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REVIEW

Metal accumulation and tolerance in wetland plants

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Abstract This paper briefly reviews the progress in studies of wetland plants in terms of heavy metal pollution. The current research mainly includes the following areas: (1) metal uptake, translocation, and distributions in wetland plants and toxicological effects on wetland plants, (2) radial oxygen loss (ROL) of wetland plants and its effects on metal mobility in rhizosphere soils, (3) constitutional metal tolerance in wetland plants, and (4) mechanisms of metal tolerance by wetland plants. Although a number of accomplishments have been achieved, many issues still remain unanswered. The future research effort is likely to focus on the ROL of wetland plants affecting metal speciation and bioavailability in rhizosphere soils, and the development of rhizosphere management technologies to facilitate and improve practical applications of phytoremediation of metal-polluted soils.

Keywords metal tolerance, wetland plants, mobility, radial oxygen loss (ROL)

1 Introduction

Metal mining has caused severe heavy metal pollution to adjacent ecosystems; especially mine tailings that have become potential sources of pollution caused by wind and water erosion (Wong, 2003). Restoration of a vegetation cover can provide benefits for environmental stabilization and pollution control. Revegetation of mine tailings generally takes place under terrestrial conditions (Saxena et al., 1999). Recently, a number of studies have verified that constructed wetlands could purify mine drainage water and establish a dense vegetative cover on mine tailings (Ye et al., 2004). As a major component in constructed wetland treatment systems, plants play an important role in metal removal *via* filtration, adsorption, and cation exchange, and through plant-induced chemical changes in the rhizosphere (Dunbabin and Bowmer, 1992; Wright and Otte, 1999). The plants chosen for constructed wetlands are usually metal tolerant, fast growing, and of high biomass, such as *Typha latifolia* and *Phragmites australis* (Ye et al., 1997a, b). Previous studies indicated that many wetland plants could colonize both uncontaminated and heavily metal-polluted areas (Deng et al., 2004). This provides important hints in choosing plant materials in constructed wetlands. Plant biodiversity in constructed wetlands should be enhanced to benefit both purification efficiency and ecology (Tilman, 1996).

Therefore, there is an urgent need to study more wetland plants in order to provide materials for constructed wetlands. In this paper, we review the studies on (1) metal uptake, translocation and distribution in wetland plants, (2) the effect of wetland plants on metal mobility of rhizosphere soil, and (3) metal tolerance in wetland plants.

2 Metal uptake, translocation, and distributions in wetland plants and toxicological effects on wetland plants

2.1 Metal uptake, translocation, and distribution

Wetland plants can accumulate heavy metals in their tissues. Previous studies indicated that some wetland plants have the ability to take up> 0.5% dry weight (DW) of a given element and bioconcentrate the element in its tissues to 1000-fold the initial element supply concentration. For instance, duck weed (*Lemna minor*) and water hyacinth (*Eichhornia crassipes*) are excellent accumulators of Cd (6000–130000 $\mathrm{mg \cdot kg^{-1}}$ DW) and Cu (6000–7000 $\mathrm{mg \cdot kg^{-1}}$ DW) (Zayed et al., 1998; Zhu et al., 1999). Other wetland plants can tolerate high concentrations of several metals in their tissues, which do not show negative effects on plant growth. Water zinnia (*Wedelia trilobata*) and smartweed (*Polygonum hydropiperoides*) accumulated

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148 $mg \cdot kg^{-1}$ Cd, and 95 $mg \cdot kg^{-1}$ Cu, respectively, in their shoot tissues without negative effects (Qian et al., 1999).

Many wetland plants accumulated higher concentrations of metals in roots than in shoots (Taylor and Crowder, 1984; Ye et al., 1997a, b; Cheng et al., 2002; Stoltz and Greger, 2002; Deng et al., 2004, 2006). The pattern of metal distribution in wetland plants is suitable to restrict metals being transported from roots to shoots. However, the degree of upward translocation is dependent on plant species, particular metal and a number of environmental conditions, such as pH, redox potential (Eh), temperature, and salinity (Chawla et al., 1991; Fitzgerald et al., 2003; Fritioff et al., 2005). In addition, other factors, such as soil particle size, organic matter content, nutrients, and the presence of other ions may also influence metal uptake by wetland plants (Salomons and Förstner, 1984; Greger, 1999). Lead uptake into the roots and shoots of rice (*Oryza sativa*) decreased with an increase in redox potential and pH, while Cd uptake increased with an increase in Eh and a decrease in pH (Reddy and Patrick, 1977). Otte et al. (1991) found that salt marsh dicots tended to have similar heavy metal concentrations in roots and shoots, whereas in monocots, the concentrations in the roots were two to three times higher than in the shoots. Fitzgerald et al. (2003) reported that Cu accumulated mainly in the roots of monocots but in the shoots of dicots. For *Aster tripolum*, Pb accumulated mostly in the roots at low salinity, while at higher salinity, a greater proportion of Pb was transported to the shoots (Fitzgerald et al., 2003). Some metals (e.g., Zn and Cu) seem to be transported easily in the plant, while other metals (e.g., Pb) appear to be rather immobile (Pitchel et al., 2000). Probably, Zn and Cu are essential to plants, while Pb is not.

2.2 Toxicological effects

Metals accumulated in wetland plant tissues can cause toxic effects, especially when translocated to aboveground tissues. Some studies focused on plant growth, showing their responses to heavy metals. When exposed to higher metal concentrations, plants showed a significant reduction in root elongation, height of seedling, leaf number, leaf area, shoot, and root biomass (Snowden and Wheeler, 1993; Ye et al., 1997a, b; Mendelssohn et al., 2001; MacFarlane and Burchett, 2002). Other studies focused on the effects of metals on the biochemical or physiological characteristics. Padinha et al. (2000) measured thiolic protein concentration, adenylate energy charge (AEC), and photosynthetic efficiency in *Spartina maritima* growing at sites with differing degrees of metal contamination. They found that thiolic protein, which binds metals, was higher in plants grown at metal-contaminated sites than at 'clean' sites. Metals bound to thiolic proteins were subsequently excreted back to the environment, maintaining lower concentrations in the leaves. Mendelssohn et al. (2001) also found photosynthesis efficiency and AEC ratio lowered in *Typha domingensis* and *Spartina alterniflora* in response to increasing levels of Cd, while chlorophyll fluorescence was relatively insensitive. These indices, reflecting toxic responses, were considered to be useful for quantifying the health of plants and could potentially be used as biomarkers.

2.3 Relation with sediment

A number of studies have analyzed the correlations between metal uptake by wetland plants and metal concentrations in sediment. Most studies showed poor correlations between them (Campbell et al., 1985; Jackson and Kalff, 1993; Cardwell et al., 2002). A possible explanation for the lack of relation is that most studies of wetland plants focused on the correlations between metals in plants and the total metal concentrations in sediment, rather than estimating the bioavailable metal concentrations (Campbell et al., 1985). The factor controlling metal uptake by plants may be metal speciation, which is determined by the physical and chemical properties of the soil. Gambrell et al. (1994) found that pH and redox potential affect Pb, Cd, and Hg accumulation in marsh plants. Jackson and kalff. (1993) also reported that low pH is favorable for metal accumulation in rooted wetland plants. In addition, soil Eh also plays an important role in determining the bioavailability of metals to wetland plants. The greatest metal mobility likely exists within a zone ($-150 < \mathrm{Eh} < 200$ mv) between strongly reduced and strongly oxidized sediments (Jackson, 1998). A greater uptake and availability of Cd was observed in a number of wetland plants under dry conditions (higher Eh) than under flooded (lower Eh) conditions (Gambrell, 1994). However, Ye et al. (1998) found that *P. australis* takes up more Pb, Zn, and Cd under flooded conditions than under dry conditions.

3 Radial oxygen loss (ROL) of wetland plants and its effects on metal mobility in rhizosphere soils

3.1 Radial oxygen loss

The greatest difference between wetland and dryland plants is the ability of rooted wetland plants to survive in saturated soil. In order to tolerate anoxic wetland environment, much plant transported oxygen to roots is not only used for respiration but also diffused to the rhizosphere [termed radial oxygen loss (ROL)] and forms an oxidative layer around the root surfaces (Armstrong et al., 1992). This oxidative layer prevents plant absorption of phytotoxic reduced substrates such as Fe^{2+}, Mn^{2+}, and sulfide (Armstrong and Armstrong, 1988; Conlin and Crowder, 1989; St-Cyr and Crowder, 1989; Christensen et al., 1994).

The ability of plants to oxygenate rhizosphere soils is determined by the oxygen demand of the surrounding medium, size of the plant root mass, permeability of the roots, and some environmental factors such as light (Chabbi et al., 2000; Colmer, 2003; Weis and Weis, 2004; Peter et al., 2005).

3.2 Effects on metal mobility in rhizosphere soils

The biogeochemical environment of waterlogged soils is generally favorable for immobilization of metals (Gambrell, 1994). A number of studies indicated that ROL could increase metal mobility (Vigneault et al., 2001; Jacob and Otte, 2004). Wetland plants can directly affect two metal mobility-regulating soil factors: redox potential (Eh) and pH *via* ROL into the medium surrounding the roots (Armstrong et al., 1992; Wigand et al., 1997; Wright and Otte, 1999). ROL usually results in the oxygenation of rhizosphere soils and thereby an increase in the sediment Eh (Wright and Otte, 1999). Some studies report increased soluble Fe concurrent with higher Eh in the root zone of waterlogged soils when compared with unvegetated plots (Chen and Barko, 1988; Wright and Otte, 1999). Rhizosphere oxidization may also lead to a decrease in pH (Kirk and Bajita, 1995), which could explain the mobilization of metals independent of changes in redox status of the soil (Jacob and Otte, 2003). However, the potential effect of wetland plants on soil Eh and pH and, thus, metal mobility, through ROL, depends on the characteristics of wetland plants and soil conditions. For example, *Glyceria fluitans* have little effect on sediment chemistry (Wright and Otte, 1999). Therefore, the effect of wetland plants on rhizosphere is species specific, which is possibly due to the varied abilities of ROL (Brix, 1993).

However, other studies showed that ROL could decrease metal mobility (Otte et al., 1989; St-Cyr and Campbell, 1996; Doyle and Otte, 1997). Another striking feature of ROL is the formation of root iron (Fe) plaques (Armstrong, 1967). The formation of Fe plaque is influenced by soil Fe availability and the oxidizing capacity of plant root (Christensen and Sand-Jensen, 1998; Hansel et al., 2002). Because of the high-absorption capacity of iron oxides, iron plaque provides a reactive substrate for metal sequestration and translocation, such as Zn, Pb, Cu, and As (Greipsson and Crowder, 1992; Ye et al., 1998; Liu et al., 2004). Metals like Zn have a high affinity for absorption and coprecipitation with Fe hydroxides, and thus, if these mobilized metals come near the Fe plaque formed on the plant roots, they become immobilized on the root surface (Otte et al., 1989; St-Cyr and Campbell, 1996). The bulk soil thus acts as a source of metals, while the Fe plaque acts as a sink, creating a diffusion gradient driven by the oxidation process in the rhizosphere (Doyle and Otte, 1997). The net effects of the presence of wetland plants and ROL on metal mobility need to be further clarified.

4 Metal tolerance in wetland plants

4.1 Tolerance and evolution

Most experimental studies of heavy metal tolerance confirm the fundamental tenet that populations surviving in metal contaminated habitats are differentiated from 'normal' populations of the same species by possessing genetically based tolerances (Antonovics et al., 1971). Ecotypes from contaminated sites show better growth on contaminated substrates than those from uncontaminated sites, while ecotypes from uncontaminated sites generally perform better on uncontaminated substrates than those from contaminated sites (Antonovics et al., 1971; Antonovics, 1975). However, the basis of metal tolerance in wetland plants seems to be different from that of dryland plants. Comparisons of populations of *T. latifolia* and *P. australis* from metal enriched sites with those from nonpolluted sites have shown that they have a similar degree of metal tolerance (McNaughton et al., 1974; Taylor and Crowder, 1983, 1984; Ye et al., 1997a, b). *Glyceria fluitans* (McCabe and Otte, 2000; Matthews et al., 2004), *Eriophorum angustifolium* (Matthews et al., 2004), *Carex rostrata* (Matthews et al., 2005), and *Leersia hexandra* (Deng et al., 2006) have also been found to possess innate metal tolerance. The information available supports the assumption that wetland plants, regardless of their origins, are able to grow in high-metal concentrations. This has led to the conclusion that wetland plants have an innate tolerance to heavy metals (McCabe et al., 2001). However, the reason why wetland plants would be innately tolerant to metals remains unclear. A possible explanation may be the biogeochemistry of the rhizosphere soils of wetland plants (Matthews et al., 2005). Wetland soils are reduced and anaerobic, but the rhizosphere soils of wetland plants are often aerobic and oxidized. This, perhaps, facilitates the mobilization of metals, which may lead to an enrichment of metals in the soils immediately surrounding the roots of wetland plants (Doyle and Otte, 1997) as compared to dryland plants. It is, therefore, possible that wetland plants are typically exposed to higher concentrations of metals than dryland plants, which has led to the evolution of tolerance in some wetland plants. However, more wetland plants need to be studied in order to provide more evidence for the presence of innate tolerance to heavy metals in wetland plants.

4.2 Tactics to tolerate heavy metals by wetland plants

4.2.1 Restricting upward movement into shoots and translocation of excessive metals into old leaves before shedding or secreting excessive metals by special organs such as salt glands

A number of studies have proved the pattern of metal distribution of wetland plants (Taylor and Crowder, 1984;

Ye et al., 1997a, b; Cheng et al., 2002; Stoltz and Greger, 2002; Deng et al., 2004). Some studies show that *Avicennia marina* and *Spartina alterniflora* can excrete metals in salt crystals released through hydathodes (salt glands) (Kraus, 1988; MacFarlane and Burchett, 2000).

4.2.2 Sequestering heavy metals in organs or subcellular compartments with little or no sensitive metabolic activity

Vesk et al. (1999) found that Cu and Zn mainly localized inside cells of roots of free-floating water hyacinth (*E. crassipes*), with a less but significant amount in the stele cell wall, whereas Zn and Cu increased centripetally in the stele cell walls. Matthews et al. (2004) reported that Zn was sequestered within the protoplasts of *G. fluitans* but was not associated with the cell walls, implying that the vacuole was also an important site for the accumulation of metals in some wetland plants. In the roots of grey mangrove (*Avicennia marina*), metals (Cu, Pb and Zn) were concentrated predominantly in cell walls (MacFarlane and Burchett, 2000).

4.2.3 Synthesizing phytochelatins (PCs), peptides and exudates to chelate-free metal ions or increasing antioxidant enzyme activities

Fediuc and Erdei (2002) studied the physiological and biochemical aspects of Cd protective mechanisms induced in *P. australis* and *T. latifolia*. Different defense strategies were operated by the two species under Cd stress. In *Typha*, the increasing accumulation of Cd was in positive correlation with the increases in free thiol content, while in *Phragmites* glutathione reductase, catalase, and peroxidase activities were increased. The authors concluded that *Typha* relies more on thiol induction and metal binding for metal avoidance, while *Phragmites* is based on increased antioxidant enzyme activities and thus scavenging of active oxygen species. Yang et al. (2000) indicated that oxalate played an important role in Pb tolerance in rice (*O. sativa*). The oxalate content in the root and root exudates increased upon Pb treatment in the tolerant varieties, whereas the opposite was observed for the sensitive ones. It was concluded that oxalate release might contribute toward the mechanism of Pb detoxification in rice. Ederli et al. (2004) found that the total PC production in *P. australis* increased after exposure to Cd, suggesting a pivotal role for phytochelatins in the sequestration of metal.

4.2.4 Presence of microbial symbionts in rhizosphere soils

De Souza et al. (1999) found that, when bacteria were inhibited with antibiotics, plants (*Scripus robustus* and *Polypogon monspeliensis*) accumulated lower concentrations of Se and Hg. Mycorrhizae provide an interface between roots and rhizosphere soil. Lakatos et al. (1999) reported that the presence of periphyton associated with the root of *P. australis* in freshwater wetlands enhanced the ability of the reed to accumulate and retain metals. However, Khan et al. (2000) suggested that mycorrhizae play a protective role, restricting the uptake of metals by plants through immobilizing metals in the fungal tissues.

4.2.5 Iron plaque as a barrier to heavy metals

A number of studies have suggested that Fe plaque may act as a barrier to toxic metals. Otte et al. (1989) found that a heavy coating of Fe plaque may act as a barrier to Zn uptake in *Aster tripolium*, but when the coating was light, Zn uptake was enhanced. Greipsson (1994) reported that Fe plaque on *O. sativa* seedlings improved growth under mildly toxic conditions of Cu and Ni exposure. Iron plaques have also been shown to act as a filter for the movement of Fe, Cu, Zn, Ni, and Cd into rhizomes and shoots. This may be achieved by adsorption onto Fe compounds or coprecipitation (Bostick et al., 2001; Hansel et al., 2002). However, other studies have shown that Fe plaque does not impede the uptake of toxic metals although it can immobilize metals (Ye et al., 1997c, 1998). Batty et al. (2000) found that when *P. australis* grew in higher pH (6.0) conditions, Fe plaques on the roots reduced the uptake of Cu and Mn but did not prevent from it. Therefore, the role of Fe plaques on wetland plants in metal uptake and tolerance remains unresolved.

5 Conclusions and research needs

Heavy metals represent a pollutant of major concern throughout the world. Substantial progress has been achieved during the last decades to understand the interactions between heavy metals and wetland plants. However, only a few literatures are available that particularly address transformation processes and interactions of heavy metals in the rhizosphere soils of wetland plants.

Future investigations related to wetland plants and heavy metals should include (1) the roles and mechanisms of ROL, Fe plaque, and microorganisms in metal uptake and tolerance in wetland plants based on rhizobox-type experiments and field trials rather than solution culture studies, (2) effects of combined heavy metal pollutions on wetland plants, and (3) development of rhizosphere management technologies (e.g., based on knowledge obtained in rhizobox studies) in order to facilitate and improve practical applications of phytoremediation of metal-polluted soils.

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