

# Phytoremediation and rehabilitation of municipal solid waste landfills and dumpsites: A brief review

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## Abstract

Environmental problems posed by municipal solid waste (MSW) are well documented. Scientifically designed landfills and/or open dumpsites are used to dispose MSW in many developed and developing countries. Non-availability of land and need to reuse the dumpsite space, especially in urban areas, call for rehabilitation of these facilities. A variety of options have been tried to achieve the goals of rehabilitation. In the last couple of decades, phytoremediation, collectively referring to all plant-based technologies using green plants to remediate and rehabilitate municipal solid waste landfills and dumpsites, has emerged as a potential candidate. Research and development activities relating to different aspects of phytoremediation are keeping the interest of scientists and engineers alive and enriching the literature. Being a subject of multi-disciplinary interest, findings of phytoremediation research has resulted in generation of enormous data and their publication in a variety of journals and books. Collating data from such diverse sources would help understand the dynamics and dimensions of landfill and dumpsite rehabilitation. This review is an attempt in this direction.

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## 1. Introduction

A landfill is an extremely variable and heterogeneous environment, as evident from the diversity of refuse composition with respect to location and time. Landfills hold wastes containing a wide range of organic molecules of both natural and xenobiotic origin. In many developed countries, municipal solid wastes (MSW) are dumped in scientifically designed sanitary landfills. In many developing countries, they are dumped in an uncontrolled manner without any precaution to deal with gas emissions and leachate generation, which pose a threat to the environment.

Natural or planted vegetation on a landfill has an important role in erosion control and removal of contaminants, besides imparting aesthetic value. Moreover, it may also be used in leachate treatment (Maurice, 1998). Landfill vegetation often shows signs of damage commonly

caused by the presence of landfill gas (LFG) in the root zone. The goal for the reconstruction of a suitable medium for landfill revegetation is to provide a capping that is deep and as favorable to root growth as is necessary to achieve desired plant performance (Vogel, 1987).

Although reviews on phytoremediation of sites contaminated with a variety of contaminants are readily available (Siciliano and Germida, 1998a; Lasat, 2002; Schwitzguebel et al., 2002), the applicability of this technology in remediation and rehabilitation of municipal solid waste dumpsites has not been given its due. The present review, an off-shoot of studies on rehabilitation of municipal solid waste dumpsites, attempts to fill this gap by leaning on research findings, especially those reported in the last two decades.

## 2. Phytoremediation

Exhaustive information on the state of the science and engineering of phytoremediation is available in McCutcheon and Schnoor (2003). These authors have approached the subject from the perspectives of biochemistry, genetics,

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toxicology, and pathway analysis. Their work covers the following aspects of phytoremediation: overview of science and applications; fundamentals of phytotransformation and control of contaminants; science and practice for aromatic, phenolic, and hydrocarbon contaminants; transformation and control of explosives; fate and control of chlorinated solvents and other halogenated compounds; modeling, design, and field pilot testing; and latest advances.

Phytoremediation, collectively referring to all plant-based technologies, uses green plants to remediate contaminated sites (Sadowsky, 1999). This technology draws its inspiration from the myriad of physical, chemical and biological interactions occurring between plants and the environmental media (Fig. 1). Phytoremediation is evolving into a cost-effective means of managing wastes, especially excess petroleum hydrocarbons, polycyclic aromatic hydrocarbons, explosives, organic matter, and nutrients. Applications are being tested for cleaning up contaminated soil, water, and air (McCutcheon and Schnoor, 2003). Several features make phytoremediation an attractive alternative to many of the currently practiced *in situ* and *ex situ* technologies. These include: low capital and maintenance

costs, non-invasiveness, easy start-up, high public acceptance and the pleasant landscape that emerges as a final product (Boyajian and Carreira, 1997). In the last several decades, phytoremediation strategies have been examined as a means to clean up a number of organic and inorganic pollutants, including heavy metals (Kumar et al., 1995; Salt et al., 1995; Chaney et al., 1997), chlorinated solvents (Walton et al., 1994; Haby and Crowley, 1996), agrochemicals (Anderson et al., 1994; Hoagland et al., 1997; Kruger et al., 1997), polycyclic aromatic hydrocarbons (Aprill and Sims, 1990; Reilly et al., 1996), polychlorinated biphenyls (Brazil et al., 1995; Donnelly and Fletcher, 1995), munitions (Schnoor et al., 1995) and radio nuclides (Entry et al., 1997). These soluble organic and inorganic contaminants, which move into plant roots or rhizosphere by the mass flow process of diffusion, appear to be most amenable to the remediation process (Schnoor et al., 1995; Cunningham et al., 1996). In several instances, plants and/or their attendant rhizosphere microbes have been shown to transform some chemical compounds to some degree (Walton et al., 1994; Crowley et al., 1996; Siciliano and Germida, 1998b).

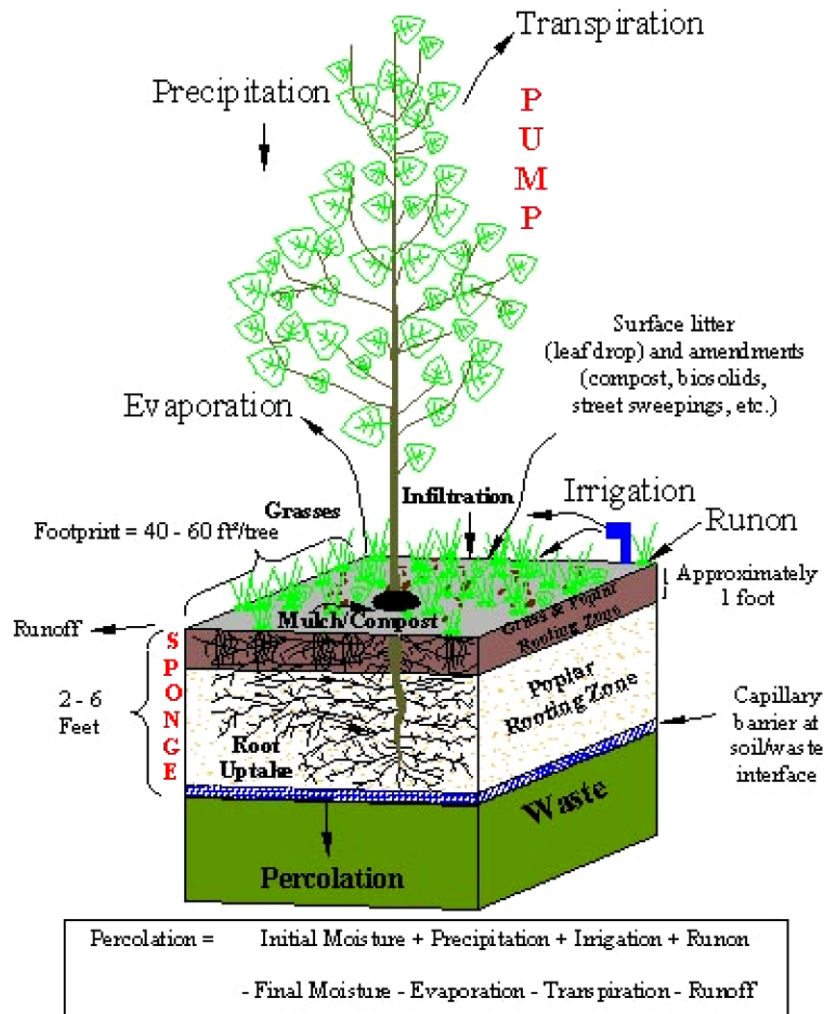


Fig. 1. Plant–environment interactions. Source: Licht and Isebrands (2005).

Plants are known to sequester, degrade and stimulate the degradation of organic contaminants in soil (Anderson et al., 1993; Shimp et al., 1993). The sequestration of heavy metals by plants is an effective method of reducing heavy metal contamination in soil (Cunningham et al., 1995). Sequestration of toxicants by plants is an important area of phytoremediation research. Plants are known to accumulate a variety of toxicants from soil (Paterson et al., 1990) and if the toxic chemical is metabolically stable and mobile, it may be transferred *via* apoplast or symplast compartments, or both, throughout most of the plant as parent compound and stored at highly bioconcentrated levels (McFarlane et al., 1987). However, the mechanisms by which plants stimulate the disappearance of hazardous organics from soil are not fully understood.

In view of its demonstrated potential, phytoremediation has been gaining importance in rehabilitation of contami-

nated sites including MSW dumpsites. Many types of phytoremediation processes have been described based on the kind of mechanism. These include: phytoextraction, rhizofiltration, phytovolatilization, phytodegradation, rhizosphere biodegradation, hydraulic pumping, phytosorption and phytocapping. Fig. 2 outlines the common processes involved in phytoremediation. Processes and contaminants dealt with by different phytoremediation processes are presented in Table 1. The selection of plant and the type of phytoremediation depends on the type of contaminants to be treated and the nature of the site.

### 3. Interactions between plants and microbes

Municipal solid waste contains a large microbial population and may be heavily contaminated with pathogenic microorganisms (Gaby, 1975). Municipal solid waste landfills often contain animal remains and feces, hospital wastes

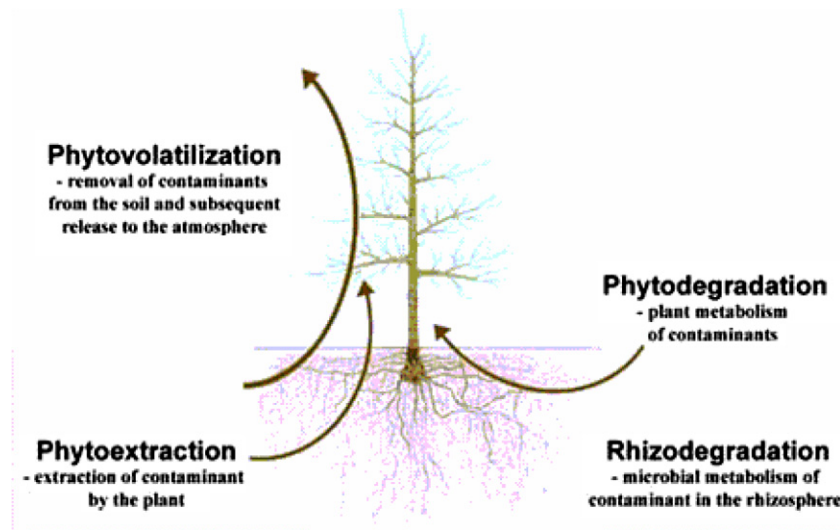


Fig. 2. Processes involved in phytoremediation. Source: <http://oldweb.northampton.ac.uk/aps/env/landfillleachate/images/phytorem.jpg>.

Table 1  
Types and processes involved in phytoremediation

Type	Contaminant	Process
Phytoextraction	Heavy metals: arsenic, cadmium, chromium, copper, mercury, lead, zinc	High biomass, metal hyperaccumulators extract metals from soil and accumulate them in shoots
Rhizofiltration		Plant roots growing in polluted water precipitate and concentrate metals
Phytostabilization		Heavy-metal tolerant plants stabilize the metal in soil and render them harmless
Phytovolatilization		Plants extract volatile metals like Hg and Se from the soil and volatilize them from the foliage
Phytodegradation		Plants absorb the contaminants and degrade them within the plant system
Rhizosphere biodegradation		Plants release exudates and enzymes which directly degrade the pollutant and/or induce the microbes which are involved in degradation
Hydraulic pumping		Plant roots grow to the water table, take up water and prevents the migration of polluted water
Phytovolatilization		Plants take up the pollutants along with water, pollutants pass through xylem and are released from foliage
Phytosorption		Adsorption of pollutants by plant roots and leaves and prevention of the pollutant movement
Phytocapping		Plants consume water from the rainfall and reduce leaching and pollutant movement

and domestic sewage sludge that pose a potentially significant health hazard.

There can be a significant bacterial population associated with municipal landfill leachates. The acute bacterial content of leachate, particularly the members of coliforms and fecal streptococci vary with the age and the chemical properties of the leachate (Senior, 1990). A limited number of bacterial pathogens have been found in leachates from commercial and experimental landfills and experimental lysimeters (Reinhart and Grosh, 1998). Ware (1980) found increase in bacterial mortality with time of leaching or refuse age due to the bactericidal effects of the leachates of the landfill. Relatively high temperatures achieved in the aerobic stage of refuse biodegradation can inhibit bacterial growth and survival (Reinhart and Grosh, 1998).

Besides sequestering or metabolizing contaminants, plant roots may increase contaminant degradation *in situ* via their root systems. Plant roots and their exudates increase microbial numbers in the soil surrounding them by one or two orders of magnitude, thus increasing microbial activity (Siciliano and Germida, 1998c).

Donnelly et al. (1994) suggest that plants specifically increase degradation of certain contaminants in soil by providing the soil microflora with polyphenolic compounds. These compounds, in turn, will induce bacterial enzymes that can degrade a variety of pollutants such as trichloroethylene (TCE) or polychlorinated biphenyls (PCBs). These authors have screened a wide range of plants for production of polyphenolics that support PCB-degrading bacteria and identified mulberry (*Morus rubia* L.) as a possible plant species suited to remediate PCB-contaminated soil sites (see also Fletcher and Hegde, 1995; Hegde and Fletcher, 1996). However, it is not clear if these plants would increase exudation in the presence of contaminants. In contrast, other workers have suggested that stimulation of bacteria may occur indirectly owing to nutrients released from roots i.e., a non-specific relationship (Schnoor et al., 1995). These nutrients, often low molecular weight organic acids, increase microbial biomass and activity but do not normally induce specific enzymatic processes that degrade xenobiotics. Consequently, plant species with deep fibrous roots that can grow in stressed environments are used in phytoremediation studies.

Plants and bacteria are known to form specific associations in which the plants provide the bacteria with a specific carbon source that induces the bacteria to reduce the phytotoxicity of the contaminated soil (Siciliano and Germida, 1998b). Alternatively, plants and bacteria can form non-specific associations in which normal plant processes stimulate the microbial community, which in the course of normal metabolic activity degrades contaminants in soil (Zablutowicz et al., 1994). Similarly, the biochemical mechanisms have been reported to increase the degradative activity of bacteria associated with plant roots. In return, bacteria can augment the degradative capacity of plants or reduce the phytotoxicity of the contaminated soil (Aprill and Sims, 1990). The specificity of the plant–bacteria inter-

action is dependent upon soil conditions (Baker et al., 1991; Brown et al., 1994; McGrath et al., 1997), which can alter contaminant bioavailability (Marschner, 1995; Baker et al., 2000), composition of root exudates (Ma and Nomoto, 1996) and nutrient levels (Mathys, 1977; Still and Williams, 1980; Kramer et al., 1996). This aspect in respect of rehabilitation of MSW dumpsites assumes great significance owing to variations in municipal solid waste characteristics. In addition, the metabolic requirements for contaminant degradation may also dictate the form of the plant–bacteria interaction i.e., specific or non-specific (Anderson et al., 1993; Shimp et al., 1993). Siciliano and Germida (1998c) have reported that no systematic framework that can predict plant–bacteria interactions in a contaminated soil has emerged, but it appears that the development of plant–bacteria associations that degrade contaminants in soil may be related to the presence of allelopathic chemicals in the rhizosphere. Investigations on plants that are resistant to or produce allelopathic chemicals may throw much light on this interesting bacterial association.

Nicholas et al. (1997) found that the number of bacteria capable of degrading the contaminant increases in contaminated soil. Higher populations of bacteria in contaminated compared with non-contaminated rhizosphere cannot demonstrate selective enhancement of degrading populations. Similarly, increased levels of degrading bacteria in the rhizosphere compared with the bulk soil cannot be taken as proof of selective enhancement. Proponents of non-specific interactions argue that specific stimulation of selected bacterial groups in soil may not be necessary for the plant to enhance contaminant degradation.

Specific plant–bacteria interactions still occur in phytoremediation, but may not be based on the strict genetic alteration seen between legumes and rhizobia. For example, Siciliano and Germida (1997) found that a combination of pseudomonas enhanced the phytoremediation activity of three different forage grasses while having no effect on other grass species. One of these strains was isolated as a plant-growth-promoting rhizobacteria (PGPR) of wheat, whereas the other was isolated from soil contaminated with 2-chlorobenzoic acid. It is unlikely that genetic alterations in the plant or bacteria are the basis for the enhanced phytoremediation activity seen when these two organisms are combined (see also Guerinot, 2000).

Walton et al. (1994) propose that plants produce specific signals in response to specific contaminants. As a result, bacteria detoxify contaminants in soil and the plant provides root exudates that either supply energy source or in some other way increase microbial detoxification activity in the rhizosphere. The key point to this association is that the plant alters its behaviour in contaminated soil to stimulate microbial communities that degrade contaminants. Plants that encounter toxicants in soil will not survive unless they can find a way to detoxify the contaminant. Over the millennia, plants have developed means of using rhizobacteria as a method to detoxify toxins in soil.



#### 4. Vegetation at dumpsites

Plants are known to increase nutrient availability by secreting cationic chelators, organic acids, or specific enzymes such as phosphatase into the soil systems. Competition for these nutrients by degrading and non-degrading species will influence the amount of contaminant degraded (Steffensen and Alexander, 1995). Increases in nutrient availability brought about by plant growth may be one mechanism by which plants stimulate biodegradation. Supporting this, Cheng and Coleman (1990) found that living roots and fertilizers had equivalent stimulatory effects on straw decomposition. Furthermore, atrazine degradation by an inoculated consortium was similar in treatments receiving fertilizer and those in which corn plants were grown (Alvay and Crowley, 1996).

Besides increasing the availability of nutrients, plants may also increase the bioavailability of the contaminant. This feature is of significance in the context of landfill vegetation. Contaminant bioavailability often limits biodegradation, and increasing it can stimulate degradation (Siciliano and Germida, 1998a). Root exudates can increase contaminant bioavailability by competing with the contaminant for binding sites on the soil matrix.

A good starting point for selection of appropriate plant species for the remediation and rehabilitation of dumpsites is to employ endemic species. Although landfills only cover a limited surface, they often offer a large diversity of environmental niches for species. Several fluxes of waste and cover materials with different origins end up at landfills and create microhabitats on which a certain type of vegetation will have a competitive advantage and develop while other species will be rare. The age of the cover also

accounts for the occurrence of landfill plants (Maurice, 1998). Similarly, in some cases the landfill plant species are related to human activities in the feeder area (Example: Kalix in Sweden – Stenberg, 1997).

Maurice et al. (1995) have reported that plants belonging to four families *viz.*, Poaceae, Asteraceae, Polygonaceae and Chenopodiaceae dominate, while other species occur only sporadically in Stockholm, Malmo and Helsingborg landfills of Sweden. Their observations further indicate that the species diversity decreases with the age of the landfill. Dwyer et al. (2000) have quantified the plant species occurring in Albuquerque, USA, with reference to different landfill covers. According to them, the perennial grass and annual weeds were abundant in different landfill covers.

At Kodungaiyur and Perungudi dumping grounds in Chennai, India, the dominant plant species recorded were *Acalypha indica*, *Lycopersicon esculentum*, *Parthenium hysteroporus*, *Cynodon dactylon* and *Cucurbita maxima* (Study of the authors of this review).

Reviewing plant species occurring at different landfills facilitates the selection of suitable plant species to deal with a range of contaminants together. It is interesting to note that the species diversity is influenced by the nature of origin of wastes, local flora and the conditions prevailing at the landfill. Hence, a single species cannot be identified as a universal indicator and the plant selection should be based on the climatic conditions and the native plants occurring in a particular landfill.

#### 5. Factors influencing landfill vegetation

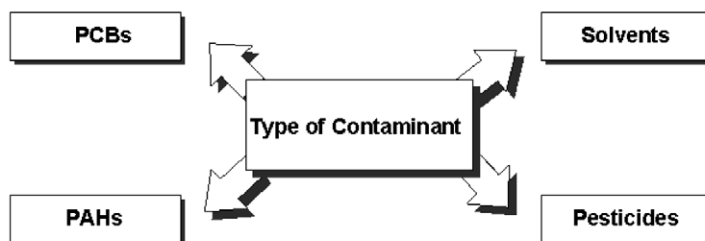
Reclamation of a landfill site must include the objective of containing the material within. This is because the pro-

Phenolic compounds in root exudate have been shown to support growth of PCB-degrading bacteria.

- Donnelly *et al.*, 1994
- Red mulberry (*Morus rubra*)

Plant species from a solvent waste site were able to enhance microbial mineralization of TCE.

- Anderson and Walton, 1995
- Lespedeza (*Lespedeza cuneata*)
- Loblolly pine (*Pinus taeda*)



Degradation of PAHs has been enhanced in the rhizosphere of prairie grasses and alfalfa.

- Aprill and Sims, 1990; Reilly *et al.*, 1996
- Prairie grasses
- Alfalfa (*Medicago sativa*)

Herbicide-resistant plants found on pesticide-contaminated sites were able to enhance microbial degradation of certain pesticides.

- Perkovich *et al.*, 1996
- Kochia (*Kochia scoparia*)

Fig. 3. Influence of different types of contaminants on vegetation.

Table 2  
Landfill conditions and their effects on plants

Landfill conditions	Consequences for plants
Good drainage	Risk for dryness
High temperature	Dryness and increase of oxygen demand
Construction above the surface	Exposure to wind, dryness
Infertile top cover soil	Bad growth conditions
Gases	Bad growth conditions
Compact soil	Extra energy required to push root tips

Source: Maurice (1998).

cesses that take place after the compaction and the covering of the waste in the site produce products of entirely new characteristics. Many of the products of these processes are toxic to several life forms including plants (Zacharias, 1995). The influence of different types of contaminants on vegetation is depicted in Fig. 3. Toxic materials in the waste include landfill gases, leachates, heavy metals, organic contaminants and others. Usual landfill conditions and their consequences on the vegetation are presented in Table 2. Vegetation in a completed landfill is often poor and damaged (Leone et al., 1977; Moffat and Houston, 1991; Gendebien et al., 1992).

## 6. Influence of landfill gases on vegetation

Landfill gas is a mixture of seven gases, namely methane, carbon dioxide, carbon monoxide, hydrogen, oxygen, nitrogen and hydrogen sulphide, in varying proportions. Methane and CO<sub>2</sub> are the dominant gases in LFG, varying in concentrations between 40% and 60% (Lelieveld et al., 1993).

Methane and CO<sub>2</sub> are both present in such high concentrations that they displace the soil O<sub>2</sub>. The resulting lack of O<sub>2</sub> causes death by asphyxia, which seems to be the first cause for vegetation kill (Gendebien et al., 1992). Most plants require O<sub>2</sub> concentration of 5–10% in the soil–gas phase. The first symptoms of asphyxia are chlorosis, i.e., leaves start to yellow due to the loss of reduced development of chlorophyll. Extreme temperatures, lack of water, infection, iron deficiency, high soil pH, etc., can also cause chlorosis (Flower et al., 1981). Asphyxia can lead to deficiencies in K, N, P, Ca and Mg (Gendebien et al., 1992).

It has been reported that methane is not phytotoxic in itself but can lead to asphyxia (Leone et al., 1977; Flower et al., 1981). Anaerobic microbial activity, induced by the lack of O<sub>2</sub>, tends to lower the organic carbon to nitrogen ratio (C:N) of the soil and may also lower soil pH to levels unfavorable to plant growth (Gendebien et al., 1992). Elevated CO<sub>2</sub> levels, common at most landfills, are directly toxic to the roots even when enough O<sub>2</sub> is available. The normal concentration of CO<sub>2</sub> in soils is between 0.04% and 2%. Normal development of plants can occur with a CO<sub>2</sub> concentration of 5%. When present in excess of 20%, CO<sub>2</sub> is generally phytotoxic (Gendebien et al., 1992).

## 7. Enhanced methane oxidation

Bergman (1995) noticed a strong methane oxidation in landfills with more vegetation than those without. He suggested that the vegetation might be used to identify methane oxidation areas. But the reliability of the vegetation coupled methane oxidation has to be confirmed with a large number of samples in various climatic conditions. In temperate regions, methane emissions are detected generally during winter when the soil is frozen; in other seasons methane is seldom detected, which, according to Maurice and Lagerkvist (1997) is indicative of oxidation of methane by bacteria. These authors have reported high CO<sub>2</sub> concentrations during summer. The optimum conditions for the methane oxidation are 30 °C and 30% moisture content (Whalen et al., 1990; Boeckx and Van Cleemput, 1996). In tropical countries, where temperatures are high, methane may be quickly oxidized to CO<sub>2</sub> and H<sub>2</sub>O. Moisture content can be a major limiting factor during summer seasons.

## 8. Influence of leachates on vegetation

A complex of sequences mediated by physical, chemical and biological events occurs within a landfill. As a consequence, refuse is degraded or transformed. As water percolates through the landfill, contaminants are leached from the solid waste. Mechanisms of contaminant removal include leaching of inherently soluble materials, leaching of soluble biodegradation products of complex organic materials, leaching of soluble products of chemical reactions and wash out of fines and colloids (Reinhart and Grosh, 1998). The quality of the leachate produced is highly variable and depends on the composition of the solid waste, depth of waste, site hydrology, compaction, waste age, interaction of leachate with the environment, landfill design and operation, available oxygen and temperature. Moisture content is an important limiting factor of plant growth and development in landfills, especially in tropical climates. In tropical climates, rainfall is the primary source of moisture and hence supports the drought tolerant vegetation and determines the species diversity in landfills. In such cases, mono species phytoremediation aided by leachate circulation may be carried out to maintain the growth, accelerate the degradation and stabilize the wastes. Moreover, leachate circulation prevents the pollutants from entering the groundwater. Toxic components in leachates such as heavy metals may reduce the growth and development of plants.

Water is also a significant factor influencing waste stabilization and leachate quality. Water addition has been demonstrated to have a stimulating effect on methanogenesis (Barlaz et al., 1990). Moisture within the landfill serves as reactant in the hydrolysis reactions, transports nutrients and enzymes, dissolves metabolites, provides pH buffering, dilutes inhibitory compounds, exposes surface area to microbial attack and controls microbial swelling (Noble

and Arnold, 1991). Relatively dry landfills have very slow stabilization rates because there will only be a small quantity of moisture to support biological degradation (Reinhart and Grosh, 1998).

The composition of the waste determines the extent of biological activity within a landfill. It determines the compaction of the waste materials, which in turn influences the root development of the plants in the landfills. Rubbish, plant residues and animal residues contribute to the organic material in leachate (Pohland and Harper, 1985). Inorganic constituents are often derived from ash wastes and construction and demolition debris (Pohland and Harper, 1985). Lignin, the primary component of paper, is resistant to anaerobic decomposition, which is the primary means of degradation. For instance, high quantities of paper in solid wastes have been shown to decrease the rate of waste decomposition (Chen and Bowerman, 1974). High concentrations of pollutants yield high strength leachates that hamper the development of plants and activities of microbes.

High concentrations of constituents are found in leachates from deeper landfills under similar conditions of precipitation and percolation. Thus deeper fills are known to require more water to reach saturation, and require longer time for decomposition and distribution of the leached material (Qasim and Chiang, 1994). Deeper landfills are also known to offer greater contact time between the liquid and solid phases and increase the leachate strength (McBean et al., 1995).

Generally, in landfills, plants with short root length are preferred so as not to interrupt the underlying geomembrane. When the depth increases, small energy rotation trees with relatively high root length can be employed so as to extract maximum pollutants from the waste. Even then, particular care should be taken to ensure the safety of geomembrane. In unorganized dumpsites without a bottom membrane, it is essential to use appropriate plants to control the movement of pollutants to groundwater.

Landfill temperature, a largely uncontrollable factor, has been shown to fluctuate with seasonal ambient temperatures. Temperature affects bacterial growth, chemical reaction within the landfill, oxygen content and moisture availability. Aerobic degradation may continue to occur just below the surface of the fill. During aerobic degradation, microorganisms degrade organic matter to CO<sub>2</sub> and H<sub>2</sub>O and produce considerable heat (McBean et al., 1995). The heat thus produced may not permit the survival of vegetation at dumpsites. After the aerobic composting, when the wastes start to stabilize, the pH approaches 7 and temperatures also slightly increase.

The effect of different environmental factors including drying, shaking, soil-to-solution ratio, competing ions, solution composition, time, pH, and temperature on adsorption of boron from landfill leachate by peat has been investigated by Majid and Leta (2005). According to them, the statistical comparison of experimental results showed that shaking of adsorption samples, soil-to-solution ratio,

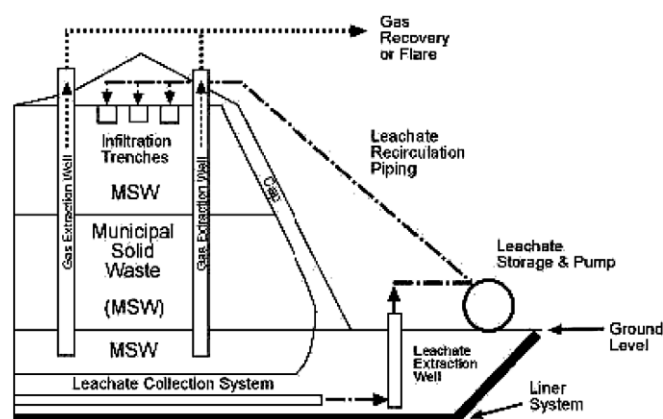


Fig. 4. Schematic of a bioreactor landfill using leachate recirculation. Source: Hughes and Christy (2003).

long-term adsorption, and competing ions did not have any significant effect on boron adsorption. However, solution composition, drying of peat prior to adsorption tests, pH, and temperature had a significant effect on the adsorption of boron by peat. Diluting leachate samples with distilled water had a negative effect on the adsorption capacity. Drying peat significantly reduced its boron adsorption capacity. Boron adsorption reached maximum level at a pH range of 9–9.5. Temperature had a negative effect on the adsorption of boron. The results of two-level factorial design experiments showed that pH had the strongest effect on the adsorption of boron by peat.

Studies on leachate circulation and development of bioreactor landfills have attracted the attention of workers in the last decade or so. Reinhart and Townsend (1998) and Reinhart et al. (2002) have reviewed several aspects of landfill bioreactors including their design and operation. A bioreactor landfill changes the goal of landfilling from the storage of waste to the treatment of waste. A bioreactor landfill is a system that is isolated from the environment and that enhances the degradation of refuse by microorganisms. Microbial degradation may be promoted by adding certain elements (nutrients, oxygen, or moisture) and controlling other elements (such as temperature or pH). The most widely used and understood method of creating a landfill bioreactor is the recirculation of leachate, since the element that usually limits microbial activity in a landfill is water. The recirculation of leachate increases the moisture content of the refuse in the landfill and, therefore, promotes microbial degradation. If leachate recirculation alone cannot raise the moisture content to levels at which microbial growth is enhanced (40% by weight, minimum), water may need to be added to the waste (Hughes and Christy, 2003). Fig. 4 presents a schematic of landfill bioreactor.

## 9. Heavy metals and volatile organic compounds (VOC)

High heavy metal concentrations tend to influence the vegetation in landfills. Metal solubility is dependant on soil

characteristics and is strongly influenced by soil pH and the degree of complexation with soluble ligands (Harter, 1983). Metals in soil exist as discrete particles or are associated with different soil components including: (1) free metal ions and soluble metal compounds in the soil solution, (2) exchangeable ions sorbed onto inorganic solid phase surfaces, (3) non-exchangeable ions and precipitated or insoluble metal compounds (e.g. oxides, hydroxides, phosphates or carbonates), and (4) metals bound in silicate minerals (Ramos et al., 1994). The metals considered readily available for plant uptake are those that exist as soluble components in the soil solution or are easily desorbed or solubilized by root exudates or other components of the soil solution, often only a small portion of the total metal content of the soil. Since effective phytoextraction is dependent on a relatively abundant source of soluble metal to achieve significant uptake in the plant shoots, the soil conditions may need to be altered to increase metal solubility and availability (Blaylock and Huang, 2000).

The amount of metal available for phytoremediation is estimated on the basis of the distribution of metal between the fractions of a sequential extraction. The results are interpreted with the understanding that the extracted fractions are operationally defined and not necessarily specific soil components. For example, the carbonate fraction consists of soluble compounds at pH 5 and is not limited solely to carbonate compounds. Chelating agents have been used to estimate metal bioavailability and are the basis for the DTPA (diethyl trinitrile penta acetic acid) soil test for micronutrient and heavy-metal availability (Lindsay and Norvell, 1978; Amacher, 1996).

Sequestration of pollutants within plants is the basis for phytoextraction of soils and water contaminated with heavy metals (Kumar et al., 1995; Raskin et al., 1997). Metals targeted by this process include Cd, Pb, Zn, Cu, Cr, Ni, Se and Hg. Phytoextraction using hyper accumulating plants is proving to be one of the most effective phytoremediation methods to clean up metal contaminated sites. Several plant species, including *Thlaspi* sp., have been shown to accumulate very high levels of Ni, Zn and Cd from soils (Baker and Brooks, 1989; Kramer et al., 2000). *Brassica juncea* has been found to be an excellent accumulator plant for metals such as Cd, Cr, Ni, Zn and Cu in soils (Kumar et al., 1995; Salt et al., 1995), and several plant species have been shown to accumulate Pb (Dushenkov et al., 1995; Cunningham et al., 1997). The enormous literature available on plant–metal interaction needs to be oriented towards the application in landfill remediation.

Generally, metal hyper accumulating plant species are herbaceous with small shoot and root systems. As they can extract metals only from top layers, their usefulness in landfill systems is limited. Screening for plant species that can explore large volumes of the wastes by their roots and ensure metal hyper accumulation is a potential field for research. It is well known that a single plant species cannot and will not accumulate all of the toxic metals. The growth of a nickel hyperaccumulator may be restricted by the pres-

ence of lead, which makes the remediation program less efficient. To overcome this problem, plants that are hyper accumulating one metal and tolerant to other metals need to be used. Plants capable of accumulating many metals are also appropriate. Research in these directions would be of immense value.

The high potential of poplar trees for successful biostabilization has been demonstrated by a number of workers (Miller et al., 1996). Rows of fast growing poplars with deep roots have been shown to absorb and transpire large quantities of water from the roots through shoots into the atmosphere. The application of trees such as poplars to control the movement of underground water in landfills is yet to gain momentum. This can be an effective tool in immobilizing water-soluble contamination and preventing migration; this may not remove contaminants such as heavy metals from the site (see also Schnoor, 2000).

Contributions of Miller et al. (1996) with regard to the management of landfill leachate using vegetation deserve a special mention. Working on phytoremediation of VOC contaminated groundwater, these authors have demonstrated the use of poplar trees in managing the plume. Based on their laboratory studies, the authors have also discussed the possible biodegradation of the contaminants in the root zone. A comparison of costs involved in traditional mechanical pump and treat method and phytoremediation has also been provided by them.

## 10. Landfill capping

Landfills are usually required to have clay caps and impermeable synthetic membranes to minimize the infiltration of rainfall and generation of leachate. Landfill capping is the most common form of remediation because it is generally less expensive than other technologies and effectively manages the human and ecological risks associated with a remediation site. Considerable research is being done to develop inexpensive and efficient layers. As outlined in Platinium International, Inc. (2002), landfill caps can be used to

- minimize exposure on the surface of the waste facility;
- prevent vertical infiltration of water into wastes that would create contaminated leachate;
- contain waste while treatment is being applied;
- control gas emissions from underlying waste;
- create a land surface that can support vegetation and/or be used for other purposes.

CH2M Hill has developed a technology by which soil profile water content was engineered by thickening soil layers of a vegetative cap coupled with capillary barriers. In this system, while the soil layer detained precipitation soaking into the cap, the plants removed the moisture that in turn, renewed the detention capacity of the cap. Significant strides made in the field of vegetative caps are detailed in Rock (2003).



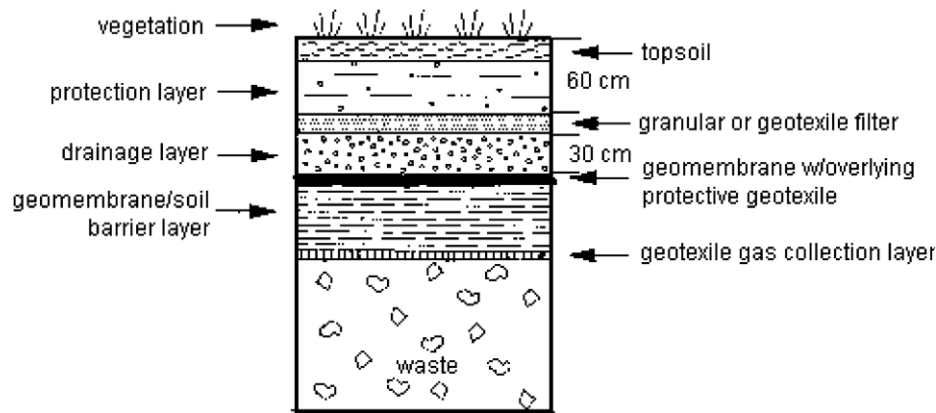


Fig. 5. Typical landfill cap system. Source: Platinum International, Inc. (2002).

A vegetative cap is a long-term, self-sustaining cover of plants growing in and/or over materials that pose environmental risk; a vegetative cap reduces that risk to an acceptable level and requires minimal maintenance. A typical landfill cap system is shown in Fig. 5. In the USA, landfill caps have been required under the Resource Conservation and Recovery Act (RCRA) since the mid-1980s. Hazardous waste landfills are regulated under Subtitle C in the Federal Register (40 CFR-264 and 265), and Subtitle D (40 CFR 257 and 258), which includes non-hazardous wastes such as municipal landfills.

Vegetative caps are also called “alternative covers” and “evapotranspiration landfill covers”. Their purpose is to increase evapotranspiration from the surface of the landfill and enhance bioremediation. A further advantage of the alternative vegetative cap is more rapid “stabilization” of the wastes, decreased gas production after 5–20 years, and earlier access to the site for alternative uses (parkland, municipal building construction). Disadvantages include the possibility of phytotoxicity, pests, or weather destroying the trees and decreasing the efficiency of the alternative cap. Other disadvantages are that it is a less proven system, and state regulations sometimes do not allow alternative caps (Schnoor, 2002).

Vegetative caps are known to significantly reduce closure costs. According to the USEPA (1997), while the cost (in US dollars) of conventional caps can range from \$24,000 to \$40,000 per ha, the cost of vegetative caps ranges from \$5500 to \$12,000 per ha.

A comprehensive evaluation of the field experiments on the landfill caps and alternative landfill covers has been made by Rock (2003).

### 11. Limitations of phytoremediation of landfill sites

The success of phytoremediation depends on the growth of the plants (Baker et al., 1994; Brown et al., 1994). More time may be required to phytoremediate a site when compared to conventional cleanup technologies. Excavation and landfill, or incineration may take weeks to months to accomplish whereas phytoextraction or degradation may

need several years (Salt et al., 1995). Therefore, for sites that pose acute risks for human and other ecological receptors, phytoremediation may not be the technique of choice (see also Raskin and Ensley, 2000).

Root contact is a primary limitation in phytoremediation applicability. Remediation with plants requires that the contaminants be in contact with the root zone of the plants. Either the plants must be able to extend their roots to the contaminants or the contaminated media must be moved to the rhizosphere of plants. This movement can be accomplished with standard agricultural practices such as deep plowing to bring soil from 60 or 90 cm deep to within 20–25 cm of the surface for shallow rooted crops and grasses or by irrigating trees and grasses with contaminated groundwater or wastewater (Rock and Sayre, 2000).

High concentrations of contaminants and/or toxicants may inhibit plant growth and thus limit application on some sites or some parts of sites. This phytotoxicity may warrant a tiered remedial approach in which high-concentration waste is handled with expensive *ex situ* techniques such as excavation and landfilling, soil washing by particle separation (Berti and Cunningham, 2000), vitrification, thermal treatment and electrokinetics (Glass, 2000). These quickly reduce acute risk whereas *in situ* phytoremediation may be used over a longer period of time to clean high volumes of lower concentrations of contaminants. Sites with widespread, medium-level contamination within the root zone are potential candidates for phytoremediation processes (Rock and Sayre, 2000).

### 12. Conclusions

Landfills and dumpsites used for disposal of municipal solid wastes require occasional rehabilitation, especially in the context of upgrading such facilities in developing economies. Rehabilitation measures and strategies must ensure that at all stages of the exercise, environmental concerns including groundwater contamination resulting from migration of leachate, transmigration of pollutants and aesthetics are not overlooked. Phytoremediation offers via-

Table 3  
Applications of phytoremediation

Sl. no.	Application	Description	Contaminants	Types of plants
<i>Soils</i>				
1	Phytotransformation	Sorption, uptake, and transformation of contaminants	Organics, including nitroaromatics and chlorinated aliphatics	Trees and grasses
2	Rhizosphere biodegradation	Microbial biodegradation in the rhizosphere stimulated by plants	Organics; e.g., PAHs, petroleum hydrocarbons, TNT, pesticides	Grasses, alfalfa, many other species including trees
3	Phytostabilization	Stabilization of contaminants by binding, holding soils, and/or decreased leaching	Metals, organics	Various plants with deep or fibrous root systems
4	Phytoextraction	Uptake of contaminants from soil into roots or harvestable shoots	Metals, inorganics, radionuclides	Variety of natural and selected hyperaccumulators, e.g., <i>Thalasspi</i> , <i>Alyssum</i> , <i>Brassica</i>
<i>Water/groundwater</i>				
5	Rhizofiltration	Sorption of contaminants from aqueous solutions onto or into roots	Metals, radionuclides, hydrophobic organics	Aquatic plants (e.g., duckweed, pennywort), also <i>Brassica</i> , sunflower
6	Hydraulic control plume capture/phytotrans	Removal of large volumes of water from aquifers by trees	Inorganics, nutrients, chlorinated solvents	Poplar, willow trees
7	Phytovolatilization	Uptake and volatilization from soil water and groundwater; conversion of Se and Hg to volatile chemical species	Volatile organic compounds, Se, Hg	Trees for VOCs in groundwater; <i>Brassica</i> , grasses, wetlands plants for Se, Hg in soil/sediments
8	Vegetative Caps	Use of plants to retard leaching of hazardous compounds from landfills	Organics, inorganics, wastewater, landfill leachate	Trees such as poplar, plants (e.g., alfalfa) and grasses
9	Constructed wetlands	Use of plants as part of a constructed ecosystem to remediate contaminants from aqueous wastestreams	Metals, acid mine drainage, industrial and municipal wastewater	Free-floating, emergent, or submergent vegetation; reeds, cattails, bamboo

Source: Schnoor (2002).

ble solutions to many environmental problems related to landfill rehabilitation and has several applications as summarized in Table 3.

Phytoremediation research has demonstrated its usefulness in waste management. Working on this aspect, Jordahl et al. (2003) elaborate the rationale for using trees with irrigation to manage contaminated water, identify key limitations and provide general design guidance. They argue that phyt irrigation provides a relatively inexpensive means of moving impaired water to a planted area or forest for treatment, greatly expanding the ways in which phytoremediation can be used. According to them, irrigation systems can be used to apply water at the land surface or below to meet the requirements of treatment and for regulatory and public acceptance. Irrigation-system designs based on trees are particularly advantageous because of the high water use, deep rooting, and low operations and maintenance costs of tree systems.

The understanding of the dynamics of phytoremediation requires a multi-disciplinary approach involving the biology, biochemistry and engineering of remediating systems. Even the advances in the processes of phytoremediation have to be changed to adapt to landfill conditions. Thus, tremendous scope exists for investigating different facets of this technology and its application to real-world conditions such as municipal solid waste landfills and dumpsites.

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