



Metal analysis of submerged aquatic vegetation in the lower St. Johns River, Florida

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Received: 22 November 2021 / Accepted: 28 May 2022
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Abstract The St. Johns River, Florida's longest river, is threatened by a variety of factors, including anthropogenic disturbances and global climate changes. Metal pollution in the Lower St. Johns River (LSJR) in Northeast Florida has been well documented. Submerged aquatic vegetation (SAV) are ecologically important plant species, known to accumulate metals and other pollutants from their surrounding environments, and have been declining in the LSJR in recent years. In this study, eight SAV taxa (*Vallisneria americana*, *Ruppia maritima*, *Chara* sp., *Najas guadalupensis*, *Eleocharis* sp., *Hydrilla verticillata*, *Zannichellia palustris*, and *Sagittaria subulata*) were collected along the LSJR from Central Florida northward to Jacksonville and the Atlantic Ocean. More than 200 SAV samples were identified, digested, and measured for a suite of metals. Mean (\pm standard deviation) metal concentrations in $\mu\text{g/g}$ dry mass in all taxa sampled were 1.76 (± 2.75) cadmium, 35.8 (± 52.24) copper, 4.16 (± 5.74) lead, 119 (± 229) nickel, 0.98 (± 1.40) silver, and 203 (± 376) zinc. SAV metal concentrations varied across species

and collection sites. In general, *Z. palustris* and *Eleocharis* sp. had the highest metal concentrations. Furthermore, a comparison of SAV metal concentrations to sediment quality guidelines for the LSJR indicated that toxicity may occur to biota exposed to these SAV. This study provides new data about SAV health in the LSJR and may help in the development of new management strategies.

Keywords Copper · Cadmium · Lead · Nickel · Zinc · Silver · SAV

Introduction

The St. Johns River is the longest river in Florida and a valued watershed for its recreational, economic, and ecological uses (DeMort, 1990; Pinto et al., 2019). Metal pollution in the Lower St. Johns River (LSJR) has increased over the years due to atmospheric emissions, direct surface discharge, and stormwater runoff from agricultural and urban areas (Bielmyer et al., 2012; Bielmyer-Fraser et al., 2020; Moore, 2009; Pinto et al., 2019). Increased metal loading, as well as flooding from storm events, has increased concerns about the health of the river and resident biota (Bielmyer-Fraser et al., 2020; Moore, 2009; Pinto et al., 2019; Tchounwou et al., 2012).

Submerged aquatic vegetation (SAV) are ecologically important plant species that are found in the LSJR (Goldberg & Trent, 2020; Goldberg et al., 2018;

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Pinto et al., 2019). SAV provide a crucial food source for a variety of waterfowl and marine mammals, including the West Indian Manatee (*Trichechus manatus latirostris*), and serve as essential habitat for many aquatic organisms (Bengtson, 1981; Li et al., 2007; Pinto et al., 2019; Poirrier et al., 2017; Scheffer & Van Nes, 2007). Furthermore, SAV accumulate contaminants, contribute to carbon and nutrient cycling, and help prevent erosion and reduce turbidity by trapping sediment (Barnett & Schneider, 1974; Bielmyer-Fraser et al., 2022; Camp et al., 2012; Hauxwell et al., 2004; Laughlin, 1982; Li et al., 2007; Poirrier et al., 2017; Scheffer & Van Nes, 2007). SAV have markedly declined worldwide over the past 50 years, and similar trends have been observed in the LSJR (Goldberg & Trent, 2020; Pinto et al., 2019; Pyati et al., 2012).

SAV and other aquatic macrophytes have been shown to significantly accumulate metals, and many SAV species have evolved different mechanisms to survive against metal toxicity including efficient accumulation, detoxification, and rapid translocation (Bai et al., 2018; Harguinteguy et al., 2014; Keskinan et al., 2004; Peng et al., 2008; Xue et al., 2010). However, elevated metal concentrations which exceed detoxification thresholds may exert toxicity in plants (Bielmyer-Fraser et al., 2022). Xing et al. (2013) reported the following toxic levels in plants: 20–100 µg/g copper, 30–300 µg/g lead, 10–100 µg/g nickel, and 100–400 µg/g zinc, although tolerance is largely species specific. A range of toxic effects from metal exposure have been reported in aquatic plants including decreased photosynthesis and reduced growth (Baker & Walker, 1989; Jarvis & Bielmyer-Fraser, 2015; Prasad & Strzałka, 1999; Prasad et al., 2001; Wang et al., 2008, 2013), ultimately reducing SAV coverage in many aquatic ecosystems. Additionally, given the importance of SAV as a food source for many organisms, trophic transfer of metals is also a concern for consumers (Hazrat et al., 2019).

The objectives of this study were to measure the concentrations of cadmium, copper, nickel, lead, silver, and zinc in eight SAV species collected along the LSJR. We hypothesized that the metal concentrations would vary among species and collection sites. Varying degrees of land development and different types of land uses (e.g., industries, wastewater treatment facilities, agriculture) have resulted in differences in the amount of contamination along the LSJR (Pinto et al., 2019). Assessing metal concentrations in SAV

species in the LSJR can provide insight into the metal tolerance among species, site-specific metal pollution, and the health of LSJR ecosystem.

Methods

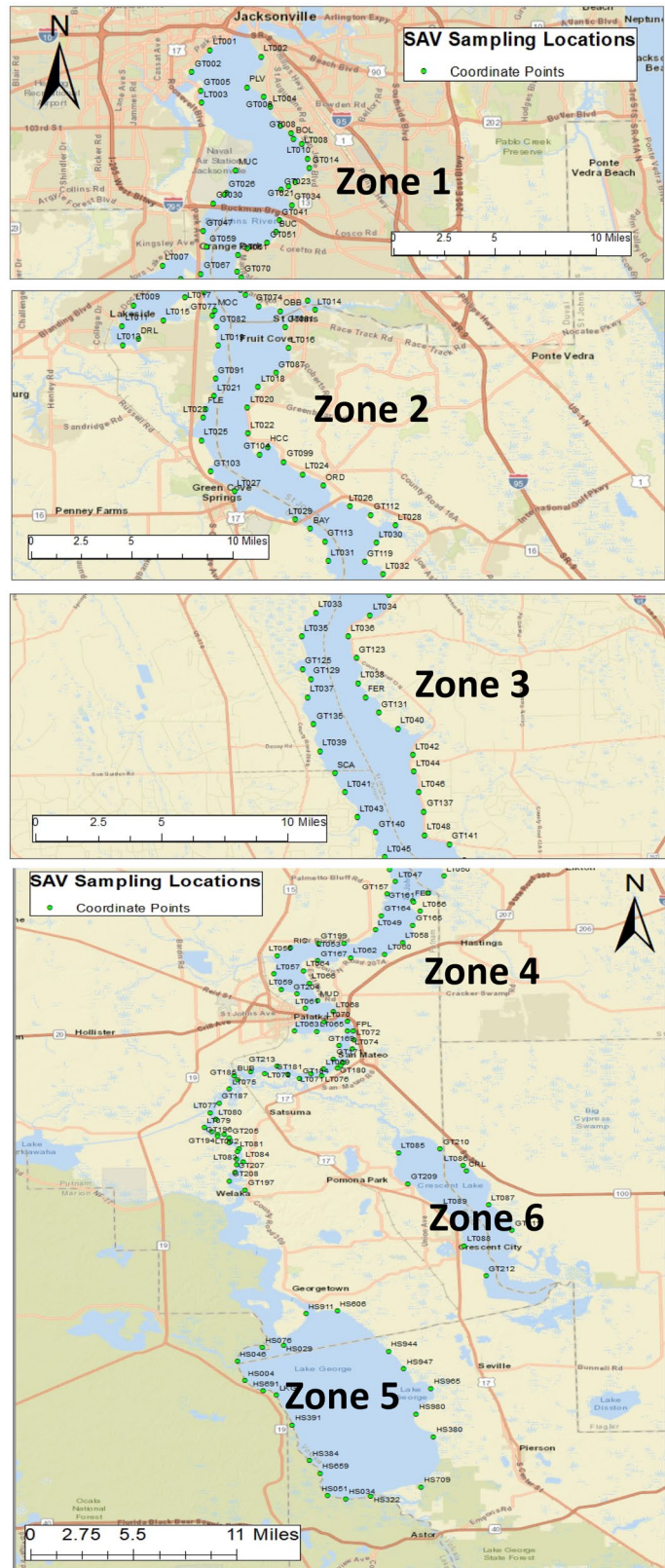
Field collection

Shoots of eight taxa of submerged aquatic vegetation including wild celery (*Vallisneria americana*), widgeon grass (*Ruppia maritima*), Southern naiad (*Najas guadalupensis*), water thyme (*Hydrilla verticillata*), horned pondweed (*Zannichellia palustris*), awl-leaf arrowhead (*Sagittaria subulata*), spike rushes (*Eleocharis* sp.), and *Chara* sp. were collected by hand from more than 200 different locations along the LSJR from May to July 2019 (Fig. 1). The LSJR was separated into six zones to compare metal concentrations in SAV across collection sites (Fig. 1). Samples were collected from the shoreline at 1-cm depth, up to 250 m from shore, and up to 1 m in water depth. Shallow water species like *Sagittaria subulata* and *Eleocharis* sp. were typically collected closer to the shoreline in shallower waters, whereas the other species were collected anywhere from the shoreline to up to 250 m from shore. Samples were placed in plastic bags and immediately frozen. Subsurface water samples (in the same location plants were obtained) were collected in polypropylene centrifuge tubes, filtered (0.45-µm filter), and acidified with trace metal-grade nitric acid.

Sample processing

In the laboratory, SAV samples were identified, and shoots of each species were then placed in a pre-weighed aluminum weigh boat and massed on an analytical balance. Samples were dried for at least 24 h at 60 °C in an oven. Samples were cooled to room temperature and then dry masses were measured. Dried SAV samples were submerged in 50% trace metal-grade nitric acid in 15-mL polypropylene tubes for digestion. All the samples were placed in a water bath (Fisher Scientific Isotemp GPD 20) at 70 °C for 18 h in total to induce complete digestion. Nine replicates of the certified reference material for trace elements

Fig. 1 GIS map of the Lower St. Johns River, FL separated into six zones. Submerged aquatic vegetation (SAV) collection sites are indicated by green circles



DORM-4 (Fish protein, National Research Council of Canada) were prepared following the same procedure.

Metal analysis

After digestion, plant tissue samples were diluted with ultra-pure 18 mΩ Milli-Q[®] water tenfold for nickel, copper, and zinc and fivefold for cadmium, lead, and silver determination. Tissue samples, certified reference materials, and spiked samples were measured for metals using atomic absorption spectrophotometry (AAS) with graphite furnace or flame detection (AAAnalyst 800, Perkin Elmer, Norwalk, CT). Metal concentrations in plant tissue are reported in µg/g dw. Water samples were measured for metals using AAS with graphite furnace detection. Certified standards were used for calibration, and re-calibration was performed every 30–40 samples. Each standard and sample were analyzed in triplicate for each element and blanks and QA/QC samples were measured throughout each analysis. Detection limits were 0.88 µg/L lead, 0.80 µg/L nickel, 0.59 µg/L copper, 0.03 µg/L cadmium, 2.06 µg/L zinc, and 0.26 µg/L silver. Mean extraction efficiencies for DORM-4 were 114.5, 109.5, 102.9, 103.0, 107.9, and 101.0% for copper, zinc, nickel, cadmium, silver, and lead, respectively.

Bioconcentration factors

Bioconcentration factors (BCFs) were calculated for each metal and each taxon within zone 2. BCFs are presented as the ratio of micrograms of metal per gram of SAV to the micrograms of metal per liter of water (environmental metal concentration; Hemond & Fechner, 2020).

Statistics

Data were analyzed for normality and equality of variance using Shapiro–Wilk and Levene tests, respectively. Because the data did not have normal distributions, nonparametric Kruskal–Wallis One Way Analysis of Variance on Ranks ($\alpha \leq 0.05$) was used to compare the concentrations of each metal among taxa for all the zones and for only zone 2, among all zones for *V. americana*, between zone 2 and 3 for *N. guadalupensis*, and between zone 1 and 2 for *Ruppia maritima*. Dunn pairwise comparisons or Mann–Whitney Rank Sum Test was conducted if the main factor was significant ($\alpha \leq 0.05$).

Data sources and toxicity measures

Sediment metal concentrations in the LSJR were obtained from WIN (from 2019) (managed by the U.S. Environmental Protection Agency and Florida Department of Environmental Protection) and were designated with a matrix of “surface water sediment,” or “sediment.” Metal concentrations in SAV were compared to the metal concentrations in LSJR sediment during the year of SAV collection (2019) and the sediment quality guidelines to assess contamination levels. The toxicity measures used in this study include the Threshold Effect Level (TEL) for marine environments and Threshold Effect Concentration (TEC) for freshwater, which is the lowest concentration of a contaminant that affects some sensitive species. Other toxicity measures used were the Probable Effect Level (PEL) for marine environments and Probable Effect Concentration (PEC) for freshwater, which is the concentration that aquatic species are likely to be affected. Metal concentrations in the water samples were compared to Florida ambient water quality criteria for a class III waterbody (DEP, 2020).

Results

Mean (\pm standard deviation) metal concentrations in all taxa sampled were 1.76 (± 2.75) µg/g cadmium, 35.8 (± 52.24) µg/g copper, 4.16 (± 5.74) µg/g lead, 119 (± 229) µg/g nickel, 0.98 (± 1.40) µg/g silver, and 203 (± 376) µg/g zinc. Metal concentrations in the shoots of SAV varied across species (Fig. 2). In general, except for lead, *Chara* sp. had the lowest shoot tissue metal concentrations (Fig. 2). *H. verticillata* and *S. subulata* also had relatively lower metal concentrations, although the sample size for these two species was small. The highest metal concentrations were observed in *Z. palustris* (cadmium, copper, and silver) and *Eleocharis* sp. (nickel, lead, and zinc) (Fig. 2). The patterns of accumulation for the essential metals, copper, nickel, and zinc were similar in each taxon, relative to each other (Fig. 2). *V. americana* and *R. maritima* had similar tissue metal concentrations for every metal measured (Fig. 2).

All taxa were not found in every zone, which limited statistical comparisons. Additionally, no SAV were found in zone 6. Zone 2 had the most robust data set for all taxa. To minimize differences in tissue metal concentrations due to collection site, tissue

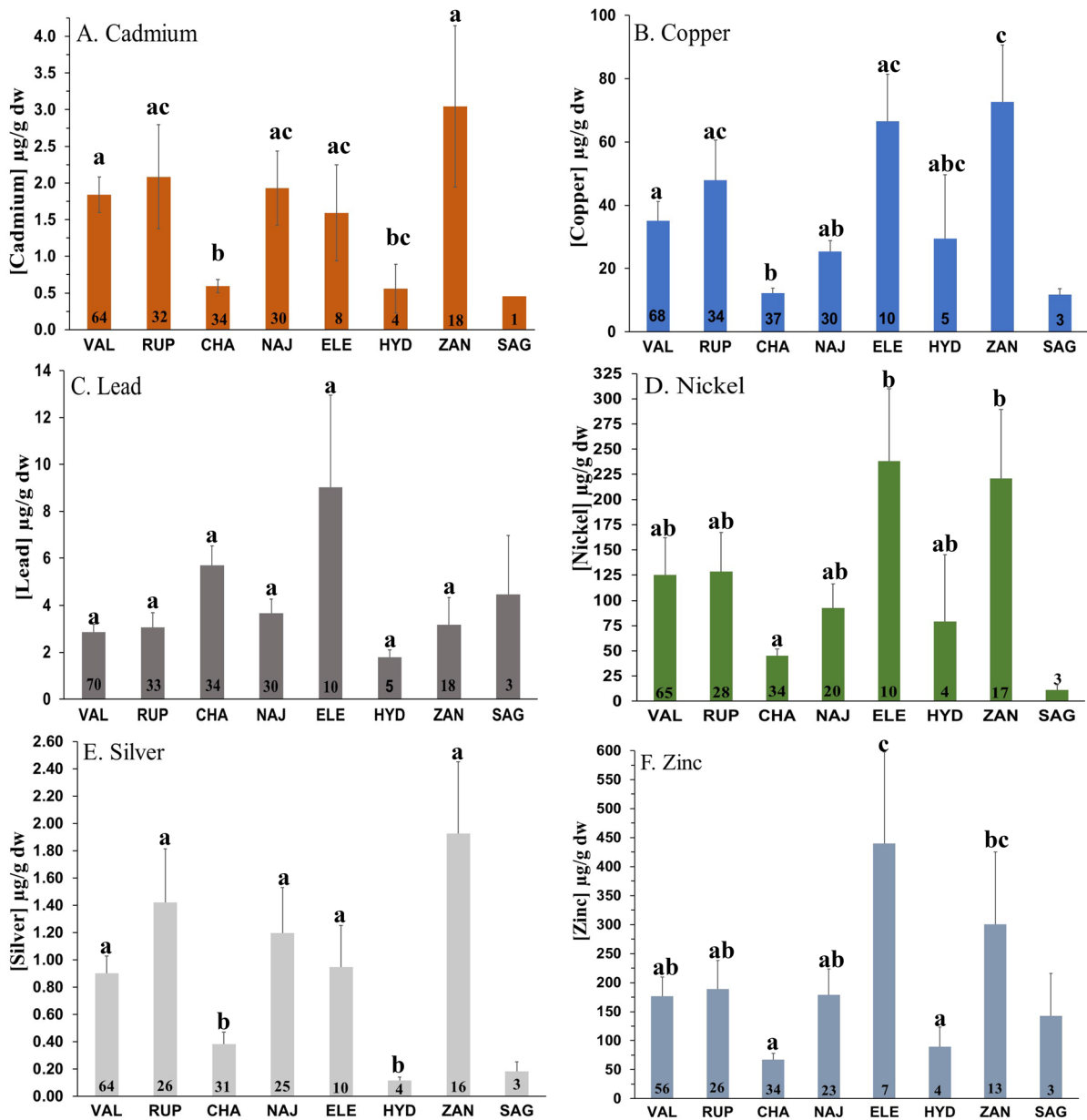


Fig. 2 Cadmium (A), copper (B), lead (C), nickel (D), silver (E), and zinc (F) concentrations in eight SAV taxa (*Vallisneria americana*, VAL; *Ruppia maritima*, RUP; *Chara* sp., CHA; *Najas guadalupensis*, NAJ; *Eleocharis* sp., ELE; *Hydrilla verticillata*, HYD; *Zannichellia palustris*, ZAN; *Sagittaria subulata*, SAG) collected in all zones of the Lower St. Johns

River, FL in 2019 (Fig. 1). Numbers within each column represent the sample size for each taxon. Different letters indicate a significant difference between taxa for a particular metal ($\alpha \leq 0.05$). The sample size for *S. subulata* was too low to detect statistical differences

metal concentrations were also compared across taxa within zone 2 only. In zone 2, zinc, cadmium, copper, and silver concentrations differed among the six taxa tested (zinc: $H=14.39$, $p=0.0133$; cadmium:

$H=23.49$, $p=0.0003$; copper: $H=23.53$, $p=0.0003$; silver: $H=17.56$, $p=0.0035$). *Chara* sp. zinc concentrations were lower than in *Eleocharis* sp., *Z. palustris*, *N. guadalupensis*, and *V. americana* (Dunn

pairwise $p < 0.0227$); cadmium concentrations were lower than in *N. guadalupensis*, *Z. palustris*, *V. americana*, and *R. maritima* (Dunn pairwise $p < 0.0248$); copper concentrations were lower than in *Eleocharis* sp., *Z. palustris*, *V. americana*, and *R. maritima* (Dunn pairwise $p < 0.0367$); and silver concentrations were lower than in *Eleocharis* sp., *Z. palustris*, *N. guadalupensis*, *V. americana*, and *R. maritima* (Dunn pairwise $p < 0.0378$). In addition, *Z. palustris* had significantly

greater copper concentrations than measured in *V. americana* and *N. guadalupensis* (Dunn pairwise $p < 0.0268$). Nickel ($H = 8.143$, $p = 0.1485$) and lead ($H = 8.880$, $p = 0.1139$) concentrations did not differ among taxa in zone 2.

The most abundant species collected was *V. americana*. For this species, comparisons of tissue metal concentrations were made across all zones (Fig. 3). Zinc, nickel, copper, and silver concentrations in *V. americana*

Fig. 3 A Copper, nickel, and zinc and B silver, cadmium, and lead concentrations in *Vallisneria americana* collected from five zones (see Fig. 1) in the Lower St. Johns River, FL. Different letters indicate a significant difference between zones for a particular metal ($\alpha \leq 0.05$)

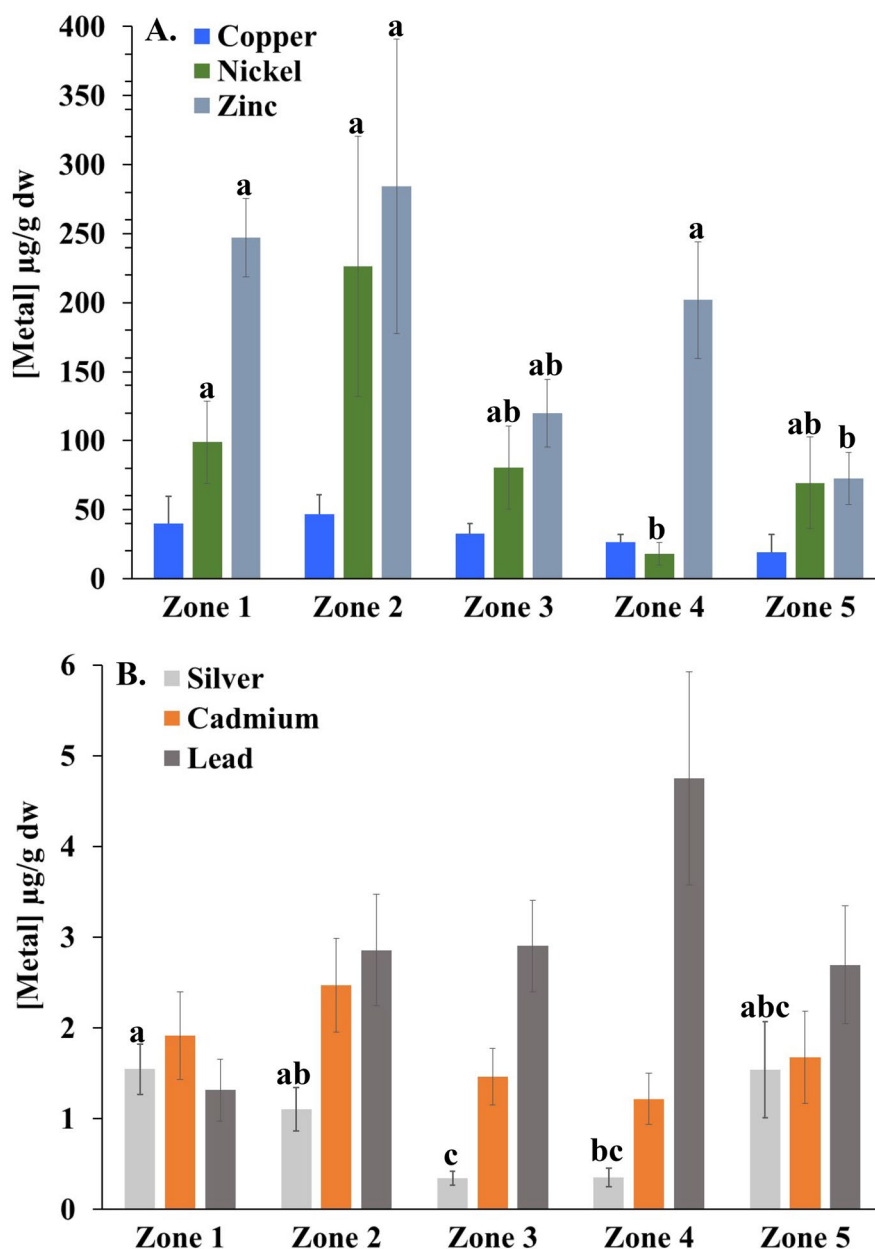


Table 1 Mean ± standard deviation of waterborne metal concentrations for each zone of the Lower St. Johns River, FL

Metal	Zone 1	Zone 2	Zone 3	Zone 4	Zone 5	All zones	Freshwater WQC	Marine WQC
Copper	2.15 ± 1.59	1.79 ± 1.52	3.57 ± 8.40	2.88 ± 3.46	1.49 ± 1.77	2.43 ± 4.68	9.3	3.7
Nickel	2.09 ± 3.00	2.27 ± 4.07	2.44 ± 1.67	2.55 ± 3.22	1.76 ± 2.06	2.27 ± 3.09	52	8.3
Zinc	8.99 ± 9.55	16.4 ± 35.4	6.57 ± 8.62	5.81 ± 4.75	4.82 ± 5.32	10.5 ± 23.2	120	83
Silver	0.30 ± 0.44	0.57 ± 1.22	0.54 ± 0.89	0.55 ± 0.66	0.15 ± 0.20	0.48 ± 0.93	0.07	0.92
Cadmium	1.12 ± 1.71	0.26 ± 0.71	0.00 ± 0.00	0.00 ± 0.00	0.01 ± 0.02	0.28 ± 0.78	0.3	8.8
Lead	0.27 ± 0.57	0.06 ± 0.18	0.04 ± 0.11	0.10 ± 0.34	0.00 ± 0.01	0.98 ± 0.26	3.2	8.5

All data are presented in µg/L

WQC water quality criterion; parts of zone 1 are marine, and all other zones are freshwater

differed significantly among zones 1–5 (zinc: $H=10.32$, $p=0.0353$; nickel: $H=11.37$, $p=0.0227$; silver: $H=14.32$, $p=0.0064$). Zinc concentrations were significantly lower in zone 5 as compared to zones 1, 2, and 4 (Dunn pairwise $p<0.0400$; Fig. 3A). Nickel concentrations were significantly lower in zone 4 as compared to zones 1 and 2 (Dunn pairwise $p<0.0172$). Copper concentrations in *V. americana* were not significantly different among zones (Fig. 3A). Silver concentrations were significantly lower in zone 3 compared to those in zones 1 and 2 (Dunn pairwise $p<0.0021$), and zone 4 was significantly lower than those in zone 1 (Dunn pairwise $p=0.0135$; Fig. 3B). Cadmium and lead concentrations in *V. americana* were not significantly different among zones (Fig. 3B). Similar to the silver accumulation in *V. americana*, cadmium and silver concentrations in *N. guadalupensis* were significantly greater in zone 2 as compared to those in zone 3 ($H=6.69$, $p=0.0097$; $H=6.063$, $p=0.0138$; data not shown).

Water samples collected in conjunction with SAV had relatively low metal concentrations, with many values below the detection limits (DL; Table 1). Cadmium ranged from <DL to 4.96 µg/L with the highest concentrations found in zones 1 and 2. The mean cadmium concentration was below the saltwater quality criterion of 8.8 µg/L and at the freshwater quality criterion of 0.3 µg/L (Table 1). Silver ranged from <DL to 5.71 µg/L, with elevated concentrations found in zones 1–4. The mean silver concentration was higher than the freshwater quality criterion of 0.07 µg/L and lower than the saltwater quality criterion of 0.92 µg/L (Table 1). Lead ranged from <DL to 1.40 µg/L with the highest values found in zone 1. The mean lead concentration was below both the freshwater and saltwater quality criteria of 3.2 and 8.5 µg/L, respectively

(Table 1). Nickel ranged from <DL to 23.0 µg/L with elevated concentrations dispersed across the zones (Table 1). All concentrations were well below the freshwater quality criteria of 52 µg/L and only one sample from zone 1 had a concentration above the saltwater quality criterion of 8.3 µg/L. Copper ranged from <DL to 39.0 µg/L. Two samples from zone 1 had concentrations that exceeded the saltwater criterion of 3.7 µg/L, and several samples from zones 3 and 4 had concentrations above the freshwater quality criterion of 9.3 µg/L (Table 1). Zinc ranged from <DL to 190.4 µg/L. The highest zinc concentrations were found in zone 2, with one value exceeding the freshwater and marine WQC (Table 1).

BCFs ranged from 44 to 645 for cadmium, 17.4 to 192 for copper, 13.4 to 66.5 for lead, 53.6 to 281 for nickel, 1.15 to 52.9 for silver, and 19.7 to 644 for zinc (Table 2). *Z. palustris* had the highest BCFs for cadmium, nickel, and silver and the second highest for copper. *V. americana* had the highest BCF for lead and second highest for nickel, and *H. verticillata* had the highest BCFs for copper and zinc (Table 2).

Discussion

Metal concentrations in these species of SAV were within range of those reported in other aquatic systems, with varying degrees of metal pollution (Ahmad et al., 2016; Biemyer-Fraser et al., 2017; Hudon, 1998; St-Cyr & Campbell, 1994, 2000; Welsh & Denny, 1980). Similar to the present study, St-Cyr and Campbell (1994) reported concentrations of 0.35–5.12 µg/g dw cadmium, 9.1–39.6 µg/g dw copper, and 1.2–8.0 µg/g dw lead in the leaves of *V. americana* collected from Lake St. Pierre, part of the St. Lawrence River in

Table 2 Bioconcentration factors for each metal and each taxon collected in the Lower St. Johns River, FL

Taxon	Cadmium	Copper	Lead	Nickel	Silver	Zinc
<i>Chara</i> sp.	85.8	17.4	66.5	53.6	14.7	23.2
<i>Eleocharis</i> sp.	48.6	132	13.4	63.0	8.75	159
<i>Hydrilla verticillata</i>	44.0	192	30.0	80.2	1.15	644
<i>Najas guadalupensis</i>	210	22.9	41.1	102	1.18	19.7
<i>Ruppia maritima</i>	168	49.7	29.5	170	2.16	63.0
<i>Vallisneria americana</i>	126	38.9	64.3	220	4.73	42.0
<i>Zannichellia palustris</i>	645	93.1	15.6	281	52.9	89.0

Canada. This area also had similar sediment metal concentrations (St-Cyr & Campbell, 1994), compared to the LSJR. Likewise, Hutchinson (1975) reported mean concentrations of 48 µg/g dw copper and 11 µg/g dw lead in various aquatic plant species. Ahmad et al. (2016) reported 71 and 85 µg/g copper, 363 and 196 µg/g nickel, 232 and 984 µg/g zinc, in *Potamogeton natans* and *Ceratophyllum demersum*, respectively. *Potamogeton* sp. have been shown to accumulate 1.2–2.2 µg/g dw cadmium, 45–50 µg/g dw copper, 5.8–7.5 µg/g dw lead, and 242–380 µg/g dw zinc (Förstner & Wittmann, 1983), like the species collected in this study.

Among the taxa sampled, *Eleocharis* sp. and *Z. palustris* had the highest concentrations of the essential metals, copper, nickel, and zinc. The differences in essential metal concentrations among the different species may have, in part, reflected differences in the nutritional requirements for these elements and/or uptake and translocation rates. *Z. palustris* also accumulated the highest concentrations of cadmium and silver, and *Eleocharis* sp. accumulated the highest lead concentrations. Uptake and accumulation of these nonessential elements may have occurred through the transport channels for the essential elements in *Z. palustris* and *Eleocharis* sp. This mimicry has been reported in other studies (Vogel-Mikus et al., 2008). Like in the present study, Bielmyer-Fraser et al. (2017) also reported higher metal bioaccumulation in *Eleocharis* sp. relative to other aquatic macrophytes. In the present study, this taxon was also less abundant, possibly indicating that *Eleocharis* sp. is less tolerant to metals. Although, it should be noted that other water quality characteristics (e.g., salinity and ammonia) could also contribute to SAV decline (Goldberg & Trent, 2020; Zhu et al., 2016).

In this study, a range of BCF values were calculated across species, with *Z. palustris* having notably

higher values. BCF values can indicate a species ability to accumulate elements from soil or water (Ladislas et al., 2012). Higher BCF values can signify higher elemental capacity when comparing across species in the same environment, or lower exposure levels if comparing among the same species in different environments (McGeer et al., 2003). BCF values may widely differ across species and for different elements (Borisova et al., 2016), which was supported by the results observed in this study. For example, *Z. palustris* had the highest BCF for cadmium, whereas *H. verticillata* had the highest BCF for zinc.

Given the relatively low metal concentrations in the water, it is likely that most of the observed metal (except for zinc) accumulation in the LSJR SAV was due to uptake from the sediments. The highest zinc concentrations in SAV were found in zone 2, which also had the highest zinc concentrations in the water. Martinez and Shu-Nyamboli (2011) have reported significant correlations between aquatic macrophytes and sediment bioavailable metal concentrations in the Gallinas River. It is also well known that SAV have a high potential to accumulate toxic elements from the surrounding environment through their thin cuticles (Prasad, 2007; Welsh & Denny, 1980), with zinc uptake occurring to a greater extent than copper or nickel (Borisova et al., 2016). During periods of elevated metal inputs into the water column of the LSJR, metals may accumulate through both water and sediment (Guilizzoni, 1991). The extent of metal uptake through each route is reportedly metal-specific (Bielmyer-Fraser et al., 2022; Welsh & Denny, 1980) and likely species-specific as well. In the present study, zone 5 had SAV with the lowest metal concentrations and it should be noted that this area is a freshwater lake, which could contribute to the observed differences (e.g., more stable sediment layers). Additionally, zones 1 and 2, which contained SAV with generally higher metal concentrations, also receive inputs from several

tributaries. Storm and hurricane events could also disrupt sediment layers and contribute to increased metal concentrations in the water (Bielmyer-Fraser et al., 2020). SAV in brown water systems like the St. Johns River (Brody, 1994) may also accumulate higher metal concentrations due to limited light availability (Hudon, 1998).

Metals can affect physiological processes, such as photosynthesis cycles and enzymatic pathways, and reduce growth in many aquatic plants (Bhalerao et al., 2015; Blaser et al., 2008; Singh et al., 2013; Zhu et al., 2016). Cadmium and lead exposure reportedly decreased chlorophyll concentration in *V. spiralis* (Wang et al., 2010) and *H. verticillata* (Singh et al., 2013). Exposure to copper-contaminated sediments has been shown to reduce chlorophyll and growth rate in *V. natans* (Zhu et al., 2016). Bielmyer-Fraser et al. (2022) reported a significant decrease in photosynthetic parameters (effective quantum yield of photosystem II) in *V. americana* exposed in the laboratory to 38 µg/L copper, 122 µg/L cadmium, and 6.9 µg/L lead for 4 days. Nickel is a strong phytotoxic metal at higher concentrations which induces changes in activity of enzymes (e.g., urease) and inhibits growth, photosynthesis, seed germination, and sugar transport and causes necrosis in plants (Bhalerao et al., 2015; Muysen et al., 2004). Exposure to metal concentrations above thresholds may cause the plants to die and degrade, thereby releasing metals back into the water column and sediment (Geng et al., 2019).

Some plant species have evolved protective mechanisms for survival in metal-rich environments including rapid element translocation, efficient accumulation, and detoxification strategies (Geng et al., 2019).

These species would be more effective for use in bioremediation (Gupta & Chandra, 1994). SAV have often been used to monitor metals in the environment and in the bioremediation of metal-contaminated environments (Gupta & Chandra, 1994). In the present study, *V. americana* and *R. maritima* had similar metal concentrations and were relatively abundant. Metal concentrations in *V. americana* also differed by site, likely due to changes in environmental metal concentrations, making this species a good bioindicator for metal pollution. Except for lead, *Chara* sp. had relatively lower metal concentrations, suggesting that this species had lower metal uptake rates and/or higher metal detoxification and excretion rates. *Chara* sp. would therefore not be indicative of environmental metal contamination levels.

In addition to metal toxicity to the SAVs, consideration should also be given to the effects on the consumers of SAV (Best, 1981; Hazrat et al., 2019; White et al., 2002). Mean metal concentrations in the sediment of the LSJR did not exceed sediment quality guidelines; however, maximum values of nickel were at the TEL, and maximum values of silver, cadmium, and lead exceeded the TEL (Table 3). Areas of increased metal concentrations in the sediment may be harmful to sensitive species. Also, sediment metal data in the LSJR were scarce. More data are needed to better estimate the threat of metal contaminants from the sediment. Except for lead, all metal concentrations were higher in SAV tissue than in the sediment in the LSJR (Table 3). In the SAV shoot tissue, copper exceeded the TEL and was at the TEC; nickel exceeded the PEL and PEC; zinc exceeded the TEL, TEC, and PEC; silver exceeded the TEL and TEC; and cadmium exceeded the TEC

Table 3 Sediment quality guidelines for marine and freshwater systems, mean (range in parenthesis) sediment metal concentrations in the LSJR in 2019, and mean SAV metal concentrations in the LSJR

Metal	PEL (marine)	TEL (marine)	PEC (freshwater)	TEC (freshwater)	Mean sediment metal concentration in LSJR (µg/g)	Mean SAV metal concentration in LSJR (µg/g)
Copper	≤ 108	≤ 19	≤ 197	≤ 36	11.18 (0.89–24.9)	35.85
Nickel	≤ 43	≤ 16	≤ 36	≤ 18	10.89 (8.90–16.0)	119.4
Zinc	≤ 271	≤ 124	≤ 315	≤ 123	41.91 (5.30–87.0)	174.4
Silver	≤ 1.8	≤ 0.7	NA	NA	0.07 (0.02–0.15)	0.98
Cadmium	≤ 4.2	≤ 0.7	≤ 3.5	≤ 0.6	0.45 (0.09–1.09)	1.76
Lead	≤ 112	≤ 30	≤ 91	≤ 35	25.43 (1.70–68.2)	3.85

TEL threshold effect level (marine), TEC threshold effect concentration (freshwater), PEL probable effect level (marine), TEC threshold effect concentration (freshwater)

and TEL (Table 3). Metal concentrations exceeding the TEL/TEC indicate that sensitive species associated with the SAV may be affected, and metal concentrations exceeding the PEL/PEC indicate that some species may be affected in the LSJR.

Conclusions

This study provides new data about SAV health in the LSJR and may help in the development of new management strategies. Species-specific metal accumulation in several species of SAV was demonstrated and could be used in conjunction with metal sequestration molecules (e.g., thiols; Borisova et al., 2016) to determine appropriate species for metal biomonitoring and bioremediation. Additionally, the results of this study highlight areas (e.g., zones 1 and 2) that may be more affected by human activities surrounding the LSJR.

Acknowledgements The authors thank Kendall Fioravante (St. Johns River Water Management District) for the help with SAV collection and Blessing Akintunde for the help with laboratory analyses.

Funding Funding was provided by the Jacksonville University Office of Research and Sponsored Programs and the Chemistry Department.

Data availability Data will be made available upon reasonable request.

Declarations

Conflict of interest The authors declare no competing interests.

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