Phytomanagement: Phytoremediation and the Production of Biomass for Economic Revenue on Contaminated Land

9

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9.1 Introduction: From Remediation to Management of Trace Element-Contaminated Land

The term phytoremediation refers to the use of plants and associated microorganisms to eliminate environmental damage or threats posed by environmental pollution. While this includes the use of plants in soil conservation such as protection against erosion or regeneration of compacted soils, the term phytoremediation is primarily used in conjunction with the decontamination/redevelopment of soils, which are contaminated by pollutants. Raskin et al. (1997) defined phytoremediation as the use of green plants to remove pollutants from the environment or to render them harmless. Phytoremediation may be applied to soils that are contaminated with toxic trace elements (TE) or organic pollutants. Depending on the targeted pollutants and the mechanisms, phytoremediation can be divided into: (a) phytoextraction, (b) phytotransformation, and (c) phytostabilization. This chapter will focus on TE as they are the most widespread and intractable soil contaminants.

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Phytoextraction describes the use of plants to remove/extract pollutants from soil. In 1583 the botanist Cesalpino, the author of the renowned book *De plantis Libri, which* contains descriptions of about 1,500 plant species, described an "alyson" that appeared to be confined to serpentine soils, which are rich in nickel (Ni), in the vicinity of Florence, Italy. In 1885, Baumann, a German botanist working near the border of Germany and Belgium, discovered that certain plant species growing on soils naturally enriched in Zn were capable of accumulating uncommonly high Zn levels. Brooks et al. (1977) coined the term hyperaccumulator for plants that accumulate >1,000 mg kg^{-1} Ni on a dry matter basis. Currently, a plant is defined as a hyperaccumulator if it reaches concentrations of at least 100 mg kg^{-1} (0.01 % dry wt.) Cd and As; 1,000 mg kg^{-1} (0.1 % dry wt.) Co, Cu, Cr, Ni, and Pb; and 10,000 mg kg^{-1} (1 % dry wt.) Mn and Zn in their aboveground tissues (Reeves and Baker 2000; Watanabe 1997). To date more than 500 plant species have been identified as natural metal hyperaccumulators, representing <0.2 % of all angiosperms most of which are Ni hyperaccumulators (450 species) (Ent et al. 2013).

In 1993, McGrath et al. proposed that hyperaccumulators could be used for the removal of TE pollutants from soil. Unfortunately, most hyperaccumulator species are slow growing and have limited biomass production. As total metal extraction is the product of biomass and tissue concentration, the speed of metal removal is accordingly limited. Field experiments by Robinson et al. (1998), Lombi et al. (2000), and McGrath et al. (2000) highlight this problem, showing that metal removal efficiency is in general not high enough to remediate contaminated soils. Subsequently, research focused more on high biomass plants, such as tobacco (*Nicotiana tabacum*) (Evangelou et al. 2004; Fässler et al. 2010; Kayser et al. 2000), maize (*Zea mays*) (Fässler et al. 2010; Keller et al. 2003), Indian mustard (*Brassica juncea*) (Keller et al. 2003; Quartacci et al. 2006), poplar (*Populus* spp.) (Mertens et al. 2004; Robinson et al. 2003b, 2006), willow (*Salix* spp.) (Cosio et al. 2006; Dickinson and Pulford 2005; Jensen et al. 2009; Klang-Westin and Eriksson 2003;

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Mleczek et al. 2009), and sunflower (*Helianthus annuus*) (Fässler et al. 2010; Madejon et al. 2003), which are fast growing, deep rooted, and easily propagated and cultivated and have a high biomass production and a relatively high metal uptake capacity. It was soon realized though that regardless of the plants used, the rate of contaminant accumulation was insufficient and thus would have to be considerably increased without diminishing their yield.

One approach to achieve this was increasing the availability of contaminating TE in soil for plant uptake, e.g., by artificial soil acidification or solubilization by means of chelating agents. Various synthetic aminopolycarboxylic acid (APCA) such as ethylene diamine tetraacetic acid (EDTA), diethylene triamino pentaacetic acid (DTPA), trans-1,2-cyclohexylene dinitrilo tetraacetic acid (CDTA), ethylenediamine-*N*, *N*′-bis (2-hydroxyphenyl) acetic acid (EDDHA), and others displayed potential to significantly increase TE uptake by plants (Evangelou et al. 2007; Lai and Chen 2004; Wu et al. 2004). However, as more and more research was put into **chelant-assisted phytoextraction**, various drawbacks arose such as their toxicity to soil microorganisms (Grcman et al. 2001) and to plants (Chen and Cutright 2001; Epstein et al. 1999), and in particular the risk that mobilized TE could leach into groundwater or surface water (Evangelou et al. 2007; Lai and Chen 2005; Luo et al. 2005; Meers et al. 2005). To reduce this risk the use of biodegradable chelating agents such as ethylene diamine disuccinate (EDDS) or nitrilotriacetic acid (NTA) was suggested. However, the degradation rates of biodegradable chelating agents such as EDDS and NTA were still too low to significantly reduce the risk of leaching (Evangelou et al. 2007; Meers et al. 2005). The risk of TE leaching was caused by the fact that in order to achieve plant shoot concentration of >1,000 mg kg^{-1}, chelants have to be applied (a) in a single large dose, to break down the endodermis in order to increase the uptake via the limited apoplastic pathway, and (b) to large excess, as most chelants are nonspecific; hence, soil components such as Ca and Fe compete with targeted TE, thus reducing the efficiency of the applied chelants (Nowack et al. 2006).

The numerous setbacks in the development of phytoremediation led to a change in focus from phytoextraction, i.e., the removal of pollutants to **phytostabilization**. The aim of phytostabilization (a) is to prevent the dispersal of particle-bound pollutants by wind and water erosion and to reduce the export of dissolved contaminants by reducing surface runoff and water flow into the subsurface and (b) to minimize the transfer of contaminants into the food chain by using plants with minimal uptake of contaminants (Collins et al. 2006). This change of concept means that quite different plant characteristics are desired compared to phytoextraction. In phytoextraction, high accumulation of contaminants was desirable, whereas in phytostabilization plants should preferably exclude contaminants from their aerial parts. In recent years, the perception of contaminated soils has changed. For decades, such soils were regarded only as a source of hazard, which required remediation. Nowadays, contaminated soils are increasingly considered as a valuable resource that can sustain plant growth, biodiversity, and other ecosystem functions. Contaminated land is an extensive underutilized resource, which could and should be used in a sustainable way to grow plants for a large variety of profitable purposes. From this new perspective, the idea of phytomanagement emerged. Phytomanagement describes the engineering or manipulation of soil–plant systems to control the fluxes of TEs in the environment, maximizing economic and/or ecological benefits while minimizing risks. Thus, the goal of phytomanagement may be to alleviate deficiencies of crops in essential TEs or to reduce the environmental risk posed by contaminating TEs. A key component of phytomanagement is that it should either cost less than other remediation or fortification technologies or be a profitable operation, by producing valuable plant biomass products (Robinson et al. 2009).Thus, the aim of phytomanagement is to produce economic revenue on a contaminated land without causing detrimental effects on human health and nature.

9.2 Contaminated Land: An Extensive but Underutilized Resource

Growth in global population, high—and growing—consumption levels in industrialized countries, rapidly increasing middle classes, related increased consumption levels, expanding urban areas, and changing diets in emerging countries combined with the increasing energy consumption are some of the key drivers behind the increasing demand for land. Demographers project that the world population will rise to 9 bn by 2050 and level off somewhere between 9 and 12 bn people by the end of the century. Accompanying the population growth is an increase in personal income. Globally, the size of the middle class could increase from 1.8 bn people to 3.2 bn by 2020 and to 4.9 bn by 2030. This results in changes in lifestyle, diets, and demographics, with meat consumption playing an important role (OECD 2010). As a consequence, the global crop demand will increase by 100–110 % from 2005 to 2050 (Tilman et al. 2011); thus, according to the OECD and UN-FAO, agricultural production has to increase by about 60 % globally and nearly 77 % in developing countries by 2050 (OECD-FAO 2012). The potential to expand the arable land areas is not great. According to the FAO, out of the world's 13.5 bn ha of total land surface, the area of land that is potentially available for expanded rain-fed crop production is ca. 2 bn ha. Of this, 1.4 bn ha are currently used for agriculture and at least

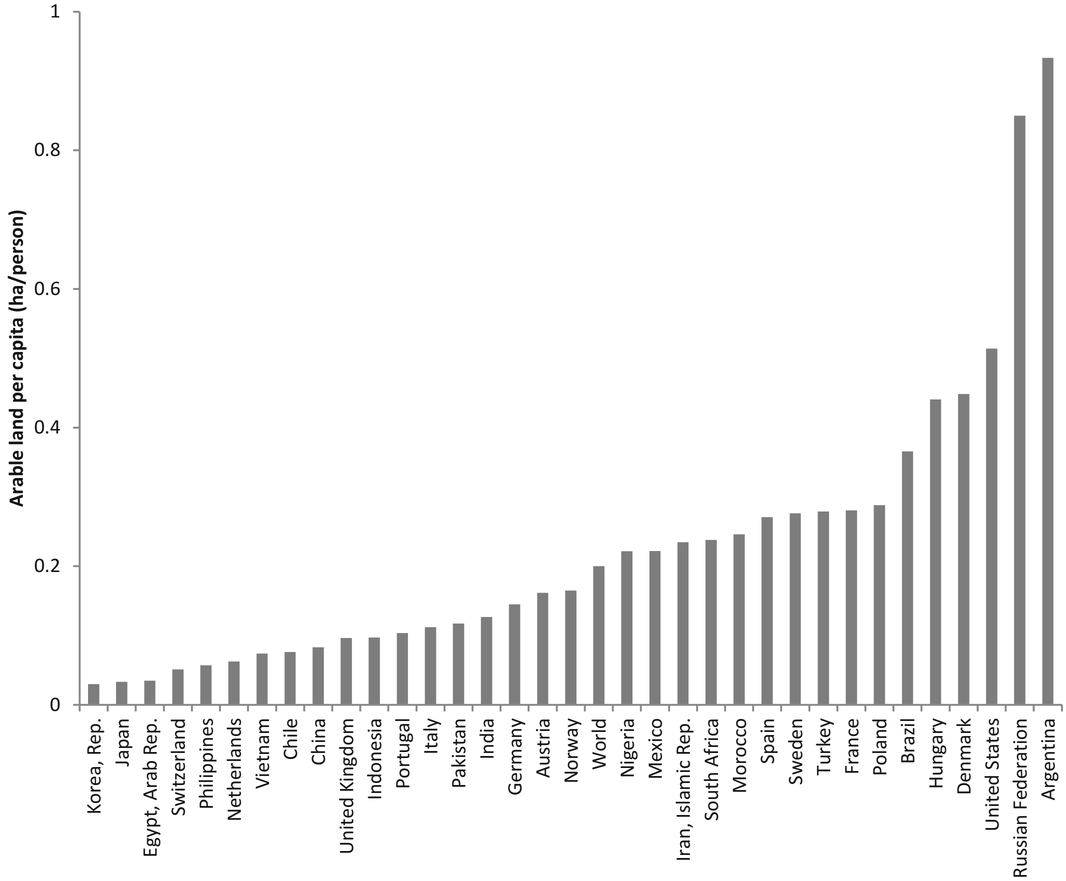

Fig. 9.1 Arable land per capita in selected countries for the year 2011 (TWB 2012)

500 M ha should remain protected from agriculture for environmental reasons (Haralambous et al. 2009). Additionally, built-up areas (currently 150 M ha) will further expand at the expense of arable or potentially arable land (FOE 2013).

Land demand for the production of food, fodder, and its use for housing will increasingly compete with the energy supply demands, in the form of bioenergy/biofuels. Biofuel growth is driven by policies and targets from over 50 countries, among them China, the EU, the United States, Brazil, the Philippines, India, and Uruguay who strive to become less reliant on foreign oil. In order to reach their biofuel demand of approximately 65 bn US gallons vast swathes of land would have to be converted from food to energy crop production (Evangelou and Schulin 2013). The OECD/FAO (2011) estimated that by 2020, 12 % of the global coarse grain production as well as 33 % of the sugar production will be used to produce ethanol. Additionally, 16 % of the global production of vegetable oil will be used to produce biodiesel. According to the International Energy Agency (IEA), in 2006 about 14 M ha of land—ca. 1 % of the world's currently available arable land—were used for the production of biofuels (IEA 2006). The FAO projects that these figures will increase up to 3.5 % by 2030 (Haralambous et al. 2009). These estimations have brought up concerns about food security and affordable food prices. Increasing population and soil loss is resulting in reduced arable land per person (Figs. 9.1 and 9.2).

Pointing to threatened food security, it has been argued that biomass production for biofuels is unsustainable (Friedemann 2007). As a result it has been suggested to use lands that are marginal, "underutilized," or "unused." However, such land, especially in developing countries, is often important for the livelihoods of poor rural communities, as it is used for grazing; as livestock transit routes; for collection of fuel wood, wild fruits and nuts, medicinal plants, and other plant products; and for access to water sources (Haralambous et al. 2009).

Contaminated land, which is not suitable for food production, is in contrast to "marginal" land often not used economically at all; thus, it could also be considered as a suitable alternative, not only for biofuels but timber and fodder as well, which would otherwise be grown on fertile non-contaminated soil that could be used to produce food. The global area of TE-contaminated soils is approximately 33 M ha (Evangelou et al. 2012). This estimation is conservative, as the extent of contaminated land in poor countries

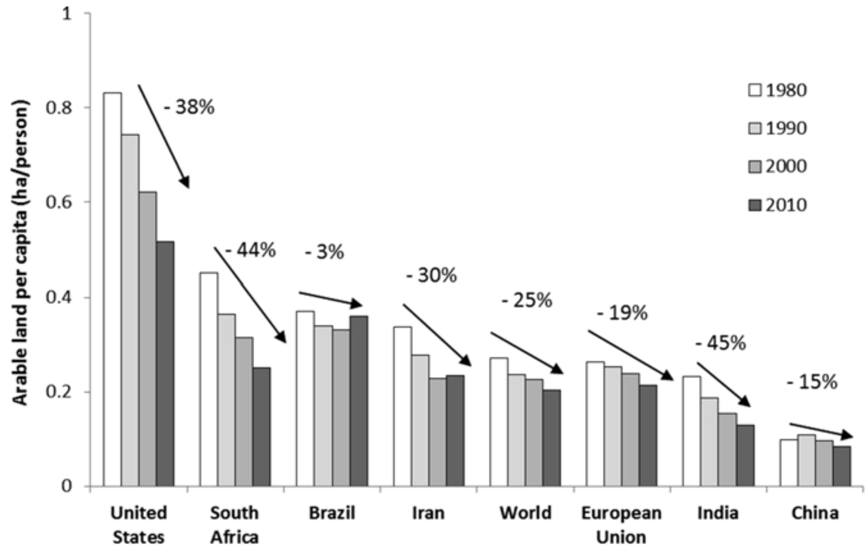

Fig. 9.2 Decrease of arable land in the years 1980–2010 (TWB 2012)

is difficult to assess due to a lack of published data. Nevertheless, it is well documented the area of contaminated land is increasing due to industrialization and lax environmental regulations in poor countries. The use of these lands would open to the affected countries new economic possibilities as most countries lack the wherewithal to remediate or secure contaminated land.

9.3 Potential Plant Species for Phytomanagement

Plants used for phytomanagement should be fast growing, deep rooted, and easily propagated and have a high biomass production as well. Their TE accumulation characteristics depend on the goal of phytomanagement. Plants that accumulate high concentrations of Se or Zn may be usefully employed on soil contaminated with these elements to provide supplementary fodder for stock in deficient areas (Banuelos and Dhillon 2011; Fässler et al. 2010). In some other cases, where accumulation of TEs may present a risk to the food chain, excluder species are desirable. In all cases the biomass should have economic or ecological value. The plants should additionally mitigate the risk originating from that soil, by, e.g., stabilizing the soil, reducing leaching, etc. Furthermore, their cultivation must be practically feasible and economically attractive under the given site and land use conditions (Robinson et al. 2009).

9.3.1 Trees

Various tree species can be used to produce biomass on contaminated land. Willow (*Salix* spp.) and poplar (*Populus* spp.) are used worldwide for bioenergy production, due to their fast growth and their capability to be coppiced (e.g., short-rotation coppice). Hardwood species such as eucalyptus (*Eucalyptus* spp.), beech (*Fagus* spp.), maple (*Acer* spp.), and birch (*Betula* spp.) as well as softwood species such as spruce, pine, fir, larch, and hemlock are used for the production of pulp, timber, and firewood. The trees should not accumulate high TE concentrations (a) in the wood, as risks could arise from the release of these TE during processing (e.g., bioenergy, pulp, burning) as well as during its use (e.g., wooden furniture, use of paper), and (b) in the leaves as TE could be spread through the dispersion of foliage into the surrounding ecosystems. Leaves usually accumulate the highest TE concentrations followed by bark and wood, as shown by Unterbrunner et al. (2007) for *Salix caprea*, *Salix purpurea*, *Salix fragilis*, *Salix* sp., *Populus tremula*, *Populus nigra*, and *Betula pendula*. Lead is an exception, as it often accumulates more in stems than in leaves (Migeon et al. 2009; Evangelou et al. 2013).

Willow (*Salix* spp.) and poplar (*Populus* spp.) are known to accumulate high concentrations of Cd and Zn (Dickinson and Pulford 2005; Migeon et al. 2009; Vamerali et al. 2009; Evangelou et al. 2012, 2013) (Table 9.1). Birch is a pioneer tree characterized by fast growth and low demand for soil nutrients. Thus, it can be a suitable candidate for the phytomanagement of contaminated soils with low nutrient contents. It does, however, take up more Pb than other tree genera such as willow (*Salix* spp.), poplar (*Populus* spp.), oak (*Quercus* spp.), beech (*Fagus* spp.), and maple (*Acer* spp.) (Migeon et al. 2009; Evangelou et al. 2012, 2013). Compared to willow (*Salix* spp.) and poplar (*Populus* spp.), birch (*Betula* spp.) usually accumulates less Cd (French et al. 2006; Hermle et al. 2006; Unterbrunner et al. 2007). Van Nevel et al. (2011) found that leaf Cd and Zn accumulation decreased in the order aspen (*Populus tremula*) > silver birch (*Betula pendula*) >> Scots pine (*Pinus sylvestris*) ≈ oak (*Quercus robur* and

Table 9.1 Bioaccumulation factors of potential trees for phytomanagement

Species	Plant part	TE	Bioaccumulation factor	References
Poplar (*Populus* spp.)	Wood	As		Laureysens et al. (2004); Mertens et al. (2007); Unterbrunner et al. (2007); Migeon et al. (2009); Vamerali et al. (2009); Evangelou et al. (2012, 2013)
		Cd	0.25–2.35	
		Cr	0.22–0.26	
		Pb	0.004–0.02	
		Zn	0.02–0.74	
	Leaves	As	0.005	Laureysens et al. (2004); Mertens et al. (2007); Unterbrunner et al. (2007); Migeon et al. (2009); Vamerali et al. (2009); Evangelou et al. (2012, 2013)
		Cd	0.56–4.63	
		Cr	0.21	
		Pb	0.006–0.01	
		Zn	0.2–1.47	
Willow (*Salix* spp.)	Wood	As		Klang-Westin and Eriksson (2003); Rosselli et al. (2003); Jensen et al. (2009); Migeon et al. (2009); Mleczek et al. (2009); Vamerali et al. (2009); Evangelou et al. (2012, 2013)
		Cd	0.72–6.5	
		Cr	0.16–0.22	
		Pb	0.002–4.5	
		Zn	0.28–1.62	
	Leaves	As	0.01	Unterbrunner et al. (2007); Reglero et al. (2008); Migeon et al. (2009); Vamerali et al. (2009); Evangelou et al. (2012, 2013)
		Cd	2.5–12.2	
		Cr	0.18–0.24	
		Pb	0.01–0.29	
		Zn	0.28–4.00	
Birch (*Betula pendula*)	Wood	As		Kozlov et al. (2000); Rosselli et al. (2003), Margui et al. 2007; Unterbrunner et al. (2007); Migeon et al. (2009); Evangelou et al. (2012, 2013)
		Cd	0.11–0.3	
		Cr	0.16	
		Pb	0.001–0.05	
		Zn	0.32–0.86	
	Leaves	As		Kozlov et al. (2000); Margui et al. (2007); Unterbrunner et al. (2007); Migeon et al. (2009); Evangelou et al. (2012, 2013)
		Cd	0.9	
		Cr	0.18	
		Pb	0.01–0.03	
		Zn	0.01–3	
Eucalyptus	Wood	As	0.01	Marchiol et al. (2013); Mok et al. (2013)
		Cd	0.1–0.90	
		Cr	0.1	
		Pb	0.03	
		Zn	0.05–7.61	
	Leaves	As		Shukla et al. (2011); Marchiol et al. (2013); Mok et al. (2013)
		Cd	0.32–0.94	
		Cr	0.1–1.2	
		Pb	0.3	
		Zn	0.37–6.14	
Oak (*Quercus* spp.)	Wood	As		Migeon et al. (2009); Evangelou et al. (2012, 2013)
		Cd	0.05–1.62	
		Cr	0.16	
		Pb	0.002–0.1	
		Zn	0.01–0.07	
	Leaves	As		Migeon et al. (2009); Evangelou et al. (2012, 2013)
		Cd	0.05–0.2	
		Cr	0.21	
		Pb	0.01–0.04	
		Zn	0.05–0.28	

Quercus petraea), while for the stem the order was aspen (*Populus tremula*) ≈ silver birch (*Betula pendula*) > Scots pine (*Pinus sylvestris*) > oak (*Quercus robur* and *Quercus petraea*). Scots pine (*Pinus sylvestris*) is a good bioindicator because it is sensitive to industrial pollution (Kosinska and Baaga 2007). Meeinkuirt et al. (2012) found a low ability of eucalypt (*Eucalyptus camaldulensis*) to accumulate Pb, while Mok et al. (2013) showed that the eucalyptus species *Eucalyptus polybractea* and *Eucalyptus cladocalyx* accumulated various TE such as Cd or Zn to high levels. Oaks (*Quercus* spp.) are defined by a high tolerance to TE but low uptake of TE (Migeon et al. 2009; Evangelou et al. 2012, 2013) (Table 9.4). But due to this slow growth, they are not very attractive for phytomanagement purposes, although their wood is valuable. Maple (*Acer* spp.) shows a low propensity to take up Zn and Cd, while it does not differ from willow (*Salix* spp.), poplar (*Populus* spp.), or birch (*Betula* spp.) in the accumulation of Pb and Cr (Migeon et al. 2009).

9.3.2 Agricultural Crop Plants

Crops used for the production of bioethanol are wheat (*Triticum* spp.), corn (*Zea mays* L.), sweet and grain sorghum (*Sorghum bicolor* (L.) Moench), and sugar beet (*Beta vulgaris* L.), as these plants accumulate large amounts of starch or sugars in plant storage organs, which can be fermented. For biodiesel, annual plants with high seed oil content are used, such as sunflower (*Helianthus annuus* L.), rapeseed (*Brassica napus* L. var. oleifera D.C.), soybean (*Glicine max* L.), and tobacco (*Nicotiana tabacum*). Besides plant parts rich in starch and sugar and oils, also stover and straw can be used to produce bioenergy. In contrast to the use of tree species or perennial herbaceous crops, annual plants require efforts for the management of harvest transport and processing. Using crops for the phytomanagement of contaminated soils that are also utilized for food or feed production, if grown on uncontaminated soil, requires particular attention, as there is an increased risk that the products of such crops could by mistake (or deliberate action) contaminate human food (Table 9.2 and 9.3).

A low TE concentration in the seeds of plants grown for the production of biodiesel is desirable as it would reduce the costs of removing TEs from the oil, which could be hazardous for human health. Soybean (*Glycine max*), wheat (*Triticum aestivum* L.), and corn (*Zea mays*) seeds accumulate significantly more Zn (up to six times more in soybean) than stems, while there are no significant differences for Cd and Pb (Lavado et al. 2001; Salazar et al. 2012). The production of methane or ethanol through anaerobic digestion (fermentation) requires low TE concentration, as they can negatively affect the enzymes responsible for the breakdown of biomass, as well as face issues concerning the fate of the digestate, such as its application on soils. Baig et al. (2011) reported the As accumulation in crop plants decreased in the order wheat (*Triticum aestivum* L.) > corn (*Zea mays* L.) ≈ sorghum (*Sorghum bicolor* L.). Sugarcane (*Saccharum* spp.) accumulates approximately 50 % less TE in the stalks than in the leaves, with the exception of Cd where the opposite occurs (Noguiera et al. 2013). Also Xia et al. (2009) found that sugarcane (*Saccharum officinarum*) has a high ability to tolerate and accumulate Cd.

The uptake of metals into the shoots may also entail potential ecological risks. Tobacco (*Nicotiana tabacum*) accumulates Cd to relatively high levels compared to other species (Kayser et al. 2000; Keller et al. 2003; Wenger et al. 2002). Concentrations of Cd in field-grown tobacco leaves were found to range from <0.5 to 5 mg Cd kg^{-1} (Lugon-Moulin et al. 2004). Corn (*Zea mays*) can accumulate Zn in the shoots (Keller et al. 2003; Luo et al. 2005), with Zn concentrations reaching >1,000 mg kg^{-1} without significant decrease in biomass (Wenger et al. 2002). In comparison soybean (*Glycine max*) reaches even higher Zn shoot concentrations (Murakami and Ae 2009). Rapeseed (*Brassica napus*) was found to accumulate more Pb than wheat (*Triticum* spp.), corn (*Zea mays* L.), and sorghum {*Sorghum bicolor* (L.)} (Tangahu et al. 2011).

Concerning potential risks deriving from biomass produced on contaminated soils, we can conclude from the available literature that tobacco (*Nicotiana tabacum*) and sugarcane (*Saccharum* spp.) would be unsuitable for Cd-contaminated soils, soybean (*Glycine max*) for Zn-contaminated soils, and wheat (*Triticum* spp.) for As-contaminated soil. Rapeseed (*Brassica napus*) is in general unsuitable for TE-contaminated sites, as it belongs to the family of the *Brassicaceae* (e.g., *Brassica species*: *B. nigra* (*L.*) Koch; *B. carinata A.* Braun; *B. oleracea* L.; *B. campestris* L.; *B. juncea* (L.) Czern.; *B. napus* L.), which includes many hyperaccumulators (Vamerali et al. 2010) (Table 9.4).

9.3.3 Herbaceous Perennial Crops

Perennial grasses have been widely used for centuries as fodder crops, often contributing significantly to energy supply on farms being used to feed draft animals. For example, as late as 1920 in the United States, 27 M animals fuelled by some 35–40 M hectares of grasslands provided traction power on farms and in cities, (Lewandowski et al. 2003). In the twenty-first century, perennial grasses may be set for a comeback, as they have a great potential to contribute to the production of bioenergy. Candidates are in particular switchgrass (*Panicum virgatum*), miscanthus (*Miscanthus* spp.), reed canary grass (*Phalaris arundinacea*), vetiver grass (*Vetiveria zizanioides* L.), elephant grass (*Pennisetum purpureum* Schumach), and giant reed (*Arundo donax*).

Table 9.2 Bioaccumulation factors of potential agricultural crops for phytomanagement

Species	Plant part	TE	Bioaccumulation factor	References
Soybean (*Glycine max*)	Shoot	As		Murakami et al. (2007, 2009); Zhuang et al. (2013)
		Cd	0.5–1.44–3.7	
		Cr	0.02	
		Pb	0.01–0.03	
		Zn	0.15–0.66	
	Grain	As		Salazar et al. (2012); Zhuang et al. (2013)
		Cd	0.57	
		Cr	0.02	
		Pb	0.01–0.13	
		Zn	0.54–4.95	
Tobacco (*Nicotiana tabacum*)	Shoot	As		Mench et al. (1989); Kayser et al. (2000); Keller et al. (2003); Evangelou et al. (2004, 2006, 2007); Fässler et al. (2010)
		Cd	0.66–2.6	
		Cr		
		Pb	0.03	
		Zn	0.1–0.22	
	Grain	As		
		Cd		
		Cr		
		Pb		
		Zn		
Rapeseed (*Brassica napus)*	Shoot	As		Solhi et al. (2005)
		Cd		
		Cr		
		Pb	0.03	
		Zn	0.16	
	Grain	As		Angelova et al. (2004)
		Cd	0.06–0.08	
		Cr		
		Pb	0.01–0.03	
		Zn	0.1–1.1	
Wheat (*Triticum aestivum* L.)	Shoot	As	0.04–0.11	Chen et al. (2004); Bermudez et al. (2011)
		Cd		
		Cr	0.5–1.3	
		Pb	0.02	
		Zn	0.2	
	Grain	As		Jamali et al. (2009). Bermudez et al. (2011)
		Cd	0.241–0.42	
		Cr	0.01	
		Pb	0.01–0.68	
		Zn	0.19–0.60	
Sunflower (*Helianthus annuus*)	Shoot	As		Kayser et al. (2000); Nehnevajova et al. (2009); Solhi et al. (2005); Marchiol et al. (2007); Sabudak et al. (2007); Fässler et al. (2010)
		Cd	0.2–2.7	
		Cr	0.2	
		Pb	0.01–0.07	
		Zn	0.1–0.7	
	Grain	As	8* 10^{-6}, & 5* 10^{-7}–0.009	Murillo et al. (1999); Angelova et al. (2004); Sabudak et al. (2007)
		Cd	0.01–0.05	
		Cr		
		Pb	5*10–5–0.009	
		Zn	0.01–0.7	

(continued)

Table 9.2 (continued)

Species	Plant part	TE	Bioaccumulation factor	References
Corn (*Zea mays*)	Shoot	As	0.03	Kayser et al. (2000) ; Chen et al. (2004); Chiu et al. (2005); Luo et al. (2005); Murakami et al. (2006, 2009)
		Cd	0.1–1.88	
		Cr		
		Pb	0.05–1.13	
		Zn	0.2–3.7	
	Fruit	As	0.045	Chiu et al. (2005)
		Cd		
		Cr		
		Pb		
		Zn	0.3	
Sorghum (*Sorghum bicolor* (L.) Moench),	Shoot	As	0.02–0.03	Murillo et al. (1999); Chen et al. (2004); Marchiol et al. (2007)
		Cd	0.05–0.1	
		Cr		
		Pb	0.01–0.02	
		Zn	0.09–0.15	
	Fruit	As		
		Cd		
		Cr		
		Pb		
		Zn		

Table 9.3 Bioaccumulation factors of potential perennial grasses for phytomanagement

Species	Plant part	TE	Bioaccumulation factor	References
Vetiver grass (*Vetiveria zizanioides* L.)	Shoot	As	0.04	Lai and Chen (2004); Chiu et al. (2005); Rotkittikhun et al. (2007)
		Cd	1.25	
		Cr		
		Pb	0.004–0.07	
		Zn	0.03–0.8	
Elephant grass (*Pennisetum purpureum* Schumach)	Shoot	As	0.5	Amonoo-Neizer et al. (1996)
		Cd		
		Cr		
		Pb		
		Zn		
Smilo grass (*Piptatherum miliaceum*)	Shoot	As	0.09	Marchiol et al. (2013)
		Cd	0.07	
		Cr		
		Pb	0.003	
		Zn	0.02	
Giant reed (*Arundo donax*)	Shoot	As	0.012	Boularbah et al. (2006); Guo and Miao (2010)
		Cd	0.04	
		Cr		
		Pb	0.007–0.0005	
		Zn	0.04–0.008	

Unlike trees or agricultural crops, research on the potential of perennial grasses to accumulate TE has not been so extensive (Table 9.3). Thus, a conclusion, about which perennial grasses are to be preferred for phytomanagement of TE-contaminated sites, cannot be drawn. Nevertheless, Hou et al. (2012) preferred hybrid *Pennisetum* followed by giant reed (*Arundo donax*), silver reed (*Thamnochortus cinereus*), and switchgrass (*Panicum virgatum*) for the phytoextraction of an As-, Hg-, Cu-, Cr-, Pb-, and Cd-contaminated soil. The ability of *Pennisetum* to accumulate Cd is supported by a study of Xia (2004), where *Pennisetum* reached higher Cd concentrations than vetiver grass (*Vetiveria zizanioides* L.).

Table 9.4 Suitability, positive (+) or negative (–), of various potential phytomanagement plants depending on their TE accumulation and soil degradation

	Trace elements		
	Cd	Zn	Pb
Trees			
Birch (*Betula pendula*)	+	+	– –
Eucalyptus (*Eucalyptus* spp.)	–	–	+
Oak (*Quercus* spp.)	++	++	+
Poplar (*Populus* spp.)	– –	– –	+
Maple (*Acer* spp.)	+	+	+
Scots pine (*Pinus sylvestris*)	++	++	+
Willow (*Salix* spp.)	––	––	+
Agricultural crops			
Corn (*Zea mays*)	+	–	+
Rapeseed (*Brassica napus*)	–	–	–
Sorghum (*Sorghum bicolor* (L.) Moench)			+
Soybean (*Glycine max*)		––	
Sugarcane (*Saccharum* spp.)	––		
Sunflower (*Helianthus annuus*)	–	––	
Tobacco (*Nicotiana tabaum*)	––		
Wheat (*Triticum aestivum* L.)			+
Perennial grasses			
Elephant grass (*Pennisetum purpureum*)	–	–	–
Giant reed (*Arundo donax*)	+/–	+/–	+/–
Switchgrass (*Panicum virgatum*)	+	+	+
Vetiver grass (*Vetiveria zizanioides* L.)	+	+	+

Giant reed (*Arundo donax*) is a tolerant plant species for Cd and Ni, which can accumulate high levels of these TE (Papazoglou 2007, 2009) (Table 9.4).

Owing to years of experience in the production of agricultural crops and timber, the optimal climatic and soil conditions for numerous species and varieties are well known. Perennial grasses are fairly new energy crops, and some, like miscanthus (*Miscanthus* spp.) and giant reed (*Arundo donax*), still retain wild-type characteristics such as high seed dormancy levels and insufficient winter rest ability. Breeding work for the development of varieties adapted to the different ecological/climatic zones, such as in the case of willow (*Salix* spp.) and poplar (*Populus* spp.), is still just beginning and has great potential to develop promising bioenergy varieties (Lewandowski et al. 2003).

9.4 Potential Products: Economic Revenue

9.4.1 Bioenergy

Bioenergy refers to renewable energy from biological sources, such as biomass that can be used for heat, electricity, and fuel, and their coproducts. The biomass may be used directly as heat (plants, wood, straw, and other plants) or processed into gases (from organic waste, landfill waste) or liquids, such as ethanol and biodiesel (derived from crops such as maize (*Zea mays*), sugarcane (*Saccharum officinarum*), wheat (*Triticum* spp.), rapeseed (*Brassica napus*), and soy (*Glycine max*) or from lignocellulosic material). Biomass has the great advantage over other renewable energy forms; it is currently the only renewable source of fixed carbon and thus is the only source in the long term for the production of transport fuels. Approximately 57.7 % of the worldwide oil consumption is used for transportation activities (IEA 2006), and the global primary demand for oil (excluding biofuels) will rise by 1 % per year on average, from 85 million barrels per day in 2007 to 106 mb day^{-1} in 2030 (IEA 2008). Thus, the market for biofuels will become very big, particularly owing to China's rapid expansion.

Biofuels have a potential but their economic viability is highly dependent on both the oil price and on governmental subsidies, the price of oil in the world market being of crucial importance. The starting point, from which the production of biofuels becomes profitable, is known as break-even point (balance point). In the European Union the break-even point for different biofuels can be reached from US\$75–80 barrel^{-1} of oil in relation to colza oil, US\$90 barrel^{-1} in relation to bioethanol, US\$100 barrel^{-1} to biodiesel, and US\$155–160 barrel^{-1} to fuels attained by second-generation technologies. In the United States the break-even point for bioethanol is currently reached when the oil price exceeds US\$40–50 barrel^{-1}. This means that bioethanol production is not economic at oil prices below US\$40 barrel^{-1}. In the case of producing ethanol in Brazil, the break-even point oscillates between US\$30 and 35 barrel^{-1}. For biofuels derived from vegetal oils, a technology in its incipient stage, the indicator is estimated to be about US\$60 barrel^{-1} (Evangelou et al. 2012, 2013). The break-even point for bioenergy produced, through combustion or fermentation from TE-enriched biomass, would probably be higher. Because filters would have to be installed to retain the volatile TE and the risk originating from the TE contained in the digestate after biogas or ethanol production would have to be mitigated.

9.4.2 Wood

Production and consumption of key wood products (roundwood, sawn-softwood, sawn-hardwood, panels, pulp, paper, and secondary products) are expected to continue past trends of 1–2 % annual increase until 2030. The global demand for wood products is driven by population increase and economic growth in particular in Asia (FAO 2009). Contemporaneously, more forests will be excluded from wood production due to new environmental policies and regulations. Additionally, there is less and less old-growth forest left for logging due to our exploitation and forest destruction. Worldwide there are >89 M ha of plantation forests (FAO 2001) with their area increasing rapidly. But also the land available for forest plantation is limited and under pressure by the demand for agricultural land. Again, areas with elevated TE concentrations (>33 M ha) could offer a viable alternative. They could be used for wood production, thus reducing the necessity to use natural forests.

Unlike biofuels, growing trees for timber does not produce rapid economic revenue. Plantation forests, depending on the tree species used and the intended product, will need 10–50 years before they become harvestable. Eucalyptus plantations, for example, intended for pulp production can be harvested after approximately 6 years (Clay 2004), while pine saw timber may need 30–50 years before it reaches economic maturity (Roth 1989). The long-term harvesting cycles are an advantage for phytomanagement, because the less the costs are for management (e.g., for harvesting, fertilizers etc.), the larger the revenue is for a given return. Furthermore, with longer duration between harvests (>25 years), the proportion of the TE rich bark can be reduced, thus reducing the overall TE concentration of the tree trunk (Evangelou et al. 2012, 2013).

The TE concentrations of wood produced on contaminated land should not exceed regulatory values. Swiss legislation and EPF industry standards require that wood panels intended for the market must not exceed concentrations of 50 mg kg^{-1} Cd, 90 mg kg^{-1} Pb, 25 mg kg^{-1} As, and 40 mg kg^{-1} Cu (ChemRRV 2005; EPF 2000). Packaging materials must not exceed the cumulative concentration limit of 100 mg kg^{-1} for Pb, Cd, Hg, and Cr as described in the EU Packaging and Packaging Waste Directive (94/62/EG) (European Parliament 1994). Thus, every product derived from phytomanagement should be monitored to ensure that it complies with the aforementioned as well as with other product related thresholds.

9.4.3 Biochar

Biochar is produced by pyrolysis (heat-induced carbonization in oxygen-poor atmosphere) of organic material. It is distinguished from charcoal by its main purposes, which are (1) to amend agricultural soils and thereby (2) to sequester carbon from organic matter and avoid its mineralization and release as carbon dioxide into the atmosphere (Lehmann and Joseph 2009). As the pyrolysis process can be used in the same time (3) to produce energy, biochar production has more than in one way the potential to make valuable use of organic residues and thus (4) offers an attractive alternative to other ways of organic wastes disposal.

Compared to biomass production for bioenergy, biochar production is still small. However, if biochar production were subsidized to a greater extent, it may result in similar challenges and problems as bioenergy production. As with bioenergy production, competition with food production should be avoided, thus feedstock sources should not reduce the availability and quality of cropland. Biomass originating from contaminated land could be such a source. A concern in the application of biochar originating from TE-contaminated soil could be the elevated concentrations of potentially toxic TE in the biochar. An important factor in this respect is the production temperature. Van Zwieten et al. (2010) found that the concentrations of Cu, Pb, Zn, Mg, Mn, Ni, and Ca were higher in biochar produced at 350 °C than in the feedstock but lower than biochar produced at 550 °C. Mercury and Cd are volatile when heated, even at 400 °C; thus, a low risk originates from these two particularly toxic TE when higher production temperatures are used. While the volatilization of toxic elements is positive for the subsequent use of biochar as soil amendment, it must be made sure that after volatilization, these contaminants are not released into the environment, but retained in the production facilities as in the case of burning biomass for energy. Trace element-enriched biomass should be converted into biochar only in production facilities, equipped with appropriate filter technology, which means that it will in general not be possible to produce biochar in small-scale biochar production facilities unless they are equally equipped, which will increase their production costs.

Even when the concentrations of toxic TE in biochar produced from plant biomass can be kept low with a suitable choice of the feedstock plants and biochar production temperature, the application is still not without risk. It is not yet sufficiently well known how the mobility and bioavailability of biochar bound TE will change with time, due to microbial activity, pH changes, organic matter interaction with biochar, etc. Trace element plant uptake and toxicological and mobility studies have to be performed to minimize the risk originating from TE-enriched biochar.

9.4.4 Biofortified Products

Deficiencies of the mineral micronutrients Fe, Zn, Se, and I affect more than half of humanity (Graham 2008). Other mineral elements, such as Ca, Mg, and Cu, can also be deficient in the diets of some populations (Zhao and McGrath 2009). One strategy for combating micronutrient malnutrition is to “biofortify” plant-based food through increased accumulation of critical elements in the edible parts of crop plants (Bouis 1996; Frossard et al. 2000; Welch 2002; Welch and Graham 1999). For this to be the case, the soil must be sufficiently rich in the elements targeted for biofortification such as Fe, Zn, Se, I, Ca, Mg, or Cu and sufficiently poor in undesired TE such as Cd, Pb, Hg, Sb, or As, depending on the capability of the plants used for selective uptake and exclusion of these elements. Hyperaccumulating plants are of particular interest in this respect, as most of them hyperaccumulate only one particular element (Assunção et al. 2003). To date >500 plant species have been classified as hyperaccumulators, with the majority (approximately 90 %) being Ni hyperaccumulators. There are also 32 Cu, 20 Se, 12 Zn, and 12 Mn hyperaccumulators that could be potentially used for the production of biofortified products (Ent et al. 2013).

It is rather rare that a soil is enriched in only one TE or metalloid. Nevertheless, these soils may require remediation or risk control, if that metal or metalloid is posing a threat to human health such as in the case of Se enrichment in seleniferous soils. Despite many anthropogenic Se sources, such as fossil fuel combustion, metal processing, applications of fertilizers, lime and manure, and disposal of sewage sludge, the Se content of most soils is primarily of geogenic origin. While most soils contain only 0.01–2.0 mg Se kg^{-1}, mean 0.4 mg Se kg^{-1}, the Se concentration of seleniferous soils can reach up to 1,200 mg Se kg^{-1}. Seleniferous soils are widespread in the Great Plains of the United States, Canada, South America, China, and Russia (White et al. 2007). Phytoremediation, i.e., cleansing of these soils using Se hyperaccumulators or *Brassica* sp. and barley (*Hordeum vulgare*), was found to be not feasible (Banuelos et al. 1997; Banuelos and Mayland 2000). However, if the aim is not cleansing but only control, then combining phytomanagement with the production of biofortified products can create a win-win situation. In the western part of the Central Valley, where soil are rich in Se concentrations, Banuelos and Mayland (2000) produced Se-enriched canola (*Brassica napus*) and utilized it as Se-biofortified forage to feed marginally Se-deficient lambs and cows. Similarly, plants such as rapeseed (*Brassica napus*), raya (*Brassica juncea*), sunflower (*Helianthus annuus*), cowpea (*Vigna sinensis*), guar (*Cyamopsis tetragonoloba*), wheat (*Triticum aestivum*), spearmint (*Mentha viridis*), sugarcane (*Saccharum officinarum*), barley (*Hordeum vulgare*), and bajra (*Pennisetum typhoides*) were used on various seleniferous soils to produce Se-enriched food, fodder, or fertilizer in India (Banuelos and Dhillon 2011). In Enshi, China, the so-called World Capital of Selenium, Yuan et al. (2012) used plants such as clover (*Trifolium repens*) and alfalfa (*Medicago sativa*) to produce Se-biofortified fodder.

The consumption of Se-enriched food or fodder is not without risks as Se readily becomes toxic at elevated concentration. Selenium concentrations of food and feed products have to be determined and controlled and Se-enriched biomass should be used only with great care. Some plants can easily accumulate Se to concentrations that are above the safety threshold for human or animal consumption. In search for alternative uses, Dhillon et al. (2007) incorporated Se-rich plant materials (up to 20 t ha^{-1}) into a non-seleniferous agricultural soil and produced wheat (*Triticum aestivum* L.) grains and straw with increased but safe levels of Se supplement in the diets of animals and humans living in Se-deficient areas. Using Se-rich plant material as an organic Se fertilizer for growing other crops could thus be a safe alternative for utilizing plant material that otherwise is too dangerous as direct Se source in animals and humans nutrition.

9.5 The Effect of Plants on the Mobility of Contaminants: Potential and Risks

9.5.1 Potential for Risk Mitigation

Plants can mitigate environmental and health risks arising from TE-contaminated soils by (a) preventing erosion through vegetation cover, (b) reducing leaching, and (c) immobilizing the contaminants. Protection against wind and water erosion is particularly important in cases where mineral and organic particles at the soil surface are loaded with high amounts of pollutants. Dense vegetation protects soil against wind and water erosion. The vegetation cover shields the soil surface against the impact of rainfall and wind. The root systems form a net that holds the soil together. Their exudates help to clog soil particles into larger aggregates and promote the activity of soil organisms which in turn promote the development of an aggregated soil structure and thus its mechanical stability. The extraction of soil water for transpiration promotes the formation of pores that are easily drained and facilitates soil aeration. As a result, the capacity of the soil to store and drain infiltrating water increases reducing the occurrence of surface runoff and thus water erosion. During dry periods vegetation also protects the soil surface against desiccation, so that the soil surface is stabilized by capillary cohesion of the soil particles against wind erosion.

Soil water consumption for transpiration also reduces contaminant leaching (Pilon-Smits 2005; Robinson et al. 2003b; Vose et al. 2003). How effective vegetation is in controlling leaching also depends on climate. The capacity of the atmosphere to take up water vapor sets upper limit on evapotranspiration. Actual evapotranspiration is due to limitations in water transfer from soil into plants often much lower than this limit. In dryer climates, evapotranspiration is usually greater from deep-rooted species because shallow-rooted species have less access to water during periods of drought and are, therefore, more likely to suffer from die-back or reduced transpiration and growth (Robinson et al. 2009). Robinson et al. (2003b, 2007) found that hybrid poplars (*Populus deltoides* × *nigra* ("Argyle" and "Selwin"), *Populus deltoides* × *yunnanensis* ("Kawa"), *Populus euramericana* × *yunnanensis* ("Toa"), *Populus alba* × *glandulosa* ("Yeogi"), *Populus nigra* × *manimowic* ("Shinsei")) reduced B leaching from a wood-waste landfill due to enhanced evapotranspiration but did not completely prevent substantial leaching driven by heavy rainfall events. Thus, collection and treatment of the discharge from the landfill would still be necessary.

Another way in which plants can immobilize pollutants in soil is binding them through their roots. Unlike in phytoextraction, not cellular uptake is necessary for this. It is sufficient that the contaminant is bound to the root cell walls. In the apoplast, the intercellular space including the cell walls separated by the intracellular space by the cell membranes, significant amounts of various substances can be bound owing to the high sorption capacity of the cell walls. Plant roots can immobilize contaminants also by modifying the chemical environment in the rhizosphere. For example, an increase in pH can reduce the solubility of metal cations. Root exudates promote the formation of soil organic matter and thus increase the sorption capacity of the soil. Also the transition from anaerobic to aerobic conditions in a soil can increase the TE sorption capacity in soils by inducing the oxidation of dissolved Fe(II) and Mn(II) to Fe(III) and Mn(IV), which then precipitate as oxides and hydroxides.

9.5.2 Possible Emerging Risks

Although deep roots seem generally more favorable than shallow roots, one must not forget that deep roots may create macropores which facilitate the preferential transport of contaminants to groundwater (Roulier et al. 2008). In a study by Knechtenhofer et al. (2003), it was shown that soil preferential flow paths below 20 cm, which are associated with roots surrounded by relatively wide root channels (Kretzschmar et al. 1999), played a significant role in the spatial distribution of Pb. In this large macropores, Pb may be transported as aqueous ions or bound by colloidal particles (Kretzschmar et al. 1999). A pH increase can decrease the mobility of metals, but it can also result in the solubilization of humic substances and facilitate the downward mobility of metals via preferential flow pathways.

Trace element-contaminated litter or harvest residues might be dispersed via wind or water erosion, thus potentially contaminating adjacent environments (Perronnet et al. 2000). Such litter and harvest residues decompose slower than non-contaminated plant material (Boucher et al. 2005; Cotrufo et al. 1995), as can be observed by the accumulation of litter on the forest floor near smelters (Berg et al. 1991; Freedman and Hutchinson 1980; Strojan 1978), resulting in the long-term availability of plant material in a form that can be dispersed. Thus, prevention measures should be taken to control plant material (e.g., leaves) dispersion, especially in situations where wind and water can be expected (Van Nevel et al. 2007). With the decomposition of contaminated litter, the contaminants may be released into the soil. Scheid et al. (2009) observed that the sorbed TE (Cu, Cd, Pb, Zn) were strongly bound in the litter even after 2 years of decomposition. However, if the contaminants become associated with dissolved organic matter, they will in fact be more mobile than contaminants adsorbed on mineral particles (Van Nevel et al. 2007).

9.6 Sustainability Aspects

9.6.1 Ecological Sustainability

The management of contaminated soils has to consider not only the established ecosystems on the site but also surrounding ecosystems. Trace element-contaminated land may be valuable as sites of specific floras and faunas such as Galmei-Vegetation in Germany (Engelen and Holtz 2000). The flora growing in metalliferous soils is a source of genetic material for research (Brady et al. 2005; Whiting et al. 2004). There is a trend to protect biodiversity that can accompany the agricultural/industrial development in these kinds of soils (Dickinson et al. 2009). Vidic et al. (2009) showed that the genome size of the species was related to their survival in TE-contaminated soils. Tolerant species had small genomes in comparison with non-tolerant ones. Such surrounding ecosystems can be affected by over extensive use or the use of not suitable plant species on phytomanaged sites. Cormish (1989) showed that when radiata pine (*Pinus radiata*) was planted in Australia to reduce soil erosion and increase slope stability, it reduced stream flow so effectively that naturally perennial streams were turned into ephemeral streams. There was thus a possibility that habitats of several fauna and flora species of surrounding ecosystems requiring perennial stream flow would be endangered.

An important debate relates to the use of non-endemic species for the production of biomass on contaminated land. Although most food, fiber, and landscape plants are nonnative, relatively few have proven invasive. However, some of those that are invasive have caused substantial socioeconomic and environmental impacts. Economic losses caused by invasive plants and costs for their control are estimated to be $34 bn annually in the United States (Ditomaso et al. 2010) and $10 bn annually in Europe (Hulme et al. 2009). The introduction and planting of invasive species/neophytes in various regions of the world, such as eucalyptus in Southern Europe or the giant reed in the Unites States, are caused by government actions. Johnson grass (*Sorghum halepense*) was originally grown as a forage grass but since has become a weed that greatly depresses yields of corn (*Zea mays*), soybeans (*Glycine max*), and other crops. It has invaded now in 16 states of the United States and incurs annual losses of more than $30 M in just three of them (Simberloff 2008). Another fast-growing perennial grass that has become invasive is miscanthus (*Miscanthus* spp.). It is primarily used for biofuel production and has been described as "Johnson grass on steroids" (Raghu et al. 2006). Therefore, the plant species used for phytomanagement should ideally be endemic. Growing noninvasive species might initially be economically less attractive. However, in the long term they would be more advantageous as no clearing costs would be involved, and public opinion would be less hostile.

9.6.2 Soil Sustainability

The production of biomass for biofuels, food, timber, etc., has often caused soil degradation/soil loss due to inadequate soil management practices. The main forms of soil degradation are water (56 %) and wind erosion (28 %). Other forms including chemical degradation and physical degradation sum up to 16 %. In total soil degradation affects about 2,000 M ha of land, which is equivalent to 15 % of the Earth's land surface (an area larger than the United States and Mexico combined). The causes of soil degradation include overgrazing (35 %), deforestation (30 %), agricultural activities (27 %), overexploitation of vegetation (7 %), and industrial activities (1 %) (UNEP 2002). Increased biofuel production has shown that soil degradation could become more severe in the near future. In Indonesia, for instance, two-thirds of oil palm expansion has occurred by converting large rainforest areas. In the United States, 1.3 M ha of lands in the Conservation Reserve Program designed to help check surpluses, maintain price levels, and promote an ecological balance were called back into production (UNEP 2012). Soil degradation can be influenced greatly by the user with an appropriate choice of plants, as well as management, which have to be adjusted accordingly to the soil type, the climate, and the geomorphology.

The choice of plant species is very important as they can either increase or decrease soil erosion as well as soil organic carbon (SOC) content. Sullivan (2004) found that traditional annual crops such as corn (*Zea mays*) and soybean (*Glycine max*) caused 50 times more soil erosion than sod crops. In general, soil is more exposed to the impacts of weather in row crops than in the latter. Also trees such as willow (*Salix* spp.) or poplar (*Populus* spp.) usually provide better protection against erosion than row crops. Pimentel and Krummel (1987) showed that under short-rotation woody crops (SRWC), the average erosion rate was 2 Mg ha^{-1} year^{-1} on a 5 % slope, whereas corn (*Zea mays*) grown on a 4 % slope resulted in a soil loss of 21.8 Mg ha^{-1} year^{-1}. Nevertheless, erosion can still be high under SRWC, if there is no herbaceous cover beneath the trees, especially when there is a high throughfall of rain (Kort et al. 1998). Perennial grasses are also effective in reducing erosion (Kemper et al. 1992) due to their dense network of fibrous roots close to the soil surface. The choice of the plant species for phytomanagement influences the SOC content. McLaughlin and Walsh (1998) reported that carbon sequestration rates under switchgrass (*Panicum virgatum*) may exceed those of annual crops by as much as 20–30 times, owing to carbon storage in the soil. Cultivation of temperate-zone perennial grasses such as miscanthus (*Miscanthus × giganteus*), switchgrass (*Panicum virgatum*), and others can increase SOC by 0.1–1 Mg ha^{-1} year^{-1} (Anderson-Teixeira et al. 2009). Short-rotation wood coppice, with willows (*Salix* spp.), may be even more effective in storing SOC than switchgrass *(Panicum virgatum*). Zan et al. (2001) found that relatively fertile soils in Canada beneath willows (*Salix* spp.) stored more SOC than under corn (*Zea mays*) or switchgrass *(Panicum virgatum*) 4 years after establishment.

Other agricultural management factors that have a major influence on soil erosion and soil carbon sequestration are whether crop residues are left on the field and incorporated into the soil as well as tillage practices. While there can be important differences between different tillage techniques, tillage in general increases the risk of soil erosion and SOC loss (Anderson-Teixeira et al. 2009; Williams et al. 2009), whereas crop residues that are left on the land protect the soil against erosion and SOC loss. If residues are completely removed, no-tillage soils can be as even more vulnerable to wind erosion than plowed soils during drought periods (Blanco-Canqui 2010). Blanco-Canqui and Lal (2009) considered a partial removal of 25 % of stover as the maximum rate that can be tolerated in no-tillage soils. This might be enough to control wind erosion, but it might still be too much to maintain optimal levels of SOC. Wilhelm et al. (2007) found that the amounts of corn (*Zea mays*) stover needed to maintain SOC at such a level by far exceed the amounts needed to control water and wind erosion.

9.7 Decision Support Systems

The success of phytomanagement crucially depends on the choice of the right plants and cultivation methods. The cultivation of candidate plants must be practical, economically attractive, and safe under the conditions of the given site and land use conditions. In practice, it is not possible to perform experimental trials in each specific case. However, the results of many pot and field studies in which plants have been grown on polluted soils have been integrated into model-based decision support systems (DSS), such as REC-Phyto-DSS (Onwubuya et al. 2009), Phyto-DSS (Robinson et al. 2003a), and Phyto-3 (Bardos et al. 2011).These can be of great help in the evaluation, design, and operation of site-adapted phytoremediation schemes. All the mentioned DSS either use a multi-criteria analysis or life cycle analysis or both. Phyto-3 is designed for US conditions. It provides guidance for regulators and practitioners, evaluating options of remedial phytotechnology available for the treatment of contaminated sites, with a strong focus on groundwater protection against organic contaminants. REC-Phyto-DSS is a European DSS specifically focusing on "gentle" site remediation techniques and in particular on phytoextraction and phytostabilization. It is implemented in the Dutch REC (Risk reduction, Environmental merits, and Cost) framework. Phyto-DSS is a generic tool designed to predict the efficiency of metal phytoextraction and evaluate its economic feasibility. It is based on a mechanistic model taking account of plant water use, soil metal solubility, and root distribution but used a lumped parameter to determine the ratio between metal concentrations in the xylem of the remediation plants and the soil solution. While the existing DSSs provide a good basis for the assessment of contaminated sites, as shown in Cano-Reséndiz et al. (2011), none of them however have yet a sufficient focus on the economic revenue. Thus, to encourage efficiency and increase the monetary output and keep possible risks derived from phytomanagement sites at a minimum, a DSS developed for phytomanagement is needed.

9.8 Conclusions

Starting with phytoextraction as a novel, low-tech, promising tool for soil cleaning around two decades ago, phytoremediation of contaminated soils will forever just remain a promising tool if it is not linked to profitable production of biomass, in the form of phytomanagement. Successful phytomanagement requires a multidisciplinary approach combining the design of appropriate crop management schemes, control of contaminant fluxes, assessment of associated risk, and optimization of economic revenues. Once accepted by regulators and decision makers in charge, phytomanagement could become a viable solution to use and even restore polluted soils. The phytomanagement of contaminated sites could offer an alternative income to people living nearby such areas and who lost their livelihood because of the contamination.

References

Amonoo-Neizer EH, Nyamah D, Bakiamoh SB (1996) Mercury and arsenic pollution in soil and biological samples around the mining town of Obuasi. Ghana Water Air Soil Pollut 91:363–373

Anderson-Teixeira KJ, Davis SC, Masters MD, Delucia EH (2009) Changes in soil organic carbon under biofuel crops. GCB Bioenergy 1:75–96

Angelova V, Ivanova R, Ivanov K (2004) Heavy metal accumulation and distribution in oil crops. Commun Soil Sci Plant Anal 35:2551–2566

Assunção AGL, Bookum WM, Nelissen HJM, Vooijs R, Schat H, Ernst WHO (2003) Differential metal-specific tolerance and accumulation patterns among Thlaspi caerulescens populations originating from different soil types. New Phytol 159:411–419

Banuelos GS, Dhillon KS (2011) Developing a sustainable phytomanagement strategy for excessive selenium in western United States and India. Int J Phytoremediation 13:208–228

Banuelos GS, Mayland HF (2000) Absorption and distribution of selenium in animals consuming canola grown for selenium phytoremediation. Ecotoxicol Environ Saf 46:322–328

Banuelos GS, Ajwa HA, Terry N, Zayed A (1997) Phytoremediation of selenium laden soils: a new technology. J Soil Water Conserv 52:426–430

Bardos RP, Bone B, Andersson-Skold Y, Suer P, Track T, Wagelmans M (2011) Crop-based systems for sustainable risk-based land management for economically marginal damaged land. Remediation 21: 11–33

Baumann A (1885) Das Verhalten von Zinksatzen gegen Pflanzen und im Boden. Die Landwirtschaftliche Versuchstation 31:1–53

Berg B, Ekbohm G, Soderstrom B, Staaf H (1991) Reduction of decomposition rates in scots pine needle litter due to heavy-metal pollution. Water Air Soil Pollut 59:165–177

Bermudez GMA, Jasan R, Plá R, Pignata ML (2011) Heavy metal and trace element concentrations in wheat grains: assessment of potential non-carcinogenic health hazard through their consumption. J Hazard Mater 193:264–271

Blanco-Canqui H (2010) Energy crops and their implications on soil and environment. Agron J 102:403–419

Blanco-Canqui H, Lal R (2009) Crop residue removal impacts on soil productivity and environmental quality. Crit Rev Plant Sci 28:139–163

Boucher U, Lamy I, Cambier P, Balabane M (2005) Decomposition of leaves of the metallophyte *Arabidopsis halleri* in soil microcosms: fate of Zn and Cd from plant residues. Environ Pollut 135: 323–332

Bouis H (1996) Enrichment of food staples through plant breeding: a new strategy for fighting micronutrient malnutrition. Nutr Rev 54:131–137

Boularbah A, Schwartz C, Bitton G, Aboudrar W, Ouhammou A, Morel JL (2006) Heavy metal contamination from mining sites in South Morocco: 2. Assessment of metal accumulation and toxicity in plants. Chemosphere 63:811–817

Brady KU, Kruckeberg AR, Bradshaw HD (2005) Evolutionary ecology of plant adaptation to serpentine soils. Annu Rev Ecol Evol Syst 36:243–266

Brooks RR, Lee J, Reeves RD, Jaffre T (1977) Detection of nickeliferous rocks by analysis of herbarium specimens of indicator plants. J Geochem Explor 7:49–57

Cano-Reséndiz O, de la Rosa G, Cruz-Jiménez G, Gardea-Torresdey JL, Robinson BH (2011) Evaluating the role of vegetation on the transport of contaminants associated with a mine tailing using the Phyto-DSS. J Hazard Mater 189:472–478

ChemRRV (2005) Verordnung vom 18. Mai 2005 zur Reduktion von Risiken beim Umgang mit bestimmten besonders gefährlichen Stoffen, Zubereitungen und Gegenständen (Chemikalien-Risikoreduktions-Verordnung, ChemRRV). In: Eidgenossenschaft DBdS (ed)

Chen H, Cutright T (2001) EDTA and HEDTA effects on Cd, Cr, and Ni uptake by Helianthus annuus. Chemosphere 45:21–28

Chen YH, Li XD, Shen ZG (2004) Leaching and uptake of heavy metals by ten different species of plants during an EDTA-assisted phytoextraction process. Chemosphere 57:187–196

Chiu KK, Ye ZH, Wong MH (2005) Enhanced uptake of As, Zn, and Cu by *Vetiveria zizanioides* and *Zea mays* using chelating agents. Chemosphere 60:1365–1375

Clay J (2004) World agriculture and the environment: a commodity-by-commodity guide to impacts and practices. Island Press, Washington, DC

Collins C, Fryer M, Grosso A (2006) Plant uptake of non-ionic organic chemicals. Environ Sci Technol 40:45–52

Cormish PM (1989) The effects of radiata pine plantation establishment and management on water yields and water quality - a review. Forestry Commission of New South Wales, Beecroft

Cosio C, Vollenweider P, Keller C (2006) Localization and effects of cadmium in leaves of a cadmium-tolerant willow (Salix viminalis L.) I. Macrolocalization and phytotoxic effects of cadmium. Environ Exp Bot 58:64–74

Cotrufo MF, Desanto AV, Alfani A, Bartoli G, Decristofaro A (1995) Effects of urban heavy-metal pollution on organic-matter decomposition in *Quercus ilex* L woods. Environ Pollut 89:81–87

Dhillon SK, Hundal BK, Dhillon KS (2007) Bioavailability of selenium to forage crops in a sandy loam soil amended with Se-rich plant materials. Chemosphere 66:1734–1743

Dickinson NM, Pulford ID (2005) Cadmium phytoextraction using short-rotation coppice *Salix*: the evidence trail. Environ Int 31:609–613

Dickinson N, Baker A, Doronila A, Laidlaw S, Reeves R (2009) Phytoremediation of inorganics: realism and synergies. Int J Phytoremediation 11:97–114

Ditomaso JM, Reaser JK, Dionigi CP, Doering OC, Chilton E, Schardt JD, Barney JN (2010) Biofuel vs bioinvasion: seeding policy priorities. Environ Sci Technol 44:6906–6910

Engelen B, Holtz F (2000) Galmeiveilchen Ein Stückchen Heimat. Zart und angepasst. Meyer & Meyer Regionalia, Aachen

Ent A, Baker AM, Reeves R, Pollard AJ, Schat H (2013) Hyperaccumulators of metal and metalloid trace elements: facts and fiction. Plant Soil 362:319–334

EPF (2000) EPF Industry standard – the use of recycled wood for wood-based panels. In: European Panel Federation (ed) European Panel Federation, Brussels

Epstein AL, Gussman CD, Blaylock MJ, Yermiyahu U, Huang JW, Kapulnik Y, Orser CS (1999) EDTA and Pb-EDTA accumulation in Brassica juncea grown in Pb-amended soil. Plant Soil 208:87–94

European Parliament (1994) European Parliament and Council Directive 94/62/EC of 20 December 1994 on packaging and packaging waste. In: European Parliament (ed)

Evangelou MWH, Schulin R (2013) Soil loss: a forgotten aspect of biofuel production. In: Allard MC (ed) Bioenergy systems, biological sources and environmental impact. Nova, Hauppauge, NY

Evangelou MWH, Daghan H, Schaeffer A (2004) The influence of humic acids on the phytoextraction of cadmium from soil. Chemosphere 57:207–213

Evangelou MWH, Ebel M, Schaeffer A (2006) Evaluation of the effect of small organic acids on phytoextraction of Cu and Pb from soil with tobacco Nicotiana tabacum. Chemosphere 63:996–1004

Evangelou MWH, Bauer U, Ebel M, Schaeffer A (2007) The influence of EDDS and EDTA on the uptake of heavy metals of Cd and Cu from soil with tobacco Nicotiana tabacum. Chemosphere 68:345–353

Evangelou MWH, Conesa HM, Robinson BH, Schulin R (2012) Biomass production on trace element-contaminated land: a review. Environ Eng Sci 29:823–839

Evangelou MWH, Robinson BH, Gunthardt-Goerg MS, Schulin R (2013) Metal uptake and allocation in trees grown on contaminated land: implications for biomass production. Int J Phytoremediation 15:77–90

FAO (2001) Global forest resources assessment 2000

FAO (2009) State of the world's forests 2009

Fässler E, Robinson BH, Stauffer W, Gupta SK, Papritz A, Schulin R (2010) Phytomanagement of metal-contaminated agricultural land using sunflower, maize and tobacco. Agric Ecosyst Environ 136:49–58

FOE (2013) Hidden impacts: how Europe's resource overconsumption promotes global land conflicts. Friends of the Earth Europe, Vienna, Austria

French CJ, Dickinson NM, Putwain PD (2006) Woody biomass phytoremediation of contaminated brownfield land. Environ Pollut 141:387–395

Freedman B, Hutchinson TC (1980) Effects of a smelter pollutants on forest leaf litter decomposition near a nickel-copper smelter at Subduy, Ontario. Can J Bot Rev Can Bot 58:1722–1736

Friedemann A (2007) Peak soil: why cellulosic ethanol, biofuels are unsustainable and a threat to America. Cultural Change

Frossard E, Bucher M, Machler F, Mozafar A, Hurrell R (2000) Potential for increasing the content and bioavailability of Fe, Zn and Ca in plants for human nutrition. J Sci Food Agric 80:861–879

Graham R (2008) Micronutrient deficiencies in crops and their global significance. In: Alloway B (ed) Micronutrient Deficiencies in Global Crop Production. Springer, Dordrecht, pp 41–61

Grcman H, Velikonja-Bolta S, Vodnik D, Kos B, Lestan D (2001) EDTA enhanced heavy metal phytoextraction: metal accumulation, leaching and toxicity. Plant Soil 235:105–114

Guo ZH, Miao XF (2010) Growth changes and tissues anatomical characteristics of giant reed (Arundo donax L.) in soil contaminated with arsenic, cadmium and lead. J Cent South Univ Technol 17:770–777

Haralambous S, Liversage H, Romano M (2009) The growing demand for land risks and opportunities for smallholder farmers

Hermle S, Gunthardt-Goerg MS, Schulin R (2006) Effects of metal-contaminated soil on the performance of young trees growing in model ecosystems under field conditions. Environ Pollut 144:703–714

Hou X, Fan X, Wu J, Zhu Y, Zhang Y (2012) Potentiality of herbaceous bioenergy plants in remediation of soil contaminated by heavy metals. Chin J Grassl 34(59–64):76

Hulme PE, Pysek P, Nentwig W, Vila M (2009) Will threat of biological invasions unite the European Union? Science 324:40–41

IEA (2006) Key world energy statisitcs. International Energy Agency, Paris

IEA (2008) World energy outlook 2008. International Energy Agency, Paris

Jamali MK (2011) Evaluation of arsenic levels in grain crops samples, irrigated by tube well and canal water. Food Chem Toxicol 49:265–270

Jamali MK, Kazi TG, Arain MB, Afridi HI, Jalbani N, Kandhro GA, Shah AQ, Baig JA (2009) Heavy metal accumulation in different varieties of wheat (*Triticum aestivum* L.) grown in soil amended with domestic sewage sludge. J Hazard Mater 164:1386–1391

Jensen JK, Holm PE, Nejrup J, Larsen MB, Borggaard OK (2009) The potential of willow for remediation of heavy metal polluted calcareous urban soils. Environ Pollut 157:931–937

Kayser A, Wenger K, Keller A, Attinger W, Felix HR, Gupta SK, Schulin R (2000) Enhancement of phytoextraction of Zn, Cd, and Cu from calcareous soil: the use of NTA and sulfur amendments. Environ Sci Technol 34:1778–1783

Keller C, Hammer D, Kayser A, Richner W, Brodbeck M, Sennhauser M (2003) Root development and heavy metal phytoextraction efficiency: comparison of different plant species in the field. Plant Soil 249:67–81

Kemper D, Dabney S, Kramer L, Dominick D, Keep T (1992) Hedging against erosion. J Soil Water Conserv 47:284–288

Klang-Westin E, Eriksson J (2003) Potential of Salix as phytoextractor for Cd on moderately contaminated soils. Plant Soil 249:127–137

Knechtenhofer LA, Xifra IO, Scheinost AC, Fluhler H, Kretzschmar R (2003) Fate of heavy metals in a strongly acidic shooting-range soil: small-scale metal distribution and its relation to preferential water flow. J Plant Nutr Soil Sci Z Pflanzenernahr Bodenkd 166:84–92

Kort J, Collins M, Ditsch D (1998) A review of soil erosion potential associated with biomass crops. Biomass Bioenerg 14:351–359

Kosinska T, Baaga M (2007) Evaluation of pine's (Pinus sylvestris) usability in bioindication of the industrial pollution and in phytoremediation. Ann WULS-SGGW For Wood Technol 61:328–332

Kozlov MV, Haukioja E, Bakhtiarov AV, Stroganov DN, Zimina SN (2000) Root versus canopy uptake of heavy metals by birch in an industrially polluted area: contrasting behaviour of nickel and copper. Environ Pollut 107:413–420

Kretzschmar R, Borkovec M, Grolimund D, Elimelech M (1999) Mobile subsurface colloids and their role in contaminant transport. Adv Agron 66:121–193

Lai HY, Chen ZS (2004) Effects of EDTA on solubility of cadmium, zinc, and lead and their uptake by rainbow pink and vetiver grass. Chemosphere 55:421–430

Lai HY, Chen ZS (2005) The EDTA effect on phytoextraction of single and combined metals-contaminated soils using rainbow pink (Dianthus chinensis). Chemosphere 60:1062–1071

Laureysens I, Blust R, De Temmerman L, Lemmens C, Ceulemans R (2004) Clonal variation in heavy metal accumulation and biomass production in a poplar coppice culture: I. Seasonal variation in leaf, wood and bark concentrations. Environ Pollut 131:485–494

Lavado RS, Porcelli CA, Alvarez R (2001) Nutrient and heavy metal concentration and distribution in corn, soybean and wheat as affected by different tillage systems in the Argentine Pampas. Soil Till Res 62:55–60

Lehmann J, Joseph S (eds) (2009) Biochar for environmental management: science and technology. Earthscan, London

Lewandowski I, Scurlock JMO, Lindvall E, Christou M (2003) The development and current status of perennial rhizomatous grasses as energy crops in the US and Europe. Biomass Bioenergy 25:335–361

Lombi E, Zhao FJ, Dunham SJ, McGrath SP (2000) Cadmium accumulation in populations of Thlaspi caerulescens and Thlaspi goesingense. New Phytol 145:11–20

Lugon-Moulin N, Zhang M, Gadani F, Rossi L, Koller D, Krauss M, Wagner GJ (2004) Critical review of the science and options for reducing cadmium in tobacco (*Nicotiana tabacum* L.) and other plants. Adv Agron 83:111–180

Luo CL, Shen ZG, Li XD (2005) Enhanced phytoextraction of Cu, Pb, Zn and Cd with EDTA and EDDS. Chemosphere 59:1–11

Madejon P, Murillo JM, Maranon T, Cabrera F, Soriano MA (2003) Trace element and nutrient accumulation in sunflower plants two years after the Aznalcollar mine spill. Sci Total Environ 307: 239–257

Marchiol L, Fellet G, Perosa D, Zerbi G (2007) Removal of trace metals by Sorghum bicolor and Helianthus annuus in a site polluted by industrial wastes: a field experience. Plant Physiol Biochem 45:379–387

Marchiol L, Fellet G, Boscutti F, Montella C, Mozzi R, Guarino C (2013) Gentle remediation at the former “Pertusola Sud” zinc smelter: evaluation of native species for phytoremediation purposes. Ecol Eng 53:343–353

Margui E, Queralt I, Carvalho ML, Hidalgo M (2007) Assessment of metal availability to vegetation (Betula pendula) in Pb-Zn ore concentrate residues with different features. Environ Pollut 145:179–184

McGrath SP, Sidoli CMD, Baker AJM, Reeves RD (1993) The potential for the use of metal-accumulating plants for the in-situ decontamination of metal-polluted soils. Kluwer, Dordrecht

McGrath SP, Dunham SJ, Correll RL (2000) Potential for phytoextraction of zinc and cadmium from soils using hyperaccumulator plants. In: Terry N, Bañuelos G (eds) Phytoremediation of contaminated soil and water. Lewis, Boca Raton, FL, pp 109–128

McLaughlin SB, Walsh ME (1998) Evaluating environmental consequences of producing herbaceous crops for bioenergy. Biomass Bioenerg 14:317–324

Meers E, Ruttens A, Hopgood MJ, Samson D, Tack FMG (2005) Comparison of EDTA and EDDS as potential soil amendments for enhanced phytoextraction of heavy metals. Chemosphere 58:1011–1022

Meeinkuirt W, Pokethitiyook P, Kruatrachue M, Tanhan P, Chaiyarat R (2012) Phytostabilization of a Pb-contaminated mine tailing by various tree species in pot and field experiments. Int J Phytoremediation 14:925–938

Mench M, Tancogne J, Gomez A, Juste C (1989) Cadmium bioavailability to Nicotiana tabacum L., Nicotiana rustica L., and Zea mays L. grown in soil amended or not amended with cadmium nitrate. Biol Fertil Soils 8:48–53

Mertens J, Vervaeke P, De Schrijver A, Luyssaert S (2004) Metal uptake by young trees from dredged brackish sediment: limitations and possibilities for phytoextraction and phytostabilisation. Sci Total Environ 326:209–215

Mertens J, Van Nevel L, De Schrijver A, Piesschaert F, Oosterbaan A, Tack FMG, Verheyen K (2007) Tree species effect on the redistribution of soil metals. Environ Pollut 149:173–181

Migeon A, Richaud P, Guinet F, Chalot M, Blaudez D (2009) Metal accumulation by woody species on contaminated sites in the north of France. Water Air Soil Pollut 204:89–101

Mleczek M, Rissmann I, Rutkowski P, Kaczmarek Z, Golinski P (2009) Accumulation of selected heavy metals by different genotypes of *Salix*. Environ Exp Bot 66:289–296

Mok HF, Majumder R, Laidlaw WS, Gregory D, Baker AJM, Arndt SK (2013) Native Australian species are effective in extracting multiple heavy metals from biosolids. Int J Phytoremediation 15:615–632

Murakami M, Ae N (2009) Potential for phytoextraction of copper, lead, and zinc by rice (Oryza sativa L.), soybean (Glycine max [L.] Merr.), and maize (Zea mays L.). J Hazard Mater 162:1185–1192

Murakami M, Ae N, Ishikawa S (2007) Phytoextraction of cadmium by rice (Oryza sativa L.), soybean (Glycine max (L.) Merr.), and maize (Zea mays L.). Environ Pollut 145:96–103

Murillo JM, Maranon T, Cabrera F, Lopez R (1999) Accumulation of heavy metals in sunflower and sorghum plants affected by the Guadiamar spill. Sci Total Environ 242:281–292

Nehnevajova E, Herzig R, Bourigault C, Bangerter S, Schwitzguebel JP (2009) Stability of enhanced yield and metal uptake by sunflower mutants for improved phytoremediation. Int J Phytoremediation 11:329–346

Nogueira TAR, Franco A, He ZL, Braga VS, Firme LP, Abreu-Junior CH (2013) Short-term usage of sewage sludge as organic fertilizer to sugarcane in a tropical soil bears little threat of heavy metal contamination. J Environ Manage 114:168–177

Nowack B, Schulin R, Robinson BH (2006) Critical assessment of chelant-enhanced metal phytoextraction. Environ Sci Technol 40: 5225–5232

OECD (2010) The emerging middle class in developing countries

OECD/FAO (2011) OECD-FAO agricultural outlook 2011-2020

OECD-FAO (2012) OECD-FAO agricultural outlook 2012. OECD

Onwubuya K, Cundy A, Puschenreiter M, Kumpiene J, Bone B, Greaves J, Teasdale P, Mench M, Tlustos P, Mikhalovsky S, Waite S, Friesl-Hanl W, Marschner B, Muller I (2009) Developing decision support tools for the selection of “gentle” remediation approaches. Sci Total Environ 407:6132–6142

Papazoglou EG (2007) Arundo donax L. stress tolerance under irrigation with heavy metal aqueous solutions. Desalination 211:304–313

Papazoglou EG (2009) Heavy metal allocation in giant reed plants irrigated with metalliferous water. Fresenius Environ Bull 18:166–174

Perronnet K, Schwartz C, Gerard E, Morel JL (2000) Availability of cadmium and zinc accumulated in the leaves of *Thlaspi caerulescens* incorporated into soil. Plant Soil 227:257–263

Pilon-Smits E (2005) Phytoremediation. Annu Rev Plant Biol 56:15–39

Pimentel D, Krummel J (1987) Biomass energy and soil erosion: assessment of resource costs. Biomass 14:15–38

Quartacci MF, Argilla A, Baker AJM, Navari-Izzo F (2006) Phytoextraction of metals from a multiply contaminated soil by Indian mustard. Chemosphere 63:918–925

Raghu S, Wiedenmann RN, Blouin ML (2006) Biofuels as invasive species? University of Arkansas Newswire. http://newswire.uark.edu/article.aspx?id=916105.09

Raskin I, Smith RD, Salt DE (1997) Phytoremediation of metals: using plants to remove pollutants from the environment. Curr Opin Biotechnol 8:221–226

Reeves RD, Baker AJM (2000) Phytoextraction of mine wastes – options and impossibilities. In: Raskin I, Ensley BD (eds) Phytoremediation of toxic metals. Using plants to clean up the environment. Wiley, New York, pp 193–230

Reglero MM, Monsalve-González L, Taggart MA, Mateo R (2008) Transfer of metals to plants and red deer in an old lead mining area in Spain. Sci Total Environ 406:287–297

Robinson BH, Leblanc M, Petit D, Brooks RR, Kirkman JH, Gregg PEH (1998) The potential of Thlaspi caerulescens for phytoremediation of contaminated soils. Plant Soil 203:47–56

Robinson B, Fernandez JE, Madejon P, Maranon T, Murillo JM, Green S, Clothier B (2003a) Phytoextraction: an assessment of biogeochemical and economic viability. Plant Soil 249:117–125

Robinson B, Green S, Mills T, Clothier B, van der Velde M, Laplane R, Fung L, Deurer M, Hurst S, Thayalakumaran T, van den Dijssel C (2003b) Phytoremediation: using plants as biopumps to improve degraded environments. Aust J Soil Res 41:599–611

Robinson B, Schulin R, Nowack B, Roulier S, Menon M, Clothier B, Green S, Mills T (2006) Phytoremediation for the management of metal flux in contaminated sites. Forest Snow Landsc Res 80:221–234

Robinson BH, Green SR, Chancerel B, Mills TM, Clothier BE (2007) Poplar for the phytomanagement of boron contaminated sites. Environ Pollut 150:225–233

Robinson BH, Banuelos G, Conesa HM, Evangelou MWH, Schulin R (2009) The phytomanagement of trace elements in soil. Crit Rev Plant Sci 28:240–266

Rosselli W, Keller C, Boschi K (2003) Phytoextraction capacity of trees growing on a metal contaminated soil. Plant Soil 256:265–272

Roth FAI (1989) Thinning to improve pine timber. Cooperative extension service – Fact sheet 5001. University of Arkansas, United States Department of Agriculture, and County Governments Cooperating

Rotkittikhun P, Chaiyarat R, Kruatrachue M, Pokethitiyook P, Baker AJM (2007) Growth and lead accumulation by the grasses *Vetiveria zizanioides* and *Thysanolaena maxima* in lead-contaminated soil amended with pig manure and fertilizer: a glasshouse study. Chemosphere 66:45–53

Roulier S, Robinson B, Kuster E, Schulin R (2008) Analysing the preferential transport of lead in a vegetated roadside soil using lysimeter experiments and a dual-porosity model. Eur J Soil Sci 59:61–70

Sabudak T, Seren G, Kaykioglu G, Dincer AR (2007) Determination of copper, zinc and lead contents in sunflower plants. J Environ Prot Ecol 8:101–106

Salazar MJ, Rodriguez JH, Nieto GL, Pignata ML (2012) Effects of heavy metal concentrations (Cd, Zn and Pb) in agricultural soils near different emission sources on quality, accumulation and food safety in soybean [Glycine max (L.) Merrill]. J Hazard Mater 233–234:244–253

Scheid S, Gunthardt-Goerg MS, Schulin R, Nowack B (2009) Accumulation and solubility of metals during leaf litter decomposition in non-polluted and polluted soil. Eur J Soil Sci 60:613–621

Shukla OP, Juwarkar AA, Singh SK, Khan S, Rai UN (2011) Growth responses and metal accumulation capabilities of woody plants during the phytoremediation of tannery sludge. Waste Manage 31:115–123

Simberloff D (2008) Invasion biologists and the biofuels boom: cassandras or colleagues. Weed Sci 56:867–872

Solhi M, Shareatmadari H, Hajabbasi M (2005) Lead and zinc extraction potential of two common crop plants, Helianthus annuus and Brassica napus. Water Air Soil Pollut 167:59–71

Strojan CL (1978) Forest leaf litter decomposition in vicinity of a zinc smelter. Oecologia 32:203–212

Sullivan P (2004) Sustainable soil management: soil systems guide. ATTRA National Sustainable Agriculture Information Service, Fayetteville, AR

Tangahu BV, Abullah SRS, Basri H, Idris M, Anuar N, Mukhlish M (2011) A review on heavy metals (As, Pb, and Hg) uptake by plants through phytoremediation. Int J Chem Eng 2011:1–31

Tilman D, Balzer C, Hill J, Befort BL (2011) Global food demand and the sustainable intensification of agriculture. Proc Natl Acad Sci USA 108:20260–20264

TWB (2012) Arable land (hectares per person). The World Bank Group. http://data.worldbank.org/indicator/AG.LND.ARBL.HA.PC. Accessed 8 June 2012

UNEP (2002) State of the environment and policy retrospective: 1972–2002 - land. In: Clarke R, Lamb R, Ward DR (eds) GEO - global environmental outlook 3 - past, present and future perspectives. Earthscan, London, pp 62–89

UNEP (2012) Part 1: State and trends of the environment – land. In: Ullstein B, de Mattos H, Hawkins C, McMullen CP, Jabbour J, Scharlemann J (eds) GEO – global environmental outlook 5 – environment for the future we want, pp 65–96

Unterbrunner R, Puschenreiter M, Sommer P, Wieshammer G, Tlustos P, Zupan M, Wenzel WW (2007) Heavy metal accumulation in trees growing on contaminated sites in Central Europe. Environ Pollut 148:107–114

Vamerali T, Bandiera M, Coletto L, Zanetti F, Dickinson NM, Mosca G (2009) Phytoremediation trials on metal- and arsenic-contaminated pyrite wastes (Torviscosa, Italy). Environ Pollut 157:887–894

Vamerali T, Bandiera M, Mosca G (2010) Field crops for phytoremediation of metal-contaminated land. A review. Environ Chem Lett 8:1–17

Van Nevel L, Mertens J, Oorts K, Verheyen K (2007) Phytoextraction of metals from soils: how far from practice? Environ Pollut 150:34–40

Van Nevel L, Mertens J, Staelens J, De Schrijver A, Tack FMG, De Neve S, Meers E, Verheyen K (2011) Elevated Cd and Zn uptake by aspen limits the phytostabilization potential compared to five other tree species. Ecol Eng 37:1072–1080

van Zwieten L, Kimber S, Morris S, Downie A, Berger E, Rust J, Scheer C (2010) Influence of biochars on flux of N2O and CO2 from Ferrosol. Aust J Soil Res 48:555–568

Vidic T, Greilhuber J, Vilhar B, Dermastia M (2009) Selective significance of genome size in a plant community with heavy metal pollution. Ecol Appl 19:1515–1521

Vose JM, Harvey GJ, Elliott KJ, Clinton BD (2003) Measuring and modeling tree and stand level transpiration. In: McCutcheon SC, Schnoor JL (eds) Phytoremediation: transformation and control of contaminants. Wiley, New York, pp 263–282

Watanabe ME (1997) Phytoremediation on the brink of commercialization. Environ Sci Technol 31:182–186

Welch RM (2002) The impact of mineral nutrients in food crops on global human health. Plant Soil 247:83–90

Welch RM, Graham RD (1999) A new paradigm for world agriculture: meeting human needs - Productive, sustainable, nutritious. Field Crop Res 60:1–10

Wenger K, Gupta SK, Furrer G, Schulin R (2002) Zinc extraction potential of two common crop plants, *Nicotiana tabacum* and *Zea mays*. Plant Soil 242:217–225

White P, Broadley MR, Bowen HC, Johnson SE (2007) Selenium and its relationship with sulfur. In: Hawkesford MJ, De Kok LJ (eds) Sulfur in plants. Springer, Dordrecht, pp 225–252

Whiting SN, Reeves RD, Richards D, Johnson MS, Cooke JA, Malaisse F, Paton A, Smith JAC, Angle JS, Chaney RL, Ginocchio R, Jaffré T, Johns R, McIntyre T, Purvis OW, Salt DE, Schat H, Zhao FJ, Baker AJM (2004) Research priorities for conservation of metallophyte biodiversity and their potential for restoration and site remediation. Restor Ecol 12:106–116

Wilhelm WW, Johnson JME, Karlen DL, Lightle DT (2007) Corn stover to sustain soil organic carbon further constrains Biomass supply. Agron J 99:1665–1667

Williams JD, Gollany HT, Siemens MC, Wuest SB, Long DS (2009) Comparison of runoff, soil erosion, and winter wheat yields from no-till and inversion tillage production systems in northeastern Oregon. J Soil Water Conserv 64:43–52

Wu LH, Luo YM, Xing XR, Christie P (2004) EDTA-enhanced phytoremediation of heavy metal contaminated soil with Indian mustard and associated potential leaching risk. Agric Ecosyst Environ 102:307–318

Xia HP (2004) Ecological rehabilitation and phytoremediation with four grasses in oil shale mined land. Chemosphere 54:345–353

Xia HL, Yan ZJ, Chi XY, Cheng WW (2009) Evaluation of the phytoremediation potential of Saccharum officinarum for Cd-contaminated soil. IEEE Computer Society, Los Alamitos, CA

Yuan L, Yin X, Zhu Y, Li F, Huang Y, Liu Y, Lin Z (2012) Selenium in plants and soils, and selenosis in Enshi, China: implications for selenium biofortification. In: Yin X, Yuan L (eds) Phytoremediation and biofortification. Springer, Dordrecht, pp 7–31

Zan CS, Fyles JW, Girouard P, Samson RA (2001) Carbon sequestration in perennial bioenergy, annual corn and uncultivated systems in southern Quebec. Agric Ecosyst Environ 86:135–144

Zhao F-J, McGrath SP (2009) Biofortification and phytoremediation. Curr Opin Plant Biol 12:373–380

Zhuang P, Li Z-A, Zou B, Xia H-P, Wang G (2013) Heavy metal contamination in soil and soybean near the Dabaoshan Mine, South China. Pedosphere 23:298–304