Remediation of Metal- and Radionuclides-Contaminated Soils by In Situ Stabilization Techniques

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CONTENTS
2.1 Introduction ................................................................................................................ ...........21
2.2 Techniques of In Situ Stabilization .....................................................................................23
  2.2.1 Lime ..............................................................................................................................23
  2.2.2 Zeolites ..........................................................................................................................25
  2.2.3 Apatite ...........................................................................................................................28
  2.2.4 Fe and Mn Oxides, Fe- and Mn-Bearing Amendments .............................................31
  2.2.5 Alkaline Composted Biosolids .....................................................................................38
  2.2.6 Other Minerals and Industrial By-Products ..............................................................39
  2.2.7 In Situ Redox Manipulation ..........................................................................................47
2.3 Summary and Conclusions .........................................................................................................53
Acknowledgments ........................................................................................................................53
References .......................................................................................................................................53

2.1 Introduction

The extent of metal and radionuclide contamination in the world is immense. In the soil environment, metals and radionuclides can be dissolved in solution, held on inorganic soil constituents through various sorption or ion exchange reactions, complexed with soil organics, or precipitated as pure or mixed solids. Soluble contaminants are subject to migration with soil water, uptake by plants or aquatic organisms, or loss due to volatilization (Smith et al., 1995). Lead (Pb), chromium (Cr), zinc (Zn), arsenic (As), and cadmium (Cd) are the most frequently identified inorganic contaminants in soil and groundwater in the order of their relative occurrence (National Research Council, 1994; Knox et al., 1999). Unlike degradable organic contaminants and even short-lived radionuclides that can become less toxic over time, metals can be considered conservative because they are not decomposed in the environment. However, many metals, especially redox-sensitive elements such as As and Cr, can undergo transformations or sorption reactions that alter both mobility and relative toxicity.

Soil contamination can have dire consequences, such as loss of ecosystem and agricultural productivity, diminished food chain quality, tainted water resources, economic loss, and human and animal illness. Public attention generally focuses on dramatic examples of
contamination such as the nuclear accident in Chernobyl, Ukraine, where significant releases of radioactivity occurred (Adriano et al., 1997). The most dramatic ecological effects, however, were confined to a 30-km radius from the reactor. In contrast, extensive areas of eastern and central Europe suffer from diseases associated with elevated levels of Pb in the air, Co in the soil, and a food chain that is contaminated by metals related to heavy industry (Tikhonov, 1996). At present, there is a critical need for the development of cost-effective remediation technologies that reduce such risks.

Attention has focused on the development of in situ stabilization methods that are generally less expensive and disruptive to the natural landscape, hydrology, and ecosystem than conventional excavation, treatment, and disposal methods. A major drawback to such an approach, however, is that the metal or radionuclide contaminant remains in place and may, due to various physicochemical and biological processes, become a health or regulatory concern at a later date.

In situ remediation techniques for metals and radionuclides typically rely on a fundamental understanding of the natural geochemical processes governing the speciation, migration, and bioavailability of a given element in the environment. Remediation techniques can be placed in one of three general categories: (1) physical methods that simply restrict access to the contamination through containment or removal; (2) chemical methods that attempt to alter contaminant speciation to either enhance mobility under various extraction scenarios or decrease mobility to reduce potential exposure hazards; and (3) biological methods that attempt to use natural or enhanced biochemical processes to either increase contaminant mobility for extraction (e.g., phytoaccumulation) or reduce mobility by altering metal speciation. For instance, natural geochemical processes such as precipitation, sorption, ion exchange, and even redox manipulation have been employed as remediation methods.

This chapter will focus on the use of natural and synthetic soil amendments to reduce the mobility and bioavailability of contaminant metals and radionuclides without drastically altering the physical or chemical properties of the soil. An overview of the important mechanisms controlling contaminant stabilization will be given for additives such as lime, zeolites, phosphate compounds (e.g., apatite minerals), Fe and Mn oxides and other minerals, biosolids, and industrial by-products. Many of these amendments are inexpensive, readily available, and can be applied to large areas of contaminated soil without the need for costly excavation.

The efficacy of in situ stabilization is usually evaluated using one of the following approaches: (1) batch studies evaluating contaminant solubility and migration potential under controlled equilibrium conditions, many of which are designed to identify and characterize resulting mineral phases or specific sorption processes using spectroscopic techniques (e.g., Berti and Cunningham, 1997; Chen et al., 1996, 1997; Lothenbach et al., 1998; Ma et al., 1993; Ruby et al., 1994; Xu et al., 1994; Zhang et al., 1997); (2) dynamic leaching studies (i.e., columns) that simulate kinetically limiting conditions (e.g., Seaman et al., 1995, 1999); and (3) plant growth experiments that indicate bioavailability and long-term stability under variable moisture conditions which are more indicative of the field environment (e.g., Chlopecka and Adriano, 1996; Laperche et al., 1997; Lothenbach et al., 1998). This reflects a general progression from well-defined systems to more complex conditions typical of the real world. Often sequential extractions such as Tessier’s method (Tessier et al., 1979) are combined with the above techniques to operationally define contaminant mineral associations, bioavailability and potential mobility, and chemical liability (Arey et al., 1999; Chlopecka and Adriano, 1996, 1997a, b, c, and d, 1999; Ma and Rao, 1997).

Selective extraction techniques successively liberate less-chemically labile phases and, ideally, their associated contaminants. The operational nature of such techniques is generally illustrated, however, by the apparent association of contaminants with specific phases
that may not actually be present in the soil or that seem to contradict the likely mechanism of metal immobilization. For example, much of the research on in situ stabilization methods has focused on the use of apatite addition to promote the precipitation of sparingly soluble metal phosphates. Although selective extraction methods have been widely applied to such studies, it is still unclear which extract would likely solubilize such precipitates (Arey et al., 1999). In addition, the possible redistribution of contaminant metals during the extraction process cannot be discounted. Numerous studies have demonstrated that various soil amendments such as hydrous ferric oxid (HFO) and lime can result in significant phase redistribution of contaminant metals, as defined by extraction, which is not reflected by plant uptake (Sappin-Didier et al., 1997a and b; Müller and Pluquet, 1997; Chlopecka and Adriano, 1996). Therefore, batch extraction methods provide a means of rapidly screening numerous alternative treatment scenarios, especially when evaluating contaminant mobility, but the limitations of such methods necessitate the use of plant growth experiments and bioassays to assess biological availability.

In a sense the study of soil additives for contaminant metal immobilization parallels earlier research efforts related to soil fertility and the development of chemical fertilizers. Such studies are often empirical in nature because the addition of a reactive component to soils can have numerous predictable and unforeseen consequences. As will be discussed, a reduction in the solubility or plant availability of a given metal can result from several different and distinct chemical mechanisms. Like previous soil fertility studies, factors such as soil heterogeneity (i.e., soil type, mineralogy, contamination level, etc.), climate (i.e., moisture conditions, temperature, etc.), and the concentration and speciation of a given contaminant, as well as the specific plant species and genotype, all play an important role in determining the relative effectiveness of a given amendment under a specific set of conditions.

2.2 Techniques of In Situ Stabilization

2.2.1 Lime

Soils differ considerably in their pH and most crops grow best when the soil pH is between 6.5 to 7.0. For centuries, lime in various forms [e.g., CaCO₃, (Ca, Mg)CO₃, CaO, Ca(OH)₂] has been used to increase soil pH and thereby improve soil fertility. Lime is a cheap and effective ameliorant for many metals; however, repeated applications are required to maintain metal immobilization. Lime is applied frequently (2 to 5 years) and in quantities larger than any other inorganic soil amendment (typically 2 to 10 t/ha). Soil pH is an important factor controlling metal mobility and bioavailability. Usually the mobility of many metals increases with decreasing pH. With increasing pH, the solubility of most trace cations will decrease (Kabata-Pendias and Pendias, 1992). Sims and coworkers have demonstrated the effects of pH on micronutrient distribution among soil fractions. They found that more Cu, Fe, Mn, and Zn were in the exchangeable and organic fractions at low pH than at high (Sims and Patrick, 1978). Also, Iyengar et al. (1981) found that exchangeable Zn generally increases in soils with decreasing pH. This observation can be explained by the precipitation of metal hydroxides, changes in the carbonate and phosphate concentrations in the soil water, adsorption and desorption of metals by hydrous oxides and organic matter, and the formation and dissolution of Fe and Mn oxides. For example, the metals Cd and Zn are illustrative of the effect of pH on their mobility. Cadmium exists in the divalent form to pH 7.8 and only 50% is converted to the Cd(OH)₂ precipitate at pH 11. In contrast, 50% of
Environmental Restoration of Metals–Contaminated Soils

Zn is in the Zn(OH)$_2$ form at pH 7.5, suggesting that at a given pH, Zn will be less mobile than Cd in a soil system.

Presented in this section are greenhouse data demonstrating the effect of lime on metal mobility and plant uptake (Knox, 1998a). In this experiment, an Appling soil was contaminated with the following metals at two treatment levels (mg kg$^{-1}$): Zn – 1000, 2000; Cd – 20, 40; Pb – 1500, 3000; Cu – 500, 1000; Ni – 350, 700. The metals were added to the soil separately as a mixture of various metal sources (40% of sulfate, 25% of carbonate, 20% of oxide, and 15% of chloride). After equilibrating 4 weeks, lime was added at variable rates to adjust the pH from 5.4 to 6.5. In the control soil, native Zn was found mostly in the residual form (71% of total), with lesser amounts in the organic and the iron-manganese oxide fractions. In the contaminated soil (two levels of Zn), Zn increased in all fractions, with the largest increase in the exchangeable fraction (30 to 42% of the total Zn concentration in the soil) (Figure 2.1). Addition of lime significantly decreased the exchangeable fraction of Zn by increasing concentrations in the carbonate, iron-manganese oxide, and residual fractions.

In treatments where Pb was added, the initial soil pH was low (5.1) and the mobility of Pb was high, with 58.8 and 123 mg kg$^{-1}$ found in the exchangeable fraction at low and high treatment levels, respectively. When the soil pH was raised to 6.1 by addition of lime, Pb mobility was significantly reduced by 54 and 45%, respectively, at the first and second level of Pb (Figure 2.2). For the other studied metals (Cd, Cu, and Ni) addition of lime to the soil significantly reduced the exchangeable fraction (Figure 2.1). In Zn, Cu, and Ni treated soil early plant mortality resulted, and yield of rye and maize was not obtained (Figure 2.2). Yield of these plants was obtained only in treatments with both levels of Pb and Cd.
When soil pH increased to 6 or 6.5, rye and maize yield was obtained in treatments with Zn, Cu, and Ni. However, at the high metal level (i.e., Zn—2000, Cu—1000, Ni—700 mg kg\(^{-1}\)) yield was much lower than in treatments with low metal level (Zn—1000, Cu—500, and Ni—350 mg kg\(^{-1}\)) (Figure 2.2). Metal concentrations in plant tissues for treatments without lime were high. For example, Cd and Pb at the high metal application rate were 79.6 and 138 mg kg\(^{-1}\) in rye tissues, respectively. In the control soil, Cd and Pb concentrations were only 0.7 and 1.3 mg kg\(^{-1}\), respectively (Figure 2.3). Lime significantly decreased metal uptake by plants (Figure 2.3).

2.2.2 Zeolites

Zeolites are framework aluminosilicates consisting of extended three-dimensional networks of linked SiO\(_4\) and AlO\(_4\) tetrahedra. They possess interconnected channels or voids that form ideal sorption sites for both water and specific alkali and alkali-earth metals. Nearly 50 natural forms of zeolite have been identified and over 100 forms having no natural analogs have been synthesized in the lab. The unique physical and chemical properties of zeolites combined with their natural abundance in sedimentary deposits and volcanic parent materials have made them useful in many industrial processes.

Zeolites derive cation exchange capacity from Al\(^{3+}\) substitution for Si\(^{4+}\) with the size of the channel determining the type of exchangeable cation that is preferred (Breck, 1974). Recent greenhouse and field studies have demonstrated the ability of zeolites to reduce the uptake of Cs, Sr, Cu, Cd, Pb, and Zn in plants (Chelishchev, 1995; Leppert, 1990; Mumpton 1984; Mineyev et al., 1990; Rebedea, 1997; Rebedea and Lepp, 1994). However, zeolites are not effective sorbents for transuranic species, such as uranyl (UO\(_2^{2+}\)), that are commonly...
found at sites with elevated Cs and Sr levels (Vaniman and Bish, 1995). To date, results from the use of zeolites as soil amendments to reduce plant uptake of radionuclides and contaminant metals have been mixed (Adriano et al., 1997; Leppert, 1990; Mineyev et al., 1990). For example, Mineyev et al. (1990) observed that clinoptilolite application to an acidic podzolic soil that had been spiked with Zn, Pb, and Cd reduced acid-extractable Zn, but not the concentration of acid-soluble Pb and Cd. Clinoptilolite application also reduced Zn and Pb accumulation in barley grain, but had no effect on yield. However, Chlopecka and Adriano (1996, 1997a) observed that the application of clinoptilolite (15 g kg$^{-1}$ soil) significantly decreased exchangeable Zn, Cd, and Pb. For example, the exchangeable Zn concentration decreased from 237 to 189 mg kg$^{-1}$ with zeolite application.

In another set of studies, Chlopecka and Adriano (1997c), Knox (1998b), and Knox and Adriano (1999) applied the natural zeolite, phillipsite, to soils from Canada, Poland, Taiwan, and the Czech Republic, which were contaminated with As, Cd, Pb, and Zn from mining, smelting, and other industrial activities. Zeolite was added at two rates, 25 and 50 g kg$^{-1}$, both of which significantly enhanced the yield of maize and oats and reduced the plant uptake of Cd, Pb, and Zn (Figure 2.4, Table 2.1). Zeolite application also influenced the plant uptake of both macro- and micronutrients for the Czech soil with an increase in plant tissue concentrations observed for Ca and Mg and a decrease of Mn from 933 to 256 mg kg$^{-1}$. For the highly polluted Czech soil, zeolite addition reduced the exchangeable Cd, Pb, and Zn by 43, 46, and 29%, respectively, but increased the concentration of each of those metals in the residual fraction (Knox, 1998b; Knox and Adriano, 1999). Other research groups have reported similar reductions in the uptake of Cd by lettuce after application of zeolite material from the foyazite group, type 4A (Rebedea et al., 1994; Rebedea, 1997). Rebedea et al. (1994) investigated the metal-binding capacity of three synthetic zeolites, 4A,
P, and Y. Each displayed a high affinity for Cd, Cu, Pb, and Zn at low solution concentrations. Greenhouse pot experiments using contaminated soils demonstrated the ability of these synthetic zeolites to reduce the phytotoxicity for maize (Rebedea et al., 1994).

More than 30 years ago, Ames (1959) demonstrated the high selectivity of clinoptilolite for Cs. Certain zeolites also display high ion exchange selectivity for Sr (Chelishchev, 1973, 1995), even in the presence of Ca and Mg (Tsitsishvili et al., 1992). Chelishchev and others (Chelishev and Chelishcheva, 1980; Vaniman and Bish, 1995) demonstrated that the presence of clinoptilolite can reduce plant uptake of Cs and Sr. The fact that isotopes of Cs and Sr represent much of the radioactivity released at Chernobyl (Kroogl, 1990) has

<table>
<thead>
<tr>
<th>Country</th>
<th>Treatment</th>
<th>Soil pH</th>
<th>Cd (mg kg⁻¹)</th>
<th>Pb (mg kg⁻¹)</th>
<th>Zn (mg kg⁻¹)</th>
</tr>
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<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>Leaves</td>
<td>Roots</td>
<td></td>
</tr>
<tr>
<td>Canada</td>
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<td>3.30a</td>
<td>4.25a</td>
<td>104a</td>
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<tr>
<td></td>
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<td></td>
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<td>0.91c</td>
<td>1.92b</td>
<td>79.2b</td>
</tr>
<tr>
<td>Czech Rep.</td>
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<td>5.41</td>
<td>11.8a</td>
<td>9.50a</td>
<td>3188a</td>
</tr>
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<td></td>
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<td>5.46</td>
<td>6.55b</td>
<td>7.25b</td>
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</tr>
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<td>5.93c</td>
<td>7.20b</td>
<td>1874c</td>
</tr>
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<td>3.73a</td>
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</tr>
<tr>
<td></td>
<td>Zeolite I</td>
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<td>2.12c</td>
<td>1.70b</td>
<td>103b</td>
</tr>
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<td></td>
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<td>1.82b</td>
<td>1.35c</td>
<td>96.7b</td>
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<tr>
<td>Taiwan</td>
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<td>19.3a</td>
<td>44.0a</td>
<td>26.2a</td>
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<tr>
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<td>17.7b</td>
<td>12.3b</td>
<td>22.8b</td>
</tr>
<tr>
<td></td>
<td>Zeolite II</td>
<td>6.36</td>
<td>14.6c</td>
<td>8.23d</td>
<td>22.3b</td>
</tr>
</tbody>
</table>

Note: Rates of amendments: I — 25 g kg⁻¹, II — 50 g kg⁻¹; means followed by letters a, b, c, d, e are significantly different at P < 0.05.

Yield of 6-week-old maize in the contaminated soils with two levels of zeolite (phillipsite): I — 25 g kg⁻¹, II — 50 g kg⁻¹; means followed by letters a, b, and c are significantly different at P<0.05 (Reprinted with permission from Chlopecka, A. and D.C. Adriano, Inactivation of metals in polluted soils using natural zeolite and apatite, in Proc. Extended Abstracts from the Fourth Int. Conf. on the Biogeochemistry of Trace Elements, Berkeley, CA, 415, 1997c.)
heightened interest in the use of clinoptilolite as a soil amendment. Selectivity for Cs and Sr is generally highest under slightly acidic conditions (pH 5.0–6.0), but drops off dramatically at lower pHs due to competition with hydrogen for exchange sites (Tsitsishvili et al., 1992). The presence of strong ligands, such as the synthetic chelate EDTA or citrate and tartrate, can also reduce Cs and Sr sorption (Tsitsishvili et al., 1992). Other common metals, such as Ca and K, may effectively compete with the target contaminant for sorption sites in some instances (Leppert, 1990).

### 2.2.3 Apatite

The use of apatite minerals as a remediation strategy is based on recognized geochemical principles. Apatite minerals form naturally and are stable across a wide range of geologic conditions (Nriagu, 1974; Wright, 1990). Wright et al. (1987) investigated the trace element composition of apatite in fossil teeth and bones and in sedimentary phosphate deposits through geologic time. They found that apatite deposited in seawater adsorbs metals and radionuclides from the seawater to millions of times the ambient concentration. The metals remain within the apatite structure indefinitely with little subsequent desorption, leaching, or exchange, even in the face of subsequent digenetic changes in the pore water chemistry, pH, or temperatures up to 1000°C. Apatite minerals act as natural collectors for metals and radionuclides. Apatite deposits in Florida, for example, have accumulated large amounts of uranium (U), enough in fact to be considered a commercial source of uranium (Eisenbud, 1987). A younger apatite deposit located in North Carolina is mined primarily for fertilizer. Young deposits have had less exposure to metals in the environment and are, therefore, generally more reactive.

Remediation studies on metal-contaminated wastes and soils using apatite or hydroxyapatite have focused mainly on Pb (Berti and Cunningham, 1997; Chen et al., 1996, 1997; Laperche et al., 1997; Ruby et al., 1994; Zhang et al., 1997). Ma et al. (1993, 1994, 1995) reported that before hydroxyapatite can be successfully used as a Pb-immobilizing material, three factors need to be considered. Hydroxyapatite must immobilize Pb²⁺ in the presence of interfering cations, anions, and dissolved organic matter; the resulting products must be stable in the contaminated environment, and the reaction should be rapid.

Current research demonstrates the successful precipitation not only of Pb, but also other metals when phosphate minerals are added to a contaminated medium (Misra and Bowen, 1981; Ma et al., 1993; Xu and Schwarz, 1994). LeGeros and LeGeros (1984) showed that there are three types of substitutions that can occur in hydroxyapatite or hydroxypyrromorphite structures. The cations Pb²⁺, Ba²⁺, Zn²⁺, Fe³⁺, and Mg²⁺ can substitute for Ca²⁺, while the oxyanions AsO₄³⁻, VO₄³⁻, CO₃²⁻, and SO₄²⁻ can replace structural PO₄³⁻. Additionally, anions such as F⁻ and Cl⁻ can substitute for OH⁻ in the apatite structure. Ma et al. (1994) showed that hydroxyypromorphite [Pb₅(PO₄)₂(OH)] precipitated after the reaction of hydroxyapatite with Pb²⁺ in the presence of NO₃⁻, SO₄²⁻, and CO₃²⁻, while chloropyromorphite [Pb₅(PO₄)₂Cl] and fluoroypromorphite [Pb₅(PO₄)₂F] formed in the presence of Cl⁻ and F⁻, respectively.

The sorption mechanisms are variable in the reaction between the apatite mineral and Pb, Cd, and Zn from contaminated soils. Lead removal results primarily from the dissolution of apatite followed by the precipitation of hydroxyl fluoropyromorphite. Minor otavite precipitation was observed in the interaction of the apatite with aqueous Cd, but other sorption mechanisms, such as surface complexation, ion exchange, and the formation of amorphous solids, are primarily responsible for the removal of aqueous Zn and Cd (Wright et al., 1995).

Other researchers found that the pH under which a reaction between metals and apatite occurs plays an important role. Wright et al. (1995) reported that the immobilization of Pb was primarily through a process of apatite dissolution followed by precipitation of various
pyromorphite-type minerals under acidic condition, or the precipitation of hydrocerussite \([\text{Pb}_3(\text{CO}_3)_2(\text{OH})_2] \) or \(\text{Pb}(_2\text{OH})_2 \cdot 2\text{PbCO}_3\) and lead oxide fluoride \((\text{Pb}_2\text{OF}_2)\) under alkaline conditions. Otavite \((\text{CdCO}_3)\) and cadmium hydroxide \((\text{Cd(OH)}_2)\), and zincite \((\text{ZnO})\) were formed in the Cd or Zn system, respectively, especially under alkaline conditions; while hopeite \([\text{Zn}_3(\text{PO}_4)\cdot 4\text{H}_2\text{O}]\) might only precipitate under alkaline conditions. Alternative sorption mechanisms other than precipitation were important in immobilizing Cd and Zn in the presence of apatite. The selectivity order of heavy metal sorption by apatite depends on pH. The selectivity order at pH below 7 was Pb>Cd>Zn, but at higher pH it was Pb>Zn>Cd. Removal of Pb, however, is less sensitive to pH (Wright et al., 1995).

Saeed and Fox (1979) studied the influence of phosphate fertilization on Zn adsorption by tropical soils and found that this component increased Zn adsorption in variable charge soils, suggesting that phosphate addition increases the negative charge on iron and aluminium oxides. However, for less-weathered soils, prior phosphate applications decreased Zn sorption because the fertilizer contained Zn as an accessory element.

Recent studies on the stabilization of metals by phosphate compounds have focused on the reduction of plant uptake, as well as the reduction in solubility/mobility. Chlopecka and Adriano (1997b, c), Knox (1998b), and Knox and Adriano (1999) found that apatite from North Carolina, consisting mainly of hydroxyapatite or fluorapatite, effectively immobilized Pb, Cd, and Zn in contaminated soils and decreased plant uptake by several crop species. In these studies, apatite \((25\text{ and } 50\text{ g kg}^{-1})\) addition to the contaminated soils significantly reduced the potential mobility of Cd, Zn, and Pb, as indicated by sequential extraction (Figure 2.5). For example, the concentration of Zn in the exchangeable fraction was reduced from 272 to 126 mg kg\(^{-1}\) at the highest apatite addition rate. The exchangeable fraction of Cd and Pb was reduced by 55 and 60%, respectively, at the highest apatite application rate, 50 g kg\(^{-1}\). Data from this study clearly show partitioning of the contaminant metals to the residual fraction, with the highest increase observed for Pb (Figure 2.5). Apatite significantly improved plant growth and yield on highly contaminated soils (Figure 2.6). Also, concentrations of these metals in plant tissues like leaves or roots significantly decreased. For example, the Cd concentration in maize leaves decreased from 11.8 mg kg\(^{-1}\) to 5.8 and 4.8 mg kg\(^{-1}\), respectively, in the control treatment (contaminated soil from Czech Republic) and treatments with 25 and 50 g apatite kg\(^{-1}\). Reduction of Zn concentrations in these tissues was higher than for Cd \((50, 59\% \text{ and } 51, 64\%, \text{ respectively}, \text{ for first and second dose of apatite for Cd and Zn})\) (Figure 2.7) (Chlopecka and Adriano, 1997c). In maize leaves the lowest reduction was obtained for Pb, 41 and 44% for first and second doses of apatite. Generally, the reduction in metal concentrations in maize roots was about 10 to 20% lower than for maize leaves (Chlopecka and Adriano, 1998). Apatite affects not only heavy metal concentrations in plants, but also essential micronutrients. Several studies have demonstrated that increasing addition of apatite to the soil decreases the concentration of P, Mn, and Fe in the plant tissues and generally increases the concentration of Ca and Mg (Boisson et al., in press; Grant and Bailey, 1992). Other studies have shown that heavy metal concentrations are not reduced in all plant tissues. Laperche et al. (1997) found the Pb content in shoot tissue decreased with increasing apatite addition. However, Pb and P contents in the plant roots increased as the quantity of added apatite increased. The authors hypothesized that Pb accumulates in/on the roots because it precipitates as lead phosphate. Without phosphate, Pb is readily translocated from roots to shoots, with similar Pb contents observed in both shoot and root tissues in unamended soils. This study also strongly suggests that, in the absence of other phosphate sources, plants can induce the dissolution of pyromorphite to facilitate P uptake. To prevent the release of Pb due to pyromorphite dissolution, soil-P levels for plants must be maintained in excess of that needed to immobilize Pb.

Arey et al. (1999) demonstrated the effectiveness of apatite addition in reducing uranium (U) solubility and TCLP-extractability in contaminated sediments from the Department of
Energy’s Savannah River Site, Aiken, SC, as well as several other contaminant metals, including Pb and Cd (Figure 2.8). However, complexation of U by dissolved organic carbon (DOC) in materials containing higher organic matter slightly decreased the effectiveness of apatite addition, presumably by lowering its free-ion activity in solution and thereby increasing U solubility for a given phosphate level.