Ecotoxicology and Environmental Safety 143 (2017) 92–101

Contents lists available at ScienceDirect

Ecotoxicology and Environmental Safety

journal homepage: www.elsevier.com/locate/ecoenv

# Comparative analysis of element concentrations and translocation in three wetland congener plants: *Typha domingensis*, *Typha latifolia* and *Typha angustifolia*

Giuseppe Bonanno[a,*], Giuseppe Luigi Cirelli[b]

[a] Department of Biological, Geological and Environmental Sciences, University of Catania, Via Longo 19, 95125 Catania, Italy
[b] Department of Agriculture, Nutrition and Environment, University of Catania, Via Santa Sofia 100, 95123 Catania, Italy

A R T I C L E I N F O

*Keywords:*
Cattails
Wetlands
Wastewater
Sediments
Heavy metals
Sicily

A B S T R A C T

This study analyzed the concentrations and distributions of Al, As, Cd, Cr, Cu, Hg, Mn, Ni, Pb and Zn in three different cattail species growing spontaneously in a natural wetland subject to municipal wastewater and metal contamination. The cattail species included *Typha domingensis*, *T. latifolia* and *T. angustifolia*. Results showed that all *Typha* species have similar element concentrations in roots, rhizomes and leaves, and similar element mobility from sediments to roots and from roots to leaves. This study corroborated three patterns of *Typha* species growing in metal contaminated environments: high tolerance to toxic conditions, bulk element concentrations in roots, and restricted element translocation from roots to leaves. This study showed that three different *Typha* species respond similarly to metal inputs under the same polluting field conditions. Given their similar metal content and similar biomass size, our results suggest that *T. domingensis*, *T. latifolia* and *T. angustifolia* may have comparable capacity of phytoremediation. High element uptake and large biomass make *Typha* species some of the best species for phytoremediation of metal contaminated environments.

## 1. Introduction

Heavy metals are a particular group of elements that, unlike organic pollutants, cannot be degraded through biological processes (Muhammad et al., 2009). Heavy metals are especially harmful in aquatic ecosystems where, once accumulated in bottom sediments, they begin to move up the food chain, often biomagnifying at higher trophic levels and ultimately causing various chronic and acute disorders in humans and animals (Gall et al., 2015). A major source of heavy metals is wastewater from agriculture due to herbicides and pesticides, and wastewater from industrial manufacture for the presence of raw materials (Dhote and Dixit, 2009). Numerous techniques have been developed to reduce the concentrations of heavy metals in the environment (e.g., chemical precipitation, membrane filtration), but most of them, although effective, proved expensive and non-ecofriendly (Olguín and Sánchez-Galván, 2012; Lara et al., 2014). As a result, the increasing need for remediation of contaminated sites led to develop cost-effective and eco-friendly biotechnologies like phytoremediation, which relies on naturally occurring plant species to extract, sequester and detoxify pollutants (Ali et al., 2013; Pandey et al., 2015; Rezania et al., 2016). Heavy metals, in particular, can be removed from contaminated soil, sludge, sediments and water, thanks to particular plants that uptake the pollutants through roots and translocate them to the upper parts of the plant (Wójcik et al., 2014; Sharma et al., 2015).

Wetland plants (or macrophytes) are widely investigated and used to treat metal contaminated surface water or remediate sites with elevated soil metal concentrations (Bonanno and Lo Giudice, 2010; Vymazal, 2011; Pandey, 2012; Bhatia and Goyal, 2014). Macrophytes, indeed, can filter out toxic metals from the surface water, often immobilize them in their sediments, and on the long term, return them to the geological part of the earth cycle (Odum, 2000; Prasad et al., 2006). Macrophytes suitable for phytoremediation are highly tolerant to heavy metal stress, have fast growth and large biomass, and can accumulate high metal concentrations in their tissues (Deng et al., 2004; Liu et al., 2016). Numerous studies showed also that uptake, accumulation and translocation of heavy metals may differ significantly among plant species (e.g., Fitzgerald et al., 2003; Yoon et al., 2006; Qian et al., 2012). Knowing the differences in heavy metal concentrations between macrophytes is thus fundamental not only to assess the suitability of one species instead of another for phytoremediation uses, but also to implement appropriate actions of ecological restoration and management, thus making phytoremediation more sustainable (Tack

* Corresponding author.
*E-mail address:* bonanno.giuseppe@unict.it (G. Bonanno).

http://dx.doi.org/10.1016/j.ecoenv.2017.05.021
Received 16 March 2017; Received in revised form 8 May 2017; Accepted 11 May 2017
Available online 16 May 2017
0147-6513/ © 2017 Elsevier Inc. All rights reserved.

and Vandecasteele, 2008; Bonanno et al., 2013; Pandey et al., 2016).

Cattail species *Typha domingensis* Pers., *Typha latifolia* L., and *Typha angustifolia* L. are common, perennial and emergent macrophytes distributed worldwide in tropic and temperate wetlands, lakes and rivers (Smith, 1987). They prefer damp soils and shallow, slow and brackish waters, and can quickly dominate a wetland plant community by forming monotypic stands (Panich-Pat et al., 2004). *Typha* ssp. are herbaceous and rhizomatous plants that can be over 3-m high (Pignatti, 1982). Given their fast growth, large biomass, high tolerance to metal-contaminated sites, and high element uptake, *T. domingensis*, *T. latifolia*, and *T. angustifolia* have been widely and successfully employed for several phytoremediation uses such as phytostabilization, phytoextraction and water treatment in constructed wetlands (Leto et al., 2013; Mufarrege et al., 2014; Gomes et al., 2014; Pandey et al., 2014). Despite the numerous studies on the heavy metal phytoremediation capacities of *T. domingensis*, *T. latifolia*, and *T. angustifolia*, to date, it has not been possible to claim with enough scientific evidence whether or not such cattail species have similar accumulation and translocation capacities of heavy metals. This study analyzed the concentrations of nine heavy metals (Al, Cd, Cr, Cu, Hg, Mn, Ni, Pb and Zn) and one metalloid (As) in roots, rhizomes and leaves of *T. domingensis*, *T. latifolia*, and *T. angustifolia* growing spontaneously in a natural sewage- and metal-contaminated wetland. This study aimed to shed further light on the possible phytoremediation similarities of such cattails by comparing their element mobility and concentrations under the same polluting field conditions.

## 2. Materials and methods

### 2.1. Study area

The study area was an urban natural wetland located near the port of Catania (Sicily, Italy), (37°29′17.69″N; 15°05′09.54″E) (Fig. 1). Catania is the second largest town of Sicily with 315,000 inhabitants that rise up to 770,000 people if the metropolitan area is included. The study wetland is the estuary of a 6-km watercourse, which was channelized to receive the domestic discharges from Catania, and is mainly affected by municipal wastewaters, road run-off and dumping from adjacent industrial activities. In this wetland, *T. domingensis*, *T. latifolia* and *T. angustifolia* grow spontaneously and form dense mono-species populations (Fig. 2). This area acts as a constructed wetland and is subject to constant polluting inputs all the year round. The average annual flow is 1.5 $m^3/s$ whereas annual rainfall and temperature are respectively 610 mm and 19.0 °C.

**Fig. 2.** Photo of sampling plots.

### 2.2. Sampling

Sampling was conducted in one-off trips conducted in April and


**Fig. 1.** Study area.

October 2014, and repeated in the same months in 2015. Five sampling plots per each of the three species were randomly selected in an area of 1000 m × 100 m. The average sampling plot was a quadrate of 4 m × 4 m, and contained a plant community characterized by the monospecies stand of one of the study *Typha* species. In each sampling plot, four mature *Typha* individuals, four samples of sediments and up to 10.0 L of water were collected. *Typha* individuals were 2-meter high on average, and partially submerged at a variable depth of 10–50 cm from the bottom. *Typha* individuals were delicately and wholly uprooted with stainless steel tools, cleaned with linen cloths to remove extraneous materials (e.g. gross ground particles), and put in sterilized airtight plastic bags. Sediment samples were instead collected from the top 30 cm of the upper layer through a Plexiglas corer (internal diameter 10 cm), and put in 1.0-L polyethylene bottles. Water samples were collected within a radius of 0.50 m from each collected plant sample, through 1.0-L sterilized glass bottles, and at a variable depth of 0.30–0.50 m from the bottom to the water surface. All samples were finally gathered in PVC containers and kept at 5 ± 1 °C until laboratory analysis.

### 2.3. Chemical analysis

The total concentrations of Al, As, Cd, Cr, Cu, Hg, Mn, Ni, Pb, Zn were analyzed in sediments and *Typha* ssp. organs. In the laboratory, the individuals of the three *Typha* species were first washed thoroughly in running tap water to avoid any surface contamination, and then rinsed with bidistilled water, obtained through a Milli-Q® Gradient distiller, to remove any further residual material on the surface. Plant samples were then dissected into roots, rhizomes and leaves, and together with sediment samples, were put in a refrigerator at 4 °C until processing. The average weight of root, rhizome, leaf and sediment samples was 2.0 kg. After defrosting, leaves, roots, rhizomes and sediment were dried to constant weight at room temperature. Once dry, plant samples were ground and homogenized in an agate mortar to ensure a homogeneous element distribution in the powder. Sediment samples were passed through a 1 mm diameter sieve. These homogenous powder and sieved sediment were then weighed at 0.1 ± 0.05 g, and oven-digested at 90 °C overnight (microwave oven Mars 6, CEM Corporation) in an acid solution ($H_2O_2/HNO_3$, 2:3 ratio; Carlo Erba) in Teflon digestion vessels. Water samples were acidified with 63% $HNO_3$ to pH ≤2, before filtering through a filter paper 2.0 µm (Whatman® GF/A glass microfiber filters). After digestion, the solid residue of plant and sediment samples was separated by centrifugation before being filtered through a filter paper 2.0 µm. Finally, supernatants were diluted with ultrapure Milli-Q water to a final volume of 25 mL (into a volumetric flask), and transferred for chemical analysis via ICP-MS (Cd, Cr, Cu, Ni, Pb, Zn), ICP-OES (Al, Fe) and FAAS (As and Hg) (respectively through PerkinElmer Elan® 6000, PerkinElmer® Optima™ 8000, PerkinElmer® AAnalyst™ 400 AA Spectrometer). The element used as internal standards was rhodium (Rh). Quality control was performed through stability of instrumental recalibration and using analytical blanks. The instruments were cyclically checked against the low level standards (once every five samples) and recalibrated either when signs of drift were noted or after every 5 samples. The validity and precision of the analytical procedures were assessed by analyzing the standard reference material *Lagarosiphon major* (Institute for Reference Materials and Measurements, IRMM, BCR® no 060).

Student's *t*-test ($\alpha = 0.05$) was used to assess whether analyzed values for the reference material were in good agreement with the certified values. The percent recovery showed values between 85% and 104%. Instrument detection limits were expressed as three times the standard deviation from the mean blank. Analyses were conducted in triplicates.

### 2.4. Statistical processing

Statistical analysis was preliminarily conducted through the Shapiro-Wilk test to determine the normal distribution of the data sets, and through the Levene's test to check the hypothesis of homoscedasticity. If both tests were accepted, a two-way analysis of variance (ANOVA) was applied to assess the possible significant influence of organs and seasons on element concentrations in *Typha* species. Specifically, the factors organ and season had respectively three and two levels: root, rhizome, leaf; spring and autumn. Tukey's *post hoc* test was performed to identify which specific mean pairs differ significantly. In case of rejection of ANOVA assumptions, data were log-transformed. Possible relationships between plant species and sediments were checked with Student's *t*-test. The level of significance was set at 0.05. Statistical processing was carried out with the statistical package IBM SPSS Version 22.0.

Bioconcentration and translocation factors were determined to assess element mobility in the study species. The values obtained were based on the following:

$$Bioconcentration\ Factor\ (BCF)\ =\ C_{root}/C_{sediment}$$

where $C_{sediment}$ and $C_{root}$ are respectively the concentrations (mg $kg^{-1}$ DW) of a given element in sediment and roots of the study species. BCF expresses the efficiency of a plant species to take up a specific element from sediments and accumulate it in its tissues. Higher BCF values imply a greater bioaccumulation capability (EPA, 2007). A BCF value higher than one may indicate that a plant species could act as a hyperaccumulator of trace elements (Zhang et al., 2002).

$$Translocation\ factors\ (TF):\ C_{leaf}/C_{root}$$

where $C_{leaf}$ and $C_{root}$ are respectively the concentrations (mg $kg^{-1}$ DW) of a given element in leaves and roots of the study species. TF provides information about the mobility of a given element from roots to leaves, and higher TF values result in a greater capacity of mobility (Deng et al., 2004).

## 3. Results and discussion

Results showed that element concentrations in water were constant during spring and autumn (Table 1). According to the Italian law (Italian Decree 260/2010), Hg concentrations in waters were significantly higher than quality limits, and As concentrations were closer to the legal threshold. Similarly, the element content in sediments was relatively constant in the study seasons (Table 1). Only Cd and Cr concentrations in sediments showed significant differences between spring and autumn. However, the different concentrations of Cd and Cr reported the same order of magnitude, suggesting a generally stable release of heavy metals into the environment. The metals Cd, Hg and Pb were the only elements to pass abundantly the legal limits of metal concentrations in sediments (Italian Decree 260/2010). Regarding plant organs, all *Typha* species showed generally similar element concentrations (Table 2), suggesting that overall *T. domingensis*, *T. latifolia* and *T. angustifolia* have comparable capacities of element uptake. All *Typha* species showed also the same patterns of element concentrations in their organs. Specifically, element concentrations decreased consistently in all *Typha* species in the order of root > rhizome > leaf. The higher element content in underground organs is a general trend of several macrophytes, including *Typha* sp., growing in metal-contaminated wetlands (Odum, 2000; Sasmaz et al., 2008; Bonanno, 2011, 2012; Vymazal, 2011; Lyubenova et al., 2013). The fact that the elements are mainly accumulated in roots may indicate a high content of mobile chemical species in sediments but it may also suggest the existence of tolerance strategies that prevent toxic levels in roots from moving to the aboveground plant organs (Hozhina et al., 2001; Bonanno et al., 2017). In particular, the protective mechanisms at the aboveground-underground boundary may be supposed to be mainly

**Table 1**
Element concentrations in waters [μg/L] and sediments [mg/kg] of the study area, and legal limits (Italian Decree 260/2010).

| | Water | | | Sediment | | |
|---|---|---|---|---|---|---|
| | Spring | Autumn | Legal quality limits | Spring | Autumn | Legal quality limits |
| Al | 0.21 ± 0.04 | 0.15 ± 0.02 | – | 24,770 ± 4340 | 23,450 ± 3290 | – |
| As | 7.04 ± 2.12 | 8.77 ± 1.59 | 10 | 8.56 ± 1.52 | 9.67 ± 1.88 | 12.0 |
| Cd | 0.38 ± 0.08 | 0.45 ± 0.09 | – | 0.89 ± 0.15* | 0.56 ± 0.11* | 0.30 |
| Cr | 1.21 ± 0.24 | 0.95 ± 0.17 | 7 | 23.8 ± 4.08* | 30.4 ± 5.21* | 50.0 |
| Cu | 18.7 ± 3.51 | 25.4 ± 5.12 | – | 109 ± 16.5 | 121 ± 18.6 | – |
| Hg | 0.15 ± 0.03 | 0.10 ± 0.02 | 0.06 | 0.37 ± 0.06 | 0.43 ± 0.07 | 0.30 |
| Mn | 0.13 ± 0.02 | 0.09 ± 0.01 | – | 890 ± 167 | 1010 ± 186 | – |
| Ni | 3.43 ± 0.65 | 5.27 ± 1.12 | 20.0 | 24.9 ± 4.21 | 27.2 ± 4.56 | 30.0 |
| Pb | 2.56 ± 0.45 | 3.68 ± 0.64 | 7.20 | 57.3 ± 8.32 | 68.5 ± 10.3 | 30.0 |
| Zn | 1.67 ± 0.31 | 1.35 ± 0.26 | – | 319 ± 52.4 | 290 ± 51.2 | – |

* Indicates significant differences between concentrations in spring and autumn for the same element.

activated in case of pronounced gradients as found in this study for the toxic elements As, Cd and Pb whose concentrations in *Typha* roots were one order of magnitude greater than leaves. The bulk element concentrations in roots, with lower concentrations in rhizomes and leaves, found by Lyubenova et al. (2013) for *T. latifolia*, are in line with As, Cd and Pb concentrations reported in all *Typha* species of this study. This result may indicate exclusion from above-ground tissues as the main tolerance mechanism in *Typha* species, as suggested for several rooted and free floating macrophytes (Sawidis et al., 1995; Chandra and Yadav, 2010; Mufarrege et al., 2010). Metal tolerance by plants, and heavy metal detoxification may be achieved through compartmentalization and metal complexation with ligands such as organic acids, amino acids and certain members of the mugineic acids occurring in plant tissues (Hall, 2002; Carrier et al., 2003).

Results showed also a positive correlation between roots and sediments for all elements in *Typha* species. Regarding rhizomes, the correlation with elements in sediments was significant for most elements, except for As and Cd for all *Typha* species, Hg for all *Typha* species in spring, and Cr for *T. domingensis* and *T. angustifolia* in spring. Statistically significant correlations between sediments and roots may suggest the prominent role of bottom sediments as the main source of elements for wetland plants. This finds general confirmation in previous studies according to which elements taken by rooted macrophytes are mainly sediment derived (Sawidis et al., 1995; Aksoy et al., 2005). Compared to roots and rhizomes, the leaves of *Typha* species showed a significant correlation with fewer elements in sediments, specifically only with Al, Cu, Mn, and Zn. This leaf-sediment pattern was consistent with all *Typha* species. Previous studies showed that metal accumulation in aboveground plant tissues are mainly determined by metal availability in the soil, species-specific uptake, translocation processes and the involvement of the metal in biological functions (Kabata-Pendias and Mukherjee, 2007; Teuchies et al., 2013). Various studies found that several species growing in metal-contaminated places, accumulate high concentrations in their roots but the metal concentra-

**Table 2**
Element concentrations in *Typha* species [mg/kg].

| | | *Typha domingensis* | | | *Typha latifolia* | | | *Typha angustifolia* | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | Root | Rhizome | Leaf | Root | Rhizome | Leaf | Root | Rhizome | Leaf |
| Al | Spring | 1756 ± 350[1,a] | 920 ± 146[a] | 38.0 ± 10.2[*,1,a] | 1780 ± 325[1,a] | 845 ± 162[*,a] | 48.5 ± 12.3[*,2,a] | 1568 ± 340[*,2,a] | 821 ± 144[a] | 36.1 ± 7.21[*,1,a] |
| | Autumn | 1890 ± 310[a] | 850 ± 188[1,a] | 50.9 ± 15.2[*,1,a] | 1740 ± 288[a] | 1055 ± 201[*,2,a] | 38.3 ± 8.81[*,2,a] | 1865 ± 378[*,a] | 962 ± 166[2,a] | 44.6 ± 8.10[*,1,2,a] |
| As | Spring | 3.21 ± 0.51[1,a] | 1.29 ± 0.23 | 0.10 ± 0.02[1] | 1.87 ± 0.32[2] | 1.21 ± 0.22 | 0.08 ± 0.01[*,1] | 2.86 ± 0.41[3,a] | 1.06 ± 0.18 | 0.05 ± 0.01[2] |
| | Autumn | 2.78 ± 0.42[1,a] | 1.34 ± 0.25 | 0.08 ± 0.02[1] | 2.21 ± 0.40[2,a] | 1.65 ± 0.25 | 0.12 ± 0.02[*,2] | 1.95 ± 0.35[2] | 1.42 ± 0.20 | 0.06 ± 0.01[1] |
| Cd | Spring | 0.61 ± 0.11[*,1,a] | 0.15 ± 0.02[1] | 0.08 ± 0.02[*,1] | 0.46 ± 0.08[2,a] | 0.16 ± 0.03[*,1] | 0.08 ± 0.01[1] | 0.38 ± 0.07[*,3] | 0.20 ± 0.03[*,2] | 0.04 ± 0.01[2] |
| | Autumn | 0.44 ± 0.08[*,a] | 0.18 ± 0.03[1] | 0.05 ± 0.01[*] | 0.39 ± 0.07[a] | 0.22 ± 0.03[*,1] | 0.06 ± 0.01 | 0.51 ± 0.10[*,a] | 0.10 ± 0.01[*,2] | 0.04 ± 0.01 |
| Cr | Spring | 3.67 ± 0.71[*,1,a] | 3.01 ± 0.52[*,1] | 1.05 ± 0.15[*] | 5.54 ± 1.10[2,a] | 3.24 ± 0.61[1,a] | 1.01 ± 0.21 | 4.26 ± 0.75[3,a] | 1.89 ± 0.32[*,2] | 0.91 ± 0.14 |
| | Autumn | 5.88 ± 1.02[*,1,a] | 4.57 ± 0.88[*,1,a] | 1.24 ± 0.20[*,1] | 6.75 ± 1.20[2,a] | 3.85 ± 0.53[2,a] | 0.95 ± 0.18[1] | 5.15 ± 1.12[3,a] | 2.48 ± 0.41[*,3,a] | 0.75 ± 0.11[2] |
| Cu | Spring | 18.5 ± 3.52[a] | 12.7 ± 2.10[a] | 4.67 ± 0.98[*,1,a] | 12.8 ± 2.10[a] | 9.87 ± 1.88[a] | 5.87 ± 1.06[2,a] | 17.4 ± 3.21[a] | 13.7 ± 2.52[a] | 5.21 ± 0.92[3,a] |
| | Autumn | 15.2 ± 3.02[1,a] | 10.4 ± 2.35[a] | 3.50 ± 0.65[*,a] | 13.1 ± 2.36[1,a] | 11.8 ± 2.06[a] | 4.66 ± 0.95[a] | 18.8 ± 4.06[2,a] | 12.5 ± 2.06[a] | 4.04 ± 0.82[a] |
| Hg | Spring | 3.21 ± 0.52[1,a] | 2.02 ± 0.31[1] | 0.97 ± 0.12[1] | 2.88 ± 0.35[*,2,a] | 1.55 ± 0.20[2] | 0.63 ± 0.10[2] | 1.98 ± 0.32[*,3,a] | 1.01 ± 0.16[*,3] | 0.55 ± 0.08[*,2] |
| | Autumn | 3.67 ± 0.35[1,a] | 2.56 ± 0.25[1,a] | 0.85 ± 0.10[1] | 3.35 ± 0.41[*,1,a] | 1.83 ± 0.26[2,a] | 0.49 ± 0.08[2] | 2.75 ± 0.35[*,2,a] | 1.96 ± 0.25[*,3,a] | 0.35 ± 0.06[*,3] |
| Mn | Spring | 151 ± 25.4[1,a] | 83.8 ± 12.3[1,a] | 51.2 ± 8.35[*,1,a] | 132 ± 25.2[2,a] | 103 ± 18.3[*,2,a] | 29.7 ± 6.21[2,a] | 95.8 ± 15.5[*,3,a] | 77.6 ± 13.4[*,1,a] | 31.6 ± 5.32[3,a] |
| | Autumn | 138 ± 22.2[1,a] | 74.2 ± 13.11[1,a] | 32.1 ± 6.12[*,a] | 155 ± 20.2[2,a] | 70.1 ± 12.3[*,1,a] | 41.0 ± 7.35[a] | 126 ± 22.5[*,3,a] | 103 ± 17.5[*,2,a] | 36.0 ± 6.21[a] |
| Ni | Spring | 36.6 ± 6.01[*,a] | 29.6 ± 4.23[*,a] | 10.9 ± 2.10[1] | 41.2 ± 7.45[a] | 28.5 ± 4.21[a] | 10.3 ± 1.86[1] | 35.7 ± 6.25[a] | 20.2 ± 3.56[a] | 12.3 ± 2.25[2] |
| | Autumn | 53.3 ± 7.56[*,1,a] | 38.7 ± 6.31[*,1,a] | 10.8 ± 1.87 | 35.6 ± 7.01[2,a] | 30.2 ± 5.31[1,a] | 8.42 ± 1.51 | 28.8 ± 5.21[3,a] | 21.6 ± 2.98[2,a] | 8.96 ± 1.73 |
| Pb | Spring | 13.7 ± 2.35[1,a] | 4.21 ± 0.73[a] | 0.71 ± 0.11 | 15.2 ± 3.12[2,a] | 4.32 ± 0.74[a] | 0.52 ± 0.10 | 8.90 ± 1.48[3,a] | 3.25 ± 0.57[a] | 0.75 ± 0.13 |
| | Autumn | 10.9 ± 1.94[1,a] | 4.33 ± 0.65[a] | 0.65 ± 0.10 | 13.5 ± 2.06[2,a] | 6.65 ± 1.32[a] | 0.44 ± 0.09 | 10.2 ± 1.95[1,a] | 5.23 ± 0.96[a] | 0.52 ± 0.10 |
| Zn | Spring | 118 ± 18.7[a] | 97.3 ± 15.3[1,a] | 38.8 ± 5.63[a] | 110 ± 18.6[a] | 103 ± 21.4[1,a] | 41.0 ± 7.21[*,a] | 107 ± 18.2[a] | 88.7 ± 13.2[*,2,a] | 39.1 ± 5.21[*,a] |
| | Autumn | 122 ± 20.5[1,a] | 103 ± 17.5[1,a] | 35.4 ± 6.05[1,a] | 115 ± 15.8[1,a] | 96.5 ± 17.6[2,a] | 28.7 ± 4.75[*,2,a] | 86.8 ± 14.6[2,a] | 68.0 ± 11.5[*,3,a] | 20.2 ± 3.56[*,3,a] |

Note: different numbers indicate different concentrations between the same organ of different species in the same season ($p < 0.05$; two-way ANOVA followed Tukey's post hoc test).
[a] Indicates significant correlation between plant organ and sediment ($p < 0.05$; Student's *t*-test).
* Indicates significant differences between the same organ of the same species in the two different seasons ($p < 0.05$; two-way ANOVA followed by Tukey's post hoc test).

**Table 3**
Bioconcentration factor (BCF) and translocation factor (TF) in *Typha* species.

| | | | *Typha domingensis* | *Typha latifolia* | *Typha angustifolia* | Mean |
|---|---|---|---|---|---|---|
| **Al** | **Spring** | **BCF** | 0.07 | 0.07 | 0.06 | 0.07 |
| | | **TF** | 0.02 | 0.03 | 0.02 | 0.02 |
| | **Autumn** | **BCF** | 0.08 | 0.07 | 0.08 | 0.08 |
| | | **TF** | 0.03 | 0.02 | 0.02 | 0.02 |
| **As** | **Spring** | **BCF** | 0.38 | 0.22 | 0.33 | 0.31 |
| | | **TF** | 0.03 | 0.04 | 0.02 | 0.03 |
| | **Autumn** | **BCF** | 0.29 | 0.23 | 0.20 | 0.24 |
| | | **TF** | 0.03 | 0.05 | 0.03 | 0.04 |
| **Cd** | **Spring** | **BCF** | 0.69 | 0.52 | 0.43 | 0.55 |
| | | **TF** | 0.13 | 0.17 | 0.10 | 0.13 |
| | **Autumn** | **BCF** | 0.79 | 0.70 | 0.91 | 0.80 |
| | | **TF** | 0.11 | 0.15 | 0.08 | 0.11 |
| **Cr** | **Spring** | **BCF** | 0.15 | 0.23 | 0.18 | 0.19 |
| | | **TF** | 0.29 | 0.18 | 0.21 | 0.23 |
| | **Autumn** | **BCF** | 0.19 | 0.22 | 0.17 | 0.19 |
| | | **TF** | 0.21 | 0.14 | 0.15 | 0.17 |
| **Cu** | **Spring** | **BCF** | 0.17 | 0.12 | 0.16 | 0.15 |
| | | **TF** | 0.25 | 0.46 | 0.30 | 0.34 |
| | **Autumn** | **BCF** | 0.13 | 0.11 | 0.16 | 0.13 |
| | | **TF** | 0.23 | 0.36 | 0.21 | 0.27 |
| **Hg** | **Spring** | **BCF** | 8.68 | 7.78 | 5.35 | 7.27 |
| | | **TF** | 0.30 | 0.22 | 0.28 | 0.27 |
| | **Autumn** | **BCF** | 8.53 | 7.79 | 6.40 | 7.57 |
| | | **TF** | 0.23 | 0.15 | 0.13 | 0.17 |
| **Mn** | **Spring** | **BCF** | 0.17 | 0.15 | 0.11 | 0.14 |
| | | **TF** | 0.34 | 0.23 | 0.33 | 0.30 |
| | **Autumn** | **BCF** | 0.14 | 0.15 | 0.12 | 0.14 |
| | | **TF** | 0.23 | 0.26 | 0.29 | 0.26 |
| **Ni** | **Spring** | **BCF** | 1.47 | 1.65 | 1.43 | 1.52 |
| | | **TF** | 0.30 | 0.25 | 0.34 | 0.30 |
| | **Autumn** | **BCF** | 1.78 | 1.31 | 1.06 | 1.38 |
| | | **TF** | 0.22 | 0.24 | 0.31 | 0.26 |
| **Pb** | **Spring** | **BCF** | 0.24 | 0.27 | 0.16 | 0.22 |
| | | **TF** | 0.05 | 0.03 | 0.08 | 0.05 |
| | **Autumn** | **BCF** | 0.16 | 0.20 | 0.15 | 0.17 |
| | | **TF** | 0.06 | 0.03 | 0.05 | 0.05 |
| **Zn** | **Spring** | **BCF** | 0.37 | 0.34 | 0.33 | 0.35 |
| | | **TF** | 0.33 | 0.37 | 0.36 | 0.35 |
| | **Autumn** | **BCF** | 0.42 | 0.39 | 0.30 | 0.37 |
| | | **TF** | 0.29 | 0.25 | 0.23 | 0.26 |

Note: BCF = [root]/[sediment]) and TF = [leaf]/[root].

tion in their leaves is relatively constant, whatever the concentrations in sediments (e.g. Jiménez et al., 2011). Such species have tolerance strategies that restrict metal accumulation in the above-ground parts of the plant. As a result, plant species are able to adapt and grow in highly polluted environments (Fischerová et al., 2006). This restricted element translocation was also found in the *Typha* species of this study, especially for the toxic elements As, Cd and Pb. In turn, the high values of micronutrient Cu, Mn and Zn in leaves and their significant correlation with sediments, may be due to the fact that these elements have a preferential uptake because essential elements of many enzyme systems, and thus needed in a larger amount (Kabata-Pendias, 2011). The higher phytoavailability for Cu, Mn and Zn, as observed in this study, is found in many systems, including tidal marshes and may be also caused by their lower sorption capacity to soil components (Gambrell, 1994; Overesch et al., 2007).

Regarding element mobility (Table 3), bioconcentration (BCF) and translocation factors (TF) showed overall similar values in all *Typha* species. BCF and TF, however, were significantly different among elements, and showed these decreasing trends in the three *Typha* species (mean values):

- BCF (spring): Hg > Ni > Cd > Zn > As > Pb > Cr > Cu > Mn > Al
- BCF (autumn): Hg > Ni > Cd > Zn > As > Cr > Pb > Mn > Cu > Al
- TF (spring): Zn > Cu > Mn = Ni > Hg > Cr > Cd > Pb > As > Al
- TF (autumn): Cu > Zn = Mn = Ni > Hg = Cr > Cd > Pb > As > Al

BCF trend was generally similar in both seasons whereas TF values showed in fact the same trend in spring and autumn. Moreover, the trends of BCF and TF were significantly different, implying that the mobility of same element in sediment-root boundary may differ from root-shoot translocation. The distribution of elements in plant organs depends on the bulk plant concentrations, the biochemical properties of the element in question, and the plant itself (Lyubenova et al., 2013). Greger (1999), in particular, observed that metal translocation can occur in the phloem, via the apoplast, and via the xylem, acropetally. BCF values were higher than TF for most elements. This was true for Al, As, Cd, Hg, Ni and Pb. In particular, BCF of Hg reported by far the highest value (> 7.0); other than Hg, Ni was the only element with BCF > 1. Values < 0.10 were found in BCF and TF of Al, and in TF of As and Pb. BCF values > 1 corroborated previous findings on the use of *Typha* species for phytostabilization (Zu et al., 2005). In general, plants having BCF greater than one and TF less than one show phytostabilization potential and can be used as for rhizoremediation (Yoon et al., 2006; Pandey, 2012). This is particularly true in those emergent tall macrophytes (e.g. *Typha* ssp, *Phragmites australis*), which contribute to metal sequestration thanks to their extensive root system (e.g., Salt et al., 1998; Otte et al., 2004; Weis and Weis, 2004; Duarte et al., 2010). TF is an important indicator to identify the phytostabilization potential of a plant species. TF values less than one mean the plant having poor translocation efficiency of metals from root to leaf, and thus can be used for phytostabilization purposes (Ma et al., 2001). In this study, TF values of *Typha* species were lower than one for all elements, thus suggesting poor element translocation efficiency from roots to leaves.

The results of this study showed that *T. domingensis*, *T. latifolia* and *T. angustifolia* have levels of element concentrations overall in agreement with numerous studies on *Typha* spp. growing in element-contaminated wetlands (Table 4). However, compared to another important and widespread macrophyte, namely *Phragmites australis* (common reed), the studied *Typha* species showed lower levels of element concentrations (Table 4). Here follows an analysis of element by element for the studied *Typha* species.

Aluminum showed the highest concentrations in sediments and plants, but also the lowest values of BCF and TF. Al thus proved the least mobile element in plants, in line with previous findings (Baker et al., 1994; Bonanno, 2013). Although the physiological function of Al in plants is not clear, studies support evidence that modest levels of Al may have beneficial effects on plant growth (Kabata-Pendias, 2011). Arsenic is a non-essential metalloid, considered as one of the most toxic elements due to its persistence in the environment and tendency to bioaccumulate (Kapaj et al., 2006). In this study, As concentrations in *Typha* species accumulated predominantly in roots, exceeding the content in leaves by one order of magnitude, in agreement with previous studies (Hozhina et al., 2001). In case of high contamination, *Typha* species may also accumulate great quantities of As without showing symptoms of toxicity (Lyubenova et al., 2013).

Cadmium is a widespread heavy metal pollutant, extremely toxic to humans and most plants (Prasad, 1995). Wetlands are especially subject to Cd contamination because Cd can enter the aquatic environment from natural as well as anthropogenic sources such as industrial effluent and agricultural runoff (Li et al., 2008). Moreover, Cd cannot be removed from water by self-purification (Liu et al., 2016). The problem of Cd contamination can be faced using green and cost-effective techniques like phytoremediation. Cd concentrations in the study *Typha* plants were generally in line with Cd-contaminated wet-

**Table 4**
Element concentrations in *T. domingensis*, *T. latifolia*, *T. angustifolia* and *P. australis* in element polluted wetlands from various studies (ppm).

| Plant species | Reference | Al | As | Cd | Cr | Cu | Hg | Mn | Ni | Pb | Zn |
|---|---|---|---|---|---|---|---|---|---|---|---|
| *T. domingensis* | This study | 38.0–1890 | 0.08–3.21 | 0.05–0.61 | 1.05–5.88 | 3.50–18.5 | 0.85–3.67 | 32.1–151 | 10.8–53.3 | 0.65–13.7 | 35.4–122 |
| | Debusk et al. (1996) | – | – | 10–600 | – | – | – | – | – | 10–1200 | – |
| | Cardwell et al. (2002) | – | – | 0.07–2.70 | – | 8.57–135 | – | – | – | 0.52–224 | 68.3–1078 |
| | Bonanno (2013) | 44.7–1711 | < 0.10–2.50 | < 0.10 | 0.89–5.44 | 2.41–16.4 | 0.87–11.4 | 26.2–108 | 8.86–48.4 | < 0.10–7.59 | 23.4–126 |
| | Teles Gomes et al. (2014) | – | – | – | – | – | 0.18–273 | – | – | – | – |
| | Eid et al. (2012) | – | – | 4.50–10.0 | – | 3.0–19.5 | – | – | – | 68–105 | 14–63 |
| | Maine et al. (2017) | – | – | – | 10.0–764 | – | – | – | 6.0–200 | – | 14.0–200 |
| | Mendoza-Carranza et al. (2016) | – | – | – | 0.31 ± 0.44 | – | – | – | 1.37 ± 0.30 | 1.24 ± 1.76 | 9.62 ± 2.70 |
| | Maine et al. (2009) | – | – | – | 25–250 | – | – | – | 10–350 | – | 12.5–180 |
| | Howitt et al. (2014) | 9.50–830 | 0.07–3.60 | < 0.01–0.09 | 0.08–2.90 | 0.83–12.0 | < 0.01 | – | 0.25–4.30 | 0.13–13.0 | 8.70–53.0 |
| *T. latifolia* | This study | 38.3–1780 | 0.08–2.21 | 0.06–0.46 | 0.95–6.75 | 4.66–13.1 | 0.49–3.35 | 29.7–155 | 8.42–41.2 | 0.44–15.2 | 28.7–115 |
| | Deng et al. (2004) | – | – | 5.00–20.0 | – | 30.0–40.0 | – | – | – | 120–3300 | 200–3000 |
| | Sasmaz et al. (2008) | – | – | 0.05–0.75 | 10.0–58.0 | 18.0–90.0 | – | 400–1800 | 15.0–98.0 | 4.00–20.0 | 150–680 |
| | Pandey et al. (2014) | – | – | 2.00–10.0 | 2.00–40.0 | 10.0–50.0 | – | 80.0–190 | 10.0–40.0 | 2.00–25.0 | 75.0–130 |
| | Salem et al. (2014) | < 1.50–10,300 | < 1.50–151 | < 0.50–1.60 | 1.00–61.3 | 1.00–53.8 | – | 55.4–20,000 | < 1.00–45.3 | – | < 1.00–175 |
| | Rai et al. (2015) | – | 9.00–36.0 | – | 15.0–58.0 | 20.0–58.0 | – | 60.0–180 | 10.0–28.0 | 10.0–62.0 | 140–230 |
| | Feng et al. (2016) | – | – | – | – | 9.35–16.0 | – | 11.2–33.6 | – | 12.6–21.3 | 13.9–43.9 |
| | Engin et al. (2017) | – | – | – | – | – | – | 59.1–594 | 107 ± 81.7 | – | 90.3–294 |
| *T. angustifolia* | This study | 36.1–1865 | 0.05–2.86 | 0.04–0.51 | 0.75–5.15 | 4.04–18.8 | 0.35–2.75 | 31.6–126 | 8.96–35.7 | 0.52–10.2 | 20.2–107 |
| | Demirezen and Aksoy (2004) | – | – | 0.03–0.80 | 0.80–8.00 | 1.00–28.0 | – | – | 1.00–20.0 | 0.50–7.00 | 10.0–100 |
| | Deng et al. (2004) | – | – | 1.00–5.00 | – | 50–500 | – | – | – | 3.00–5.00 | 10–100 |
| | Samecka-Cymerman and Kempers (2001) | 54.0 ± 3.00 | – | 0.50 ± 0.10 | 3.40 ± 0.20 | 6.10 ± 0.50 | – | 59.0 ± 5.00 | 1.10 ± 0.10 | 6.00 ± 0.40 | 14.0 ± 0.70 |
| | Yadav et al. (2012) | – | – | – | 15.0–841 | 15.0–826 | – | – | 21.0–980 | – | 30.0–1343 |
| | Laffont-Schwob et al. (2015) | – | – | 0.003–0.008 | – | 0.50–2.20 | – | – | – | 0.33–0.62 | 1.75–8.00 |
| *P. australis* | Stoltz and Greger (2002) | – | 1.00–32.5 | 1.00–4.60 | – | 6.40–80.1 | – | – | – | 4.10–523 | 68.0–1310 |
| | Deng et al. (2004) | – | – | < 1.00–5.00 | – | 5.00–60.0 | – | – | – | 40.0–2300 | 180–1400 |
| | Bragato et al. (2006) | – | – | – | < 1.00–120 | 5.00–14.0 | – | – | < 1.00–65.0 | – | 8.00–14.0 |
| | Quan et al. (2007) | – | – | – | – | < 5.00–70.0 | – | – | – | 2.00–28.0 | 20.0–140 |
| | Vymazal et al. (2009) | 181–11,900 | 0.15–4.95 | 0.01–0.21 | 0.13–14.0 | 4.89–38.0 | 0.01–0.06 | 52.0–266 | 1.18–16.7 | 0.08–7.10 | 20.5–86.0 |
| | Bonanno and Lo Giudice (2010) | – | – | 0.68–1.13 | 0.40–6.97 | 2.31–14.9 | 1.05–5.22 | 27.9–475 | 0.48–9.12 | 9.87–16.5 | 10.0–104 |
| | Obolewski et al. (2011) | – | – | 1.35–2.10 | 4.10–11.4 | 1.50–11.1 | – | – | 3.10–6.10 | 14.4–26.0 | 8.60–44.4 |
| | Phillips et al. (2015 | – | – | – | – | 2.38–39.5 | – | – | – | 0.57–4.33 | 14.7–51.7 |
| | Bonanno et al. (2017) | – | 0.17–3.17 | 0.57–1.36 | 0.73–4.12 | 4.75–18.2 | 0.27–0.91 | 44.5–558 | 0.79–4.78 | 0.35–7.78 | 15.3–144 |

Concentrations are expressed as range values or mean values ± SD in plant tissues.

lands (Hozhina et al., 2001). This result strengthens the previous studies on the use of *Typha* species as phytoremediators given their tolerance to high Cd contamination. Cd-tolerant plants generally have also higher Cd concentrations in roots than in leaves (Gussarsson, 1994), in line with this study. Xu et al. (2011) conducted a study on *T. angustifolia* tolerance to Cd exposure, and found that root Casparian band, cell wall, vacuole, glutathione (GSH), and glutathione peroxidase (GPX) play important roles in Cd detoxification; mechanisms of Cd detoxification differ in leaf cell cytoplasm (mainly a GSH-related antioxidant defense system) and root cell cytoplasm (mainly a GSH-related chelation system). Xu et al. (2011) concluded that *T. angustifolia* has multiple detoxification mechanisms for Cd and acts as a promising species for phytoremediation of Cd-polluted environments. Xu et al. (2011) investigated Cd detoxification in *T. angustifolia* but the results of our study suggest that *T. domingensis* and *T. latifolia* may have similar tolerance mechanisms given their comparable values of concentrations and mobility of Cd.

Chromium is generally considered toxic for plants because it may alter N metabolism by affecting protein formation (Chatterjee and Chatterjee, 2000). Cr concentrations in all study *Typha* species were in line with Cr-contaminated wetlands (Samecka-Cymerman and Kempers, 2001; Demirezen and Aksoy, 2004). In particular, this study found that Cr concentrations in all *Typha* species were higher than Cr toxic threshold in plants ($>1.0\ mg\ kg^{-1}$), according to Markert (1992). However, no toxicity signs were observed in the study *Typha* species. Pais and Jones (2000) reported, indeed, that there may be some evidence of toxic effects on plants for Cr concentrations higher than $10\ mg\ kg^{-1}$. In general, species of *Typha* genus show high tolerance to Cr contamination (Sasmaz et al., 2008). Nichel is another dangerously toxic metal that may affect plant growth, metabolism and physiology (Kabata-Pendias, 2011). In this study, Ni concentrations in all *Typha* species were in line with values detected in contaminated environments (Pandey et al., 2014). Ni concentrations in *Typha* species exceed the level of $5\ mg\ kg^{-1}$, considered toxic to most plants (Allen, 1989). However, similarly to Cr, despite the high levels of Ni, *Typha* species showed no symptoms of toxicity.

Mercury is one of the most dangerous natural elements on Earth (Boening, 2000), and may be converted into toxic forms as methyl–-mercury, easily assimilated by aquatic and terrestrial biota (Bargagli, 1998). Mercury is particularly dangerous in aquatic ecosystems, which are characterized by a greater number of trophic levels. In wetlands, indeed, inorganic Hg can be converted into organic forms, which are more toxic, and thus, the processes of biomagnification are even higher (Morel et al., 1998). This study showed that *Typha* species have remarkably high values of BCF for Hg (BCF > 7.0) compared to other elements. These results recommend the use of *Typha* species for Hg stabilization, in line with previous studies (Willis et al., 2010; Gomes et al., 2014). Hg concentrations in study *Typha* species were within the phytotoxic range of $1–3\ mg\ kg^{-1}$, according to Massa et al. (2010), but no species showed clear effects of toxicity. This study found also high levels of Hg in the leaves of all *Typha* species, a pattern identified by previous studies in Hg-contaminated sites (Lominchar et al., 2015). Our results specifically found higher Hg levels in *Typha* leaves ($0.35–0.97\ mg\ kg^{-1}$), as compared to normal levels ($0.04\ mg\ kg^{-1}$) in wetland plants growing in areas with no Hg pollution (Moore et al., 1995). Millán et al. (2014) analyzed the Hg concentrations in the leaves of *Typha domingensis*, *Phragmites australis*, *Flueggea tinctoria*, *Tamarix canariensis*, and *Nerium oleander*, in a Spanish mercury mining district. These authors found the highest concentrations of Hg in the leaves of *T. domingensis*, whose values were in line with all *Typha* species of this study. These results suggest that *Typha* species may be suitable for multiple options of Hg phytoremediation, including phytoextraction.

Lead is not an essential element for plant metabolism and is considered among the most toxic metals, even at low concentrations (Prasad, 2004; Kabata-Pendias and Mukherjee, 2007). Lead is relatively immobile in soil and tends to accumulate in roots, with a consequent low translocation into the above-ground organs (Siedlecka et al., 2001; Fischerová et al., 2006; Carranza-Álvarez et al., 2008). Moreover, Pb concentrations found in leaves may be often the result of wind contamination rather than soil concentrations (Dalenberg and van Driel, 1990). According to Blaylock et al. (1997), Pb shows limited solubility in soil water and limited availability for plant uptake due to complexation with organic matter, sorption on oxides and clays, and precipitation as carbonates, hydroxides and phosphates. Our results corroborated these general trends of *Typha* genus found in previous studies: tolerance to Pb-contaminated soils, Pb bulk concentrations in roots, Pb restricted translocation into leaves. The ability of wetland species like *Typha* ssp. to accumulate high Pb concentrations in roots was demonstrated by numerous studies (e.g., Ye et al., 1998; Panich-Pat et al., 2004). In particular, it has been proposed that the presence of iron plaques on root surface of wetland species may act as a barrier to the uptake of potentially phytotoxic metals into plant tissues owing to adsorption and immobilization of metals by iron plaques (Greipsson, 1994; Wang and Peverly, 1996). Panich-Pat et al. (2004) showed that *T. angustifolia* can tolerate unusually high addition of Pb into the soil ($267\ mg\ kg^{-1}$). The tolerance of another *Typha* species, *T. latifolia* to high Pb concentrations has been also demonstrated. For example, McNaughton et al. (1974) exposed *T. latifolia* to Pb concentrations 16 times higher than normal levels with no evidence of chlorosis or any other visible symptoms. Berry (1986) suggested three basic strategies of response to metal toxicity: avoidance, detoxification, and biochemical tolerance, each of them affecting plant metal concentrations in different ways. This study showed in particular that all *Typha* species maintain low Pb concentrations in leaves, thus avoiding toxicity symptoms. These results may suggest that the tolerance of *Typha* leaves to Pb depend mainly on metal exclusion ability. The avoidance strategy of high metal concentrations from *Typha* leaves found confirmation in several studies (e.g., Fernandes and Henriques, 1990; Ye et al., 1997).

The concentrations of Cu, Mn and Zn in all *Typha* species were correlated with the levels in sediments. These elements are important micronutrients for plant growth and metabolism, and this may explain the significant correlation between concentrations in plant species and sediments (Memon et al., 2001; Carranza-Álvarez et al., 2008). However, Cu, Mn and Zn may reach high concentrations in plants, thus proving more toxic than non-essential elements as a consequence of active uptake mechanisms in the former and tolerance strategies in the latter (Ralph and Burchett, 1998; Smillie, 2015). The levels of Cu, Mn and Zn in the *Typha* species of this study were in line with previous findings in metal-contaminated wetlands (Hozhina et al., 2001; Pandey et al., 2014). According to Kabata-Pendias (2011) and Allen (1989), phytotoxic ranges of Mn and Cu are respectively $50–500\ mg\ kg^{-1}$ and $2.1–8.4\ mg\ kg^{-1}$. In this study, the mean concentrations of Mn and Cu in *Typha* species were within and passed respectively these phytotoxic levels. These results further corroborated previous findings that showed the great tolerance of *Typha* plants to metal contaminated environments (Carranza-Álvarez et al., 2008; Sasmaz et al., 2008).

When metal concentrations are as high as those found in the sediments of the study area, it becomes invaluable to use plant species to extract such metals but it is equally important to identify those plant species capable of stabilizing and preventing the spread of contaminants in the environment (Salt et al., 1995; Hashimoto et al., 2008). The efficiency of phytoremediation is often limited by the small biomass of hyperaccumulator plants (Xu et al., 2011). However, several plants with large biomass have been successfully used for phytoremediation (Pilon-Smits, 2005; Vymazal, 2011). The efficiency of phytoremediation relies on the ability of such large-biomass plants to accumulate high metal concentrations in roots and shoots over long periods without showing any signs of toxicity. To allow this, large-biomass plants must have efficient mechanisms of metal tolerance (Cosio et al., 2006). Selecting suitable plants for phytoremediation should also follow ecological criteria such as climatic adaptability and invasiveness risk. In particular, *T. domingensis*, *T. latifolia* and *T. angustifolia* should be considered as some of the best species for phytoremediation because of their large biomass, high metal tolerance and ecological amplitude. These *Typha* species are also invasive plants and strong competitors. In a constructed wetland for effluent treatment, invasiveness is a desirable plant characteristic; however, in a natural wetland, treating a pollution discharge through *Typha* spp. could be a problem due to its invasiveness difficult to control. Overall, in an ever more polluted world, natural, efficient and cost-effective systems of remediation are suitable responses to mitigate the progressive degradation of soil and water, and cattail species may play a prominent role in metal remediation of contaminated wetlands.

## 4. Conclusions

In ecological engineering, using one species instead of another can resolve the practical problem of replacing one species with another having similar remediation capacities but more abundant. Cattail species *Typha domingensis*, *T. latifolia* and *T. angustifolia* should be considered as interchangeable plants for phytoremediation of metal contaminated wetlands. This study found indeed similar element concentrations in all *Typha* species under the same field polluting conditions. Moreover, given their similar biomass size, these three *Typha* species may also have comparable capacities of accumulation. These three *Typha* species could be also simultaneously used in a constructed wetland due to the fact they have no differences in contaminant removal and tolerance. Besides, collecting the three mixed *Typha* species simultaneously in the field could be easier and quicker. Our results corroborated previous findings on these cattail species, namely high tolerance to metal contamination, bulk concentrations in roots, and restricted element-translocation into leaves. This study provided a clear example of how different *Typha* species can have similar capacities to assimilate heavy metals in their organs.

## Acknowledgements

This study was financed by Horizon 2020 ERANET cofund WATER WORKS 2014 - project Smart decentralised water management through a dynamic integration of technologies – WATINTECH. This study received financial support also by EcoStat srl – Spin-off of Catania University, and by the Italian Ministry of Education, University and Research. The authors are thankful to the staff involved in sampling, chemical analysis and data processing.

## References

Aksoy, A., Duman, F., Sezen, G., 2005. Heavy metal accumulation and distribution in narrow-leaved cattail (*Typha angustifolia*) and common reed (*Phragmites australis*). J. Freshw. Ecol. 20 (4), 783–785.
Ali, H., Khan, E., Sajad, M.A., 2013. Phytoremediation of heavy metals—concepts and applications. Chemosphere 91, 869–881.
Allen, S.E., 1989. Chemical Analysis of Ecological Material, 2nd edition. Blackwell Scientific Publications, Oxford, pp. 368.
Baker, A., McGrath, S.P., Sidoli, C.M., Reeves, R.D., 1994. The possibility of in situ heavy metal decontamination of polluted soils using crops of metal-accumulating plants. Resour. Conserv. Recycl. 11, 41–49.
Bargagli, R., 1998. Trace Elements in Terrestrial Plants. An Ecophysiological Approach to Biomonitoring and Biorecovery. Springer, Berlin, pp. 324.
Berry, W.L., 1986. Plant factors in influencing the use of plant analysis as a tool for biogeochemical prospecting. In: Carlisle, D., Berry, W.L., Kaplan, I.R., Watterson, J.R. (Eds.), Mineral Exploration: Biological Systems and Organic Matter, Rubey 5. Prentice-Hall, Englewood Cliffs, NJ, U.S.A, pp. 13–32.
Bhatia, M., Goyal, D., 2014. Analyzing remediation potential of wastewater through wetland plants: a review. Environ. Prog. Sustain. Energy 33, 9–27.
Blaylock, M.J., Salt, D.E., Dushenkov, S., Zakharova, O., Gussman, C., Kapulnik, Y., Ensley, B.D., Raskin, I., 1997. Enhanced accumulation of Pb in Indian mustard by soil-applied chelating agents. Environ. Sci. Technol. 31, 860–865.
Boening, D.W., 2000. Ecological effects, transport, and fate of mercury: a general review. Chemosphere 40, 1335–1351.
Bonanno, G., 2011. Trace element accumulation and distribution in the organs of *Phragmites australis* (common reed) and biomonitoring applications. Ecotoxicol. Environ. Saf. 74, 1057–1064.
Bonanno, G., 2012. *Arundo donax* as a potential biomonitor of trace element contamination in water and sediment. Ecotoxicol. Environ. Saf. 80, 20–27.
Bonanno, G., 2013. Comparative performance of trace element bioaccumulation and biomonitoring in the plant species *Typha domingensis*, *Phragmites australis* and *Arundo donax*. Ecotox. Environ. Saf. 97, 124–130.
Bonanno, G., Lo Giudice, R., 2010. Heavy metal bioaccumulation by the organs of *Phragmites australis* (common reed) and their potential use as contamination indicators. Ecol. Indic. 10, 639–645.
Bonanno, G., Cirelli, G.L., Toscano, A., Lo Giudice, R., Pavone, P., 2013. Heavy metal content in ash of energy crops growing in sewage-contaminated natural wetlands: potential applications in agriculture and forestry? Sci. Total Environ. 452–453, 349–354.
Bonanno, G., Borg, J.A., Di Martino, V., 2017. Levels of heavy metals in wetland and marine vascular plants and their biomonitoring potential: a comparative assessment. Sci. Total Environ. 576, 796–806.
Cardwell, A.J., Hawker, D.W., Greenway, M., 2002. Metal accumulation in aquatic macrophytes from southeast Queensland, Australia. Chemosphere 48, 653–663.
Carranza-Álvarez, C., Alonso-Castro, A.J., Alfaro-De La Torre, M.C., García-De La Cruz, R.F., 2008. Accumulation and distribution of heavy metals in *Scirpus americanus* and *Typha latifolia* from an artificial Lagoon in San Luis Potosí, México. Water Air Soil Pollut. 188, 297–309.
Carrier, P., Baryla, A., Havaux, M., 2003. Cadmium distribution and microlocalization in oilseed rape (*Brassica napus*) after long-term growth on cadmium- contaminated soil. Planta 216, 939–950.
Chandra, R., Yadav, S., 2010. Potential of *Typha angustifolia* for phytoremediation of heavy metals from aqueous solution of phenol and melanoidin. Ecol. Eng. 36, 1277–1284.
Chatterjee, J., Chatterjee, C., 2000. Phytotoxicity of cobalt, chromium and copper in cauliflower. Environ. Pollut. 109, 69–74.
Cosio, C., Vollenweider, P., Keller, C., 2006. Localization and effects of cadmium in leaves of a cadmium-tolerant willow (*Salix viminalis* L.): Part I Macrolocalization and phytotoxic effects of cadmium. Environ. Exp. Bot. 58, 64–74.
Dalenberg, J.W., van Driel, W., 1990. Contribution of atmospheric deposition to heavy metal concentration in field crops, Netherlands. J. Agric. Sci. 38, 369–379.
Debusk, T.A., Laughlin Jr, R.B., Schwartz, L.N., 1996. Retention and compartmentalization of lead and cadmium in wetland microcosms. Water Res. 30 (11), 2707–2716.
Demirezen, D., Aksoy, A., 2004. Accumulation of heavy metals in *Typha angustifolia* (L.) and *Potamogeton pectinatus* (L.) living in Sultan Marsh (Kayseri, Turkey). Chemosphere 56, 685–696.
Deng, H., Ye, Z.H., Wong, M.H., 2004. Accumulation of lead, zinc, copper and cadmium by 12 wetland plant species thriving in metal-contaminated sites in China. Environ. Pollut. 132 (1), 29–40.
Dhote, S., Dixit, S., 2009. Water quality improvement through macrophytes—a review. Environ. Monit. Assess. 152, 149–153.
Duarte, B., Caetano, M., Almeida, P.R., Vale, C., Caçador, I., 2010. Accumulation and biological cycling of heavy metal in four salt marsh species, from Tagus estuary (Portugal). Environ. Pollut. 158, 1661–1668.
Eid, E.M., Shaltout, K.H., El-Sheikh, M.A., Asaeda, T., 2012. Seasonal courses of nutrients and heavy metals in water, sediment and above- and below-ground *Typha domingensis* biomass in Lake Burullus (Egypt): perspectives for phytoremediation. Flora 207, 783–794.
Engin, M.S., Uyanik, A., Cay, S., 2017. Investigation of trace metals distribution in water, sediments and wetland plants of Kızılırmak Delta, Turkey. Int. J. Sed. Res. 32, 90–97.
Feng, H., Qian, Y., Gallagher, F.J., Zhang, W., Yu, L., Liu, C., Jones, K.W., Tappero, R., 2016. Synchrotron micro-scale measurement of metal distributions in *Phragmites australis* and *Typha latifolia* root tissue from an urban brownfield site. J. Environ. Sci. 41, 172–182.
Fernandes, J.C., Henriques, F.S., 1990. Metal levels in soils and attail (*Typha latifolia* L.) plants in a pyrites mine area at Lousal, Portugal. Intern. J. Environ. Stud. 36, 205–210.
Fischerová, Z., Tlustos, P., Száková, J., Sichorová, K., 2006. A comparison of phytoremediation capability of selected plant species for given trace elements. Environ. Pollut. 144, 93–100.
Fitzgerald, E.J., Caffrey, J.M., Nesaratnam, S.T., McLoughlin, P., 2003. Copper and lead concentrations in salt marsh plants on the Suir estuary, Ireland. Environ. Pollut. 123, 67–74.
Gall, J.E., Boyd, R.S., Rajakaruna, N., 2015. Transfer of heavy metals through terrestrial food webs: a review. Environ. Monit. Assess. 187, 1–21.
Gambrell, R.P., 1994. Trace and toxic metals in wetlands — a review. J. Environ. Qual. 23, 883–891.
Gomes, M.V.T., de Souza, R.R., Teles, V.S., Mendes, E.A., 2014. Phytoremediation of water contaminated with mercury using *Typha domingensis* in constructed wetland. Chemosphere 103, 228–233.
Greger, M., 1999. Metal availability and bioconcentration in plants. In: Prasad, M.N.V., Hagemeyer, J. (Eds.), Heavy Metals Stress in Plants: From Molecule to Ecosystems. Springer-Verlag, Heidelberg, Berlin, pp. 1–27.
Greipsson, S., 1994. Effects of iron plaque on roots of rice on growth and metal concentration of seeds and plant tissues when cultivated in excess copper. Commun. Soil Sci. Plant Anal. 25, 2761–2769.
Gussarsson, M., 1994. Cadmium-induced alterations in nutrient composition and growth of *Betula pendula* seedlings: the significance of fine roots as a primary target for cadmium toxicity. J. Plant Nutr. 17, 2151–2163.
Hall, J.L., 2002. Cellular mechanisms for heavy metal detoxification and tolerance. J. Exp. Bot. 53 (366), 1–11.
Hashimoto, Y., Matsufuru, H., Sato, T., 2008. Attenuation of lead leachability in shooting range soils using poultry waste amendments in combination with indigenous plant

species. Chemosphere 73, 643–649.
Howitt, J.A., Mondon, J., Mitchell, B.D., Kidd, T., Eshelman, B., 2014. Urban stormwater inputs to an adapted coastal wetland: role in water treatment and impacts on wetland biota. Sci. Total Environ. 485–486, 534–544.
Hozhina, E.I., Khramov, A.A., Gerasimov, P.A., Kumarkov, A.A., 2001. Uptake of heavy metals, arsenic, and antimony by aquatic plants in the vicinity of ore mining and processing industries. J. Geochem. Explor. 74, 153–162.
Italian Decree 260/2010. Decreto Ministero dell'ambiente e della tutela del territorio e del mare 8 Novembre 2010, N. 260. Regolamento Recante I Criteri Tecnici per La Classificazione Dello Stato Dei Corpi Idrici Superficiali, per La Modifica Delle Norme Tecniche Del Decreto Legislativo 3 Aprile 2006, N. 152, Recante Norme in Materia Ambientale, Predisposto Ai Sensi dell'articolo 75, Comma 3, Del Medesimo Decreto Legislativo. Supplemento N. 31 Alla Gazzetta Ufficiale 7 Febbraio 2011 N. 30.
Jiménez, M.N., Bacchetta, G., Casti, G.M., Navarro, F.B., Lallena, A.M., Fernández-Ondoño, E., 2011. Potential use in phytoremediation of three plant species growing on contaminated mine-tailing soils in Sardinia. Ecol. Eng. 37, 392–398.
Kabata-Pendias, A., 2011. Trace Elements in Soils and Plants, Fourth edition. Taylor & Francis Group, Boca Raton, FL, USA.
Kabata-Pendias, A., Mukherjee, A.B., 2007. Trace Elements from Soil to Human. Springer, Berlin Heidelberg.
Kapaj, S., Peterson, H., Liber, K., Bhattacharya, P., 2006. Human health effects from chronic arsenic poisoning – a review. J. Environ. Sci. Health [A] 41, 2399–2428.
Laffont-Schwob, I., Triboit, F., Prudent, P., Soulié-Märsche, I., Rabier, Despréaux, J., Thiéry, A., 2015. Trace metal extraction and biomass production by spontaneous vegetation in temporary Mediterranean stormwater highway retention ponds: freshwater macroalgae (*Chara* spp.) vs. cattails (*Typha* spp.). Ecol. Eng. 81, 173–181.
Lara, M.A.M., Blazquez, G., Trujillo, M.C., Perez, A., Calero, M., 2014. New treatment of real electroplating wastewater containing heavy metal ions by adsorption onto olive stone. J. Clean. Prod. 81, 120–129.
Leto, C., Tuttolomondo, T., La Bella, S., Leone, R., Licata, M., 2013. Effects of plant species in a horizontal subsurface flow constructed wetland – phytoremediation of treated urban wastewater with *Cyperus alternifolius* L. and *Typha latifolia* L. in the West of Sicily (Italy). Ecol. Eng. 61, 282–291.
Li, M., Zhang, L.J., Tao, L., Li, W., 2008. Ecophysiological responses of *Jussiaea rapens* to cadmium exposure. Aquat. Bot. 88, 347–352.
Liu, J., Zhang, W., Qu, P., Wang, M., 2016. Cadmium tolerance and accumulation in fifteen wetland plant species from cadmium-polluted water in constructed wetlands. Front. Environ. Sci. Eng. 10 (2), 262–269.
Lominchar, M.A., Sierra, M.J., Millán, R., 2015. Accumulation of mercury in Typha domingensis under field conditions. Chemosphere 119, 994–999.
Lyubenova, L., Pongrac, P., Vogel-Mikuš, K., Kukek Mezek, G., Vavpetič, P., Grlj, N., Regvar, M., Pelicon, P., Schröder, P., 2013. The fate of arsenic, cadmium and lead in *Typha latifolia*: a case study on the applicability of micro-PIXE in plant ionomics. J. Hazard. Mater. 248–249, 371–378.
Ma, L.Q., Komar, K.M., Tu, C., Zhang, W., Cai, Y., 2001. A fern that hyper accumulates arsenic. Nature 409, 579.
Maine, M.A., Suñe, N., Hadad, H., Sánchez, G., Bonetto, C., 2009. Influence of vegetation on the removal of heavy metals and nutrients in a constructed wetland. J. Environ. Manag. 90, 355–363.
Maine, M.A., Hadad, H.R., Sánchez, G.C., Di Luca, G.A., Mufarrege, M.M., Caffaratti, S.E., Pedro, M.C., 2017. Long-term performance of two free-water surface wetlands for metallurgical effluent treatment. Ecol. Eng. 98, 372–377.
Markert, B., 1992. Presence and significance of naturally occurring chemical elements of the periodic system in the plant organism and consequences for future investigations on inorganic environmental chemistry in ecosystems. Vegetatio 103, 1–30.
Massa, N., Andreucci, F., Poli, M., Aceto, M., Barbato, R., Berta, G., 2010. Screening for heavy metal accumulators amongst autochthonous plants in a polluted site in Italy. Ecotoxicol. Environ. Saf. 73, 1988–1997.
McNaughton, S.J., Folsom, T.C., Lee, T., Park, F., Price, C., Roeder, D., Schmitz, J., Stokwell, C., 1974. Heavy metal tolerance in *Typha latifolia* without the evolution of tolerant races. Ecology 55, 1163–1165.
Memon, A.R., Aktoprakligil, D., Ozdemir, A., Vertii, A., 2001. Heavy metal accumulation and detoxification mechanisms in plants. Turk. J. Bot. 25, 111–121.
Mendoza-Carranza, M., Sepúlveda-Lozada, A., Dias-Ferreira, C., Geissen, V., 2016. Distribution and bioconcentration of heavy metals in a tropical aquatic food web: a case study of a tropical estuarine lagoon in SE Mexico. Environ. Pollut. 210, 155–165.
Millán, R., Lominchar, M.A., Rodríguez-Alonso, J., Schmid, T., Sierra, M.J., 2014. Riparian vegetation role in mercury uptake (Valdeazogues River, Almadén, Spain). J. Geochem. Explor. 104, 104–110.
Moore, T.R., Bubier, J.L., Heyes, A., Flett, R.J., 1995. Methyl and total mercury in boreal wetland plants, experimental lakes area, Northwestern Ontario. J. Environ. Qual. 24, 845–850.
Morel, F.M.M., Kraepiel, A.M.L., Amyot, M., 1998. The chemical cycle and bioaccumulation of mercury. Annu. Rev. Ecol. Syst. 29, 543–566.
Mufarrege, M.M., Hadad, H.R., Maine, M.A., 2010. Response of *Pistia stratiotes* to heavy metals (Cr, Ni, and Zn) and phosphorous. Arch. Environ. Contam. Toxicol. 58 (1), 53–61.
Mufarrege, M.M., Hadad, H.R., Di Luca, G.A., Maine, M.A., 2014. Metal dynamics and tolerance of *Typha domingensis* exposed to high concentrations of Cr, Ni and Zn. Ecotoxicol. Environ. Saf. 105, 90–96.
Muhammad, D., Chen, F., Zhao, J., Zhang, G.P., Wu, F.B., 2009. Comparison of EDTA and citric acid-enhanced phytoextraction of heavy metals in artificially metal contaminated soil by *Typha angustifolia*. Int. J. Phytoremediat. 11, 558–574.
Odum, H.T., 2000. Heavy Metals in the Environment. Using Wetlands for Their Removal. CRC Press LLC, Boca Raton, Florida, USA.
Olguín, E.J., Sánchez-Galván, G., 2012. Heavy metal removal in phytofiltration and phycoremediation: the need to differentiate between bioadsorption and bioaccumulation. New Biotechnol. 30, 3–8.
Otte, M.L., Matthews, D.J., Jacob, D.L., Moran, B.M., Baker, A.J.M., 2004. Biogeochemistry of metals in the rhizosphere of wetland plants — an explanation for "innate" metal tolerance? In: Wong, M.H. (Ed.), Wetlands ecosystems in Asia: function and management. Elsevier Science, Amsterdam, Netherlands.
Overesch, M., Rinklebe, J., Broll, G., Neue, H.U., 2007. Metals and arsenic in soils and corresponding vegetation at Central Elbe river floodplains (Germany). Environ. Pollut. 145, 800–812.
Pais, I., Jones, J.B., 2000. The Handbook of Trace Elements. St. Luice Press, Florida.
Pandey, V.C., 2012. Phytoremediation of heavy metals from fly ash pond by *Azolla caroliniana*. Ecotoxicol. Environ. Saf. 82, 8–12.
Pandey, V.C., Singh, N., Singh, R.P., Singh, D.P., 2014. Rhizoremediation potential of spontaneously grown *Typha latifolia* on fly ash basins: study from the field. Ecol. Eng. 71, 722–727.
Pandey, V.C., Pandey, D.N., Singh, N., 2015. Sustainable phytoremediation based on naturally colonizing and economically valuable plants. J. Clean. Prod. 86, 37–39.
Pandey, V.C., Bajpai, O., Singh, N., 2016. Energy crops in sustainable phytoremediation. Renew. Sustain. Energy Rev. 54, 58–73.
Panich-Pat, T., Pokethitiyook, P., Kruatrachue, P.M., Upatham, E.S., Srinives, P., Lanza, G.R., 2004. Removal of lead from contaminated soils by *Typha angustifolia*. Water Air Soil Pollut. 155, 159–171.
Pignatti, S., 1982. Flora d'Italia. Edagricole, Bologna.
Pilon-Smits, E., 2005. Phytoremediation. Annu. Rev. Plant Biol. 56, 15–39.
Prasad, M.N.V., 1995. Cadmium toxicity and tolerance in vascular plants. Environ. Exp. Bot. 35, 525–545.
Prasad, M.N.V., 2004. Heavy Metal Stress in Plants: From Biomolecules to Ecosystems, second ed. Springer-Verlag, Berlin Heidelberg, New York.
Prasad, M.N.V., Sajwan, K.S., Naidu, R., 2006. Trace Elements in the Environment: Biogeochemistry, Biotechnology and Bioremediation. CRC Press LLC, Boca Raton, Florida, USA.
Qian, Y., Gallagher, F.J., Feng, H., Wu, M., 2012. A geochemical study of toxic metal translocation in an urban brownfield wetland. Environ. Pollut. 166, 23–30.
Rai, U.N., Upadhyay, A.K., Singh, N.K., Dwivedi, S., Tripathi, R.D., 2015. Seasonal applicability of horizontal sub-surface flow constructed wetland for trace elements and nutrient removal from urban wastes to conserve Ganga River water quality at Haridwar, India. Ecol. Eng. 81, 115–122.
Ralph, P.J., Burchett, M.D., 1998. Photosynthetic responses of *Halophila ovalis* to heavy metals stress. Environ. Pollut. 103, 91–101.
Rezania, S., Taib, S.M., Din, M.F.M., Dahalan, F.A., Kamyab, H., 2016. Comprehensive review on phytotechnology: heavy metals removal by diverse aquatic plants species from wastewater. J. Hazard. Mater. 318, 587–599.
Salem, Z.B., Laffray, X., Ashoour, A., Ayadi, H., Aleya, L., 2014. Metal accumulation and distribution in the organs of Reeds and Cattails in a constructed treatment wetland (Etueffont, France). Ecol. Eng. 64, 1–17.
Salt, D.E., Blaylock, M., Nanda Kumar, P.B.A., Dushenkov, V., Ensley, B.D., Chet, I., Raskin, I., 1995. Phytoremediation: a novel strategy for the removal of toxic metals from the environment using plants. Biotechnology 13, 468–474.
Salt, D.E., Smith, R.D., Raskin, I., 1998. Phytoremediation. Annu. Rev. Plant Physiol. Plant Mol. Biol. 49, 643–668.
Samecka-Cymerman, A., Kempers, A.J., 2001. Concentrations of heavy metals and plant nutrients in water, sediments and aquatic macrophytes of anthropogenic lakes (former open cut brown coal mines) differing in stage of acidification. Sci. Total Environ. 281, 87–98.
Sasmaz, A., Obek, E., Hasar, H., 2008. The accumulation of heavy metals in *Typha latifolia* L. grown in a stream carrying secondary effluent. Ecol. Eng. 33, 278–284.
Sawidis, T., Chettri, M.K., Zachariadis, G.A., Stratis, J.A., 1995. Heavy metals in aquatic plants and sediment from systems in Macedonia Greece. Ecotoxicol. Environ. Saf. 32, 73–83.
Sharma, S., Singh, B., Manchanda, V.K., 2015. Phytoremediation: role of terrestrial plants and aquatic macrophytes in the remediation of radionuclides and heavy metal contaminated soil and water. Environ. Sci. Pollut. Res. 22, 946–962.
Siedlecka, A., Tukendorf, A., Skórzyńska-Polit, E., Maksymiec, W., Wójcik, M., Baszyński, T., Krupa, Z., 2001. Angiosperms (Asteraceae, Convolvulaceae, Fabaceae and Poaceae; other than Brassicaceae). In: Prasad, M.N.V (Ed.), Metals in the Environment. Analysis by Biodiversity. Marcel Dekker, Inc, New York, pp. 171–217.
Smillie, C., 2015. *Salicornia* spp. as a biomonitor of Cu and Zn in salt marsh sediments. Eco. Indic. 56, 70–78.
Smith, S.G., 1987. Typha: its taxonomy and the ecological significance of hybrids. Arch. Hydrobiol. Beih. 27, 129–138.
Tack, F.M.G., Vandecasteele, B., 2008. Cycling and ecosystem impact of metals in contaminated calcareous dredged sediment-derived soils (Flanders, Belgium). Sci. Total Environ. 400, 283–289.
Teles Gomes, M.V., Rodrigues de Souza, R., Silva Teles, V., Araújo Mendes, E.,, 2014. Phytoremediation of water contaminated with mercury using *Typha domingensis* in constructed wetland. Chemosphere 103, 228–233.
Teuchies, J., Jacobs, S., Oosterlee, L., Bervoets, L., Meire, P., 2013. Role of plants in metal cycling in a tidal wetland: implications for phytoremediation. Sci. Total Environ. 445–446, 146–154.
Vymazal, J., 2011. Constructed wetlands for wastewater treatment: five decades of experience. Environ. Sci. Technol. 45, 61–69.
Wang, T., Peverly, J.H., 1996. Oxidation states and fractionation of plaque iron on root of common reeds. Soil Sci. Soc. Am. J. 60, 323–329.
Weis, J.S., Weis, P., 2004. Metal uptake, transport and release by wetland plants: implications for phytoremediation and restoration. Environ. Int. 30, 685–700.
Willis, J.M., Gambrell, R.P., Hester, M.W., 2010. Growth response and tissue

accumulation trends of herbaceous wetland plants species exposed to elevated aqueous mercury levels. Int. J. Phytoremediat. 12, 586–598.
Wójcik, M., Sugier, P., Siebielec, G., 2014. Metal accumulation strategies in plants spontaneously inhabiting Zn-Pb waste deposits. Sci. Total Environ. 487, 313–322.
Xu, W., Shi, W., Yan, F., Zhang, B., Liang, J., 2011. Mechanisms of cadmium detoxification in cattail (*Typha angustifolia* L.). Aquat. Bot. 94, 37–43.
Yadav, A.K., Abbassi, R., Kumar, N., Satya, S., Sreekrishnan, T.R., Mishra, B.K., 2012. The removal of heavy metals in wetland microcosms: effects of bed depth, plant species, and metal mobility. Chem. Eng. J. 211–212, 501–507.
Ye, Z.H., Baker, A.J.M., Wong, M.H., Willis, A.J., 1997. Zinc, lead and cadmium tolerance, uptake and accumulation by *Typha latofolia*. New Phytol. 136, 469–480.
Ye, Z.H., Baker, A.J.M., Wong, M.H., Willis, A.J., 1998. Zinc, lead and cadmium accumulation and tolerance in *Typha latifolia* as affected by iron plaque on the root surface. Aquat. Bot. 61, 55–67.
Yoon, J., Cao, X., Zhou, Q., Ma, L.Q., 2006. Accumulation of Pb Cu and Zn in native plants growing on a contaminated Florida site. Sci. Total Environ. 368, 456–464.
Zhang, W., Cai, Y., Tu, C., Ma, L.Q., 2002. Arsenic speciation and distribution in a arsenic hyperaccumulating plant. Sci. Total Environ. 300, 167–177.
Zu, Y.Q., Li, Y., Chen, J.J., Chen, H.Y., Qin, L., Schvartz, C., 2005. Hyperaccumulation of Pb, Zn and Cd in herbaceous grown on lead-zinc mining area in Yunnan. China. Environ. Int. 31, 755–762.