



HAL
open science

Bioaccumulation of some trace elements in tropical mangrove plants and snails (Can Gio, Vietnam)

Nguyen Thành-Nho, Cyril Marchand, Emilie Strady, Nguyen Huu-Phat,
Tran-Thi Nhu-Trang

► **To cite this version:**

Nguyen Thành-Nho, Cyril Marchand, Emilie Strady, Nguyen Huu-Phat, Tran-Thi Nhu-Trang. Bioaccumulation of some trace elements in tropical mangrove plants and snails (Can Gio, Vietnam). *Environmental Pollution*, 2019, 248, pp.635-645. 10.1016/j.envpol.2019.02.041 . hal-02297594

HAL Id: hal-02297594

<https://hal.science/hal-02297594v1>

Submitted on 30 Sep 2019

HAL is a multi-disciplinary open access archive for the deposit and dissemination of scientific research documents, whether they are published or not. The documents may come from teaching and research institutions in France or abroad, or from public or private research centers.

L'archive ouverte pluridisciplinaire **HAL**, est destinée au dépôt et à la diffusion de documents scientifiques de niveau recherche, publiés ou non, émanant des établissements d'enseignement et de recherche français ou étrangers, des laboratoires publics ou privés.

1 **Bioaccumulation of some trace elements in tropical mangrove plants and snails**

2 **(Can Gio, Vietnam)**

3 **Nguyen Thanh-Nho^{a,b,*}, Cyril Marchand^{a,b,f}, Emilie Strady^{c,d}, Nguyen Huu-Phat^a, Tran-Thi Nhu-Trang^e**

4 *a. Department of Analytical Chemistry, Faculty of Chemistry, University of Science, Vietnam National University Ho Chi Minh*
5 *City.*

6 *b. IMPMC, Institut de Recherche pour le Développement (IRD), UPMC, CNRS, MNHN, Noumea, New Caledonia, France.*

7 *c. Univ. Grenoble Alpes, CNRS, IRD, Grenoble INP*, IGE, F-38000 Grenoble, France.*

8 *d. CARE-HCMUT, Ho Chi Minh City, Vietnam.*

9 *e. Faculty of Chemical Engineering and Food Technology, Nguyen Tat Thanh University, Vietnam.*

10 *f. Université de la Nouvelle-Calédonie (UNC), PPME, EA 3325, BP R4, 98 851 Noumea, New Caledonia, France.*

11 ** Corresponding author: ntnho@hcmus.edu.vn*

12 **ABSTRACT**

13 Mangrove sediments can store high amount of pollutants that can be more or less bioavailable
14 depending on environmental conditions. When in available forms, these elements can be subject
15 to an uptake by mangrove biota, and can thus become a problem for human health. The main
16 objective of this study was to assess the distribution of some trace elements (Fe, Mn, Co, Ni, Cr,
17 As, and Cu) in tissues of different plants and snails in a tropical mangrove (Can Gio mangrove
18 Biosphere Reserve) developing downstream a megacity (Ho Chi Minh City, Vietnam). In addition,
19 we were interested in the relationships between mangrove habitats, sediment quality and
20 bioaccumulation in the different tissues studied. Roots and leaves of main mangrove trees
21 (*Avicennia alba* and *Rhizophora apiculata*) were collected, as well as different snail species:
22 *Chicoreus capucinus*, *Littoraria melanostoma*, *Cerithidea obtusa*, *Nerita articulata*. Trace
23 elements concentrations in the different tissues were determined by ICP-MS after digestion with
24 concentrated HNO₃ and H₂O₂. Concentrations differed between stands and tissues, showing the
25 influence of sediment geochemistry, species specific requirements, and eventually adaptation
26 abilities. Regarding plants tissues, the formation of iron plaque on roots may play a key role in
27 preventing Fe and As translocation to the aerial parts of the mangrove trees. Mn presented higher
28 concentrations in the leaves than in the roots, possibly because of physiological requirements. Non-
29 essential elements (Ni, Cr and Co) showed low bioconcentration factors (BCF) in both roots and
30 leaves, probably resulting from their low bioavailability in sediments. Regarding snails, essential
31 elements (Fe, Mn, and Cu) were the dominant ones in their tissues. Most of snails were
32 “macroconcentrators” for Cu, with BCF values reaching up to 42.8 for *Cerithidea*. We suggest
33 that high quantity of As in all snails may result from its high bioavailability and from their ability
34 to metabolize As.

35 *Keywords: Bioavailability; Bioconcentration factor, Translocation; macroconcentrator.*

36 **1. Introduction**

37 Mangroves, complex intertidal forests, are considered as sinks for contaminants including trace
38 metals (Tam and Wong 2000). It was previously suggested that mangrove plants can play key roles
39 in metals removal from mangrove sediments (Alongi et al. 2004, Yang et al. 2008). On the one
40 hand, depending on the metals properties and the mangrove species, they can provide
41 “phytostabilization”, meaning that metals are immobilized and stored in the sediment or in the
42 below-ground biomass (MacFarlane et al. 2007). On the other hand, they can provide
43 “phytoextraction”, and mangrove trees can be considered as accumulators, providing that they can
44 transfer metals from the sediment and concentrate them in above-ground tissues (Křibek et al.
45 2011). Because of different rates in metals uptake and the specific influence of mangrove roots on
46 sediment geochemistry, metals dynamic in mangrove sediments may be affected by the
47 composition of plant communities (Verkleij and Schat 1990). In fact, some mangrove trees can
48 oxidize the sediments via the movement of oxygen from their rhizosphere downwards to
49 aerenchyma tissues (Moorhead and Reddy 1988). This oxidation process can remobilize the stable
50 forms of metals, e.g. the ones bound to sulphides (Marchand et al. 2006, Noël et al. 2015), thus
51 increasing metals’ bioavailability. The bioaccumulation and/or fixation of trace metals in
52 mangrove plant tissues, especially roots which can then redistribute metals proportion into
53 aboveground tissues (i.e. especially leaves, see review of (Weis and Weis 2004)), may limit metals
54 concentrations in the water column and restrict the transfer of these contaminants into mangrove
55 biota. However, the excess of essential and non-essential trace elements could affect the growth,
56 metabolism activities and cell structure of plants (Cox and Hutchinson 1981, Wang et al. 2003).

57 Bioaccumulations of trace elements by snail species have been addressed in many research
58 projects in the past two decades (Berandah et al. 2010, Dias and Nayak 2016, Reed-Judkins et al.

59 1997). The variations of trace metals concentrations in snails depend on their feeding regime, their
60 digging activities, and metals bioavailability within their habitat. Snails can accumulate higher
61 metals concentrations than any other groups of mollusks (Zhou et al. 2008) and can offer the
62 possibility to assess metals contaminations (De Wolf and Rashid 2008, Samsi et al. 2017, Yap and
63 Cheng 2013). It was demonstrated that elevated trace metals concentrations in sediments can
64 influence negatively the number of snail species in an ecosystem as well as the community
65 structure (Amin et al. 2009).

66 In Vietnam, an emerging country, the fast economic development (i.e. urbanization,
67 industrialization, etc.) and the population growth induce high pressure on rivers and estuaries
68 (Babut et al. 2019, Costa-Boddeker et al. 2017, Strady et al. 2017). Can Gio estuary is located at
69 the edge of the biggest industrial city in Vietnam-Ho Chi Minh City (*i.e.* a megacity of almost 10
70 million inhabitants), and half of the Can Gio area is covered by mangrove forests. This estuary is
71 also a unique gate for drainages of sewages from the land to the ocean. Recently, Thanh-Nho et
72 al. (2018) highlighted that metals can be transferred over long distance from watersheds to the
73 mangrove forest, and that elevated inputs of metals in the estuary were the result of enhanced
74 runoff and soil leaching during the monsoon season. It was proved that the enrichment of
75 mangrove-derived organic matter played a key role in controlling partitioning and availability of
76 trace elements in the sediment beneath different mangrove species (Thanh-Nho et al. 2019).
77 Consequently, taking into account mangrove specific geochemical characteristics, the lack of
78 wastewater treatment plants in emerging countries as Vietnam and the important local ecosystem
79 services provided by the Can Gio Mangrove (Cormier-Salem et al. 2017, Kuenzer and Tuan 2013),
80 more attention should be paid on metals bioaccumulation and transfer in mangrove ecosystems.
81 Because of their persistence, trace metals may present a major threat to the mangrove diversity

82 and also to human health via food chain. Notably, the Can Gio mangrove is home to high
83 biodiversity with 20 species of flora, in which two mangrove species are dominant: *Avicennia alba*
84 and *Rhizophora apiculata* (Luong et al. 2015), and with more than 200 species of fauna (e.g.
85 benthic organisms, fish, mollusks, planktonic, etc.). Thus, the knowledge of trace elements'
86 contents accumulated by different mangrove organisms would be useful to get a better
87 understanding on the behaviors of these contaminants in the mangrove, and to contribute to an
88 important information on the protected species of tropical mangroves in Vietnam and their
89 potential element poisoning.

90 The main objective of this study was to assess the distribution of some trace elements (Fe, Mn,
91 Co, Ni, Cr, As, and Cu) in the tissues of different mangrove plants and snails receiving elevated
92 amounts of contaminants from watersheds, i.e. lateritic soils and Ho Chi Minh City. In addition,
93 we were interested in the relationships between the bioaccumulation in the different tissues
94 studied, mangrove stands, and sediments quality, bioconcentration factors were calculated using
95 data published in a previous paper and concerning trace metals dynamics in the sediment of the
96 studied mangrove stands (Thanh-Nho et al. 2019). To reach our goals, we collected and analyzed
97 different parts (i.e. roots and leaves) of the main mangrove trees (i.e. *Avicennia alba* and
98 *Rhizophora paticulata*) in the Can Gio Biosphere Reserve, and different snail species within
99 mangrove stands: *Chicoreus capucinus* (*C.capucinus*), *Littoraria melanostoma* (*L.melanostoma*),
100 *Cerithidea obtuse* (*C.obtusa*) and *Nerita articulata* (*N.articulata*). The field sampling was
101 conducted at the end of the monsoon season in 2015, since we showed previously that elements
102 inputs and sediment reactivity were highest at this period, and since our main objective was to
103 compare the two main mangrove stands.

104 2. Materials and methods

105 2.1. Study area

106 The present study was conducted in the Can Gio mangrove Biosphere Reserve, Vietnam
107 (10°22'-10°44'N and 106°46'-107°01'E, Fig. 1). This mangrove covers approximately 35,000 ha
108 (Tuan and Kuenzer 2012), being usually classified as a “Mangrove afforestation and re-forestation
109 area” (Blasco et al. 2001). It is also a district of the densely populated megacity of Ho Chi Minh
110 City (HCMC), with almost 10 million of inhabitants. The Can Gio mangrove is situated 35 km
111 downstream of the city urban center and industrial zones (mainly plastic and rubber production,
112 mechanical engineering, electrical engineering, packaging, textile and dyes industry, oil activities
113 and cement production) (Strady et al. 2017, Vo 2007). The main economic activities of the local
114 people in Can Gio are aquacultures, salt production, fishing, and forest management. The
115 topography of Can Gio mangrove is generally low-lying. This coastal area is subject to an
116 asymmetric semi-diurnal tidal regime and to the typical tropical monsoon climate, with two
117 distinct seasons. The dry season extends from November to April and the wet season lasts from
118 May to October. The climatic conditions were not much different during consecutive years, from
119 2014 to 2016, prior and after sampling campaign (data from HCMC statistical Years Book, 2016).
120 The highest precipitation can reach up to 500 mm in September or October, while it is usually less
121 than 80 mm per month during the dry season. The annual mean precipitation ranged from 1,800 to
122 2,000 mm, with almost 90 % of the precipitation falling during the rainy season. The annual mean
123 temperature varied from 28.4 °C to 30 °C while the annual humidity ranged from 72 to 74 %. The
124 two dominant mangrove species are *Avicennia alba* and *Rhizophora apiculata*. Because of high
125 commercial values, *Rhizophora apiculata* was widely replanted, being often found on elevated
126 ground. *Avicennia alba* is a pioneering species with high salinity tolerance and ability to grow on
127 weak, unconsolidated sediment. Consequently, there is a zonation of the ecosystem (Fig. 1) with

128 *Rhizophora* trees growing at higher elevation than the *Avicennia* ones, and being thus less
129 immersed by tides. Many marine organisms live within and around the mangroves, e.g. shrimp,
130 fish, snail, or crab, etc. Therefore, the mangrove ecosystem provides not only various goods and
131 services to local people such as timber, seedlings, medicines, but also foods (Kuenzer and Tuan
132 2013).

133 2.2. Field sampling

134 The sampling campaign was carried out at the end of the wet season (October 2015) in the
135 center area of the Can Gio mangrove (Fig. 1).

136 Leaf and root samples were collected from different types of trees (saplings and mature trees
137 of the *Rhizophora apiculata* and *Avicennia alba* species). For both species, each leaf sample
138 consisted of 30 leaves collected from 15 trees, and we analyzed 3 samples per each type of tree.
139 Root samples were collected in the upper 50 cm from sediment cores collected using an
140 Eijkelkamp gouge auger (inner diameter: 80 mm) at low tide ($n = 3$ per each type of tree). All
141 samples were rinsed with deionized water and were then dried at 50 °C for > 48 h to constant mass
142 in an oven. Dried samples were then ground and sieved using 100 μm pore size for trace elements
143 analysis.

144 Four species of snails were collected in the mangrove stands: *C.capucinus*, *L.melanostoma*,
145 *C.obtusa* and *N.articulata*. Notably for *L.melanostoma*, they were collected on mangrove trees. In
146 the *Rhizophora* stand, due to their excessive height (> 15 m), *L.melanostoma* were collected on
147 young trees. In the *Avicennia* stand, due to low density of saplings, *L.melanostoma* were collected
148 on mature trees. For each species, we analyzed 3 samples per mangrove stand, and each sample
149 comprised 15 to 20 snails. Their shell sizes (length and width) were measured prior to taking out
150 their tissues by pegging out (using a clean pestle) the shell carefully. Due to the fragile

151 characteristic of the shell, a mild force was sufficient to break the shell (strong force might destroy
152 the internal organ of the snail). Snails were chosen in the same range of size throughout the work
153 to reduce possible variations in trace elements concentrations due to size (age). The whole bodies
154 were preserved at -18 °C until performing processes of freezer-drying. Dried samples were ground
155 and sieved through 100 µm pore size for analysis of trace elements. Snails' soft tissues are food
156 sources for local people, and thus their trace elements contents may be a risk to human health.
157 Therefore, only the soft tissues of snails were analyzed in the present study.

158 *Specific features of selected snail's species*

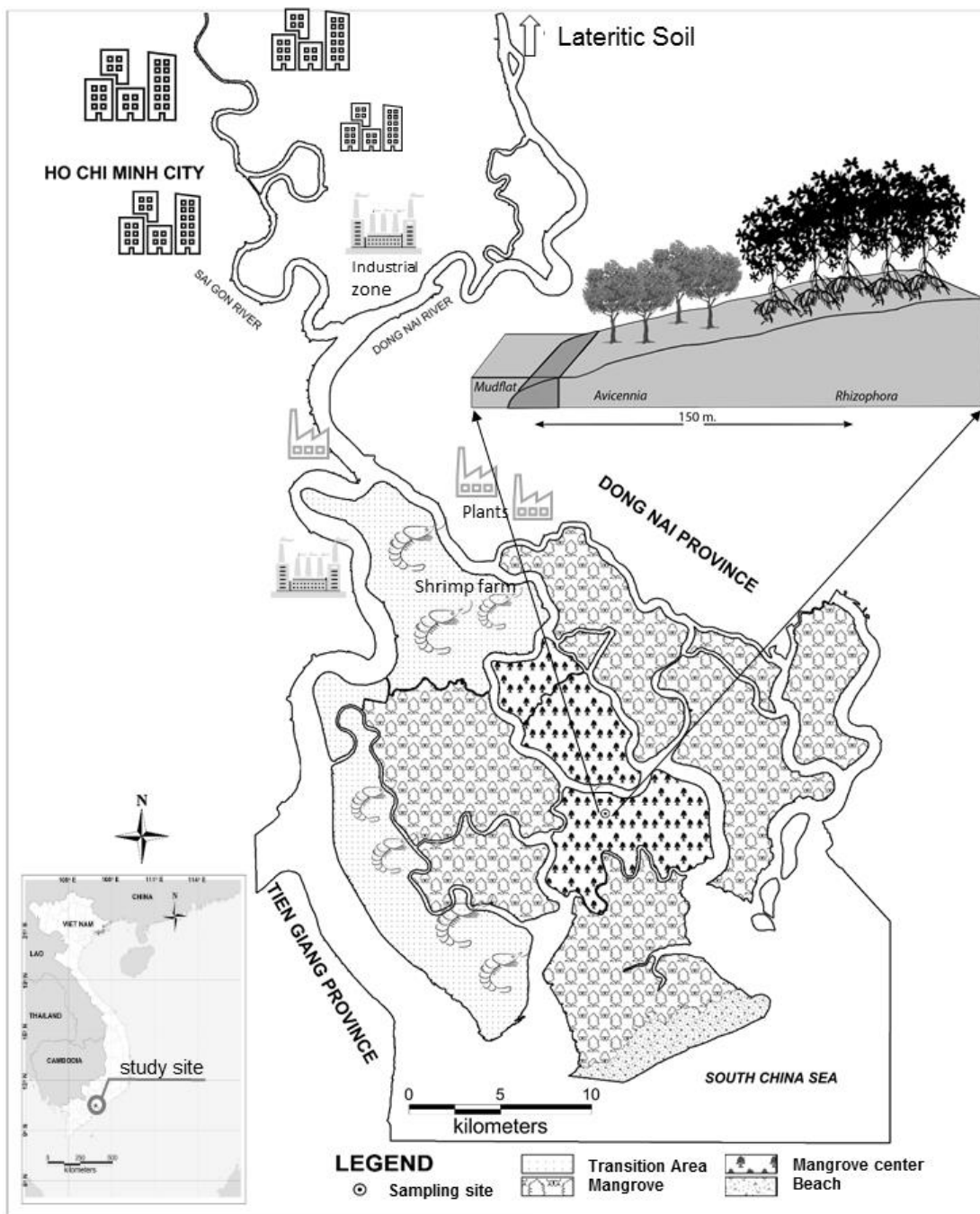
159 *C.capucinus* has elaborately textured shell with uniformly dark brown and six convex whorls.
160 These shells are heavy and solid and can reach a size of 40 to 120 mm. They are sculptured with
161 prominent spiral cords, axial ribs and striae. The aperture is rounded or oviform, brown tinged and
162 the inner labial edge show 14 to 17 denticles. The siphonal canal is quite long. The operculum is
163 dark brown.

164 *L.melanostoma* can reach 2 to 3 cm length. Shell has a narrower tip, smooth, yellowish to moss
165 green or dark brown with fine spiral pattern of brown dots. Sometimes the shell is bleached white.
166 "*Melanostoma*" means "black mouth". There is short fat tentacles on soft head.

167 *C.obtusa* has Obtuse Horn Shell (large, solid, broad, periphery rounded; aperture strongly
168 thickened and flared; 14 to 37 rounded axial ribs on penultimate whorl; 5 spiral cords on spire, 5
169 to 9 above periphery on last whorl). Shell is also known as Mud Creeper. It is a relatively common
170 snail found in muddy coastal areas, growing to around 5 to 6 cm (Suwanjarat and Klepal 2001).

171 *N.articulata* is also known as *Nerita lineata* or *N. balteata*, with 2 to 3 cm length. Shell is
172 sturdy oval, spire does not stick out characterized by beige, greyish or pinkish with fine, spiralling
173 and black ribs. The flat underside is smooth, white, sometimes with yellow patches. Small notched

174 'teeth', usually three, on the straight edge at the shell opening. Operculum thick, evenly covered in
 175 tiny bumps, pinkish with black portions. Body pale with fine black bands on the foot and long thin
 176 black tentacles (data from wildsingapore).



177
 178 *Fig. 1. Map of the study area showing the location of the Can Gio mangrove in Vietnam and the location of the*
 179 *collected biological samples in the mangrove.*

180 2.3. Analytical methods and calculations

181 2.3.1. Samples digestion and analysis

182 The plant and snail samples were digested in triplicate with concentrated nitric acid and
183 hydrogen peroxide (MacFarlane et al. 2003). The samples (250 mg to 500 mg of dried weight)
184 were put into PTFE vessels, in which 10 mL of concentrated HNO₃ was added. These samples
185 were homogenized in an ultrasonic bath for 15 min and were then digested at 110 °C for 12h on
186 an electrical oven. After cooling, 2 mL of H₂O₂ was added into these samples, which were again
187 digested at 110 °C for 30 min. The residual HNO₃ was eliminated at 160 °C. The samples were
188 centrifuged to reject any residues and then diluted to 25 mL using deionized water and stored at 4
189 °C until analysis. The trace elements were determined by ICP – MS (Agilent 7700x at Institute of
190 Public Health, Ho Chi Minh City, Vietnam), using spiked ¹⁰³Rh and ¹⁹⁷Au as internal standards.
191 The precision and accuracy of analytical method were controlled using certified reference material
192 muscle tissues (SRM-2976: certified by National Institute of Standard and Technology/NIST,
193 USA). The recoveries of trace elements were 94 % to 106.5 % with relative standard deviation
194 from 4.2 % to 9.8 % (Table SD1). All chemicals were analytical grade (Merck). HNO₃ was purified
195 using a sub-boiling quartz distillation equipment.

196 2.3.2. Data calculations

197 Bioconcentration factor (BCF) was proposed by Babukutty and Chacko (1995) to assess the
198 bioaccumulation of pollutants into an aquatic and terrestrial organisms via any route i.e., by active
199 or/and passive accumulations. It is defined as $BCF = C_{tissues}/C_{sediment}$, where $C_{tissues}$ and $C_{sediment}$ are
200 total concentration of trace element in organism tissues and total concentration of trace element in
201 the sediment, respectively. To do this calculation, we used the trace elements concentrations in the

202 sediment that we published in a previous paper (Thanh-Nho et al. 2019); sediment samples were
203 collected at the same sites and at the same period of the present study.

204 *BCF of trace elements in roots and leaves*

205 BCF was calculated from a mean concentration of trace element in sediment down to 50 cm
206 depth (Table SD2). This depth was chosen because in mangrove environment, the highest root
207 density is in the upper sediment, between 20 and 50 cm depth (Ha et al. 2018, Komiyama et al.
208 2000, Tamoooh et al. 2012).

209 Translocation factors (TF) was calculated, which are used to assess the transfer of trace
210 elements from roots to aboveground components and to evaluate the potential phytoextraction of
211 plants (Marchiol et al. 2004). In the present study, it is defined as $TF = C_{leaves}/C_{root}$, where C_{leaves}
212 and C_{root} are total concentration of trace element in leaves and total concentration of trace element
213 in roots, respectively.

214 *BCF of trace elements in snails*

215 Depending on feeding regimes of individual snail species and their habitat, the BCF was
216 calculated from a mean concentration of trace element corresponding to a mean concentration of
217 trace element in each type of feeding source. *L.melanostoma* feeds on mangrove trees, mainly
218 fresh leaves and materials on the plant surfaces as phylloplane and fungi (Lee et al. 2001). BCF of
219 trace element was estimated based on the mean concentration of trace element measured in
220 *Avicennia* and *Rhizophora* leaves. *C.obtusa* is known as sediment eater (Tue et al. 2012), BCF of
221 trace element was calculated based on a mean concentration corresponding to a mean
222 concentration of trace element in available fractions at the upper 5 cm of the sediment (Table SD2).
223 *N.articulata* eats microalgae (Eichhorst 2016), and *C.capucinus* is a versatile predator, feeding on
224 the barnacles growing on mangroves and on mussels, snails and worms in their habitat (Tan and

225 Oh 2003, Tan 2008). We were not able to sample their feeding sources, consequently, we did not
226 calculate any BCF for two snails' species.

227 **3. Results**

228 *3.1. Trace elements in roots and leaves of mangrove plants*

229 Mean concentrations of Fe, Mn, Co, Ni, Cr, As, and Cu ($\mu\text{g g}^{-1}$) in the roots and leaves of the
230 *Avicennia alba* and the *Rhizophora apiculata* are presented in Table 1. Trace elements in roots of
231 the *Avicennia* saplings and mature trees presented a similar distribution: Fe > Mn > Cu > Cr > As
232 > Ni > Co. For the roots of the *Rhizophora* trees, the concentrations of trace elements were Fe >
233 Mn > As > Cr > Cu ~ Ni > Co in the saplings, and were Fe > Mn > Cr > Ni ~ Cu > As ~ Co in the
234 mature trees. Most trace elements were characterized by low values of BCF in the roots, i.e. less
235 than 1, except for Cu and As for the roots of *Avicennia* saplings (i.e. BCF: 1.02 for As and 2.95
236 for Cu) (Fig. 2). Leaves of the *Avicennia* saplings were characterized by the following trace
237 elements concentrations: Fe > Mn > Cu > Cr > Ni > Co ~ As, while in the mature trees, the
238 distribution was Mn > Fe > Cu > Cr > Ni > Co ~ As. Mn presented higher concentrations in the
239 leaves of mature trees than in the saplings. For leaves of the *Rhizophora*, the concentration of trace
240 elements were Fe > Mn > Cr ~ Cu > Ni > Co ~ As in the saplings, and were Fe > Mn > Cu > Cr >
241 Ni > As ~ Co in the mature trees. Mn and Cu concentrations were higher in the leaves of the mature
242 trees than in the saplings. The bioconcentration factors were lower than 1 for most trace elements
243 in both mangrove species, with the exception of Mn in saplings and mature trees (Fig. 2). The
244 translocation factors (TF) were higher than 1 for Mn, Cu in the *Rhizophora* saplings and mature
245 trees, for Cr in the *Rhizophora* saplings, while it was lower than 1 for the other trace elements
246 studied (Fig. 3).

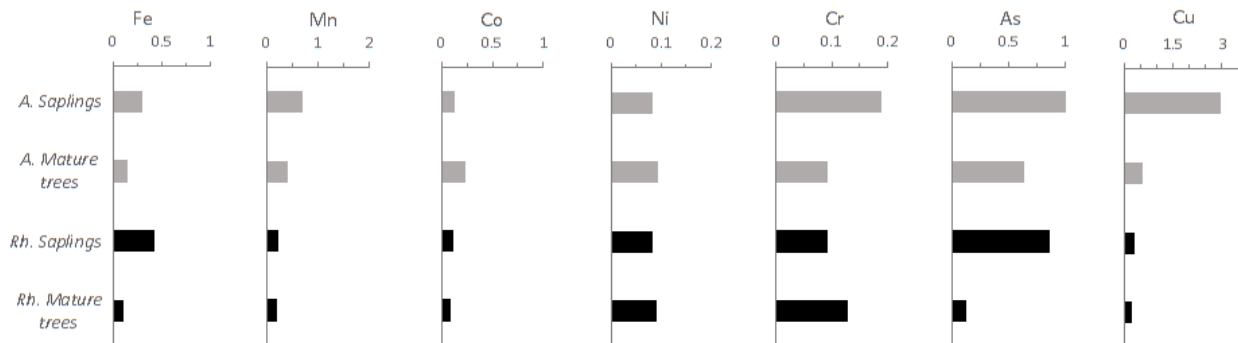
247

248 *Table 1. Mean concentrations of trace elements (n = 3) in the roots and leaves of the Avicennia alba and the*
 249 *Rhizophora apiculata, expressed in $\mu\text{g g}^{-1}$ (mean, SD). The roots of trees were collected in the upper part of 50 cm*
 250 *depth in the sediments.*

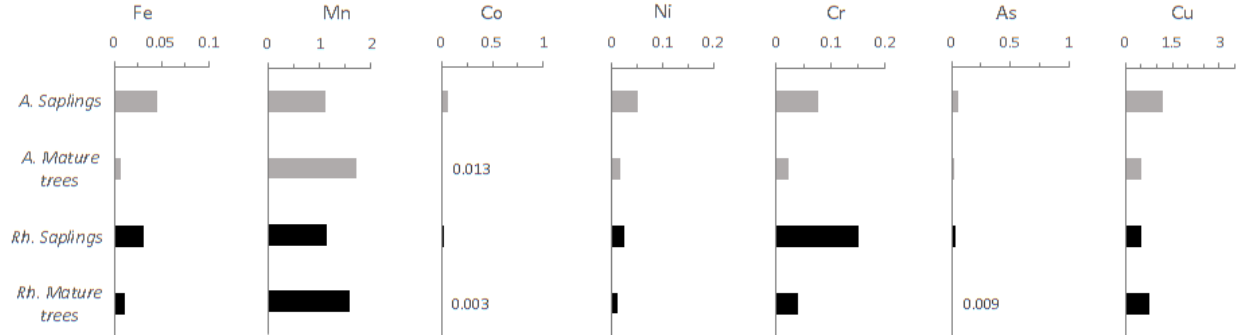
Roots									
	Depth (cm)	Type of trees	Fe	Mn	Co	Ni	Cr	As	Cu
<i>Avicennia</i>	< 50	Saplings	14,948 ± 239	376 ± 4	2.8 ± 0.06	5.1 ± 0.16	17.8 ± 0.11	11.9 ± 0.12	54.7 ± 1.8
	< 50	Mature trees	7,433 ± 308	217 ± 1.7	5.0 ± 0.13	5.8 ± 0.09	8.9 ± 0.42	7.5 ± 0.19	10.2 ± 0.29
<i>Rhizophora</i>	< 50	Saplings	19,897 ± 1,265	60 ± 1.2	2.4 ± 0.05	5.0 ± 0.11	8.8 ± 0.25	12.7 ± 0.15	6.3 ± 0.12
	< 50	Mature trees	5,458 ± 145	57 ± 1.1	2.0 ± 0.01	5.4 ± 0.08	11.9 ± 0.33	2.0 ± 0.05	4.7 ± 0.13

Leaves									
	Height of tree	Type of trees	Fe	Mn	Co	Ni	Cr	As	Cu
<i>Avicennia</i>	< 80 cm	Saplings	2,287 ± 39	583 ± 5	1.43 ± 0.01	3.18 ± 0.28	7.31 ± 0.22	0.61 ± 0.02	22.14 ± 0.30
	> 7 m	Mature trees	359 ± 21	908 ± 20	0.29 ± 0.10	1.09 ± 0.14	2.13 ± 0.29	0.30 ± 0.03	8.93 ± 0.30
<i>Rhizophora</i>	< 80 cm	Saplings	1,416 ± 50	267 ± 3	0.48 ± 0.02	1.60 ± 0.02	13.19 ± 0.79	0.40 ± 0.01	10.63 ± 0.55
	> 15 m	Mature trees	498 ± 21	405 ± 38	0.07 ± 0.03	0.70 ± 0.25	3.72 ± 0.30	0.14 ± 0.03	15.92 ± 0.80

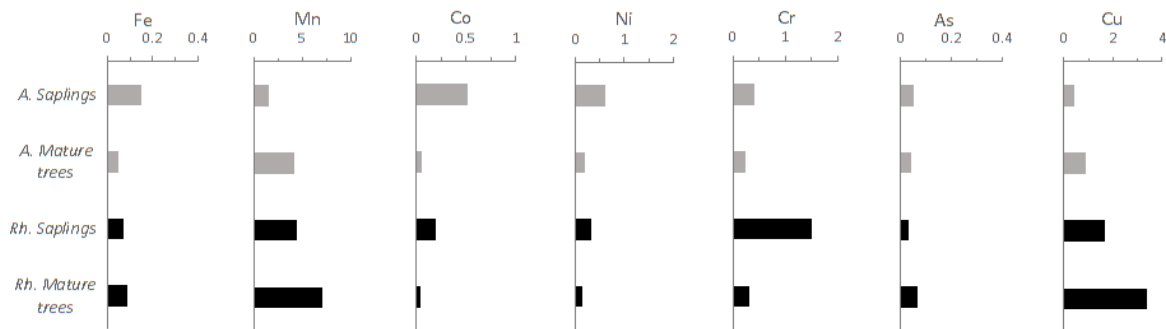
BCF in Roots



BCF in Leaves



251
 252 *Fig. 2. Bioconcentration factors (BCF = concentration of trace element in tissues/ concentration of trace*
 253 *element in sediment) of the Avicennia alba (gray column) and the Rhizophora apiculata (black column).*



254

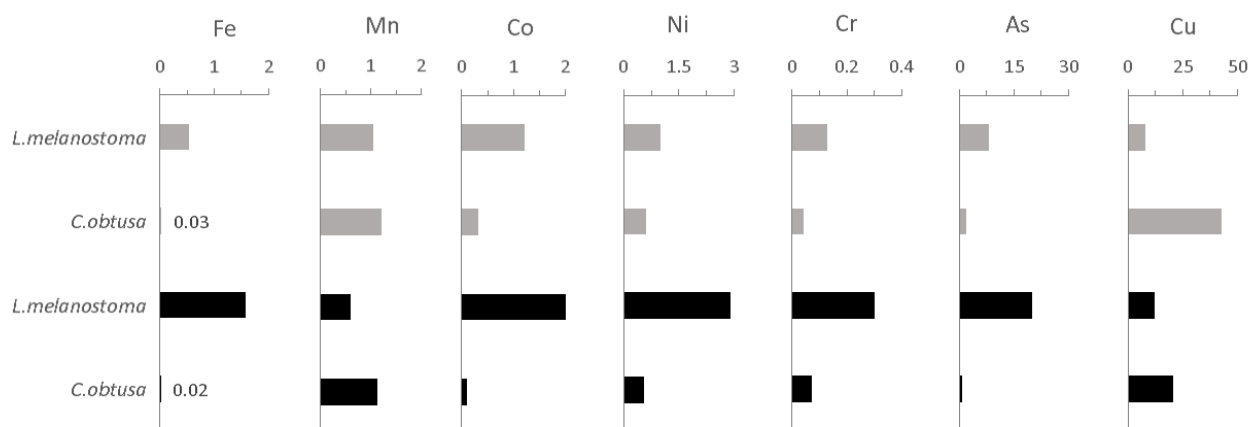
255 *Fig. 3. Translocation factors (TF = concentration of trace element in leaves/ concentration of trace element in*
 256 *roots) of the Avicennia alba (gray column) and the Rhizophora apiculata (black column).*

257 **3.2. Trace elements in snails**

258 Mean concentrations of Fe, Mn, Co, Ni, Cr, As, and Cu ($\mu\text{g g}^{-1}$) in snails' soft tissues are given
 259 in Table 2. The *C.capucinus* and *N.articulata* presented similar trace elements concentrations
 260 beneath two mangrove stands: $\text{Fe} > \text{Cu} > \text{Mn} > \text{As} > \text{Ni} \sim \text{Cr} \sim \text{Co}$ for *C.capucinus* and $\text{Fe} > \text{Mn}$
 261 $> \text{Cu} > \text{As} \sim \text{Ni} \sim \text{Cr} > \text{Co}$ for *N.articulata*. *L.melanostoma* and *C.obtusa* exhibited higher
 262 concentrations in Fe, Mn and Cu than As, Ni, Cr, and Co, and concentrations differed between
 263 mangrove stands (*Avicennia alba* and *Rhizophora apiculata*). Bioconcentration factors of all trace
 264 elements are presented in Fig. 4. The BCF of each trace element varied between snail species,
 265 ranging from 0.02 to 1.58; 0.6 to 1.13; 0.1 to 5.09; 0.56 to 2.9; 0.04 to 0.3; 0.85 to 20.1 and 7.7 to
 266 42.8 for Fe, Mn, Co, Ni, Cr, As, and Cu respectively.

267 *Table 2. Concentrations of trace elements in soft tissues of various snails (expressed in $\mu\text{g g}^{-1}$): C.capucinus*
 268 *(predator), L.melanostoma (leaves eater), C.obtusa (sediment eater) and N.articulata (algae eater).*

Mangrove stand	Species	Length (cm)	Width (cm)	Fe	Mn	Co	Ni	Cr	As	Cu
Avicennia	<i>C.capucinus</i>	2.5 - 4	1 - 1.5	536 ± 40	139 ± 10	0.88 ± 0.11	1.52 ± 0.16	1.11 ± 0.17	10.9 ± 1.1	481 ± 38
	<i>L.melanostoma</i>	1.5 - 2.5	1 - 2	714 ± 95	766 ± 29	1.04 ± 0.03	2.14 ± 0.08	0.61 ± 0.12	3.64 ± 0.17	120.6 ± 9.1
	<i>C.obtusa</i>	2 - 4	1 - 1.5	360 ± 44	780 ± 81	2.64 ± 0.26	5.93 ± 0.44	0.70 ± 0.11	4.32 ± 0.88	113.9 ± 5.7
	<i>N.articulata</i>	2 - 2.6	1.5 - 2	271 ± 26	42.5 ± 2.6	0.20 ± 0.13	1.16 ± 0.15	0.90 ± 0.25	2.83 ± 0.22	7.84 ± 0.65
Rhizophora	<i>C.capucinus</i>	3 - 4	1.5 - 2.5	885 ± 80	119 ± 11	1.14 ± 0.03	2.07 ± 0.12	1.27 ± 0.31	14.3 ± 2.2	389 ± 13
	<i>L.melanostoma</i>	1.5 - 3	1 - 2	1,517 ± 120	201 ± 20	1.41 ± 0.02	3.34 ± 0.30	2.5 ± 1.3	5.39 ± 0.25	163 ± 17
	<i>C.obtusa</i>	3 - 4	1 - 1.2	338 ± 29	158 ± 16	0.99 ± 0.09	6.15 ± 0.74	1.58 ± 0.41	6.95 ± 1.15	120 ± 20
	<i>N.articulata</i>	1.8 - 2.5	1.2 - 1.5	795 ± 82	96 ± 14	0.88 ± 0.27	3.58 ± 1.74	5.99 ± 1.43	6.45 ± 1.69	16.6 ± 0.9



269

270 *Fig. 4. Bioconcentration factors (BCF = concentration of trace element in soft tissues/ concentration of trace*
 271 *element correspond to its concentration in feeding of individual species) of various snails living in the Avicennia*
 272 *alba (gray column) and the Rhizophora apiculata (black column).*

273 4. Discussions

274 4.1. Bioaccumulation of trace elements in roots and leaves of Avicennia alba and

275 Rhizophora apiculata

276 4.1.1. Essential elements: Fe, Mn, Cu

277 Iron is an important component of chlorophyll, being useful for protein synthesis and root
 278 growth (Jones Jr et al. 1991). In the present study, despite low bioconcentration factors (BCF), Fe
 279 was the most abundant element in the roots whatever the mangrove species, reaching up to 19,897
 280 $\mu\text{g g}^{-1}$ (Table 1). These mean concentrations of Fe in the plants' roots of the Can Gio mangrove
 281 were substantially higher than in most mangroves, like for instance in Indian mangroves
 282 (Kathiresan et al. 2014). We suggest that the high concentrations of measured Fe in the roots
 283 resulted from the high iron concentrations in the sediments (Table SD2) and in pore-waters
 284 (Thanh-Nho et al. 2019) (e.g. where it can be highly bioavailable (Abohassan 2013)), as observed
 285 for *Avicennia marina* and *Rhizophora stylosa* in New Caledonian mangroves, where the mangrove
 286 sediments are rich in iron because these ecosystems develop downstream lateritic soils (Marchand

287 et al. 2016). However, if the Fe concentrations were high in the roots, the translocation factors to
288 the leaves were low for this element (Fig. 3), suggesting that the root acted as a barrier preventing
289 iron translocation to aerial parts. Machado et al. (2005) suggested that the seedlings of mangrove
290 species can exclude some trace elements through iron plaque formation on the roots. The oxygen
291 released by the roots of some mangrove species may promote oxidizing conditions within the
292 rhizosphere, which results in trace elements precipitation at the root surface, creating iron-rich root
293 coatings, generally called iron plaques (Chaudhuri et al. 2014, Koch and Mendelssohn 1989, Zhou
294 et al. 2011). Machado et al. (2005) also demonstrated that the washing of the roots prior to analysis
295 influences the fixation of the iron plaque on the roots: in contrary to distilled water, the dithionite–
296 citrate–bicarbonate (DCB) solution is able to extract the iron plaque from the roots. Considering
297 that we only washed the roots with deionized water, the presence of iron plaques at the root surface
298 may be a possible explanation for the elevated iron concentrations in the plants' roots in the Can
299 Gio mangrove. For the mature trees, iron concentrations were higher in the *Avicennia* roots than
300 the *Rhizophora* ones, which may be the results of the higher iron concentrations both in the solid
301 and the dissolved phases (Table SD2) in the sediment of the *Avicennia* stand (Thanh-Nho et al.
302 2019). Also, the *Avicennia* root system is known to release oxygen (Marchand et al. 2004,
303 Scholander et al. 1955), which may increase iron plaque precipitation. Notably, whatever the
304 mangrove species, iron concentrations in saplings leaves presented higher values than the mature
305 ones as a result of higher iron concentrations in saplings roots (Table 1). We suggest that iron
306 plaque may be a source of Fe for plants uptake by bacterial reductive dissolution of plaque during
307 organic matter decay processes on dead roots surface. Additionally, the root system of the saplings
308 may develop in the upper layer, and not as deep as the one of the mature trees. This upper layer is
309 more subject to variations in the redox conditions, and possibly to enhanced alternation of

310 precipitations/dissolution processes. Consequently, elements, specifically iron, may be more
311 available at the depth of the saplings' roots. Eventually, it is possible that the adaptation strategy
312 of the mangrove trees was not fully developed at the sapling stage, as a result, the role in barrier
313 of the root system may be limited.

314 Manganese is a major contributor to various biological processes, including photosynthesis
315 (Millaleo et al. 2010). In Can Gio mangrove, whatever the mangrove species, Mn was the second
316 abundant element in their tissues, with average concentrations in the range of those measured
317 worldwide in mangroves (see review of Lewis et al. (2011) and Bayen (2012)). Conversely to Fe,
318 Mn accumulated more in the leaves than in the roots (Table 1), as previously observed in New
319 Caledonia by Marchand et al. (2016). TF values (i.e. maximum TF: 4.08) and Mn concentrations
320 in roots and leaves of mangrove plants measured in New Caledonia were lower than in the present
321 study (e.g. TF reached up to 7.07). We suggest that the high dissolved Mn concentrations measured
322 in the pore-waters and the difference of Mn partitioning in the mangrove sediments may be
323 responsible for the elevated values measured in the present study. Within the top 50 cm of the
324 sediments in the Can Gio mangrove, approximately 60 % of Mn was associated with the
325 bioavailable fractions (Table SD2) whereas in New Caledonia, 90 % of Mn was bound to the
326 refractory fraction, preventing its transfer to mangrove plants (Marchand et al. 2016). In the Can
327 Gio mangrove, Mn was only the element to potentially present an ecological risk due to its high
328 bioavailability (ratio in concentrations of trace element in the bioavailable fractions
329 (exchangeable/carbonate bound) to the total concentrations of trace element) (Thanh-Nho et al.
330 2019). Since high TF of Mn was already reported for *Rhizophora* species, we suggest that it may
331 be related to metabolic requirements, this element playing an important role in enzymes reactions

332 is also needed for water splitting at photosystemII (McEvoy and Brudvig 2006). Consequently,
333 *Rhizophora* trees have a high potential for Mn phytoextraction.

334 Regarding copper, it is known as an essential element for plants growth, being required in
335 enzyme system related to photosystemII electron transport, mitochondria and chloroplast reaction,
336 carbohydrate metabolism, cell wall lignification, and protein synthesis (Yruela 2009). Cu
337 concentrations in the roots and leaves of mangrove plants measured in the present study were in
338 the range of those in other mangroves (Bayen 2012) such as in China (He et al. 2014), in southern
339 Brazil (Madi et al. 2015), or in Indonesia (Martuti et al. 2016). It was reported that the Cu
340 partitioning in sediments of the Can Gio mangrove was mainly related to sulphide precipitation
341 and to its complexes with organic compounds (Thanh-Nho et al. 2019). Thus, we suggest that the
342 low Cu concentrations measured in the mangrove roots may be related to Cu association with
343 sulphide, which limit its bioavailability. However, this element presented the second highest values
344 of TF, beside Mn, with TF reaching 1.69 for *Avicennia* mature trees and 3.37 for *Rhizophora*
345 mature ones. MacFarlane and Burchett (2002) showed that Cu can be toxic to plants growth if its
346 concentration in the sediments exceeds 400 $\mu\text{g g}^{-1}$, which is almost 20 fold higher than the
347 concentrations measured in the Can Gio mangrove sediments. Consequently, we suggest that the
348 high values in TF of Cu resulted from specific metabolic requirements (Baker 1981, Dudani et al.
349 2017) as Cu has a direct impact on photosynthesis I, being a constituent of plastocyanin, which is
350 involved in the photosynthetic electron transport chain (Maksymiec 1998).

351 4.1.2. *Non-essential elements: Co, Ni, Cr, As*

352 Non-essential elements such as Co, Ni, Cr, and As mainly accumulated in the roots (Table 1),
353 with low translocation to the leaves (Fig. 3). This phenomenon may result from different
354 mechanisms developed by mangrove plants to prevent the uptake of toxic elements and to limit

355 their transport within the components of plants (Almeida et al. 2006), notably through cell wall
356 immobilization and/or sequestration in the epidermal layers (MacFarlane et al. 2007). They, thus,
357 accumulate in the perennial tissues, especially the roots (Carbonell et al. 1998, Zhou et al. 2011).

358 Concerning Arsenic, some forms can be subject to plant uptakes, involving arsenites-As(III)
359 and arsenates-As(V) (Asher and Reay 1979). However, arsenites can be toxic for radicular
360 membranes because of As reaction with sulfhydryl groups in proteins, causing disruption of roots
361 functions and cellular death (Speer 1973, Wu et al. 2015). In the present study, arsenic
362 concentrations in the roots and leaves of mangrove plants were higher than those of other
363 mangroves like in the Indian Sundarban (Chowdhury et al. 2015), or in the Chinese Futian (He et
364 al. 2014). Arsenic was also characterized by higher BCF in the roots, specifically for the *Avicennia*.
365 These results may be related to the high As concentrations in bioavailable fractions in the Can Gio
366 mangrove sediments (Table SD2). In Vietnam, soils are naturally rich in As (Gustafsson and Tin
367 1994, Nguyen et al. 2016), which may explain its accumulation in mangrove sediments and then
368 its transfer to mangrove roots. We suggest that As may be incorporated into iron plaque at the roots
369 surface, which restricted As transfer to aerial parts. This hypothesis may be supported by a positive
370 correlation between the concentrations of Fe and As in the roots (i.e. $r = 0.92$, Pearson correlation).
371 Like iron, As was more bioavailable beneath the *Rhizophora* stand, where the sediment was more
372 enrich in organic matter and the oxidizable fraction represented up to 36 % of total As
373 concentrations. However, BCF of As was higher for the *Avicennia* than *Rhizophora* roots, which
374 comforts our hypothesis in enhanced plaque formation beneath the *Avicennia* rhizosphere due to
375 specific redox conditions resulting from oxygen release via the roots system. Similar to Fe, As
376 concentrations in sapling's leaves of both *Avicennia* and *Rhizophora* species were higher than
377 mature ones. We suggest that the As incorporated in iron plaque could be a As source for plant

378 uptake from plaque reductive dissolution by bacteria during organic matter decay processes of
379 death roots tissues. Arsenic uptake by mangrove leaves may concomitantly occur to Fe (i.e. a
380 positive correlation was observed between concentration of As and Fe in the leaves, $r = 0.91$,
381 Pearson correlation). Again, the extension of the root system in the upper layer, and possibly the
382 less developed adaptation strategy of saplings may also be responsible for the As transfer to
383 saplings' leaves.

384 Regarding Ni and Cr, their concentrations in the roots and leaves of *Avicennia* and *Rhizophora*
385 in the Can Gio mangrove were in the range of those presented in the review of Lewis et al. (2011)
386 but lower than those measured in the Indian Sundarban mangroves (Chowdhury et al. 2017) or in
387 New Caledonia (Marchand et al. 2016). In the later environments, the authors reported higher Ni
388 concentrations in plants tissues, with values up to $700 \mu\text{g g}^{-1}$ due to high Ni concentrations in
389 mangrove sediments resulting from the proximity of lateritic soils enriched in Ni. Thus, we suggest
390 that the moderate Ni and Cr concentrations in the Can Gio mangrove plants may result from their
391 association to the refractory fraction (i.e. more than 80 %, Table SD2). For plants, Ni is a
392 micronutrient that is required at very low concentrations (Gajewska and Skłodowska 2007), and
393 which can inhibit plant growth at high concentrations (Rao and Sresty 2000). The effect of Ni on
394 plants varies according to plant species as well as the Ni concentration in sediment; one example
395 of its toxic symptom is chlorosis or yellowing of the leaves (Mishra and Kar 1974). Cr is also toxic
396 for plant growth (Shanker et al. 2005). When mangrove seedlings are exposed to excessive Cr
397 concentrations, their roots would be shortened, and their height and biomass would be limited
398 (Fang et al. 2008). However, considering the low BCF and the low concentrations of those trace
399 elements in the Can Gio mangrove plants, their impacts on plant growth are probably limited. Cr
400 concentrations in the roots of *Rhizophora* trees were lower although its concentrations in

401 bioavailable forms were higher than in the *Avicennia* (Table SD2). This phenomenon, supported
402 by higher TF values in the *Rhizophora* trees, may be related to a more intense phytoextraction of
403 *Rhizophora* species. In addition, translocation factor of Cr in the saplings of both *Avicennia* and
404 *Rhizophora* trees were higher than in the mature ones, which may be due to higher allocation of
405 this element to leaf (Weis and Weis 2004) in the saplings. Similar result was also observed for Ni
406 (Fig. 3).

407 Co is a micronutrient, which can be used in redox processes to stabilize molecules through
408 electrostatic interactions as components of various plant enzymes. However in excess, it can be
409 toxic, inducing enzyme modification, disturbing cellular function (Palit et al. 1994). In the present
410 study, Co concentrations in the roots and leaves of the different mangrove species were higher
411 than in Australian mangroves (Nath et al. 2014), in French Guianan mangroves (Marchand et al.
412 2006), or New Caledonian mangroves (Marchand et al. 2016). We suggest that high Co
413 concentrations in the Can Gio mangrove plants may result from its high concentrations in the
414 bioavailable fractions in the sediments, reaching up to 44 % (Table SD2). However, if Co
415 concentrations in the roots were relatively high, TF were lower than 0.5 (Fig. 3), which suggest its
416 limited physiological role and/or the fact that the roots acted as a physical barrier. As a
417 consequence, the present study showed a narrow range of Co level in the leaves (i.e. 0.07 to 1.43
418 $\mu\text{g g}^{-1}$). The BCF values of Co in the tissues of *Avicennia* were much higher than in the *Rhizophora*,
419 which may reflect the specific physiological requirements of the trees, considering that its
420 bioavailability was higher beneath *Rhizophora* stand (Table SD2).

421 4.2. Bioaccumulation of trace elements in snails

422 In the Can Gio mangrove, whatever the snails' species and their habitat, the concentrations of
423 Fe, Mn and Cu were the most abundant in their soft tissues (Table 2). Previous studies showed that

424 soft tissues of mollusks accumulated higher concentrations of Cu and Fe than in the shells because
425 of specific physiological requirements (Szefer et al. 1999, Vukašinović-Pešić et al. 2017, Yap and
426 Cheng 2009). In fact, Fe, Mn, and Cu play important roles in metabolic biomolecules like enzymes
427 or metalloenzymes (Langston et al. 1998, Rainbow 1997). Snails also need elevated amount of Cu,
428 being a constituent of hemocyanin (Dallinger et al. 2005). Conversely, the other elements studied
429 can be toxic, inducing growth retardation, edema and thinning of the shell (Factor and de Chavez
430 2012). Similar results were reported concerning snails and different mollusks in mangroves in
431 Malaysia (Yap and Cheng 2013), in India (Palpandi and Kesavan 2012), in Costa Rica (Vargas et
432 al. 2015), or in Senegal (Sidoumou et al. 2006).

433 We also suggest that the different amounts of trace elements in the food sources and feeding
434 habits of individual snail affect the degree and extent accumulation of trace elemental in their
435 tissues. The snail *C.obtusa*, a sediment eater, exhibited higher trace elements concentrations in the
436 *Rhizophora* stand than in the *Avicennia* one, except for Mn and Co. This could result from higher
437 concentrations of trace elements in the available fractions in sediments beneath the *Rhizophora*
438 stand (Table SD2). We evidenced that because of the enhanced organic enriched-sediment beneath
439 the *Rhizophora* stand, reductive dissolution of Fe-Mn oxihydroxides by bacteria during organic
440 matter decay processes is a source of dissolved trace elements in pore-waters, these elements being
441 more bioavailable (Thanh-Nho et al. 2019). As a consequence, concentrations of trace elements in
442 *C.obtusa* tissues in the present study were far higher than those measured in other mangroves
443 (Joseph and Ramesh 2016) or estuary with strong anthropogenic pressure (Kesavan et al. 2013).
444 Concerning *L.melanostoma*, a leaf eater, we suggest that the higher concentrations of trace
445 elements (except Mn), combined to higher BCF (Fig. 4), in snails tissues living beneath the
446 *Rhizophora* stand compared with the *Avicennia* one (Table 2) is likely related to the higher trace

447 elements concentrations in the *Rhizophora* leaves than the *Avicennia* ones (Table 1). In addition,
448 *L.melanostoma* presented the highest concentrations of Fe and Mn in the tissues compare to other
449 snail species in the Can Gio mangrove. This result suggest that leaf eater is more sensitive to
450 accumulation of trace elements or that trace elements in leaves are more bioavailable. Furthermore,
451 *L.melanostoma* in the present study showed higher Fe and Mn concentrations than those in
452 *L.scabra* collected from a polluted mangrove (De Wolf and Rashid 2008). *C.capucinus*, a predator
453 of barnacles, bivalves and other mollusks (Berandah et al. 2010), presented the highest Cu
454 concentrations, which may result from copper biomagnification via their food chains because Cu
455 is known to be very compatible with protein binding in organisms (Jiang and Qiu 2009). Cu
456 biomagnification in snails body is usually observed, and they are considered as
457 “macroconcentrators” species for Cu (Nica et al. 2012). However, concerning *N.articulata*, the
458 microalgae eater, Mn and Cu concentration in their tissues were lower than those in other species
459 whatever the mangrove stands, and were lower than those measured in *N.articulata* living in other
460 mangroves like in Malaysia (Yap and Cheng 2013) or Southeast coast of India (Palpandi and
461 Kesavan 2012). This result may be related to low Mn and Cu accumulation in microalgae in the
462 Can Gio mangrove. Unfortunately, in the present study, we were not able to measure
463 concentrations of trace elements in microalgae. Considering the key role of microphytobenthos in
464 mangrove trophic food chain (Lee 2008, Raw et al. 2017), we suggest that further investigation
465 should be carried out on accumulation of trace elements in these microalgae.

466 Eventually, among non-essential elements, all snails contained high quantity of As, which can
467 result from its availability in their diet, and snails’ ability to metabolize As and retain it (Kirby et
468 al. 2002, Zhang et al. 2013). Arsenic also presented the highest BCF whatever the snail species
469 (Fig. 4). In the Can Gio mangrove, dissolved As concentrations in pore-waters reached up to almost

470 20 $\mu\text{g L}^{-1}$. In addition at the *Rhizophora* stand, characterized by higher organic content, the organic
471 fraction, being bioavailable, represented up to 36 % of total As concentrations (Table SD2).
472 Khokiattiwong et al. (2009) studied on two mollusks species (*C.capucinus* and *Telescopium*
473 *telescopium*), and they showed that *C.capucinus*, as a predator, contained more As than the detritus
474 and algae eater species, *T.telescopium*. We also observed that *C.capucinus* contained the highest
475 As concentrations in their body (2 to 4 fold higher level tissues than other snail species) in the Can
476 Gio mangrove, which could be related to their predator diet and magnification along food chain
477 (Goessler et al. 1997).

478 According to Dallinger (1993), snail tissues can be classified in macroconcentrators (BCF >
479 2), microconcentrators ($1 < \text{BCF} < 2$) or deconcentrators (BCF < 1). Thus, in the present study, the
480 *L.melanostoma* and *C.obtusa* can be classified as macroconcentrators for Cu. The *L.melanostoma*
481 was macroconcentrators for As, Co and Ni. Conversely, all snails were deconcentrators to
482 microconcentrators for Mn and Fe. Due to negative effects of toxic trace elements on community
483 structure, gender, size of snails (Amin et al. 2009, Yap and Cheng 2013), the obtained results in
484 the present study provided a further claim that these snails can be used as good biomonitors for
485 environmental quality (Samsi et al. 2017).

486 5. Conclusions

487 Trace elements accumulation in the tissues of mangrove plants and snails studied in the Can
488 Gio mangrove reflected their concentrations in the sediment, their bioavailability, and specific
489 adaptation strategies or physiological processes of the biota. Clear differences between the two
490 stands were evidenced, notably because of different sediment geochemistry. The main conclusions
491 concerning mangrove trees can be summarized as follow:

- 492 - Fe and As were characterized by low translocation to the leaves of the *Avicennia* and
493 *Rhizophora* trees. We suggest that the formation of iron plaque on roots by oxygen release
494 in the rhizosphere may be a key factor preventing this translocation. Consequently,
495 mangroves roots can be considered as “phytostabilizers”, immobilizing trace elements
496 and limiting their transfer in the environment.
- 497 - Co exhibited higher concentrations in roots and leaves of the Can Gio mangrove trees than
498 the world average. However, this element was characterized by low BCF and TF, which
499 may be related to its high concentrations in available forms in the sediment
- 500 - High translocation factors for Mn and Cu may possibly result from physiological
501 requirements, both elements being useful in photosynthetic processes, but also from
502 elevated bioavailability in the sediment. We also suggest that the *Rhizophora* trees have a
503 high potential for Mn “phytoextraction”.
- 504 - Ni and Cr were characterized by low BCF and TF whatever the mangrove plants, which
505 may result from their low bioavailability in the sediment and from the limited physiological
506 roles of those elements.
- 507 - Consequently, we suggest that the mangrove trees species studied can act as sinks when
508 elements are stored in their root systems, or sources for the trophic chain when they
509 accumulate in the leaves. Therefore, studies regarding the turnover of trace elements from
510 decomposing of leaf litter should be developed to get a better understanding of the cycling
511 of trace elements in mangroves ecosystem.

512 The main conclusions concerning mangrove snails can be summarized as follow:

- 513 - The great variability in concentrations of studied trace elements in the various snails
514 indicated that bioaccumulation of trace elements depended not only on characteristics of

515 trace elements (i.e. physiological properties and biological functions), and metabolic
516 requirements of each species but also on available concentrations of these trace elements
517 in their feeding sources.

518 - Fe, Mn and Cu, which are essential elements, presented higher concentrations in all snails
519 tissues than As, Ni, Cr and Co, which can be toxic.

520 - Most of the snails living in the *Rhizophora* stand presented higher levels of trace elements
521 concentrations (except Mn and Co) than those living in the *Avicennia* stand, specifically
522 for *L.melanostoma* and *C.obtusa*, due to higher concentrations of trace elements in the
523 leaves or in the bioavailable fractions in the sediment.

524 - Concerning the predators, *C.capucinus*, possible biomagnification of Cu via the food chain
525 induced elevated Cu concentrations in their tissues.

526 - In this study, we also measured that all snails contained high amount of As, most probably
527 because it is highly bioavailable in mangrove sediments.

528 - Eventually, most snails can be classified as “maroconcentrators” of some trace elements
529 such as Cu, As, Co and Ni. Because of their potential toxic effects, a further study should
530 be performed to assess the possibility of using snails as bioindicators of environmental
531 quality. In addition, it would be relevant to compare trace elements dynamics between
532 seasons. We suggest that during the dry season, lower elements inputs and lower sediment
533 reactivity may induce lower accumulation in mangrove plants and snails tissues.

534 **Acknowledgements**

535 This research is funded by Vietnam National University Ho Chi Minh City (VNU-HCM) under
536 grand number C2016-18-07. Nho received a BEST grant from IRD (France).

537

538 **References**

- 539 Abohassan R. A. (2013). Heavy metal pollution in *Avicennia marina* Mangrove systems on the Red Sea
540 Coast of Saudi Arabia. *Journal of King Abdulaziz University: Metrology, Environment and Arid Land*
541 *Agricultural Sciences* 24(1), 35-53.
- 542 Almeida C. M. R., Mucha A. P. and Vasconcelos M. T. S. (2006). Comparison of the role of the sea club-
543 rush *Scirpus maritimus* and the sea rush *Juncus maritimus* in terms of concentration, speciation and
544 bioaccumulation of metals in the estuarine sediment. *Environmental Pollution* 142(1), 151-159.
- 545 Alongi D. M., Wattayakorn G., Boyle S., Tirendi F., Payn C. and Dixon P. (2004). Influence of roots and
546 climate on mineral and trace element storage and flux in tropical mangrove soils. *Biogeochemistry* 69,
547 105–123.
- 548 Amin B., Ismail A., Arshad A., Yap C. K. and Kamarudin M. S. (2009). Gastropod assemblages as indicators
549 of sediment metal contamination in mangroves of Dumai, Sumatra, Indonesia. *Water, Air, and Soil*
550 *Pollution* 201(1-4), 9-18.
- 551 Asher C. and Reay P. (1979). Arsenic uptake by barley seedlings. *Functional Plant Biology* 6(4), 459-466.
- 552 Babukutty Y. and Chacko J. (1995). Chemical partitioning and bioavailability of lead and nickel in an
553 estuarine system. *Environmental Toxicology and Chemistry* 4(13), 427-434.
- 554 Babut M., Mourier B., Desmet M., Simonnet-Laprade C., Labadie P., Budzinski H., De Alencastro L. F., Tu
555 T. A., Strady E. and Gratiot N. (2019). Where has the pollution gone? A survey of organic contaminants
556 in Ho Chi Minh city / Saigon River (Vietnam) bed sediments. *Chemosphere* 217, 261-269.
- 557 Baker A. J. (1981). Accumulators and excluders-strategies in the response of plants to heavy metals.
558 *Journal of plant nutrition* 3(1-4), 643-654.
- 559 Bayen S. (2012). Occurrence, bioavailability and toxic effects of trace metals and organic contaminants in
560 mangrove ecosystems: a review. *Environment International* 48, 84-101.
- 561 Berandah F. E., Kong Y. C. and Ismail A. (2010). Bioaccumulation and distribution of heavy metals (Cd, Cu,
562 Fe, Ni, Pb and Zn) in the different tissues of *Chicoreus capucinus lamarck* (Mollusca: Muricidae)
563 collected from Sungai Janggut, Kuala Langat, Malaysia. *Environmental Asia* 3(1), 65-71.
- 564 Blasco F., Aizpuru M. and Gers C. (2001). Depletion of the mangroves of Continental Asia. *Wetlands*
565 *Ecology and Management* 9, 245-256.
- 566 Carbonell A., Aarabi M., DeLaune R., Gambrell R. and Patrick Jr W. (1998). Arsenic in wetland vegetation:
567 availability, phytotoxicity, uptake and effects on plant growth and nutrition. *Science of The Total*
568 *Environment* 217(3), 189-199.
- 569 Chaudhuri P., Nath B. and Birch G. (2014). Accumulation of trace metals in grey mangrove *Avicennia*
570 *marina* fine nutritive roots: the role of rhizosphere processes. *Marine Pollution Bulletin* 79(1-2), 284-
571 292.
- 572 Chowdhury R., Favas P. J. C., Jonathan M. P., Venkatachalam P., Raja P. and Sarkar S. K. (2017). Bioremoval
573 of trace metals from rhizosediment by mangrove plants in Indian Sundarban Wetland. *Marine*
574 *Pollution Bulletin* 124(2), 1078-1088.
- 575 Chowdhury R., Favas P. J. C., Pratas J., Jonathan M. P., Ganesh P. S. and Sarkar S. K. (2015). Accumulation
576 of Trace Metals by Mangrove Plants in Indian Sundarban Wetland: Prospects for Phytoremediation.
577 *International Journal of Phytoremediation* 17(9), 885-894.
- 578 Cormier-Salem M.-C., Van Trai N., Burgos A., Durand J.-D., Bettarel Y., Klein J., Duc Huy H. and Panfili J.
579 (2017). The mangrove's contribution to people: Interdisciplinary pilot study of the Can Gio Mangrove
580 Biosphere Reserve in Viet Nam. *Comptes Rendus Geoscience* 349(6-7), 341-350.
- 581 Costa-Boddeker S., Hoelzmann P., Thuyen L. X., Huy H. D., Nguyen H. A., Richter O. and Schwalb A. (2017).
582 Ecological risk assessment of a coastal zone in Southern Vietnam: Spatial distribution and content of
583 heavy metals in water and surface sediments of the Thi Vai Estuary and Can Gio Mangrove Forest.
584 *Marine Pollution Bulletin* 114(2), 1141-1151.

585 Cox R. and Hutchinson T. (1981). Multiple and cotolerance to metals in the grass *Despitosa* Beauv from
586 the sudbury smelting area. *Journal of plant nutrition* 3, 731-741.

587 Dallinger R. (1993). Strategies of metal detoxification in terrestrial invertebrates. *Ecotoxicology of metals*
588 in invertebrates 245.

589 Dallinger R., Chabicovsky M., Hödl E., Prem C., Hunziker P. and Manzl C. (2005). Copper in *Helix pomatia*
590 (*Gastropoda*) is regulated by one single cell type: differently responsive metal pools in rhogocytes.
591 *American Journal of Physiology-Regulatory, Integrative and Comparative Physiology* 289(4), R1185-
592 R1195.

593 De Wolf H. and Rashid R. (2008). Heavy metal accumulation in *Littoraria scabra* along polluted and pristine
594 mangrove areas of Tanzania. *Environmental Pollution* 152(3), 636-643.

595 Dias H. Q. and Nayak G. N. (2016). Geochemistry and bioavailability of mudflats and mangrove sediments
596 and their effect on bioaccumulation in selected organisms within a tropical (Zuari) estuary, Goa, India.
597 *Marine Pollution Bulletin* 105(1), 227-236.

598 Dudani S. N., Lakhmapurkar J., Gavali D. and Patel T. (2017). Heavy Metal Accumulation in The Mangrove
599 Ecosystem of South Gujarat Coast, India. *Turkish Journal of Fisheries and Aquatic Sciences* 17(4), 755-
600 766.

601 Eichhorst T. (2016). *Neritidae of the world. Vol. 1.* ConchBooks, Hackenheim, Germany.

602 Factor C. J. B. and de Chavez E. R. C. (2012). Toxicity of arsenic, aluminum, chromium and nickel to the
603 embryos of the freshwater snail, *Radix quadrasi* von Möellendorf 1898. *Philippine Journal of Science*
604 141(2), 207-216.

605 Fang Y., Zheng W., Wan Y., Chen C. and Sheng H. (2008). Effects of chromium (III) on the seedling growth
606 of mangrove species *Avicennia marina* [J]. *Chinese Journal of Ecology* 3, 021.

607 Gajewska E. and Skłodowska M. (2007). Effect of nickel on ROS content and antioxidative enzyme activities
608 in wheat leaves. *Biometals* 20(1), 27-36.

609 Goessler W., Maher W., Irgolic K., Kuehnelt D., Schlagenhaufen C. and Kaise T. (1997). Arsenic compounds
610 in a marine food chain. *Fresenius' journal of analytical chemistry* 359(4-5), 434-437.

611 Gustafsson J. P. and Tin N. T. (1994). Arsenic and selenium in some Vietnamese acid sulphate soils. *Science*
612 of The Total Environment 151(2), 153-158.

613 Ha T. H., Marchand C., Aimé J., Dang H. N., Phan N. H., Nguyen X. T. and Nguyen T. K. C. (2018).
614 Belowground carbon sequestration in a mature planted mangroves (Northern Viet Nam). *Forest*
615 *Ecology and Management* 407, 191-199.

616 He B., Li R., Chai M. and Qiu G. (2014). Threat of heavy metal contamination in eight mangrove plants
617 from the Futian mangrove forest, China. *Environmental Geochemistry and Health* 36(3), 467-476.

618 Ho Chi Minh City Statistical Yearbook, 2016. HCMC Statistical Office. Thanh Nien
619 Publisher.

620 Jiang J. and Qiu H. (2009). Assessment of content of heavy metals in seafood from sea area of Guangdong
621 province. *Journal of Environment and Health* 26(9), 814-816.

622 Jones Jr J. B., Wolf B. and Mills H. A. (1991). *Plant analysis handbook. A practical sampling, preparation,*
623 *analysis, and interpretation guide,* Micro-Macro Publishing, Inc.

624 Joseph T. and Ramesh K. (2016). Heavy Metal Risk Assessment in Bhavanapadu Creek Using Three
625 Potamidid Snails-*Telescopium telescopium*, *Cerithidea obtusa* and *Cerithidea cingulata*. *Journal of*
626 *Environmental and Analytical Toxicology* 6(385), 2161-0525.1000385.

627 Kathiresan K., Saravanakumar K. and Mullai P. (2014). Bioaccumulation of trace elements by *Avicennia*
628 *marina*. *Journal of Coastal Life Medicine* 2(11), 888-894.

629 Kesavan K., Murugan A., Venkatesan V. and Kumar V. (2013). Heavy metal accumulation in molluscs and
630 sediment from Uppanar estuary, southeast coast of India. *Thalassas* 29(2), 15-21.

631 Khokiattiwong S., Kornkanitnan N., Goessler W., Kokarnig S. and Francesconi K. A. (2009). Arsenic
632 compounds in tropical marine ecosystems: similarities between mangrove forest and coral reef.
633 Environmental Chemistry 6(3), 226-234.

634 Kirby J., Maher W., Chariton A. and Krikowa F. (2002). Arsenic concentrations and speciation in a
635 temperate mangrove ecosystem, NSW, Australia. Applied Organometallic Chemistry 16(4), 192-201.

636 Koch M. S. and Mendelssohn I. (1989). Sulphide as a soil phytotoxin: differential responses in two marsh
637 species. The Journal of Ecology, 565-578.

638 Komiyama A., Havanond S., Srisawatt W., Mochida Y., Fujimoto K., Ohnishi T., Ishihara S. and Miyagi T.
639 (2000). Top/root biomass ratio of a secondary mangrove (*Ceriops tagal* (Perr.) CB Rob.) forest. Forest
640 Ecology and Management 139(1-3), 127-134.

641 Křibek B., Mihaljevič M., Sracek O., Knésl I., Ettler V. and Nyambe I. (2011). The extent of arsenic and of
642 metal uptake by aboveground tissues of *Pteris vittata* and *Cyperus involucratus* growing in copper-
643 and cobalt-rich tailings of the Zambian Copperbelt. Archives of Environmental Contamination and
644 Toxicology 61(2), 228-242.

645 Kuenzer C. and Tuan V. Q. (2013). Assessing the ecosystem services value of Can Gio Mangrove Biosphere
646 Reserve: Combining earth-observation- and household-survey-based analyses. Applied Geography 45,
647 167-184.

648 Langston W. J., Bebianno M. J. and Burt G. R. (1998). Metal handling strategies in molluscs. Metal
649 metabolism in aquatic environments, Springer, 219-283.

650 Lee O. H., Williams G. A. and Hyde K. D. (2001). The diets of *Littoraria arduiniana* and *L. melanostoma* in
651 Hong Kong mangroves. Journal of the Marine Biological Association of the United Kingdom 81(6), 967-
652 973.

653 Lee S. Y. (2008). Mangrove macrobenthos: Assemblages, services, and linkages. Journal of Sea Research
654 59(1-2), 16-29.

655 Lewis M., Pryor R. and Wilking L. (2011). Fate and effects of anthropogenic chemicals in mangrove
656 ecosystems: a review. Environmental Pollution 159(10), 2328-2346.

657 Luong N. V., Tateishi R. and Hoan N. T. (2015). Analysis of an impact of succession in mangrove forest
658 association using remote sensing and GIS technology. Journal of Geography and Geology 7(1), 106.

659 MacFarlane G. and Burchett M. (2002). Toxicity, growth and accumulation relationships of copper, lead
660 and zinc in the grey mangrove *Avicennia marina* (Forsk.) Vierh. Marine Environmental Research 54(1),
661 65-84.

662 MacFarlane G., Pulkownik A. and Burchett M. (2003). Accumulation and distribution of heavy metals in
663 the grey mangrove, *Avicennia marina* (Forsk.) Vierh.: biological indication potential. Environmental
664 Pollution 123(1), 139-151.

665 MacFarlane G. R., Koller C. E. and Blomberg S. P. (2007). Accumulation and partitioning of heavy metals in
666 mangroves: a synthesis of field-based studies. Chemosphere 69(9), 1454-1464.

667 Machado W., Gueiros B. B., Lisboa-Filho S. D. and Lacerda L. D. (2005). Trace metals in mangrove seedlings:
668 role of iron plaque formation. Wetlands Ecology and Management 13(2), 199-206.

669 Madi A. P. L. M., Boeger M. R. T. and Reissmann C. B. (2015). Distribution of Cu, Fe, Mn, and Zn in Two
670 Mangroves of Southern Brazil. Brazilian Archives of Biology and Technology 58(6), 970-976.

671 Maksymiec W. (1998). Effect of copper on cellular processes in higher plants. Photosynthetica 34(3), 321-
672 342.

673 Marchand C., Baltzer F., Lallier-Vergès E. and Albéric P. (2004). Pore-water chemistry in mangrove
674 sediments: relationship with species composition and developmental stages (French Guiana). Marine
675 geology 208(2-4), 361-381.

676 Marchand C., Fernandez J. M. and Moreton B. (2016). Trace metal geochemistry in mangrove sediments
677 and their transfer to mangrove plants (New Caledonia). Science of The Total Environment 562, 216 -
678 227.

679 Marchand C., Lallier-Vergès E., Baltzer F., Albéric P., Cossa D. and Baillif P. (2006). Heavy metals
680 distribution in mangrove sediments along the mobile coastline of French Guiana. *Marine Chemistry*
681 98(1), 1-17.

682 Marchiol L., Assolari S., Sacco P. and Zerbi G. (2004). Phytoextraction of heavy metals by canola (*Brassica*
683 *napus*) and radish (*Raphanus sativus*) grown on multicontaminated soil. *Environmental Pollution*
684 132(1), 21-27.

685 Martuti N. K. T., Widianarko B. and Yulianto B. (2016). Copper Accumulation on *Avicennia marina* In Tapak,
686 Tugurejo, Semarang, Indonesia. *Waste Technology* 4(1), 40-45.

687 McEvoy J. P. and Brudvig G. W. (2006). Water-splitting chemistry of photosystem II. *Chemical reviews*
688 106(11), 4455-4483.

689 Millaleo R., Reyes-Díaz M., Ivanov A., Mora M. and Alberdi M. (2010). Manganese as essential and toxic
690 element for plants: transport, accumulation and resistance mechanisms. *Journal of soil science and*
691 *plant nutrition* 10(4), 470-481.

692 Mishra D. and Kar M. (1974). Nickel in plant growth and metabolism. *The Botanical Review* 40(4), 395-
693 452.

694 Moorhead K. and Reddy K. R. (1988). Oxygen transport through selected aquatic macrophytes. *Journal of*
695 *Environmental Quality* 17(1), 138-142.

696 Nath B., Birch G. and Chaudhuri P. (2014). Assessment of sediment quality in *Avicennia marina*-dominated
697 embayments of Sydney Estuary: The potential use of pneumatophores (aerial roots) as a bio-indicator
698 of trace metal contamination. *Science of The Total Environment* 472, 1010-1022.

699 Nguyen V. T., Ozaki A., Nguyen Tho H., Nguyen Duc A., Tran Thi Y. and Kurosawa K. (2016). Arsenic and
700 Heavy Metal Contamination in Soils under Different Land Use in an Estuary in Northern Vietnam.
701 *International Journal of Environmental Research and Public Health* 13(11).

702 Nica D. V., Bura M., Gergen I., Harmanescu M. and Bordean D.-M. (2012). Bioaccumulative and
703 conchological assessment of heavy metal transfer in a soil-plant-snail food chain. *Chemistry Central*
704 *Journal* 6(1), 55.

705 Noël V., Morin G., Juillot F., Marchand C., Brest J., Bargar J. R., Muñoz M., Marakovic G., Ardo S. and Brown
706 G. E. (2015). Ni cycling in mangrove sediments from New Caledonia. *Geochimica et Cosmochimica*
707 *Acta* 169, 82-98.

708 Palit S., Sharma A. and Talukder G. (1994). Effects of cobalt on plants. *The Botanical Review* 60(2), 149-
709 181.

710 Palpandi C. and Kesavan K. (2012). Heavy metal monitoring using *Nerita crepidularia*-mangrove mollusc
711 from the Vellar estuary, Southeast coast of India. *Asian Pacific Journal of Tropical Biomedicine* 2(1),
712 S358-S367.

713 Rainbow P. S. (1997). Trace metal accumulation in marine invertebrates: marine biology or marine
714 chemistry? *Journal of the Marine Biological Association of the United Kingdom* 77(1), 195-210.

715 Rao K. M. and Sresty T. (2000). Antioxidative parameters in the seedlings of pigeonpea (*Cajanus cajan* (L.)
716 Millspaugh) in response to Zn and Ni stresses. *Plant Science* 157(1), 113-128.

717 Raw J. L., Perissinotto R., Miranda N. A. F. and Peer N. (2017). Feeding dynamics of *Terebralia palustris*
718 (Gastropoda: Potamididae) from a subtropical mangrove ecosystem. *Molluscan Research* 37(4), 258-
719 267.

720 Reed-Judkins D. K., Farris J. L., Cherry D. S. and Cairns J. (1997). Foodborne uptake and sublethal effects
721 of copper and zinc to freshwater snails. *Hydrobiologia* 364(2-3), 105-118.

722 Samsi A. N., Asaf R., Sahabuddin S., Santi A. and Wamnebo M. I. (2017). Gastropods As A Bioindicator and
723 Biomonitoring Metal Pollution. *Aquacultura Indonesiana* 18(1), 1-8.

724 Scholander P., Van Dam L. and Scholander S. I. (1955). Gas exchange in the roots of mangroves. *American*
725 *Journal of Botany*, 92-98.

726 Shanker A. K., Cervantes C., Loza-Tavera H. and Avudainayagam S. (2005). Chromium toxicity in plants.
727 Environment International 31(5), 739-753.

728 Sidoumou Z., Gnassia-Barelli M., Siau Y., Morton V. and Romeo M. (2006). Heavy metal concentrations in
729 molluscs from the Senegal coast. Environment International 32(3), 384-387.

730 Speer H. L. (1973). The effect of arsenate and other inhibitors on early events during the germination of
731 lettuce seeds (*Lactuca sativa* L.). Plant physiology 52(2), 142-146.

732 Strady E., Dang V. B., Nemery J., Guedron S., Dinh Q. T., Denis H. and Nguyen P. D. (2017). Baseline
733 seasonal investigation of nutrients and trace metals in surface waters and sediments along the Saigon
734 River basin impacted by the megacity of Ho Chi Minh (Vietnam). Environmental Science and Pollution
735 Research International 24(4), 3226–3243.

736 Suwanjarat J. and Klepal W. (2001). Ultrastructural Investigations of Euspermatogenesis and
737 Euspermatozoa in *Cerithidea obtusa* (Lamarck 1822)(Caenogastropoda: Potamididae). Marine
738 Ecology 22(1-2), 23-34.

739 Szefer P., Ali A., Ba-Haroon A., Rajeh A., Geldon J. and Nabrzyski M. (1999). Distribution and relationships
740 of selected trace metals in molluscs and associated sediments from the Gulf of Aden, Yemen.
741 Environmental Pollution 106(3), 299-314.

742 Tam N. F. Y. and Wong Y. S. (2000). Spatial variation of heavy metals in surface sediments of Hong Kong
743 mangrove swamps. Environmental Pollution 110(2), 195-205.

744 Tamooh F., Van den Meersche K., Meysman F., Marwick T. R., Borges A. V., Merckx R., Dehairs F., Schmidt
745 S., Nyunja J. and Bouillon S. (2012). Distribution and origin of suspended matter and organic carbon
746 pools in the Tana River Basin, Kenya. Biogeosciences 9(8), 2905-2920.

747 Tan K.-S. and Oh T. M. (2003). Feeding habits of *Chicoreus capucinus* (Neogastropoda: Muricidae) in a
748 Singapore mangrove. Bollettino Malacologico 38, 43-50.

749 Tan K. (2008). Mudflat predation on bivalves and gastropods by *Chicoreus capucinus* (Neogastropoda:
750 Muricidae) at Kungkrabaen Bay, Gulf of Thailand. Raffles Bulletin of Zoology Supplement 18, 235-245.

751 Thanh-Nho N., Marchand C., Strady E., Vinh T.-V. and Nhu-Trang T.-T. (2019). Metals geochemistry and
752 ecological risk assessment in a tropical mangrove (Can Gio, Vietnam). Chemosphere 219, 365-382.

753 Thanh-Nho N., Strady E., Nhu-Trang T. T., David F. and Marchand C. (2018). Trace metals partitioning
754 between particulate and dissolved phases along a tropical mangrove estuary (Can Gio, Vietnam).
755 Chemosphere 196, 311-322.

756 Tuan V. Q. and Kuenzer C. (2012). Can Gio Mangrove Biosphere Reserve Evaluation 2012: Current status,
757 Dynamics and Ecosystem Services Ha Noi, Viet Nam: IUCN, 102 pp.

758 Tue N. T., Hamaoka H., Sogabe A., Quy T. D., Nhuan M. T. and Omori K. (2012). Food sources of macro-
759 invertebrates in an important mangrove ecosystem of Vietnam determined by dual stable isotope
760 signatures. Journal of Sea Research 72, 14-21.

761 Vargas J. A., Acuña-González J., Gómez E. and Molina J. (2015). Metals in coastal mollusks of Costa Rica.
762 Revista de biología tropical 63(4), 1007-1019.

763 Verkleij J. and Schat H. (1990). Mechanisms of metal tolerance in higher plants, CRC Press, Boca Raton, FL.

764 Vo P. L. (2007). Urbanization and water management in Ho Chi Minh City, Vietnam-issues, challenges and
765 perspectives. GeoJournal 70(1), 75-89.

766 Vukašinić-Pešić V., Blagojević N., Vukanović S., Savić A. and Pešić V. (2017). Heavy Metal Concentrations
767 in Different Tissues of the Snail *Viviparus mamillatus* (Küster, 1852) from Lacustrine and Riverine
768 Environments in Montenegro. Turkish Journal of Fisheries and Aquatic Sciences 17(3), 557-563.

769 Wang S., Yang Z. and Xu L. (2003). Mechanisms of copper toxicity and resistance of plants. Ecology and
770 Environment 3, 336-341.

771 Weis J. S. and Weis P. (2004). Metal uptake, transport and release by wetland plants: implications for
772 phytoremediation and restoration. Environment International 30(5), 685-700.

773 Wildsingapore. <http://www.wildsingapore.com/wildfacts/mollusca/gastropoda/neritidae/articulata.htm>

774 Wu G.-R., Hong H.-L. and Yan C.-L. (2015). Arsenic accumulation and translocation in mangrove (*Aegiceras*
775 *corniculatum* L.) grown in Arsenic Contaminated Soils. *International Journal of Environmental*
776 *Research and Public Health* 12(7), 7244-7253.

777 Yang Q., Tam N. F., Wong Y. S., Luan T., Su W., Lan C., Shin P. K. and Cheung S. G. (2008). Potential use of
778 mangroves as constructed wetland for municipal sewage treatment in Futian, Shenzhen, China.
779 *Marine Pollution Bulletin* 57(6-12), 735-743.

780 Yap C. and Cheng W. (2013). Distributions of heavy metal concentrations in different tissues of the
781 mangrove snail *Nerita lineata*. *Sains Malaysiana* 42(5), 597-603.

782 (2009). Heavy metal concentrations in *Nerita lineata*: the potential as a biomonitor for heavy metal
783 bioavailability and contamination in the tropical intertidal area. *Marine Biodiversity Records* 2.

784 Yruela I. (2009). Copper in plants: acquisition, transport and interactions. *Functional Plant Biology* 36(5),
785 409-430.

786 Zhang W., Wang W.-X. and Zhang L. (2013). Arsenic speciation and spatial and interspecies differences of
787 metal concentrations in mollusks and crustaceans from a South China estuary. *Ecotoxicology* 22(4),
788 671-682.

789 Zhou Q., Zhang J., Fu J., Shi J. and Jiang G. (2008). Biomonitoring: an appealing tool for assessment of
790 metal pollution in the aquatic ecosystem. *Analytica Chimica Acta* 606(2), 135-150.

791 Zhou Y.-w., Peng Y.-s., Li X.-l. and Chen G.-z. (2011). Accumulation and partitioning of heavy metals in
792 mangrove rhizosphere sediments. *Environmental Earth Sciences* 64(3), 799-807.

793