Metal fractionation in a contaminated soil after reforestation: Temporal changes versus spatial variability

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Abstract

In a lysimeter experiment, topsoils were polluted with filter dust from a non-ferrous metal smelter and then planted with trees. Sequential extractions were used to follow the changes in metal fractionation of Cu, Zn, Cd, and Pb over 42 months. Plant-free and uncontaminated soils served as reference. In the contaminated and planted soils, the largest changes in speciation occurred within the first 6 months. The relative amounts of certain metal fractions were linearly related to each other, indicating systematic redistribution between fractions. The results indicate that under natural conditions with high heterogeneity in total metal contents spatial differences are more important than temporal variations in determining the fractionation and solubility of metals in contaminated soils. In the absence of plants soils exhibited a completely different fractionation 30 months after pollution, with much higher proportions in the more refractory phases. This suggests that plant activity kept the metals in a more soluble form.

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1. Introduction

Soils represent an important sink for metals released into the environment by human activities, e.g. through industry, mining, agriculture, and traffic (Alloway, 1995). On one hand, metals entering soils may – often adversely but sometimes also beneficially – affect soil fertility. On the other hand they may be leached into groundwater, exported with surface runoff or erosion into surface water or may enter food chains. Particularly in the initial stages of a contamination, the mobility, reactivity and bioavailability of polluting metals can strongly depend on the form in which they entered the soil (Birkefeld et al., 2006). Knowledge about phase transformations and their kinetics during these stages can be crucial for the assessment of the exposure arising from the pollution even in the long-term (Dudka and Adriano, 1997).

Metals added to a soil in soluble form will redistribute between the soil’s liquid and solid phases over time (Lock and Janssen, 2003). In general, they will rapidly be transferred into exchangeable or easily mobilizable fractions, while redistribution into more crystalline or even residual fractions will occur over time scales of months or even years (Sposito et al., 1983; Mandal and Mandal, 1986). In various studies such redistribution processes have been monitored over a few weeks (Ma and Uren, 1998; Sinaj et al., 1999; Lu et al., 2005), up to months (Almas et al., 1999; Han and Banin, 1999), but very rarely over years (Gunkel et al., 2002). All these studies found that metal solubility decreased, while the most refractory fractions increased over time. Spectroscopic investigations have shown that the formation of crystalline metal phases (e.g. phyllosilicates or layered double hydroxides) occurred within a few months in soils with a near-neutral pH (Sparks, 2000).

A widely used technique to characterize the binding of elements in soils is sequential extraction. Many different sequential extraction schemes have been proposed. They all are based on a series of reagents successively applied to solubilize step by step metal fractions differing in their binding to the various soil constituents (Tessier et al., 1979; Zeien and Brümmer, 1989). The major shortcoming of this methodology is that the extracted fractions are only operationally defined and are often difficult to be interpreted in terms of chemical species or binding forms (Gleyzes et al., 2002). Nevertheless, sequential extractions have provided valuable insights into the variation of metal binding characteristics among soils, their changes over time and the effects of soil treatments.

There are only few studies in which the influence of plants and their associated microorganisms on metal leaching from contaminated soils was investigated under controlled conditions. Römkens et al. (1999) compared soil Cu speciation in pots with and without

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plants and found much higher Cu solubility in the planted pots. Turpeinen et al. (2000) on the other hand observed that pine seedlings reduced Pb solubility by up to 93%. Conflicting results were also found in column and lysimeter experiments. In column studies, in which either effluent was collected at the bottom of the column (Track et al., 1998; Keller et al., 2002) or soil solution was sampled by means of suction cups at various depths (Jacob and Otte, 2004; Zhao et al., 2007), plants were found to exhibit both mobilizing and immobilizing effects on metals in the root zone. These effects depended on the plant species, the metal, the soil properties and the experimental setup. Using soil lysimeters spiked with sewage sludge, McLaren et al. (2004) found slightly larger concentrations of Cd and Zn in the leachate from bare forest soils than from grass covered pasture soils and ascribed this to larger metal uptake by the grass.

The aim of this work was to monitor the development of metal binding in a factorial lysimeter experiment investigating potential phytostabilization of a metal-contaminated agricultural topsoil by afforestation under quasi-field conditions (Menon et al., 2005; Nowack et al., 2006; Brunner et al., 2008; Luster et al., 2008). Phytostabilization is a remediation strategy in which vegetation is used to stabilize a contaminated soil against erosion and to prevent contaminant export by surface runoff and also to control contaminant dispersal through food chains (Robinson et al., 2006). In addition, uptake of water and elements by plant roots can help to keep polluting metals in the root zone and, thus, reduce their leaching to the groundwater. On the other hand, vegetation may also have effects that increase metal mobility and bioavailability, in particular through chemical changes in the rhizosphere due to root exudation and respiration and through direct chemical or indirect microbial effects induced by the decomposition of leaf and root litter. Such effects need to be understood in order to make phytostabilization a viable long-term option for the management of metal-contaminated soils.

2. Materials and methods

2.1. Description of the lysimeter experiment

A factorial lysimeter experiment was performed in which model forest ecosystems were subjected to combinations of the following parameters: metal-contaminated versus uncontaminated topsoil, uncontaminated acid subsoil versus calcareous subsoil and acid versus ambient rain. Treatments (2 × 2 × 2) were replicated 4 times each in a total of 16 open-top chambers (OTC) arranged in a Latin square. Each OTC hosted one lysimeter with acid subsoil and one lysimeter with calcareous subsoil.

The lysimeters had a surface area of 3 m² and a depth of 1.5 m. At the bottom was a 50 cm drainage layer consisting of three sublayers of fire dried quartz gravel (grain size 5–8, 1.5–2.2, and 0.7–1.2 mm). The soil above the drainage layer was composed of an 80 cm subsoil layer and a 15 cm topsoil layer. The topsoil was a weakly acid loam collected from an agricultural field at Birr, Kanton Aargau, Switzerland. Some physical and chemical characteristics of the topsoil are given in Table 1. For more details on the soils, their preconditioning and the filling of the lysimeters see Luster et al. (2008).

<table>
<thead>
<tr>
<th>Texture (% sand; silt; clay)</th>
<th>pH (0.01 M CaCl2)</th>
<th>CEC [mmolc/kg]</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>36; 49; 15</td>
<td>6.55</td>
<td>15.1</td>
<td>Nowack et al., 2006</td>
</tr>
<tr>
<td>Corg [g/kg]</td>
<td>&lt;1</td>
<td></td>
<td>Nowack et al., 2006</td>
</tr>
<tr>
<td>Corg [g/kg]</td>
<td>1.2</td>
<td></td>
<td>Nowack et al., 2006</td>
</tr>
<tr>
<td>CEC [mmolc/kg]</td>
<td>102</td>
<td></td>
<td>Nowack et al., 2006</td>
</tr>
</tbody>
</table>

Control | Contaminated
Cu (mg/kg) | 28 ± 4 | 588 ± 206 | this study |
Zn (mg/kg) | 97 ± 3 | 2854 ± 872 | this study |
Cd (mg/kg) | 0.1 | 0.2 ± 0.4 | this study |
Pb (mg/kg) | 37 ± 3 | 103 ± 27 | this study |

In the treatments with added metals, filter dust from a non-ferrous metal smelter containing approximately 170 mg kg⁻¹ Cu, 800 mg kg⁻¹ Zn, 0.27 g kg⁻¹ Cd, and 15.5 g kg⁻¹ Pb was manually mixed into the topsoil by means of a barrow in April 2000. In May 2000, each lysimeter was planted with 14 trees (6 Picea abies (L.) Karst., 4 Populus tremula L., 2 Salix viminalis L. and 2 Betula pendula Roth.) and understorey plants (for details see Dumle et al. (2007)). The leaves of the trees were harvested each autumn and therefore no litter layer was formed on the soil (Scheid et al., 2009).

During the vegetation period (May through October), the roofs of the OTC closed automatically during rain events to exclude natural precipitation. The lysimeters were irrigated with an aqueous solution mimicking either ambient or acid rain (pH 3.5). The ionic composition of the “ambient rain” was (in mM): 70 NH₄, 70 NO₃, 3.2 PO₄, 17 Cl, 3.1 SO₄, 4.3 Na, 7.7 K, 5 Ca, 1.3 Mg, 0.15 Zn; the pH was adjusted with HCl. During winter (November through April) the roofs were open all the time, and the lysimeters received natural precipitation.

2.2. Sampling

Shortly before the trees were planted in May 2000, the topsoil (15 cm) of the contaminated lysimeters was sampled by taking 8 samples at random from each lysimeter and bulking them into one composite sample per lysimeter. Further topsoil samples were taken in November each experimental year (2000, 2001, 2002, and 2003) from the contaminated lysimeters with “acid rain” irrigation, using a cylindrical sampler (diameter 5 cm; length: 15 cm), taking three samples at a distance of about 20–30 cm from the three spruce trees per lysimeter. The locations of the sampling points shifted by about 45° around the spruce trees from sampling to sampling. The three samples from each lysimeter were pooled for analysis. Additional samples were taken in 2001 from the uncontaminated soils and in 2003 from the “ambient rain” treatment. All samples were dried at 40 °C and sieved to 2 mm.

As plant-free control treatment, four soil columns made of stainless steel were installed in the vicinity of the OTC at a shady place. The columns were 145 cm high and 78 cm wide. Prior to filling the columns with soil, the inside part was lined with a polyethylene foil in order to avoid direct contact with the steel walls. The same materials (quartz sand, subsoil and topsoil) were filled into the columns in the same way as for the OTC lysimeters. Irrigation was applied only as synthetic “ambient” rain (pH 5.5). Otherwise the irrigation regime was the same as for the OTC lysimeters. A roof excluded natural precipitation during the vegetation period. During winter, the roof was removed and only natural precipitation irrigated the soil columns. The columns were operated from 2003 until 2006. Topsoil samples were taken in March 2006 from the centre of the columns in order to minimize edge effects.

2.3. Sequential extraction procedure

The sequential extraction scheme of Zeien and Brümmer (used to fractionate soil metals (Zeien and Brümmer, 1989). The extraction was carried out progressively in duplicate for each of the four replicates samples collected per treatment at each sampling date using 2.0 g subsamples. The extrants and the extractants for the operationally defined fractions are given in Table 2. All suspensions were centrifuged at 2500 rpm for 10 min and passed through Schleicher & Schull 790 1/2 paper filters before they were analyzed. The metals (Cu, Zn, Cd, Pb, Fe, Mn) in the filtrates from F1–F6 were measured by flame atomic absorption spectrometry (SpectraAA 220/FS, Varian). The residual fraction (F7) was analyzed by X-ray fluorescence spectrometry (Spectro X-Lab 2000).

2.4. Statistical analysis

Two-sample t-tests were performed to elucidate significant differences between treatments and an ANOVA to detect significant temporal trends. For all tests, DataDesk 6.2.1 for Mac was used.

Table 2

<table>
<thead>
<tr>
<th>Fraction</th>
<th>Chemical interpretation</th>
<th>Extraction conditions</th>
<th>Duration</th>
</tr>
</thead>
<tbody>
<tr>
<td>F1</td>
<td>mobile</td>
<td>1 M NH₄NO₃</td>
<td>24 h</td>
</tr>
<tr>
<td>F2</td>
<td>easily mobilizable</td>
<td>1 M NH₄-acetate (pH 6)</td>
<td>24 h</td>
</tr>
<tr>
<td>F3</td>
<td>bound to Mn oxides</td>
<td>0.1 M NH₄OH–HCl + 1 M NH₄-acetate (pH 6)</td>
<td>30 min.</td>
</tr>
<tr>
<td>F4</td>
<td>bound to organic matter</td>
<td>0.025 M NH₄–EDTA (pH 4.6)</td>
<td>90 min.</td>
</tr>
<tr>
<td>F5</td>
<td>bound to amorphous</td>
<td>0.2 M Na₂O₃–oxalate (pH 3.25)</td>
<td>4 h in the dark</td>
</tr>
<tr>
<td>F6</td>
<td>bound to crystalline</td>
<td>0.1 M ascorbic acid + 0.2 M NH₄–oxalate (pH 3.25)</td>
<td>30 min.</td>
</tr>
<tr>
<td>F7</td>
<td>residual fraction</td>
<td>X-ray fluorescence</td>
<td>at 96 °C</td>
</tr>
</tbody>
</table>

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3. Results and discussion

3.1. Total contents

The contaminated soil samples displayed a large small-scale heterogeneity in the spatial distribution of the metals (Table 1). The observed ranges were 231–1219 mg kg⁻¹ for Cu, 1390–5176 mg kg⁻¹ for Zn, 4–22 mg kg⁻¹ for Cd and 61–164 mg kg⁻¹ for Pb. Because the source was the same for all metals added, it is no surprise that close linear correlations were found between their total contents, even 42 months after addition (Fig. 1). The ratios of Cu, Cd and Pb to Zn in the added filter dust were 0.21 for Cu, 0.0033 for Cd and 0.019 for Pb. The respective ratios in the soil samples were 0.18 for Cu, 0.0026 for Cd and 0.026 for Pb. The Pb:Zn ratio was significantly higher in the soil than in the dust, because the metal added with the dust increased the Pb value compared to the control soil only by a factor of 3–4, whereas Cu, Zn, and Cd concentrations were increased by the addition of the dust by factors of 10–30 for Cu, 20–50 for Zn, and 50–200 for Cd. This explains why the linear relationship between Pb and Zn has an intercept that is significant compared to the total amount. Overall, the observed soil metal ratios agreed with the expected ones.

The small-scale heterogeneity in the chambers resulted from the manual incorporation of the surface-applied filter dust into the topsoil by harrowing. Small-scale variations of heavy metal contents in polluted soils have been reported e.g. from the area of Dornach, Switzerland, where Cu, Zn and Cd contents in topsoil sampled from a meadow in the immediate vicinity of a metal smelter varied within 0.5 m by a factor of up to 3 (Keller et al., 1999), similar to the soil samples in this study. High micro-scale variability in metal contents of soils polluted by atmospheric deposition has also been observed in other places, e.g. in forests around metal smelters (Wopereis et al., 1988; Arrouays et al., 1996; MacDonald and Hendershot, 2003). Thus, the high small-scale spatial variability of total heavy metals contents in our soil is in the normal range of polluted soils.

Table 3 presents the average values at each sampling for pH and the total contents of the four studied metals. While there was a highly significant decrease in pH, occurring mainly during the first 1.5 years, ANOVA showed that there was a significant decrease only in Cu and Pb contents over time, whereas Zn and Cd showed no such effect. A higher leaching loss of Cu and Pb compared to Zn and Cd concomitant with a strong decrease in pH may be explained by a dominance of metal transport bound to dissolved or particulate organic matter in these near-neutral soils.

3.2. Fractionation of metals

The large short-range variation of total metal contents allowed us to follow the effect of different total metal contents on the extractability over time under otherwise identical conditions. Fig. 2 shows the observed changes in the average distribution of the sequential extraction fractions for Cu, Zn, Cd, and Pb in the lysimeter
Table 4

Mean values of percent extractable metal contents after 42 months for different treatments (2-sample t-test, alpha = 0.05, means of replicates per chamber, n = 4).

<table>
<thead>
<tr>
<th>Fraction</th>
<th>Acid rain</th>
<th>Neutral rain</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Mean %</td>
<td>Std.Dev. %</td>
</tr>
<tr>
<td>a) Cu</td>
<td></td>
<td></td>
</tr>
<tr>
<td>F1</td>
<td>7.9</td>
<td>2.6</td>
</tr>
<tr>
<td>F2</td>
<td>33.0</td>
<td>7.8</td>
</tr>
<tr>
<td>F3</td>
<td>15.1</td>
<td>0.7</td>
</tr>
<tr>
<td>F4</td>
<td>28.4</td>
<td>5.8</td>
</tr>
<tr>
<td>F5</td>
<td>10.7</td>
<td>2.8</td>
</tr>
<tr>
<td>F6</td>
<td>1.9</td>
<td>0.7</td>
</tr>
<tr>
<td>F7</td>
<td>3.1</td>
<td>1.4</td>
</tr>
<tr>
<td>b) Zn</td>
<td></td>
<td></td>
</tr>
<tr>
<td>F1</td>
<td>30.7</td>
<td>10.3</td>
</tr>
<tr>
<td>F2</td>
<td>36.4</td>
<td>12.5</td>
</tr>
<tr>
<td>F3</td>
<td>8.4</td>
<td>1.3</td>
</tr>
<tr>
<td>F4</td>
<td>9.9</td>
<td>1.0</td>
</tr>
<tr>
<td>F5</td>
<td>6.3</td>
<td>1.7</td>
</tr>
<tr>
<td>F6</td>
<td>3.6</td>
<td>1.3</td>
</tr>
<tr>
<td>F7</td>
<td>4.7</td>
<td>1.4</td>
</tr>
<tr>
<td>c) Cd</td>
<td></td>
<td></td>
</tr>
<tr>
<td>F1</td>
<td>36.8</td>
<td>3.5</td>
</tr>
<tr>
<td>F2</td>
<td>49.3</td>
<td>5.7</td>
</tr>
<tr>
<td>F3</td>
<td>9.4</td>
<td>2.3</td>
</tr>
<tr>
<td>F4</td>
<td>4.4</td>
<td>1.5</td>
</tr>
<tr>
<td>F5</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>F6</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>F7</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>d) Pb</td>
<td></td>
<td></td>
</tr>
<tr>
<td>F1</td>
<td>1.3</td>
<td>0.5</td>
</tr>
<tr>
<td>F2</td>
<td>9.7</td>
<td>5.1</td>
</tr>
<tr>
<td>F3</td>
<td>19.8</td>
<td>3.6</td>
</tr>
<tr>
<td>F4</td>
<td>38.7</td>
<td>2.6</td>
</tr>
<tr>
<td>F5</td>
<td>14.9</td>
<td>4.2</td>
</tr>
<tr>
<td>F6</td>
<td>2.9</td>
<td>1.9</td>
</tr>
<tr>
<td>F7</td>
<td>12.3</td>
<td>2.9</td>
</tr>
</tbody>
</table>

Fig. 2. Partitioning of Cu, Zn, Cd and Pb among the seven fractions determined by means of sequential extraction using the scheme of Zeien and Brümmer, (1989). Each column corresponds to the average of 8 replicates.

top soils over the 42-month monitoring period. There were no significant differences in metal fractionation between the ambient and the acid-rain treatments (2-sample t-test, alpha = 0.05, means of replicates per chamber, n = 4; see Table 4), indicating that over three years the pH of the rain had no influence on the distribution of the metal fractions in the soil. The fractionation of all four metals differed substantially between the uncontaminated and the contaminated soil. For Cu, Zn and Pb the uncontaminated soil had much larger proportions of more refractory phases and only small percentages of easily soluble metals. For Cd the uncontaminated soil contained only the fraction F1 whereas in the contaminated soil also F4—F2 were present. However, one has to note that the residual fraction could not be determined for the uncontaminated soil because its total Cd content (0.1 mg kg⁻¹) was below the detection limit of the XRF method used for the sequential extraction. In the first six months after the incorporation of the dust strong changes in metal fractionation occurred for Cu and Zn. For Zn there was a decrease in the most soluble fraction, for Cu a decrease in the most refractory fraction. Only small changes occurred in the contaminated soil from 6 to 42 months. Cu exhibited a decreasing trend of the most soluble fraction and an increasing trend for the more refractory fractions.

Drainage water monitoring indicated that 9 months after filling the soils had reached chemical equilibrium conditions in the lysimeter (Luster et al., 2008). The initial stage was governed by dissolution of the filter dust containing mainly zincite (ZnO) and brass (Cu—Zn alloy) (Luster et al., 2008). The ZnO dissolved rapidly in the soil, as revealed by XAFS spectroscopy (Voegelin et al., 2005),
suggesting that the high availability of Zn in the initial sample was due to the fact that most of the easily soluble Zn was still present as ZnO. Most dissolution, adsorption of the dissolved metal onto the solid phase and formation of new phases took place within the first 6 months.

We normally expect that contaminating metals are initially present in easily available fractions and then redistribute to more refractory fractions (Lock and Janssen, 2003). However, for Cu we observed the opposite trend. Initially, there was more Cu in fractions 5–7 than after 6 months. The added Cu was mainly contained in brass particles, which are only dissolved under harsh conditions. Slow alteration of these particles could thus explain the slow increase in easily soluble Cu with time. In most other studies about changes in metal speciation, the metals were added as soluble compounds that did not have to be dissolved first.

3.3. Relations between different fractions

The large spatial variability in metal contents allowed us to derive relationships between different metal fractions. Many fractions showed a clear dependence of their relative magnitude on total soil metal contents. Fig. 3 summarizes the most important of these relationships for Cu, Zn, and Pb. Copper fractions F1 and F2 increased with increasing total Cu content, F3 remained constant and the other fractions decreased (only shown for F4 and F5). Fig. 5 shows that F1 was inversely related to F4 and F2 to F4 and F5. F4 was positively correlated to F5.

Zinc fraction F1 slightly decreased, while F2 increased with increasing total Zn content. The inverse relationship between Zn fractions F1 and F2 is shown in Fig. 4. The other Zn fractions either remained constant or decreased slightly. The sum of the two fractions F1 and F2 remained constant. A similar behavior has been observed for acidic and calcareous mine tailings (Conesa et al., 2008), where F1 was much larger in the acidic compared to the neutral tailing while the sum of both showed little variation.

Lead fractions F2 and F3 increased with increasing total Pb contents, while the other Pb fractions slightly decreased as shown for F4 in Fig. 3.

3.4. Reduced partition index

In order to quantify the relative binding intensity of metals in soils derived from sequential extractions, a so-called "reduced partition index" \( I_R \) has been proposed (Han et al., 2003). Here we used the following definition for \( I_R \):

\[
I_R = \frac{1}{k^2} \sum_{i=1}^{k} F_i^2
\]

where \( i \) is the index number of the extraction step, progressing from 1 for the weakest to \( k \), the most aggressive extractant (\( k = 7 \) in our case), and \( F_i \) is the relative content of the considered metal in fraction \( i \). The maximum value \( I_R \) can have is 1. It means that all the metal is present in fraction \( F_i \). Low values of \( I_R \) represent distribution patterns in which most of the metal resides in the most soluble fractions, whereas high values result from a high proportion in the more residual fractions. Fig. 6 shows the \( I_R \) of Cu, Zn, and Pb as...
functions of the respective total soil metal content. For all three metals we can observe a clear decrease in $I_R$ with increasing total metal content. In particular, much higher values were found for the uncontaminated soil than for the contaminated soil.

3.5. Comparison between planted and plant-free soil

Metal fractionation was markedly different in the plant-free and planted soils (Fig. 2). For all four metals the most soluble fractions (F1 and F2) were larger in the planted soil, while for Cu, Zn and Cd the more refractory phases (F4–F7) were larger in the plant-free soil, as also indicated by a higher reduced partition index. These differences are especially noteworthy for Zn and Cd, where fraction F5–F7 were much more important in the plant-free soil compared to the planted one.

Also in the planted soil, the main changes of metal fractionation occurred in the first 6 months. At this time the trees had still not grown much and also their root systems were not much developed yet. In the following years, only small changes occurred in the speciation of metals. There have been several indications in the literature that the presence of plants increases dissolved metal concentrations in soil (Römkens et al., 1999; Zhu et al., 1999). Metals incorporated into fine roots will be extracted primarily with the most soluble fractions in sequential extraction schemes. However, this contribution was very small in our case according to results of Brunner et al. (2008), who found that metals associated with fine roots made up only 0.03–0.2% of the metals in the soils of our experiment. Metal speciation may have been affected also indirectly by the treatments through changes in microbial activity. In fact, Frey et al. (2006), who investigated the same soils as described here, found large differences in microbial activity between the contaminated and uncontaminated soils. In summary, our results indicate that plant root activity, directly or indirectly via effects on the microbial communities, mobilized soil metals, counteracting the tendency of the metals to partition into more refractory phases. However, it should be kept in mind that metal mobilizing or immobilizing processes occurring in or due to plant litter are not considered in our study because of the litter removal each autumn.

4. Conclusions

Overall, we found that under natural conditions with high heterogeneity in total metal contents spatial differences were more important than temporal variations in determining the fractionation and solubility of metals in contaminated soils.

Both the relations between different fractions and the reduced partition indices indicated increasing trends of more soluble fractions with increasing total metal contents. This can be explained by saturation of binding sites on minerals and organic matter and higher contents of precipitates and newly formed solid phases. The observed effect of plants to keep metals in soils in more soluble fractions would be contra-productive in a phytostabilization scheme, counteracting, to some degree at least, the beneficial effects
of reducing deep seepage and surface water runoff (Robinson et al., 2006). Thus, although metal concentrations in drainage water collected from the lysimeters were governed by the subsols and no increased leaching of metals was observed under contaminated topsoils (Luster et al., 2008), phytoextraction may have to be complemented by amendments for the chemical immobilization of polluting metals in the topsoil.

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