CHAPTER

Bioremediation and soils

9

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9.1 Introduction

Land degradation resulting from anthropogenic activities worldwide has multiple and complex impacts on the global environment and public health through direct and indirect processes, which affects a wide array of ecosystem functions and services (Rodríguez-Eugenio et al., 2018). The pollution of soils and water caused by anthropogenic activities is often associated with modern urbanization, industrialization, and agricultural activities such as industrial mining of metals, extraction of petroleum oils and gas, landfill waste, and applications of pesticides and herbicides for food production.

Soil pollution is one of the major effects of human technological advancement. A variety of pollutants affect topsoil and subsoil, including fuel and oil products, heavy metals, hydrocarbon waste, excessive nutrients (e.g., nitrate and phosphate), pesticides, and herbicides. Thousands of chemical pollutants, which are commercially produced on a large scale, are released into terrestrial and aquatic environments on a daily basis, resulting in about 33% of all global soils being at risk of degradation (Rodríguez-Eugenio et al., 2018). For example, agrichemicals, which can help meet the world's growing demand for food, lead to soil pollution and degraded agroecosystems.

According to the Food and Agriculture Organization (FAO) of the United Nations, more than 22 million ha of soil have been affected by soil pollution (Rodríguez-Eugenio et al., 2018). In particular, more than 16% of all Chinese soils and 19% of Chinese agricultural soils are categorized as polluted [China Council for International Cooperation on Environment and Development (CCICED) 2015].

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In Europe, nearly 60% of the top agricultural soils in 11 countries are contaminated with multiple persistent pesticides, and approximately 3 million potentially polluted sites contaminated with industrial pollutants have been identified in the European Economic Area and cooperating countries in the West Balkans (European Environment Agency, 2014). In the United States of America, over 1300 sites are included on the Superfund National Priorities List, with contamination from either heavy metals or hydrocarbon pollutants (US Environmental Protection Agency, 2013). On a larger magnitude the total number of contaminated sites is estimated at 80,000 across Australia (Australia Department of Environment and Conservation, 2010). According to the FAO, approximately 50 million tons of e-waste (i.e., discarded electrical or electronic devices) is generated every year, making it one of the world's fastest growing sources of pollutants that contaminate soil and water (Rodríguez-Eugenio et al., 2018). In addition, land becomes polluted by contaminants not only from industrial waste but also from municipal waste as well. For example, in 2014, Americans produced about 258 million tons of solid waste (US Environmental Protection Agency, 2013). These waste materials release a multitude of hazardous substances (e.g., flame retardants, dioxin-like compounds, polycyclic aromatic hydrocarbons, and heavy metals) that jeopardize environmental quality and human health (Perkins et al., 2014). A little over half of the waste (i.e., 136 million tons) was gathered in landfills, resulting in the soils and leachate at these sites and surrounding areas often being saturated with chemicals and hazardous substances. In addition, according to the National Oceanic and Atmospheric Administration, 80% of pollution in marine environment comes from land through sources such as soil sediments in runoff (Rodríguez-Eugenio et al., 2018).

Furthermore, the aforementioned ecological degradation caused by anthropogenic activities worldwide has resulted in the need to mitigate damage to essential ecosystem services in both rural and urban areas. Phytoremediation is a promising and environmental friendly approach for reclamation of contaminated sites that removes contaminants from systems through enhanced degradation, transformation, extraction, and immobilization (Mirck et al., 2005). There are several advantages of phytoremediation over traditional chemical and physical remediation approaches. In particular, phytoremediation is (1) cost-effective and affordable, (2) easy to implement and maintain, (3) solar-driven, (4) esthetically appealing and socially accepted, (5) minimally invasive, and (6) sustainable in closed-loop systems (Tsao, 2003).

This technical approach has been successfully used to treat soils and surface runoff or leachate that are contaminated with inorganic and organic pollutants (Epps, 2006; Jones et al., 2006; Lin, 2002; Lin et al., 2008, 2011a; Placek et al., 2016; Russell, 2005; Tsao, 2003). Phytoremediation and associated phytotechnologies provide essential ecosystem services during times of accelerated ecological degradation (Epps, 2006). This remediation approach helps restore ecosystem functions, preserve landscapes, and repair degraded lands, while protecting the quality of human life by reducing the risk of exposure to pollutants. Phytoremediation also offers additional ecosystem benefits, such as improved nutrient cycling, carbon sequestration, water flow regulation, and erosion control [Epps, 2006]. As a result, the restoration of contaminated sites offers the highest possible net ecosystem

benefit in terms of ecosystem services that are socially, environmentally, and economically beneficial to society.

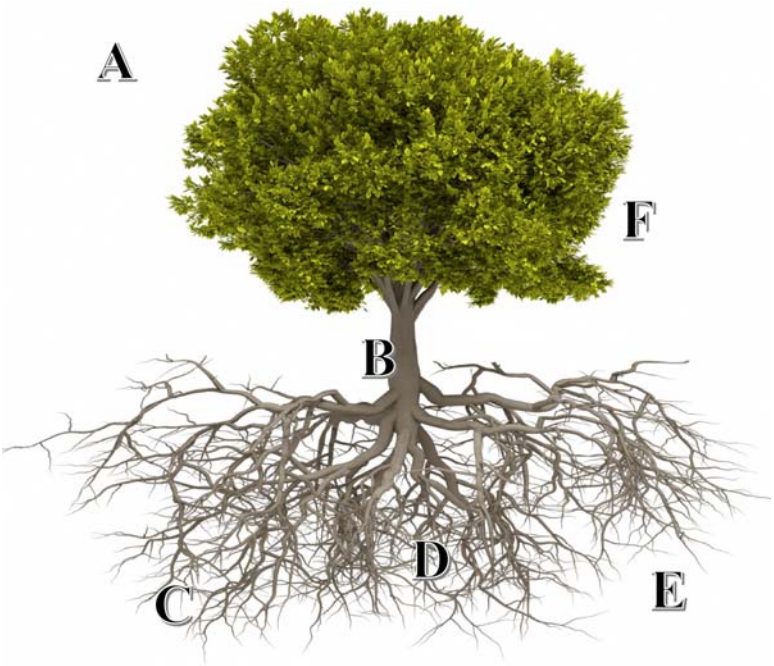
Phytoremediation is a plant-based technology for restoring contaminated land and water resources (Zalesny et al., 2016b). The success of remediation relies on several fundamental physical, chemical, and biological mechanisms (Fig. 9.1). Transport of the pollutants can be significantly reduced through enhanced infiltration/evapotranspiration, therefore reducing the volume of soil infiltration of the pollutants (Jones et al., 2006; Lin et al., 2011b; Placek et al., 2016; Zupančič-Justin et al., 2010). Organic pollutants can be immobilized and stabilized via enhanced physical adsorption, filtration, or enzymatic conjugation (Chu et al., 2010). A wide range of rhizobacteria have been known to quickly metabolize or transform the contaminants (e.g., explosives, metals, nutrients, herbicides, and pesticides) through biochemical mechanisms, including enzymatic detoxification, nitrification, and denitrification (Lin et al., 2004, 2005, 2008, 2009, 2011a). Direct plant uptake may also help to eliminate herbicides, heavy metals, and nutrients from subsurface flow (Burken and Schnoor, 1997; Lin et al., 2008). Plants have been known to extract and absorb metal contaminants such as Pb, Cd, Cr, Ar, and various radionuclides from soils. One mechanism of phytoremediation, phytoextraction, has been successfully used to remove inorganics from soil through the uptake of heavy metals that are essential for plant growth (e.g., Fe, Mn, Zn, Cu, Mg, Mo, and Ni) (Borghi et al., 2008; Pulford and Watson, 2003). Furthermore, the improvement of soil characteristics by vegetation (e.g., increases in organic matter content) helps enhance the rhizosphere's capacity for adsorption and chemical hydrolysis of pollutants (Chu et al., 2010; Mandelbaum et al., 1993). Recently, many biodefense secondary metabolites released by root exudates, such as benzoxazinones, have been identified as bioactive agents that can rapidly degrade organic pollutants (Willett et al., 2013, 2014, 2016).

The remainder of the chapter addresses the remediation of contaminated soils using plants, focusing on selection of appropriate plant materials and soil factors important for designing remediation systems. The last section of the chapter contains five real-world examples of such systems, including: (1) grasslands used for phytoremediation of soil phosphorus, (2) urban afforestation used to create forests in cities, (3) riparian buffer systems used to reduce agrichemical transport from agroecosystems, (4) short rotation woody crops (SRWCs) used to enhance ecosystem services at landfills, and (5) woody species used for surface mine reclamation.

9.2 Selection of appropriate plant materials

9.2.1 Functional groups

Grasses are probably the most common functional group of herbaceous plants used for phytoremediation, partly because they are highly diverse with a wide range of stress tolerances, they are often capable of forming sod or dense cover that may have multiple



A. Phytovolatilization
The uptake and transpiration of a contaminant by a plant. The contaminant, or a modified form of it, is released into the atmosphere.

B. Phytoextraction
The uptake and translocation of contaminants by plant roots from the soil into plant parts.

C. Phytostabilization
Plants immobilize contaminants in the soil and groundwater through adsorption onto roots or precipitation within the root zone of the plant.

D. Rhizofiltration
The use of plant roots to remove pollutants from water or soil solution. Rhizofiltration is similar to phytoextration, but the plant species are used primarily to remediate contaminated groundwater rather than soil.

E. Rhizodegradation
The breakdown of contaminants in the soil through microbial activity that is enhanced by the presence of the rhizosphere (i.e., root zone).

F. Phytodegradation
The breakdown of contaminants taken up by plants through metabolic processes within the plant or the breakdown of contaminants external to the plant through the effect of compounds produced by the plant, such as enzymes.

FIGURE 9.1

Six processes of phytoremediation that involve contaminant degradation, sequestration, or volatilization in the root zone and in tree roots, wood, and leaves.

Definitions from Mirck, J., Isebrands, J.G., Verwijst, T., Ledin, S., 2005. Development of short-rotation willow coppice systems for environmental purposes in Sweden. Biomass Bioenergy 28, 219–228.

uses, and there is an extensive worldwide seed industry to support commercial distribution of grasses for many purposes. Numerous other monocots include rushes, sedges, and flowering ornamentals for less extensive projects. Dicots may be represented by numerous functional groups of plants that include, but are not limited to agricultural crops, ground covers, or ornamentals that originate from a wide range of habitats.

SRWCs such as poplars (*Populus* spp.), willows (*Salix* spp.), and eucalypts (*Eucalyptus* spp.) are among the most productive temperate forest trees (Zalesny et al., 2011) and, therefore, are the most commonly used trees for phytoremediation (Zalesny et al., 2016b). Selected species and their intra- and interspecific hybrids are phreatophytes, exhibiting extensive root systems, and high biomass production potential relative to other temperate trees, as well as the capability to utilize high volumes of water on moisture-rich sites or exhibit high-water use efficiency on water-limited areas (depending on the genotype selected) (Zalesny et al., 2019a). Given the broad genetic diversity of these genera, there is a high probability of selection gains within and among genomic groups for tolerance and/or uptake of both inorganic and organic pollutants during phytoremediation.

Despite the focus on SRWCs for phytoremediation, slower growing, later successional tree species have gained visibility for phytotechnologies in recent years. In particular, urban greening has become an important policy focus for many cities around the world and, in addition to planting more street trees, cities are also focusing on creating forests in cities (Oldfield et al., 2013). Available land for creating urban forests can include brownfields (Gallagher et al., 2008) and vacant lots (Anderson and Minor, 2017), some of which could benefit from phytoremediation. However, remediation of a site is not the only goal for most urban afforestation projects. This means that even if fast growing "workhorse" trees like poplars and willows are used initially, there is the expectation that longer lived, more esthetically desirable species will be established in the end. However, this does not mean later successional species lack value in situations where soils need remediation. These species can play a role in long-term phytostabilization of a contaminated site (Pulford and Watson, 2003) and can take up metal contaminants. For instance, oak (*Quercus* spp.), maple (*Acer* spp.), and birch (*Betula* spp.) species accumulate Cd, Zn, Cr, and Pb to varying degrees (Evangelou et al., 2015), albeit at much lower concentrations than the SRWCs. A criticism of using later successional species is that the wood is a valuable commodity that could release accumulated metals during processing or expose consumers when they use wood products. This is not a concern for trees growing in urban forests because the primary goal of growing trees in cities is not lumber but rather the ecosystem services they provide (Nowak and Dwyer, 2000). Immobilizing and sequestering contaminants can be added to the long list of ecosystem services provided by trees to cities.

9.2.2 Selection criteria and testing

Multiple criteria are used to select plant materials for phytoremediation projects. Usually, the first decision to be made is with regard to the environment and what

functional groups of plants are reasonable choices for that environment. For example, terrestrial versus aquatic environments would lead to completely different functional groups. Within those two broad environmental categories there are many possible subdivisions that will lead to different functional and/or phenotypic differences, for example, savanna, grassland, agricultural, forest, urban (e.g., brownfields and landfills), or ornamental (terrestrial sites) and riverine, estuarial, seashore, or marine (aquatic sites). The question of native versus introduced plants may be the second decision criterion, depending on the needs or desires of the land managers or customer groups. In some cases, natives may not have the desired or necessary traits, so introduced species may be required (Paquin et al., 2004). These decision processes will significantly narrow the range of species under consideration, down to a point where a combination of literature reviews and physical testing and evaluation may be the only additional means of making informed choices.

Testing and evaluation can be conducted at one or more of three levels: bench, pilot, and field−scales. There is no hard-and-fast rule that all three scales be employed for any particular project, but rather the nature of testing will depend on both the results of the literature review and the level and duration of funding available for testing. Bench-scale studies can be conducted at several scales and under a range of conditions that involve the use of pot containers in glasshouses or growth chambers, or small field plots in common-garden experiments that are randomized and replicated to allow statistical comparisons among several species and/or genotypes within species (Zalesny and Bauer, 2007). The fundamental requirement for these studies is to create the appropriate environmental screen that mimics the impacted area or remediation application. For the former, this could be levels of soil contamination with heavy metals, petroleum products, radioactive isotopes, explosives, effluents, or other specific pollutants. For the latter, applications may entail mimicking field phytoremediation applications (e.g., landfill leachate irrigation and wastewater recycling). If the measurements to be made in bench-scale studies are quantitative in nature, statistical rules and principles generally applicable to agronomic field studies should be followed (Casler, 2015). Conversely, if assessments and decisions can be made visually based on vigor and/or survivorship, this may reduce the requirements to follow all of the normal statistical rules of replication and randomization. Bench-scale studies may also be conducted in multiple stages, with visual prescreening of a large number of species, followed by quantitative assessment of a smaller number of candidate species. Bench-scale studies are designed to eliminate unadapted species and focus on a small number of species for pilot-scale (e.g., nursery or small field sizes) or field scale (e.g., large field or production sizes) studies, depending on amount and duration of funding, as well as the level of confidence in the results from the bench-scale studies.

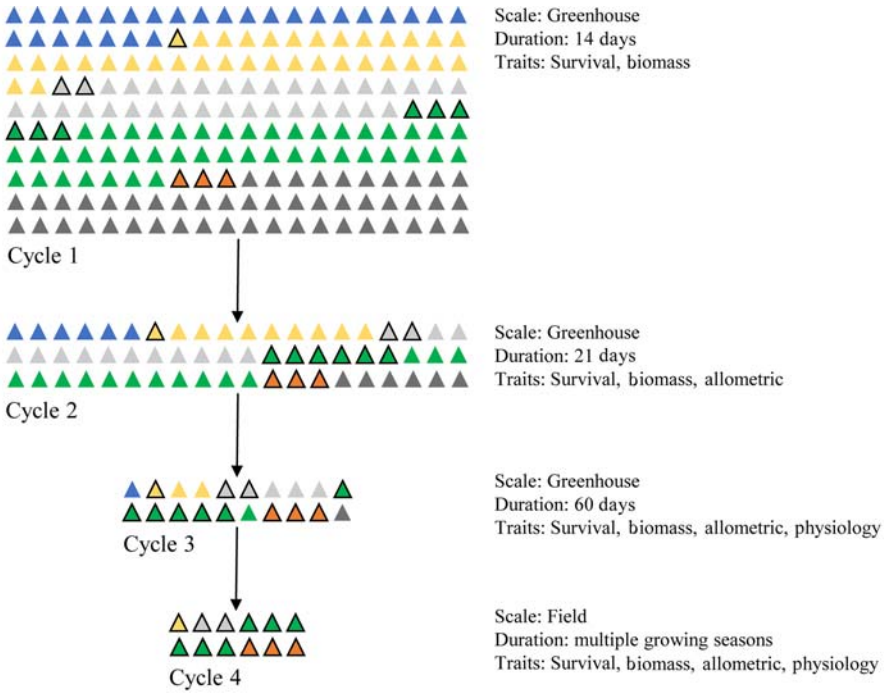
Phyto-recurrent selection is an example of a methodology used for such testing and evaluation (Zalesny et al., 2007b). Phyto-recurrent selection builds upon decades of plant breeding experience in agronomy, horticulture, and other plant

sciences to match SRWC genotypes and their tissues of uptake (i.e., root, wood, and leaf) to specific contaminants from soil- and water-based phytoremediation systems. In particular, these genotypes are tested in 2–5 selection cycles in order to make informed decisions on what varieties should be out planted and tested at the field scale. Early selection cycles are short in duration, conducted in controlled conditions, and include hundreds of genotypes that vary greatly in their genetic backgrounds. As phyto-recurrent selection progresses, cycles get longer, trees are grown in nurseries and/or field conditions, and the number of genotypes decreases. In addition, the complexity of data increases with each subsequent selection cycle, with early cycles focusing on survival and biomass traits and later cycles incorporating additional allometric and physiological parameters. The ultimate goal of phyto-recurrent selection is to choose a combination of genotypes with high phytoremediation potential and adequate genetic variation (i.e., selecting a suite of clones rather than just the best clone). In doing so, two different categories of genotypes are identified: (1) generalist clones that perform well across varying site conditions and pollutants and (2) specialist clones that grow well at specific sites and with particular contaminants (Orlović et al., 1998; Zalesny et al., 2016b). Fig. 9.2 illustrates an example of phyto-recurrent selection used for choosing poplar genotypes for landfill phytoremediation.

With similar objectives and needs as for SRWCs and other trees, the selection of grasses, forbs, and other herbaceous species for bioremediation projects often is evaluated by (1) the remediation mechanisms, (2) the plants' tolerance to the pollutants, and (3) other biotic and abiotic stress gradients in the soil microenvironments. Phytoremediation of herbicides with grasses, forbs, and other herbaceous species provides a meaningful example of such selection and testing.

Specifically, the remediation mechanisms are determined by plant-rhizosphere interactions, plant detoxification mechanisms, physiological and morphological characteristics, and chemistry of the herbicides (i.e., their phytotoxicity, solubility, and hydrophobicity) (Lin et al., 2003, 2008, 2011b). More specifically, for mobile- and degradation-resistant organic pollutants, such as herbicides like isoxaflutole and glyphosate, surface- and ground-water mitigation has been achieved through enhanced infiltration/evapotranspiration, therefore reducing the volume of percolating and surface water transport (Lerch et al., 2017; Lin et al., 2008). On the other hand, for organic herbicides or pollutants that are more sensitive to degradation, mitigation has occurred via enhanced biological, chemical, and enzymatic transformation of organic pollutants into less-toxic and less-mobile metabolites in the rhizosphere (Lin et al., 2008, 2009, 2010, 2011a). Many biodefense secondary metabolites, such as benzoxazinones that are released by several native warm season grasses, also play an important role in enhancing the degradation of organic pollutants in the bioremediation systems (Willett et al., 2013, 2014, 2016).

Identification of biotic and abiotic stress gradients in the microenvironment of plants is crucial in the design of tree–shrub–grass multispecies systems for pollutant remediation. This includes knowledge of temporal and spatial characteristics of each stress as well as mechanisms by which specific stresses are reduced



Number of clones tested in four phyto-recurrent selection cycles

Genomic group / cycle	1	2	3	4
P. deltoides Bartr. ex Marsh	27	6	1	0
P. deltoides × *P. deltoides*	35	10	3	1
P. deltoides × *P. maximowiczii* A. Henry	35	15	5	2
P. deltoides × *P. nigra* L.	50	20	7	6
P. nigra × *P. maximowiczii*	3	3	3	3
(*P. trichocarpa* Torr. & Gray × *P. deltoides*) × *P. deltoides*	50	6	1	0
Total	200	60	20	12

FIGURE 9.2

Example of phyto-recurrent selection used to choose poplar (*Populus* spp.) genotypes for phytoremediation. Four selection cycles are illustrated showing hypothetical genotypes (represented by *triangles*) belonging to six genomic groups. Clones advancing to subsequent cycles are indicated with bold outlines. Note that testing moves from the greenhouse to the field, duration of testing increases, and data become more complex with later cycles.

(Lin et al., 2004). With regard to bioremediation of organic herbicides by tree–shrub–grass riparian systems, understanding the tolerance of the selected understory grass species to both stresses of shade and high concentrations of pollutants, as well as their detoxification mechanisms, is critical for success.

For example, the C_4 warm season switchgrass (*Panicum virgatum* L.) and eastern gammagrass (*Tripsacum dactyloides* L.) provide an ideal first line of defense along fields of corn (*Zea mays* L.) where atrazine (ATR) concentrations are expected to be highest, thereby providing a sound tree–shrub–grass vegetative buffer system for mitigation of this commonly used herbicide. These grasses not only tolerate high levels of ATR but also have strong capacity to quickly neutralize the ATR through rapid chemical and biological degradation processes (Lin et al., 2003, 2008, 2011a). Switchgrasses and eastern gamagrasses not only help to prevent channelized flow that is generated but they encourage a more uniform sheet flow due to their stem morphology (Lee et al., 1997) and help to decrease the surface transport through encouraged flow infiltration (Lin et al., 2011b). Shade tolerant C_3 species, such as smooth bromegrass (*Bromus inermis* Leyss.) or tall fescue (*Festuca arundinacea* Schreb.), can be ideal choices to be incorporated below the C_4 warm season species near the stream bank where ATR is diluted and tree shading is a concern. These cool-season C_3 species have shown higher annual evapotranspiration rates relative to C_4 grasses (Lin et al., 2003, 2004, 2008, 2011b). As a result, these C_3 species will rapidly remove soil moisture and facilitate the physical trapping of herbicides in the soil. Also, these C_3 species are expected to be tolerant of the moderate shade projected by tree crowns closer to the stream bank (Lin et al., 1998, 2004).

Finally, microbial symbionts may also play a significant role in the success of phytoremediation efforts and in the choice of specific species or genotypes. Arbuscular mycorrhizal (AM) fungi can enhance the adaptation and survivorship of some host plants by enhancing nutrient uptake, absorbing heavy metals, and protecting the host from metal toxicity. Numerous examples of herbaceous monocots and dicots with AM fungal associations acting as hyperaccumulators of heavy metals were provided in the review by Leung et al. (2013); and there are examples from SRWCs, as well (Gunderson et al., 2007; Jordahl et al., 1997). Endophytic fungi (EF) are widely distributed within the grass family and present in many other monocots and dicots. However, the ability of EF to enhance hyperaccumulation characteristics of host plants has received relatively little study to date (Deng and Cao, 2017). There is evidence that EF within the fescues (*Festuca* L. spp.) can lead to increased hyperaccumulation of petroleum pollutants (Soleimani et al., 2010).

9.2.3 Traditional breeding and selection approaches

Traditional breeding and selection approaches are generally conducted under field conditions that are intended to mimic real-world production environments. Generally, this means using appropriate environments, managements, and

selection screens. Environments are generally defined by eco-geographic factors such as temperature, precipitation, soil type, and photoperiod. However, when breeding new cultivars for the purpose of phytoremediation, the anthropogenic soil or aquatic factors that have created this demand must be considered when defining both the environment and the selection screen (Zalesny and Bauer, 2007). If the initial germplasm to be used possesses genetic variability for resistance, tolerance, and/or uptake of the targeted anthropogenic element, breeding schemes can be simplified to focus largely on screening plants and progeny for tolerance, uptake, vigor, survivorship, and any other relevant measures necessary to generate superior genotypes (e.g., stable hyperaccumulators, varieties with favorable water use efficiency) (Ernst, 2006).

The rate at which plant breeders can develop new cultivars to solve potential problems is dependent on four general factors: (1) the heritability of the trait, (2) the selection pressure applied, (3) the breeding procedure, and (4) the time required to complete a generation of selection (Falconer and Mackay, 1996). Heritability can generally be increased by the use of replicated families or clonally replicated individuals, obviously with some added expense. Heritability can also be increased by any mechanism that allows the breeder to generate more accurate or precise data used to make selection decisions. Selection pressure is increased by the use of larger population sizes, which tends to lead to space, time, and funding limitations. Different breeding procedures utilize additive and dominance genetic variation to different degrees, which can influence the rate of genetic progress. Lastly, generation time is a major factor determining rate of gain, with woody species, especially forest trees, requiring the longest time, and annual plants or algae species the shortest times. Plant breeders are generally well versed in the myriad of trade-offs that are required to design the most efficient and effective breeding and selection schemes, so project managers who have resources to use these approaches to develop new genotypes or cultivars should consult with a plant breeder who can provide advice and counsel in making many of these decisions. While perennial plants require longer generation times than annual plants, they might have the advantage of clonal propagation for commercial cultivars, allowing the single "best" individual of each generation to be chosen for commercialization and dissemination. Another advantage of clonally propagated species is that they allow the breeder to utilize all forms and amounts of genetic variability within the population; they are very efficient in this regard.

Creation of an effective, efficient, and relevant screen is perhaps the most critical aspect of a breeding and selection component of a phytoremediation project. If the screen is too severe, it might kill all the subjects, while a too-mild screen would not provide sufficient discrimination to be effective in identifying the best genotypes (Ernst, 2006). If improved vigor or survivorship is the goal, screening can be simplified to development of a soil or aqueous medium that provides a concentration of the toxin or pollutant that results in some loss of vigor or some mortality, sufficient to allow discrimination of a small number of individuals with a high level of confidence, for example, often targeting a selection intensity of

0.01%−1% of the population. Conversely, if the goal is hyperaccumulation of a specific element, selection is more complicated, requiring the breeder to measure biomass accumulation, and collect, process, and measure tissue samples using some high-throughput mechanism. Tolerance and hyperaccumulation are not the same trait, hence requiring different measurements and selection approaches (Ernst, 2006). This would add time and expense to the selection process but may be critical or necessary for some project goals.

For woody species, results from tree development programs are often slow or limited given multiyear timeframes between breeding activities and sexual maturity of the trees. Although vegetative propagation can accelerate selection processes of favorable genotypes, information about their full performance (e.g., biomass productivity, disease resistance, and phytoremediation potential) is yet to be obtained until after the end of each production cycle, which can last greater than 10 years for SRWCs. Nevertheless, throughout plantation development, tree performance is typically assessed via growth parameters, which are a reflection of numerous allometric, anatomical, physiological, and biochemical traits (Orlović et al., 1998; Zalesny and Bauer, 2007). Similarly, the effect of different factors (e.g., water and nutrient availability and presence of xenobiotics) on plant growth can be obtained through yield assessment, which is a composite trait that can be tested directly or indirectly via individual traits that affect plant performance (Marron and Ceulemans, 2006). The most common allometric traits include biomass, diameter, height, number of leaves, leaf area, root area, number of roots, and root length (Zalesny and Bauer, 2007), while those related to internal structure and function are parameters related to nitrogen assimilation (Matraszek, 2008; Pilipović et al., 2012a), photosynthesis, transpiration, and water use efficiency (Borghi et al., 2008; Pajević et al., 2009, 2012a, 2019), and biochemical processes (e.g., proline, glutathione, and antioxidant activity) (Di Baccio et al., 2005; Kebert et al., 2017; Nikolić et al., 2008). In addition, morphological changes resulting from variability in these physiological parameters are also useful traits when selecting SRWCs for phytoremediation (Di Baccio et al., 2003; Nikolić et al., 2008; Rogers et al., 2019; Zalesny et al., 2009a). While many of these parameters also are relevant for nonwoody genera, none are more important and cross-cutting than contaminant concentrations in roots, stems, and leaves, which is the greatest measure of remediation success.

The starting plant materials used to develop cultivars for phytoremediation purposes are critical. There must be sufficient genetic variation for the key traits to allow the breeder to reliably choose the "best" individuals and, in so doing, to accumulate the necessary genes in the selected genotypes or their progeny to ensure adequate performance of the new cultivar. There are numerous examples in which heavy metal tolerance and hyperaccumulation abilities have evolved naturally in perennial grasses that are subjected to mine spoils, tailings, Zn-coated electricity pylons, or contaminated soils (Macnair, 1987). Even though the frequency of "tolerant" plants might start out as low as <0.01%, this frequency can increase through long-term on-site exposure or by using artificial selection

approaches with large population sizes in the laboratory, for example, screening seedlings on contaminated soil or using a hydroponic system. The low frequency of heavy metal tolerances in these populations, as an example, drives home the critical point that plant breeding is a "numbers game"—the larger the population size that is screened, the greater the likelihood of finding desirable genotypes.

Finally, genotypes or cultivars that are candidates for phytoremediation should have their field performance validated or verified before too many resources are put toward multiplication of seed or vegetative cuttings and before large-scale remediation plantings are initiated. For example, selecting poplar clones that are resistant to diseases such as leaf rust (e.g., *Melampsora* spp.) is important. This phase would involve pilot-scale or field-scale evaluations as with phyto-recurrent selection mentioned earlier in this chapter. Ideally, this would include multiple plantings or environments if there are multiple target sites for phytoremediation. In these cases, pilot-scale trials could be conducted on a small area of each site, for the purposes of confirming that the candidate cultivar has the required levels of tolerance and/or hyperaccumulation ability. If researchers have a high degree of confidence in the future performance of a candidate cultivar, seed or clonal stock multiplication can proceed at the same time as the pilot-scale trials to save time.

9.3 Soil factors important for designing remediation systems

Soil health or quality is a topic of much discussion and research (Bünemann et al., 2018), and the importance of healthy, good quality soils to agricultural and natural systems is not disputed. In fact, definitions of soil quality stress the importance of a soil's ability to buffer "potential pollutants such as agricultural chemicals, organic wastes, and industrial chemicals" (National Research Council, 1993). "Storing, filtering, and transformation of compounds" is listed as one of the seven soil functions by a study group convened by the Royal Academy of Sciences of the Netherlands (Bouma, 2010). Bouma (2010) goes on to discuss a knowledge gap in soil science and highlights the fact that technical "end-of-pipe" solutions to environmental pollution are missing out on the potential for soils to act upstream as a living filter.

Soil degradation from urbanization and industrialization often results in the loss or destruction of plant life from a given site and frequently involves chemical dumping. In addition, severe and long-term agricultural practices often result in nonpoint source pollution of surface waters due to excessive nutrient loads (Sharpley et al., 1994; Sims et al., 1998). Restoration or remediation is needed when human impacts cause the soil system to become compromised or overloaded thereby limiting a soil's ability to filter or transform naturally occurring chemicals (e.g., N and P) or organic wastes and industrial chemicals, all of which can have negative impacts on ecosystem and human health (Galloway et al., 2017; Nieder et al., 2018; Sarwar et al., 2017).

There is plenty of literature on optimal soil physicochemical properties to maximize agricultural yield, promote optimal tree health and growth, or for maintaining healthy ecosystems of many types (Verheye, 2010a–c). In a critical review of soil quality indicators, Bünemann et al. (2018) found that total organic matter and pH are the most frequently used soil quality indicators. In the case of degraded sites the emphasis shifts from optimal conditions to finding genera and/or species that can survive on the site and subsequently help remediate the site so the soil can become functional once again. The key factors to consider when designing phytoremediation systems for degraded sites are highlighted in Table 9.1.

Table 9.1 Soil factors to consider when designing phytoremediation systems for degraded sites.

Scale[a,b]	
Small (<0.25 ha)	Abandoned homes/lots in urban areas, concentrated dumping of chemicals or spills, small agricultural operations.
Medium (>0.25, <10 ha)	Urban construction projects, landfills, brownfields, agricultural operations, chemical disposal.
Large (>10 ha)	Landfills, agricultural operations, mining operations.
Level of physical disturbance[b,c]	
No disturbance	Chemical disposal or leakage. Soil column left intact.
Agricultural	Some agricultural disturbance. Primarily dealing with nonpoint source pollution issues.
Construction	Common in urban settings where construction debris (e.g., concrete, rebar, and asphalt) is deposited and covered with clean fill.
Historical land-use	Abandoned homes and railroad beds.
Landfills	Engineered (e.g., lined and capped) or not engineered will be handled differently.
Mining	Strip mines and tailings will be handled differently.
Chemical contamination[b,c,d,e]	
No contamination	Common in urban setting where clean fill has been used to cover construction debris creating anthrosols. Typically characterized by low levels of organic matter, available nutrients and microbial biomass.
Heavy metals	Factors that impact bioavailability should be considered. Heavy metal bioavailability is influenced by pH, competitive ion concentrations, soil organic matter, cation exchange capacity, and texture.
Mercury	Levels of soil organic matter are important.
Organic contaminants	Includes aromatic hydrocarbons, PAH, PCB, pesticides and polychlorinated dibenzo-*p*-dioxins, among others.
Excessive nutrients	Phosphorus and/or nitrates in soil are often susceptible to erosion or leaching, eventually entering surface waters and causing algae blooms or eutrophication.

(*Continued*)

Table 9.1 Soil factors to consider when designing phytoremediation systems for degraded sites. *Continued*

Hydrologic flow paths[c,f]	
Contained	No movement of contaminant.
Overland flow	Agricultural systems.
Effluent or leachate	Landfills, agricultural operations, mining operations.
Connected to groundwater	Landfills, agricultural operations, mining operations.

PAH, *Polycylic aromatic hydrocarbons;* PCB, *polychlorinated biphenyls.*
[a]*Anderson and Minor (2017).*
[b]*Oldfield et al. (2014).*
[c]*Gallagher et al. (2008).*
[d]*Gramatica et al. (2002).*
[e]*Lin et al. (2011a).*
[f]*Zalesny et al. (2008).*

9.4 Applications and experiences

9.4.1 Grasslands as a mechanism for phytoremediation of excessive soil phosphorus to reverse eutrophication and improve water quality

Decades of phosphorus fertilization and manure applications from livestock agriculture in excess of crop requirements have resulted in millions of hectares of soils with excessive phosphorus (P) concentrations (Sims et al., 1998). High-P conditions have resulted in increased P loss to surface waters, leading to rapid eutrophication and degradation of water quality, significantly impacting natural resources important for conservation, recreation, drinking water, and fresh and marine water food sources (Steinman et al., 2017). Eutrophication of surface waters leads to inversion of the food pyramid in fresh and marine waters, decreasing the abundance of consumers and predators that are important sources of food and recreation (Binzer et al., 2016).

Phosphorus inputs from point and nonpoint sources can accumulate at many points along the transport pathways from farm fields to surface waters, resulting in numerous sources of legacy P (Sharpley et al., 2013). Accumulated legacy P can be remobilized back into the transport pathway by severe storms or other disturbances or by relaxation of soil conservation practices that may have been put in place decades ago. As a result, many large-scale conservation programs that have been in place for several decades have not delivered the promised increases in water quality within the expected timescales (Sharpley et al., 2013, 2015; Vadas et al., 2018). The only practical way to reduce legacy P in agricultural soils is crop uptake and export. Because annual P removal in crops is generally less than 100 kg P ha^{-1}, it may require several decades to reduce legacy P to reasonable levels on high-P soils that may have up to 4,000 kg P ha^{-1} in the upper 30 cm of soil.

Grasslands are a highly effective mechanism to stabilize soil and reduce soil erosion (Jackson, 2017; Panagos et al., 2015a,b). Grasslands are frequently composed of highly diverse communities of many species with a range of characteristics (Fig. 9.3). Both among species and within species diversity can be used to identify species and genotypes that have the required adaptive characteristics to be used for phytoremediation projects.

Vegetative buffer strips (VBS) or grass margins (GM) are being used with increasing frequency throughout Europe and much of North America to reduce the entry of eroded soil and nutrients from cropland into surface waters (Jackson, 2017; Habibiandehkordi et al., 2019; Panagos et al., 2015a). Traditional use of VBS or GM is based on the use of unharvested perennials, in some cases to the extent of grassland restoration or recreation of seminatural habitats for wildlife and recreation (Holland et al., 2016; Jackson, 2017). Despite the promise of this approach, results have been disappointing, partly due to (1) compromises that can negatively impact the continuity and contiguity of VBS, (2) uncertainty of the best management practices for VBS, and (3) uncertainty of optimal placement to maximize effectiveness and efficiency of

FIGURE 9.3

Thousands of tallgrass prairie and savanna remnants remain throughout the eastern and central United States, providing excellent sources of germplasm for direct release as cultivars or as source materials for selection nurseries to develop cultivars suitable for phytorestoration projects.

Photo by M.D. Casler (USDA Agricultural Research Service).

VBS (Jackson, 2017). While unharvested VBS may have considerable natural appeal as wildlife habitat and for human recreation purposes, this management practice results in no impact on legacy P in high-P soils (Sharpley et al., 2013, 2015; Vadas et al., 2018).

Recent efforts have focused on the use of multifunctional grasslands that are capable of mining and exporting P from high-P soils, providing a crop of economic value to farmers, and (in some cases) providing wildlife habitat and human recreational benefits associated with seminatural habitats. This multifunctionality requires partnerships between agricultural operations and soil conservation organizations, as well as additional partners that are involved in wildlife conservation for some applications. Perennial grasslands can meet these needs in one of the two ways: (1) as a source of forage or feed to support livestock agriculture or (2) as a source of biomass to support renewable energy production systems. Grasslands for use as forage or feed crops can be established not only on VBS or GM scales but also on whole-field scales, but the key element of these systems is to grow productive crops that stabilize the soil and extract P from high-P soils (Fiorellino et al., 2017; Habibiandehkordi et al., 2019; Pant et al., 2004). These feeds must be exported from the farm and their greatest benefit would be derived from sale and feeding to support livestock operations on low-P soils (Vadas et al., 2018). Similarly, a number of perennial grasses are undergoing development as perennial biomass crops for conversion to bioenergy and many of these are also suitable for VBS, GM, or whole-field biomass production systems (Jackson, 2017; Silveira et al., 2013). Biomass crops are generally exported from the farm and used to support conversions systems with energy as the primary product, but always with a coproduct or by-product. In the case of pyrolysis to produce bio-oils, phosphorus from high-P soils ends up in the form of biochar, a soil amendment that is being packaged and used in both farming and gardening applications. Ironically, biochar enhances soil-P uptake and extraction, providing a potential opportunity to improve soil-P management (Biedermann and Harpole, 2013; Gao et al., 2019).

9.4.2 Forest creation in the city: Testing an anthropogenic forest succession strategy

Urban areas around the world are embarking on efforts to increase green space within city limits. Tree planting is one of the areas of focus for these efforts. “Million Trees” programs have been instituted in cities like New York, Los Angeles, London, and Shanghai. Other cities such as Chicago have adopted a tree canopy cover goal. This focus, by cities, on increasing urban tree canopy and maintaining healthy urban trees has been brought about by increasing recognition of the socio economic value trees provide cities (Berman et al., 2008; Nowak et al., 2018). One of the challenges faced by urban land management agencies when trying to implement these programs is finding places to put the trees. One

approach is to "restore" city property that has been taken over by exotic invasive plant species. Another approach is to "reclaim" areas that have been severely impacted by construction activities where the soil is primarily made up of fill material of varying quality. These planting goals are particularly important in the light of recent findings showing declining urban and community tree cover in the United States (Nowak and Greenfield, 2018).

The sites that can be reclaimed in the city often bear little resemblance to the places where trees evolved to grow. In fact, the soils on many of these sites are described as human altered and human transported (HAHT) soils (Galbraith, 2018). The combination of these soils in an urban environment and the competition from exotic invasive plant species (Vidra et al., 2006) makes the creation of a forest in the city challenging. These challenges mean that in urban systems human intervention is likely to be a necessary component of a sustainable urban forest starting with establishment and continuing throughout the development of mature canopy dominant native trees (Sasaki et al., 2018; Simmons et al., 2016). Current research on urban afforestation is focused on species palettes, diversity, and soil treatments (Oldfield et al., 2013, 2015). Another element of sustainable forestry is the recruitment of desirable species into the understory. Here again, reclaimed lands in urban areas prove to be challenging. Robinson and Handel (2000) found that on an abandoned municipal landfill that natural recruitment would not support the development of a more diverse woodland and that human intervention would be necessary. Doroski et al. (2018) found that 6 years post-planting, site treatments and conditions strongly influence natural regeneration but that urban sites will need continued human intervention to be sustainable in the long term.

Tree species that are being used to increase urban canopy cover are usually native to the local geographical area (Oldfield et al., 2013). In addition, consideration is given to species that are tolerant of the harsh chemical and climatic conditions of the city where they are being planted. One of the goals of these afforestation efforts is to get the young trees established and to achieve canopy closure as quickly as possible in order to survive amongst exotic invasive plant species that can outcompete native vegetation.

Classical succession theory (Clements, 1916) has been applied to rural forest vegetation dynamics describing the changes in plant communities that occur across time (hundreds to thousands of years) after catastrophic disturbances (e.g., fire and landslides). Despite the extended time trajectory, there are elements of succession theory that can be applied to urban afforestation projects. Anthropogenic succession theory combines elements of classical succession theory with phytoremediation techniques like phyto-recurrent selection of early successional genera (*Populus* and *Salix*) to find fast growing genotypes that are more likely to succeed on HAHT soils in urban areas and perhaps be able to compete with exotic invasive plant species (Fig. 9.4) (Zalesny et al., 2014, 2016b). Once established, these early successional species can improve soil conditions and create an environment where later successional species can thrive. In SRWC

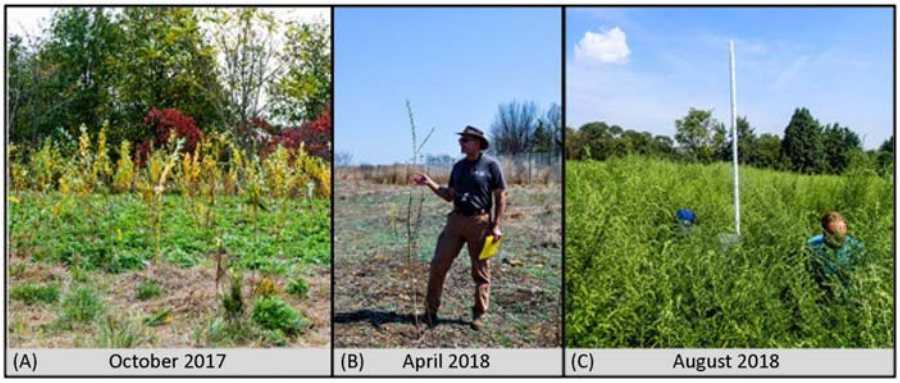


FIGURE 9.4

Freshkills Anthropogenic Succession study, Staten Island, New York, United States. (A) *Salix* spp. planted on legacy dump October 2017. (B) *Salix* spp. early flowering spring 2018 with mugwort (*Artemisia vulgaris* L.) just starting to green up. (C) Mugwort is 2 m tall by August. Can fast growing *Salix* spp. compete?

Photo panel from R.A. Hallett (USDA Forest Service).

and phytoremediation applications, rapid growth and establishment is a priority (Zalesny et al., 2016b). In urban afforestation, this rapid growth and establishment can be leveraged by thinning established early successional trees within a few years and underplanting with slower growing, more desirable, later successional planting palettes including shade tolerant tree and shrub species. This process compresses the time trajectory of natural succession and could result in reduced costs for establishing a forest in the city.

9.4.3 Riparian buffer systems to reduce agrichemical transport from agroecosystems

Well-engineered multispecies riparian buffer strips can be utilized as a cost-effective bioremediation measure to reduce agrichemical transport from agroecosystems and provide a broad range of long-term ecological and environmental benefits.

Over the past decade the University of Missouri's Center for Agroforestry has been dedicated to developing riparian buffer technologies for remediating point and nonpoint sources of pollution (Fig. 9.5). Recent work investigating bioremediation of herbicides and veterinary antibiotics in VBS systems has shown promising results in terms of contaminant load reduction and enhanced degradation (Lin et al., 2003, 2008, 2010, 2011a,b,c; Lin and Thompson, 2013; Lerch et al., 2017). For example, 8 m of riparian buffer strips consisting of native warm season grasses removed 75%–80% of ATR, metolachlor, glyphosate, sulfamethazine,

FIGURE 9.5

The implementation of a vegetative riparian buffer system to remove agrichemicals in soils, surface runoff, and subsurface flow from agricultural fields in Iowa, United States.

Photo from the Center for Agroforestry at University of Missouri, and Department of Natural Resource Ecology and Management at Iowa State University.

tylosin, and enrofloxin in surface runoff. Perennial riparian buffer strips systems tend to harbor soil microbial communities that express greater enzymatic activity that facilitate the degradation of the agrichemicals. Subsequently, ATR degradation was found to be significantly greater in soils previously planted to warm season riparian buffer species relative to bare-soil controls (90% and 24%, respectively), and the half-life of sulfamethazine was 4.25 days shorter in soils collected from the rooting zone of a hybrid poplar (*Populus deltoides* × *Populus nigra*) tree than in control samples (Lin et al., 2010). Enhanced oxytetracycline and sulfadimethoxine sorption to riparian buffer soils has been documented as well (Chu et al., 2010). These results suggested VBS can significantly alter the fate and transport of agrichemicals in agroecosystems, and findings from these studies can be used to design riparian buffer systems that more effectively provide ecosystem services and minimize acreage removed from crop production.

In addition, there are several factors that impact the efficacy of VBS for mitigating surface transport of organic contaminants, including the selection of species, soil type, buffer width, soil erodibility, source to buffer area ratio, buffer placement, runoff flow type (i.e., sheet vs concentrated flow), slope, rainfall intensity, antecedent soil moisture, and the chemical properties of the contaminants (Liu et al., 2008;

Reichenberger et al., 2007; Sabbagh et al., 2009). A broad range of trapping efficiencies resulting from the variation in these factors has been reported in the literature. For example, in a watershed study conducted in central Texas, a 44%–50% reduction in herbicide levels was observed when a filter strip was implemented, while other studies reported 17%–80% removal efficiencies of herbicides in surface runoff (Lin et al., 2011b; Lerch et al., 2017; Hoffman et al., 1995; Mersie et al., 1999; Seybold et al., 2001).

To maximize the removal efficiency, vegetation type, buffer width, and buffer placement are the factors that can be managed. The selection of the plant species and community strongly influences physical, chemical, and biological soil properties that are involved in the bioremediation processes of the pollutants. Many species in various riparian buffer designs have shown the capacity to enhance degradation of herbicides trapped in the rhizospheres because of their ability to stimulate microbial growth and enzyme activities (Lin et al., 2008, 2011a,c; Staddon et al., 2001). Buffer width has been shown to be another important factor to influence the contaminant transport and sediment trapping, with greater buffer widths required to trap fine-grained particles and moderately sorbed pesticides (Liu et al., 2008; Reichenberger et al., 2007). With regard to the placement of the buffer, to prevent the occurrence of concentrated flow through the buffer, the buffer system should be located in close proximity to the source of contamination (Reichenberger et al., 2007). In general, VBS effectiveness will increase with decreasing source to buffer area ratio (Liu et al., 2008). When prioritizing the placement of the system, factors such as the contributing source area, soil wetness, and soil erodibility should be taken into consideration during the design phase (Tomer et al., 2009).

Lastly, there are several physical, chemical, and biological mechanisms involved in the process of bioremediation within the riparian buffer zone. The organic pollutants and nutrients can be intercepted by the roots and residue of the vegetation via the enhanced physical adsorption and filtration (Chu et al., 2010). Rhizobacteria growing in the root zone may have the capacity to metabolize herbicides and nutrients through various biochemical mechanisms, including enzymatic detoxification, nitrification, and denitrification (Lin et al., 2004, 2005, 2008, 2009, 2011a). Direct plant uptake may also help to eliminate the herbicides and nutrients from the subsurface flow (Burken and Schnoor, 1997; Lin et al., 2008). Furthermore, the improvement of soil characteristics by vegetation (e.g., increases in organic matter content and improved porosity) may enhance the rhizosphere's capacity for adsorption and chemical hydrolysis of pollutants (Chu et al., 2010; Mandelbaum et al., 1993).

9.4.4 Using phytoremediation to enhance ecosystem services of landfills

Increasing human population growth and associated industrial development in the last 50 years have contributed to degradation of essential ecosystem services

throughout landscapes along the rural to urban continuum (Donohoe, 2003; McDonnell and Pickett, 1990; Wu et al., 2016). For example, human activities ranging from rural farming to industrial production in large cities have greatly impacted soil and water quality in the Great Lakes basin of the United States and Canada (Quinn et al., 2001; Stites and Kraft, 2001). Similar to other areas throughout the world, municipal and industrial landfills in Great Lakes watersheds have contributed to nonpoint source pollution of soils and water, especially given potential impacts of their runoff and leakage (Ferro et al., 2001; Minogue et al., 2012; Zalesny et al., 2016b). Finding methods to reduce these impacts, remediate these sites, and restore these ecosystems is of paramount importance given that the Great Lakes are the largest collection of fresh water in the world and that they provide a tremendous quantity and magnitude of additional ecosystem services (Steinman et al., 2017).

As described previously, selecting appropriate plant materials and identifying key soil factors is essential for designing remediation systems. This is especially true for tree-based phytotechnologies given their broad variation in site conditions, contaminant chemistries, and management objectives (Smesrud et al., 2012; Zupanc and Zupančič-Justin, 2010; Zalesny et al., 2016b). Similar to bioenergy and bioproducts applications, understanding genotype × environment interactions is crucial for making decisions about what plants to utilize in phytotechnology portfolios and how to maximize biomass productivity from those trees (Headlee et al., 2013; Zalesny et al., 2007a, 2009b). This is especially important at landfills where contaminants range in complexity from inorganics (e.g., heavy metals) to organics (e.g., polycyclic aromatic hydrocarbons), and management priorities are site specific (e.g., landfill leachate recycling, runoff reduction, and contaminant removal) (Christensen and Kjeldsen, 1989; Duggan, 2005; Kjeldsen et al., 2002). While many phytotechnologies have been used at landfills, none have been designed and implemented more than phytoremediation (Zalesny et al., 2016b, 2019b). This frequent use of phytoremediation is due to its broad applicability of having multiple processes that may simultaneously take place in the rhizosphere, roots, wood, and leaves, thus collectively increasing the potential success of the system relative to those that are limited to individual contaminants or plant tissues (Fig. 9.1) (Mirck et al., 2005).

SRWCs such as poplars (*Populus* spp.) and willows (*Salix* spp.) are ideal for phytoremediation of landfills because they grow quickly and have extensive root systems and hydraulic control potential, all of which serve as biological systems that capture and remediate soil and water pollution (Nichols et al., 2014; Nixon et al., 2001; Rockwood et al., 2004). Production gains from poplar and willow breeding programs have been successful in traditional applications given the broad genetic variability of both genera and the subsequent potential to select superior pure species and intra-/interspecies hybrids from within parental and progeny populations (Aravanopoulos et al., 1999; Mahama et al., 2011; Nelson et al., 2018; Rajora and Zsuffa, 1990). Knowledge gained from these traditional tree development activities has translated well into testing and selecting poplar

and willow genotypes for phytoremediation (Licht and Isebrands, 2005; Mirck et al., 2005; Zalesny et al., 2016a,b). As described previously, USDA Forest Service researchers have developed phyto-recurrent selection, a tool for choosing generalist tree varieties that remediate a broad range of contaminants, or specialists that are matched to specific pollutants (Zalesny et al., 2007b, 2014). The ability to select varieties across contaminants allows for broad applicability of these phytoremediation systems (Zupančič-Justin et al., 2010).

Recently, phyto-recurrent selection has been used to choose poplar and willow genotypes for phytoremediation buffer systems that are being developed to reduce untreated runoff, recycle wastewater, and groundwater and manage stormwater from landfills within the Lake Superior and Lake Michigan watersheds and, ultimately, to mitigate nonpoint source pollution impacts on nearshore health (Gardiner et al., 2018; GLRI, 2019). In particular, since June 2017, over 20,000 trees have been planted across 16 buffer systems in Wisconsin and Michigan (Fig. 9.6). Key management implications include (1) projecting and measuring the volume of untreated runoff captured or treated, (2) delineating potential

FIGURE 9.6

Poplar (*Populus* spp.) trees 14 months after planting at a landfill in southeastern Wisconsin for runoff reduction and phytoremediation.

Photo by R.S. Zalesny Jr. (USDA Forest Service).

landfill leachate leakage plumes through the use of phytoforensic technologies (Burken and Schnoor, 1998; Limmer et al., 2011), (3) developing a "green tool" to provide site managers with biological treatment options, (4) developing tree health assessment protocols that can be used in phyto-recurrent selection indices, and (5) assessing phytoremediation potential (via phytostabilization and phytovolatilization). Overall, these phytoremediation activities are reducing uncertainty about the efficacy of using trees to remediate landfills, dumps, and similar sites while improving water quality and soil health, stabilizing stream banks, increasing forest cover, and enhancing ecosystem services.

9.4.5 Surface mine reclamation

Surface mining is one of the most extreme forms of land and soil degradation, being a technology that requires physical removal of overlying soil deposits to access materials such as coal, metals, and minerals (Lima et al., 2016). An example of the significance of such environmental impact is that opencast coal mining damages 2–11 times more land than underground mining (Bai et al., 1999). Such activities result in broad scale disturbance of the landscape, integrity of the habitat, environmental flows, and ecosystem functions (Miller and Zégre, 2014), therefore becoming a continuous source of air and water pollution (Mukhopadhyay et al., 2013). Contemporary mining sites are designed in such a manner to mitigate their impact on the environment. As one of the activities to reduce the risks of mining waste for the environment, Bradshaw and Johnson (1992) recommended revegtation as the most promising approach rather than the application of physicochemical treatments (Ortega-Larrocea et al., 2010).

Unfavorable conditions for plant growth at mining sites present the most crucial limitation in using revegetation for soil remediation (Mulligan et al., 2001). In particular, contamination levels, low-soil fertility, lack of organic matter, disturbance of soil chemical and physical properties, and disappearance of soil microbiota comprise the most common obstacles in revegetation (Borišev et al., 2018). Considering these limitations, selection of proper plant species (and genotypes, where applicable) is a prerequisite for future success of surface mine reclamation. While there are three afforestation strategies for restoration (i.e., pioneer species, climax species, or the biodynamic method combining pioneer and climax species) (Pietrzykowski et al., 2015a), in most cases the use of phytoremediation and erosion control with pioneer species is applied to promote natural succession with more demanding species (Pietrzykowski et al., 2015a,b). Sometimes well-adapted pioneer species like white poplar (*Populus alba* L.) and false indigo-bush (*Amorpha fruticosa* L.) naturally colonize mine sites (Pavlović et al., 2004), which can serve as a meaningful indicator for selection of species for revegetation. Low fertility soils can be enriched by the use of nitrogen fixating species like black locust (*Robinia pseudoacacia* L.) or elder (*Sambucus* L. spp.) (Haynes, 2009). Fast growing species can be used to establish SRWC plantations (Caterino et al., 2017; Quinkenstein and Jochheim, 2016) and improve soil and other ecosystem services at mining sites. However,

for better plant survival and performance, soil amendments play a crucial role and are often necessary. For example, amendments such as fertilizers (Hao et al., 2004), seeds of grasses with biosolids (Pietrzykowski et al., 2015a,b), or water holding polymers and microbial fertilizers (Pilipović et al., 2012b) substantially contribute to establishment, survival, and growth of planted seedlings. In addition, esthetic benefits may be achieved through the visual effect of greening the environment with fast growing tree species in relatively short time. However, in the long term, stability of the ecosystem is most often obtained through the use of climax species, which is similar to the urban forests showcased previously. The establishment of plantations with climax or biodynamic species ensures sustainability of the ecosystem. Benefits of such vegetation types can be observed through higher CO_2 sequestration (Brunori et al., 2017; Yuan et al., 2016), enhanced soil health properties (Maiti, 2007), and creation of favorable conditions for soil microbiota (that could last for more than 20 years) (Anderson et al., 2008).

Often, decreased pollution caused by mining activities is considered during the remediation of degraded mine sites, in addition to the restoration of soil and vegetation cover. Copper, Pb, Zn, and other metal mining activities leave land-area footprints that are much larger than the actual size of the mining/disposal sites, which creates the need for application of phytoremediation at these operations. The presence of contamination limits the spectrum of woody species that can be used for this purpose. Most of the research for phytoremediation of heavy metal contaminated sites includes poplars, willows, black locust, and other fast-growing species (Fig. 9.7) (Borghi et al., 2008; Borišev et al., 2016; Di Baccio et al., 2003; Nikolić et al., 2008; Župunski et al., 2016), which can be used for phytoextraction and phytomining of metals. The fast growth of these genera can be combined with their phytoextraction capability in order to obtain both economic and environmental benefits of surface mine reclamation.

The outcomes from research results and practical experience worldwide indicate a high level of complexity associated with remediation of mining sites. In particular, each site has its own soil and contaminant peculiarities that should be analyzed in order to choose the proper remediation technology. First, selection of suitable woody species for mine site recultivation is limited both by environmental factors and objectives of the applied activities. When considering environmental factors, the presence of contamination is most important and, as such, is used to inform what further activities are needed on site. Based on this contamination, selection of species must be matched to their efficiency in phytoremediation, followed by the selection of plantation type. On the other hand, the lack of contamination (e.g., where reduced runoff may be the primary objective) slightly increases the spectrum of potential species, which is then subjected to habitat limitations caused by soil and climate properties (i.e., genotype $\times$ environment interactions). Habitat limitations can be mitigated to some extent by the use of different amendments and measures to promote plant survival during establishment. But in the long run, to establish sustainable ecosystems, there is a need to promote measures aimed at climax phytocenosis. Therefore it is necessary to acquire knowledge about interactions among existing plant species,

FIGURE 9.7

Mine tailings at copper mine RTB "Bor" in Serbia with (A) naturally occurring pioneer tree species of white poplar (*Populus alba* L.), silver birch (*Betula pendula* L.), and black locust (*Robinia pseudoacacia* L.) in the bottom of the photo, and (B) black locust and false indigo-bush (*Amorpha fruticosa* L.) revegetated on the reclaimed plateau in the upper right of the photo.

Photo by A. Pilipović (University of Novi Sad).

microbiota, and soil prior to defining site-specific silvicultural measures. Such measures should be included in all phases of surface mine development to secure benefits of ecosystem services provided by mining activities.

9.5 Summary

Anthropogenic activities worldwide have caused ecological degradation that has resulted in the need to mitigate damage to essential ecosystem services in rural and urban areas. Phytoremediation and associated phytotechnologies are ideal for such applications and require extensive knowledge of soil–plant interactions for restoration to be successful. The information presented previously detailed remediation of contaminated soils using plants, focusing on selection of appropriate

plant materials and soil factors important for designing remediation systems. Critical principles and key points include:

- Functional plant groups used for phytoremediation include grasses, SRWCs, and later successional forests species (e.g., oak, maple, and birch). Grasses are used more than other herbaceous species given their broad genetic diversity, wide range of stress tolerances, capability of forming sod or dense cover, and existing worldwide seed industry for commercial distribution. SRWCs are desirable for remediation systems given their extensive root systems, fast growth, and elevated hydraulic control potential, while slower growing species are used given their longevity and esthetics.
- The two primary criteria used to select plant materials for remediation include environment (i.e., terrestrial vs aquatic) and functional group.
- Testing is conducted at bench, pilot, and field scales using methodologies such as phyto-recurrent selection (i.e., using multiple testing cycles to identify superior genotypes with elevated phytoremediation potential).
- Microbial symbionts such as AM fungi can enhance adaptation and survivorship of host plants during phytoremediation.
- While traditional plant breeding approaches are generally conducted under field conditions to mimic real-world production environments, methodologies for phytoremediation may need to include screens of plant growth and development in controlled conditions utilizing soils or other conditions (e.g., wastewater irrigation) from the remediation site to test whether genotypes will survive the contaminants before investing in field trials.
- When advanced to field-scale testing, genotypes should have their field performance validated prior to large-scale deployment to verify efficacy of the system before too many resources are put toward multiplication of seed or vegetative cuttings.
- Similar to traditional plant breeding, development of genotypes for phytoremediation depends on (1) heritability of the traits of interest, (2) selection pressure applied, (3) breeding procedure, and (4) time required to complete a selection generation.
- The primary parameters used to test plant material during phytoremediation include allometric, anatomical, physiological, and biochemical traits, although the most important parameter for measuring remediation success is contaminant concentrations in roots, stems, and leaves.
- There must be sufficient genetic variation in base populations to allow for reliable selection of the “best” individuals, based on specific soil/contaminant conditions and primary traits of interest.
- The key functions of soils related to environmental pollution are to store, filter, and transform compounds—which is complementary to those of plants and should not be overlooked during phytoremediation.

- The most frequently used soil quality indicators are total organic matter and pH, while key factors to consider when designing remediation systems include (1) scale, (2) level of physical disturbance, (3) concentration of contaminants, and (4) hydrologic flow paths.
- Successful remediation technologies have been used across the rural to urban continuum, with examples, including (1) grasslands used for phytoremediation of soil phosphorus, (2) urban afforestation used to create forests in cities, (3) riparian buffer systems used to reduce agrichemical transport from agroecosystems, (4) SRWCs used to enhance ecosystem services at landfills, and (5) woody species used for surface mine reclamation.

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