



RESEARCH PAPER

# Phytoremediation Potential of Two Hydrophytes Introduced to Egypt: *Hydrocotyle umbellata* L. and *Salvinia auriculata* Aubl.

Asmaa M. El-Makawy<sup>1</sup> · Selim Z. Heneidy<sup>1</sup> · Amal M. Fakhry<sup>1</sup> · Marwa Waseem A. Halmy<sup>2</sup>

Received: 12 August 2025 / Revised: 16 January 2026 / Accepted: 10 February 2026 / Published online: 6 April 2026  
© The Author(s) 2026

## Abstract

Phytoremediation represents an eco-friendly and cost-effective approach for removing heavy metals from aquatic systems. This study evaluated the phytoremediation potential of *Hydrocotyle umbellata* L. and *Salvinia auriculata* Aubl. for copper (Cu), lead (Pb), zinc (Zn), and chromium (Cr) under two hydrological conditions, deep water (DW) and constructed wetlands (CW), in Egypt over a 12-month experimental period. Phytoremediation performance was assessed using rhizofiltration efficiency, translocation factor (TF), and bioaccumulation factor (BAF). Both species demonstrated substantial capacity to remediate the investigated metals, with an overall uptake pattern of Zn>Cu>Cr>Pb. Clear interspecific differences were observed in remediation strategies. *Hydrocotyle umbellata* exhibited strong rhizofiltration capacity, particularly for Zn and Cu, but limited metal translocation (TF<1 for most metals), indicating a phytostabilization-dominated strategy. Metals were largely retained in the root system, a recognized tolerance mechanism that reduces toxicity in aerial tissues. In contrast, *S. auriculata* showed higher rhizofiltration, translocation (TF≥1 for several metals), and bioaccumulation, reflecting a stronger phytoextraction potential, albeit with evidence of metal-induced growth inhibition. Seasonal variation significantly influenced metal removal, with spring showing the highest rhizofiltration and accumulation for most metals, particularly in *S. auriculata*. Growing *H. umbellata* in CW enhanced its rhizofiltration efficiency, whereas hydrological conditions had a lesser effect on *S. auriculata*. Overall, the findings highlight the complementary remediation roles of both species and emphasize the importance of species selection, hydrological setting, and seasonality in optimizing phytoremediation strategies for metal-contaminated waters.

---

✉ Marwa Waseem A. Halmy  
marwa.w.halmy@alexu.edu.eg

Asmaa M. El-Makawy  
asmaa.elmakawy\_PG@alexu.edu.eg

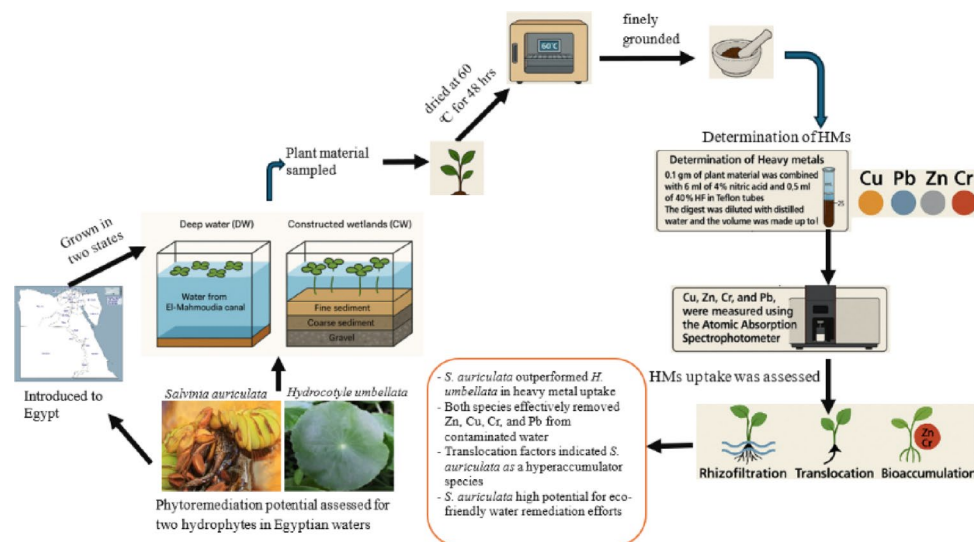
Selim Z. Heneidy  
selim.heneidy@alexu.edu.eg

Amal M. Fakhry  
amal.fakhry@alexu.edu.eg

<sup>1</sup> Department of Botany and Microbiology, Faculty of Science, Alexandria University, Alexandria 21511, Egypt

<sup>2</sup> Department of Environmental Sciences, Faculty of Science, Alexandria University, P.O. Box: 21511, Alexandria, Egypt

## Graphical Abstract



## Highlights

- Phytoremediation potential assessed for two hydrophytes in Egyptian waters.
- *Salvinia auriculata* outperformed *Hydrocotyle umbellata* in heavy metal uptake.
- Both species effectively removed Zn, Cu, Cr, and Pb from contaminated water.
- Translocation factors indicated *S. auriculata* as a hyperaccumulator species.
- *S. auriculata* demonstrates high potential for eco-friendly water remediation efforts.

**Keywords** Heavy metals · Translocation · Water treatment · Bioindicators · Bioaccumulation · Hyperaccumulators · Rhizofiltration

## Introduction

Anthropogenic activities including agricultural practices, industrial activities and urbanization result in heavy metal contamination of soil and water (Alloway 2013), which pose threats to living organisms, human health, and ecological services provisioning (Nsanganwimana et al. 2014; Oladoye et al. 2022).

Heavy metals (HMs) contamination poses a serious threat to freshwater ecosystems because of the persistence, toxicity, and bioaccumulative nature of many metals. Elements such as copper (Cu), zinc (Zn), chromium (Cr), and lead (Pb) occur naturally at low concentrations; however, their levels in aquatic environments have increased markedly due to anthropogenic activities, including industrial discharges, urban runoff, and agricultural practices (Alloway 2013; Tchounwou et al. 2012). Copper and zinc are essential micronutrients for plant metabolism, but at elevated concentrations they can become toxic and disrupt key biological processes. Among commonly reported heavy metal pollutants (e.g., Cd, Cr, Cu, Hg, Pb, and Zn), metals such as Co, Cu, and

Zn are required only in trace amounts for plant growth, and their excessive accumulation poses risks to plants and other living organisms (Varol and Sünbül 2018; Khan et al. 2019).

Anthropogenic inputs represent the dominant sources of heavy metals in aquatic systems, primarily through industrial activities, wastewater disposal, chemical runoff, and intensive agriculture (Oladoye et al. 2022; Tóth et al. 2016). Copper contamination is largely associated with pesticide production and application, battery manufacturing, and electroplating industries (Guida et al. 2008; Song et al. 2011), and it commonly enters surface waters via industrial effluents, antifouling agents, agricultural fungicides (Alloway 2013; Kabata-Pendias 2011). Zinc originates mainly from galvanization processes, tire wear, sewage discharge, and fertilizer use (Alloway 2013; Kabata-Pendias 2011). Chromium pollution is closely linked to tanning, electroplating, and textile industries, with hexavalent chromium being particularly mobile and toxic in aquatic environments (Saha et al. 2011). Lead, a non-essential and highly toxic metal, is primarily derived from traffic emissions, battery

recycling, industrial effluents, and aging infrastructure, and can accumulate in aquatic organisms even at low concentrations (Jaishankar et al. 2014). Understanding the sources and environmental behavior of Cu, Zn, Cr, and Pb is therefore critical for interpreting metal uptake patterns and assessing ecological risks in contaminated freshwater systems (Kabata-Pendias 2011; Tchounwou et al. 2012).

In recent years, phytoremediation—defined as a set of approaches that use plants and their associated rhizosphere microorganisms to remove, transform, or stabilize pollutants in contaminated soils, sediments, and water bodies—has received considerable attention (Abhilash et al. 2009a, b; Ali et al. 2020). The term phytoremediation was first described in 1991 after several research efforts in oil spills, wastewater polishing wetlands, ecological treatment systems, and agricultural plant accumulation of heavy metals (Rock et al. 2000). There are different techniques and approaches of phytoremediation which include phytoextraction, phytofiltration, phytostabilization, phytodegradation, and phytovolatilization (Alkorta et al. 2004). Plants possess highly efficient and selective mechanisms for the uptake of trace elements from their surroundings, even when these elements occur at very low concentrations. Toxic metals with chemical properties similar to essential nutrients may be absorbed, translocated, and stored through the same physiological pathways (Tangahu et al. 2011). Aquatic macrophytes, in particular, are widely used as bioindicators and phytoremediators due to their ability to absorb and accumulate heavy metals from both water and sediments (Ali et al. 2020; Mustafa and Hayder 2021). There are several criteria used for selection of aquatic plant species as a phytoremediator, including the remediation potential of the plant, the type of elements to be removed, and the local geographical and environmental conditions prevailing (Khilji and Bareen 2008). The rapid growth and high biomass production of most aquatic plants provide potential for utilization in phytoremediation in aquatic ecosystems. Previous studies revealed that metal accumulation is typically higher in roots than in shoots, reflecting a common tolerance and retention strategy in aquatic macrophytes (Fritioff and Greger 2006; Khilji and Bareen 2008). Additionally, aquatic plants have certain tolerance mechanisms that qualify them to avoid metal toxicity including metal sequestration, compartmentalization in certain cell organelles, exclusion, and inactivation by exudation of organic ligands (Sarwar et al. 2017). These special characteristics of macrophytes increase their capability for being used in the eco-friendly and cost-effective technology in pollution abatement (Abida and HariKrishna 2010).

Several species have demonstrated effective heavy metal removal, including *Pontederia crassipes* Mart. (Zhu et al. 1999), *Phragmites australis* (Cav.) Trin. ex Steud., *Potamogeton pectinatus* L., and *Typha domingensis* (Pers.) Poir. ex Steud., which are recognized for playing considerable role in phytoremediation, as efficient and economic clean-up technologies for contaminated water, groundwater, and wastewater. *Hydrocotyle umbellata* L. (Dierberg et al. 1987; Khilji and Bareen 2008), and species within the genus *Salvinia* exhibit a strong capacity to accumulate substantial amounts of heavy metals (Dhir and Srivastava 2011).

Several studies have demonstrated that *H. umbellata* exhibits considerable phytofiltration potential for multiple metals (e.g., Ni, Cr, Cd, Pb, Cu, and Zn), with high removal efficiencies observed in wastewater treatment contexts and significant accumulation in plant biomass (roots and shoots) over time (Bokhari et al. 2022). Transcriptomic and physiological investigations further confirm that *H. umbellata* can accumulate cadmium (Cd) at concentrations that result in bioaccumulation factor (BCF) values greater than 1, indicating considerable uptake capacity under controlled exposure conditions (Saeed et al. 2024a, b).

Although both *Hydrocotyle umbellata* L. and *Salvinia auriculata* Aubl. have been recognized in the phytoremediation literature as effective agents for the removal and bioaccumulation of heavy metals from contaminated water, the body of work is still emerging relative to some other aquatic macrophytes. Specific studies on *S. auriculata* are more limited, broader research on *Salvinia* species consistently highlights their strong capacity to accumulate a range of heavy metals, including Zn, Pb, Cr, and Cu, often with greater metal concentrations in roots than in aerial parts (Muliyadi et al. 2023). *Salvinia* spp. are widely included in comparative bioaccumulation surveys of aquatic plants, demonstrating reliable metal removal from wastewater and supporting their use in phytoremediation systems (Dhir and Srivastava 2011; Muliyadi et al. 2023).

Previous studies indicate that *H. umbellata* and *S. auriculata* possess physiological and ecological traits favorable for heavy metal phytoremediation. However, their comparative performance, seasonal variability, and system-dependent responses under field-relevant conditions remain insufficiently explored. Accordingly, the present study provides a preliminary yet integrative assessment of the phytoremediation potential of *H. umbellata* and *S. auriculata* in a newly colonized habitat. Specifically, the objectives were to: (1) compare the seasonal efficiency of rhizofiltration, translocation, and bioaccumulation of Cu, Zn, Cr, and Pb by the two

macrophyte species; and (2) evaluate the effects of system type (deep water versus constructed wetlands) on heavy metal removal performance. By addressing these objectives, the study aims to inform the selection and management of aquatic macrophytes for sustainable phytoremediation and biomonitoring of metal-contaminated freshwater systems.

## Materials and Methods

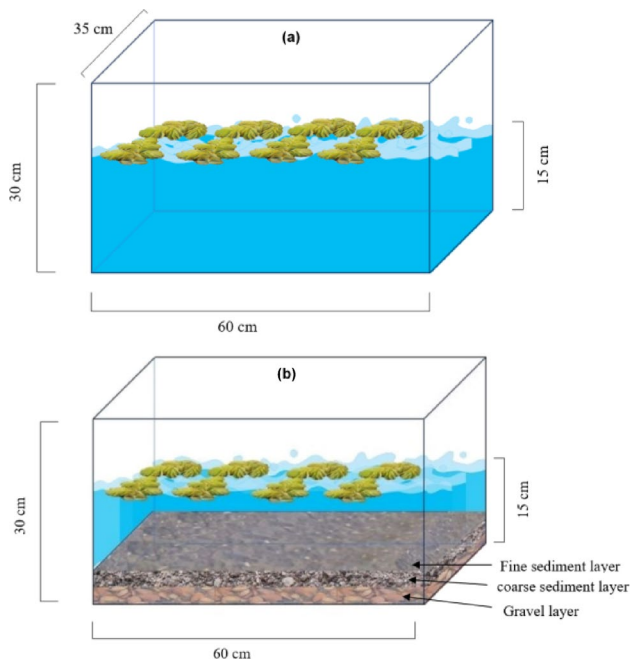
The two investigated aquatic species *H. umbellata* and *S. auriculata* have been recently introduced in private gardens in Egypt as ornamental plants (Heneidy et al. 2019). Experimental studies were conducted on the individuals of the two introduced hydrophyte species *Salvinia auriculata* and *Hydrocotyle umbellata* in open air conditions in the Alexandria University Botanic Garden (AUBG) at Faculty of Science (Figure S1 & S2). The experiment was conducted outdoors in the AUBG Botanic Garden, and the plants received natural sunlight throughout the study period. No artificial lighting was used. During the experimental period, water sourced from El-Mahmoudia Canal was partially renewed to maintain stable growth conditions and minimize nutrient depletion and metal concentration drift. Water levels were maintained at constant depth throughout the experiment. All tanks were maintained under ambient temperature and natural photoperiod conditions throughout the experiment. Plant growth and water levels were monitored regularly, and no external fertilizers or chemical additives were

applied during the experimental period to avoid interference with metal uptake processes. The tanks were supplied with continuous low-intensity water circulation using submersible pumps. This controlled water movement was applied to prevent water stagnation and ensure uniform exposure of the plant to dissolved oxygen throughout the experimental period. The phytoremediation efficiency of the two species was assessed in two states; (1) in deep water ecosystems represented by tanks filled by water only; and (2) in shallow water represented by 'constructed wetlands' (CW) tanks (Fig. 1).

### Deep Water (DW)

The plant individuals were planted in glass tanks. Each with the dimension of 60 in length  $\times$  35 cm in width  $\times$  30 cm in height. In this study, for *S. auriculata* the individual was considered as a grouping of several ramets (an average of 10 ramets) with an average width of 5 cm and average length of 6 cm. Individuals of each species utilized in the study were selected to have uniform size and vegetative appearance. All plant individuals were placed simultaneously and therefore shared a common life-history prior to the seasonal measurements. While individual plant age and size was controlled, seasonal variation in metal uptake and accumulations efficiency is expected due to changes according to external environmental conditions such as temperature, photoperiod, nutrient availability, and water chemistry, which strongly influence plant metabolism, growth dynamics, and metal uptake capacity. Using plants of the same size, age and establishment history, would minimize confounding ontogenetic effects and ensure that seasonal differences in metal uptake and accumulations performance primarily reflect environmental variability and plant physiological activity rather than legacy effects from previous growth seasons. Similar experimental designs have been widely applied to assess seasonal dynamics in metal uptake under controlled or semi-controlled conditions (e.g., Nan et al. 2002; Duman et al. 2007; Vymazal and Březinová 2015).

Two tanks were utilized for growing each species, where 32 individual plant specimens of each species were planted in each tank. The tanks were completed with water collected from El-Mahmoudia canal and the growth indicators of the individuals of the two species were observed for over 12 months. Continuous low-intensity water circulation was maintained in the tanks using submersible pumps to prevent stagnation and ensure uniform aeration to plants during the experiment. The glass tanks were placed in the AUBG in open air conditions. Observations were made to monitor changes in the morphology and growth parameters of individual plants from each species cultivated in tanks on a regular basis.



**Fig. 1** Experimental setup for **a** deep water (DW) and **b** constructed wetlands (CW) treatment

## Constructed Wetlands (CW)

To simulate the natural habitat, the constructed wetlands were established through obtaining water from El-Mahmoudia canal and adding it to glass tanks lined with sediments brought from the same site from which water samples were collected. Each tank was first lined with a gravel layer, followed by a coarse sediment layer, a fine sediment layer, then it was filled up to its half height (the mean water depth was  $15.0 \pm 3.0$  cm) with water acquired from El-Mahmoudia canal. Over the course of the experiment water was partially renewed to compensate for evaporation, maintain stable water level, growth conditions and minimize nutrient depletion and metal concentration drift. The individuals of the two species were then added to separate CW tanks, where duplicate tanks were constructed and used for growing the plants of each species. For *H. umbellata*, 32 individual plant specimens were planted in each CW tank, where each individual plant possessed four leaves. For *S. auriculata*, 32 individual plant specimens were planted in each CW tank, where each individual is a ramets with an average width of 5 cm and average length of 6 cm. The water level was kept constant during the experiment. The growth and survival of the individuals of the two species were observed.

## Plant Material Sampling

Plant material of each species was collected from the tanks where the macrophytes were planted as explained in the previous section seasonally. The collected plant material was rinsed with tap water, then with distilled water. Individuals of each species were divided into 3 parts: leaves, stolon, and rhizoids. The parts were weighed separately and then dried in an electric oven at 60 °C for 48 h. The plant material was weighed until reaching constant weight and then was finely grounded in a porcelain mortar. Six individuals of each species were collected from the water tanks and processed for subsequent analysis as mentioned above.

Water samples from El-Mahmoudia Canal, used in the experiments, were analyzed (Table S1). In addition, seasonal water samples were gathered in triplicate from each tank using plastic vials for subsequent analysis.

## Determination of HMs

HMs concentrations were determined for plant specimens of *S. auriculata*, and *H. umbellata*, soil and water samples. Powdered plant specimens were acid digested, where 0.1 gm of plant material was combined with 6 ml of 4% nitric acid and 0.5 ml of 40% HF in Teflon tubes. The Teflon tubes were tied in a system to be more sealed when heated in oven to over 130 °C for 24 h. The digest was diluted with distilled

water to reach 25 ml. HMs concentrations in water samples were determined after preparation according to US EPA (1994) “Method 200.7”, where 50 ml of unfiltered water samples were digested by acidifying the samples with 4%  $\text{HNO}_3$  then heated on hot plate at 60–70 °C for 3 h. Then the samples were heated in the oven overnight at 55 °C. For preparation of soil samples for analysis of metal concentrations, 0.1 gm of soil samples were digested with 3 ml of 4% nitric acid and 2 ml HF in Teflon tube. Teflon tubes were tied in a system to be more sealed when heated in oven to over 55 °C for 24 h. The digest was diluted with distilled water and the volume was made up to 25 ml. Determination of copper (Cu), zinc (Zn), chromium (Cr), and lead (Pb), were done following the method of APHA using the Atomic Absorption Spectrophotometer (Model: ANALYTIK JENA CONTRAA 300).

## Estimating Phytoremediation Indicators

Phytoremediation indicators were evaluated over multiple seasons to identify when each species reaches its optimal phytoremediation performance. Previous studies have documented pronounced seasonal patterns in metal accumulation by aquatic plants such as *Pistia stratiotes*, *Eichhornia crassipes*, and *Salvinia* spp., emphasizing the need to account for seasonal dynamics when evaluating phytoremediation performance (Nan et al. 2002; Galal et al. 2018). Seasonal assessments are particularly important for rhizofiltration studies, as root activity and rhizosphere interactions which govern metal adsorption and uptake are sensitive to seasonal changes including fluctuation in temperature and redox conditions, pH, dissolved oxygen, and salinity as well as plant physiological activity including growth rate, root biomass, and metabolic demand (Cheng 2003; Eid and Shaltout 2014; Vymazal and Březinová 2015). The rhizofiltration potential represents the total amount of HMs extracted from the defined amount of solution in mg heavy metal  $\text{m}^{-2} \text{year}^{-1}$ . It was calculated according to Veselý et al. (2011) as follows:

$$\text{RP} = \left( \frac{C_{\text{leaves}} * M_{\text{leaves}}}{M_{\text{total}}} + \frac{C_{\text{roots}} * M_{\text{roots}}}{M_{\text{total}}} \right) * M_{\text{plant}}$$

Where  $M_{\text{leaves}}$  is the leaf dry weight in g;  $M_{\text{total}}$  is the total dry weight of leaves and roots in g;  $C_{\text{leaves}}$  is the concentration of heavy metal in leaves in mg/kg;  $M_{\text{roots}}$  is the roots dry weight in g;  $C_{\text{roots}}$  is the concentration of heavy metal in roots in mg/kg and  $M_{\text{plant}}$  is the annual mean yield of plant in g dry weight.  $\text{m}^{-2} \text{year}^{-1}$ .

The bio-concentration factor (BAF) is a useful parameter to estimate the potentiality of plants for accumulating metals. While the translocation factor (TF) shows the capability

**Table 1** Shows the mean conc. of HMs (mg/kg) accumulated by different organs of *S. auriculata* and *H. umbellata* from deep water treatment and constructed wetland.

| Treatment | Plant part |        | <i>S. auriculata</i> |       |        |       | <i>H. umbellata</i> |       |        |       |
|-----------|------------|--------|----------------------|-------|--------|-------|---------------------|-------|--------|-------|
|           |            |        | Cu                   | Pb    | Zn     | Cr    | Cu                  | Pb    | Zn     | Cr    |
| DW        | Shoot      | Mean   | 41.72                | 37.17 | 697.50 | 27.90 | 28.04               | 17.68 | 508.86 | 25.42 |
|           |            | CV (%) | 0.23                 | 0.94  | 0.73   | 0.17  | 0.47                | 0.50  | 1.48   | 0.43  |
|           | Rhizome    | Mean   | 33.59                | 22.19 | 394.81 | 28.23 | 40.23               | 20.51 | 160.09 | 24.99 |
|           |            | CV (%) | 0.37                 | 0.32  | 0.29   | 0.43  | 0.39                | 0.38  | 0.75   | 0.24  |
|           | Root       | Mean   | 53.58                | 34.11 | 508.75 | 26.56 | 45.23               | 45.41 | 574.80 | 53.03 |
|           |            | CV (%) | 0.29                 | 0.28  | 0.28   | 0.19  | 0.49                | 0.90  | 0.74   | 0.88  |
| CW        | Shoot      | Mean   | 73.13                | 10.04 | 119.49 | 33.65 | 15.79               | 11.35 | 66.56  | 30.92 |
|           |            | CV (%) | 1.00                 | 0.63  | 0.88   | 0.31  | 0.32                | 1.12  | 0.59   | 0.17  |
|           | Rhizome    | Mean   | 53.39                | 13.56 | 324.74 | 23.91 | 38.08               | 12.47 | 120.57 | 24.94 |
|           |            | CV (%) | 0.24                 | 0.85  | 0.50   | 0.28  | 0.30                | 1.09  | 0.51   | 0.35  |
|           | Root       | Mean   | 80.18                | 26.49 | 648.75 | 34.09 | 141.35              | 17.37 | 468.56 | 27.20 |
|           |            | CV (%) | 0.08                 | 0.55  | 0.68   | 0.48  | 0.14                | 0.62  | 0.33   | 0.16  |

Maximum and minimum values are italics. *DW* deep water treatment, *CW* constructed wetland, *CV* coefficient of variation

of the plant to translocate HMs from roots to leaves. The BAF and TF were determined according to Abhilash et al. (2009a, b) as follows:

$$TF = \frac{\text{Concentration of metal in plant shoot (mg/kg)}}{\text{Concentration of metal in plant root (mg/kg)}}$$

$$BAF = \frac{\text{Concentration of metal in plant root (mg/kg)}}{\text{Concentration of metal in water (mg/l)}}$$

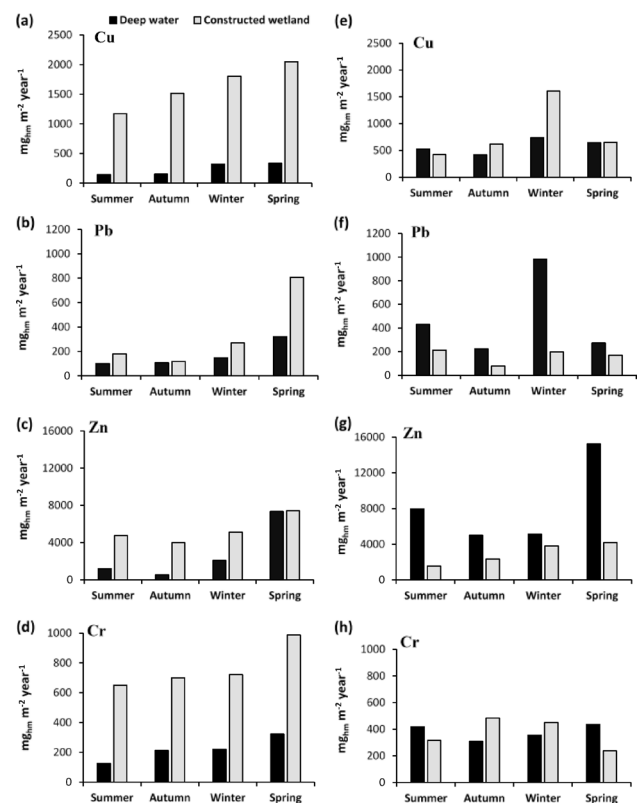
The bioaccumulation factor (BAF) and the translocation factor (TF) were calculated to determine the level of metal accumulation in the plants grown in El-Mahmoudia canal water.

## Results

### The Phytoremediation Capacity of the Two Macrophytes

The results indicated that *Salvinia auriculata* accumulated higher amounts of Cu in its rhizoids under the constructed wetland (CW) treatment compared to the deep water (DW) treatment. The shoots of *S. auriculata* showed the highest accumulation of Pb and Zn under the DW treatment, while the lowest concentrations of these metals were observed in the shoots under the CW treatment (Table 1). Additionally, *S. auriculata* accumulated the highest concentration of Cr in its shoot and the lowest in its root under the CW treatment.

For *Hydrocotyle umbellata*, the highest Cu concentration was found in the shoot under the CW treatment, whereas the lowest concentration occurred in the root. The roots of *H. umbellata* accumulated the highest levels of Pb



**Fig. 2** Rhizofiltration of Cu, Pb, Zn, and Cr by *H. umbellata* (a), (b), (c) and (d); and *S. auriculata* (e), (f), (g) and (h) grown in deep water and constructed wetland

and Zn under the DW treatment, while the lowest concentrations were recorded in the shoot under the CW treatment. The highest Cr concentration in *H. umbellata* was observed in the root under DW treatment, while the lowest was found in the rhizome under CW treatment (Table 1). Overall, *H. umbellata* tended to accumulate

**Table 2** Rhizofiltration potential (RP) in  $\text{mg. m}^{-2} \cdot \text{year}^{-1}$  of both *H. umbellata* and *S. auriculata* grown in deep water (DW) and constructed wetland (CW). The highest mean RP values are in italics. CV: coefficient of variation

| Treatment |        | Rhizofiltration of <i>S. auriculata</i><br>( $\text{mg. m}^{-2} \cdot \text{year}^{-1}$ ) |        |         |        | Rhizofiltration of <i>H. umbellata</i><br>( $\text{mg. m}^{-2} \cdot \text{year}^{-1}$ ) |        |         |               |
|-----------|--------|-------------------------------------------------------------------------------------------|--------|---------|--------|------------------------------------------------------------------------------------------|--------|---------|---------------|
|           |        | Cu                                                                                        | Pb     | Zn      | Cr     | Cu                                                                                       | Pb     | Zn      | Cr            |
| DW        | mean   | 582.23                                                                                    | 477.07 | 8325.26 | 379.30 | 237.58                                                                                   | 168.51 | 2781.50 | 220.23        |
|           | CV (%) | 0.24                                                                                      | 0.73   | 0.58    | 0.15   | 0.44                                                                                     | 0.61   | 1.12    | 0.37          |
| CW        | mean   | 824.51                                                                                    | 163.79 | 2974.42 | 372.00 | <i>1635.00</i>                                                                           | 343.84 | 5320.96 | <i>764.41</i> |
|           | CV (%) | 0.64                                                                                      | 0.37   | 0.42    | 0.31   | 0.23                                                                                     | 0.92   | 0.28    | 0.20          |

higher concentrations of the studied elements in the root compared to the rhizome or shoot.

## Rhizofiltration

In general, both species demonstrated greater rhizofiltration potential under colder conditions (winter and spring) compared to warmer seasons (summer and autumn), as reflected by higher values of the rhizofiltration index during the colder periods (Fig. 2).

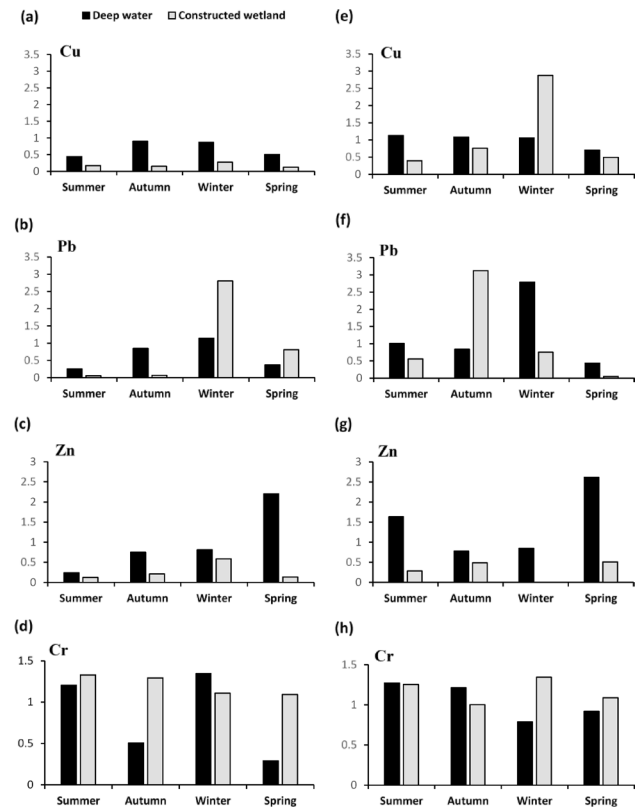
*Hydrocotyle umbellata* grown in deep water (DW) treatment exhibited the highest rhizofiltration potential for Cr, Cu, Zn, and Pb during spring. Rhizofiltration by *H. umbellata* in DW was notably enhanced in colder seasons, showing increased values in winter and spring compared to summer and autumn. Similarly, when grown in the constructed wetland (CW) treatment, *H. umbellata* followed a comparable seasonal trend, reaching its peak rhizofiltration capacity in spring for all tested heavy metals (HMs).

Interestingly, *H. umbellata* grown in CW showed a higher rhizofiltration capacity for Cu than those in DW. For both treatments, rhizofiltration of Zn was generally higher than for the other investigated HMs, whereas Pb showed the lowest rhizofiltration potential. Overall, *H. umbellata* exhibited greater rhizofiltration capacity in CW compared to DW for all metals examined.

*Salvinia auriculata* also showed improved rhizofiltration performance for Cu under colder conditions in DW, with the rhizofiltration index significantly higher in winter and spring than in summer and autumn (Fig. 2). A similar seasonal trend was observed for Cr, where rhizofiltration declined from summer to autumn, then gradually increased through winter, reaching a peak in spring.

The rhizofiltration of Zn by *S. auriculata* in DW peaked in spring, following a decrease from summer to autumn and a subsequent rise through winter. In contrast, Pb rhizofiltration declined from summer to its lowest in autumn, peaked in winter, and then declined again in spring.

Among the four heavy metals, *S. auriculata* exhibited the highest rhizofiltration potential for Zn and the lowest for Cr. When grown in both DW and CW treatments, the highest rhizofiltration of Pb occurred in winter. Notably,

**Fig. 3** Translocation of Cu, Pb, Zn, and Cr by *H. umbellata* (a), (b), (c) and (d); and *S. auriculata* (e), (f), (g) and (h) grown in deep water and constructed wetland

Pb rhizofiltration decreased from summer to autumn, increased sharply in winter, and declined again in spring.

Cu rhizofiltration trends in CW-grown *S. auriculata* mirrored those observed in DW. Rhizofiltration increased from summer to autumn, peaked in winter, and dropped in spring. Similarly, Zn rhizofiltration increased gradually throughout the year, reaching its highest levels in spring. Cr rhizofiltration peaked in early autumn and winter, then decreased during the transition from winter to spring, reaching its lowest levels in the latter season.

Overall, the rhizofiltration potential of *S. auriculata* for Pb, Zn, and Cr was higher in the DW treatment, while Cu rhizofiltration was greater in CW (Table 2). In contrast, *H. umbellata* demonstrated a higher rhizofiltration capacity

**Table 3** Translocation of both *H. umbellata* and *S. auriculata* grown in deep water (DW) and in the constructed wetland (CW)

| Treatment |        | <i>S. auriculata</i> |      |      |      | <i>H. umbellata</i> |      |      |      |
|-----------|--------|----------------------|------|------|------|---------------------|------|------|------|
|           |        | Cu                   | Pb   | Zn   | Cr   | Cu                  | Pb   | Zn   | Cr   |
| DW        | Mean   | 0.99                 | 1.27 | 1.47 | 1.05 | 0.68                | 0.65 | 1.00 | 0.84 |
|           | CV (%) | 0.20                 | 0.82 | 0.58 | 0.22 | 0.35                | 0.64 | 0.84 | 0.62 |
| CW        | Mean   | 1.13                 | 1.12 | 0.32 | 1.17 | 0.18                | 0.94 | 0.26 | 1.21 |
|           | CV (%) | 1.04                 | 1.22 | 0.72 | 0.13 | 0.37                | 1.38 | 0.85 | 0.10 |

CV coefficient of variation

for Cu and Cr in CW, whereas *S. auriculata* was more effective at rhizofiltration of Pb and Zn in DW.

## Translocation

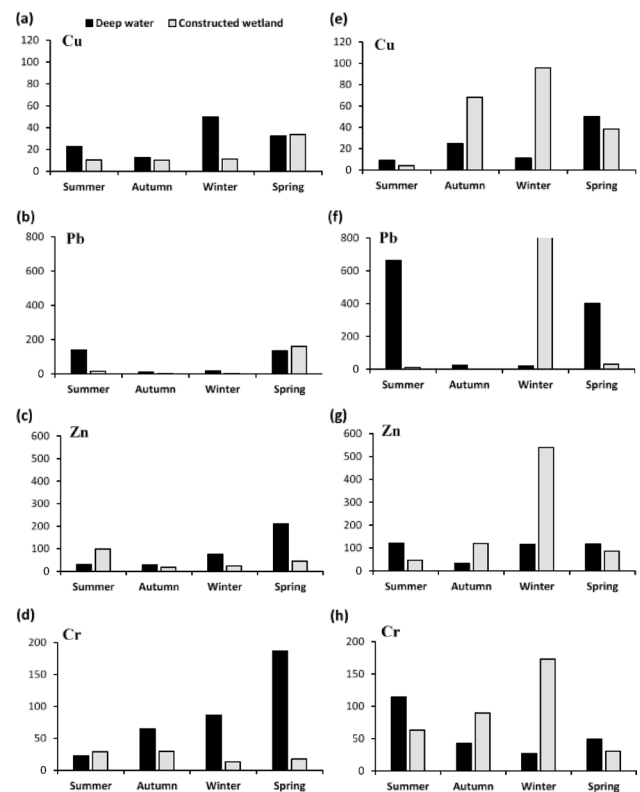
For *Hydrocotyle umbellata* grown under the deep water (DW) treatment, translocation of Cu was lowest in summer, peaked in autumn, and then declined through winter and spring (Fig. 3). Pb translocation gradually increased from summer, reaching its maximum in winter before decreasing in spring. Zn translocation also followed a seasonal upward trend, increasing across the four seasons and reaching its highest value in spring. In contrast, Cr translocation decreased in autumn relative to summer, peaked in winter, and then dropped to its lowest level in spring.

Under the constructed wetland (CW) treatment, *H. umbellata* exhibited a different pattern for Cu translocation, which decreased from summer to autumn, peaked in winter, and declined again in spring. For Pb and Zn, the translocation trends were similar: both recorded their lowest values in summer, increased slightly in autumn, peaked in winter, and dropped in spring. Cr translocation consistently declined throughout the seasons, reaching its lowest value in spring.

For *Salvinia auriculata* in the DW treatment, Cu translocation declined steadily from summer to spring, reaching its minimum in the latter season (Fig. 3). Pb translocation decreased from summer to autumn, peaked in winter, and declined again in spring. The highest Zn translocation occurred in spring, following a decrease from summer to autumn and a subsequent increase in winter. Cr translocation was more effective in warmer conditions, with higher values observed in summer and autumn compared to winter and spring.

In the CW treatment, *S. auriculata* exhibited an increase in Cu translocation from summer to autumn, with a peak in winter followed by a decline in spring. Pb translocation reached its maximum in early autumn, then decreased through winter and reached its lowest point in spring. Zn translocation increased from summer to autumn, dropped in winter, and then peaked in spring. Cr translocation showed a decrease from summer to autumn, a peak in winter, and a decline again in spring.

Overall, *S. auriculata* displayed higher translocation of Pb and Zn in DW treatment compared to CW, while



**Fig. 4** Bioaccumulation of Cu, Pb, Zn, and Cr by *H. umbellata* (a), (b), (c) & (d); and *S. auriculata* (e), (f), (g) and (h) grown in deep water and constructed wetland

translocation of Cu and Cr was greater in CW (Table 3). For *H. umbellata*, Cu and Zn translocation was higher in DW than in CW, whereas Pb and Cr translocation was greater in CW. Specifically, *S. auriculata* showed the highest translocation of Cu in CW and the highest translocation of Pb and Zn in DW. *H. umbellata*, on the other hand, demonstrated superior Cr translocation in CW treatment.

## Bioaccumulation

*Hydrocotyle umbellata* grown in the deep water (DW) treatment exhibited higher bioaccumulation of Cu during colder seasons, with greater levels recorded in winter and spring compared to summer and autumn (Fig. 4). Pb bioaccumulation peaked in summer, declined in autumn and winter, and then rose again in spring. Zn bioaccumulation decreased

**Table 4** Bioaccumulation of both *H. umbellata* and *S. auriculata* grown in deep water (DW) and constructed wetland (CW)

| Treatment |        | <i>S. auriculata</i> |           |        |       | <i>H. umbellata</i> |       |       |       |
|-----------|--------|----------------------|-----------|--------|-------|---------------------|-------|-------|-------|
|           |        | Cu                   | Pb        | Zn     | Cr    | Cu                  | Pb    | Zn    | Cr    |
| DW        | Mean   | 23.83                | 276.63    | 96.55  | 58.12 | 29.38               | 74.95 | 86.31 | 90.34 |
|           | CV (%) | 0.79                 | 1.13      | 0.44   | 0.66  | 0.54                | 0.95  | 1.00  | 0.77  |
| CW        | Mean   | 51.66                | 249333.74 | 197.95 | 88.99 | 16.35               | 44.32 | 46.15 | 22.37 |
|           | CV (%) | 0.76                 | 2.00      | 1.16   | 0.68  | 0.71                | 1.75  | 0.80  | 0.36  |

CV coefficient of variation

from summer to autumn but gradually increased in winter and reached its maximum in spring. In the case of Cr, bioaccumulation increased progressively from summer to reach its highest level in spring.

Under the constructed wetland (CW) treatment, *H. umbellata* showed a decrease in Cu bioaccumulation from summer to autumn, followed by a gradual increase in winter and a marked peak in spring (Fig. 4). Pb bioaccumulation reached its maximum in spring. For Zn, bioaccumulation was high in summer, dropped to its lowest in autumn, and then gradually increased through winter and spring. *H. umbellata* exhibited better Cr bioaccumulation under warmer conditions, with higher values during summer and autumn compared to winter and spring.

In *Salvinia auriculata* grown under DW treatment, Cu bioaccumulation increased from summer through autumn and reached its highest level in spring (Fig. 4). Pb bioaccumulation peaked in summer, decreased in autumn and reached its lowest level in winter, before increasing again in spring. For Zn, the maximum bioaccumulation was observed in summer, followed by a decrease in autumn and a gradual increase through winter and spring. Cr bioaccumulation was higher under warmer conditions, with greater levels during summer and autumn compared to winter and spring.

In the CW treatment, *S. auriculata* exhibited the lowest Cu bioaccumulation in summer, which increased in autumn, peaked in winter, and declined in spring (Fig. 4). Pb bioaccumulation decreased in autumn compared to summer, reached its maximum in winter, and dropped again in spring. The trend for Zn was similar to Cu: lowest in summer, increasing through autumn, peaking in winter, and declining in spring. Cr bioaccumulation increased from summer to winter and then dropped to its minimum level in spring.

Overall, *S. auriculata* showed the highest potential for bioaccumulation of the investigated heavy metals (HMs) during winter compared to other seasons.

As shown in Table 4, *S. auriculata* accumulated higher levels of all investigated HMs in the CW treatment compared to the DW treatment. In contrast, *H. umbellata* exhibited greater bioaccumulation for all metals under DW conditions than in CW. Specifically, *S. auriculata* showed higher bioaccumulation for Cu, Pb, and Zn in CW treatment, whereas *H. umbellata* exhibited higher Cr bioaccumulation in DW treatment.

## Discussion

Phytoremediation is increasingly recognized as an environmentally sustainable and cost-effective alternative to conventional remediation technologies (Rakshit et al. 2017). This study evaluated the phytoremediation performance of *Hydrocotyle umbellata* and *Salvinia auriculata* under contrasting hydrological systems and seasonal conditions, with a particular focus on rhizofiltration, translocation, and bioaccumulation of Cu, Pb, Zn, and Cr. The discussion below is organized around (1) seasonal drivers, (2) system-level effects, and (3) species-specific responses, integrating environmental controls such as temperature, pH as, salinity, and water depth.

### Seasonal Drivers of Phytoremediation Efficiency

Metal absorption by aquatic plants is governed by multiple interacting factors, including plant species, metal type, bioavailability, and environmental conditions such as temperature, pH, redox potential, cation exchange capacity, dissolved oxygen, and root exudates (Cheng 2003). Seasonal variation in metal capture arises from changes in environmental conditions (e.g., metal concentrations, pH, organic carbon, and element interactions), plant physiological processes such as growth-related dilution, or interactions between these drivers (Nan et al. 2002; Duman et al. 2007; El-Liethy et al. 2022).

Metal accumulation also differs among elements due to variations in solubility, mobility, and bioavailability (Duman et al. 2007; Du et al. 2020). Differences in uptake and translocation have been attributed to metal-specific mobility and plant uptake preferences (Miretzky et al. 2004; Dhir et al. 2011). Metals with limited bioavailability are generally more resistant to phytoextraction (Oladoye et al. 2022), with Cu and Zn being more readily available than Pb and Cr in aquatic systems (Prasad 2003).

Seasonality emerged as a key control on rhizofiltration efficiency in both investigated macrophytes, with generally higher metal uptake during colder seasons (winter and spring) than during summer and autumn. This pattern aligns with previous studies reporting higher metal concentrations in plant tissues during early or cooler growth

stages, followed by declines associated with biomass dilution and metabolic stress during warmer periods (Vymazal and Březinová 2015). Seasonal variation in rhizofiltration reflects combined effects of environmental controls on metal solubility and plant physiological activity, including temperature-driven changes in metabolism, growth rate, and root function (Nan et al. 2002; Duman et al. 2007).

Rhizofiltration operates through the adsorption and uptake of dissolved contaminants by plant roots within the rhizosphere, which is directly exposed to contaminated water. This mechanism is typically evaluated under controlled or semi-controlled aquatic conditions to isolate root-mediated processes (Sharma et al. 2020). *Hydrocotyle umbellata* is relatively cold tolerant and exhibits strong nutrient uptake capacity (Chandra et al. 1997). Its ability to remove heavy metals from aquatic systems has been widely reported (Panyakhan et al. 2006; Khilji and Bareen 2008; Bokhari et al. 2022). In the present study, *H. umbellata* showed its highest rhizofiltration potential for Cu, Cr, Pb, and Zn during spring in both DW and CW systems, with consistently greater uptake of Zn, corroborating earlier findings (Khilji and Bareen 2008; Bokhari et al. 2022).

For *Salvinia auriculata*, rhizofiltration of Cu and Pb was higher in colder seasons, while Zn uptake peaked in spring. In contrast, Cr uptake was highest in early autumn and winter before declining toward spring. A significant positive correlation between water salinity and Pb rhizofiltration was observed, suggesting that increased ionic strength enhances Pb bioavailability. Similar effects have been reported for Cd under saline conditions, where salinity increased metal mobility and plant uptake (Zhou et al. 2019).

Metal-specific behavior was evident across seasons. Zn and Cu consistently exhibited higher rhizofiltration than Pb and Cr, reflecting their greater solubility and bioavailability (Prasad 2003; Oladoye et al. 2022). In contrast, Pb and Cr showed lower uptake due to reduced mobility and stronger binding to sediments (Miretzky et al. 2004; Dhir et al. 2011). Seasonal changes in redox conditions, dissolved oxygen, and organic carbon availability further influenced metal behavior in the aquatic systems (Cheng 2003; Du et al. 2020).

Seasonal assessment was intentionally incorporated to capture the dynamic nature of rhizofiltration under field-relevant conditions. Uptake and retention of metals by aquatic macrophytes vary seasonally due to fluctuations in environmental parameters and plant physiological activity, including biomass allocation and root development (Nan et al. 2002; Duman et al. 2007; Vymazal and Březinová 2015). Such seasonal patterns have been widely reported for aquatic plants, including *Pistia stratiotes*, *Eichhornia crassipes*, *Salvinia* spp., *Phragmites australis*, *Typha domingensis*,

and other aquatic species (Nan et al. 2002; Galal et al. 2018; Mohammed et al. 2023).

Translocation of Cu, Pb, Zn, and Cr also showed seasonal variability. The reported cold tolerance of *H. umbellata* (Khilji and Bareen 2008), together with negative correlations between metal translocation and both temperature and pH, likely explains its higher translocation potential during winter (1 Supplementary Material: Table S5). In contrast, *S. auriculata* exhibited maximum translocation during spring, except for Cr, which peaked in winter. Plant-mediated processes such as chelation, redox reactions, and pH modification enhance metal solubilization and uptake, even at low ambient concentrations. In this study, positive correlations between temperature, pH, and Cr and Pb bioaccumulation suggest enhanced uptake under warmer conditions, consistent with previous observations in other hydrophytes (San-yahumbi et al. 1998).

Although seasonal variation in phytoremediation efficiency was assessed, all experimental plants were established simultaneously and therefore shared a common life history prior to seasonal measurements. Consequently, observed seasonal patterns reflect short-term physiological and environmental responses within a single growth cycle rather than cumulative effects of plant age or prior growing seasons. This design minimizes ontogenetic bias and allows seasonal differences in metal uptake to be attributed primarily to variations in temperature, pH, salinity, water chemistry, and plant metabolic activity (Nan et al. 2002; Duman et al. 2007). Similar approaches have been widely applied in phytoremediation studies, where seasonal fluctuations in metal accumulation were interpreted as responses to environmental forcing rather than long-term plant history (Eid and Shaltout 2014; Vymazal and Březinová 2015; Mohammed et al. 2023). Nonetheless, longer-term, multi-year experiments incorporating plant age structure would further refine understanding of interannual variability in rhizofiltration performance.

## System Effects: Deep Water versus Constructed Wetlands

Hydrological system design strongly influenced phytoremediation performance. Constructed wetlands (CW) generally enhanced rhizofiltration efficiency compared with deep water (DW) systems, particularly for *Hydrocotyle umbellata*. CW provides shallow water depth, longer hydraulic residence time, increased root-water contact, and more stable physicochemical conditions, all of which promote metal uptake and retention (Stefanakis 2018; Zapana et al. 2020).

Cultivation of *H. umbellata* in CW markedly improved its rhizofiltration effectiveness, whereas system type had a weaker effect on *Salvinia auriculata*. In CW, *H. umbellata*

exhibited higher Cu rhizofiltration potential than in DW systems. For both macrophytes, a consistent trend in rhizofiltration capacity was observed, with Zn showing the highest uptake, followed by Cu, Cr, and Pb. Previous studies have reported tolerance of *H. umbellata* to Zn, Cu, and Cr at various concentrations in wastewater (e.g., Khilji and Barea 2008). In contrast, Pb and Cr toxicity has been shown to reduce biomass production and induce leaf chlorosis in *H. umbellata* (Yongpisanphop et al. 2005).

Metal bioaccumulation rankings differed between species and systems. For *H. umbellata*, the order of accumulation shifted from Cr>Zn>Pb>Cu in DW to Zn>Pb>Cr>Cu in CW. In contrast, *S. auriculata* showed a consistent accumulation pattern (Pb>Zn>Cr>Cu) regardless of system type. The higher Pb accumulation by *H. umbellata* in CW compared to DW aligns with findings by Yongpisanphop et al. (2005), suggesting that sediment interactions in CW enhance Pb availability, whereas DW conditions limit Pb uptake.

Copper accumulation negatively affected growth parameters in both species. Increased Cu bioaccumulation was inversely correlated with shoot dry weight and rhizoids length, and Cu translocation was negatively correlated with rhizoids biomass in *S. auriculata* grown in CW (1 Supplementary Material: Table S4). Strong Cu sorption can damage root tissues and impede translocation to shoots (McBride 2001; Uchimiya et al. 2020). Consistent with this, Cu translocation was negatively associated with shoot concentration and fresh weight of *S. auriculata*, potentially affecting overall plant size. Similar Cu-induced growth inhibition has been reported for corn (McBride 2001), woody plants (Alaoui-Sossé et al. 2004), *S. auriculata* (Bizzo et al. 2014), and *Salvinia minima* (Al-Hamdani and Blair 2004). Negative correlations between Pb and Zn bioaccumulation and fresh weight further indicate growth constraints under elevated metal exposure, consistent with the general principle that both metal excess and deficiency can restrict plant development (Musielńska et al. 2016).

Seasonal patterns interacted with system type, particularly for *S. auriculata*. In CW, *S. auriculata* exhibited its highest bioaccumulation potential during winter, with a negative correlation between temperature and Zn accumulation in rhizoids. In DW systems, temperature was negatively correlated with Cu concentrations in shoots. Cr dynamics showed distinct seasonal behavior, with positive correlations between temperature and Cr translocation, as well as between Cr translocation and plant fresh weight, indicating seasonal regulation of Cr uptake. Similar associations between seasonal growth and phytoremediation capacity have been reported for *Pontederia crassipes*, although growth reduction was attributed to metal toxicity (Muramoto and Oki 1983). Chromium toxicity has also been shown to

reduce biomass and growth rates in *S. minima* (Nichols et al. 2000; Wolff et al. 2012). In contrast, Cr bioaccumulation in this study positively influenced rhizoids length, and Cr translocation was strongly associated with rhizoids and total dry weight, suggesting metal-specific physiological responses.

For *H. umbellata*, rhizofiltration of Cu, Cr, Pb, and Zn was consistently higher in CW than in DW, particularly during spring. This enhanced performance likely reflects improved rhizosphere development and sediment-root interactions in CW, where rooted growth forms increase exposure to both dissolved and sediment-bound metals (Zwolsman et al. 1993). Rooted macrophytes are known to accumulate higher metal concentrations due to direct access to contaminated sediments (Sawidis et al. 1995; Bonanno and Giudice 2010).

In contrast, *S. auriculata* showed less sensitivity to system type, likely due to its free-floating habit and reliance on water-column metals. Nevertheless, CW conditions still promoted higher overall bioaccumulation and translocation, underscoring the importance of hydrodynamics, depth, and sediment interactions in shaping phytoremediation outcomes.

Beyond phytoremediation performance, CW support dense stands of aquatic vegetation and diverse microbial and faunal communities, enhancing contaminant transformation and ecosystem functioning. As engineered systems that mimic natural wetlands, CW offer multiple economic and environmental advantages, including low energy requirements, minimal chemical inputs, ease of operation, and cost-effectiveness relative to conventional wastewater treatment technologies (Stefanakis 2018; Zapana et al. 2020).

## Species-Specific Phytoremediation Responses

Clear interspecific differences were observed in rhizofiltration, translocation, and bioaccumulation strategies. *Hydrocotyle umbellata* exhibited strong rhizofiltration capacity, particularly for Zn and Cu, but limited metal translocation (TF<1 for most metals), indicating a phytostabilization-dominated strategy. Metals were largely retained in the root system, a recognized tolerance mechanism that reduces toxicity in aerial tissues (Yanqun et al. 2005; Eid and Shaltout 2014; Bokhari et al. 2022).

Rhizofiltration by both species varied seasonally, with spring generally showing the highest uptake, particularly for *H. umbellata*. In contrast, *Salvinia auriculata* exhibited higher overall rhizofiltration potential than *H. umbellata*, and metal uptake was proportional to metal concentrations in the surrounding water. This pattern is consistent with previous findings showing increased trace metal accumulation in aquatic plants with rising environmental concentrations

(Favas et al. 2012; Eid and Shaltout 2014; Mirza et al. 2014; Galal and Shehata 2015). Similar correlations between metal concentrations in plant tissues and water have been reported for *Pistia stratiotes* (Galal et al. 2018). The genus *Salvinia* is generally recognized for its high metal accumulation capacity (Dhir and Srivastava 2011), although metal-specific uptake is influenced by differences in mobility, species preference, and translocation rates (Miretzky et al. 2004; Dhir et al. 2011).

Despite higher translocation and bioaccumulation in *S. auriculata*, rhizofiltration by *H. umbellata* grown in DW exceeded that of *S. auriculata*. This likely reflects the emergent and rooted growth form of *H. umbellata*, which facilitates interaction with sediment-associated metals. Rooted macrophytes are more strongly influenced by sediment-bound contaminants than free-floating species and therefore often exhibit higher metal accumulation (Zwolsman et al. 1993). Roots and rhizomes of emergent plants such as *Phragmites australis* and *Typha domingensis* have been shown to accumulate substantial metal loads due to extensive intercellular air spaces in cortical tissues (Sawidis et al. 1995; Bonanno and Giudice 2010).

Mean translocation factor values for *H. umbellata* were generally  $< 1$  (except for Zn in DW and Cr in CW), confirming limited root-to-shoot transport and supporting its suitability for phytostabilization of Cu, Pb, Zn, and Cr (Eid and Shaltout 2014). Restricting metal movement to above-ground tissues is a key tolerance mechanism that minimizes phytotoxicity (Yanqun et al. 2005; Bokhari et al. 2022). Additional tolerance strategies may include conversion of metals into less toxic forms and their subsequent release through transpiration (Fritioff and Greger 2006). These findings align with earlier studies demonstrating the effectiveness of *H. umbellata* in phytofiltration of multiple heavy metals from wastewater (Yongpisanphop et al. 2005; Khilji and Barea 2008; Ali et al. 2018; Bokhari et al. 2022).

In contrast, *S. auriculata* exhibited higher translocation factors ( $TF \geq 1$  for several metals), particularly in spring, indicating stronger phytoextraction potential (Masarovičová et al. 2010; Bader et al. 2019). However, this strategy was associated with physiological costs. Significant negative correlations between Zn and Cu accumulation and growth-related traits (size index, biomass, and rhizoid length) indicate metal-induced stress. Similar growth inhibition due to Cu and Pb toxicity has been widely reported for *Salvinia* species (Alaoui-Sossé et al. 2004; Bizzo et al. 2014; Al-Hamdani and Blair 2004).

Bioaccumulation factors ( $BAF > 1$ ) for both species across metals confirm their suitability as accumulators and biomonitors (Bose et al. 2008; Bonanno 2013). Overall, *S. auriculata* showed higher BAF values than *H. umbellata* in both DW and CW systems, consistent with previous reports

that free-floating macrophytes often have greater accumulation capacity than emergent species (Zayed et al. 1998). Nevertheless, *H. umbellata* showed higher bioaccumulation when grown in DW and outperformed *S. auriculata* for most metals except Pb, for which *S. auriculata* remained the more effective accumulator.

Metal-specific differences were also evident. Cu exhibited the lowest bioaccumulation among the investigated metals in both species, despite previous reports highlighting *H. umbellata* as a good Cu accumulator (Prasad 2003). Higher bioconcentration factors for Cr and Zn suggest greater affinity for these metals, consistent with earlier studies (Khilji and Barea 2008). Elevated Zn levels may also inhibit Cu uptake, further explaining the reduced Cu accumulation (Tóth et al. 2016). Bioaccumulation of Cu and Zn peaked in spring, and metals were predominantly retained in roots rather than shoots, particularly in *H. umbellata*.

Environmental variables further modulated bioaccumulation patterns. Higher temperature and pH enhanced Cr and Pb uptake, while increased salinity promoted Pb accumulation, reinforcing the importance of abiotic controls on phytoremediation efficiency (Sanyahumbi et al. 1998; Zhou et al. 2019).

## Implications for Phytoremediation and Wetland Design

The outcomes revealed the bioaccumulation capability of both species; and their potential for remediation of the contaminated water, and possibility of using the macrophytes in biomonitoring of the examined HMs. Overall, both macrophytes demonstrated substantial potential for the remediation and biomonitoring of heavy metals in aquatic systems. *H. umbellata* is particularly suited for phytostabilization in constructed wetlands, while *S. auriculata* offers stronger phytoextraction potential under controlled conditions. These findings underscore the importance of matching species traits with system design, seasonal timing, and environmental conditions to maximize remediation efficiency.

Constructed wetlands, by mimicking natural wetland processes and providing stable hydrological and physico-chemical conditions, represent an effective and low-cost platform for integrating such macrophytes into wastewater treatment and pollution mitigation strategies (Stefanakis 2018; Zapana et al. 2020).

The observed species-, season-, and system-specific differences in metal uptake highlight the importance of adaptive phytoremediation design in contaminated aquatic environments. Constructed wetlands, when planted with functionally complementary macrophytes, can be optimized to enhance metal removal efficiency by aligning species selection with hydrological conditions and seasonal

dynamics. Rooted species such as *Hydrocotyle umbellata* are more suitable for phytostabilization in shallow wetland systems, where prolonged root–sediment interaction limits metal mobility, whereas free-floating species such as *Salvinia auriculata* are better suited for phytoextraction from the water column under controlled conditions. Seasonal timing should be considered in management plans, as metal uptake efficiency was highest during cooler periods. Integrating these considerations into wetland design and operation can improve contaminant removal while minimizing ecological stress, supporting the use of nature-based solutions for sustainable water quality management.

## Conclusion

This study demonstrates that both *Hydrocotyle umbellata* and *Salvinia auriculata* possess substantial potential for the phytoremediation of Cu, Pb, Zn, and Cr in aquatic systems. However, clear interspecific differences were evident in their remediation strategies. *H. umbellata* primarily exhibited a phytostabilization-dominated mechanism, characterized by strong rhizofiltration and restricted metal translocation, resulting in the retention of metals within the root system. In contrast, *S. auriculata* showed higher rhizofiltration, translocation, and bioaccumulation capacities, indicating a phytoextraction-oriented strategy, although this was accompanied by signs of metal-induced physiological stress. Hydrological conditions played a species-specific role in remediation efficiency. Cultivation of *H. umbellata* in constructed wetlands enhanced its rhizofiltration performance, whereas *S. auriculata* exhibited relatively stable remediation efficiency across both deep water and constructed wetland systems. Seasonal variation further modulated phytoremediation performance, with spring conditions generally promoting higher metal uptake and accumulation, highlighting the importance of environmental controls on remediation efficiency.

Overall, the complementary remediation behaviors of these species suggest that their combined or site-specific application could enhance the effectiveness of macrophyte-based treatment systems. Further research is needed to optimize operational conditions, including hydrology, seasonal management, and metal loading rates, as well as to develop environmentally safe strategies for the handling, disposal, or valorization of metal-enriched biomass. The cultivation of *H. umbellata* and *S. auriculata* can therefore contribute not only to the remediation of metal-contaminated and eutrophic waters but also to the sustained operation and optimization of constructed wetland systems as nature-based solutions for wastewater treatment.

**Supplementary Information** The online version contains supplementary material available at <https://doi.org/10.1007/s41742-026-01062-z>.

**Acknowledgements** This work was conducted using specimens from two hydrophytic species that were introduced to the Alexandria University Botanic Garden (AUBG) at the Faculty of Science, Alexandria University (ALEX). The research team extends sincere gratitude to Prof. S. Z. Heneidy, Director of the Botanic Garden, for his invaluable support and for facilitating the implementation of this study.

**Author Contributions** M.W.H. and S.Z.H.: Conceptualization. S.Z.H. and A.M.F.: Data Curation. A.M.E. and M.W.H.: Methodology, experimental design, sample analysis, Software, Formal Analysis, Visualization, Writing original draft, review and editing. S.Z.H., M.W.H. and A.M.F.: Investigation, Data Curation, supervision, project administration, funding acquisition, review and editing.

**Funding** Open access funding provided by The Science, Technology & Innovation Funding Authority (STDF) in cooperation with The Egyptian Knowledge Bank (EKB). “The author declare that no funds, grants, or other support were received during the preparation of this manuscript.”

## Declarations

**Conflict of interest** “The authors have no relevant financial or non-financial interests to disclose.”

**Open Access** This article is licensed under a Creative Commons Attribution 4.0 International License, which permits use, sharing, adaptation, distribution and reproduction in any medium or format, as long as you give appropriate credit to the original author(s) and the source, provide a link to the Creative Commons licence, and indicate if changes were made. The images or other third party material in this article are included in the article’s Creative Commons licence, unless indicated otherwise in a credit line to the material. If material is not included in the article’s Creative Commons licence and your intended use is not permitted by statutory regulation or exceeds the permitted use, you will need to obtain permission directly from the copyright holder. To view a copy of this licence, visit <http://creativecommons.org/licenses/by/4.0/>.

## References

- Abhilash PC, Jamil S, Singh N, Elsevier (2009) Transgenic plants for enhanced biodegradation and phytoremediation of organic xenobiotics. *Biotechnol Adv* 27(4):474–488. <https://doi.org/10.1016/j.biotechadv.2009.04.002>
- Abhilash PC, Pandey VC, Srivastava P, Rakesh PS, Chandran S, Singh N, Thomas AP (2009) Phytofiltration of cadmium from water by *Limncharis flava* (L.) Buchenau grown in free-floating culture system. *J Hazard Mater* 170(2–3):791–797. <https://doi.org/10.1016/j.jhazmat.2009.05.035>
- Abida B, HariKrishna S (2010) Bioaccumulation of trace metals by aquatic plants. *Int J ChemTech Res* 2(1):250–254
- Al-Hamdani SH, Blair SL (2004) Influence of copper on selected physiological responses in *Salvinia minima* and its potential use in copper remediation. *Am Fern J* 94(1):47–56. [https://doi.org/10.1640/0002-8444\(2004\)094\[0047:IOCOSP\]2.0.CO;2](https://doi.org/10.1640/0002-8444(2004)094[0047:IOCOSP]2.0.CO;2)
- Alaoui-Sossé B, Genet P, Vinit-Dunand F, Toussaint ML, Epron D, Badot PM (2004) Effect of copper on growth in cucumber plants

- (*Cucumis sativus*) and its relationships with carbohydrate accumulation and changes in ion contents. *Plant Sci* 166(5):1213–1218. <https://doi.org/10.1016/j.plantsci.2003.12.032>
- Ali Z, Mohammad A, Riaz Y, Quraishi UM, Malik RN (2018) Treatment efficiency of a hybrid constructed wetland system for municipal wastewater and its suitability for crop irrigation. *Int J Phytoremediation* 20(11):1152–1161. <https://doi.org/10.1080/15226514.2018.1460311>
- Ali S, Abbas Z, Rizwan M, Zaheer IE, Yavas I, Ünay A, Abdel-Daim MM, Bin-Jumah M, Hasanuzzaman M, Kalderis D (2020) Application of floating aquatic plants in phytoremediation of heavy metals polluted water: a review. *Sustain (Switzerland)* 12(5):1927. <https://doi.org/10.3390/su12051927>
- Alkorta I, Hernández-Allica J, Becerril JM, Amezaga I, Albizu I, Garbisu C (2004) Recent findings on the phytoremediation of soils contaminated with environmentally toxic heavy metals and metalloids such as zinc, cadmium, lead, and arsenic. *Rev Environ Sci Biotechnol* 3(1):71–90. <https://doi.org/10.1023/B:RESB.0000040059.70899.3d>
- Alloway B (2013) Heavy metals in soils: trace metals and metalloids in soils and their bioavailability. In *Springer, Dordrecht* (3rd ed., Vol. 50, Issue 07). <https://doi.org/10.5860/choice.50-3862>
- Bader N, Alsharif E, Nassib M, Alshelmani N, Alalem A (2019) Asian journal of green chemistry original research article phytoremediation potential of *Suaeda vera* for some heavy metals in roadside soil in Benghazi, Libya. *Asian Journal of Green Chemistry* 3:82–90. <https://doi.org/10.22034/ajgc.2018.67060>
- Bizzo ALT, Intorne AC, Gomes PH, Suzuki MS, Esteves B (2014) Short-term physiological responses to copper stress in *Salvinia auriculata* Aubl. *Acta Limnol Bras* 26(3):268–277. <https://doi.org/10.1590/s2179-975x2014000300006>
- Bokhari SH, Nawaz G, Azizullah A, Mahmood-Ul-Hassan M, Ali Z (2022) Heavy metals phytofiltration potential of *Hydrocotyle umbellata* from Nullah Lai wastewater and its environmental risk. *Int J Phytoremediation*. <https://doi.org/10.1080/15226514.2022.2035673>
- Bonanno G (2013) Comparative performance of trace element bioaccumulation and biomonitoring in the plant species *Typha domingensis*, *Phragmites australis* and *Arundo donax*. *Ecotoxicol Environ Saf* 97:124–130. <https://doi.org/10.1016/j.ecoenv.2013.07.017>
- 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(3):639–645. <https://doi.org/10.1016/j.ecolind.2009.11.002>
- Bose S, Vedamati J, Rai V, Ramanathan AL (2008) Metal uptake and transport by *Typaha angustata* L. grown on metal contaminated waste amended soil: an implication of phytoremediation. *Geoderma* 145(1–2):136–142. <https://doi.org/10.1016/j.geoderma.2008.03.009>
- Chandra P, Sinha S, Rai UN (1997) Bioremediation of chromium from water and soil by vascular aquatic plants. *ACS Symp Ser* 664:274–282. <https://doi.org/10.1021/bk-1997-0664.ch020>
- Cheng S (2003) Heavy metals in plants and phytoremediation. *Environ Sci Pollut Res* 10(5):335–340. <https://doi.org/10.1065/espr2002.11.141.3>
- Dhir B, Srivastava S (2011) Heavy metal removal from a multi-metal solution and wastewater by *Salvinia natans*. *Ecol Eng* 37(6):893–896. <https://doi.org/10.1016/j.ecoleng.2011.01.007>
- Dhir B, Sharmila P, Pardha Saradhi P, Sharma S, Kumar R, Mehta D (2011) Heavy metal induced physiological alterations in *Salvinia natans*. *Ecotoxicol Environ Saf* 74(6):1678–1684. <https://doi.org/10.1016/j.ecoenv.2011.05.009>
- Dierberg FE, Debusk TA, Goulet jr N (1987) Removal of copper and lead using a thin-film technique. In: Reddy KR, Smith, WH (eds) *Aquatic plants for water treatment and resource recovery*. Magnolia Publishing, New York, 497–504. <https://agris.fao.org/agris-search/search.do?recordID=US9037307>
- Du X, Gao L, Xun Y, Feng L (2020) Comparison of different sequential extraction procedures to identify and estimate bioavailability of arsenic fractions in soil. *J Soils Sediments* 20(10):3656–3668. <https://doi.org/10.1007/s11368-020-02694-0>
- Duman F, Cicek M, Sezen G (2007) Seasonal changes of metal accumulation and distribution in common club rush (*Schoenoplectus lacustris*) and common reed (*Phragmites australis*). *Ecotoxicology* 16(6):457–463. <https://doi.org/10.1007/s10646-007-0150-4>
- Eid EM, Shaltout KH (2014) Monthly variations of trace elements accumulation and distribution in above- and below-ground biomass of *Phragmites australis* (Cav.) Trin. ex Steudel in Lake Burullus (Egypt): a biomonitoring application. *Ecol Eng* 73:17–25. <https://doi.org/10.1016/j.ecoleng.2014.09.006>
- El-Liethy MA, Dakhil MA, El-Keblawy A, Abdelaal M, Halmy MWA, Elgarhy AH, Kamika I, El-Sherbeny GA, Mwaheb MA (2022) Temporal phytoremediation potential for heavy metals and bacterial abundance in drainage water. *Sci Rep* 12(1):8223
- Favas PJC, Pratas J, Prasad MNV (2012) Accumulation of arsenic by aquatic plants in large-scale field conditions: opportunities for phytoremediation and bioindication. *Sci Total Environ* 433:390–397. <https://doi.org/10.1016/j.scitotenv.2012.06.091>
- Fritioff Å, Greger M (2006) Uptake and distribution of Zn, Cu, Cd, and Pb in an aquatic plant *Potamogeton natans*. *Chemosphere* 63(2):220–227. <https://doi.org/10.1016/j.chemosphere.2005.08.018>
- Galal TM, Shehata HS (2015) Impact of nutrients and heavy metals capture by weeds on the growth and production of rice (*Oryza sativa* L.) irrigated with different water sources. *Ecol Ind* 54:108–115. <https://doi.org/10.1016/j.ecolind.2015.02.024>
- Galal TM, Eid EM, Dakhil MA, Hassan LM (2018) Bioaccumulation and rhizofiltration potential of *Pistia stratiotes* L. for mitigating water pollution in the Egyptian wetlands. *Int J Phytoremediation* 20(5):440–447. <https://doi.org/10.1080/15226514.2017.1365343>
- Guida M, Inglese M, Meriç S (2008) A multi-battery toxicity investigation on fungicides. *Desalination* 226(1–3):262–270. <https://doi.org/10.1016/j.DESAL.2007.04.096>
- Heneidy SZ, Halmy MWA, Fakhry AM, El-Makawy AM (2019) The status and potential distribution of *Hydrocotyle umbellata* L. and *Salvinia auriculata* Aubl. under climate change scenarios. *Aquat Ecol* 53(4):509–528
- Jaishankar M, Tseten T, Anbalagan N, Mathew BB, Beeregowda KN (2014) Toxicity, mechanism and health effects of some heavy metals. *Interdiscip Toxicol* 7(2):60
- Kabata-Pendias A (2011) Trace elements in soils and plants, 4th ed. CRC Press. <https://doi.org/10.1201/b10158>
- Khan I, Aftab M, Shakir SU, Ali M, Qayyum S, Rehman MU, Haleem KS, Touseef I (2019) Mycoremediation of heavy metal (Cd and Cr)–polluted soil through indigenous metallotolerant fungal isolates. *Environ Monit Assess* 191(9):1–11. <https://doi.org/10.1007/S10661-019-7769-5/FIGURES/4>
- Khilji S, Bareen F (2008) Rhizofiltration of heavy metals from the tannery sludge by the anchored hydrophyte, *Hydrocotyle umbellata* L. *Afr J Biotechnol* 7(20):3714–3720. <https://doi.org/10.5897/AJB08.608>
- Masarovičová E, Kráľová K, Kummerová M (2010) Principles of classification of medicinal plants as hyperaccumulators or excluders. *Acta Physiol Plant* 32(5):823–829. <https://doi.org/10.1007/s11738-010-0474-1>
- McBride MB (2001) Cupric ion activity in peat soil as a toxicity indicator for maize. *J Environ Qual* 30(1):78–84. <https://doi.org/10.2134/JEQ2001.30178X>
- Miretzky P, Saralegui A, Cirelli AF (2004) Aquatic macrophytes potential for the simultaneous removal of heavy metals (Buenos

- Aires, Argentina). *Chemosphere* 57(8):997–1005. <https://doi.org/10.1016/j.chemosphere.2004.07.024>
- Mirza N, Mahmood Q, Maroof Shah M, Pervez A, Sultan S, Hindawi Limited (2014) Plants as useful vectors to reduce environmental toxic arsenic content. *Sci World J*. <https://doi.org/10.1155/2014/921581>
- Mohammed SJ, Fadhil AA, Ali HH (2023) Seasonal dynamics of heavy metal uptake in some aquatic plants of the Tigris River. *Int J Aquat Biol* 11(6):577–582
- Muliyadi M, Purwanto P, Sumiyati S, Mussadun M (2023) Bioaccumulation of heavy metals using aquatic plants in wastewater. *Int J Public Health Sci* 12(2):690–698
- Muramoto S, Oki Y (1983) Removal of some heavy metals from polluted water by water hyacinth (*Eichhornia crassipes*). *Bull Environ Contam Toxicol* 30:170–177
- Musielińska R, Kowol J, Kwapuliński J, Rochel R (2016) Antagonism between lead and zinc ions in plants. *Arch Environ Prot* 42(2):78–91. <https://doi.org/10.1515/aep-2016-0022>
- Mustafa HM, Hayder G (2021) Recent studies on applications of aquatic weed plants in phytoremediation of wastewater: a review article. *Ain Shams Eng J* 12(1):355–365. <https://doi.org/10.1016/J.ASEJ.2020.05.009>
- Nan Z, Li J, Zhang J, Cheng G (2002) Cadmium and zinc interactions and their transfer in soil-crop system under actual field conditions. *Sci Total Environ* 285(1–3):187–195. [https://doi.org/10.1016/S0048-9697\(01\)00919-6](https://doi.org/10.1016/S0048-9697(01)00919-6)
- Nichols P, Couch J, Al-Hamdani S (2000) Selected physiological responses of *Salvinia minima* to different chromium concentrations. *Aquatic Botany* 68 (2000) 313–319
- Nsanganwimana F, Marchand L, Douay F, Mench M, Taylor & Francis Group (2014) *Arundo donax* L., a candidate for phytomanaging water and soils contaminated by trace elements and producing plant-based feedstock. A review. *Int J Phytoremediation* 16(10):982–1017. <https://doi.org/10.1080/15226514.2013.810580>
- Oladoye PO, Olowe OM, Asemoloye MD (2022) Phytoremediation technology and food security impacts of heavy metal contaminated soils: a review of literature. *Chemosphere* 288(P2):132555. <https://doi.org/10.1016/j.chemosphere.2021.132555>
- Panyakhan S, Kruatrachue M, Pokethitiyook P, Soonthornsarathoon V, Upatham S (2006) Toxicity and accumulation of cadmium and zinc in *Hydrocotyle umbellata*. *ScienceAsia* 32(3):323–328. <https://doi.org/10.2306/scienceasia.1513-1874.2006.32.323>
- Prasad MNV (2003) Phytoremediation of metal-polluted ecosystems: hype for commercialization. *Russ J Plant Physiol* 50(5):686–701. <https://doi.org/10.1023/A:1025604627496>
- Rakshit A, Pal S, Parihar M, Singh HB (2017) Bioremediation of soils contaminated with Ni and Cd: An overview. *Adapt Soil Management: Theory Practices* 339–357. [https://doi.org/10.1007/978-981-10-3638-5\\_17](https://doi.org/10.1007/978-981-10-3638-5_17)
- Rock S, Pivetz B, Madalinski K, Adams N, Wilson T (2000) Introduction to Phytoremediation. U.S. Environmental. In US Environmental Protection Agency, Washington, DC. EPA/600/R-99/107 (NTIS PB2000-106690)
- Saeed SH, Shah GM, Mahmood Q, Shaheen S, Zeb BS, Nawazish S, Abd\_Allah EF (2024a) Phytoremediation ability and selected genetic transcription in *Hydrocotyle umbellata*-under cadmium stress. *Int J Phytoremediation* 26(7):1144–1153
- Saeed SH, Gazder U, Gillani GMS, Mahmood Q (2024b) Unveiling Cu extractability and statistical insights: *Hydrocotyle umbellata* L. as a catalyst for environmental remediation. *Environ Sci Pollut Res* 31(57):65778–65789
- Saha R, Nandi R, Saha B (2011) Sources and toxicity of hexavalent chromium. *J Coord Chem* 64(10):1782–1806
- Sanyahumbi D, Duncan JR, Zhao M, Van Hille R (1998) Removal of lead from solution by the non-viable biomass of the water fern *Azolla filiculoides*. *Biotechnol Lett* 20(8):745–747. <https://doi.org/10.1023/A:1005386703592>
- Sarwar N, Imran M, Shaheen MR, Ishaque W, Kamran MA, Matloob A, Rehman A, Hussain S (2017) Phytoremediation strategies for soils contaminated with heavy metals: modifications and future perspectives. *Chemosphere* 171:710–721. <https://doi.org/10.1016/j.chemosphere.2016.12.116>
- Sawidis T, Chettri MK, Zachariadis GA, Stratis JA (1995) Heavy metals in aquatic plants and sediments from water systems in Macedonia, Greece. *Ecotoxicol Environ Saf* 32(1):73–80. <https://doi.org/10.1006/eesa.1995.1087>
- Sharma P, Singh SP, Pandey S, Thanki A, Singh NK (2020) Role of potential native weeds and grasses for phytoremediation of endocrine-disrupting pollutants discharged from pulp paper industry waste. *Bioremediat Pollutants* 17–37. <https://doi.org/10.1016/b978-0-12-819025-8.00002-8>
- Song L, Marsh TL, Voice TC, Long DT (2011) Loss of seasonal variability in a lake resulting from copper sulfate algacide treatment. *Phys Chem Earth* 36(9–11):430–435. <https://doi.org/10.1016/j.pce.2010.04.016>
- Stefanakis AI (2018) Constructed wetlands for industrial wastewater treatment. Wiley, Chichester, p 578
- Tangahu BV, Sheikh Abdullah SR, Basri H, Idris M, Anuar N, Mukhlisin M (2011) A review on heavy metals (As, Pb, and Hg) uptake by plants through phytoremediation. *Int J Chem Eng*. <https://doi.org/10.1155/2011/939161>
- Tchounwou PB, Clement G, Yedjou AK, Patlolla, Dwayne J (2012) Sutton. Heavy metal toxicity and the environment. *Molecular, clinical and environmental toxicology: volume 3: environmental toxicology* : 133–164
- Tóth G, Hermann T, Da Silva MR, Montanarella L (2016) Heavy metals in agricultural soils of the European Union with implications for food safety. *Environ Int* 88:299–309. <https://doi.org/10.1016/J.ENVINT.2015.12.017>
- Uchimiya M, Bannon D, Nakanishi H, McBride MB, Williams MA, Yoshihara T (2020) Chemical speciation, plant uptake, and toxicity of heavy metals in agricultural soils. *J Agric Food Chem* 68(46):12856–12869. <https://doi.org/10.1021/acs.jafc.0c00183>
- US EPA (1994) Method 200.7: Determination of Metals and Trace Elements in Water and Wastes by Inductively Coupled Plasma-Atomic Emission Spectrometry, Revision 4.4. Cincinnati, OH
- Varol M, Sünbül MR (2018) Biomonitoring of trace metals in the Keban Dam Reservoir (Turkey) using mussels (*Unio elongatulus eucirrus*) and crayfish (*Astacus leptodactylus*). *Biol Trace Elem Res* 185(1):216–224. <https://doi.org/10.1007/S12011-017-1238-1/TABLES/7>
- Veselý T, Tlustoš P, Száková J (2011) The use of water lettuce (*Pistia stratiotes* L.) for rhizofiltration of a highly polluted solution by cadmium and lead. *Int J Phytoremediation* 13(9):859–872
- Vymazal J, Březinová T (2015) Heavy metals in plants in constructed and natural wetlands: concentration, accumulation and seasonality. *Water Sci Technol* 71(2):268–276. <https://doi.org/10.2166/wst.2014.507>
- Wolff G, Pereira G, Castro E, Louzada J, Coelho F (2012) The use of *Salvinia auriculata* as a bioindicator in aquatic ecosystems: biomass and structure dependent on the cadmium concentration. *Braz J Biol* 72(1):71–77. <https://doi.org/10.1590/s1519-69842012000100009>
- Yanqun Z, Yuan L, Jianjun C, Haiyan C, Li Q, Schwartz C (2005) Hyperaccumulation of Pb, Zn and Cd in herbaceous grown on lead-zinc mining area in Yunnan, China. *Environ Int* 31(5):755–762. <https://doi.org/10.1016/j.envint.2005.02.004>
- Yongpisanphop J, Kruatrachue M, Pokethitiyook P (2005) Toxicity and accumulation of lead and chromium in *Hydrocotyle umbellata*. *J Environ Biol* 26(1):79–89

- Zapana JSP, Arán DS, Bocardo EF, Harguinteguy CA (2020) Treatment of tannery wastewater in a pilot scale hybrid constructed wetland system in Arequipa, Peru. *Int J Environ Sci Technol* 17(11):4419–4430. <https://doi.org/10.1007/s13762-020-02797-8>
- Zayed A, Gowthaman S, Terry N (1998) Phytoaccumulation of trace elements by wetland plants: I. Duckweed. *J Environ Qual* 27(3):715–721. <https://doi.org/10.2134/jeq1998.00472425002700030032x>
- Zhou M, Engelmann T, Lutts S (2019) Salinity modifies heavy metals and arsenic absorption by the halophyte plant species *Kosteletzkya pentacarpos* and pollutant leaching from a polycontaminated substrate. *Ecotoxicol Environ Saf* <https://doi.org/10.1016/j.ecoen.2019.109460>
- Zhu YL, Zayed AM, Qian J, Souza M, Terry N (1999) Phytoaccumulation of trace elements by wetland plants: II. Water hyacinth. *J Environ Qual* 28(1):339–344. <https://doi.org/10.2134/jeq1999.00472425002800010042x>
- Zwolsman JJG, Berger GW, Van Eck GTM (1993) Sediment accumulation rates, historical input, postdepositional mobility and retention of major elements and trace metals in salt marsh sediments of the Scheldt estuary, SW Netherlands. *Mar Chem* 44(1):73–94. [https://doi.org/10.1016/0304-4203\(93\)90007-B](https://doi.org/10.1016/0304-4203(93)90007-B)

**Publisher's Note** Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.