personal care products) were detected in runoff, the concentrations were below the probable no-effect levels, demonstrating that the use of biosolids would not pose any impact on the aquatic environment. The authors recommend that the use of best management practices, such as runoff control measures to prevent sediment loss until the establishment of vegetation, would further help in addressing some of these concerns at the sites being restored.

Recovering Nutrients from Wastewater for Reuse

Improving the fertility of urban soils should not be a problem as cities are centers of resource consumption and as a result generate a wide variety of by-products (e.g., plant residues, kitchen waste, old newspapers, tree leaves, biosolids) that could be composted and applied to urban soils (Fig. 1, modified from Brose et al., 2014). A large amount of nutrients is brought to the cities in the form of food from neighboring rural areas and from around the world. A significant proportion of these nutrients is released in human feces and urine that are sent to WWTPs (Brose et al., 2014). On average, 0.6 kg P is excreted per year per person, and a major proportion (~58%) of this is excreted in the soluble form in urine (Kumar et al., 2012). High concentration of P in effluent discharged from WWTPs may cause eutrophication of surface waters (De-Bashan and Bashan, 2004; Parsons and Smith, 2008). Due to stringent regulations imposed on WWTPs, many municipalities have invested in P recovery technologies to harvest struvite, magnesium ammonium phosphate ($MgNH_4PO_4.6H_2O$) fertilizer. These technologies offer sustainable option for reducing P-loading in receiving waters as well producing biosolids with relatively favorable P/N ratio. In their study, Venkatesan et al. (2016) modeled the feasibility of P recovery from a typical WWTP serving a population of 160,000 in Arizona. Modeling results showed that about 71 to 96% of the P being lost in effluent discharge could be recovered, resulting 491 ± 64 t yr^{-1} of struvite fertilizer. The amount of P recovered from this WWTP could fertilize about 2000 ha of agricultural land. The study conducted by Venkatesan et al. (2016) shows that there is a potential of recovering between 20 and 50% of excreted P in the form of struvite; the process was projected to be economically feasible for WWTPs with a payback period of ~3 yr. Furthermore, for every 1 t struvite production, approximately 10 t CO_2 equivalent emissions would be offset compared with conventional fertilizer production. Thus, not only can the nutrients be recovered from the waste right where it is generated, but they can also be utilized where the demand is, reducing the import of nutrients from distant places.

Benefits of Urban Green Spaces

Thermal Comfort

Greening of the urban land vacated due to the deindustrialization of metropolitan areas may provide several environmental benefits. One such benefit is improvement of the thermal comfort condition of city residents during peak summer months by mitigating the urban heat island effect. The study conducted by Brandani et al. (2016) provides useful information on thermal regimes of urban soils and surfaces and demonstrates that exposed surfaces became less heated if their albedo was high, which led to significant reduction in surface temperatures even under the sun's direct exposure. The authors compared four different surfaces, showing that green surfaces were always cooler than asphalt, gray sandstone, and white gravel, as indicated by lower daytime surface and air temperatures. Thus, replacing impervious land surfaces with green groundcover is important to improving thermal comfort during the peak summer months in the city, and urban planners and policymakers must take heed of this while developing urban transformations and renovation plans.

Urban Soils and Stormwater Management

Development and urbanization have altered the drainage system of most metropolitan areas by increasing the impermeable surfaces at the cost of green permeable surfaces, resulting in greater volumes of stormwater runoff and flashier storm peaks, which overwhelm the capacity of combined sewers and cause localized flooding, flow surge to the downstream WWTPs, and combined sewer overflows to receiving waters (Kumar et al., 2016). Recently, the focus has been shifting from "end-of-pipe" type traditional drainage systems to more sustainable drainage systems often referred to as "green infrastructure" (GI) for managing stormwater runoff. The general principle behind the idea of GI technologies is simply "collect, treat, and freely infiltrate stormwater to recharge groundwater" such that the stormwater bypasses the collection system sewers. In comparison to traditional drainage systems, GI technologies, like bioretention systems (rain gardens, bioswales, planter boxes), are deemed sustainable and are often cost-effective for urban areas (Center for Neighborhood Technology, 2010).

These systems use soil to enhance stormwater (runoff from surrounding impervious surfaces) infiltration into the soil. The soils used in the bioretention systems, while supporting plant growth, must also be capable of rapid water infiltration and have high retention or filtration capacity for stormwater pollutants and reducing pollutants being conveyed to surface water bodies via WWTPs, hence reducing the impact of the "first flush" effect that is commonly associated with urban runoff (Rajapakse and Ives, 1990; Andersen et al., 1999; Kumar et al., 2016). Urban stormwater may contain a wide range of contaminants, including particulates, nutrients, metals, and organic matter like fats, oils, and grease. In general, the focus has been on heavy metals and nutrients as contaminants in urban runoff, and the reported concentrations have been typically <1 mg L^{-1} and <2 mg L^{-1}, respectively (Brown et al., 2016b). Removal efficiencies or retention of these pollutants will depend on the characteristics of soils used in the GI retention systems. Brown et al. (2016b), referring to a review of soil specification from 16 states by Carpenter and Hallam (2010), indicate that composition of soils used in GI systems varies from 30 to 60% sand, compost ranging from 20 to 40%, and top soil ranging from 20 to 30%.

In their study, Brown et al. (2016b) propose that the P saturation index (PSI) could be used as a tool to evaluate whether the soil mixes used in GI systems could become a sink or source of nutrients, like N and P, and also of metals, like Zn and Cu. The PSI was developed as a predictive tool to determine the potential

of P leaching from soils amended with organic residuals like manures and biosolids (Elliott et al., 2002; Agyin-Birikorang and O'Connor, 2007) and is based on a strong correlation between the ratio of total P to Fe and Al oxides as determined by oxalate extraction (PSI = $P_{ox}/(Fe_{ox} + Al_{ox})$ and the P found in leachate. Brown et al. (2016b) evaluated three different composts adjusted with Fe-based drinking water treatment residuals and P salts to PSI values of 0.1, 0.5, and 1.0 on nutrient and metal leaching using a synthetic stormwater solution and also evaluated plant performance. Results indicate that compost from manure/sawdust performed poorly in terms of plant performance. The PSI proved to be an effective tool to predict P movement in GI soils. Although all compost materials tested showed high contaminant removal, removal of metals was much higher when PSI of the soil mix was low.

If, however, the soil mixes show a high contaminant removal or, in other words, the metal contaminants are retained in the soil mix, an important question arises as to whether the concentration of metals in the GI soil will exceed the threshold for human health impacts of urban soil remediation standards due to the long residence time of metals in soils. This may have important implications for the people who manage these systems or the public who may come in contact with these systems. Thus, it is important to study the accumulation of elements of concern in GI soils, as pointed out by Kondo et al. (2016). These authors rightly point out that most GI projects are located on public or institutional lands, such as street right-of-ways, in parks, or school grounds and that due to their location, it is important to characterize GI soils in relation to human exposure and health risks. Kondo et al.'s (2016) study provides a unique and much needed evaluation of soil elemental concentrations in GI projects constructed over a decade in Philadelphia, PA. Soil elemental concentrations, categorized as macro- and micronutrients, heavy metals, and other elements at 59 GI sites and soil samples 3 to 5 m upland of these sites were compared. The comparisons were adjusted for the age of GI, underlying soil type, street drainage, and surrounding land use. The results indicate only calcium and iodine concentrations were significantly higher than background levels at GI sites, which might be the result of winter deicing salt from road runoff to GI soils. These elements do not pose a human health risk.

Summary and Recommendations

The collection of papers published in this special section highlights the need to educate the public and policymakers about urban soil quality. Characterization of urban soils in relation to priority pollutants is the first step for decision making for their intended use to provide various ecosystem services. Results presented in these papers clearly indicate that although urban soils may be degraded and contaminated, the soils could be improved or restored to provide various ecosystem services by simply amending them with locally available resources such as composted municipal solid waste, dredged sediments, and wastewater treatment residuals like biosolids and also by reusing nutrients (e.g., struvite fertilizer) recovered from wastewater. These kinds of locally available resources, once seen as waste materials, are becoming valuable for improving the quality of urban soils in a more sustainable manner. From the presentations made at the "Soil in The City—2014" conference held in Chicago, and from this extraordinary collection of papers, we highlight the following recommendations:

- The industrial revolution during the early 20th century and deindustrialization of urban cities in the last three decades throughout the world left a legacy of heavy metal contamination in the urban soils. There is an urgent need of establishing an urban heavy metal accumulation baseline not only for developing ameliorative approaches but also for monitoring and evaluating future changes in urban soil quality.
- Although an elevated level of lead could be present in many urban soils, only a fraction of it is bioaccessible and may not pose any elevated risk for adverse human health. The best management practice of adding organic amendments in urban gardens to grow crops under adequate soil nutrient levels has been shown to reduce the bioavailability and bioaccessibility of soil lead. It is now widely accepted as a remediation method for urban soils.
- Modified USEPA Method 1340 without glycine and extracting solution pH of 2.5 has the potential to predict reductions in lead bioavailability resulting from the addition of various P amendments to the lead-contaminated urban soils.
- Planting prairie gardens may improve urban soils in the long term when converted from typical turfgrass, and urban soils may provide improved ecosystem services including higher carbon sequestration and improved stormwater management in urban landscapes.
- Biosolids, compost, and dredged sediments are important and sustainable locally available resources to improve degraded urban soils. In addition, mixtures of biosolids and dredged sediments can be used for greening brownfields.
- Nutrients may be recovered from wastewater right where it is generated and also utilized where the demand is, reducing the import of nutrients from distant places. There is potential of recovering a significant proportion of excreted P in the form of struvite from WWTPs if enhanced biological P removal coupled with P recovery is adopted.
- Greening/farming of the urban land vacated due to deindustrialization of metropolitan areas may improve the thermal comfort condition of city residents during peak summer heat waves by mitigating the urban heat island effect.
- Characterizing soils used in the green infrastructure stormwater management projects is important to ensure that they act as a sink for legacy pollutants received in runoff from the catchment areas. The PSI of different soils amended with compost and biosolids was shown to be an effective tool not only to predict P movement in green infrastructure soils but also to retain heavy metals. Most green infrastructure projects for stormwater management are located on public or institutional lands, such as street right-of-ways, in parks, or school grounds. Due to their location, it is important to characterize soils in relation to human exposure and health risks.

Conclusions

There is a need to develop the local knowledge and skills for managing and improving urban soils so that the urban soils provide the required ecosystem functioning. These skills must integrate knowledge from agronomic, ecological, environmental, economic, and social sciences for increasing urban food production, greening of urban landscape, and green infrastructure for stormwater management with the overarching objective of not only improving the nutrition and health of urban populations but also improving the overall environment and living conditions of urban communities.

References

Agyin-Birikorang, S., and G.A. O'Connor. 2007. Lability of drinking-water treatment residuals (WTR) immobilized phosphorus: Aging and pH effects. J. Environ. Qual. 36:1076–1085. doi:10.2134/jeq2006.0535

Andersen, C.T., I.D.L. Foster, and C.J. Pratt. 1999. The role of urban surfaces (permeable pavements) in regulating drainage and evaporation: Development of a laboratory simulation experiment. Hydrol. Processes 13:597–609. doi:10.1002/(SICI)1099-1085(199903)13:4<597::AID-HYP756>3.0.CO;2-Q

Basta, N.T., D.M. Busalacchi, L.S. Hundal, K. Kumar, R.P. Dick, R.P. Lanno, J. Carlson, A.E. Cox, and T.C. Granato. 2016. Restoring ecosystem function in degraded urban soil using biosolids, biosolids blend, and compost. J. Environ. Qual. 45:74–83. doi:10.2134/jeq2015.01.0009

Brandani, G., M. Napoli, L. Massetti, M. Petralli, and S. Orlandini. 2016. Urban soil: Assessing ground cover impact on surface temperature and thermal comfort. J. Environ. Qual. 45:90–97. doi:10.2134/jeq2014.12.0521

Brandon, D.L., and R.A. Price. 2007. Summary of available guidance and best practices for determining suitability of dredged material for beneficial uses. USEPA ERDC/EL TR-O7-27. http://el.erdc.usace.army.mil/elpubs/pdf/trel07-27.pdf (accessed 2 Nov. 2015).

Brose, D.A., L.S. Hundal, O.O. Oladeji, K. Kumar, T.C. Granato, A. Cox, and Z. Abedin. 2016. Greening a steel mill slag brownfield with biosolids and sediments: A case study. J. Environ. Qual. 45:53–61. doi:10.2134/jeq2015.09.0456

Brose, D.A., K. Kumar, L.S. Hundal, A. Cox, H. Zhang, and T.C. Granato. 2014. A resource recovery perspective for improving the productivity of urban soils. Poster presented at: 2014 Soil in the City Conference, Chicago, IL. 29 June–2 July, 2014. USDA W-3170 Committee, Chicago, IL.

Brown, K.H., and A.L. Jameton. 2000. Public health implications of urban agriculture. J. Public Health Policy 21:20–39. doi:10.2307/3343472

Brown, S., R.L. Chaney, and G.M. Hettiarachchi. 2016a. Lead in urban soils: A real or perceived concern for urban agriculture? J. Environ. Qual. 45:26–36. doi:10.2134/jeq2015.07.0376

Brown, S., A. Corfman, K. Mendrey, K. Kurtz, and F. Grothkopp. 2016b. Stormwater bioretention systems: Testing the phosphorus saturation index and compost feedstocks as predictive tools for system performance. J. Environ. Qual. 45:98–106. doi:10.2134/jeq2014.10.0414

Brown, S., K. Kurtz, A. Bary, and C. Cogger. 2011. Quantifying benefits associated with land application of organic residuals in Washington state. Environ. Sci. Technol. 45:7451–7458. doi:10.1021/es2010418

Carpenter, D.D., and L. Hallam. 2010. Influence of planting soil mix characteristics on bioretention cell design and performance. J. Hydrol. Eng. 15:404–416. doi:10.1061/(ASCE)HE.1943-5584.0000131

Center for Neighborhood Technology. 2010. The value of green infrastructure. A guide to recognizing its economic, environmental and social benefits. http://www.cnt.org/publications/the-value-of-green-infrastructure-a-guide-to-recognizing-its-economic-environmental-and (accessed 2 Nov. 2015).

Darmody, R.G., J.C. Marlin, J. Talbott, R.A. Green, E.F. Brewer, and C. Stohr. 2004. Dredged Illinois River sediments. J. Environ. Qual. 33:458–464. doi:10.2134/jeq2004.4580

De-Bashan, L.E., and Y. Bashan. 2004. Recent advances in removing phosphorus from wastewater and its future use as fertilizer (1997–2003). Water Res. 38:4222–4246. doi:10.1016/j.watres.2004.07.014

Delbecque, N., and A. Verdoodt. 2016. Spatial patterns of heavy metal contamination by urbanization. J. Environ. Qual. 45:9–17. doi:10.2134/jeq2014.11.0508

Elliott, H.A., G.A. O'Connor, and S. Brinton. 2002. Phosphorus leaching from biosolids-amended sandy soils. J. Environ. Qual. 31:681–689. doi:10.2134/jeq2002.0681

Hundal, L.S., A.E. Cox, P. Lindo, G. Tian, T.C. Granato, and B. Sawyer. 2005. Use of biosolids for establishing vegetation on a steel mill slag brownfield in Chicago USX site: A research and demonstration project. Rep. 05-6. Monitoring and Research Department, Metropolitan Water Reclamation District of Greater Chicago, Chicago, IL.

Johnston, M.R., N.J. Balster, and J. Zhu. 2016. Impact of residential prairie gardens on the physical properties of urban soil in Madison, Wisconsin. J. Environ. Qual. 45:45–52. doi:10.2134/jeq2015.02.0093

Kelly, J.J., E. Favila, L.S. Hundal, and J.C. Marlin. 2007. Assessment of soil microbial communities in surface applied mixture of Illinois River sediments and biosolids. Appl. Soil Ecol. 36:176–183. doi:10.1016/j.apsoil.2007.01.006

Kondo, M.C., R. Sharma, A.F. Plante, Y. Yang, and I. Burstyn. 2016. Elemental concentrations in urban green stormwater infrastructure soils. J. Environ. Qual. 45:107–118. doi:10.2134/jeq2014.10.0421

Koropchak, S.C., W.L. Daniels, A. Wick, G.R. Whittecar, and N. Haus. 2016. Beneficial use of dredge materials for soil reconstruction and development of dredge screening protocols. J. Environ. Qual. 45:62–73. doi:10.2134/jeq2014.12.0529

Kumar, K., A. Cox, and H. Zhang. 2012. Phosphorus in service areas of the Stickney, Calumet, and north side water reclamation plants: Tracking sources and feasibility of recovery to meet future limits. Rep. 12-44. Monitoring and Research Department, Metropolitan Water Reclamation District of Greater Chicago, Chicago, IL.

Kumar, K., J. Kozak, L. Hundal, A. Cox, H. Zhang, and T. Granato. 2016. In-situ infiltration performance of different permeable pavements in an employee used parking lot: A four-year study. J. Environ. Manage. 167:8–14. doi:10.1016/j.jenvman.2015.11.019

McIvor, K., C. Cogger, and S. Brown. 2012. Effects of biosolids based soil products on soil physical and chemical properties in urban gardens. Compost Sci. Util. 20:199–206. doi:10.1080/1065657X.2012.10737049

Mielke, H.W., J.C. Anderson, K.J. Berry, P.W. Mielke, R.L. Chaney, and M.L. Leech. 1983. Lead concentrations in inner city soils as a factor in the child lead problem. Am. J. Public Health 73:1366–1369. doi:10.2105/AJPH.73.12.1366

Mitchell, R.G., H.M. Spliethoff, L.N. Ribaudo, D.M. Lopp, H.A. Shayler, L.G. Marquez-Bravo, V.T. Lambert, G.S. Ferenz, J.M. Russell-Anelli, E.B. Stone, and M.B. McBride. 2014. Lead (Pb) and other metals in New York City community garden soils: Factors influencing contaminant distributions. Environ. Pollut. 187:162–169. doi:10.1016/j.envpol.2014.01.007

Montgomery, J.A., C.A. Klimas, J. Arcus, C. DeKnock, K. Rico, Y. Rodriguez, K. Vollrath, E. Webb, and A. Williams. 2016. Soil quality assessment is a necessary first step for designing urban green infrastructure. J. Environ. Qual. 45:18–25. doi:10.2134/jeq2015.04.0192

North East Biosolids and Residuals Association (NEBRA). 2007. A national biosolids regulation, quality, end use, and disposal survey: Final report. www.nebiosolids.org/s/NtlBiosolidsReport-20July07-mvqy.pdf (accessed July 2014).

Obrycki, J.F., N.T. Basta, K. Scheckel, B.N. Stevens, and K.K. Minca. 2016. Phosphorus amendment efficacy for in situ remediation of soil lead depends on the bioaccessible method. J. Environ. Qual. 45:37–44. doi:10.2134/jeq2015.05.0244

Parsons, S.A., and J.A. Smith. 2008. Phosphorus removal and recovery from municipal wastewaters. Elements 4:109–112. doi:10.2113/GSELEMENTS.4.2.109

Rajapakse, J.P., and K.J. Ives. 1990. Pre-filtration of very highly turbid waters using pebble matrix filtration. J. Inst. Water Environ. Manage. 4:140–146. doi:10.1111/j.1747-6593.1990.tb01569.x

USEPA. 1993. 40 CFR Part 503, Standards for the use or disposal of sewage sludge: Final rules. Fed. Reg. 58(32):9248–9415.

USEPA. 2001. 40 CFR Part 745, Lead: Identification of dangerous levels of lead: Final rules. Fed. Reg. 66(4):1206–1240.

Venkatesan, A.K., A.-H.M. Hamdan, V.M. Chavez, J.D. Brown, and R.U. Halden. 2016. Mass balance model for sustainable phosphorus recovery in a US wastewater treatment plant. J. Environ. Qual. 45:84–89. doi:10.2134/jeq2014.11.0504

Published January 11, 2016

Journal of Environmental Quality

SPECIAL SECTION

SOIL IN THE CITY

Greening a Steel Mill Slag Brownfield with Biosolids and Sediments: A Case Study

Dominic A. Brose, Lakhwinder S. Hundal,* Olawale O. Oladeji, Kuldip Kumar, Thomas C. Granato, Albert Cox, and Zainul Abedin

Abstract

The former US Steel Corporation's South Works site in Chicago, IL, is a 230-ha bare brownfield consisting of steel mill slag fill materials that will need to be reclaimed to support and sustain vegetation. We conducted a case study to evaluate the suitability of biosolids and dredged sediments for capping the steel mill slag to establish good quality turfgrass vegetation. Eight study plots were established on a 0.4-ha parcel that received biosolids and dredged sediment blends of 0, 25, 50, or 100% biosolids (v/v). Turfgrass was successfully established and was thicker and greener in biosolids-amended sediments than in unamended sediments. Concentrations of N, P, K, and micronutrients in turfgrass tissues increased with increasing biosolids. Soil organic carbon, N, P, and micronutrients increased with increasing biosolids. Cadmium, Cu, Ni, and Zn concentrations in biosolids-amended sediments also increased with increasing biosolids but were far below phytotoxicity limits for turfgrass. Lead and Cr concentrations in biosolids-amended plots were comparable to concentrations in unamended sediments. Groundwater monitoring lysimeters and wells below the study site and near Lake Michigan were not affected by nutrients leaching from the amendments. Overall, the results from this case study demonstrated that blends of biosolids and dredged sediments could be successfully used for capping steel mill slag brownfield sites to establish good quality turfgrass vegetation.

Core Ideas

- Biosolids and sediment blends provide ideal growing medium for greening steel mill slag brownfield sites.
- Use of biosolids for greening steel mill slag sites has no impact on the surface and groundwater quality.
- Turfgrass performs better in biosolids-amended plots.

J. Environ. Qual. 45:53–61 (2016)
doi:10.2134/jeq2015.09.0456
Received 9 Sept. 2015.
Accepted 19 Sept. 2015.
*Corresponding author (lakhwinder.hundal@mwrd.org).

The former US Steel Corporation's South Works site in Chicago, IL, is a 230-ha brownfield situated along the southern lakefront that will need to be reclaimed to support and sustain vegetation before development. The site consists mostly of a deep (up to 40 m) heterogeneous fill of steel mill slag materials resulting from the former iron and steel operations. The steel mill slag material is incapable of sustaining vegetation, and the site has been vacant since the 1970s (Fig. 1); however, parks and residential and commercial development are planned for the site. The slag will need to be capped with topsoil (~0.3-m-thick layer to support healthy root growth) for establishing turfgrass. The Chicago Park District estimates that up to 380,000 m^3 of topsoil will be needed for new parkland planned for the site, which would be cost prohibitive. Thus, a more cost-effective alternative is needed.

One approach to defraying revegetation cost is to substitute topsoil with locally available dredged sediments from navigable waterways. Dredging navigable waterways is essential for keeping them functional, but the hauling and disposal of dredged sediments can be costly. Dredged sediments were shown to support crops comparable to fertile farmland soil (Darmody et al., 2004); however, low organic matter content and poor structure resulted in surface crusting and sealing upon drying when sediments alone were used for establishing vegetation. The addition of biosolids was shown to mitigate this problem and to have a significant positive effect on soil organic carbon (OC), total Kjeldahl nitrogen (TKN), total phosphorous (TP), and microbial biomass and activity 1 yr after blending with sediments (Kelly et al., 2007).

Biosolids are processed sewage sludge that undergo further treatment to meet the USEPA's requirements that permit their application to land as a nutrient-rich soil amendment (USEPA, 1993). Biosolids have been shown to increase soil productivity by improving soil structure, decreasing bulk density of compacted soils, and increasing soil organic matter and nutrient content (Brown et al., 2011; McIvor et al., 2012). Blending locally available dredged sediments with biosolids could be an economical option as a soil amendment for reclaiming brownfields and revitalizing urban areas (Kelly et al., 2007).

Monitoring and Research Dep., Metropolitan Water Reclamation District of Greater Chicago. Assigned to Technical Editor Tsutomu Ohno.

Abbreviations: EC, electrical conductivity; EQ, exceptional quality; MWRDGC, Metropolitan Water Reclamation District of Greater Chicago; OC, organic carbon; TKN, total Kjeldahl nitrogen; TP, total phosphorous.

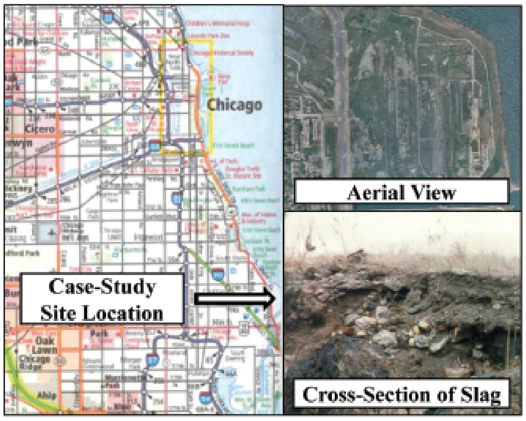

Fig. 1. Location of steel mill slag case-study site (left), aerial photo of site (top right), and cross-sectional view of slag material (bottom right).

Previous studies blending dredged sediments with nutrient-rich biosolids have demonstrated improved tilth and fertility of sediments (Kelly et al., 2007; Sigua, 2009; Ruiz Diaz et al., 2010). Biosolids were shown to increase the fertility of dredged sediments in a greenhouse study and to improve barley (*Hordeum vulgare* L.) and snap bean (*Phaseolus vulgaris* L.) growth (Ruiz Diaz et al., 2010). Biosolids were also shown to immobilize trace metals, rendering them less bioavailable and thus reducing uptake from sediments by plants. The reduced bioavailability of heavy metals in contaminated soils by the addition of biosolids has been demonstrated due to immobilization of heavy metals by Fe and Al oxides present in biosolids (Basta et al., 2001; Brown et al., 2003; Brown et al., 2012, Navarro, 2012).

To our knowledge, blending biosolids into dredged sediments for capping and re-vegetating steel mill slag brownfields has not been demonstrated at the field scale. The Chicago Park District and the Metropolitan Water Reclamation District of Greater Chicago (MWRDGC) considered the use of biosolids blended with sediments for capping a portion of the former US Steel Corporation's site in Chicago, IL, as an option for establishing parkland vegetation. The MWRDGC conducted a case study to evaluate and demonstrate the feasibility of this option. Biosolids were blended into dredged sediment at increasing rates (0–100%, v/v) and were evaluated for establishment and performance of turf vegetation, soil fertility, and impact on surface and groundwater quality.

Materials and Methods

Case Study Design

The case study was designed to compare turfgrass performance and soil fertility over a 4-yr period in field-scale plots amended with four rates of biosolids and dredged sediment blends: 0, 25, 50, and 100% biosolids (v/v). The plots were established on an approximately 0.4-ha parcel of slag at the US Steel Corporation's South Works site in Chicago, IL (Fig. 2). Briefly, four plots were established on the eastern and western sections of the parcel and were separated by a 1.2-m-high berm. On the eastern section, a 0.15-m layer of silty clay loam soil was placed on the slag

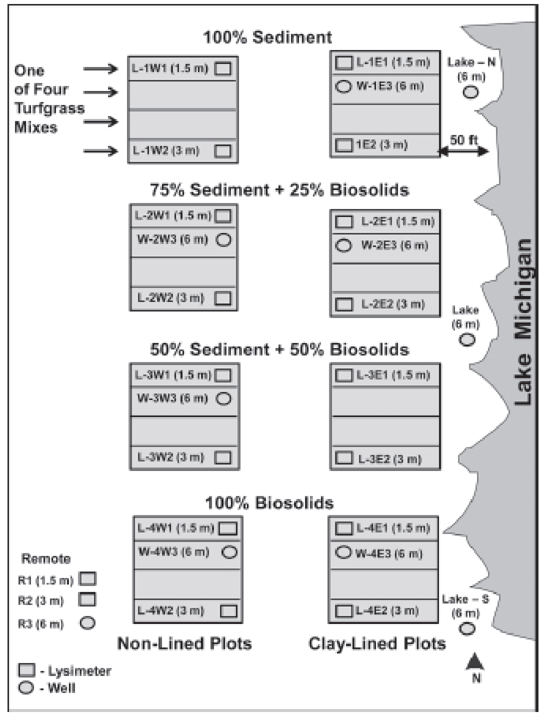

Fig. 2. Layout of plots, lysimeters, and wells in the case study.

to regulate water movement and to simulate a heavy-textured B-horizon found in many Midwestern soil profiles before placing the sediments and biosolids-amended sediments (clay-lined plots). On the western section, the biosolids-amended sediments were placed directly on the slag (nonlined plots).

Before spreading the biosolids-amended sediments, two suction lysimeters (1.5 and 3 m deep) and one well (6 m deep) were installed in each plot and at a remote location approximately 60 m southwest of the plots for the purpose of monitoring subsurface water and groundwater quality (Fig. 2). The depth of 6 m below the slag surface was selected to represent the static water level of Lake Michigan. Due to frequent obstructions encountered during drilling, wells were not installed in the eastern 50% biosolids-amended and western 0% biosolids-amended plots. An additional three wells (6 m deep) were installed east of the plots and approximately 15 m offshore of Lake Michigan to monitor the quality of leachate from the slag material.

Biosolids and Sediment

Exceptional quality (EQ), anaerobically digested, lagoon-aged, and air-dried biosolids from the MWRDGC's Calumet Water Reclamation Plant and dredged sediments from a local waterway were used. The chemical analysis of the biosolids, sediment, and slag is presented in Table 1. The biosolids and sediment were blended in situ on a vol/vol basis using front loaders and were applied as a cap on the surface of slag material to achieve an approximately 0.3-m uniform layer using the same equipment. A 1.8-m-wide berm and a 0.3-m-high berm were constructed around the entire plot area to control surface runoff leaving the plots due to close proximity to Lake Michigan. A 1-m-wide sod

Table 1. Chemical characteristics of biosolids, dredged sediments, and steel mill slag.

Parameter†	Biosolids	Sediment	Slag
pH	6.3	6.6	7.6
		dS m^{-1}	
EC	2.9	0.40	0.60
		g kg^{-1}	
Organic C	160	21	ND‡
TKN	16	1.7	0.68
NH_4–N	0.13	3.0×10^{-4}	3.0×10^{-5}
NO_2–N + NO_3–N	0.29	5.0×10^{-3}	0.01
Total P	24	0.67	0.50
K	1.6	ND	1.1
SO_4–S	3.3	0.28	3.0×10^{-3}
Cd	0.01	$<2.0 \times 10^{-4}$	0.01
Cr	0.15	0.30	0.22
Cu	0.30	0.03	0.38
Fe	24	19	61
Mn	0.39	0.50	6.6
Mo	0.01	$<3.0 \times 10^{-4}$	0.02
Ni	0.03	0.02	0.07
Pb	0.13	0.02	0.37
Zn	1.4	0.09	0.81

† EC, electrical conductivity; TKN, total Kjeldahl nitrogen.

‡ Not determined.

buffer was placed between the main plots to minimize runoff and cross-contamination between plots. During the period of study, weed control in the plots was periodically conducted by manually pulling out weeds and spraying commercial herbicides.

Turfgrass

Four turfgrass blends commonly used in the Chicago area (Swink and Wilhelm, 1994) were planted in one of four subplots within each of the eight main plots (0, 25, 50, and 100% biosolids in clay-lined and nonlined plots) in the fall of year 1 of the study (Fig. 2; Table 2). Similar to the National Turfgrass Evaluation Program, performance was evaluated using scores based on turf density and color. Turf density was measured in each of the subsequent 3 yr (years 2–4) by visually determining scores ranging from 0 to 100, which were assigned based on turf thickness, occurrence of bare spots, and density of weeds: the higher the number of bare spots and the greater the density of weeds, the lower the score. The highest scores were assigned to plots with dense turf, no bare spots, and minimal weeds.

Turf color was measured using the Munsell color chart for plant tissues, which defines colors by assigning a code for the hue (the predominant basic color) and a ratio of two numbers (e.g., 7/8) to represent the value (lightness or darkness) and chroma (color intensity). Turfgrass color ranged from light green (e.g., 7.5 GY 7/8 to 7/10) to dark green (e.g., 7.5 GY 5/4 to 5/8) and was assigned scores ranging from 0 to 100. The dark green color was assigned the highest score, and the scoring decreased as the turfgrass color approached pale yellow.

The overall performance evaluation scores for turfgrass were calculated by using a weighted sum of the turf density and color scores. Weighting factors of 0.75 and 0.25 were assigned to turf density and color, respectively, to give more weight to turf density because it is a key characteristic that turfgrass managers seek. The overall performance scores were calculated as follows: Overall Performance Score = Density Score × 0.75 + Color Score × 0.25. The turfgrass performance was rated based on the overall performance scores as follows: excellent, >60; well, 46 to 60; poor, 30 to 45; very poor, <29.

Turfgrass Sampling and Analysis

Turfgrass tissue samples were collected in the fall of each of the three consecutive years by clipping four 0.1-m^2 areas from each subplot to a height of 0.025 m above the soil surface. Samples were washed in distilled, deionized water with a mild phosphate-free detergent to remove adhering particulate matter, dried at 65°C, and ground in a Wiley mill to pass a 2×10^{-3} m screen. The samples were stored in capped glass jars for chemical analyses. Samples were analyzed for N and P using total Kjeldahl digestion, and micronutrients and total trace elements were analyzed by digestion in concentrated nitric acid (USEPA, 1996).

Biosolids-Amended Sediments Sampling and Analysis

Sediment and biosolids-amended sediment were sampled using a soil probe from each of the four subplots (each subplot contained one of the four turfgrass mixes) immediately after placing the biosolids-amended sediments (year 1) and every fall for three subsequent years (years 2–4). Ten subsamples were collected from each plot at depths of 0 to 0.15 m and 0.15 to 0.30 m and were combined and thoroughly mixed to create a composite soil sample for each depth. The samples were air-dried, ground

Table 2. Turfgrass seed mixtures planted in the research plots.

Name	Composition
Metropolitan Water Reclamation District of Greater Chicago (MWRDGC)	70% tall fescue (*Festuca arundinacea*)
	30% Kentucky bluegrass (*Poa pratensis*)
Standard Chicago Park District Turf Blend (SCPD)	70% Kentucky bluegrass (*Poa pratensis*)
	15% creeping red fescue (*Festuca rubra*)
	10% perennial rye (*Lolium perenne*)
	5% red top (*Agrostis gigantean*)
Illinois Department of Transportation (IDOT)	75% tall fescue (*Festuca arundinacea*)
	15% perennial rye (*Lolium perenne*)
	10% creeping red fescue (*Festuca rubra*)
Variation of Illinois Department of Transportation Lawn Mixture (VIDOT)	50% perennial rye (*Lolium perenne*)
	30% Kentucky bluegrass (*Poa pratensis*)
	20% creeping red fescue (*Festuca rubra*)

with a pestle and mortar, passed through a 0.002-m sieve, and stored in capped glass jars for chemical analyses.

The pH and electrical conductivity (EC) were measured in 1:2 (soil/water) slurry using a Fisher Model 50 pH/ion/conductivity meter (Thomas, 1996; Rhoades, 1996). Ammonia nitrogen and NO_3–N were extracted at a 1:2 solid-to-water ratio (Mulvaney, 1996) and analyzed with a Lachat flow injection analysis auto analyzer (Hach Co.). Total Kjeldahl N (TKN) and total P were analyzed from total Kjeldahl digested samples by UV-VIS spectrometry (Clesceri, 1998). Organic carbon was determined by Walkley–Black wet oxidation (Nelson and Sommers, 1996) for the 0- to 0.15-m samples only. For total trace elements, the 0- to 0.15-m samples were digested in concentrated nitric acid, and elements were determined by ICP–AES (USEPA, 1996).

Lysimeter and Well Sampling and Analysis

Lysimeters and wells in the plots and remote locations were sampled monthly from June to December in the first year of the study period to assess the loss of nutrients from the soil profile (Fig. 2). The water samples were analyzed for pH, EC, TP, NO_3–N, NH_4–N, and TKN. The NO_3–N and NH_4–N were analyzed by Lachat using a continuous flow injection system (Hach Co.). Total P and TKN were determined by the colorimetric method after digestion with H_2SO_4 in the presence of potassium sulfate and mercuric sulfate (USEPA, 1983).

Data Analysis and Presentation

The purpose of this case study was to qualitatively describe observable differences in turfgrass performance among four different turfgrass mixes and between four different biosolids rates. The data were not subjected to a robust statistical regimen, but rather a qualitative comparative analysis is presented.

Results and Discussion

Turfgrass Performance

The case study site consisted mostly of a deep heterogeneous fill of steel mill slag material incapable of supporting adequate vegetation (Fig. 1). The slag material had no detectable OC, 0.68 g kg^{-1} TKN, and 0.50 g kg^{-1} TP (Table 1). The heterogeneous nature of slag and the lack of OC result in very little to no water-holding capacity in the slag material. Vegetation was established with the addition of sediments or biosolids-amended sediments; however, turfgrass performance in unamended sediments was lower than in biosolids-amended sediments (Table 3). In the unamended sediments, the Standard Chicago Park District Turf Blend had the lowest score (53 ± 0.8), and the Variation of Illinois Department of Transportation Lawn Mixture had the highest score (59 ± 2.5). The scores in the 25 and 50% biosolids-amended sediments ranged from 68 ± 6.4 to 71 ± 4.5 for all four turfgrass mixes. The MWRDGC blend had the highest score (74 ± 2.4), which was in the 100% biosolids-amended sediments. Overall, the turfgrass blends were darker green and more robust in the biosolids-amended plots (Fig. 3).

In year 4, the last year of the study period, turfgrass performance scores for all turfgrass mixes in the unamended sediment plots ranged from 45 ± 17 to 56 ± 5.2, which was a lower range than scores in year 2 (Table 3). Turfgrass performance scores in biosolids-amended sediments were higher in year 4 than in year 2. The MWRDGC blend had the highest score (83 ± 3.8) in the 25% biosolids-amended sediments. The range for all turfgrass mixes for all rates of biosolids was 72 ± 14 to 83 ± 3.8. Halofsky and McCormick (2005) demonstrated that seeded turfgrass strips on a reclaimed coal mine site amended with municipal biosolids had significant cover and density. Improved turfgrass performance in biosolids-amended sediments can likely be attributed primarily to higher soil fertility and improved soil physical properties in the root zone (Loschinkohl and Boehm, 2001; Schnell et al., 2010). Although the four turfgrass mixes were established in unamended and biosolids-amended sediments, the turfgrass in the biosolids-amended sediments was darker green and denser, as determined by performance scores. The range in performance scores for the turfgrass in unamended sediments appeared to decrease in the last year of the study, whereas the range of scores

Table 3. Mean† turfgrass performance scores.

Turfgrass mixture‡	Biosolids amendments			
	0%	25%	50%	100%
	Year 2			
MWRDGC	55 ± 4.1§	69 ± 7.3	69 ± 5.2	74 ± 2.4
SCPD	53 ± 0.8	71 ± 3.7	70 ± 2.7	72 ± 3.7
IDOT	55 ± 3.1	69 ± 5.8	68 ± 6.4	73 ± 2.4
VIDOT	59 ± 2.5	71 ± 4.2	71 ± 4.5	73 ± 2.8
	Year 4			
MWRDGC	45 ± 17	83 ± 3.8	72 ± 14	74 ± 18
SCPD	54 ± 0.70	82 ± 4.0	80 ± 5.9	75 ± 16
IDOT	54 ± 8.1	74 ± 11	73 ± 22	80 ± 12
VIDOT	56 ± 5.2	74 ± 5.9	74 ± 1.9	76 ± 13

† Mean of two samples.

‡ IDOT, Illinois Department of Transportation; MWRDGC, Metropolitan Water Reclamation District of Greater Chicago; SCPD, Standard Chicago Park District Turf Blend; VIDOT, Variation of Illinois Department of Transportation Lawn Mixture.

§ Excellent, >60; well, 46–60; poor, 30–45; very poor, <29.

Fig. 3. Photo of the steel mill slag case study site (top) and photo of the site after establishment of turfgrass (bottom).

in the biosolids-amended sediments appeared to increase (Table 3). It would be expected that the biosolids-amended sediments would have a longer duration of plant nutrients available to sustain healthy vegetation for a longer period.

Turfgrass Macronutrients

Concentrations of the macronutrients N, P, and K in turfgrass tissue samples in year 2, the first year the turfgrass emerged, were greater in biosolids-amended plots than in unamended sediments (Table 4). The concentration of N in turfgrass tissue from unamended sediments was 24 ± 3.4 g kg^{-1}, and the range from biosolids-amended plots was 43 ± 4.2 to 52 ± 2.7 g kg^{-1}. The concentration of N in turfgrass in the biosolids amendments was above the general sufficiency range for turfgrass of 22 to 40 g kg^{-1} (Kabata-Pendias, 2001); lower N concentrations in the tissues from unamended sediment plots likely contributed to the poorer turfgrass performance observed.

Similarly, the total P concentration in tissues from unamended sediments was 4.1 ± 0.30 g kg^{-1}, and the range in tissues from the biosolids-amended plots was 5.2 ± 0.55 to 6.1 ± 0.29 g kg^{-1}. The concentration of K in tissues from the unamended sediments was 21 ± 2.9 g kg^{-1}, and the range in tissues from the biosolids-amended plots was 31 ± 2.5 to 40 ± 1.9 g kg^{-1}. Concentrations of the macronutrients N and K in turfgrass tissue samples were greater in biosolids-amended plots than in unamended sediments in year 4; however, total P concentrations were similar between the unamended sediments and biosolids-amended plots (Table 4).

Turfgrass Trace Elements

The concentrations of Cd, Cu, and Ni in the turfgrass tissues in year 2 of the study were greater in the biosolids-amended plots than in the unamended sediments but were below phytotoxicity limits (Table 4). The concentration of Zn in turfgrass tissues from unamended sediments was 0.04 ± 0.01 g kg^{-1}, and the range in biosolids-amended plots was 0.10 ± 0.03 to 0.19 ± 0.05 g kg^{-1}. Although Zn concentrations were within the toxicity range of 0.10 to 0.40 g kg^{-1} (Cockerham and Miner, 2001; Plank, 2002), there are generally natural limitations to trace metal bioavailability, including a plateau response, which maintains concentrations below toxic levels (Barbarick et al., 1995; Chaney, 1994; Basta et al., 2005). Additionally, Granato et al. (2004) demonstrated in a greenhouse study that tall fescue, perennial ryegrass, Kentucky

Table 4. Mean† concentrations of nutrients and trace elements in turfgrass tissues in years 2 and 4 of the study in the 0, 25, 50, and 100% biosolids-amended plots.

Parameter	Biosolids amendment				Optimum range‡	Toxicity range§
	0%	25%	50%	100%		
			Year 2¶			
			g kg^{-1}			
N	24 ± 3.4	43 ± 4.2	44 ± 7.1	52 ± 2.7	22–40	–
P	4.1 ± 0.30	5.2 ± 0.55	5.4 ± 0.91	6.1 ± 0.29	3.0–7.0	–
K	21 ± 2.9	31 ± 2.5	33 ± 2.3	40 ± 1.9	15–30	–
Mn	0.22 ± 0.09	0.12 ± 0.02	0.12 ± 0.03	0.10 ± 0.01	0.02–0.30	–
Zn	0.04 ± 0.01	0.10 ± 0.03	0.11 ± 0.02	0.19 ± 0.05	0.02–0.07	0.10–0.40
			mg kg^{-1}			
Cd	0.19 ± 0.10	0.37 ± 0.20	0.51 ± 0.20	0.68 ± 0.30		5.0–30
Cr	0.66 ± 0.30	0.41 ± 0.20	0.45 ± 0.20	0.41 ± 0.0	0.10–55	5.0–30
Cu	7.7 ± 1.3	18 ± 2.2	20 ± 2.2	29 ± 3.3	5.0–20	20–100
Mo	9.5 ± 1.8	8.5 ± 2.4	10 ± 1.9	14 ± 5.2	0.20–5.1	10–50
Ni	2.4 ± 0.50	2.8 ± 0.60	3.4 ± 0.60	8.3 ± 2.3	0.10–55	10–100
Pb	0.72 ± 0.30	0.29 ± 0.50	0.31 ± 0.20	0.11 ± 0.00		30–300
			Year 4			
			g kg^{-1}			
N	26 ± 2.5	46 ± 4.4	43 ± 8.3	42 ± 8.8	22–40	–
P	5.0 ± 0.65	4.2 ± 0.48	3.9 ± 0.28	4.3 ± 0.33	3.0–7.0	–
K	22 ± 2.4	28 ± 2.6	28 ± 3.5	33 ± 2.9	15–30	–
Mn	0.05 ± 0.09	0.03 ± 0.00	0.03 ± 0.00	0.11 ± 0.02	0.02–0.30	–
Zn	0.04 ± 0.01	0.06 ± 0.01	0.06 ± 0.01	0.10 ± 0.12	0.02–0.07	0.10–0.40
			mg kg^{-1}			
Cd	0.22 ± 0.10	0.12 ± 0.10	0.36 ± 0.30	0.67 ± 0.40		5–30
Cr	0.69 ± 0.20	0.38 ± 0.10	0.43 ± 0.20	0.52 ± 0.10	0.1–55	5–30
Cu	7.6 ± 0.80	14 ± 2.3	14 ± 2.8	18 ± 3.6	5–20	20–100
Mo	8.7 ± 2.1	4.9 ± 0.50	4.3 ± 0.70	5.1 ± 1.3	0.2–5.05	10–50
Ni	1.6 ± 0.30	2.0 ± 0.50	2.7 ± 0.50	6.8 ± 1.2	0.1–55	10–100
Pb	0.53 ± 0.40	0.26 ± 0.20	0.23 ± 0.10	0.30 ± 0.10		30–300

† Mean of eight samples.

‡ Kabata-Pendias (2001).

§ Cockerham and Miner (2001); Plank (2002).

¶ Turfgrass emerged in year 2; thus, concentrations in years 2 and 4 are compared.

bluegrass, and reed canary grass had Zn toxicity thresholds of 1.3, 1.2, 0.69, and 1.5 g kg^{-1}, respectively.

The concentrations of Cr and Pb in the turfgrass tissues from the biosolids-amended plots in year 2 of the study were comparable to unamended sediments (Table 4). The turfgrass mixtures in this case study were dark green and dense and did not visually exhibit any phytotoxicity effects in any of the biosolids-amended plots. Metal oxides in biosolids, particularly Fe-oxides, play a major role in the immobilization of trace elements, likely due to coprecipitation with and adsorption by Fe compounds (Sukkariyah et al., 2005; Hettiarachchi et al., 2006). Overall, the concentrations of trace elements were below phytotoxicity levels, and these results further corroborate that biosolids-amended sediments are not a significant source of heavy metals in turfgrass tissues.

Soil Fertility

The soil samples from all turfgrass subplots were pooled for analysis, and the mean concentrations of nutrients and soil fertility parameters in the 0- to 0.15-m and the 0.15- to 0.30-m depths of the four biosolids rates are discussed (Tables 5 and 6). The biosolids used in this case study provided 16 g kg^{-1} TKN, 24 g kg^{-1} total P, 160 g kg^{-1} OC, and a suite of other micronutrients beneficial to soil fertility (Table 1). As expected, there were observable increases in all soil fertility parameters with increasing biosolids rates in the 0- to 0.15-m and 0.15- to 0.30-m depths (Tables 5 and 6).

Soil pH values in the 100% biosolids-amended plots at both depths were lower than any other biosolids rate (Tables 5 and 6). The lower soil pH in the 100% biosolids amendment is most likely due to acidification associated with the higher amount of nitrification and organic matter mineralization in the 100% biosolids amendment. The soil pH values in all plots at both depths ranged from 6.1 ± 0.1 to 7.9 ± 0.3, which is within the optimum range for plant growth (Soil and Plant Analysis Council, 1999). Concentrations of OC in the 0- to 0.15-m depth ranged from 21 ± 2.9 to 160 ± 8.1 g kg^{-1} in year 1 and from 24 ± 1.7 to 150 ± 25 g kg^{-1} in year 4 (Table 4). The greater amount of OC associated with biosolids addition helps increase nutrient and water retention in the rhizosphere (Williamson et al., 2011).

Soil EC values also increased with increasing biosolids rates at both depths (Tables 5 and 6). At the 0- to 0.15-m depth, soil EC values ranged from 0.39 ± 0.16 dS m^{-1} in unamended sediments to 2.9 ± 0.30 dS m^{-1} in 100% biosolids-amended plots in year 1. In year 4, soil EC decreased slightly and ranged from 0.18 ± 0.04 to 1.7 ± 0.36 dS m^{-1} (Table 5). A similar trend in decreasing EC

Table 5. Mean† concentrations of soil fertility parameters and nutrients in year 1 of the study when the 0, 25, 50, and 100% biosolids-amended plots were first blended and in year 4 at the 0- to 0.15-m depth.

Biosolids amendment	pH	EC‡	OC‡	TKN‡	NO_3–N	TP‡	NH_3–N
%		dS m^{-1}	g kg^{-1}				mg kg^{-1}
				Year 1			
0	6.6 ± 0.0	0.39 ± 0.16	21 ± 2.9	1.7 ± 0.24	5.0×10^{-3} ± 0.00	0.67 ± 0.67	0.28 ± 0.05
25	6.8 ± 0.15	1.8 ± 0.29	73 ± 10	4.5 ± 0.23	0.15 ± 0.04	6.8 ± 15	1.7 ± 0.78
50	6.9 ± 0.30	1.8 ± 0.27	86 ± 15	6.7 ± 0.52	0.14 ± 0.07	9.4 ± 2.5	2.2 ± 0.77
100	6.1 ± 0.13	2.9 ± 0.30	160 ± 8.1	13 ± 0.52	0.37 ± 0.08	24 ± 1.8	8.5 ± 5.9
				Year 4			
0	7.9 ± 0.33	0.18 ± 0.04	24 ± 1.7	1.9 ± 0.27	1.7×10^{-3} ± 0.00	0.58 ± 0.26	0.59 ± 0.59
25	7.6 ± 0.67	0.39 ± 0.07	60 ± 9.5	5.0 ± 1.3	0.03 ± 0.00	7.4 ± 2.2	0.52 ± 0.26
50	7.8 ± 0.45	0.46 ± 0.10	81 ± 19	7.4 ± 0.98	0.04 ± 0.03	11 ± 3.0	0.94 ± 0.92
100	6.5 ± 0.12	1.7 ± 0.36	150 ± 25	16 ± 1.1	0.19 ± 0.08	28 ± 0.47	2.7 ± 0.94

† Mean of eight samples.

‡ EC, electrical conductivity; OC, organic carbon; TKN, total Kjeldahl nitrogen; TP, total P.

Table 6. Mean† concentrations of soil fertility parameters and nutrients in year 1 of the study when the 0, 25, 50, and 100% biosolids-amended plots were first blended and in year 4 at the 0.15- to 0.30-m depth.

Biosolids amendment	pH	EC‡	TKN‡	NO_3–N	TP‡	NH_3–N
%		dS m^{-1}	g kg^{-1}			mg kg^{-1}
			Year 1			
0	6.7 ± 0.08	0.66 ± 0.29	1.7 ± 0.12	5.9×10^{-3} ± 0.00	NA§	0.21 ± 0.10
25	7.0 ± 0.16	1.8 ± 0.27	6.1 ± 1.7	0.10 ± 0.05	NA	2.1 ± 0.53
50	7.2 ± 0.20	1.9 ± 0.30	10 ± 2.0	0.11 ± 0.06	NA	3.2 ± 1.3
100	6.4 ± 0.14	3.0 ± 0.52	16 ± 1.4	0.25 ± 0.08	NA	33 ± 36
			Year 4			
0	7.9 ± 0.29	0.30 ± 0.13	2.4 ± 0.29	1.5×10^{-3} ± 0.00	1.1 ± 0.75	1.0 ± 0.57
25	7.5 ± 0.35	0.39 ± 0.08	5.5 ± 1.1	0.02 ± 0.01	6.3 ± 2.1	1.0 ± 0.76
50	7.7 ± 0.16	0.50 ± 0.12	8.0 ± 1.0	0.04 ± 0.03	11 ± 3.3	1.3 ± 0.65
100	6.7 ± 0.50	2.3 ± 0.56	16 ± 3.5	0.22 ± 0.16	25 ± 6.4	2.8 ± 1.5

† Mean of eight samples.

‡ EC, electrical conductivity; TKN, total Kjeldahl nitrogen; TP, total P.

§ No data available.

values from year 1 to year 4 was observed at the 0.15- to 0.30-m depth (Table 6). An initial increase in EC values was expected due to the addition of a higher proportion of water-soluble plant nutrients, such as NO_3–N, other micronutrients, and salts with increasing biosolids rates. Although EC values at the 0- to 0.15-m depth were as high as 2.9 dS m^{-1}, vegetation was not adversely affected. Salt-sensitive turfgrasses can tolerate EC levels up to 1.5 dS m^{-1}, and moderately sensitive turfgrasses can tolerate up to 3.0 dS m^{-1} (Carrow and Duncan, 1998); other than Kentucky bluegrass, the turfgrasses used in this study are tolerant of soil EC values >3.0 dS m^{-1}.

The NO_3–N concentrations in year 1 ranged from $5.0 \times 10^{-3} \pm 0.00$ g kg^{-1} in unamended sediments to 0.37 ± 0.08 g kg^{-1} in 100% biosolids-amended plots. In year 4 of the study, NO_3–N concentrations decreased from year 1 and ranged from $1.7 \times 10^{-3} \pm 0.00$ g kg^{-1} in unamended sediments to 0.19 ± 0.08 g kg^{-1} in the 100% biosolids-amended plots. Nitrate concentrations were comparable at the 0.15- to 0.30-m depth but did not decrease appreciably between year 1 and year 4 (Table 6). A decrease in surface NO_3–N concentrations due to some leaching would be expected. Zhang et al. (2006) applied composted biosolids to degraded soils to improve soil fertility and demonstrated downward trends in N concentrations in the years after the application.

Concentrations in the soil fertility parameters TKN and TP at the 0- to 0.15-m depth increased with increasing biosolids rates but were consistent between year 1 and year 4 (Table 5). Concentrations of TKN ranged from 1.7 ± 0.24 g kg^{-1} in unamended sediments to 13 ± 0.52 g kg^{-1} in the 100% biosolids-amended plots in year 1. In year 4, TKN concentrations ranged from 1.9 ± 0.27 to 16 ± 1.1 g kg^{-1}. Total P concentrations ranged from 0.67 ± 0.67 to 24 ± 1.8 g kg^{-1} in year 1 and from 0.58 ± 0.26 to 28 ± 0.47 g kg^{-1} in year 4. Concentrations of TKN and TP also increased with increasing biosolids rates at the 0.15- to 0.30-m depth (Table 6). These data demonstrate that the addition of biosolids to dredged sediments increases the concentration of plant nutrients and significantly improves the soil fertility status of the sediments.

Soil Trace Elements

Trace elements Cd, Cu, Cr, Mo, Pb, and Zn in the 0- to 0.15-m depth of the amendments increased with increasing biosolids rates (Table 7). The concentrations, however, were consistent between year 1 and year 4, indicating that these trace metals were not appreciably taken up by turfgrass or leached from the soil profile (Table 7). Although Zn concentrations in the 100% biosolids-amended sediments reached 1.3 g kg^{-1}, which is in range of concentrations shown to stunt soybean growth in farm fields receiving cattle manure (Pierzynski and Schwab, 1993), the turfgrass in the 100% biosolids-amended sediments did not show any visual signs of phytotoxicity. This is likely due to the plateau effect limiting bioavailability of trace elements and tissue concentrations of Zn below phytotoxicity thresholds in turfgrass (Table 4).

The concentrations of Ni did not increase appreciably with increasing biosolids amendment and were similar to the concentrations in unamended sediments (Table 7). The 25 and 50% biosolids-amended sediments in year 1 resulted in concentrations of 0.06 ± 0.00 and 0.07 ± 0.2 g kg^{-1} Pb, respectively, which are below the 0.10 g kg^{-1} limit the USEPA recommends for establishing community gardens (USEPA, 2013). Soil Pb levels at 0.10 g kg^{-1} or lower are considered to be of negligible risk with no restrictions on types of vegetables grown. Remediation is not recommended until a soil Pb concentration >0.40 g kg^{-1} is reached. The 100% biosolids amendment resulted in 0.13 ± 0.02 g kg^{-1} Pb, which is below the 0.40 g kg^{-1} limit. Additionally, the application of biosolids to Pb- and As-contaminated urban soils was shown to reduce the bioavailability of Pb and As in carrots, lettuce, and tomatoes in an urban garden from 50 to 71% and 46 to 80%, respectively (Defoe et al., 2014).

Subsurface Water Quality

Overall, there were no consistent differences between concentrations of water quality parameters in well samples from the clay-lined (east) and nonlined (west) plots (Table 8). Total Kjeldahl N, NH_3–N, and NO_3–N concentrations generally increased in samples from the 1.5-m lysimeters in the biosolids-amended plots compared with the unamended sediments (Table 8). Total Kjeldahl N in lysimeter samples at 1.5 m from the clay-lined unamended sediments plots was 2.4 ± 2.7 mg L^{-1} and ranged from 3.5 ± 1.9 to 15 ± 11 mg L^{-1} in samples from the clay-lined biosolids-amended plots at the same depth. The concentration of TKN from well samples at the 6-m depth in all plots was <1.0 mg L^{-1} and was <0.50 mg L^{-1} in well samples closest to the lake.

Table 7. Mean† soil total element concentrations in year 1 of the study when the 0, 25, 50, and 100% biosolids-amended plots were blended and in year 4 at the 0- to 0.15-m depth.

Biosolids amendment	Cd	Ni	Cu	Cr	Pb	Zn
%	mg kg^{-1}		g kg^{-1}			
			Year 1			
0	0.84 ± 0.00	24 ± 3.3	0.03 ± 0.00	0.03 ± 0.00	0.02 ± 0.00	0.08 ± 0.03
25	3.1 ± 0.72	29 ± 2.1	0.11 ± 0.02	0.10 ± 0.03	0.06 ± 0.00	0.51 ± 0.10
50	3.6 ± 1.1	32 ± 3.0	0.14 ± 0.03	0.18 ± 0.06	0.07 ± 0.02	0.64 ± 0.16
100	8.9 ± 2.2	32 ± 3.1	0.28 ± 0.06	0.15 ± 0.01	0.13 ± 0.02	1.3 ± 0.28
			Year 4			
0	1.1 ± 0.19	23 ± 6.8	0.02 ± 0.00	0.07 ± 0.08	0.03 ± 0.00	0.07 ± 0.01
25	4.3 ± 0.80	32 ± 4.5	0.12 ± 0.03	0.16 ± 0.08	0.06 ± 0.01	0.41 ± 0.10
50	5.5 ± 0.88	36 ± 8.1	0.16 ± 0.03	0.29 ± 0.12	0.07 ± 0.01	0.53 ± 0.11
100	12 ± 0.32	35 ± 0.90	0.38 ± 0.02	0.19 ± 0.03	0.14 ± 0.00	1.3 ± 0.05

† Mean of eight samples.

Table 8. Mean concentrations of water chemistry parameters and nutrients in lysimeter and well samples collected once monthly from June through December (seven samples) in year 1 of the study from clay-lined and nonlined plots.

ID†	Device	Depth	Biosolids	pH	Water parameters‡				
					EC	TKN	NH_3–N	NO_3–N	TP
		m	%		dS m^{-1}	mg L^{-1}			
L-1E1§	Lys¶	1.5	0	7.8 ± 0.4	1.3 ± 1.2	2.4 ± 2.7	0.28 ± 0.11	6.8 ± 5.8	0.26 ± 0.14
L-1E2#	Lys	3.0	0	7.3 ± 0.6	1.0 ± 1.4	1.4 ± 0.81	0.29 ± 0.15	6.7 ± 4.5	0.18 ± 0.04
L-1W2§	Lys	3.0	0	7.5 ± 0.6	1.7 ± 1.6	1.4 ± 0.98	0.24 ± 0.05	0.24 ± 0.05	0.14 ± 0.07
W-1E3	Well	6.0	0	7.5 ± 0.2	1.2 ± 0.72	0.83 ± 0.27	0.19 ± 0.11	0.96 ± 1.8	0.09 ± 0.07
L-2E1§	Lys	1.5	25	7.9 ± 0.2	1.2 ± 1.2	3.5 ± 1.9	0.43 ± 0.21	49 ± 49	0.14 ± 0.09
L-2W1††	Lys	1.5	25	8.1 ± 0.2	1.3 ± 1.5	4.2 ± 4.2	0.58 ± 0.47	18 ± 35	0.16 ± 0.15
L-2E2§	Lys	3.0	25	7.6 ± 0.6	1.3 ± 1.0	2.0 ± 1.0	0.33 ± 0.11	4.1 ± 4.2	0.11 ± 0.05
L-2W2§	Lys	3.0	25	7.9 ± 0.4	1.4 ± 1.4	3.2 ± 1.9	0.34 ± 0.22	5.2 ± 5.5	0.15 ± 0.08
W-2E3	Well	6.0	25	7.4 ± 0.3	1.4 ± 0.83	0.73 ± 0.82	0.20 ± 0.03	0.17 ± 0.16	0.06 ± 0.04
W-2W3	Well	6.0	25	7.5 ± 0.2	1.5 ± 0.66	0.94 ± 0.61	0.39 ± 0.25	0.41 ± 0.26	0.10 ± 0.06
L-3E1§	Lys	1.5	50	7.6 ± 0.5	2.4 ± 2.0	7.1 ± 4.3	0.90 ± 0.51	143 ± 128	0.12 ± 0.06
L-3W1§	Lys	1.5	50	7.8 ± 0.4	1.6 ± 1.5	6.3 ± 4.6	0.87 ± 0.62	53 ± 69	0.18 ± 0.02
L-3E2§	Lys	3.0	50	7.6 ± 0.5	1.8 ± 1.4	5.7 ± 4.1	1.9 ± 1.4	63 ± 66	0.16 ± 0.07
L-3W2§	Lys	3.0	50	7.7 ± 0.5	1.9 ± 1.8	2.2 ± 1.6	0.28 ± 0.23	3.9 ± 7.0	0.10 ± 0.05
W-3W3	Well	6.0	50	7.6 ± 0.4	1.3 ± 0.59	0.62 ± 0.31	0.22 ± 0.03	0.13 ± 0.10	0.11 ± 0.07
L-4E1	Lys	1.5	100	8.7 ± 0.9	1.1 ± 0.3	15 ± 11	9.8 ± 8.3	3.1 ± 8.4	0.15 ± 0.05
L-4W1††	Lys	1.5	100	7.7 ± 0.3	2.2 ± 2.4	21 ± 23	6.9 ± 8.5	117 ± 172	0.42 ± 0.24
L-4E2††	Lys	3.0	100	8.0 ± 0.2	1.7 ± 1.4	2.7 ± 2.2	0.33 ± 0.13	20 ± 9.4	0.14 ± 0.06
L-4W2§	Lys	3.0	100	7.5 ± 0.5	1.1 ± 0.75	11 ± 12	7.1 ± 7.8	9.1 ± 16	0.21 ± 0.11
W-4W3	Well	6.0	100	7.6 ± 0.5	1.4 ± 0.59	0.89 ± 0.61	0.45 ± 0.23	0.36 ± 0.72	0.06 ± 0.03
R1§	Lys	1.5	0	7.9 ± 0.4	1.4 ± 0.89	1.4 ± 0.79	0.31 ± 0.18	0.28 ± 0.29	0.11 ± 0.06
R2§	Lys	3.0	0	8.0 ± 0.3	1.9 ± 1.7	1.4 ± 0.47	0.28 ± 0.11	6.0 ± 10	0.11 ± 0.06
R3	Well	6.0	0	7.6 ± 0.3	1.2 ± 0.54	0.70 ± 0.42	0.38 ± 0.18	0.24 ± 0.32	0.08 ± 0.07
Lake-N††	Well	6.0	0	7.6 ± 0.7	0.31 ± 0.01	0.17 ± 0.12	0.06 ± 0.02	0.30 ± 0.06	0.07 ± 0.03
Lake‡	Well	6.0	0	7.7 ± 0.5	0.31 ± 0.01	0.32 ± 0.12	0.07 ± 0.02	0.33 ± 0.15	0.08 ± 0.04
Lake-S††	Well	6.0	0	7.6 ± 0.7	0.32 ± 0.02	0.40 ± 0.19	0.07 ± 0.03	0.37 ± 0.17	0.07 ± 0.02

† E indicates clay-lined pots; W indicates nonlined plots. Plots 1W1 and 4E3 omitted due to lack of data collected from devices.

‡ EC, electrical conductivity; TKN, total Kjeldahl nitrogen; TP, total P.

§ Mean of six samples.

¶ Lys, lysimeter; Well, monitoring well.

Mean of four samples.

†† Mean of five samples.

There was a general trend of increasing NO_3–N concentrations with increasing biosolids rates within the first 1.5-m depth of the cap. Nitrate concentrations in lysimeter samples at the 1.5-m depth from the clay-lined and nonlined 100% biosolids-amended plots were 3.1 ± 8.4 and 117 ± 172 mg L^{-1}, respectively. Although surface NO_3–N concentrations increased with increasing biosolids, NO_3–N concentrations were <1.0 mg L^{-1} in all well samples at 6 m depth from all plots and were <0.50 mg L^{-1} in well samples closest to the lake (Table 8). These values are well below Illinois's EPA's potable groundwater pollutant limit of 10 mg L^{-1} (ILEPA, 1991). Long-term agricultural research has shown that the addition of biosolids and other organic amendments enhances soil organic N content by up to 90% but stores it for mineralization in later cropping seasons without increasing nitrate concentrations in groundwater (Diacono and Montemurro, 2010).

The highest total P concentration observed in lysimeters was 0.42 ± 0.24 mg L^{-1} in the 1.5-m lysimeter in the 100% biosolids-amended sediment, followed by 0.26 ± 0.14 mg L^{-1} in the 1.5-m lysimeter in the 0% biosolids-amended sediment (Table 8). All well samples, including well samples closest to the lake, were <0.11 mg L^{-1}. Phosphorus in biosolids is complexed by Fe- and Al-oxides and is much less water soluble than P in fertilizers applied to agricultural soils (Sharpley et al., 1994; Maguire et al., 2001; Ippolito et al., 2007). Amending sediments with biosolids increased total P concentrations, which were retained in the biosolids-amended plots and did not appreciably leach from the soil profile.

Conclusions

This case study provides a qualitative assessment of using EQ biosolids blended into dredged sediments as an effective cap for establishing turfgrass on steel mill slag brownfields and thus potentially other marginal soils. Amending sediments with biosolids provides sufficient nutrients leading to the improved performance of turfgrass. The rate of biosolids application in the sediment blend depends on the nutrient requirements of the vegetation specified for the site and proximity to surface water or groundwater. When groundwater is not shallow and surface water is not in close proximity to the reclamation site, higher rates of biosolids up to 100% could be used to improve soil fertility and long-term plant performance. Conservative rates of 50% or lower are recommended if there is groundwater within the first 3 m of the underlying soil profile or surface water in close proximity to the reclamation site to ensure that water quality is not affected by excess nutrients.

Because of the initial low concentrations of metals in EQ biosolids and their ability to chelate heavy metals, amended sediments are not limited for use in residential soils or for gardening purposes in states following USEPA regulations. The concentrations of most heavy metals in turfgrass from the biosolids-amended plots were similar to those in the unamended sediment plots. Overall, this case study demonstrated that dredged sediments from local waterways, blended with biosolids, would provide a good quality soil amendment for capping and re-vegetating steel mill slag brownfields and other marginal lands, which is a critical aspect of urban revitalization.

References

Barbarick, K.A., J.A. Ippolito, and D.G. Westfall. 1995. Biosolids effect on phosphorus, copper, zinc, nickel, and molybdenum concentrations in dryland wheat. J. Environ. Qual. 24:608–611. doi:10.2134/jeq1995.00472425002400040009x

Basta, N.T., R. Gradwohl, K.L. Snethena, and J.L. Schrodera. 2001. Chemical immobilization of lead, zinc, and cadmium in smelter-contaminated soils using biosolids and rock phosphate. J. Environ. Qual. 30:1222–1230. doi:10.2134/jeq2001.3041222x

Basta, N.T., J.A. Ryan, and R.L. Chaney. 2005. Trace element chemistry in residual-treated soil: Key concepts and metal bioavailability. J. Environ. Qual. 34:49–63.

Brown, S.L., I. Clausen, M.A. Chappell, K.G. Scheckel, M. Newville, and G.M. Hettiarachchi. 2012. High-iron biosolids compost–induced changes in lead and arsenic speciation and bioaccessibility in co-contaminated soils. J. Environ. Qual. 41:1612–1622. doi:10.2134/jeq2011.0297

Brown, S.L., K. Kurtz, A. Bary, and C. Cogger. 2011. Quantifying benefits associated with land application of organic residuals in Washington state. Environ. Sci. Technol. 45:7451–7458. doi:10.1021/es2010418

Brown, S.L., R.L. Chaney, J.G. Hallfrisch, and Q. Xue. 2003. Effect of biosolids processing on lead bioavailability in an urban soil. J. Environ. Qual. 32:100–108. doi:10.2134/jeq2003.1000

Carrow, R.N., and R.R. Duncan. 1998. Salt affected turfgrass sites: Assessment and management. Ann Arbor Press, Chelsea, MI.

Chaney, R.L. 1994. Trace metal movement: Soil-plant systems and bioavailability of biosolids-applied metals. In: C.E. Clapp, W.E. Larson, and R.H. Dowdy, editors, Sewage sludge: Land utilization and the environment. ASA, CSSA, SSSA, Madison, WI. p. 27–31.

Clesceri, L.S. 1998. Phosphorus (4500-Norg)/block digestion and flow injection analysis. In: L.S. Clesceri, A.D. Eaton, and A.E. Greenberg, editors, Methods for the examination of water and wastewater. 20th ed. American Public Health Association, Washington, DC. p. 126–128.

Cockerham, S.T., and D.D. Miner. 2001. Turfgrass Nutrition and Fertilizers. In: D.E. Aldous, editor, International turf management handbook. CRC Press, New York.

Darmody, R.G., J.C. Marlin, J. Talbott, R.A. Green, E.F. Brewer, and C. Stohr. 2004. Dredged Illinois River sediments. J. Environ. Qual. 33:458–464. doi:10.2134/jeq2004.4580

Defoe, P.P., G.M. Hettiarachchi, C. Benedict, and S. Martin. 2014. Safety of gardening on lead- and arsenic-contaminated urban brownfields. J. Environ. Qual. 43:2064–2078. doi:10.2134/jeq2014.03.0099

Diacono, M., and F. Montemurro. 2010. Long-term effects of organic amendments on soil fertility: A review. Agron. Sustainable Dev. 30:401–422. doi:10.1051/agro/2009040

Granato, T.G., O. Dennison, and G. Knafl. 2004. Determination of phytotoxic zinc thresholds in leaves of grasses and food and fiber crops. Report No. 04-23. Metropolitan Water Reclamation District of Greater Chicago, Chicago, IL.

Halofsky, J.E., and L.H. McCormick. 2005. Effects of unseeded areas on species richness of coal mines reclaimed with municipal biosolids. Restor. Ecol. 13:630–638. doi:10.1111/j.1526-100X.2005.00081.x

Hettiarachchi, G.M., K.G. Scheckel, J.A. Ryan, S.R. Sutton, and M. Newville. 2006. u-XRF investigations of metal binding mechanisms in biosolids. J. Environ. Qual. 35:342–351. doi:10.2134/jeq2004.0259

ILEPA. 1991. Groundwater quality. IL Admin. Code. Title 35 Subtitle F Chapter I Part 620 Subpart D. ftp://www.ilga.gov/JCAR/AdminCode/035/03500620sections.html (accessed 2 Sept. 2015).

Ippolito, J.A., K.A. Barbarick, and K.L. Norvell. 2007. Biosolids impact soil phosphorus recovery, fractionation, and potential environmental risk. J. Environ. Qual. 36:764–772. doi:10.2134/jeq2006.0308

Kabata-Pendias, A. 2001. Trace elements in soils and plants, 3rd ed. CRC Press, Boca Raton, FL.

Kelly, J.J., E. Favila, L.S. Hundal, and J.C. Marlin. 2007. Assessment of soil microbial communities in surface applied mixtures of Illinois River sediments and biosolids. Appl. Soil Ecol. 36:176–183. doi:10.1016/j.apsoil.2007.01.006

Loschinkohl, C., and M.J. Boehm. 2001. Composted biosolids incorporation improves turfgrass establishment on disturbed urban soil and reduces leaf rust severity. HortScience 36:790–794.

Maguire, R.O., J.T. Sims, S.K. Dentel, F.J. Coale, and J.T. Mah. 2001. Relationships between biosolids treatment process and soil phosphorus availability. J. Environ. Qual. 30:1023–1033. doi:10.2134/jeq2001.3031023x

McIvor, K., C. Cogger, and S. Brown. 2012. Effects of biosolids based soil products on soil physical and chemical properties in urban gardens. Compost Sci. Util. 20:199. doi:10.1080/1065657X.2012.10737049

Mulvaney, R.L. 1996. Nitrogen-inorganic forms. In: D.L. Sparks, A.L. Page, P.A. Helmke, and R.H. Loeppert, editors, Methods of soil analysis. Part 3. Chemical methods. SSSA Book Ser. 5. ASA and SSSA, Madison WI. p. 1123–1184.

Navarro, A. 2012. Effect of sludge amendment on remediation of metal contaminated soils. Minerals 2:473–492. doi:10.3390/min2040473

Nelson, D.W., and L.E. Sommers. 1996. Total carbon, organic carbon and organic matter. In: D.L. Sparks, A.L. Page, P.A. Helmke, and R.H. Loeppert, editors, Methods of soil analysis. Part 3. Chemical methods. SSSA Book Ser. 5. ASA and SSSA, Madison, WI. p. 961–1010.

Pierzynski, G.M., and A.P. Schwab. 1993. Bioavailability of zinc, cadmium, and lead in a metal-contaminated alluvial soil. J. Environ. Qual. 22:247–254. doi:10.2134/jeq1993.00472425002200020003x

Plank, C.O. 2002. Plant analysis handbook for Georgia. The Cooperative Extension Service, The University of Georgia, Athens, GA.

Rhoades, J.D. 1996. Salinity: Electrical conductivity and total dissolved solids. In: D.L. Sparks, editor, Methods of soil analysis. SSSA, Madison, WI. p. 417–435.

Ruiz Diaz, D.A., R.G. Darmody, J.C. Marlin, G.A. Bollero, and F.W. Simmons. 2010. Trace metal bioaccumulation and plant growth on dredged river sediments and biosolids. Water Air Soil Pollut. 206:321–333. doi:10.1007/s11270-009-0108-7

Sukkariyah, B.F., G. Evanylo, L. Zelazny, and R.L. Chaney. 2005. Recovery and distribution of biosolids-derived trace metals in a clay loam soil. J. Environ. Qual. 34:1843–1850. doi:10.2134/jeq2004.0180

Schnell, R.W., D.M. Vietor, C.L. Munster, R.H. White, and T.L. Provin. 2010. Effect of turfgrass establishment practices and composted biosolids on water quality. J. Environ. Qual. 39:697–705. doi:10.2134/jeq2009.0198

Sharpley, A.N., S.C. Chapra, R. Wedepohl, J.T. Sims, T.C. Daniel, and K.R. Reddy. 1994. Managing agriculture phosphorus for protection of surface waters: Issues and options. J. Environ. Qual. 23:437–451. doi:10.2134/jeq1994.00472425002300030006x

Sigua, G.C. 2009. Recycling biosolids and lake-dredged materials to pasture-based animal agriculture: Alternative nutrient sources for forage productivity and sustainability: A review. Agron. Sustainable Dev. 29:143–160. doi:10.1051/agro:2008037

Soil and Plant Analysis Council. 1999. Soil analysis handbook of reference methods. CRC Press, Boca Raton, FL.

Swink, F., and G. Wilhelm. 1994. Plants of the Chicago region. 4th Ed. Indiana Academy of Science, Indianapolis, IN.

Thomas, G.W. 1996. Soil pH and soil acidity. In: D.L. Sparks, A.L. Page, P.A. Helmke, and R.H. Loeppert, editors, Methods of soil analysis. Part 3. Chemical methods. SSSA Book Ser. 5. ASA and SSSA, Madison, WI. p. 475–515.

USEPA. 1983. Test method for chemical analysis of water and wastes. EPA-600/4-79-020. USEPA, Washington, DC.

USEPA. 1993. Standards for the use and disposal of wastewater sludge, final rule. Fed. Regist. 55:9248–9415.

USEPA. 1996. Test methods for evaluating solid waste: Physical/chemical methods. USEPA, Washington, DC.

USEPA. 2013. Technical review workgroup recommendations regarding gardening and reducing exposure to lead-contaminated soils. OSWER 9200.2-142. Office of Solid Waste and Emergency Response, USEPA, Washington, DC.

Williamson, J.C., E.C. Rowe, P.W. Hill, M.A. Nason, D.L. Jones, and J.R. Healey. 2011. Alleviation of both water and nutrient limitations is necessary to accelerate ecological restoration of waste rock tips. Restor. Ecol. 19:194–204. doi:10.1111/j.1526-100X.2009.00575.x

Zhang, M., D. Heaney, B. Henriquez, E. Solberg, and E. Bittner. 2006. A four-year study on influence of biosolids/MSW co-compost application in less productive soils in Alberta: Nutrient dynamics. Compost Sci. Util. 14:68–80. doi:10.1080/1065657X.2006.10702265

Published January 11, 2016

Journal of Environmental Quality

SPECIAL SECTION

SOIL IN THE CITY

Restoring Ecosystem Function in Degraded Urban Soil Using Biosolids, Biosolids Blend, and Compost

N. T. Basta,* D. M. Busalacchi, L. S. Hundal, K. Kumar, R. P. Dick, R. P. Lanno, J. Carlson, A. E. Cox, and T. C. Granato

Abstract

Many soils at former industrial sites are degraded. The objective of this research was to determine the ability of compost, biosolids, and biosolids blends to improve soil ecosystem function with minimal potential impact to surface water. Treatments rototilled into the top 12.5 cm of soil were biosolids at 202 Mg ha^{-1}; biosolids at 404 Mg ha^{-1}; compost at 137 Mg ha^{-1}; or a blend consisting of biosolids applied at 202 Mg ha^{-1}, drinking water treatment residual, and biochar. Rainfall runoff from experimental plots was collected for 3 yr. One year after soil amendments were incorporated, a native seed mix containing grasses, legumes, and forbs was planted. Soil amendments improved soil quality and nutrient pools, established a dense and high-quality vegetative cover, and improved earthworm reproductive measures. Amendments increased soil enzymatic activities that support soil function. Biosolids treatments increased the Shannon–Weaver Diversity Index for grasses. For the forbs group, control plots had the lowest diversity index and the biosolids blend had the highest diversity index. Biosolids and compost increased the number of earthworm juveniles. In general, biosolids outperformed compost. Biosolids increased N and P in rainfall runoff more than compost before vegetation was established. Several microconstituents (i.e., pharmaceutical and personal care products) were detected in runoff water but at concentrations below the probable no-effect levels and therefore should pose little impact to the aquatic environment. Future restoration design should ensure that runoff control measures are used to control sediment loss from the restored sites at least until vegetation is established.

Core Ideas

- Compost and biosolids restored urban degraded soil with minimal impact.
- Biosolids and compost greatly increased critical soil ecosystem services.
- Biosolids outperformed compost in soil quality, vegetation, and earthworm measures.
- Pharmaceuticals in runoff water were below the probable no-effect levels.
- Biosolids-treatments increased diversity in established native prairie.

J. Environ. Qual. 45:74–83 (2016)
doi:10.2134/jeq2015.01.0009
Received 8 Jan. 2015.
Accepted 6 July 2015.
*Corresponding author (basta.4@osu.edu).

The deindustrialization of cities has created vacant land that needs redevelopment. However, many soils at former industrial sites have been degraded by the loss of topsoil, contamination, and/or soil conditions that will not support plant or microbial life (i.e., salinity, acidity, and compaction). Research and demonstration studies have shown that organic soil amendments (e.g., biosolids, anaerobic digester waste, and compost) are effective soil amendments to restore degraded land (Basta, 2000; Brown et al., 2014; Haering et al., 2000; Stuczynski et al., 2007; Siebielec and Chaney, 2012; Tian et al., 2006). The Lake Calumet Cluster Site (LCCS) in Calumet, IL, a Superfund Site that was formerly an industrial steel mill and a heavy industrial complex, is an extensive area where soils are degraded and is listed as high priority for ecological functional restoration. Restoration goals for the area are rehabilitation and restoration of degraded soils to produce a functioning, complex ecosystem (Calumet Ecotoxicology Roundtable Technical Team, 2007). Much of the LCCS slated for ecological restoration is characterized by degraded soils with little topsoil and the presence of fill (e.g., steel slag), low levels of soil organic matter, and, consequently, little vegetative cover. Soil erosion by wind and water then becomes a serious problem and affects aquatic habitats adjacent to these sites.

Organic soil amendments (i.e., biosolids and compost) can be used to restore soil ecosystem function. The use of biosolids from the Metropolitan Water Reclamation District of Greater Chicago, used for land restoration and remediation on many sites in the Greater Chicago region and on coal strip mining sites in southern Illinois (Basta, 2000; Tian et al., 2006), has been proposed for restoration at LCCS. This study was performed to determine the use of biosolids as an appropriate byproduct material for ecological restoration. The primary objective of this research was to determine the ability of biosolids and biosolids

N.T. Basta, D.M. Busalacchi, R.P. Dick, and J. Carlson, School of Environment and Natural Resources, The Ohio State University, Columbus, OH 43210; L.S. Hundal, K. Kumar, A.E. Cox, and T.C. Granato, Metropolitan Water Reclamation District of Greater Chicago, Cicero, IL 60804; R.P. Lanno, Dep. of Evolution, Ecology, and Organismal Biology, The Ohio State Univ., Columbus, OH 43210; D.M. Busalacchi, current address, Ohio Environmental Protection Agency, Columbus, OH 43216; J. Carlson, current address, Ohio Environmental Protection Agency, Twinsburg, OH 44087. Assigned to Associate Editor Kang Xia.

Abbreviations: BL, blend treatment; DEET, N,N-Diethyl-meta-toluamide; HDPE, high-density polyethylene; HQ, hazard quotient; ICP–OES, inductively coupled plasma–atomic emission spectroscopy; LCCS, Lake Calumet Cluster Site; MWRD, Metropolitan Water Reclamation District of Greater Chicago; NOAEL, no observable adverse effect level; PSI, phosphorus saturation index; TSS, total suspended solids; WTR, water treatment residual.

blends to improve soil ecosystem function. The second objective was to compare the performance of biosolids with US Compost Council–certified compost, which is the material selected for restoration in parts of the LCCS. The third objective was to evaluate the potential impact of biosolids on runoff water quality and to evaluate a best management practice to reduce the potential environmental impacts. The use of organic amendments to restore disturbed land is not new. The objective of this research was to evaluate key ecosystem measures that compare biosolids and compost for restoration in a comprehensive field study.

Materials and Methods

A 0.10-ha research site was established on the property of the Metropolitan Water Reclamation District of Greater Chicago (MWRD) East Calumet wastewater treatment and biosolids processing facility. Treatments applied to 20 experimental plots were (i) biosolids applied at 202 Mg ha^{-1} (2.5-cm layer, B1); (ii) biosolids applied at 404 Mg ha^{-1} (5.1-cm layer, B2); (iii) USCC certified hardwood vegetative compost applied at 137 Mg ha^{-1} (compost, 2.5-cm layer); (iv) a "blend treatment" (BL) consisting of MWRD biosolids applied at 202 Mg ha^{-1}, drinking water treatment residual (WTR) material from Ogden Dunes, IN, applied at the rate of 10.3 Mg ha^{-1} (dry weight basis), and 5.7 Mg ha^{-1} of biochar; or (v) control. All treatments were incorporated by rototilling the top 15.2 cm of soil. After 1 yr of plot treatment, additional WTR was surface applied at 11.6 Mg ha^{-1} to the bottom (i.e., downslope) 25% of the plot area in the BL area.

Fabricated steel liner borders were installed for each experimental plot. A rainfall runoff collection system for each plot was established by channeling runoff water from each plot into 3.8-L, high-density polyethylene (HDPE) bucket suspended in a 210-L HDPE barrel sunken into the ground. Plots were seeded with a restoration mix of six prairie grasses, three legumes, and 26 forbs in spring 2010 (Busalacchi, 2012). Soil ecosystem services were evaluated throughout the 3-yr study, including (i) improvements in soil quality focused on nutrient pools that support plant growth, C sequestration, and soil microbial functions that support plant/animal debris decomposition and nutrient recycling; (ii) establishment of a high-quality, diverse, native vegetative cover; (iii) improvements in earthworm quantity and quality; and (iv) the potential impact of biosolids and vegetative compost treatments on surface water quality evaluated by measuring concentrations of nutrients, trace metals, and several organic microconstituents in stormwater runoff.

Selected properties measured for site soil, amendments, and components used to make amendments included pH; oxalate-extractable Fe, Al, and P (Fe_{ox}, Al_{ox}, and P_{ox}) (McKeague and Day, 1996); total P; total C; total organic C and N; and P saturation index (PSI) calculated from PSI (%) = $[(P_{ox})/(Al_{ox} + Fe_{ox})] \times 100\%$.

Soil Quality

Soil samples were collected at 1 and 2 yr after plot establishment. The vegetative layer was removed from the top 6 cm, and soil was collected from a depth of 7 to 12 cm. All samples were transported within 24 h to the lab, where they were frozen at −20°C until analysis.

Salinity was determined by measuring electrical conductivity, soil pH was determined in 1:2 soil:0.01 mol L^{-1} $CaCl_2$ solution by glass electrode, total metals were determined by USEPA 3051A digestion, and subsequent analysis was performed by inductively coupled plasma–atomic emission spectroscopy (ICP–OES). Total N and C were determined by dry combustion, and total organic C was determined by heated dichromate digestion and colorimetry (Basta et al., 2012). Plant-available P, K, Ca, Mg, Mn, Cu, and Zn amounts were determined by Mehlich-3 soil extraction (Mehlich, 1984). Six enzyme assays (arylsulfatase, acid phosphatase, β-glucosaminidase, β-glucosidase, fluorescein diacetate hydrolysis, and urease) were performed to evaluate nutrient cycling and organic matter transformations. Soil collection, preservation, processing, and analysis for enzymatic activity is described in Carlson et al. (2015) and Tvergyak (2012).

Vegetative Performance and Quality

After treatment application on 26 Apr. 2010, plots were treated with glyphosate to kill broadleaf weeds. After hoeing, raking, and weeding, a native seed mix containing grasses, legumes, and forbs was planted on 24 May 2010. Seed was mixed with sand and weighed for precise application and was hand broadcast to all plots. Cellulose seed germination mats were then rolled onto and staked on top of the plots to protect seeds and improve germination. On 15 Sept. 2010, a second planting of native seed mix was broadcast on all plots.

More than 90% of plant species were identified in Year 3. A Shannon–Weaver diversity index was calculated (Eq. [1]) (Ludwig and Reynolds, 1988).

$$H' = -\sum p_i \ln(p_i) \qquad [1]$$

where H' is the Shannon–Weaver diversity index (a low value expresses less diversity as a relative portion of the group compared with the whole population) and p_i is the relative abundance of each group of organisms.

An index for species evenness was calculated (Magurran, 1988) (Eq. [2]).

$$E = \frac{e^D}{s} \qquad [2]$$

where e is the natural ln of 2.7; E is evenness, a measure of population distribution of species; D is the Shannon–Weaver function expressed as H'; and s is the number of species in a sample.

Plant samples were collected in September of Years 2 and 3 of the experiment. Transects were drawn across each plot from southeast to northwest corner. Plants were collected from three 1-m^2 subplots. Total plant biomass (dry weight basis) in g m^{-2} was measured for each plot. The nutrient and trace element content of plant tissue was determined in duplicate by dry combustion for N and acid digestion and subsequent analysis by ICP–OES for other nutrients.

Earthworm Performance

Earthworm [*Eisenia fetida* (L.)] 56-d reproduction bioassays were conducted in accordance with Environment Canada (2007). Deionized water was added to plot soils to achieve approximately 70% water holding capacity. Jars were capped

with aluminum or nylon screens (fine mesh) screwed on with ring tops. This allowed for greater gas exchange to compensate for CO_2 evolution from the biosolids treatments and suffocation of the earthworms. Test chambers were maintained under continuous fluorescent lighting at 20 ± 2°C. In each treatment vessel, worms were fed 5 mL hydrated, cooked Quaker Quick Oats every 14 d. Moisture was checked and adjusted every few days, and the condition of the worms was noted. Earthworm mortality, reproduction, and conditions were recorded on Days 14, 28, and 56. Juveniles and cocoons were counted and recorded on Days 28, 42, and 56. Bioassays were terminated on Day 56. Adult worms were depurated for 24 h on moist tissue paper, rinsed with deionized water, and frozen at −80°C in falcon tubes until further analysis.

Storm Water Runoff Quality

Recessed barrels were placed at the end of screened pipes at the bottom of each runoff plot. Runoff samples were collected in 3.8-L HDPE buckets suspended in 208-L HDPE barrels. Total volume of water collected per rain event was recorded with a calibrated measuring rod. Subsamples of <2 L were pumped from buckets, placed in coolers with ice packs, and shipped to our lab where they were frozen at −20°C until analysis. Samples of stormwater runoff from the first rainfall event after plot treatment (i.e., first flush) were collected on 23 Oct. 2009. Samples collected subsequent to first flush were collected from each precipitation event, composited quarterly each season (autumn, winter, spring, and summer), and collected through summer 2011. The composite sample was split into an unfiltered and filtered (0.7-μm glass fiber Whatman filter) sample. Flow-weight composite unfiltered and filtered samples, prepared for each plot-quarterly season, were analyzed as follows. Total suspended solids (TSS), electrical conductivity, and total N and P were measured on whole, unfiltered water samples. Unfiltered water samples were subjected to a persulfate digestion. with subsequent analysis of N and P by ion chromatography. Dissolved ammonium, nitrite, nitrate, P, and trace metals were determined for composite filtered water samples. Ammonium N was determined on filtered runoff water by a modified USEPA Nessler Method 8038. Nitrate (NO_3^-) was determined potentiometrically using an Orion Nitrate Ion Selective combination electrode. Nitrite N (NO_2^-–N) was determined on filtered runoff water and read colorimetrically by a modified Griess–Ilovsay method (Keeney and Nelson, 1982). Runoff samples were analyzed for Al, As, Ba, B, Cd, Co, Cr, Cu, Fe, Mn, Mo, Ni, P, Pb, Sb, Se, Tl, V, and Zn by ICP–OES.

Analysis of microconstituents (i.e., pharmaceutical and personal care products) in filtered runoff water was performed by Axys Analytical Laboratories using Method MLA-075 (Axys Analytical Services, 2010), a modified USEPA Method 1694. Nonylphenol ethoxylates analysis in runoff was performed by Axys Analytical Laboratories using MLA-004 (Axys Analytical Services, 2010).

Data Analysis

The experimental design was five treatments with four replicates per treatment. One-way ANOVA was conducted. If ANOVA results were significant, multiple means comparisons were conducted by Fisher LSD. The level of significance for all statistical comparisons was $\alpha = 0.10$. All statistical analyses were conducted by Minitab Statistical Software, Release 15 for Windows (Minitab Inc., 2007).

Results and Discussion

The soil at the experimental site was calcareous and had a pH of 7.39 (Table 1). The site soil had a substantial organic C content of 37.4 g kg^{-1}. The site was located slightly upslope from a small marsh and had a high water table, which may account for accumulated soil organic C. The soil had adequate plant nutrients for establishing vegetation but was deficient in plant-available P. The soil test values of 10.6 mg kg^{-1} Mehlich-3 P were below the desirable minimum value of 30 mg kg^{-1} recommended for plant growth (Johnson et al., 2000). Biosolids contained more reactive Fe and Al oxides than other soil amendments or components because of the use of Al and Fe salts in advanced wastewater treatment process by the City of Chicago.

Soil Quality

Soils were analyzed for 3 yr (Basta et al., 2012; Busalacchi, 2012) during the study, but for brevity only Year 3 results are presented (Table 2). With the exception of P, plant-available nutrient concentrations were adequate for the untreated control soil (Basta et al., 2012; Busalacchi, 2012) (Table 2). However, increasing plant-available N increases plant growth and vegetative cover. Biosolids are very rich in N and are often land applied to provide plant-available N for crop production. Soil N was greater in the biosolids and compost treatments than in the con-

Table 1. Selected properties for site soil, amendment, and components used to make amendments.†

Material	pH‡	P_{ox}§	Al_{ox}§	Fe_{ox}§	PSI¶	Total P	Total C	TOC#	Total N
		g kg^{-1}				mg kg^{-1}	g kg^{-1}		
Site soil	7.39	0.21	1.28	5.13	0.05	465	–	37.4	1.62
Biosolids	6.20	19.6	5.70	60.2	1.06	18,730	201	201	15.7
Compost	7.98	1.57	0.48	2.20	0.83	2,797	161	161	10.1
WTR††	6.94	0.61	1.03	71.7	0.02	–	97.2	–	4.29
Biochar	7.25	0.10	0.10	0.10	2.66	281	772	–	0.00

† All results are expressed on a dry weight basis.

‡ 1:2 soil: 0.01 mol L^{-1} $CaCl_2$.

§ Oxalate-extractable P, Al, and Fe.

¶ Phosphorus saturation index (mol P/mol Al + Fe).

Total organic C.

†† Water treatment residual.

trol plots. Increased soil N pools in biosolids plots resulted in greater plant biomass production (Basta et al., 2012; Busalacchi, 2012). This is consistent with the higher N content of the biosolids (15.7 g kg^{-1}) compared with the compost (10.1 g kg^{-1}) (Table 1).

Biosolids contained 18.7 g kg^{-1} total P, which was much greater than compost (2.80 g kg^{-1}) (Table 1). Both compost and biosolids treatments increased plant-available Mehlich-3 P and corrected potential soil P deficiencies in site soil (Table 2). Because of Fe and Al additions during wastewater treatment, biosolids had much more reactive Fe and Al oxide than compost. The PSI is directly related to P solubility and to the runoff threat to surface water. Elliott et al. (2002) reported that biosolids with a PSI > 1.1 were correlated with offsite P runoff risk. The MWRD biosolids had a PSI of 1.06 (Table 1). The BS1 treatment plots resulted in a soil with a PSI of 0.79, whereas the higher biosolids application in the BS2 treatment resulted in a PSI of 1.15, which suggests potential P runoff problems. The addition of WTR to the 1-inch BL reduced the PSI from 0.79 to 0.65 and decreased potential offsite P runoff risk significantly (Table 2). Compost treatment had a PSI of 0.14, which suggests a minimal concern for P runoff (Table 2).

Biosolids, but not compost, increased plant-available pools of micronutrients such as Cu and Zn (Table 2). Soil treatments did not increase micronutrient Fe or Mn in soils (Basta et al., 2012; Busalacchi, 2012) (Table 2). Biosolids had no effect or had small increases in the soil content of most trace elements and metals compared with compost and the control soil (Table 3). The most significant increases due to biosolids application were in soil Cu and Zn. This is not surprising because Cu and Zn are usually elevated in biosolids due to household plumbing and other domestic or industrial inputs of these metals. These soil concentrations of Cu and Zn (Table 3) are well below the levels of concern. However, biosolids increased plant-available pools of these micronutrients. Slightly elevated levels of Cu and Zn from biosolids do benefit the terrestrial food web because they are essential nutrients to plants and other forms of life. Biosolids application increased plant-available Mehlich-3 Fe, Cu, and Zn of the site soil (Table 2). Compost increased plant-available Mehlich-3 Fe but not Cu and Zn.

Soil organic C values were BS2 > BS1, BL ≥ compost ≥ control (Table 2). More total organic C was found in the BS plots than in the compost and control plots. Additional monitoring is

Table 2. Selected chemical properties and plant-available nutrients of control soil and amended plot soils.

Parameter	Control	Soil amendments†			
		Compost	B1	B2	Blend
Soil pH	7.44c‡	7.39c	7.06b	6.80a	7.12b
TOC§ g kg^{-1}	47.8a	62.0ab	71.9bc	86.7c	71.9bc
Total N g kg^{-1}	2.25a	4.52b	5.17bc	7.10c	4.38b
Mehlich-3 P	10.6a	78.0a	679bc	821c	598b
PSI¶	0.05a	0.14a	0.79bc	1.15c	0.65b
Mehlich-3 K	140a	221b	144a	154a	154a
Mehlich-3 Cu	10.8a	9.50a	25.0bc	28.7c	23.8b
Mehlich-3 Fe	157a	193bc	241c	241c	255c
Mehlich-3 Mn	53.0a	61.7a	58.5a	53.2a	62.4a
Mehlich-3 Zn	15.7a	20.2a	85.6b	107b	87.5b

† B1, biosolids applied at 202 Mg ha^{-1}; B2, biosolids applied at 404 Mg ha^{-1}.

‡ Treatments means within a row measured with same letter are not different at $P < 0.10$.

§ Total organic C.

¶ Phosphorus saturation index.

Table 3. Trace element metal content of control soil and amended soils.

Element	Control	Soil amendments†			
		Compost	B1	B2	Blend
	mg kg^{-1}				
As	11.4a‡	12.1a	12.7ab	13.5b	14.0b
Ba	128a	130a	189c	210c	182b
Be	0.37c	0.12a	0.23b	0.20b	0.28bc
Cd	0.98a	0.88a	1.29bc	1.43c	1.19ab
Co	3.26b	2.43a	2.52a	2.40a	2.43a
Cr	31.4a	28.4ab	393c	50.8d	36.4bc
Cu	38.3a	39.5a	109bc	124c	89.3b
Mo	4.72ab	4.21a	5.40b	7.03c	5.12b
Ni	26.7c	23.8a	27.5c	28.7c	24.9b
Pb	50.3a	52.4a	70.7b	70.7b	62.6ab
V	40.6b	34.1a	36.2ab	37.4ab	37.5ab
Zn	110a	121a	277bc	311c	241b

† B1, biosolids applied at 202 Mg ha^{-1}; B2, biosolids applied at 404 Mg ha^{-1}.

‡ Treatments means within a row measured followed by the same letter are not different at $P < 0.10$.

needed to determine the long-term effects of biosolids and compost additions on sequestration of SOC.

Application of organic amendments increased soil enzymatic activities, with the exception of urease (Table 4). Urease was unchanged by soil amendments (Table 4). Total enzymatic activity decreased with time likely because the amendments had largely been depleted of available substrates. However, the enzymatic activity in treated soils was still greater than in the control soils at the end of the study, with the exception of urease. These results are similar to other studies that have shown that microbial properties of remediated soils tend to decrease to near background levels after a few years, especially when treatment applications are not reoccurring (Speir et al., 2003; García-Gil et al., 2004).

Arylsulfatase is an extracellular enzyme that catalyzes the hydrolysis of organic sulfate esters and releases plant-available SO_4. Most organic amendments increased arylsulfatase activity compared with the control soil (Table 4). Phosphatases are responsible for the mineralization of organic P in soils (Juma and Tabatabai, 1977). Acid phosphatase is an indicator of the potential of soil to release phosphate from organic matter in soils. In general, all organic amendments increased phosphatase activity compared with the control soil (Table 4). β-Glucosaminidase hydrolyzes N-acetyl-β-D-glucosamine, a component of chitin, the second most abundant polymer on earth. Chitin is found in cell walls of insects and fungi. In the process, chitin is converted to amino sugars and releases inorganic N in soils (Ekenler and Tabatabai, 2002). β-Glucosaminidase showed increased levels of activity for the BS treatments in comparison to compost (Basta et al., 2012). In general, all organic amendments increased β-glucosaminidase activity compared with the control soil (Table 4). The largest increases were associated with biosolids treatments after application. β-Glucosaminidase activity in the BS treatments decreased until it was equal to compost at the last sampling of August 2011. The increase in β-glucosaminidase activity in our experiment may represent the stimulation of fungal biomass by mainly biosolids treatments. However, fungal and other microbial populations were equal between biosolids and compost treatment after 3 yr (Carlson et al., 2015). β-Glucosidase plays a major role in microbial metabolism by releasing low-molecular-weight sugars that serve as energy sources (Tabatabai, 1994; Bandick and Dick, 1999). It is involved in the final step of cellulose degradation and is sensitive to residue management (Bandick and Dick, 1999; Acosta-Martinez et al., 2003). Most organic amendments increased glucosidase activity compared with the control soil after 2 yr (Table 4). Pérez de Mora et al. (2005) found increased β-glucosidase activity in a municipal compost compared with a biosolids-treated soil. Conversely, this study found little difference between the biosolids-treated compost and the BL soils. This indicates similar substrate availability in each of the treatments regardless of application rate or amendment source. Many enzymes in soil can hydrolyze fluorescein diacetate. These enzymes can be classified as lipases, esterases, and proteases. For these reasons, β-glucosidase activity is a good indicator of overall biological activity (Schnürer and Rosswall, 1982; Dick, 1994). More specifically, fluorescein diacetate hydrolysis is important for determining the decomposition of fungal and bacterial biomass (Schnürer and Rosswall, 1982). In general, treatments increased fluorescein diacetate hydrolysis activity.

Vegetative Performance and Quality

Biosolids treatments had a greater production of plant biomass than the compost plots (Table 5). Both biosolids and compost produced more plant biomass than the control plots. These differences were visible, with much more vegetative cover on the biosolids treatment than the compost or control. Plant N, P, and K nutrient plant tissue levels were directly related to the plant biomass. Plant macronutrients (N, P, K) were significantly higher in plant tissue of biosolids-treated plots compared with compost and control (Table 5). Nutrient-rich biosolids boosted plant nutrient levels and biomass compared with the nutrient-poor compost. Because N is a component of amino acids and proteins, plant tissue N is used to estimate plant protein content. High plant protein content is an important indicator of food quality for grazing animals and the terrestrial food web. Higher plant N is consistent with higher protein content in plant tissue associated with the biosolids treatments. Several trace elements that are not essential for plant development (e.g., Pb and Cd) can cause phytotoxicity in excessive amounts. Plant tissue analysis showed that the concentration levels of all these trace elements (results not shown) (Basta et al., 2012; Busalacchi, 2012) are below potential plant toxicity ranges (Kabata-Pendias, 2001). Biosolids did not increase the tissue concentration of these trace elements or degrade the quality of plant vegetation as ecosystem forage or feed.

A comparison of the total number of all species counted and comparisons between the plant species planted to the invasives are summarized in Table 6 and in Busalacchi (2012). Grasses

Table 4. Response of arylsulfatase, acid phosphatase, β-glucosaminidase, β-glucosidase, fluorescein diacetate hydrolysis, and urease after 3 yr of soil amendment.

Enzyme	Control	Soil amendments†			
		Compost	B1	B2	Blend
	mg kg^{-1} h^{-1}				
Arylsulfatase	67.3a‡	147b	107b	98.7ab	123b
Acid phosphatase	205a	388b	374ab	470b	425b
β-Glucosaminidase	20.0a	60.3ab	69.1b	75.0b	85.2b
β-Glucosidase	46.7a	111b	90.5ab	99.2ab	124b
FDA§	2.45a	3.42ab	4.10ab	5.00ab	5.36b
Urease	63.1a	76.1a	77.2a	84.9a	84.8a

† B1, biosolids applied at 202 Mg ha^{-1}; B2, biosolids applied at 404 Mg ha^{-1}.

‡ Treatments means within a row measured followed by the same letter are not different at $P < 0.10$.

§ Fluorescein diacetate hydrolysis.

Table 5. Plant biomass and plant tissue macro- and secondary nutrients.

Biomass	Control	Soil amendments†			
		Compost	B1	B2	Blend
	kg ha^{-1}				
	2080a‡	2990a	7440c	4590b	5660b
Plant tissue content					
	g kg^{-1}				
N	9.10a	10.3a	22.5b	27.4c	24.5b
Ca	16.1a	14.7a	16.0a	13.1a	15.7a
K	11.3a	12.7a	17.6b	20.0b	19.1b
Mg	2.80a	2.66a	4.31b	4.16b	4.37b
P	1.57a	2.44b	3.40c	3.62c	3.56c
S	5.82b	5.44b	4.44b	3.47b	3.09a

† B1, biosolids applied at 202 Mg ha^{-1}; B2, biosolids applied at 404 Mg ha^{-1}.

‡ Treatments means within a row measured followed by the same letter are not different at $P < 0.10$.

were the predominant vegetative group on the control plots. Grass height of 15 to 30 cm was the lowest in the control compared with compost and biosolids. Biosolids-treated plots had the highest plant density, the greatest growth (120–200 cm), and the most plant species diversity of both native and invasive species. Results were grouped into grasses and forbs. Legumes were grouped with the grasses because there was only one species observed. A Shannon–Weaver index was calculated for each plot at the end of Year 3 (Table 7). Three years after planting is a short time for establishment of native prairie forbs and grasses (Webb, 1997; Walker and Del Moral, 2003) but should indicate the trajectory of the plant community in the near term. All biosolids treatment plots (B1, B2, and BL) showed a higher Shannon–Weaver index than control plots in the grass group. For the forbs group, control plots had the lowest diversity index, and the biosolids blend had the highest diversity index. Both BL

Table 6. Total number and percent of plant species by group for all runoff plots on the study site.

Group	Species	All plots (*n*)	Type†	Percentage of group	Total group (*n*)
Grasses	little bluestem	217	gp	63.8	
	Canadian wild rye	94	gp	27.6	
	big bluestem	14	gp	4.1	
	prairie cordgrass	12	gp	3.5	
	crabgrass	2	ig	0.6	
	tall dropseed	1	gp	0.3	340
Legume	Illinois tickfoil	1	lp	100	1
Native forbs	autumn sneezeweed	147	fn	31.7	
	yellow coneflower	102	fp	22.0	
	black-eyed Susan	99	fp	21.4	
	Canadian goldenrod	30	fn	6.5	
	Canadian horseweed	26	an	5.6	
	smooth ironweed	13	fp	2.8	
	wild bergamot	8	fp	1.7	
	common sunflower	6	fn	1.3	
	daisy fleabane	6	fn	1.3	
	wild lettuce	6	fn	1.3	
	tall coreopsis	4	fp	0.9	
	brown-eyed Susan	4	fp	0.9	
	rosinweed	3	fp	0.6	
	common milkweed	3	fp	0.6	
	New England aster	2	fp	0.4	
	gray goldenrod	2	fp	0.4	
	compass plant	1	fp	0.2	
	common evening primrose	1	fn	0.2	463
Invasives	purple loosestrife	46	ip	65.7	
	unknown invasives	19	ip	27.1	
	knapweed	2	ip	2.9	
	lambs quarters	2	ia	2.9	
	curly dock	1	ip	1.4	70
	totals				874

† Fn, forb native; gp, grass planted; ia, invasive annual; ip, invasive perennial; lp, legume planted.

Table 7. Comparison of selected plant species indexes and measures between control soil and amended soils.

Group	Control	Soil amendments†			
		Compost	B1	B2	Blend
		Shannon–Weaver index values			
Grasses	1.22a‡	1.83ab	2.13b	2.39b	1.95b
Forbs	3.06a	4.41bc	3.75ab	4.37abc	5.24c
		Species evenness index			
Grasses	2.39a	2.47a	3.84bc	4.14c	3.18ab
Forbs	5.37a	9.93a	13.1a	18.0ab	31.1b
		Total, different species			
Grasses	2a	4a	3a	4a	3a
Forbs	13a	10a	10a	12a	15a
		Total, all species			
Grasses	54a	79b	79b	80b	46a
Forbs	108ab	134ac	94ab	88a	109b
		Occurrence PL§ as % total (Forbs)			
Forbs	14a	10a	11a	6a	5a

† B1, biosolids applied at 202 Mg ha^{-1}; B2, biosolids applied at 404 Mg ha^{-1}.

‡ Means within parameter measured followed by the same letter are not different (LSD; $p < 0.10$).

§ Purple loosestrife.

and compost had high levels of stable C (i.e., cellulosic and charcoal) materials. In the case of the BL, this was perhaps due to the biochar addition in this treatment. Studies have linked biochar soil amendment to plant native species diversity by stimulation of beneficial microbial interactions. For example, Kulmatiski (2011) showed that additions of activated charcoal, the principal component of biochar, restored native plant dominance to an old farm field, compared with native seed planting alone, as well as microbial community changes that were linked to native plant success.

Purple loosestrife (*Lythrum salicaria* L.) is a major invasive species that is spreading across many habitats in the United States. An invasive from Eurasia, it crowds out native plants and has negative impacts on native pollinators. Purple loosestrife was observed at our field site before plot installation and was then found in control and all amended plots. The occurrence of purple loosestrife was similar across all treatments and in the untreated control soil (Table 7). The Species Evenness Index is a measure of the evenness of the species distribution within a treatment. The biosolids treatments (B1 and B2) had higher evenness in the grasses group compared with compost, BL, and control soil. Only the BL had significantly higher evenness values for Forbs. The Species Evenness Index is expected to increase as the site moves to a more mature, established climax community structure of a mesic, intermediate height prairie with mixed wetland woody species in hydric locations.

Earthworm Performance

Soil treatments had no effect on earthworm mortality (Fig. 1). Soil amendments did affect reproductive endpoints of juveniles and cocoons. Earthworm cocoon counts are a measure of potential reproductive success, but the more important juvenile counts are a measure of actual reproductive success. The average number of juveniles was significantly higher across all treatments compared with the control soil. There was no significant difference between the number of cocoons in amended soils and the untreated control soil. Similarly, when cocoons and juvenile counts are measured as a combined measure of reproductive success, there were no significant differences among treatments. Both biosolids and compost treatments increased the number of juveniles and earthworm reproductive success.

Stormwater Runoff Quantity and Quality

Total runoff for all events was summarized quarterly for each plot (Fig. 2). In 2010, total precipitation (95 cm) was slightly above the area annual historical average of 87 cm for the region (ClimateStations.com, 2014). With total precipitation of 127 cm, 2011 was the second wettest year recorded since 1871. Most of the record amounts of rainfall in 2011 occurred during the first half of the year, which was the collection period for 2011 for our study. Runoff volume was 150 times higher in 2011 compared with 2010. The first runoff event (23 Oct. 2009) was also a single collection event; the rest were totals of several events for each seasonal interval.

Total suspended solids were measured on a concentration and mass basis. The first runoff event on 23 Oct. 2009 had the high-

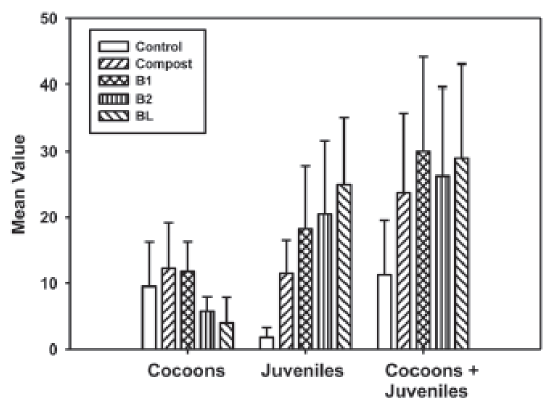

Fig. 1. Earthworm reproductive performance in amended and control soils. Soil treatments are Control, Compost, B1 (biosolids at 202 Mg ha^{-1}), B2 (biosolids at 404 Mg ha^{-1}), BL (blend consisting of biosolids applied at 202 Mg ha^{-1}), drinking water treatment residual, and biochar.

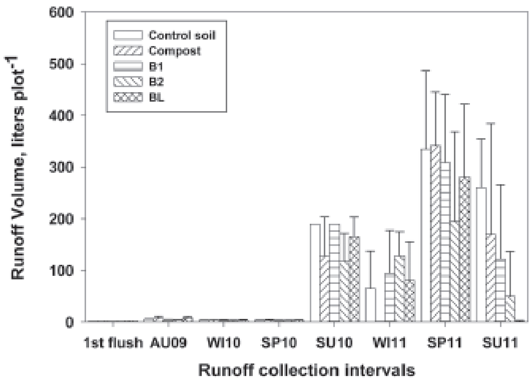

Fig. 2. Storm water runoff volume from experimental plots. Soil treatments are Control, Compost, B1 (biosolids at 202 Mg ha^{-1}), B2 (biosolids at 404 Mg ha^{-1}), BL (blend consisting of biosolids applied at 202 Mg ha^{-1}), drinking water treatment residual, and biochar. AU09, autumn 2009 (October–December); WI10, winter 2010 (January–March); SP10, spring 2010 (April–June); SU10, summer 2010 (July–September); WI11, winter 2011; SP11, spring 2011; SU11, summer 2011.

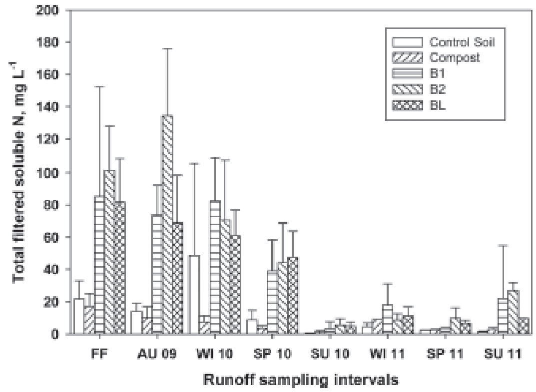

Fig. 3. Soluble N in storm water runoff. Soil treatments are Control, Compost, B1 (biosolids at 202 Mg ha^{-1}), B2 (biosolids at 404 Mg ha^{-1}), BL (blend consisting of biosolids applied at 202 Mg ha^{-1}), drinking water treatment residual, and biochar. FF, first flush (23 Oct. 2009). Sampling intervals are defined in Fig. 2.

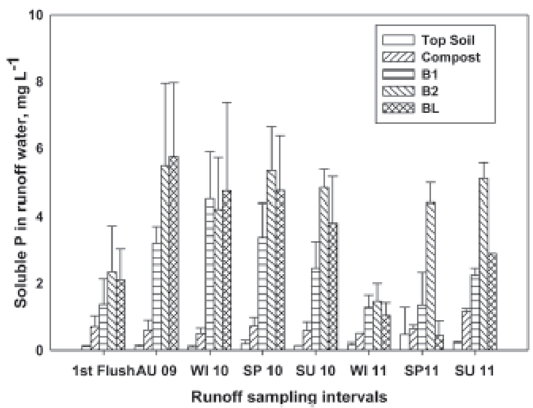

Fig. 4. Soluble P in storm water runoff. Soil treatments are Control, Compost, B1 (biosolids at 202 Mg ha^{-1}), B2 (biosolids at 404 Mg ha^{-1}), BL (blend consisting of biosolids applied at 202 Mg ha^{-1}), drinking water treatment residual, and biochar. Sampling intervals are defined in Fig. 2.

est concentration of >4 mg kg^{-1} TSS. The plots had been freshly rototilled, and the uncovered surface did not have any erosion control. Vegetation was not yet growing on the plots, so there was significant sediment loss carried into the runoff. The concentration of TSS declined markedly with time to background levels of <2 mg kg^{-1} by the summer of 2010; TSS values were not significantly different among treatments.

Total N in runoff water was determined for both filtered (soluble) and total (unfiltered) water samples. The largest losses of soluble N were in the first half of the study (Fig. 3). Similar trends were found for unfiltered total N losses (Basta et al., 2012; Busalacchi, 2012). Vegetation was not yet growing on the plots, so there was significant loss of N in sediment, and soluble forms of N easily leached from organic treatments, especially N-rich biosolids treatments. The concentration of N declined markedly with time, approaching background levels by the summer of 2010, which were not significantly different among treatments.

Dissolved P (Fig. 4) and total P (results not shown) (Basta et al., 2012; Busalacchi, 2012) in runoff water followed the trend B2 ≥ B1, BL > compost ≥ control soil. Dissolved P could be explained by the amount of total P added to plots. The highest dissolved P was associated with the highest application of biosolids (B2) and the lowest with the compost in runoff water. Dissolved P amounts from biosolids treatments were greater than the compost because of the P content of these materials (Table 1). Similar dissolved P loss was found for the BS1 and the BL treatments (Fig. 4). The B1 and BL treatments had the same amount of biosolids, but the BL treatment contained WTR. Results suggest that WTR had no effect on reducing dissolved P runoff. Previous studies have shown WTR can significantly reduce P runoff when blended with biosolids or animal manures (Dayton and Basta, 2005). Apparently not enough WTR was added to reduce dissolved P. Unlike soluble N, increases in dissolved runoff P for soil amendments persisted throughout the experiment (Fig. 4). However, the amount of P (i.e., mass) lost from the treatments in runoff water was almost entirely (>90%) associated with solids washed off the plots (organic P and soil P) and not dissolved P (Basta et al., 2012; Busalacchi, 2012). Most P lost from plots was from erosion from before vegetation was established (Fig. 5). However, after vegetation was well established and erosion losses were reduced, P loss decreased to background levels in 2011 (Fig. 5).

Soluble metals concentrations were at or below detection limits throughout the study. With the exception of 56.0 μg L^{-1} Cu for the BS2 treatment, all runoff metal values were below the Calumet Ecotoxicology Protocol benchmark values (Basta et al., 2012; Busalacchi, 2012; Calumet Ecotoxicology Roundtable Technical Team, 2007). Several microconstituents were detected in the control soil, compost, and biosolids (Basta et al., 2012; Busalacchi, 2012). Thus, it was expected that some microconstituents would be present in runoff water. Microconstituents were determined in the first rainfall event (i.e., first flush) because this would likely be the highest concentration in runoff water and would simulate a worse-case scenario. Runoff water samples collected from the plots were analyzed for analgesics, antihistamines, antipsychotics, analeptics, antibacterial and antimicrobial

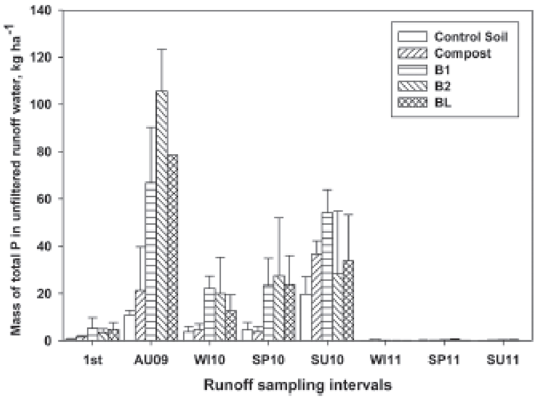

Fig. 5. Total P in storm water runoff. Soil treatments are Control, Compost, B1 (biosolids at 202 Mg ha^{-1}), B2 (biosolids at 404 Mg ha^{-1}), BL (blend consisting of biosolids applied at 202 Mg ha^{-1}), drinking water treatment residual, and biochar. Sampling intervals are defined in Fig. 2.

compounds, illicit drugs, metabolites, insecticides, rodenticides, stimulants, diuretics, and steroidal drugs, among others (Basta et al., 2012; Busalacchi, 2012). Of the 119 compounds tested, 20 were detected at the ng L^{-1} level, and only four compounds were detected in runoff from all treatments, including the compost-treated and control soil plots. Five compounds (carbamazepine, gemfibrozil, ibuprofen, N,N-Diethyl-meta-toluamide [DEET], and valsartan) were detected at concentrations 10 times greater than the detection limit (Table 8). The B2 plot typically had the highest levels of several of these compounds, with ibuprofen detected at 1.76 μg L^{-1}. The control and compost plots also contained significant levels of ibuprofen. One of the control soil plots had the highest levels of DEET, but all plots across all soil amendments had measureable amounts of DEET in runoff water because it was frequently used by students and other personnel as a mosquito repellant during site preparation and plot establishment. Gemfibrozil, carbamazepine, ibuprofen, naproxen, and triclocarban did not exceed the no observable adverse effect level (NOAEL) (Table 8). A hazard quotient (HQ) was calculated according to Waiser et al. (2011). The HQ was determined by dividing the maximum environmental concentration for the plot by a probable no-effect concentration provided by Waiser et al. (2011). Hazard quotient values were calculated for all of the compounds for which a probable no-effect concentration was provided. All microconstituent runoff concentrations had a HQ <1.0, indicating low to no risk to aquatic receptors. Only ibuprofen had a HQ >1 of 1.07. However, the ibuprofen concentration of 1760 ng L^{-1} was below the chronic NOAEL of 5000 ng L^{-1} (Lin et al., 2008), indicating low risk to aquatic receptors. Nonylphenol occurred at 242 ng L^{-1} in runoff from the B2 treatment. This level is below the recommended chronic NOAEL and National Ambient Water Quality level of 5900 ng L^{-1} (Staples et al., 2004).

Conclusions

The initial goals of site restoration should maximize habitat by rapid development of the microbiological soil community and plant assemblies while minimizing contaminant exposure to plant and animal communities and their environment. The organic amendments used in this study allowed rapid development of microbiological soil community and plant assemblies. Biosolids, compost, and biosolids-blend greatly increased soil ecosystem services, including (i) improved soil quality, (ii) rapid establishment of a dense and high-quality vegetative cover and placing the restored site on a trajectory of establishment of native plant populations, and (iii) improved soil earthworm populations and improved terrestrial food web. In general, biosolids outperformed compost by providing greater improvements in soil quality, vegetative performance and quality, earthworm performance and quality, and microbial ecology and function. Biosolids did affect surface water quality more than compost in the first year of the experiment. During this time, loss of soil and organic solids in runoff water resulted in much greater loss of N and P from biosolids-treated plots than after establishment of vegetation. However, most of the nutrient loss was greatly decreased once the vegetation was established. Future restoration design should ensure that runoff control measures, such as cover crop, mulch, or construction of berms, are used to control sediment loss from the restored sites at least until vegetation is established. The use of WTRs can control dissolved P after vegetation is established, but application of WTRs with more P sorption potential is needed than that used in this study. Runoff losses of trace elements and heavy metals were negligible. Several organic microconstituents (i.e., pharmaceutical and personal care products) were detected in runoff water. However, the concentrations

Table 8. Microconstituents in storm water runoff.

Microconstituent	Control soil	Soil amendment			NOAEL‡	HQ§
		Compost	B2†	Blend		
	ng L^{-1}					
Carbamazepine	BDL¶	BDL	66.0–206	BDL	25,000	0.02
DEET#	57.9–420	57.9–86.5	43.0–154	58.2–176	NA††	0.004
Gemfibrozil	3.41–15.0	7.05–84.0	35.8–119	90.3–324	100,000	0.13
Ibuprofen	BDL–202	89.7–568	527–1760	854–1490	5,000	1.07
Valsartan	BDL–17.3	BDL–78.0	58.4–200	102–233	NA	NA

† Biosolids applied at 404 Mg ha^{-1}.

‡ No observable adverse effect level.

§ Hazard quotient.

¶ Below detection limit.

N,N-Diethyl-meta-toluamide.

†† Not available.

of these organic microconstituents in runoff water were below the probable no-effect levels and should pose little impact on the aquatic environment.

Acknowledgments

The authors thank the Metropolitan Water Reclamation District of Greater Chicago for providing funding and personnel, equipment, and other support for this project. Other support included salary for N.T. Basta by state and federal funds appropriated to the Ohio Agricultural Research and Development Center, The Ohio State University, Columbus, OH.

References

Acosta-Martinez, V., T.M. Zobeck, T.E. Gill, and A.C. Kennedy. 2003. Enzyme activities and microbial community structure in semiarid agricultural soils. Biol. Fertil. Soils 38:216–227. doi:10.1007/s00374-003-0626-1

Axys Analytical Services. 2010. www.axysanalytical.com (accessed 6 Sept. 2015).

Bandick, A.K., and R.P. Dick. 1999. Field management on soil enzyme activities. Soil Biol. Biochem. 31:1471–1479. doi:10.1016/S0038-0717(99)00051-6

Basta, N.T. 2000. Examples and case studies of beneficial reuse of municipal by-products. In: J.F. Power, editor, Land application of agricultural, industrial, and municipal by-products. SSSA Book Ser. 6. SSSA, Madison, WI. p. 481–504.

Basta, N.T., D.M. Busalacchi, R.P. Dick, J. Tvergyak, and R.P. Lanno. 2012. Final report: Ecosystem services study of degraded soils amended with biosolids, Metropolitan Water Reclamation District of Greater Chicago, Chicago, IL.

Brown, S., M. Mahoney, and M. Sprenger. 2014. A comparison of the efficacy and ecosystem impact of residual-based and topsoil-based amendments for restoring historic mine tailings in the Tri-State mining district. Sci. Total Environ. 485-486:624–632. doi:10.1016/j.scitotenv.2014.03.029

Busalacchi, D. 2012. Evaluation of biosolids as a soil amendment for use in ecological restoration. M.S. thesis, The Ohio State University, Columbus, OH.

Calumet Ecotoxicology Roundtable Technical Team. 2007. Calumet area ecotoxicology protocol. City of Chicago, Chicago, IL.

Carlson, J., J. Saxena, N. Basta, L. Hundal, and D. Busalacchi. 2015. Application of organic amendments to restore degraded soil: Effects on microbial properties. Environ. Monit. Assess. 187:1–15. doi:10.1007/s10661-015-4293-0

ClimateStations.com. 2014. Chicago annual precipitation: 1871-2014. www.climatestations.com/chicago/ (accessed 2 Jan. 2015).

Dayton, E.A., and N.T. Basta. 2005. Use of drinking water treatment residuals as a potential best management practice to reduce phosphorus risk index scores. J. Environ. Qual. 34:2112–2117. doi:10.2134/jeq2005.0083

Dick, R.P. 1994. Soil enzyme activities as indicators of soil quality. In: J.W. Doran, D.C. Coleman, D.F. Bezdicek, and B.A. Stewart, editors, Defining soil quality for a sustainable environment. SSSA Special Publication 35. SSSA, Madison, WI. p. 107–124.

Ekenler, M., and M.A. Tabatabai. 2002. β-Glucosaminidase activity of soils: Effect of cropping systems and its relationship to nitrogen mineralization. Biol. Fertil. Soils 36:367–376. doi:10.1007/s00374-002-0541-x

Elliott, H.A., G.A. O'Connor, and S. Brinton. 2002. Influence of water treatment residuals on phosphorus solubility and leaching. J. Environ. Qual. 31:1362–1369. doi:10.2134/jeq2002.1362

Environment Canada. 2007. Tests for toxicity of contaminated soil to earthworms (*Eisenia andrei, E. fetida and Lumbricus terrestris*). EPS 1/RM/43. Environment Canada, Gatineau, Quebec.

García-Gil, J.C., C. Plaza, N. Senesi, G. Brunetti, and A. Polo. 2004. Effects of sewage sludge amendment on humic acids and microbiological properties of a semiarid Mediterranean soil. Biol. Fertil. Soils 39:320–328. doi:10.1007/s00374-003-0709-z

Haering, K.C., W.L. Daniels, and S.E. Feagley. 2000. Reclaiming mined lands with biosolids, manures, and papermill sludges. In: R.I. Barnhisel, R.G. Darmody, and W. Lee Daniels, editors, Reclamation of drastically disturbed lands. Agron. Monogr. 41. ASA, CSSA, SSSA, Madison, WI. p. 615–644.

Johnson, G.V., R.R. Raun, H. Zhang, and F.A. Hattey. 2000. Oklahoma soil fertility handbook. Division of Agric. Sciences and Natural Resources, Oklahoma State Univ., Stillwater, OK.

Juma, N.G., and M.A. Tabatabai. 1977. Effects of trace elements on phosphatase activity in soils. Soil Sci. Soc. Am. J. 41:343–346. doi:10.2136/sssaj1977.03615995004100020034x

Kabata-Pendias, A. 2001. Trace elements in soil and plants. CRC Press, Boca Raton, FL.

Keeney, D.R., and D.W. Nelson. 1982. Nitrogen-inorganic forms. In: A.L. Page et al., editors, Methods of soil analysis. Part 2. 2nd ed. Agron. Monogr. 9. ASA and SSSA, Madison, WI. p. 643–698.

Kulmatiski, A. 2011. Changing soils to manage plant communities: Activated carbon as a restoration tool in ex-arable fields. Restor. Ecol. 19:102–110. doi:10.1111/j.1526-100X.2009.00632.x

Lin, A.Y., T.H. Yu, and C.F. Lin. 2008. Pharmaceutical contamination in residential, industrial, and agricultural waste streams: Risk to aqueous environments in Taiwan. Chemosphere 74:131–141.

Ludwig, J.A., and J.F. Reynolds. 1988. Statistical ecology. John Wiley & Sons, New York.

Magurran, A. 1988. Ecological diversity and its measurement. Princeton Univ. Press, Princeton, NJ.

McKeague, J.A., and J.H. Day. 1996. Ammonium oxalate extraction of amorphous iron and aluminum. In: M.R. Carter, editor, Soil sampling and methods of analysis. Canadian Society of Soil Science. Lewis Publ., Boca Raton, FL.

Mehlich, A. 1984. Mehlich-3 soil test extractant: A modification of Mehlich-2 extractant. Commun. Soil Sci. Plant Anal. 15:1409–1416. doi:10.1080/00103628409367568

Minitab Inc. 2007. Minitab Statistical Software, Release 15 for Windows. Minitab, State College, PA.

Pérez de Mora, A., J.J. Ortega-Calvo, F. Cabrera, and E. Madejón. 2005. Changes in enzyme activities and microbial biomass after "in situ" remediation of a heavy metal-contaminated soil. Appl. Soil Ecol. 28:125–137. doi:10.1016/j.apsoil.2004.07.006

Schnürer, J., and T. Rosswall. 1982. Fluorescein diacetate hydrolysis as a measure of total microbial activity in soil and litter. Appl. Environ. Microbiol. 43:1256–1261.

Siebielec, G., and R.L. Chaney. 2012. Testing amendments for remediation of military range contaminated soil. J. Environ. Manage. 108:8–13. doi:10.1016/j.jenvman.2012.04.028

Speir, T.W., A.P. van Schaik, A.R. Lloyd Jones, and H.A. Kettles. 2003. Temporal response of soil biochemical properties in a pastoral soil after cultivation following high application rates of undigested sewage sludge. Biol. Fertil. Soils 38:377–385. doi:10.1007/s00374-003-0656-8

Staples, C., E. Mihaich, J. Carbone, K. Woodburn, and G. Klecka. 2004. A weight of evidence analysis of the chronic ecotoxicity of nonylphenol ethoxylates, nonylphenol ether carboxylates, and nonylphenol. Hum. Ecol. Risk Assess. 10:999–1017. doi:10.1080/10807030490887122

Stuczynski, T., G. Siebielec, W.L. Daniels, G. McCarty, and R.L. Chaney. 2007. Biological aspects of metal waste reclamation with biosolids. J. Environ. Qual. 36:1154–1162. doi:10.2134/jeq2006.0366

Tabatabai, M.A. 1994. Soil enzymes. In: R.W. Weaver, J.S. Angle, P.J. Bottomley, D. Bezdicek, S. Smith, A. Tabatabai, and A. Wollum, editors, Methods of soil analysis. Part 2. Microbiological and biochemical properties. SSSA Book Ser. 5. SSSA, Madison, WI. p. 775–833.

Tian, G., T.C. Granato, R.I. Pietz, C.R. Carlson, and Z. Abedin. 2006. Effetc of long-term application of biosolids for land reclamation on surface water chemistry. J. Environ. Qual. 35:101–113. doi:10.2134/jeq2004.0354

Tvergyak, J.L. 2012. Organic by-product materials as soil amendments. M.S. thesis, School of Environment and Natural Resources, The Ohio State University, Columbus, OH.

Waiser, M.J., D. Humphries, Z.V. Tumber, and J. Holmy. 2011. Effluent-dominated streams. Part 2: Presence and possible effects of pharmaceuticals and personal care products in Wascana Creek, Saskatchewan, Canada. Environ. Toxicol. Chem. 30:508–519. doi:10.1002/etc.398

Walker, L., and R. Del Moral. 2003. Primary succession and ecosystem rehabilitation. Cambridge Univ. Press, New York.

Webb, N.R. 1997. The development of criteria for ecological restoration. In: K. Urbanska, N. Webb, and P.H. Enkell, editors, Restoration ecology and sustainable development. Cambridge Univ. Press, Cambridge, UK. pp. 133–160.