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Ground Water Issue

Phytoremediation of Contaminated Soil and Ground Water at Hazardous Waste Sites

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Background

The EPA Regional Ground Water Forum is a group of EPA professionals representing Regional Superfund and Resource Conservation and Recovery Act (RCRA) Offices, committed to the identification and resolution of ground-water issues impacting the remediation of Superfund and RCRA sites. The Forum is supported by and advises the Superfund Technical Support Project. Emerging technologies that could provide effective cleanup at hazardous waste sites are of interest to the Forum. Phytoremediation, the use of plants in remediation, is one such technology. This issue paper focuses on the processes and applications of phytoremediation for remediation of hazardous waste sites.

The purpose of this issue paper is to provide a concise discussion of the processes associated with the use of phytoremediation as a cleanup or containment technique for remediation of hazardous waste sites. Introductory material on plant processes is provided. The different forms of phytoremediation are defined and their applications are discussed. The types of contaminated media and contaminants that are appropriate for phytoremediation are summarized. Information is provided on the types of vegetation that have been studied or used in phytoremediation. The advantages and disadvantages of phytoremediation are discussed, and some cost information is provided. Considerations for design of a phytoremediation system are introduced; however, this issue paper is not a design manual. Citations and references are provided for the reader to obtain additional information. The issue paper is intended for remedial project managers, on-scene coordinators, and others involved in remediation of hazardous waste sites. It provides a basic understanding of the numerous issues that should be examined when considering the use of phytoremediation. The issue paper is intended to be an updated, more concise version of information presented in the *Introduction to Phytoremediation* (EPA/600/R-99/107), in a format that will facilitate use of this information.

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Introduction

Phytoremediation is the use of plants to partially or substantially remediate selected contaminants in contaminated soil, sludge, sediment, ground water, surface water, and waste water. It utilizes a variety of plant biological processes and the physical characteristics of plants to aid in site remediation. Phytoremediation has also been called green remediation, botano-remediation, agroremediation, and vegetative remediation. Phytoremediation is a continuum of processes, with the different processes occurring to differing degrees for different conditions, media, contaminants, and plants. A variety of terms have been used in the literature to refer to these various processes. This discussion defines and uses a number of terms as a convenient means of introducing and conceptualizing the processes that occur during phytoremediation. However, it must be realized that the various processes described by these terms all tend to overlap to some degree and occur in varying proportions during phytoremediation. Phytoremediation encompasses a number of different methods that can lead to contaminant degradation, removal (through accumulation or dissipation), or immobilization:

1. Degradation (for destruction or alteration of organic contaminants).

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A. *Rhizodegradation*: enhancement of biodegradation in the below-ground root zone by microorganisms.

B. *Phytodegradation*: contaminant uptake and metabolism above or below ground, within the root, stem, or leaves.

2. Accumulation (for containment or removal of organic and/or metal contaminants).

   A. *Phytoextraction*: contaminant uptake and accumulation for removal.

   B. *Rhizofiltration*: contaminant adsorption on roots for containment and/or removal.

3. Dissipation (for removal of organic and/or inorganic contaminants into the atmosphere).

   A. *Phytovolatilization*: contaminant uptake and volatilization.

4. Immobilization (for containment of organic and/or inorganic contaminants).

   A. *Hydraulic Control*: control of ground-water flow by plant uptake of water.

   B. *Phytostabilization*: contaminant immobilization in the soil.

Vegetated caps, buffer strips, and riparian corridors are applications that combine a variety of these methods for contaminant containment, removal, and/or destruction. The different forms of phytoremediation are discussed individually below. With each phytoremediation method, it is necessary to ensure that unwanted transfer of contaminant to other media does not occur. Phytoremediation is potentially applicable to a variety of contaminants, including some of the most significant contaminants, such as petroleum hydrocarbons, chlorinated solvents, metals, radionuclides, nutrients, pentachlorophenol (PCP), and polycyclic aromatic hydrocarbons (PAHs).

Phytoremediation requires more effort than simply planting vegetation and, with minimal maintenance, assuming that the contaminant will disappear. Phytoremediation requires an understanding of the processes that need to occur, the plants selected, and what needs to be done to ensure plant growth. Given the great number of candidates, a relatively limited number of plants have been investigated. Screening studies will be important in selecting the most useful plants. Extrapolation of results from hydroponic or greenhouse studies to actual field situations will require caution. Further field studies will be necessary. Verification of the applicability and efficacy of phytoremediation is likely to be required on a site-specific basis, at least until the technology becomes firmly proven and established. Phytoremediation requires a commitment of resources and time, but has the potential to provide a lower-cost, environmentally acceptable alternative to conventional remedial technologies at appropriate sites.

Plant Processes

Phytoremediation takes advantage of the natural processes of plants. These processes include water and chemical uptake, metabolism within the plant, exudate release into the soil that leads to contaminant loss, and the physical and biochemical impacts of plant roots.

Growth of plants depends on photosynthesis, in which water and carbon dioxide are converted into carbohydrates and oxygen, using the energy from sunlight. Roots are effective in extracting water held in soil, even water held at relatively high matric and osmotic negative water potentials; extraction is followed by upward transport through the xylem. Transpiration (water vapor loss from plants to the atmosphere) occurs primarily at the stomata (openings in leaves and stems where gas exchange occurs), with additional transpiration at the lenticels (gas exchange sites on stem and root surfaces).

Carbon dioxide uptake from the atmosphere occurs through the stomata, along with release of oxygen. Respiration of the carbohydrates produced during photosynthesis, and production of ATP, necessary for the active transport of nutrients by roots, requires oxygen. Diffusion and advection of oxygen into the soil are necessary for continued plant survival; and a high or saturated soil water content will greatly slow oxygen transport. Plants do not transport oxygen into roots (or into the surrounding water or soil), except for a relatively small number of plants (mostly aquatic, flood-adapted, or wetland plants) using specialized structures or mechanisms such as aerenchyma, lacunae, or pneumatophores. Few woody species can transport oxygen to the root zone; flood tolerance of some trees, such as poplar, is likely due to coping mechanisms other than transport of oxygen.

Plants require macronutrients (N, P, K, Ca, Mg, S) and micronutrients (B, Cl, Cu, Fe, Mn, Mo, Zn and possibly Co, Ni, Se, Si, V, and maybe others). Lack of chlorophyll due to stresses on the plant, such as lack of nutrients, can result in chlorosis (the yellowing of normally green plant leaves). Nutrient uptake pathways can take up contaminants that are similar in chemical form or behavior to the nutrients. Cadmium can be subject to plant uptake due to its similarity to the plant nutrients calcium and zinc, although poplar leaves in a field study did not accumulate significant amounts of cadmium (Pierzynski et al., 1994). Arsenic (as arsenate) might be taken up by plants due to similarities to the plant nutrient phosphate; however, poplars growing in soil containing an average of 1250 mg/kg arsenic did not accumulate significant amounts of arsenic in their leaves (Pierzynski et al., 1994). Selenium replaces the nutrient sulfur in compounds taken up by a plant, but does not serve the same physiological functions (Brooks, 1998b).

For uptake into a plant, a chemical must be in solution, either in ground water or in the soil solution (i.e., the water in the unsaturated soil zone). Water is absorbed from the soil solution into the outer tissue of the root. Contaminants in the water can move through the epidermis to and through the Casparian strip, and then through the endodermis, where they can be sorbed, bound, or metabolized. Chemicals or metabolites passing through the endodermis and reaching the xylem are then transported in the transpiration stream or sap. The compounds might react with or partition into plant tissue, be metabolized, or be released to the atmosphere through stomatal pores (Paterson et al., 1990; Shimp et al., 1993).

The uptake and translocation of organic compounds is dependent on their hydrophobicity (lipophilicity), solubility, polarity, and molecular weight (Briggs et al., 1982; Bell, 1992; Schnoor, 1997). Briggs et al. (1982) found that translocation of non-ionized compounds to shoots was optimum for intermediate polarity compounds that were moderately hydrophobic (with log of the octanol-water partition coefficient, i.e., log k_{ow}, between 1.5 to 2.0), with less translocation for more polar compounds. A slightly wider range of log k_{ow} values (approximately 1.0 to 3.5) was provided by Schnoor (1997) for prediction of translocation to the shoot. More hydrophobic compounds are more strongly bound to root surfaces or partition into root solids, resulting in less translocation within the plant (Briggs et al., 1982; Schnoor

et al., 1995; Cunningham et al., 1997). Very soluble organic compounds (with low sorption) will not be sorbed onto roots as much as lower solubility compounds, or translocated within the plant (Schnoor et al., 1995). In contrast to the very soluble organic compounds, soluble inorganic compounds, such as nutrients, can be readily taken up by plants. Uptake of the inorganic compounds (which are generally in ionic or complexed form) is mediated by active or passive uptake mechanisms within the plant (Brady, 1974), whereas uptake of organic compounds is generally governed by log k_{ow} (hydrophobicity) and polarity. Ryan et al. (1988) provide more discussion of plant uptake of organic compounds.

Plant uptake of organic compounds can also depend on the type of plant, age of the contaminant, and many other physical and chemical characteristics of the soil. One study identified greater than 70 organic chemicals, which represented many classes of compounds, that were taken up and accumulated by 88 species of plants and trees (Paterson et al., 1990). Definitive conclusions cannot always be made about a particular chemical. For example, when PCP was spiked into soil, 21% was found in roots and 15% in shoots after 155 days in the presence of grass (Qiu et al., 1994); in another study, minimal uptake of PCP by several plants was seen (Bellin and O'Connor, 1990).

The breaking up of soil aggregates is a physical effect of root tips pushing through soil as the root tips grow. Roots can form large openings (macropores) in the soil, especially as the roots decay, which can contribute to water, gas, and contaminant transport through the soil and change the aeration and water status of the soil. The increased 'workability' of soil due to the incorporation of organic matter by plants might make the soil conditions more amenable to various types of soil treatment. Plant materials and plant roots can have chemical and biological impacts in the soil. Exudates such as simple phenolics and other organic acids can be released from living cells or from the entire cell contents during root decay. These exudates can change metals speciation (i.e., form of the metal), and the uptake of metal ions and simultaneous release of protons, which acidifies the soil and promotes metal transport and bioavailability (Ernst, 1996). In some cases, the changed metals speciation can lead to increased precipitation of the metals. The organic compounds in the root exudates can stimulate microbial growth in the rhizosphere (the region immediately surrounding plant roots). Fungi associated with some plant roots (i.e., mycorrhizae) can also influence the chemical conditions within the soil. Decaying roots and above-ground plant material that is incorporated into the soil will increase the organic matter content of the soil, potentially leading to increased sorption of contaminants and humification (the incorporation of a compound into organic matter). Contaminant loss may also increase as roots decay, due to release of substrates and the creation of air passages in the soil; increased TPH loss occurred as white clover was dying and the roots were degrading in a field study (AATDF, 1998). Decaying plant material can also have biochemical impacts on the soil; for example, compounds may be released that suppress growth of other plants.

Phytoremediation Processes

There are a number of different forms of phytoremediation, discussed immediately below. Defining these forms is useful to clarify and understand the different processes that can occur due to vegetation, what happens to a contaminant, where the contaminant remediation occurs, and what should be done for effective phytoremediation. The different forms of phytoremediation may apply to specific types of contaminants or contaminated media, and may require different types of plants (the terms 'plant' and 'vegetation' will be used interchangeably to indicate all plant life, whether trees, grasses, shrubs, or other forms).

Phytoextraction

Phytoextraction is contaminant uptake by roots with subsequent accumulation in the aboveground portion of a plant, generally to be followed by harvest and ultimate disposal of the plant biomass. It is a contaminant removal process. Phytoextraction applies to metals (e.g., Ag, Cd, Co, Cr, Cu, Hg, Mn, Mo, Ni, Pb, Zn), metalloids (e.g., As, Se), radionuclides (e.g., ^{90}Sr, ^{137}Cs, ^{234}U, ^{238}U), and non-metals (e.g., B) (Salt et al., 1995; Kumar et al., 1995; Cornish et al., 1995; Bañuelos et al., 1999), as these are generally not further degraded or changed in form within the plant. Phytoextraction has generally not been considered for organic or nutrient contaminants taken up by a plant, as these can be metabolized, changed, or volatilized by the plant, thus preventing accumulation of the contaminant. However, some studies have shown accumulation of unaltered organic contaminants within the aboveground portion of a plant. The target medium is generally soil, although contaminants in sediments and sludges can also undergo phytoextraction. Soluble metals in surface water or extracted ground water could conceivably be cleaned using phytoextraction, perhaps in conjunction with rhizofiltration.

Phytoextraction is also known as phytoaccumulation, phytoabsorption, and phytosequestration (which can all also apply to contaminant accumulation within the roots). Some practitioners define the term phytoremediation to mean extraction of metals by plants; however, as discussed throughout this issue paper, there are many types of phytoremediation, and thus phytoremediation should remain a broad, over-all term. Phytoextraction has also been referred to as phytomining or biomining. A narrower definition of phytomining is the use of plants to obtain an economic return from metals extracted by a plant, whether from contaminated soils or from soils having naturally high concentrations of metals (Brooks, 1998a); this more specialized application will not be discussed here, as the primary goal and motivation for this issue paper is the remediation of hazardous waste sites.

Interest in metal-accumulating plants initially focused on hyperaccumulators, plants that accumulate a metal from metal-rich soil to a much greater degree (such as 100-fold or 1000-fold) than do other plants in that soil, and reach some specified unusually high concentration of metal in some part of the plant. These plants are generally relatively rare and found only in localized areas around the world, with less than four hundred identified species for eight heavy metals (Brooks, 1998a). Heavy metals are generally phytotoxic to plants; however, hyperaccumulators have developed on heavy-metal-rich soils. A possible physiological reason for metals hyperaccumulation could be as a tolerance strategy for these high soil concentrations of metals. Other potential reasons for metals hyperaccumulation include a possible competitive advantage, a means to resist drought, inadvertent metal uptake, or a defense against herbivores or pathogens such as bacteria and fungi (Brooks, 1998a; Boyd, 1998). More research is required to determine the reasons for hyperaccumulation (Boyd, 1998).

Brooks (1998b) discusses the processes involved in hyperaccumulation. It is not clear if a plant's tolerance to one metal will induce tolerance to another metal (Reeves and Brooks, 1983). Some hyperaccumulators of one metal can

hyperaccumulate other metals if present; for example, copper or cobalt hyperaccumulators will hyperaccumulate both (Brooks, 1998c). Other hyperaccumulators will take up only a specific metal even if others are present.

Plant roots generally contain higher metal concentrations than the shoots despite the translocation mechanisms. An upper limit to the metal concentration within the root can occur. Root uptake of lead by hydroponically-grown plants reached a maximum concentration and did not increase further as the lead concentration of the solution increased (Kumar et al., 1995). Metals are generally unevenly distributed throughout a plant, although in hyperaccumulators the metal content of the leaves is often greater than other portions of the plant; for example, the greatest proportion of nickel in *Alyssum heldreichii* was found in the leaves (Brooks, 1998b). Cadmium and zinc were found in both roots and shoots, although the shoots had higher concentrations of zinc (Brooks, 1998b). High concentrations of zinc were found in small hemispherical bodies located on the surface of some leaves of *Thlaspi caerulescens* (Brooks, 1998b).

Phytoextraction occurs in the root zone of plants. The root zone may typically be relatively shallow, with the bulk of roots at shallower rather than deeper depths. This can be a limitation of phytoextraction. Remediation of lead-contaminated soil using *Brassica juncea* was limited to the top 15 cm, with insignificant lead removal from 15 to 45 cm (Blaylock et al., 1999).

Due to the scarcity, small biomass, slow growth rate, uncertain or specialized growing conditions of many hyperaccumulators, or lack of hyperaccumulators for some of the most serious contaminants, such as chromium, the effectiveness of hyperaccumulators for phytoextraction has been uncertain, especially if they can remove only a relatively small mass of metals from the soil. Solutions to this uncertainty include increased screening of hyperaccumulator candidate plants, plant breeding, genetic development of better hyperaccumulators, genetic transfer of hyperaccumulating abilities to higher-biomass plants, fertilization strategies that increase the biomass of hyperaccumulators, or use of faster-growing, greater biomass metal-accumulating plants that are not hyperaccumulators. Metals can be taken up by other plants that do not accumulate the high concentrations of hyperaccumulators, for example, corn (*Zea mays*), sorghum (*Sorghum bicolor*), alfalfa (*Medicago sativa* L.), and willow trees (*Salix* spp.). The greater biomass of these plants could result in a greater mass of metals being removed from the soil even though the concentrations within the plants might be lower than in hyperaccumulators, since the metal concentration in the plant multiplied by the biomass determines the amount of metal removal. McGrath (1998) points out, however, that the much higher metals concentrations achievable in hyperaccumulators more than compensate for their lower biomass. The suitability of hyperaccumulators as compared to non-hyperaccumulators will need to be resolved through further research and field trials of phytoextraction.

Metals are taken up to different degrees. In one greenhouse study, phytoextraction coefficients (the ratio of the metal concentration in the shoot to the metal concentration in the soil) for different metals taken up by Indian mustard (*Brassica juncea* (L.) Czern) were 58 for Cr(VI), 52 for Cd(II), 31 for Ni(II), 17 for Zn(II), 7 for Cu(II), 1.7 for Pb(II), and 0.1 for Cr(III), with the higher phytoextraction coefficients indicating greater uptake (Kumar et al., 1995). The effectiveness of phytoextraction can be limited by the sorption of metals to soil particles and the low solubility of the metals; however, the metals can be solubilized by addition of chelating agents to allow uptake of the contaminant by the plant. The chelating agent EDTA was used in a growth chamber study to solubilize lead to achieve relatively high lead concentrations in Indian mustard (Blaylock et al., 1997) and EDTA and HBED solubilized lead for uptake by corn under greenhouse conditions (Wu et al., 1999). Potential adverse impacts of chelating agent addition, such as high water solubility leading to negative impacts on ground water, or impacts on plant growth, have to be considered (Wu et al., 1999). In addition, increased uptake might be specific for one metal, such as lead, while decreasing uptake of other metals; for example, addition of citric acid or EDTA decreased uptake of nickel in the hyperaccumulator *Berkheya coddii* (Robinson et al., 1997).

Some research with hyperaccumulating plants has achieved high levels of metal uptake when using plants grown in hydroponic solution. Extrapolation of the results of hydroponic studies to phytoextraction of metals from soils could be misleading, even using the same plants, due to the much greater bioavailability of metals in the hydroponic solution as compared to metals in soil. Such research indicates that uptake is possible, and identifies appropriate plant species, rather than providing estimates of the actual concentrations. Phytoextraction coefficients under field conditions are likely to be less than those determined in the laboratory (Kumar et al., 1995).

A small-scale field test application of phytoextraction was successfully conducted at the "Magic Marker" site in Trenton, NJ, by a commercial phytoremediation firm (Phytotech, Inc., which was acquired by Edenspace Systems Corporation in 1999) under the Superfund Innovative Technology Evaluation (SITE) program. Lead was removed from soil using three crops of Indian mustard in one growing season, with a decrease in soil concentrations of lead to acceptable levels (Blaylock et al., 1999).

Phytoextraction of organic contaminants is not as straightforward as for metals, in that transformations of the contaminants within the plant are more likely to occur. Ashing of metal-contaminated biomass and recovery of the metals may raise less concerns than would the combustion of plant biomass containing organic contaminants, due to potential concerns over incomplete destruction of the organics and release of contaminants in the off-gases and particulate matter. Phytoaccumulation of organic contaminants has occurred. The explosive hexahydro-1,3,5-trinitro-1,3,5-triazine (RDX) was found to have accumulated in an unaltered form in the leaves of hybrid poplar, after uptake from a hydroponic solution (Thompson et al., 1999). This was viewed as a potential impediment to other forms of phytoremediation of RDX, such as rhizodegradation and phytodegradation, rather than as an application of phytoextraction. Accumulation of RDX in the poplar leaves could potentially result in food chain contamination.

Phytostabilization

Phytostabilization is the use of vegetation to contain soil contaminants in situ, through modification of the chemical, biological, and physical conditions in the soil. Contaminant transport in soil, sediments, or sludges can be reduced through absorption and accumulation by roots; adsorption onto roots; precipitation, complexation, or metal valence reduction in soil within the root zone; or binding into humic (organic) matter through the process of humification. In addition, vegetation can reduce wind and water erosion of the soil, thus preventing dispersal of the contaminant in runoff or fugitive dust emissions, and may reduce or prevent leachate generation. Phytostabilization is also known as in-place inactivation or

phytoimmobilization. Phytostabilization research to date has generally focused on metals contamination, with lead, chromium, and mercury being identified as the top potential candidates for phytostabilization (U.S. EPA, 1997). However, there may be potential for phytostabilization of organic contaminants, since some organic contaminants or metabolic byproducts of these contaminants can be attached to or incorporated into plant components such as lignin (Harms and Langebartels, 1986). This form of phytostabilization has been called phytolignification (Cunningham et al., 1995). One difference, however, is that phytostabilization of metals is generally intended to occur in the soil, whereas phytostabilization of organic contaminants through phytolignification can occur aboveground.

Metals within the root zone can be stabilized by changing from a soluble to an insoluble oxidation state, through root-mediated precipitation. For example, roots can mediate the precipitation of lead as insoluble lead phosphate (Salt et al., 1995). Stabilization of metals also includes the non-biological process of surface sorption, due to chelation, ion exchange, and specific adsorption (Salt et al., 1995). Lead, which is generally toxic to plants, is usually not accumulated in plants under natural conditions, possibly due to precipitation of lead as sulfate at the plant roots (Reeves and Brooks, 1983). Soil pH can be changed by the production of CO_2 by microbes degrading the plant root exudates, possibly changing metal solubility and mobility or impacting the dissociation of organic compounds. Effective phytostabilization requires a thorough understanding of the chemistry of the root zone, root exudates, contaminants, and fertilizers or soil amendments, to prevent unintended effects that might increase contaminant solubility and leaching. Cunningham et al. (1995) indicate that phytostabilization might be most appropriate for heavy-textured soils and soils with high organic matter contents.

A form of phytostabilization may occur in water into which plant roots release plant exudates such as phosphate. Insoluble precipitated forms of contaminants may occur, such as lead phosphate, thus removing the contaminant from solution without having it taken up into the plant. The formation of a lead phosphate precipitate in a hydroponic solution was identified by Dushenkov et al. (1995).

Advantages of phytostabilization are that soil removal is unnecessary, disposal of hazardous materials or biomass is not required, the cost and degree of disruption to site activities may be less than with other more vigorous soil remedial technologies, and ecosystem restoration is enhanced by the vegetation.

Disadvantages of phytostabilization include the necessity for long-term maintenance of the vegetation or verification that the vegetation will be self-sustaining. This is necessary since the contaminants remain in place and future re-release of the contaminants and leaching must be prevented. A plant system that produces an irreversible stabilization process is preferred, but must be verified. If not, phytostabilization might have to be considered an interim containment measure. Plant uptake of metals and translocation to the aboveground portion should be avoided, to prevent the transfer of metals to the food chain.

Phytostabilization requires a plant that is able to grow in the contaminated soil (i.e., metal-tolerant plants for heavy-metal contaminated soils), with roots growing into the zone of contamination, and that is able to alter the biological, chemical, or physical conditions in the soil. In a field study, mine wastes containing copper, lead, and zinc were stabilized by grasses (*Agrostis tenuis* cv. Goginan for acid lead and zinc mine wastes, *Agrostis tenuis* cv. Parys for copper mine wastes, and *Festuca rubra* cv. Merlin for calcareous lead and zinc mine wastes) (Smith and Bradshaw, 1979). Indian mustard appeared to have potential for effective phytostabilization. In a laboratory study, leachate from sand planted with seedlings of the Indian mustard contained 22 μg/mL lead, compared to 740 g/mL lead from sand without plants (Salt et al., 1995). A laboratory rhizofiltration study indicated that Indian mustard roots apparently reduced Cr(VI) to Cr(III) (Dushenkov et al., 1995); this process occurring in soil would promote phytostabilization.

Some hazardous waste sites are former mining or mining-waste sites that can have large areal expanses of contaminated and severely degraded soil. Saline-affected soils can also cover large areas. Reclamation and revegetation of these soils will reduce wind and water erosion and subsequent dispersal of contaminated soil, as well as promote restoration of the local ecosystem. Phytostabilization is the primary strategy to be used at these sites, but if appropriate for the contaminant, extractive phytoremediation methods, such as phytoextraction, could be used. The use of phytoextraction, however, raises concerns about transfer of the contaminants to the broader ecosystem; thus, it should not be used unless the biomass containing accumulated metals is removed for disposal. Reclamation and revegetation of mining-impacted and saline soils has been researched for many years, well before the concept of phytoremediation was applied to hazardous waste sites, so a large body of literature and experience exists for these conditions.

Plant re-establishment at these waste sites may be difficult for reasons such as phytotoxicity of the contaminant, the physical condition of the soil, adverse pH, arid climate, or lack of organic matter. Metal-tolerant, non-accumulator plants are appropriate, as they would tolerate, but not accumulate high levels of metals. Hyperaccumulator plants generally would not be used due to their slow growth rate and propensity to accumulate metals.

Stabilizing covers of native metal-tolerant grasses were successfully established on metalliferous mine wastes in the United Kingdom, and grew vigorously during a nine-year investigation (Smith and Bradshaw, 1979). Hybrid poplars in experimental plots at the Whitewood Creek Superfund site, South Dakota, grew to 12 m by the end of the first growing season, and established dense root masses. Analysis of leaves, stems, and roots for arsenic and cadmium indicated that laboratory studies had overestimated the amount of uptake (Pierzynski et al., 1994). Revegetation was proposed for the Galena Superfund site in southeastern Kansas as a phytostabilization strategy that would decrease wind erosion of contaminated soil. Experimental studies using native and tame grasses and leguminous forbs, including big bluestem (*Andropogon gerardi* Vit.) and tall fescue (*Festuca arundinacea* Schreb.), revealed the importance of mycorrhizae and adding organic waste amendments in establishing plants on the metal-contaminated mine wastes at the Galena site (Pierzynski et al., 1994). Investigations have also been conducted using metal-tolerant plants to examine the feasibility of phytostabilizing large areas of cadmium- and zinc-contaminated soils at a Superfund site in Palmerton, Pennsylvania. The IINERT (In-Place Inactivation and Natural Ecological Restoration Technologies) Soil-Metals Action team under the Remediation Technologies Development Forum (RTDF) program has also investigated the use of plants to physically stabilize metal-contaminated soil in order to decrease off-site movement of contaminants.

Rhizofiltration

Rhizofiltration (also known as phytofiltration) is the removal by plant roots of contaminants in surface water, waste water, or extracted ground water, through adsorption or precipitation onto

the roots, or absorption into the roots. The root environment or root exudates may produce biogeochemical conditions that result in precipitation of contaminants onto the roots or into the water body. The contaminant may remain on the root, within the root, or be taken up and translocated into other portions of the plant, depending on the contaminant, its concentration, and the plant species.

Rhizofiltration and phytoextraction are similar in that they each result in accumulation of the contaminant in or on the plant. However, in rhizofiltration this accumulation can occur in the roots or in the portion of the plant above water, whereas for effective phytoextraction the accumulation occurs aboveground, not in the roots. In addition, rhizofiltration differs from phytoextraction in that the contaminant is initially in water, rather than in soil.

Rhizofiltration is a contaminant removal process, in which contaminant removal from the site is accomplished by harvesting the roots and, if necessary, the above-water portion of the plant, followed by proper disposal of the contaminated plant mass. Thus, rhizofiltration differs from phytostabilization occurring in soil, in which the contaminant remains in the root zone.

Rhizofiltration is generally applicable to treating large volumes of water with low contaminant concentrations (in the ppb range). It has primarily been applied to metals (Pb, Cd, Cu, Fe, Ni, Mn, Zn, Cr(VI) (Dushenkov et al., 1995; Wang et al., 1996; Salt et al., 1997)) and radionuclides (^{90}Sr, ^{137}Cs, ^{238}U, ^{236}U (Dushenkov et al., 1997)).

Either aquatic or terrestrial plants can be used. Given a support platform to enable growth on water, terrestrial plants offer the advantage of greater biomass and longer, faster-growing root systems than aquatic plants (Dushenkov et al., 1995). The use of seedlings has been proposed in place of mature plants since seedlings can take up metals but do not require light or nutrients for germination and growth for up to two weeks (Salt et al., 1997).

Rhizofiltration can be conducted in situ to remediate contaminated surface water bodies, or ex situ, in which an engineered system of tanks can be used to hold the introduced contaminated water and the plants. Either of these systems will require an understanding of the contaminant speciation and interactions of all contaminants and nutrients. Monitoring and possible modification of the water pH, or of the flow rate and contaminant concentration of influent water, may be necessary. Predictions of metal immobilization and uptake from laboratory studies and greenhouse studies might not be achievable in the field. However, in an engineered ex-situ system, the ability to control conditions may allow results to approach those predicted in the laboratory. Effluent from engineered flow-through rhizofiltration systems will need to meet relevant discharge limits. Proper disposal of the contaminated plant biomass will be required.

Applications of rhizofiltration are currently at the pilot-scale stage. Phytotech tested a pilot-scale rhizofiltration system in a greenhouse at a Department of Energy uranium-processing facility in Ashtabula, Ohio (Dushenkov et al., 1997). This engineered ex-situ system used sunflowers to remove uranium from contaminated ground water and/or process water. Phytotech also conducted a small-scale field test of rhizofiltration to remove radionuclides from a small pond near the Chernobyl reactor, Ukraine. Sunflowers were grown for four to eight weeks in a floating raft on a pond, and bioaccumulation results indicated that sunflowers could remove ^{137}Cs and ^{90}Sr from the pond.

Rhizodegradation

Rhizodegradation is the enhancement of naturally-occurring biodegradation in soil through the influence of plant roots, and ideally will lead to destruction or detoxification of an organic contaminant. Other terms have been used by some authors as synonyms for rhizodegradation, such as enhanced rhizosphere biodegradation.

Organic contaminants in soil can often be broken down into daughter products or completely mineralized to inorganic products such as carbon dioxide and water by naturally occurring bacteria, fungi, and actinomycetes. The presence of plant roots will often increase the size and variety of microbial populations in the soil surrounding roots (the rhizosphere) or in mycorrhizae (associations of fungi and plant roots). Significantly higher populations of total heterotrophs, denitrifiers, pseudomonads, BTX (benzene, toluene, xylenes) degraders, and atrazine degraders were found in rhizosphere soil around hybrid poplar trees in a field plot (*Populus deltoides* × *nigra* DN-34, Imperial Carolina) than in non-rhizosphere soil (Jordahl et al., 1997). The increased microbial populations are due to stimulation by plant exudates, compounds produced by plants and released from plant roots. Plant exudates include sugars, amino acids, organic acids, fatty acids, sterols, growth factors, nucleotides, flavanones, enzymes, and other compounds (Shimp et al., 1993). The increased microbial populations and activity in the rhizosphere can result in increased contaminant biodegradation in the soil, and degradation of the exudates can stimulate cometabolism of contaminants in the rhizosphere. Rhizodegradation occurs primarily in soil, although stimulation of microbial activity in the root zone of aquatic plants could potentially occur.

Stimulation of soil microbes by plant root exudates can also result in alteration of the geochemical conditions in the soil, such as pH, which may result in changes in the transport of inorganic contaminants. Plants and plant roots can also affect the water content, water and nutrient transport, aeration, structure, temperature, pH, or other parameters in the soil, often creating more favorable environments for soil microorganisms, regardless of the production of exudates. This effect has not been addressed in most phytoremediation research. One laboratory study did raise the possibility that transpiration due to alfalfa plants drew methane from a saturated methanogenic zone up into the vadose zone where the methane was used by methanotrophs that cometabolically degraded trichloroethylene (TCE) (Narayanan et al., 1995). Lin and Mendelssohn (1998) indicate that the salt marsh grasses *Spartina alterniflora* and *S. patens* could potentially increase subsurface aerobic biodegradation of spilled oil by transporting oxygen to their roots.

Appealing features of rhizodegradation include destruction of the contaminant in situ, the potential complete mineralization of organic contaminants, and that translocation of the compound to the plant or atmosphere is less likely than with other phytoremediation technologies since degradation occurs at the source of the contamination. Harvesting of the vegetation is not necessary since there is contaminant degradation within the soil, rather than contaminant accumulation within the plant. Root penetration throughout the soil may allow a significant percentage of the soil to be contacted. However, at a given time only a small percentage of the total soil volume is in contact with living roots. It can take a long time for root dieback and root growth into new areas of the soil for contact with most of the soil to occur. Also, inhospitable soil conditions or areas of high contaminant

concentrations can decrease root penetration, leading to some portions of the soil never being contacted by roots.

Perhaps the most serious impediment to successful rhizodegradation is its limitation to the depth of the root zone. Many plants have relatively shallow root zones, and the depth of root penetration can also be limited by soil moisture conditions or by soil structures such as hard pans or clay pans that are impenetrable by roots. However, in some cases roots may extend relatively deep (e.g., 110 cm) and extend into soil with high contaminant concentrations (Olson and Fletcher, 2000). Other potential impediments to successful rhizodegradation include the often substantial time that may be required to develop an extensive root zone. The rhizosphere extends only about 1 mm from the root and initially the volume of soil occupied by roots is a small fraction of the total soil volume; thus, the soil volume initially affected by the rhizosphere is limited. However, with time, new roots penetrate more of the soil volume and other roots decompose. This root turnover adds exudates to the rhizosphere (Olson and Fletcher, 1999). Uptake of the contaminant by the plant is an undesirable trait in rhizodegradation, and plants must be selected to avoid uptake, unless it is shown that phytodegradation also occurs within the plant. Stimulation of rhizosphere organisms does not always lead to increased contaminant degradation, as populations of microorganisms that are not degraders might be increased at the expense of degraders. Competition between the plants and the microorganisms can also impact the amount of biodegradation. In addition, organic matter from the vegetation might be used as a carbon source instead of the contaminant, which would decrease the amount of contaminant biodegradation (Molina et al., 1995).

In some studies, rhizodegradation has increased the initial rate of degradation compared to a non-rhizosphere soil, but the final extent or degree of degradation was similar in both rhizosphere and non-rhizosphere soil. That the rhizosphere has a significant beneficial effect on biodegradation under most conditions has not been conclusively proven, although a forensic phytoremediation field investigation provided evidence that contaminant loss did occur in the root zone (Olson and Fletcher, 2000). The effectiveness of rhizodegradation may be site-specific and not universal. The chances for successful rhizodegradation can be enhanced in several ways. A useful preliminary step is the screening of plants for root exudates that have been experimentally determined to be effective in stimulating contaminant cometabolism (Fletcher and Hegde, 1995). Seeds can be inoculated with bacteria that are capable of degrading the contaminant (Pfender, 1996).

A wide range of organic contaminants are candidates for rhizodegradation, such as petroleum hydrocarbons, PAHs, pesticides, chlorinated solvents, PCP, polychlorinated biphenyls (PCBs), and surfactants. Higher populations of benzene-, toluene-, and *o*-xylene-degrading bacteria were found in soil from the rhizosphere of poplar trees than in non-rhizosphere soil, although it was not clear that the populations were truly statistically different. Root exudates contained readily biodegradable organic macromolecules (Jordahl et al., 1997). Schwab and Banks (1999) investigated total petroleum hydrocarbon (TPH) disappearance at several field sites contaminated with crude oil, diesel fuel, or petroleum refinery wastes, at initial petroleum hydrocarbon contents of 1,700 to 16,000 mg/kg TPH. Plant growth varied by species, but the presence of some species led to greater TPH disappearance than with other species or in unvegetated soil. At the crude oil-contaminated field site near the Gulf of Mexico, an annual rye-soybean rotation plot and a St. Augustine grass-cowpea rotation plot had significantly ($P < 0.05$) greater TPH disappearance than did sorghum-sudan grass or unvegetated plots, at 21 months. At the diesel fuel-contaminated Craney Island field site in Norfolk, Virginia, the fescue plot had significantly ($P < 0.10$) greater TPH disappearance than did an unvegetated plot. At the refinery waste site, statistical analyses were not presented due to the short time since establishment of the plots, but Schwab and Banks (1999) reported that qualitatively, the vegetated plots had greater TPH disappearance than the unvegetated plots.

For PAHs, a greater disappearance in vegetated soil than in non-vegetated soil was found for 10 mg/kg of chrysene, benz(a)anthracene, benzo(a)pyrene, and dibenz(a,h)anthracene (Aprill and Sims, 1990). This laboratory study used a mix of prairie grasses: big bluestem (*Andropogon gerardi*), little bluestem (*Schizachyrium scoparius*), indiangrass (*Sorghastrum nutans*), switchgrass (*Panicum virgatum*), Canada wild rye (*Elymus canadensis*), western wheatgrass (*Agropyron smithii*), side oats grama (*Bouteloua curtipendula*), and blue grama (*Bouteloua gracilis*). In a greenhouse study, statistically greater loss of fluoranthene, pyrene, and chrysene occurred in soil planted with perennial ryegrass (*Lolium perenne*) than in unplanted soil (Ferro et al., 1999). Fescue, a cool-season grass; sudangrass (*Sorghum vulgare* L.) and switchgrass, warm-season grasses; and alfalfa, a legume, were used in a greenhouse study of the disappearance of 100 mg/kg anthracene and pyrene; greater disappearance was seen in the vegetated soils than in unvegetated soils (Reilley et al., 1996).

Pesticide biodegradation has been found to be influenced by plants. *Kochia* species (sp.) rhizosphere soil increased the degradation of herbicides (0.3 μg/g trifluralin, 0.5 μg/g atrazine, and 9.6 μg/g metolachlor) relative to non-rhizosphere soil. These laboratory experiments used rhizosphere soil but were conducted in the absence of plants to minimize any effects of root uptake (Anderson et al., 1994). In a laboratory study, bush bean (*Phaseolus vulgaris* cv. "Tender Green") rhizosphere soil had higher mineralization rates for 5 μg/g of the organophosphate insecticides parathion and diazinon than non-rhizosphere soil. Diazinon mineralization in soil without roots did not increase when an exudate solution was added, but parathion mineralization did increase (Hsu and Bartha, 1979). A greenhouse study indicated that rice (*Oryza sativa* L.) rhizosphere soil with 3 μg/g propanil herbicide had increased numbers of Gram-negative bacteria that could rapidly transform the propanil. It was hypothesized that the best propanil degraders would benefit from the proximity to plant roots and exudates (Hoagland et al., 1994). Microorganisms capable of degrading 2,4-dichlorophenoxyacetic acid (2,4-D) occurred in elevated numbers in the rhizosphere of sugar cane, compared to non-rhizosphere soil (Sandmann and Loos, 1984). The rate constants for 2,4-D and 2,4,5-trichlorophenoxyacetic acid (2,4,5-T) herbicide biodegradation in a laboratory evaluation were higher in field-collected rhizosphere soil than in non-rhizosphere soil (Boyle and Shann, 1995).

Chlorinated solvents may be subject to rhizodegradation. In a growth chamber study, TCE mineralization was increased in soil planted with a legume (*Lespedeza cuneata* (Dumont)), Loblolly pine (*Pinus taeda* (L.)), and soybean (*Glycine max* (L.) Merr., cv. Davis), compared to non-vegetated soil (Anderson and Walton, 1995). In another laboratory study, the presence of alfalfa possibly contributed to the dissipation of 100 and 200 μL/L TCE and 50 and 100 μL/L 1,1,1-trichloroethane (TCA) in ground water, through the effect of root exudates on soil bacteria

(Narayanan et al., 1995). Newman et al. (1999) did not find any rhizodegradation of TCE in a two-week long laboratory experiment using hybrid poplars; however, they could not conclusively rule out the occurrence of microbial degradation of TCE in the soil.

Other contaminants are also candidates for rhizodegradation, as indicated by a variety of greenhouse, laboratory, and growth chamber studies. Mineralization rates of 100 mg/kg PCP were greater in soil planted with Hycrest crested wheatgrass than in unplanted controls (Ferro et al., 1994). Proso millet (*Panicum miliaceum* L.) seeds treated with a PCP-degrading bacterium germinated and grew well in soil containing 175 mg/L PCP, compared to untreated seeds (Pfender, 1996). Compounds (such as flavonoids and coumarins) found in leachate from roots of specific plants stimulated the growth of PCB-degrading bacteria (Donnelly et al., 1994; Gilbert and Crowley, 1997). Spearmint (*Mentha spicata*) extracts contained a compound that induced cometabolism of a PCB (Gilbert and Crowley, 1997). Red mulberry (*Morus rubra* L.), crabapple (*Malus fusca* (Raf.) Schneid), and osage orange (*Maclura pomifera* (Raf.) Schneid) produced exudates with relatively high levels of phenolic compounds, at concentrations capable of supporting growth of PCB-degrading bacteria (Fletcher and Hegde, 1995). A variety of ectomycorrhizal fungi, which grow symbiotically with the roots of a host plant, metabolized various congenors of PCBs (Donnelly and Fletcher, 1995). The surfactants linear alkylbenzene sulfonate (LAS) and linear alcohol ethoxylate (LAE) at 1 mg/L had greater mineralization rates in the presence of cattail (*Typha latifolia*) root microorganisms than in non-rhizosphere sediments (Federle and Schwab, 1989).

Phytodegradation

Phytodegradation is the uptake, metabolizing, and degradation of contaminants within the plant, or the degradation of contaminants in the soil, sediments, sludges, ground water, or surface water by enzymes produced and released by the plant. Phytodegradation is not dependent on microorganisms associated with the rhizosphere. Contaminants subject to phytodegradation include organic compounds such as munitions, chlorinated solvents, herbicides, and insecticides, and inorganic nutrients. Phytodegradation is also known as phytotransformation, and is a contaminant destruction process.

For phytodegradation to occur within the plant, the plant must be able to take up the compound. Uptake of contaminants requires that they have a moderate log k_{ow}, and laboratory experiments at the University of Washington indicated that short chain halogenated aliphatic compounds could be taken up by plants (Newman et al., 1998). Plants can metabolize a variety of organic compounds, including TCE (Newman et al., 1997), trinitrotoluene (TNT) (Thompson et al., 1998), and the herbicide atrazine (Burken and Schnoor, 1997). Partial metabolism by wheat and soybean plant cell cultures was found for a variety of compounds, including 2,4-dichlorophenoxyacetic acid (2,4-D); 2,4,5-trichlorophenoxyacetic acid (2,4,5-T), 4-chloroaniline; 3,4-dichloroaniline; PCP; diethylhexylphthalate (DEHP); perylene; benzo(a)pyrene; hexachlorobenzene; DDT; and PCBs (Sandermann et al., 1984; Harms and Langebartels, 1986; and Wilken et al., 1995). In phytodegradation applications, transformation of a contaminant within the plant to a more toxic form, with subsequent release to the atmosphere through transpiration, is undesirable. The formation and release of vinyl chloride resulting from the uptake and phytodegradation of TCE has been a concern. However, although low levels of TCE metabolites have been found in plant tissue (Newman et al., 1997), vinyl chloride has not been reported.

Plant-produced enzymes that metabolize contaminants may be released into the rhizosphere, where they can remain active in contaminant transformation. Plant-formed enzymes have been discovered in plant sediments and soils. These enzymes include dehalogenase, nitroreductase, peroxidase, laccase, and nitrilase (Schnoor et al., 1995). These enzymes are associated with transformations of chlorinated compounds, munitions, phenols, the oxidative step in munitions, and herbicides, respectively. In one week, the dissolved TNT concentrations in flooded soil decreased from 128 ppm to 10 ppm in the presence of the aquatic plant parrot feather (*Myriophyllum aquaticum*), which produces nitroreductase enzyme that can partially degrade TNT (Schnoor et al., 1995). The nitroreductase enzyme has also been identified in a variety of algae, aquatic plants, and trees (Schnoor et al., 1995). Hybrid poplar trees metabolized TNT to 4-amino-2,6-dinitrotoluene (4-ADNT), 2-amino-4,6-dinitrotoluene (2-ADNT), and other unidentified compounds in laboratory hydroponic and soil experiments (Thompson et al., 1998).

Uptake and degradation of TCE has been confirmed in poplar cell cultures and in hybrid poplars. About one to two percent of applied TCE was completely mineralized to carbon dioxide by cell cultures (Newman et al., 1997). After exposure to ground water containing about 50 ppm TCE, unaltered TCE was present in the stems of hybrid poplars (Newman et al., 1997). In addition to unaltered TCE, TCE metabolites were detected in the aboveground portion of hybrid poplars exposed to TCE in ground water in a controlled field experiment. These metabolites included trichloroethanol, trichloroacetic acid, and dichloroacetic acid, as well as reductive dechlorination products, but vinyl chloride was not reported (Newman et al., 1999).

Laboratory studies have demonstrated the metabolism of methyl tertiary-butyl ether (MTBE) by poplar cell cultures, and provided some indication of MTBE uptake by eucalyptus trees (Newman et al., 1998).

Atrazine degradation has occurred in hybrid poplars (*Populus deltoides* × *nigra* DN34, Imperial Carolina). Atrazine in soil was taken up by trees and then hydrolyzed and dealkylated within the roots, stems, and leaves. Metabolites were identified within the plant tissue, and a review of atrazine metabolite toxicity studies indicated that the metabolites were less toxic than atrazine (Burken and Schnoor, 1997).

The herbicide bentazon was degraded within black willow (*Salix nigra*) trees, as indicated by loss during a nursery study and by identification of metabolites within the tree. Bentazon was phytotoxic to six tree species at concentrations of 1000 and 2000 mg/L, but allowed growth at 150 mg/L. At this concentration, bentazon metabolites were detected within tree trunk and canopy tissue samples. Black willow, yellow poplar (*Liriodendron tulipifera*), bald cypress (*Taxodium distichum*), river birch (*Betula nigra*), cherry bark oak (*Quercus falcata*), and live oak (*Quercus viginiana*) were all able to support some degradation of bentazon (Conger and Portier, 1997).

Deep-rooted poplars have also been used to remove nutrients from ground water. Nitrate can be taken up by plants and incorporated into proteins or other nitrogen-containing compounds, or transformed into nitrogen gas (Licht and Schnoor, 1993). Deep-rooting techniques can increase the effective depth of this application.

Plant-derived materials have been used in waste water treatment. Waste water contaminated with chlorinated phenolic compounds was treated in ex-situ reactors using oxidoreductase enzymes derived from horseradish roots, and minced horseradish roots

successfully treated wastewater containing up to 850 ppm of 2,4-dichlorophenol (Dec and Bollag, 1994). Application of phytoremediation, however, has more typically focused on using the whole, living plant.

Research and pilot-scale field demonstration studies of phytodegradation have been conducted for a number of sites, primarily Army Ammunition Plants (AAPs) contaminated with munitions waste, including the Iowa AAP, Volunteer AAP, and Milan AAP. At the Milan AAP, emergent aquatic plants in a field demonstration decreased TNT concentrations from over 4,000 ppb to the remedial goal of less than 2 ppb, except during the winter months (ESTCP, 1999). Phytodegradation of munitions is part of the remedy in the Record of Decision (ROD) for the Iowa AAP.

Phytovolatilization

Phytovolatilization is the uptake of a contaminant by a plant, and the subsequent release of a volatile contaminant, a volatile degradation product of a contaminant, or a volatile form of an initially non-volatile contaminant. For effective phytoremediation, the degradation product or modified volatile form should be less toxic than the initial contaminant. Phytovolatilization is primarily a contaminant removal process, transferring the contaminant from the original medium (ground water or soil water) to the atmosphere. However, metabolic processes within the plant might alter the form of the contaminant, and in some cases transform it to less toxic forms. Examples include the reduction of highly toxic mercury species to less toxic elemental mercury, or transformation of toxic selenium (as selenate) to the less toxic dimethyl selenide gas (Adler, 1996). In some cases, contaminant transfer to the atmosphere allows much more effective or rapid natural degradation processes to occur, such as photodegradation. Because phytovolatilization involves transfer of contaminants to the atmosphere, a risk analysis of the impact of this transfer on the ecosystem and on human health may be necessary.

Phytovolatilization can occur with soluble inorganic contaminants in ground water, soil, sediment, or sludges. In laboratory experiments, tobacco (*Nicotiana tabacum*) and a small model plant (*Arabidopsis thaliana*) that had been genetically modified to include a gene for mercuric reductase converted ionic mercury (Hg(II)) to the less toxic metallic mercury (Hg(0)) and volatilized it (Meagher et al., 2000). Similarly transformed yellow poplar (*Liriodendron tulipifera*) plantlets had resistance to, and grew well in, normally toxic concentrations of ionic mercury. The transformed plantlets volatilized about ten times more elemental mercury than did untransformed plantlets (Rugh et al., 1998). Indian mustard and canola (*Brassica napus*) may be effective for phytovolatilization of selenium, and, in addition, accumulate the selenium (Bañuelos et al., 1997).

Phytovolatilization can also occur with organic contaminants, such as TCE, generally in conjunction with other phytoremediation processes. Over a three year-period, test cells containing hybrid poplar trees exposed under field conditions to 50 ppm TCE in ground water lost from 98% to 99% of the TCE from the water, compared to about 33% TCE lost in an unplanted test cell (Newman et al., 1999). Of this amount of TCE loss, a companion study indicated that about 5% to 7% of added TCE was mineralized in the soil. Uptake of TCE by the trees occurred, with unaltered TCE being found within the trees. Oxidation of TCE also occurred within the trees, indicated by the presence of TCE oxidative metabolites. Analysis of entrapped air in bags placed around leaves indicated that about 9% of the applied TCE was transpired from the trees during the second year of growth, but no TCE was detected during the third year (Newman et al., 1999).

It is not clear to what degree phytovolatilization of TCE occurs under different conditions and with different plants, since some other studies have not detected transpiration of TCE. However, measurement of transpired TCE can be difficult, and measurements must differentiate between volatilization from the plant and volatilization from the soil. In addition, it is almost certain that several phytoremediation processes (rhizodegradation, phytodegradation, and phytovolatilization) occur concurrently in varying proportions, depending on the site conditions and on the plant. Questions remain as to chlorinated solvent metabolism within plants and transpiration from the plants.

In a study (Burken and Schnoor, 1998, 1999) of poplar cuttings in hydroponic solution, about 20% of the benzene and TCE in the initial solution was volatilized from the leaves, with little remaining within the plant. About 10% of toluene, ethylbenzene, and *m*-xylene was volatilized. There was little volatilization of nitrobenzene and no volatilization of 1,2,4-trichlorobenzene, aniline, phenol, pentachlorophenol, or atrazine. The percentage of applied compound taken up into the plant was 17.3% for 1,2,4-trichlorobenzene, 40.5% for aniline, 20.0% for phenol, 29.0% for pentachlorophenol, and 53.3% for atrazine. For 1,2,4-trichlorobenzene, aniline, phenol, and pentachlorophenol, the largest percentage of compound taken up was found in the bottom stem, as opposed to the root, upper stem, or leaves. For atrazine, the largest percentage of compound taken up was found in the leaves. Of the eleven compounds tested, nine had 2.4% or less of the applied compound in the leaves, but aniline had 11.4% and atrazine had 33.6% in the leaves. All compounds had 3.8% or less in the upper stem (Burken and Schnoor, 1998, 1999). However, the chemical fate and translocation is most likely concentration-dependent, and other concentrations may give different results.

Hydraulic Control

Hydraulic control (or hydraulic plume control) is the use of vegetation to influence the movement of ground water and soil water, through the uptake and consumption of large volumes of water. Hydraulic control may influence and potentially contain movement of a ground-water plume, reduce or prevent infiltration and leaching, and induce upward flow of water from the water table through the vadose zone. Other phytoremediation processes, such as rhizodegradation, phytodegradation, and phytovolatilization, may occur as the contaminated water is brought to and into the plant. In some cases and under certain conditions, vegetative hydraulic control may be used in place of, or to supplement, an engineered pump-and-treat system. Root penetration throughout the soil can help counteract the slow flow of water in low-conductivity soils.

Vegetation water uptake and transpiration rates are important for hydraulic control and remediation of ground water. Water uptake and the transpiration rate depend on the species, age, mass, size, leaf surface area, and growth stage of the vegetation. They also are affected by climatic factors, such as temperature, precipitation, humidity, insolation, and wind velocity, and will vary seasonally. Deciduous trees will be dormant for part of the year, resulting in lowered transpiration and water uptake rates. Thus, well-defined typical rates are difficult to provide for a given type of vegetation. For this reason, design and operation of phytoremediation hydraulic control will likely require site-specific

observations of water levels, flow patterns, and water uptake rates. Some estimates of water uptake rates indicate the possible magnitude: 100 to 200 L/day for a five-year old poplar tree (Newman et al., 1997); 5000 gal/day transpired by a single willow tree, comparable to the transpiration rate of 0.6 acre of alfalfa (Gatliff, 1994); between 50 and 350 gal/day per tree for individual 40-foot tall cottonwood trees in southwestern Ohio, based on analysis of drawdown near the trees (Gatliff, 1994); and approximately 5 to 13 gal/day for four-year-old hybrid poplars (Hinckley et al., 1994). A phreatophyte is a plant or tree, such as tamarisk and eucalyptus, that is deep-rooted and that can draw a large amount of water from a deep water table. Phreatophytes may be desirable for hydraulic control of ground water, especially from deeper zones.

Cottonwood and hybrid poplar trees were used at seven sites in the eastern and midwestern United States to contain and treat shallow ground water contaminated with heavy metals, nutrients, or pesticides. At one site, poplar trees were combined with an engineered pump-and-treat system to control a contaminated ground-water plume (Gatliff, 1994). At least five U.S. companies are active in installing phytoremediation systems that incorporate hydraulic control.

Vegetated Caps

A vegetated cap (or cover) is a long-term, self-sustaining cap of plants growing in and/or over contaminated materials, designed to minimize exposure pathways and risk. The primary purpose of the vegetation is to provide hydraulic control and prevent or minimize infiltration of precipitation and snowmelt into the contaminated subsurface, thus preventing or minimizing leachate formation. This is done by maximizing evapotranspiration and maximizing the storage capacity of the soil. A cap designed for this purpose is called an evapotranspiration cap or water-balance cover. The vegetation can also increase stability of the soil, thus preventing erosion, and could potentially destroy or remove contaminants through rhizodegradation, phytodegradation, or phytovolatilization. A cap designed to incorporate contaminant destruction or removal in addition to the prevention of infiltration is called a phytoremediation cap. A vegetated cap can be constructed over landfills, or over contaminated soil or ground water. Long-term maintenance of the cap might be required, or the cap vegetation may be designed to allow an appropriate plant succession that will maintain the cap integrity.

Significant issues remain with the use of vegetative caps on landfills for evapotranspirative control or for contaminant destruction. These include the equivalency to standard, regulatory-approved landfill covers; the potential for contaminant uptake; the possibility of plant roots breaching the cap integrity; and the generation of gas in landfills.

Plants for evapotranspiration covers should have relatively shallow root depths so that the cap is not breached; however, trees with weak root systems should be avoided as they may topple in high winds and jeopardize the integrity of the cap. In cases where prevention of infiltration is not a concern, a phytoremediation cover may use deeper-rooted plants to allow penetration of the roots into the underlying waste. Plants for evapotranspiration covers should also be capable of evapotranspiring the desired amount of water. Poplar trees and grasses have been used commercially to construct vegetative covers over landfills. The soils used in a vegetative cover should also be carefully selected. Soils with a high capacity to store water are desired, and soils with rapid drainage are to be avoided. In humid areas, there might be inadequate evapotranspiration on a seasonal basis, and soil layers will need to be thicker than in arid regions.

Buffer Strips and Riparian Corridors

Buffer strips are areas of vegetation placed downgradient of a contaminant source or plume, or along a waterway (i.e., riparian corridor). The vegetation contains, extracts, and/or destroys contaminants in soil, surface water, and ground water passing underneath the buffer through hydraulic control, phytodegradation, phytostabilization, rhizodegradation, phytovolatilization, and perhaps phytoextraction. The use of buffer strips might be limited to easily assimilated and metabolized compounds. Relatively soluble contaminants, such as nutrients and some organics (especially pesticides), have been addressed using buffer strips and riparian corridors. Agricultural runoff has been a target of buffer strips and riparian corridors. Additional benefits of riparian corridors are the stabilization of stream banks and prevention of soil erosion, and the improvement of aquatic and terrestrial habitats. To be remediated, ground water must be within the depth of influence of the roots. Sufficient land must be available for the establishment of the vegetation. Monitoring is likely to be required to ensure that contaminant removal has occurred. Poplars have been used successfully in riparian corridors and buffer strips to remove nitrate (Licht, 1990). Laboratory and field experiments have indicated that soil planted with poplars can degrade atrazine (CO_2 production presumably indicated mineralization in the root zone) and slow migration of volatile organics (Licht and Schnoor, 1993; Nair et al., 1993). Commercial installation of buffer strips and riparian corridors has been successfully accomplished. Correll (1999) provides an extensive annotated and indexed bibliography on vegetated riparian zones.

Constructed Wetlands

Constructed wetlands or treatment wetlands are artificial wetlands that are used for treating organic, inorganic, and nutrient contaminants in contaminated surface water, municipal waste water, domestic sewage, refinery effluents, acid mine drainage, or landfill leachate. A considerable amount of research and applied work has been conducted using constructed wetlands for these applications. Cole (1998) provides an overview of constructed wetlands, and more detailed discussions are provided in Kadlec and Knight (1996). Natural wetlands have also been examined for treatment of these wastes. Ground-water treatment is less common, though conceivable. Except in a few cases, constructed wetlands generally have not been used in remediation of hazardous waste sites; however, constructed and natural wetlands have been investigated for the phytodegradation of munitions-contaminated water. In the future, constructed wetlands might become an option for treatment of water extracted from hazardous waste sites, using rhizofiltration and phytodegradation. Integration of hazardous waste site phytoremediation and constructed wetland technologies might increase in the future.

Combinations of Phytoremediation Processes

At a phytoremediation site, combinations of the phytoremediation processes discussed above may occur simultaneously or in sequence for a particular contaminant, or different processes may act on different contaminants or at different exposure concentrations. For example, TCE in soil can be subject to biodegradation in the root zone (rhizodegradation) and metabolism within the plant (phytodegradation), with loss of some contaminant or metabolite through volatilization from the

plant (phytovolatilization). Some metals or radionuclides in water can be accumulated on or within roots (rhizofiltration) while other metals or radionuclides are simultaneously taken up into the aerial portion of the plant (phytoextraction).

Forensic Phytoremediation

Some undisturbed contaminated sites, such as inactive land treatment units, will naturally revegetate. Vegetation may become established after the phytotoxic contamination has been reduced through naturally-occurring biodegradation, abiotic processes such as volatilization, or through intentional traditional remedial technologies. In these cases, the vegetation would indicate that the contaminants are no longer bioavailable or toxic to the established plant species. Alternatively, a plant that can withstand the contaminant might preferentially become established, and perhaps then contribute to additional contaminant loss through the phytoremediation processes discussed above. This has apparently happened at a petroleum refinery waste sludge impoundment, in which mulberry trees became established through natural revegetation on sludges containing PAHs (Olson and Fletcher, 2000). Root exudates from mulberry trees were found to be good substrates for microbial degradation of recalcitrant compounds such as PAHs (Hegde and Fletcher, 1996). Examination of naturally-revegetated sites has been termed forensic phytoremediation, in which the beneficial effects of the vegetation, reasons for the vegetation re-establishment, contaminant loss mechanisms, and prediction of future impacts have to be deduced after the vegetation has appeared. Forensic phytoremediation investigations seek to verify and quantify naturally-occurring phytoremediation at a contaminated site. The study cited above is the only in-depth forensic phytoremediation study to date, although naturally-revegetated sites have been examined in a number of studies in an attempt to identify potentially useful plant species. Natural revegetation of a site that leads to contaminant attenuation could be considered a form of or enhancement of natural attenuation. As phytoremediation is likely to be a lengthy process due to the relatively slow growth of vegetation, naturally-revegetated sites are useful because they provide the equivalent of a long-established research plot.

Environmental Monitoring and Bioassays (Phytoinvestigation)

Bioassays using plants have been used routinely in the environmental sciences. In some cases, the effectiveness of bioremediation efforts at hazardous waste sites has been assessed using plant bioassays. Phytotoxicity testing was used to determine the extent of bioremediation of a contaminated soil (Baud-Grasset et al., 1993). A plant assay was used on site to test for levels of arsenic, chromium, and copper in soil (Sandhu et al., 1991). In other cases, potential impacts on the environment have been investigated or monitored using plants, such as in assessing the uptake of metals from land-applied sludges. Air pollution has been monitored by analysis of plant tissues and of particulates deposited on leaves.

In geobotany, the presence of a particular plant species such as a hyperaccumulator can be indicative of an underlying ore body. In biogeochemistry, the change in metals concentrations within a particular plant species, over a wide area, can also indicate a host rock for an ore body. Analysis of previously collected herbarium specimens can also lead to identification of areas that could contain ore bodies (Brooks, 1998d).

The presence of different species of plants also provides clues as to the presence and depth of ground water (Meinzer, 1927). Ground-water plume movement has been investigated through analysis of tree samples. Tree ring data indicated the direction and velocity of a chloride plume in ground water near a landfill (Vroblesky and Yanosky, 1990) and were correlated with nickel concentrations in a ground-water plume near a landfill and stainless steel plant (Yanosky and Vroblesky, 1992). Failure in portions of a phytoremediation project might provide information about the contaminated soil or ground water, as unhealthy or dying vegetation might indicate previously undetected hot spots of higher contamination.

Applicable Media

Ground Water

For selected site conditions, contaminants in ground water may be addressed using phytodegradation, phytovolatilization, hydraulic control, vegetative caps, constructed wetlands, riparian corridors, and buffer strips. Extracted ground water may be treated using rhizofiltration, or in some cases, used as irrigation water that then undergoes rhizodegradation and phytodegradation.

The primary considerations for ground-water contamination are the depth to the ground water and the depth to the contaminated zone. In-situ ground-water phytoremediation is essentially limited to unconfined aquifers in which the water table depths are within the reach of plant roots and to a zone of contamination in the uppermost portion of the water table that is accessible to the plant roots. Plant roots will not grow through clean ground water to a deeper contaminated zone. If in-situ remediation of deeper contaminated water is desired, modeling may be useful to determine if the water table can be lowered by the plants or through pumping, or if ground water movement can be induced towards the roots. However, modeling may be hindered by the uncertainty and seasonality of water uptake rates by plants. Careful field measurements and conservative estimates of water uptake will be necessary, and modeling results should be confirmed by observations of the water table. Deep ground water that is beyond the reach of plant roots could be remediated by phytoremediation after the water is pumped from the subsurface using extraction wells, and then applied to a phytoremediation treatment system. For ground-water containment, the rate of ground-water flow into the phytoremediation area should be matched by the rate of water uptake by the plants to prevent migration past the vegetation.

Surface Water and Waste Water

Surface water can be treated using rhizofiltration or phytodegradation, in ponds, engineered tanks, natural wetlands, or constructed wetlands. In some cases, the contaminated water can be used as irrigation water in which the contaminants then undergo rhizodegradation and phytodegradation.

Soil, Sediment, and Sludge

Contaminated soil, sediment, or sludge can be treated using phytoextraction, phytostabilization, rhizodegradation, phytodegradation, and phytovolatilization, or through vegetative cap applications. Phytoremediation is most appropriate for large areas of a relatively thin surface layer of contaminated soil, within the root depth of the selected plant. Deeper soil contamination, high contaminant concentrations, or small soil volumes might be more effectively treated using conventional technologies, although through future phytoremediation research, the capabilities of phytoremediation might be increased. Soil characteristics, such as texture and water content (degree of saturation), should be conducive to plant growth.

Air

Phytoremediation research and application have focused on contaminated solid or liquid media. There has been little discussion of phytoremediation of contaminated air or soil gas, and no such application of phytoremediation. However, airborne contaminants can be directly withdrawn from the atmosphere through uptake of gaseous contaminants by plant leaves or by deposition of contaminated particulate matter onto the leaves. Some plants appear to remove volatile compounds from air, in addition to removing the contaminants through the action of roots and soil microbes. In one study, potted mums removed 61% of formaldehyde, 53% of benzene, and 41% of TCE (Raloff, 1989). A critical review paper on phytoremediation cited a study indicating that planted soil was a sink for benzene vapor in air, with subsequent soil biodegradation of benzene; the rate of benzene depletion for the planted soil was twice as great as for unplanted soil (Shimp et al., 1993). Contaminated air has been remediated by drawing the air through soil beds in which microbial activity helps to degrade the contaminants. It is conceivable that this application of biodegradation could be enhanced by the presence of the root zone of plants. Phytoremediation of contaminated air and soil gas may become a subject for future research.

Applicable Contaminants

Inorganic contaminants amenable to phytoremediation include metals (Table 1) and metalloids, non-metals, radionuclides, and nutrients (Table 2). Organic contaminants amenable to phytoremediation include petroleum hydrocarbons, chlorinated solvents, pesticides, munitions, wood-preserving wastes, surfactants, and some others (Table 3). These tables list contaminants, phytoremediation processes, and media, along with some examples of contaminant concentrations and plants that have been investigated for phytoremediation. Research and application of phytoremediation has provided a large body of knowledge which cannot be given in these summary tables and which can be obtained from the phytoremediation literature. Some additional sources include the *International Journal of Phytoremediation*, for all contaminants; INEEL (2000) and Terry and Bañuelos (2000) for inorganic contaminants; and Frick et al. (1999) for petroleum hydrocarbon contamination.

Phytoremediation may be limited by high contaminant concentrations, as these concentrations are likely to be phytotoxic or could cause an unacceptable decrease in plant growth. Areas of higher, phytotoxic contaminant concentrations may have to be treated using other technologies, or excavated and landfilled, with phytoremediation being used for the lower contaminant concentration areas of a site. Phytoremediation (such as rhizodegradation) may be suited for a "polishing" or final step, for example, if active land treatment bioremediation has ended without having achieved a desired low contaminant concentration. Future long-term field studies with additional plant species may indicate that there are fewer limitations than currently thought.

The contaminant concentrations that are phytotoxic to specific plants are likely to be site-specific, and affected by soil, climate, and bioavailability. Aged compounds in soil can be much less bioavailable. This will decrease phytotoxicity, but can also decrease the effectiveness of phytoremediation. Site-specific phytotoxicity or treatability studies should use contaminated soil from the site rather than uncontaminated soil spiked with the contaminant. Phytotoxic concentration levels will need to be determined on a site-specific basis, although literature values can provide a first approximation. Information on concentrations from one site or from a laboratory study may not be applicable to another site with different soil and geochemical conditions.

Robinson et al. (1997) found that nickel content in a hyperaccumulator plant was correlated with the ammonium acetate-extractable nickel concentration of the soil. This suggests that the potential for successful hyperaccumulation of a metal from a soil might best be predicted by the soil concentration given by a specific extraction that reflects bioavailable metal, rather than by the total metal content of the soil.

The presence of dense non-aqueous phase liquids (DNAPLs) or light non-aqueous phase liquids (LNAPLs) will adversely affect plant growth due to the relatively high contaminant concentrations resulting from the NAPL and the physical impact of the NAPL fluid which interferes with oxygen and water transfer. The pH of a contaminated medium can also affect plant growth by changing the bioavailability of nutrients or toxic compounds. Mixtures of different contaminants might not be effectively treated using one plant or individual phytoremediation method. The use of several plants, or a treatment train approach with other remedial technologies, might be required. When applying the results of laboratory studies that examined contaminants individually, synergistic or antagonistic effects need to considered when treating mixtures of wastes. For example, the phytoremediation behavior of a plant (i.e., uptake of metals) may be different for mixtures of metals than for one metal alone (Ebbs et al., 1997).

Vegetation

Root morphology and depth are important plant characteristics for phytoremediation. A fibrous root system, such as found in grasses (e.g., fescue), has numerous fine roots spread throughout the soil and will provide maximum contact with the soil due to the high surface area of the roots. A tap root system (such as in alfalfa) is dominated by one larger central root. Many hyperaccumulators, such as *Thlaspi caerulescens*, have a tap root system, which limits root contact to relatively small volumes of soil (Ernst, 1996).

Root depth directly impacts the depth of soil that can be remediated or depth of ground water that can be influenced, as close contact is needed between the root and the contaminant or water. The fibrous root systems of some prairie grasses can extend to about 6 to 10 feet. Alfalfa roots can potentially reach quite deep, down to about 30 feet. However, these values represent maximum depths that are not likely to occur in most cases. The effective depth for phytoremediation using most non-woody plant species is likely to be only one or two feet. Most metal accumulators have root zones limited to the top foot of soil, which restricts the use of phytoextraction to shallow soils. The effective depth of tree roots is likely to be in the few tens of feet or less, with one optimistic estimate that trees will be useful for extraction of ground water up to 30 feet deep (Gatliff, 1994). Ground water from depths below the root zone can be pumped to the surface using extraction wells and then applied to a phytoremediation system.

Root depth can be manipulated to some degree during planting by placement of a root ball at a desired depth or by using planting tubes, or during growth, by restricting water infiltration, thus forcing roots to extend deeper to obtain water. Root depth varies greatly among different types of plants, and can also vary significantly for one species depending on local conditions such as depth to water, soil water content, soil structure, depth of a hard pan, soil fertility, cropping pressure, contaminant concentration, or other conditions. The bulk of root mass will be found at shallower depths, with much less root mass at deeper

depths. For example, measurement of the distribution of roots of three-year old oak trees (*Quercus phellos* L.) in the top 30 cm of soil indicated that 90% of the root density was found in the top 20 cm (Katul et al., 1997). Another survey of tree root systems indicated that most roots were in the first one or two meters (Dobson and Moffat, 1995). Also, deeper roots will provide a very small proportion of the water needed by the plant, except in cases of drought.

A large root mass and large biomass may be advantageous for various forms of phytoremediation, for example, to allow a greater mass of metals accumulation, greater transpiration of water, greater assimilation and metabolism of contaminants, or production of a greater amount of exudates and enzymes. However, there may be characteristics of a plant, such as the types of exudates produced by the roots, that are more important to phytoremediation effectiveness than biomass. Screening studies could help identify such characteristics. If a large biomass is important, a fast growth rate could potentially decrease the time required for remediation. Literature values for growth rates and biomass production may be from studies in which vegetation was grown under normal agricultural practices (i.e., in uncontaminated soil) and thus may not reflect the lower values that are likely to occur under stressed conditions in contaminated soils.

The different forms of phytoremediation require different general plant characteristics for optimum effectiveness. In rhizofiltration and phytostabilization, these are the ability to remove metals, no translocation of metals from the roots to the shoots, and rapidly growing roots. For phytoextraction, the plant should tolerate, translocate, and accumulate high concentrations of heavy metals in the shoots and leaves, and have a rapid growth rate and high biomass production. For rhizodegradation, a plant should release appropriate enzymes and other substances that enhance biodegradation, not take up the contaminant, and have the appropriate depth, rate, and extent of root growth and decay. Phytodegradation requires a plant that can take up and metabolize the contaminant, without producing toxic degradation products. For phytovolatilization, the plant must be able to take up and transform the contaminant to a less toxic volatile form.

Phytoremediation research studies have examined numerous plants, but interest has focused on a smaller group for reasons such as widespread distribution, ready availability, ease of growth, an existing large knowledge base, or even the plant's commodity value. Terrestrial plants are more likely to be effective for phytoremediation than aquatic plants due to their larger root systems. Poplar (or hybrid poplar) and cottonwood trees, such as the Eastern cottonwood (*Populus deltoides*), are fast-growing trees (some can grow more than 3 m/year (Newman et al., 1997)) with a wide geographic distribution that have the ability to take up or degrade contaminants. Indian mustard is a relatively high biomass and fast-growing accumulator plant which has the ability to take up and accumulate metals and radionuclides. Sunflower (*Helianthus annuus*) can accumulate metals and has about the same biomass as Indian mustard. Examples of metal hyperaccumulators that have been investigated include *Thlaspi caerulescens* (Alpine pennycress), but which is slow-growing and has a low biomass; *Thlaspi rotundifolium* spp. *cepaeifolium*, the only known hyperaccumulator of Pb (Brooks, 1998e); and other *Thlaspi* species that can hyperaccumulate cadmium, nickel, or zinc (Brooks, 1998c). Grasses have been investigated for rhizodegradation and phytostabilization due to their widespread growth and their extensive root systems. Examples include ryegrass, prairie grasses, and fescues. Some grasses, such as *Festuca ovina*, can take up metals but are not hyperaccumulators (Ernst, 1996). Alfalfa, a legume, has been investigated due to its deep root system, its ability to fix nitrogen, and a large knowledge base about this plant. Although these plants are some that have been popular for research to date, future screening studies will undoubtedly add many more candidates, some of which may prove to be much more effective for phytoremediation.

Aquatic plants such as the floating plants water hyacinth (*Eichhornia crassipes*), pennywort (*Hydrocotyle umbellata*), duckweed (*Lemna minor*), and water velvet (*Azolla pinnata*) (Salt et al., 1995) have been investigated for use in rhizofiltration, phytodegradation, and phytoextraction. These plants have been used in water treatment, but are smaller and have smaller, slower-growing root systems than terrestrial plants (Dushenkov et al., 1995). Based on metals content and degree of bioaccumulation, Zayed et al. (1998) found that duckweed could be an effective phytoremediator of cadmium, selenium, and copper in waste water, and Zhu et al. (1999) found that water hyacinth was a promising candidate for phytoremediation of cadmium, chromium, copper, and selenium. Other aquatic plants that have been investigated include parrot feather, *Phragmites* reeds, and cattails.

The uptake of metals into plants was investigated during research in the 1970s and 1980s on land application of sludges and wastes, in order to study the potential impacts on consumers (human or animal) of the plants grown on sludge-amended land (Chaney, 1983). Information on potentially useful plants, as well as on their cultural requirements, may be found in the literature resulting from this research (U.S. EPA, 1983).

Careful selection of the plant and plant variety is critical, first, to ensure that the plant is appropriate for the climatic and soil conditions at the site, and second, for effectiveness of the phytoremediation. Plant species that are long-term competitors and survivors under adverse changing conditions will have an advantage. Depending on the climatic and soil conditions, the plant may need resistance to or tolerance of disease, heat, cold, insects, drought, chemicals, and stress. In some cases, salt-resistant plants (halophytes), such as salt cedar, might be necessary in cases of saline soils or ground water. The use of phreatophytes can enhance hydraulic control of ground water. Other considerations in plant selection include the use of annuals or perennials, the use of a monoculture or several plant species, and the use of deciduous trees. The seeds or plants (or variety of the plant) should be from, or adapted to, the climate of the phytoremediation site. Viable seeds and disease-free plants are important in establishing the vegetation. There should be no transport, import, quarantine, or use restrictions. A sufficient quantity of plants or seeds should be available when needed.

Variability in phytoremediation efficacy in varieties, cultivars, or genotypes of a given species has been encountered in alfalfa for hydrocarbon rhizodegradation (Wiltse et al., 1998), *Brassica juncea* for metals uptake (Kumar et al., 1995), and possibly in poplars. Biomass and zinc content varied significantly between different populations of *Thlaspi caerulescens* (Brooks, 1998b) and cadmium, copper, and zinc uptake varied widely among willow clones (Greger and Landberg, 1999). The type, amount, and effectiveness of exudates and enzymes produced by a plant's root will vary between species and even within subspecies or varieties of one species. A screening of phytotoxicity and effectiveness of cultivars/varieties might be required on a site-specific basis as an initial step in plant selection.

Genetic engineering of plants has the potential to increase the effectiveness and use of phytoremediation, as plants can be genetically modified using specific bacterial, fungal, animal, or plant genes that are known to have useful properties for contaminant uptake, degradation, or transformation. Stomp et al. (1994) discuss the potential benefits of genetic engineering for phytoremediation, with some examples of what genetically-engineered plants can achieve. Numerous examples of promising research into genetic engineering for phytoremediation are given by Gleba et al. (1999). Genetically-modified canola and tobacco were able to survive concentrations of Hg(II) that killed non-modified control plants, and the tobacco converted the toxic Hg(II) to the less toxic metallic mercury and volatilized it (Meagher et al., 2000). These results were also seen with genetically-modified yellow poplar plantlets (Rugh et al., 1998). Genetic engineering of tobacco seedlings to express a bacterial nitroreductase increased their tolerance ten-fold to TNT and nitroglycerine, and apparently doubled the rate of nitroglycerine degradation by the seedlings (Meagher, 2000). Transgenic tobacco plants containing mammalian cytochrome P450 2E1 had higher concentrations of a metabolite of TCE in the plant tissue than did transgenic control plants without P450 2E1, indicating increased transformation of TCE within the plant (Doty et al., 2000). A similar experiment indicated an apparent increase in dehalogenation of ethylene dibromide by plants containing P450 2E1 (Doty et al., 2000). The use of appropriate genes could increase the accumulation of toxic metals by faster-growing, higher-biomass plants, and bacterial genes that enhance PCB biodegradation could assist in degradation of PCBs by plants (Meagher, 2000). In conjunction with research on genetically-engineered plants for phytoremediation, however, regulatory and public concerns will have to be addressed for this relatively new technology.

Cost Information

When research of phytoremediation began, initial cost estimates predicted that phytoremediation would have lower costs than other remedial technologies. Actual cost data for phytoremediation technologies are sparse, and currently are from pilot-scale or experimental studies that may not accurately reflect expected costs once the technology matures. Phytoremediation costs will include preliminary treatability studies to select the proper plant and to assess its effectiveness; soil preparation; planting; maintenance such as irrigation and fertilization; monitoring, which may include plant nutrient status, plant contaminant concentrations, as well as soil or water concentrations, and air monitoring in the case of phytovolatilization; and disposal of contaminated biomass.

Estimated costs for an actual field-scale research study of rhizodegradation of petroleum hydrocarbons in soil were $240/yd^3 or $160/ton. The costs for a full-scale system were estimated to be significantly lower, at $20/yd^3 or $13/ton, due to economy of scale and lack of research-oriented expenses (AATDF, 1998). Based on a small-scale field application of lead phytoextraction, predicted costs for removal of lead from surface soils using phytoextraction were 50% to 75% of traditional remedial technology costs (Blaylock et al., 1999). The cost for phytoremediation of 60-cm deep lead-contaminated soil was estimated at $6/m^2 (in 1996 dollars), compared to the range of about $15/m^2 for a soil cap to $730/m^2 for excavation, stabilization, and off-site disposal (Berti and Cunningham, 1997). Cost estimates made for remediation of a hypothetical case of a 20 in.-thick layer of cadmium-, zinc-, and cesium-137-contaminated sediments from a 1.2 acre chemical waste disposal pond indicated that phytoextraction would cost about one third the amount of soil washing (Cornish et al., 1995). Costs were estimated to be $60,000 to $100,000 using phytoextraction for remediation of one acre of 20 in.-thick sandy loam compared to a minimum of $400,000 for just excavation and storage of this soil (Salt et al., 1995).

The estimated cost for removal of explosives contamination (TNT, RDX, HMX) from ground water using aquatic plants in a full-scale gravel-based system was $1.78 per thousand gallons (ESTCP, 1999). The estimated cost of removing radionuclides from water with sunflowers in a rhizofiltration system was $2.00 to $6.00 per thousand gallons (Cooney, 1996). For phytostabilization, cropping system costs have been estimated at $200 to $10,000 per hectare, equivalent to $0.02 to $1.00 per cubic meter of soil, assuming a one-meter root depth (Cunningham et al., 1995). Estimated costs for hydraulic control and remediation of an unspecified contaminant in a 20-foot deep aquifer at a one-acre site were $660,000 for conventional pump-and-treat, and $250,000 for phytoremediation using trees (Gatliff, 1994). Cost estimates have been presented that indicate a very substantial savings for an evapotranspiration cap compared to excavation, a RCRA Subtitle C cap, or a RCRA Subtitle D cap (RTDF, 1998).

Recovery of some remedial costs through the sale of recovered metals when using phytoextraction has been proposed; however, it might be difficult to find a processor and market for the metal-contaminated plant material. Similarly, recovery of costs by selling a commodity type of vegetation, such as alfalfa, lumber, or other wood products, could be difficult due to potential concerns about contaminant residues in the crop. Confirmation that the vegetation is uncontaminated may be required. In one case, however, a contaminant in one geographic location may be a desired nutrient in another location. Biomass that contains selenium (an essential nutrient) potentially could be transported from areas with excessive selenium to areas that are deficient in selenium and used for animal feed (Bañuelos et al., 1997). Cost recovery, and the appropriateness of including it as a plant selection criterion, is an issue that will likely have to wait until greater experience has been gained in phytoremediation, and its application becomes more accepted and widespread.

Advantages

(1) Early estimates of the costs of phytoremediation indicated a substantial savings over the cost of traditional technologies. As actual cost data are developed during pilot-scale studies, it appears that phytoremediation will be a lower-cost technology, although actual costs of routine application of phytoremediation are still unclear.

(2) Phytoremediation has been perceived to be a more environmentally-friendly “green” and low-tech alternative to more active and intrusive remedial methods. As such, public acceptance could be greater.

(3) Phytoremediation can be applied in situ to remediate shallow soil and ground water, and can be used in surface water bodies.

(4) Phytoremediation does not have the destructive impact on soil fertility and structure that some more vigorous conventional technologies may have, such as acid extraction and soil washing (Greger and Landberg, 1999). Instead, the presence of plants is likely to improve the overall condition of the soil, regardless of the degree of contaminant reduction.

(5) Vegetation can also reduce or prevent erosion and fugitive dust emissions.

Disadvantages

(1) A significant disadvantage of phytoremediation is the depth limitation due to the generally shallow distribution of plant roots. Effective phytoremediation of soil or water generally requires that the contaminants be within the zone of influence of the plant roots. Selection of deep-rooted plants and the use of techniques to induce deep rooting could help alleviate this disadvantage.

(2) A longer time period is likely to be required for phytoremediation, as this technology is dependent on plant growth rates for establishment of an extensive root system or significant above-ground biomass. For example, in one estimate the low growth rate and biomass of hyperaccumulators meant that remediation of metals could not be achieved within even 10 to 20 years (Ernst, 1996). Another estimate was that a heavy-metal-contaminated site would require 13 to 14 years to be remediated, based on a field trial using *Thlaspi caerulescens* (Salt et al., 1995). Strategies to address this potential difficulty include the selection of faster-growing plants than hyperaccumulators, and the harvesting of the vegetation several times a year. A field demonstration of lead phytoextraction had three harvests of Indian mustard in one growing season to achieve acceptable levels of lead in the soil (Blaylock et al., 1999). However, a long time for remediation may still occur with a high biomass plant; a period of 12 years was calculated for removal of 0.6 mg/kg of cadmium, based on realistic willow tree biomass production rates and experimentally-determined cadmium uptake rates (Greger and Landberg, 1999).

A need for rapid attainment of remedial goals or imminent re-use of the land could eliminate some forms of phytoremediation (such as phytoextraction and rhizodegradation) as an alternative. However, other forms of phytoremediation, for other media, might occur at faster rates, such as rhizofiltration for cleaning up contaminated water.

(3) Plant matter that is contaminated will require either proper disposal or an analysis of risk pathways. Harvesting and proper disposal is required for plant biomass that accumulates heavy metals or radionuclides in phytoextraction and rhizofiltration, and may be necessary for other forms of phytoremediation if contaminants accumulate within the plant. The biomass may be subject to regulatory requirements for handling and disposal, and an appropriate disposal facility will need to be identified. For example, sunflower plants that extracted ^{137}Cs and ^{90}Sr from surface water were disposed of as radioactive waste (Adler, 1996). The growth of plant matter represents an addition of mass to a contaminated site, since 94% to 99.5% of fresh plant tissue is made up of carbon, hydrogen, and oxygen (Brady, 1974) which come from offsite and the atmosphere. Should the phytoremediation effort fail, an increased mass of material will need to be remediated.

(4) A phytoremediation system can lose its effectiveness during winter (when plant growth slows or stops) or when damage occurs to the vegetation from weather, disease, or pests. A back-up remedial technology might be necessary.

(5) As with all remedial technologies, in some cases, there may be uncertainty about attainment of remedial goals, such as meeting concentration goals in soil or ground water, or in achieving hydraulic containment. Bench-scale or pilot-scale tests to assess attainment might not be possible in some cases if rapid remediation is desired, due to the potential relatively long periods of time for some forms of phytoremediation.

(6) High initial contaminant concentrations can be phytotoxic, and prevent plant growth. Preliminary phytotoxicity studies are likely to be necessary to screen candidate plants.

(7) There are a number of potential adverse effects. The plant species should be selected, in part, to minimize these potential problems, or managed to prevent such problems.

(a) Introduction or spread of an inappropriate or invasive plant species (e.g., tamarisk or saltcedar) should be avoided. Noxious or invasive vegetation can negatively impact the local ecosystem by escaping the phytoremediation site and then outcompeting and eliminating local species. This could negatively impact animals and other plants in the ecosystem. Other potential problems caused by inappropriate plant species include allergy-causing pollens; odors from vegetation decay or at certain growth stages; plant debris such as fallen leaves or released seeds; hazards arising from the plant itself (such as adverse human health effects of parts of the plant); attraction of pests; or root damage to underground utilities or foundations.

(b) Potential transfer of contaminants to another medium, the environment, and/or the food chain should be prevented, especially if there is transformation of the contaminant into a more toxic, mobile, or bioavailable form. Bioconcentration of toxic contaminants in plants and ingestion of those contaminants by ecosystem consumers is a concern. However, pathogen or herbivore predation on metal-accumulating plants might not occur or might be reduced due to the presence of the metal (Boyd, 1998). Phytovolatilization does involve release of the contaminant to the atmosphere. In this case, it should be confirmed that an adequate destruction mechanism, such as photodegradation in the atmosphere, will occur. A risk analysis might be required in cases such as elemental mercury volatilization, to ensure that the degree of risk is lessened through the use of phytovolatilization.

(c) Potential adverse impacts on surrounding areas include drift of sprayed pesticides and hybridization of certain plant species.

(8) Plant species or varieties of one species can vary significantly in their efficacy for phytoremediation. There can be a wide range in their response to a contaminant and concentration of that contaminant, in their uptake or metabolism of the contaminant, or in their ability to grow under specific soil and climatic conditions. Due to these factors, phytoremediation may not be an “off-the-shelf” technology; rather, site-specific studies may always be necessary prior to implementation.

(9) Cultivation of vegetation often requires great care due to stresses of climate and pests; under the adverse conditions

of contaminated soil or ground water, successful cultivation can be much more difficult. Additions to or modifications of normal agronomic practices might be required, and may have to be determined through greenhouse or pilot-scale tests. However, stressing of vegetation might have beneficial impacts as this may increase root exudate production.

(10) Phytoremediation might require use of a greater land area than other remedial methods. This might interfere with other remediation or site activities.

(11) Amendments and cultivation practices might have unintended consequences on contaminant mobility. For example, application of many common ammonium-containing fertilizers can lower the soil pH, which might result in increased metal mobility and leaching of metals to ground water. Potential effects of soil amendments should be understood before their use.

Design Considerations

Phytoremediation is one more potential technology to be applied to remediating a hazardous waste site. If initially proposed for a site, the phytoremediation alternative will need to be compared to other remedial technologies to determine which best suits the remedial goals. It is possible that other technologies will be able to remediate the site more effectively, that several technologies will be used, or that phytoremediation can fit into a treatment train.

Successful vegetation growth depends strongly on the proper climatic conditions. The correct amount and timing of precipitation, sunlight, shade, and wind, and the proper air temperature and growing season length are necessary to ensure growth. Local conditions and the suitability of the selected plant for these conditions should always be assessed. Sources of knowledge of these local conditions, such as agricultural extension agents, can be very beneficial.

Soil amendments such as compost, manure, or fertilizers generally benefit vegetation growth when added to soil before or after planting. However, potential adverse effects of their addition must be considered before they are added. These include the mobilization of contaminants through changes in soil chemistry, immobilization of contaminants through sorption onto organic matter or through humification, changes in microbial populations, or reduction of phytoremediation efficiency through competitive uptake of nutrients rather than contaminants.

Vegetation growth should be optimized through monitoring and maintenance of proper soil or water pH, nutrient levels, and soil water content. Weeds and plant diseases can be controlled through cultural practices such as tilling or pesticide application, and by removal of diseased plant matter. Pests (insects, birds, or herbivores) can be controlled through the use of pesticides, netting, fencing, or traps. This can also help prevent undesirable transfer of contaminants to the food chain.

Monitoring Considerations

The primary monitoring requirement and measure of remedial effectiveness is likely to be the contaminant concentration in the contaminated media. Due to the role of plant roots in phytoremediation, the location of the roots will be important in planning sample collection and in assessing sampling results. Sampling and analysis of plant tissues may be necessary to measure accumulation of contaminants within the plant and formation of metabolites. Information on methods for sampling and analyzing plant matter and transpiration gases will need to become available to phytoremediation practitioners and to laboratories, and development of new analytical methods may be necessary. Involvement of agricultural and botanical scientists will be crucial in this effort. Their knowledge will also be important in maintaining and optimizing the plant system, as climatic, seasonal, plant growth stage, and other factors will impact the effectiveness of the phytoremediation. Development of protocols for performance monitoring are part of the goal of the Total Petroleum Hydrocarbons (TPH) in Soil and the Alternative Cover Subgroups of the Phytoremediation of Organics Action Team sponsored by the RTDF.

The phytoremediation process and location of the contaminants within the plant-soil-water system need to be known to ensure that unplanned transfer of the contaminant to the environment does not occur. For example, a remediation system that is designed to use rhizodegradation should not have excessive contaminant uptake and accumulation within the plant to the point where a risk is introduced. Risk analyses are likely to be important and necessary, given the potential pathways for contaminant transformation and transfer. Sampling and analysis of the aboveground plant matter and of the transpiration gases for contaminants and degradation products may be necessary to ensure that the contaminants are not transferred to the food chain. Calculation of a bioconcentration factor, the concentration in the plant relative to the concentration in the soil or ground water, can be done to estimate the effectiveness of the remediation as well as potential transfer to the food chain. Proper sample collection and analysis protocols should be followed to ensure correct results; for example, windborne dust could contaminate plant samples (Brooks, 1998b) or contaminants taken in through aboveground foliage could lead to an erroneous conclusion that uptake is occurring.

Visual and chemical analysis of the plant tissues may be necessary to recognize phytotoxicity symptoms, diagnose nutrient deficiencies, and optimize nutrient additions. Sap and transpiration stream measurement might be necessary to determine water usage rates and contaminant uptake. Temperature and precipitation data can be used to time watering and fertilizing of the plants. Sequential extraction steps may be necessary to determine the bioavailability of the contaminants to the plants. Field-validated hydrologic and plant uptake models will need to be developed and used to optimize the phytoremediation system or to predict behavior.

Status of Phytoremediation

Phytoremediation has been investigated in the laboratory and field by government, industry, and university research groups. The Phytoremediation of Organics Action Team of the RTDF is a phytoremediation research collaboration between industry and EPA. Team Subgroups include the Total Petroleum Hydrocarbons (TPH) in Soil, Chlorinated Solvents, and Alternative Cover Subgroups. Goals of the team are to develop protocols for phytoremediation site evaluation, designs for implementation, and monitoring for efficacy/risks; and to establish standardized field test plots in different regions of the country. The TPH in Soil Subgroup established several field test plots starting in 1998. Information about the activities and meetings of this Action Team is accessible at http://www.rtdf.org. The Petroleum Environmental Research Forum (PERF) is a consortium of industries that is examining phytoremediation of petroleum hydrocarbon contamination. Greenhouse and field studies have been conducted by member industries.

Phytoremediation has been investigated for inclusion as part of the remedy at over a dozen Superfund sites, and has been included in the ROD for some of these (Idaho National Engineering and Environmental Laboratory, Naval Surface Warfare - Dahlgren, Tibbetts Road, Calhoun Park Area, and Naval Undersea Warfare Station Superfund sites).

Phytoremediation research to date indicates that some of the most promising applications are for chlorinated solvents (especially TCE) in shallow ground water; metals in water; radionuclides (especially ^{137}Cs and ^{90}Sr) in soil and water; petroleum hydrocarbons in soil; munitions such as TNT and RDX in soil and surface water; excessive nutrients in ground water; and selenium from soil and ground water. Some successful extraction of metals from soil has been accomplished, although more research is needed before full-scale applications can be done. Indian mustard, poplar trees, and certain grasses and legumes have been popular plants for phytoremediation studies; however, screening of many other candidate plants will likely be beneficial to find the most effective plant species. Field studies will be necessary and may have to include a range of contaminant concentrations, mixtures of contaminants, and varied experimental treatments; be longer-term; and examine additional types of contaminants. In general, phytoremediation appears to be one alternative, innovative technology that might be applied at hazardous wastes sites. Careful evaluation of its applicability and effectiveness at these sites will be required. Successful phytoremediation is likely to be achieved only through the combining of expertise from numerous scientific disciplines.

General guidance and recommendations for application of phytoremediation are now available. Documents prepared by government and industry groups (ITRC, 1999; CH2M HILL, 1999; and U.S. EPA, 2000) present general decision-making guidance and recommendations on practices to be followed in conducting phytoremediation projects. As greater field experience is gained, it is likely that more detailed and specific practices will be available in design manual or handbook form for routine use. However, it is unclear that universally-applicable specific guidelines will become available as the technology matures; phytoremediation may continue to require site-specific studies.

Notice/Disclaimer

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Quality Assurance Statement

All research projects making conclusions or recommendations based on environmentally-related measurements and funded by the Environmental Protection Agency are required to participate in the Agency Quality Assurance (QA) program. This project did not involve physical measurements and as such did not require a QA plan.

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Table 1. Phytoremediation of Metals

CONTAMINANT	MEDIUM	PROCESS	CONCENTRATION[1]	PLANT[2]	REFERENCE
			RESULTS/NOTES		
Cadmium	Soil	Phytoextraction	7.9 mg/kg	Willow (*Salix viminalis*)	Greger and Landberg, 1999
			The calculated removal rate of Cd from soil was 216.7 g/ha per year.		
		Phytostabilization	9.4 mg/kg total Cd (in mine tailings)	Hybrid poplars	Pierzynski et al., 1994
			Poplar leaves did not accumulate significant amounts of Cd when grown in an outdoor plot on mine tailings.		
	Water	Rhizofiltration	2 mg/L	Indian mustard	Dushenkov et al., 1995
			Bioaccumulation coefficient of 134 after 24 hours.		
			0.18 to 18 µM (20 to 2000 µg/L) in hydroponic solution	Indian mustard	Salt et al., 1997
			Bioaccumulation coefficients of 500 to 2000. Seedlings removed 40 to 50% of the Cd within 24 hours.		
			1 to 16 mg/L	Water milfoil (*Myriophyllum spicatum*)	Wang et al., 1996
			Minimum residual concentration of about 0.01 mg/L.		
			0.1 to 10 mg/L	Duckweed, water hyacinth	Zayed et al., 1998; Zhu et al., 1999
			Bioconcentration factor of 500 to 1300, in duckweed whole plant tissue. Water hyacinth bioconcentration factor of 185 in shoots and 2150 in roots, at 0.1 mg/L.		
Chromium	Soil	Phytoextraction	*NA*	None	Brooks, 1998b
			Brooks (1998b) indicates that there is no evidence of Cr hyperaccumulation by any vascular plants.		
		Phytostabilization	Unspecified.	Indian mustard (*Brassica juncea*)	Salt et al., 1995
			Some laboratory evidence indicated that Cr(VI) might be reduced to Cr(III) by *B. juncea* roots.		
	Sludge	Phytoextraction	214 mg/kg	Aquatic macrophytes: *Bacopa monnieri*, *Scirpus lacustris*, *Phragmites karka*.	Chandra et al., 1997
			Maximum Cr accumulation was in *Phragmites karka* (816 mg/kg dry weight) at 12 weeks.		
	Water	Rhizofiltration	4 mg/L Cr(VI)	Indian mustard	Dushenkov et al., 1995
			Bioaccumulation coefficient of 179 after 24 hours. The roots contained Cr(III), indicating reduction of Cr(VI).		
			0.1 to 10 mg/L Cr(VI)	Water hyacinth	Zhu et al., 1999
			Maximum water hyacinth bioconcentration factor was 1823 in roots, at 0.1 mg/L.		

Table 1. continued

CONTAMINANT	MEDIUM	PROCESS	CONCENTRATION[1]	PLANT[2]	REFERENCE
			RESULTS/NOTES		
Cobalt	Soil	Phytoextraction	*NA*	*NA*	McGrath, 1998
			Although Co (and Cu) hyperaccumulators exist, McGrath indicates that no demonstration of phytoextraction of Co and Cu has been made.		
Copper	Soil	Phytoextraction	*NA*	*NA*	McGrath, 1998
			Although Cu (and Co) hyperaccumulators exist, McGrath indicates that no demonstration of phytoextraction of Cu and Co has been made.		
		Phytostabilization	25 to 15,400 ppm in mine tailings.	Perennial grasses such as *Agrostis tenuis* and *Festuca rubra*. *Agrostic tenuis* cv. Parys is available commercially.	Smith and Bradshaw, 1979
			Naturally-occurring metal-tolerant populations of grasses grew well, provided that fertilization was sufficient.		
	Water	Rhizofiltration	6 mg/L Cu(II)	Indian mustard	Dushenkov et al., 1995
			Bioaccumulation coefficient of 490 after 24 hours.		
			1 to 16 mg/L	Water milfoil (*Myriophyllum spicatum*)	Wang et al., 1996
			Minimum residual concentration of about 0.01 mg/L.		
			0.1 to 10 mg/L	Duckweed, water hyacinth	Zayed et al., 1998; Zhu et al., 1999
			Bioconcentration factor of 200 to 800, in duckweed whole plant tissue. Maximum water hyacinth bioconcentration factor was 595.		
Lead	Soil	Phytoextraction	At 0 to 15 cm depth: 40% of site had >400 mg/kg; 7% of site had >1000 mg/kg	Indian mustard	Blaylock et al., 1999
			"Magic Marker" site. After 3 crops, 28% area had > 400 mg/kg, 0% area had > 1000 mg/kg. The 15-30 cm and 30-45 cm depths did not change significantly. Plant shoot lead was <100 mg/kg to >3000 mg/kg. Projected cost is 50-75% of traditional costs.		
			625 µg/g (dry weight) in sand	*Brassica juncea* cultivars	Kumar et al., 1995
			The best cultivar could theoretically remove 630 kg Pb/ha if the shoots were harvested.		
		Phytostabilization	625 mg/kg	*Brassica juncea* seedlings	Salt et al., 1995
			Leachate concentration was 740 mg/L without plants and was 22 mg/L with plants.		

Table 1. continued

CONTAMINANT	MEDIUM	PROCESS	CONCENTRATION[1]	PLANT[2]	REFERENCE
			RESULTS/NOTES		
	Water	Rhizofiltration	2 to 500 mg/L in hydroponic solution	Indian mustard	Dushenkov et al., 1995
			Bioaccumulation coefficient of 563 after 24 hours for 2 mg/L solution.		
			0.096 to 9.6 µM (20 to 2000 µg/L) in hydroponic solution	Indian mustard	Salt et al., 1997
			Bioaccumulation coefficients of 500 to 2000.		
			1 to 16 mg/L	Water milfoil (*Myriophyllum spicatum*)	Wang et al., 1996
			Minimum residual concentration was below 0.004 mg/L.		
Manganese	Water	Rhizofiltration	1 mg/L in hydroponic solution	Twelve wetland plants	Qian et al., 1999
			Smart weed (*Polygonum hydropiperoides* L.) was most effective of the plants tested in removing Mn, and could remove 306 g Mn/ha per day.		
Mercury	Water	Rhizofiltration	1 mg/L in hydroponic solution	Twelve wetland plants	Qian et al., 1999
			Smart weed (*Polygonum hydropiperoides* L.) was most effective of the plants tested in removing Hg, and could remove 71 g Hg/ha per day.		
	Soil and ground water	Phytovolatilization	5 µM Hg(II) (1 mg/L) in hydroponic solution	Genetically altered *Arabidopsis thaliana* and tobacco (*Nicotiana tabacum*)	Meagher et al., 2000
			At seven days, tobacco plants had decreased Hg(II) in solution from 5 to 1.25 µM by reducing it to less toxic metallic mercury, which was volatilized.		
Nickel	Soil	Phytoextraction	14 to 3,333 mg/kg	*Berkheya coddii*	Robinson et al., 1997
			It was estimated that plants could achieve a Ni content of 5000 µg/g and remove 110 kg Ni per ha. Plants did not grow in soils with 10,000 mg/kg Ni.		
		Phytostabilization	Unspecified concentrations in mine tailings.	Native plants (herbs, shrubs, and trees) including hyperaccumulators and legumes.	Brooks et al., 1998
			Plants and soil amendments have been used to reclaim mine tailings in New Caledonia.		
	Water	Rhizofiltration	1 mg/L in hydroponic solution	Twelve wetland plants	Qian et al., 1999
			Smart weed (*Polygonum hydropiperoides* L.) was most effective of the plants tested in removing Ni, and could remove 108 g Ni/ha per day.		

Table 1. continued

CONTAMINANT	MEDIUM	PROCESS	CONCENTRATION[1]	PLANT[2]	REFERENCE
			RESULTS/NOTES		
Thallium	Soil	Phytoextraction	0.321 to 18 mg/kg	Winter rape (*Brassica napus* L.), winter wheat, corn, cabbage, leek	Tremel et al., 1997
			The highest average Tl concentration was 20 mg/kg in rape shoots, on the highest Tl-concentration soil. On a dry weight basis, the maximum accumulation in rape was 2.5x the concentration of Tl in the soil.		
Zinc	Soil	Phytoextraction	124 to 444 mg/kg Zn (other metals were also present at lesser concentrations).	Seven species of metals hyperaccumulators	McGrath et al., 2000
			Thlaspi caerulescens and *Cardaminopsis halleri* accumulated high levels of Zn, averaging 1232 to 3472 mg/kg. These species removed 4.6 to 17.6 kg Zn/ha per year.		
		Phytostabilization	Up to 43,750 mg/kg in mine rock waste material.	Native grasses, tame grasses, leguminous forbs.	Pierzynski et al., 1994
			Mycorrhizae and organic amendments in soil enhanced plant growth. Some uptake of Zn by plants occurred.		
	Water	Rhizofiltration	100 mg/L in hydroponic solution	*Brassica juncea*	Dushenkov et al., 1995
			Over 13,000 µg/g Zn accumulated in roots after 24 hours. Some Zn may have been translocated into shoots and root exudates may have precipitated Zn from solution.		

[1]Concentration units are given as they were reported in the original reference. Conversions of molar concentrations have been added.
[2]Plant names are given as they were reported in the original reference.

Table 2. Phytoremediation of Metalloids, Non-metals, Radionuclides, and Nutrients

CONTAMINANT	MEDIUM	PROCESS	CONCENTRATION[1]	PLANT[2]	REFERENCE
			RESULTS/NOTES		
Metalloids/non-metals					
Arsenic	Surface water	Phytoextraction Rhizofiltration	>0.05 µg/mL	Aquatic plants *Ceratophyllum demersum*, *Egeria densa*, and *Lagarosiphon major*	Brooks and Robinson, 1998
			Arsenic concentrations in plants were up to 1200 µg/g dry weight. In *Ceratophyllum demersum*, the concentrations were about 10,000 times the arsenic concentration in the water. The use of these plants was suggested as a means of reducing arsenic concentrations in the water.		
	Soil	Phytoextraction Phytostabilization	Unspecified.	Couch grass, i.e., bermudagrass, (*Cynodon dactylon*), thatch grass (*Pinicum sativum*), amaranths (*Amaranthus hybridus*)	Jonnalagadda and Nenzou, 1997
			Couch grass grew on metal-rich mine dumps and accumulated 10,880 mg/kg arsenic in the roots and 1,660 mg/kg in the stem/leaves (dry weight).		
		Phytostabilization Phytoextraction	1250 mg/kg (in mine tailings)	Lambsquarters, poplars	Pierzynski et al., 1994
			Lambsquarters leaves had relatively higher As concentrations (14 mg/kg As) than other native plant or poplar leaves (8 mg/kg) in mine-tailing wastes.		
Selenium	Ground water (irrigation drainage water)	Phytoextraction	100, 300, 500 µg/L	Hybrid poplar clones (*Populus*)	Bañuelos et al., 1999
			Maximum Se content was 9.1 mg/kg dry matter in 500 µg/L treatment, but Se content did not exceed 1 mg/kg dry matter in 100 and 300 µg/L treatments. Poplars accumulated more Se than other tree species; however, net accumulation of Se was said to be minimal.		
	Water	Rhizofiltration	0.1 to 10 mg/L	Duckweed, water hyacinth	Zayed et al., 1998; Zhu et al., 1999
			Duckweed maximum bioconcentration factor was 850, in whole plant tissue.		
	Soil	Phytoextraction Phytovolatilization	40 mg/kg	Canola (*Brassica napus* cv. Westar) kenaf (*Hibiscus cannabinus* L. cv. Indian), tall fescue (*Festuca arundinacea* Schreb cv. Alta)	Bañuelos et al., 1997
			The plants accumulated Se. Canola reduced total soil by 47%, kenaf reduced it by 23%, and tall fescue reduced it by 21%.		

Table 2. continued

CONTAMINANT	MEDIUM	PROCESS	CONCENTRATION[1]	PLANT[2]	REFERENCE
			RESULTS/NOTES		
Boron	Water (soil solution)	Phytoextraction	10 mg water-extractable B/L	Indian mustard (*Brassica juncea* Czern L.), tall fescue (*Festuca arundinacea* Schreb. L.), birdsfoot trefoil (*Lotus corniculatus* L.), kenaf (*Hibiscus cannibinus* L.)	Bañuelos, 1996
			Mean B concentrations in shoots ranged from 122 mg B/kg dry matter in birdsfoot trefoil to 879 mg B/kg dry matter in kenaf leaves. In two years, each plant species lowered extractable B concentrations in soil by at least 25%.		
Perchlorate	Water (hydroponic solution)	Phytoextraction Phytodegradation Rhizodegradation	10, 22, 100 mg/L	Willow (*Salix nigra*), Eastern cottonwood (*Populus deltoides* and hybrid *populus*), and eucalyptus (*Eucalyptus cineria*).	Nzengung et al., 1999
			Willows had best growth under hydroponic conditions and degraded perchlorate from 10 mg/L to below detection (2 µg/L) in about 20 days, from 22 mg/L to BD in about 35 days, and from 100 mg/L to BD in about 53 days. 1.3% of initial perchlorate mass was found in willow plant tissues, especially leaves and upper stems (branches). Leaves had 813.1 mg/kg perchlorates. Some evidence for perchlorate degradation within leaves. Perchlorate degradation rates decreased as nitrate concentration increased, and type of nitrogen source affects perchlorate degradation rate.		
Radionuclides					
Cesium or ^{137}Cs	Soil	Phytoextraction	2600 Bq/kg average	Amaranth species	Dushenkov et al., 1999
			Extraction of ^{137}Cs was limited by binding to soil; addition of amendments did not increase bioavailability.		
	Water	Rhizofiltration Rhizofiltration	C_o = 200 µg/L	Sunflowers	Dushenkov et al., 1997
			In bench-scale and pilot-scale engineered systems using the stable isotope, C_o decreased noticeably after 6 hours, then went below 3 µg/L after 24 hours.		
			20 to 2000 µg/L	Indian mustard	Salt et al., 1997
			Accumulation by the roots, with bioaccumulation coefficients of 100 to 250.		
Eu(III) [surrogate for radionuclide Am(III)]	Water	Rhizofiltration	3.3 x 10^{-4} M (50 mg/L)	Water hyacinth (*Eichhornia crassipes*)	Kelley et al., 1999
			26% of the Eu(III) in solution was removed. Eu(III) on roots was 0.01 g/g dry weight root material. Almost all of the removed Eu(III) was found on the roots.		
^{60}Co	Soil	Phytoextraction	C_o = 1.59 Bq ^{60}Co/g soil	Yellow sweetclover (*Melilotus officinalis* (L.) Lam) and Sudan grass (*Sorghum sudanese* (Piper) Stapf.)	Rogers and Williams, 1986
			2.6% of the total ^{60}Co in soil was removed by two harvests of clover, at 65 and 93 days. 1.2% of the total ^{60}Co in soil was removed by two harvests of grass, at 85 and 119 days.		

Table 2. continued

CONTAMINANT	MEDIUM	PROCESS	CONCENTRATION[1]	PLANT[2]	REFERENCE
			RESULTS/NOTES		
^{226}Ra, ^{222}Rn	Soil	Phytoextraction	C_o = 4.4 x 10^3 Bq ^{226}Ra/kg soil	Corn (*Zea mays* L.), dwarf sunflower (*Helianthus annuus* L.), tall fescue (*Festuca arundinacea* Schreb.)	Lewis and MacDonell, 1990
			^{226}Ra and ^{222}Rn are taken up by plants, although percent removal per year of ^{226}Ra from soil is likely to be very low. A negative impact is that ^{222}Rn taken up is released by plants, with the amount released dependant on leaf area. Plants could increase the escape of ^{222}Rn through a soil cover.		
Strontium or ^{90}Sr	Ground water	Rhizofiltration	200 µg/L	Sunflowers	Dushenkov et al., 1997
			Using the stable isotope, C_o decreased to 35 µg/L within 48 hours, then to 1 µg/L by 96 hours		
			20 to 2000 µg/L	Indian mustard	Salt et al., 1997
			Root bioaccumulation coefficients were 500 to 2000		
	Soil	Phytoextraction	100 Bq ^{137}Cs/g soil 112 Bq ^{90}Sr/g soil	Bahia grass (*Paspalum notatum*), johnson grass (*Sorghum halpense*), switchgrass (*Panicum virgatum*)	Entry et al., 1999
			In 3 harvests in 24 weeks, aboveground biomass of bahia grass accumulated 26.3 to 46.7% of applied ^{137}Cs and 23.8 to 50.1% of applied ^{90}Sr; johnson grass accumulated 45.5 to 71.7% of applied ^{137}Cs and 52.6 to 88.7% of applied ^{90}Sr; and switchgrass accumulated 31.8 to 55.4% of applied ^{137}Cs and 36.8 to 63.4% of applied ^{90}Sr. Mycorrhizal infection of all plant species increased amount of uptake of ^{137}Cs and ^{90}Sr. Favorable experimental conditions may have enhanced uptake.		
Uranium	Soil	Phytoextraction	280 and 750 mg/kg total U	Amaranth (*Amaranth cruentus* L.), *Brassica juncea* (various cultivars), bush bean (*Phaseolus vulgaris* L.), Chinese cabbage (*Brassica chinensis* L.), Chinese mustard (*Brassica narinosa* L.), corn (*Zea mays*), cow pea (*Pisum sativum* L.), field pea (*Pisum sativum* L.), sunflower (*Helianthus annuus* L.), and winter wheat (*Triticum aestivum* L.)	Huang et al., 1998
			Amaranth, *Brassica juncea*, Chinese cabbage, and Chinese mustard had significant U concentration in shoots, when soil was treated with citric acid. Maximum shoot U concentrations were more than 5000 mg/kg. Different cultivars had significantly different accumulation of U.		
	Water	Rhizofiltration	10 to 2430 µg/L	Sunflowers	Dushenkov et al., 1997
			Initial concentrations declined significantly within 48 hours, in some cases to below the remedial goal.		

Table 2. continued

CONTAMINANT	MEDIUM	PROCESS	CONCENTRATION[1]	PLANT[2]	REFERENCE
			RESULTS/NOTES		
Nutrients					
NO_3^-	Ground water	Hydraulic control Riparian corridors Buffer strips	150 mg/L	Poplars (*Populus* spp.)	Licht and Schnoor, 1993
			Nitrate in ground water decreased from 150 mg/L at the edge of a corn field, to 8 mg/L below a downgradient poplar buffer strip, and then to 3 mg/L downgradient at the edge of a stream.		

[1]Concentration units are given as they were reported in the original reference. Conversions of molar concentrations have been added.
[2]Plant names are given as they were reported in the original reference.

Table 3. Phytoremediation of Organic Contaminants

CONTAMINANT	MEDIUM	PROCESS	CONCENTRATION[1]	PLANT[2]	REFERENCE
			RESULTS/NOTES		
Petroleum hydrocarbons/PAHS					
Crude oil	Soil	Rhizodegradation	8200 to 16,000 mg/kg TPH	St. Augustine grass (*Stenotaphrum secundatum* L.), rye (*Secale cereale* L.)-soybean-rye rotation, sorghum-sudan grass (*Sorghum bicolor sudanese* L.)	Schwab and Banks, 1999
			At this Gulf Coast pipeline site, dissipation of TPH at 21 months ranged from 35 to 50% in planted plots, compared to 21% in unplanted plots.		
Diesel (weathered)	Soil	Rhizodegradation	3000 mg/kg TPH	Tall fescue, bermudagrass, ryegrass, white clover	AATDF, 1998
			Craney Island field phytoremediation site. TPH decreased 50% in clover plots, 45% in fescue plots, about 40% in bermuda grass plots, and about 30% in unvegetated plots in 24 months. TPH and PAH decreases in vegetated plots were statistically significantly greater than in unvegetated plots.		
No. 2 fuel oil	Soil	Rhizodegradation	40 to 5000 mg/kg DRO	Hybrid willow	Carman et al., 1998
			Willow roots were established in contaminated soil. Contribution of rhizosphere to biodegradation had not yet been assessed.		
BTEX	Soil	Rhizodegradation	Uncontaminated soil was used for aerobic MPN cultures containing 5 mg/L each of benzene, toluene, and *o*-xylene.	Hybrid poplar trees (*Populus deltoides* × *nigra* DN-34, Imperial Carolina)	Jordahl et al., 1997
			Ratio of BTX degraders in rhizosphere soil compared to surrounding soil was 5:1.		
	Water	Phytovolatilization	50 mg/L in hydroponic solution	Hybrid poplar	Burken and Schnoor, 1998, 1999
			18% of applied benzene, 10% of applied toluene and ethylbenzene, and 9% of applied *m*-xylene were volatilized from poplar cuttings in hydroponic solution.		
PAHs	Soil	Rhizodegradation	10 mg/kg chrysene, benz(a)anthracene, benzo(a)pyrene, and dibenz(a,h)anthracene	Eight types of prairie grasses	Aprill and Sims, 1990
			The disappearance of PAHs was greater in vegetated soils than in unvegetated soils. From greatest to least, the order of disappearance was benz(a)anthracene > chrysene > benzo(a)pyrene > dibenz(a,h)anthracene.		
			298 $\pm$ 169 mg/kg total PAHs for 15 PAHs	Perennial ryegrass (*Lolium perenne*)	Ferro et al., 1999
			Fluoranthene, pyrene, and chrysene had greater disappearance in planted soils compared to unplanted soils.		

Table 3. continued

CONTAMINANT	MEDIUM	PROCESS	CONCENTRATION[1]	PLANT[2]	REFERENCE
			RESULTS/NOTES		
			100 mg/kg anthracene and pyrene	Fescue (*Festuca arundinacea* Schreb.), sudangrass (*Sorghum vulgare* L.), switchgrass (*Panicum virgatum* L.), and alfalfa (*Medicago sativa* L.)	Reilley et al., 1996
			PAH disappearance was greater in vegetated treatments than in unvegetated treatments.		
MGP site wastes (PAHs)	Soil	Rhizodegradation	100 to 200 mg/kg total PAHs	Alfalfa (*Medicago sativa*), switchgrass (*Panicum virgatum*), little bluegrass (*Schizachyrium scoparium*)	Pradhan et al., 1998
			Planted soils had a greater percent reduction in total and carcinogenic PAHs than unplanted soils.		
Chlorinated solvents					
Tetrachloroethene (PCE)	Water	Rhizofiltration	0.5 to 10 mg/L	Waterweed (*Elodea canidensis*), a submergent aquatic plant	Nzengung et al., 1999
			PCE rapidly sorbed to plant matter.		
Trichloroethene (TCE)	Soil	Rhizodegradation Phytodegradation	Not specified.	Chinese lespedeza (*Lespedeza cuneata* (Dumont)), a legume; a composite herb (*Solidago* sp.); Loblolly pine (*Pinus taeda* L.); soybean (*Glycine max* (L.) Merr., cv. Davis)	Anderson and Walton, 1995
			Significantly greater TCE mineralization occurred in soil vegetated with *L. cuneata,* Loblolly pine, and soybean than in non-vegetated soil, but not with composite herb - *Solidago* sp.		
	Ground water	Phytodegradation Phytovolatilization	Average of 0.38 mM (50 mg/L) during first season and 0.11 mM (14.5 mg/L) during second season.	Hybrid poplar (*Populus trichocarpa* × *P. deltoides*)	Newman et al., 1999
			Trees removed over 99% of added TCE. TCE transpiration was detected by leaf bag gas analysis, but not detected by open-path Fourier transform infrared spectroscopy. During second and third years, less than 9% was transpired. TCE was found in leaves, branches, trunks, and roots (and was generally highest in branches). Low levels of metabolites and reductive dechlorination products were found in plant tissues.		
1,1,1-trichloroethane (TCA)	Ground water	Phytodegradation Rhizodegradation	Not given.	Hybrid poplar	Newman et al., 1998
			In laboratory tests, hybrid poplars could take up TCA. In the field application, an underground irrigation system applied contaminated ground water to the tree roots.		

Table 3. continued

CONTAMINANT	MEDIUM	PROCESS	CONCENTRATION[1]	PLANT[2]	REFERENCE
			RESULTS/NOTES		
Munitions					
TNT	Water	Phytodegradation	1,250 to 4,440 ppb TNT and 3,250 to 9,200 ppb total nitrobodies (TNT, RDX, HMX, TNB, 2A-DNT, 4A-DNT)	Emergent aquatic plants: canary grass (*Phalaris arundinacea*), wool grass (*Scirpus cyperinus*), sweetflag (*Acorus calamus*), parrotfeather (*Myriophyllum aquaticum*)	ESTCP, 1999
			An *ex-situ* gravel-based system with aquatic plants removed TNT to goal of <2 ppb, except during winter months.		
	Soil	Rhizodegradation	41 mg/kg	Meadow bromegrass (*Bromus erectus* Huds.), perennial ryegrass (*Lolium perenne* L.), sweet vernalgrass (*Anthoxanthum odoratum* L.)	Siciliano and Greer, 2000
			Perennial ryegrass and sweet vernalgrass in contaminated soil did not survive when inoculated with a TNT-degrading bacterium. Inoculated meadow bromegrass reduced TNT in soil to about 70% of the levels in unvegetated soil.		
RDX	Water	Phytodegradation	1,250 to 4,440 ppb TNT and 3,250 to 9,200 ppb total nitrobodies (TNT, RDX, HMX, TNB, 2A-DNT, 4A-DNT)	Emergent aquatic plants: canary grass (*Phalaris arundinacea*), wool grass (*Scirpus cyperinus*), sweetflag (*Acorus calamus*), parrotfeather (*Myriophyllum aquaticum*)	ESTCP, 1999
			An *ex-situ* gravel-based system with aquatic plants removed total nitrobodies to goal of <50 ppb, except during winter months.		
	Hydroponic solution	Phytoextraction Phytodegradation	10 ppm	Bush bean (*Phaseolus vulgaris*, var. tendergreen)	Harvey et al., 1991
			Laboratory experiment was to study environmental fate of RDX, not for phytoremediation. After 7 days, roots had 6 ppm, stem had 11 ppm, and leaves had 97 ppm RDX. Limited metabolism of RDX occurred within the plant.		
	Soil irrigated with contaminated water	Phytoextraction Phytodegradation	1.0 µg/mL	Corn (*Zea mays*), tomato (*Lycopersicon esculentum*)	Larson et al., 1999
			Corn leaves contained 22 µg RDX/g dry weight and tassels contained 16 µg RDX/g dry weight; stalks and husks did not contain appreciable RDX. Tomato fruit contained 10 µg RDX/g fresh weight. High molecular weight RDX transformation products were detected in plant tissues.		

Table 3. continued

CONTAMINANT	MEDIUM	PROCESS	CONCENTRATION[1]	PLANT[2]	REFERENCE
			RESULTS/NOTES		
HMX	Water	Phytodegradation	1,250 to 4,440 ppb TNT and 3,250 to 9,200 ppb total nitrobodies (TNT, RDX, HMX, TNB, 2A-DNT, 4A-DNT)	Emergent aquatic plants: canary grass (*Phalaris arundinacea*), wool grass (*Scirpus cyperinus*), sweetflag (*Acorus calamus*), parrotfeather (*Myriophyllum aquaticum*)	ESTCP, 1999
			An *ex-situ* gravel-based system with aquatic plants removed total nitrobodies to goal of <50 ppb, except during winter months.		
Nitroglycerine	Water	Phytodegradation	1.8 mM (410 mg/L)	Sugar beet (*Beta vulgaris*) cell cultures	Goel et al., 1997
			In flask cell cultures, complete disappearance of nitroglycerine occurred in 24 to 35 hours, with formation of degradation products.		
Pesticides					
Atrazine	Soil	Phytodegradation	60.4 µg/kg	Hybrid poplar (*Populus deltoides* × *nigra* DN34, Imperial Carolina)	Burken and Schnoor, 1997
			Trees took up and metabolized atrazine to less toxic compounds. Atrazine degradation in unplanted soil was similar to degradation in planted soil.		
		Rhizodegradation	0.5 µg/g atrazine in contaminated soils, spiked with additional atrazine to >10 ppm.	*Kochia* sp.	Anderson et al., 1994
			Enhanced biodegradation of atrazine occurred in soil collected from the rhizosphere.		
	Ground water and soil water (hydroponic solution)	Phytodegradation	Unspecified: 48.3 µg atrazine in less than 270 mL solution was used in the hydroponic reactor.	Hybrid poplar (*Populus deltoides* × *nigra* DN34, Imperial Carolina)	Burken and Schnoor, 1999
			There was no volatilization of atrazine from the poplars. 53.3% of the applied atrazine was taken up into the plant. The largest percentage of atrazine taken up was found in the leaves.		
	Surface water	Phytodegradation	200 µg/L	Aquatic plants: coontail or hornwort (*Ceratophyllum demersum*), American elodea or Canadian pondweed (*Elodea canadensis*), common duckweed (*Lemna minor*)	Rice et al., 1997a
			After 16 days, atrazine concentrations were significantly reduced in presence of *Ceratophyllum demersum* (41.3% of applied atrazine remained) and *Elodea canadensis* (63.2% of applied atrazine remained) but not in unvegetated system or in presence of *Lemna minor* (85% of applied atrazine remained).		

Table 3. continued

CONTAMINANT	MEDIUM	PROCESS	CONCENTRATION[1]	PLANT[2]	REFERENCE
			RESULTS/NOTES		
Carbofuran	Soil	Rhizodegradation Phytodegradation	3 kg/ha active ingredient in field experiment	Corn (*Zea mays* L.)	Buyanovsky et al., 1995
			In first 30 days of greenhouse experiment, mineralization was greater in soil close to the roots than in soil without roots or far from roots. Uptake of carbofuran and/or degradation products occurred. In field experiment, concentrations in top 10 cm of planted soil were half the concentrations in unplanted soil.		
EDB	Ground water	Phytodegradation	25 ppm	Koa haole	Newman et al., 1998
			Uptake of EDB was investigated.		
2,4-D	Soil	Rhizodegradation	Uncontaminated soil was used.	Sugarcane (*Saccharum officinarum*)	Sandmann and Loos, 1984
			MPN counts of 2,4-D degraders were significantly greater for sugarcane rhizosphere soil than for non-rhizosphere soil.		
Other					
PCP	Soil	Rhizodegradation	100 mg/kg	Hycrest crested wheatgrass (*Agropyron desertorum* (Fisher ex Link) Schultes)	Ferro et al., 1994
			After 155 days, 22% of PCP was mineralized in planted system, but only 6% in unplanted.		
PCBs	Soil	Rhizodegradation	Not specified	Osage orange and mulberry	Fletcher et al., 1995
			Phenolics in leachates from osage orange and mulberry supported growth of PCB-degrading bacteria.		
MTBE	Ground water	Phytodegradation Phytovolatilization	Not specified	Poplar tree cell cultures, hybrid poplars, eucalyptus	Newman et al., 1998, 1999
			Eucalyptus transpired 16.5% of applied radio-labeled MTBE and hybrid poplar transpired 5.1% in laboratory chambers. Detectable transpiration of MTBE was not measured from mature eucalyptus in the field. Metabolism of MTBE by the trees is one hypothesis.		
Surfactants (LAE, LAS)	Surface water	Rhizodegradation	1 mg/L	Cattail (*Typha latifolia*), duckweed (*Lemna minor*)	Federle and Schwab, 1989
			Mineralization of LAS and LAE was faster and more extensive in water with cattails than in root-free sediment. LAE was mineralized by duckweed but LAS was not.		

Table 3. continued

CONTAMINANT	MEDIUM	PROCESS	CONCENTRATION[1]	PLANT[2]	REFERENCE
			RESULTS/NOTES		
Ethylene glycol	Soil	Rhizodegradation	1000 μg/g	Tall fescue (*Festuca arundinacea*), perennial rye grass (*Lolium perenne* L.), Kentucky bluegrass (*Poa pratensis* L.), alfalfa (*Medicago sativa*), birdsfoot trefoil (*Lotus corniculatus*), and mixed (all except birdsfoot trefoil).	Rice et al., 1997b
			At 30 days, in soils tested at 0°C, there was significantly greater CO_2 produced from soil planted with tall fescue, perennial rye grass, Kentucky bluegrass, alfalfa, and mixed than from unplanted soil. At 20°C, there was significantly greater CO_2 produced from all planted soils than from unplanted soil.		

[1]Concentration units are given as they were reported in the original reference. Conversions of molar concentrations have been added.
[2]Plant names are given as they were reported in the original reference.