



HAL
open science

Metals geochemistry and ecological risk assessment in a tropical mangrove (Can Gio, Vietnam)

Nguyen Thành-Nho, Cyril Marchand, Emilie Strady, Truong-Van Vinh,
Tran-Thi Nhu-Trang

► **To cite this version:**

Nguyen Thành-Nho, Cyril Marchand, Emilie Strady, Truong-Van Vinh, Tran-Thi Nhu-Trang. Metals geochemistry and ecological risk assessment in a tropical mangrove (Can Gio, Vietnam). *Chemosphere*, 2019, 219, pp.365-382. 10.1016/j.chemosphere.2018.11.163 . hal-02297576

HAL Id: hal-02297576

<https://hal.science/hal-02297576>

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 **Metals geochemistry and ecological risk assessment**
2 **in a tropical mangrove (Can Gio, Vietnam)**

3
4
5 **Nguyen Thanh-Nho^{1,2}, Cyril Marchand^{1,2,*}, Emilie Strady^{3,4}, Truong-Van Vinh^{2,5}, Tran-Thi Nhu-Trang^{1,6}**

6 *1. Department of Analytical Chemistry, Faculty of Chemistry, University of Science, Vietnam National University- Ho Chi Minh*
7 *City*

8 *2. IMPMC, Institut de Recherche pour le Développement (IRD), UPMC, CNRS, MNHN, Nouméa New Caledonia, France.*

9 *3. Univ. Grenoble Alpes, CNRS, IRD, Grenoble INP*, IGE, F-38000 Grenoble, France*

10 *4. CARE-HCMUT, Ho Chi Minh City, Vietnam*

11 *5. Nong Lam University, Ho Chi Minh City, Vietnam*

12 *6. Faculty of Chemical Engineering and Food Technology, Nguyen Tat Thanh University, Vietnam.*

13 ** Corresponding author: Cyril.marchand@ird.fr*

29 **ABSTRACT**

30 Mangrove sediments act as natural biogeochemical reactors, modifying metals partitioning after
31 their deposition. The objectives of the present study were: to determine distribution and partitioning
32 of metals (Fe, Mn, Ni, Cr, Cu, Co and As) in sediments and pore-waters of Can Gio Mangrove; and
33 to assess their ecological risks based on Risk Assessment Code. Three cores were collected within
34 a mudflat, beneath *Avicennia alba* and *Rhizophora apiculata* stands. We suggest that most metals
35 had a natural origin, being deposited in the mangrove mainly as oxyhydroxides derived from the
36 upstream lateritic soils. This hypothesis could be supported by the high proportion of metals in the
37 residual fraction (mean values (%): 71.9, 30.7, 80.7, 80.9, 67.9, 53.4 and 66.5 for Fe, Mn, Ni, Cr,
38 Cu, Co, and As respectively, in the mudflat). The enrichment of mangrove-derived organic matter
39 from the mudflat to the *Rhizophora* stand (i.e. up to 4.6 % of TOC) played a key role in controlling
40 metals partitioning. We suggest that dissolution of Fe and Mn oxyhydroxides in reducing condition
41 during decomposition of organic matter may be a major source of dissolved metals in pore-waters.
42 Only Mn exhibited a potential high risk to the ecosystem. Most metals stocks in the sediments were
43 higher in the *Avicennia* stand than the *Rhizophora* stand, possibly because of enhanced dissolution
44 of metal bearing phases beneath later one. In a context of enhanced mangrove forests destruction,
45 this study provides insights on the effects of perturbation and oxidation of sediments on metal
46 release to the environment.

47

48 *Keywords: Mangrove; Sequential extraction; Metal geochemistry; Partitioning; Ecological risk;*
49 *Vietnam.*

50

51

52

53 **1. Introduction**

54 Mangrove ecosystems are closely tied to anthropogenic activities. Due to their specific
55 characteristics, i.e. their richness in fine particles and organic matter, mangrove sediments can act
56 as natural sinks for metals originating from natural and anthropized watersheds (Clark et al. 1998,
57 Marchand et al. 2011a, Tam and Wong 1996). In mangrove sediments, the partitioning of metals
58 between different phases, i.e. organic matter, iron-manganese oxyhydroxides, sulphides, carbonates
59 (Tam and Wong 1996, Zhang et al. 2014) is strongly influenced by sediment characteristics, i.e.
60 pH, redox potential, particle size (Otero et al. 2009). Accordingly, the mangrove zonation, the
61 hydrological conditions and the presence of different tree species will influence the above
62 characteristics (Marchand et al. 2004, Marchand et al. 2003) which will in turn influence metal
63 accumulation and speciation in the sediments. In fact metals can be released from sediments into
64 pore-waters in suboxic/anoxic conditions by the dissolution of bearing phases (Chakraborty et al.
65 2016, Marchand et al. 2012) during diagenetic processes, which can subsequently re-precipitate
66 with another bearing phase (Noël et al. 2014). Then, dissolved metals can be exported to adjacent
67 surface waters (Holloway et al. 2016, Sanders et al. 2012) and also be transferred to mangrove's
68 living organism such as fishes, crabs, snails and mangrove's leaves and roots (Chakraborty et al.
69 2014, De Wolf and Rashid 2008, Parvaresh et al. 2011). In a context of enhanced mangrove forests
70 destruction at a worldwide scale i.e. at a rate close to 1 % per year (Duke et al. 2007), the
71 perturbation and oxidation of sediments can induce the dissolution of sulfide minerals, which can
72 result in sediment acidification (Dent 1986) and can enhance the release of metals from the
73 sediment into the aquatic environment. Thus, the understanding of metal behaviors in mangrove
74 ecosystems is highly relevant for the environment itself but also for human, as mangroves provide
75 important ecosystem services (Lee et al. 2014, M. Brander et al. 2012).

76 In Vietnam, large mangrove areas were devastated by herbicide mixture during the war in the
77 70's, and the Can Gio Mangrove was one of the most heavily sprayed areas (Hong 2001, Ross
78 1975). However, mangrove restoration efforts have been realized and almost 40 years after, the
79 rehabilitated mangrove is now more diverse in community structure than prior to the war (Hong

80 2001). Nowadays, this mangrove is largely dominated by *Rhizophora apiculata* and *Avicennia alba*
81 species, the latter one developing at lower elevation along tidal creek. The Can Gio Mangrove is
82 located downstream Ho Chi Minh City, the economic capital of Vietnam with almost 10 million
83 inhabitants. Mangrove river network acts as a unique gate for water outlet from Ho Chi Minh City
84 to the South China Sea. Industrial activities, economic developments and rapid population growth
85 are inducing high pressure on water and sediment quality of the adjacent river, the Sai Gon River
86 (Babut et al. 2019, Nguyen et al. 2018, Strady et al. 2017). Strady et al. (2017) stated that the main
87 rivers and canals in the city were moderately contaminated by major metal(oid)s. Recently, Thanh-
88 Nho et al. (2018) highlighted that metals can be transferred over long distance from upstream to
89 the mangrove forest, and that elevated inputs of metals in the estuary were the result of enhanced
90 runoff and soil leaching during the monsoon season. Consequently, taking into account the
91 mangrove specific geochemical characteristics, the lack of wastewater treatment plants in emerging
92 countries as Vietnam and the important local ecosystem services provided by the Can Gio
93 Mangrove (Cormier-Salem et al. 2017, Kuenzer and Tuan 2013), more attention should be paid on
94 metals distribution, speciation, bioaccumulation and transfer in mangrove ecosystems. We note that
95 so far, in the Can Gio Mangrove, only total metal concentrations in surface water, suspended
96 sediment and surface sediment were investigated (Costa-Boddeker et al. 2017, Thanh-Nho et al.
97 2018) and the evaluation of the metal speciation in the sediment has not yet been assessed. It can
98 be evaluated by the technique of sequential extraction (Tessier et al. 1979) which provides a
99 classification of metals bound to different geochemical fractions. This technique also allow to
100 characterize the diagenetic processes taking place in the sediment over a depth profile and to
101 characterize the geogenic components and so origin of the sediments.

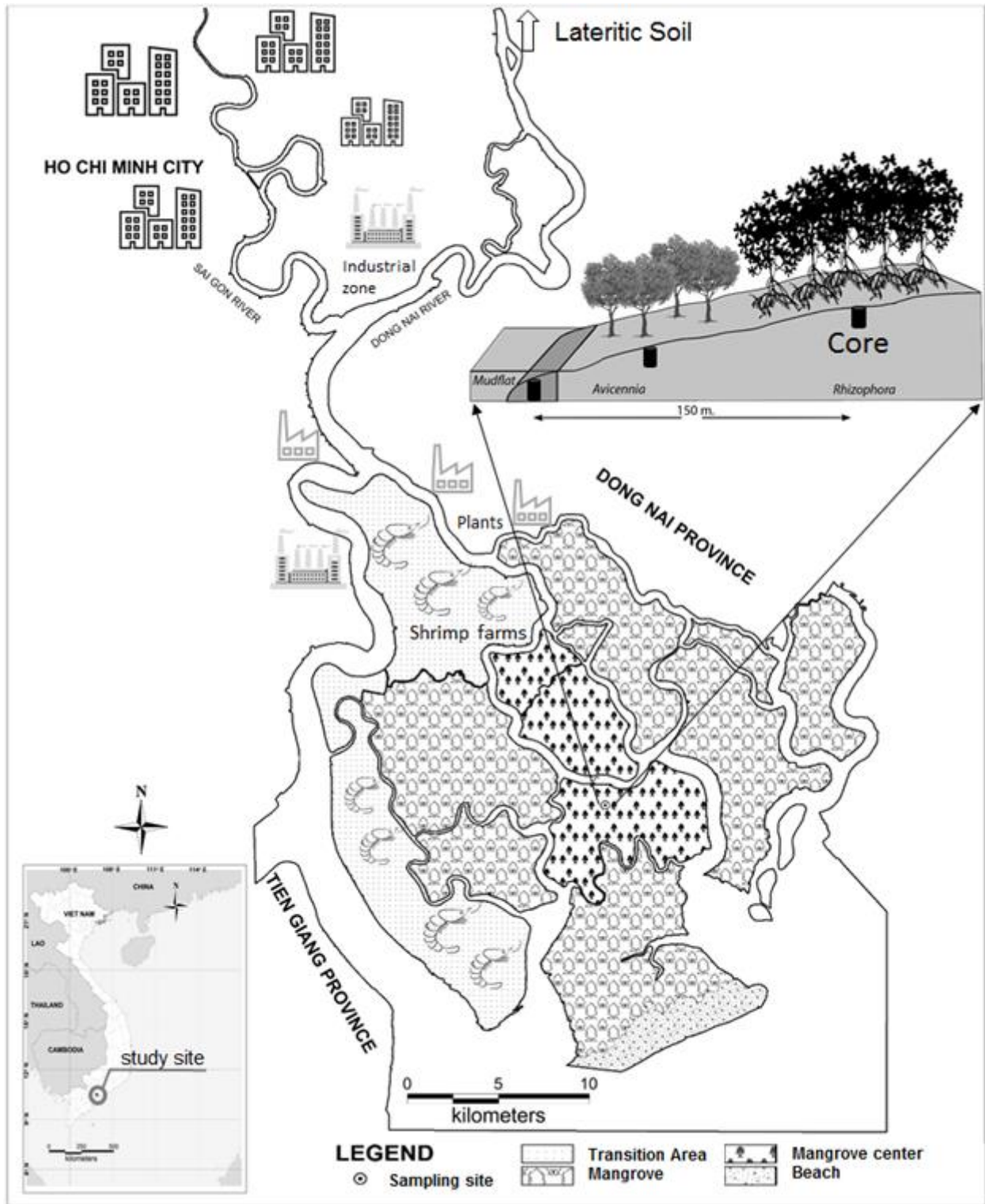
102 Within this context, the objectives of this study were: *i*) to investigate the geochemistry of Fe,
103 Mn, Ni, Cr, Cu, Co and As in sediment cores of the Can Gio Mangrove by focusing on the metal
104 speciation in the sediments and their concentrations in pore-waters; and *ii*) to assess the potential
105 ecological risks of these metals accumulated in the sediments on the mangrove ecosystem by using
106 Risk Assessment Code (Benson et al. 2017). The sediment cores were sampled on three specific

107 environments characterized by different elevation (i.e. tidal inundation), organic matter content,
108 redox stratification and plant cover: in the mudflat, beneath *Avicennia* stand and beneath
109 *Rhizophora* stand.

110 **2. Materials and methods**

111 *2.1. Study site*

112 The Can Gio Mangrove (approximately 35,000 ha, extending from 10°22'-10°44'N and
113 106°46'-107°01'E (Tuan and Kuenzer 2012)) is located in the south of Vietnam, at the downstream
114 part of the Sai Gon and Dong Nai Rivers watershed and in the South China Sea coastal zone (Fig.
115 1). It is a Biosphere Reserve of UNESCO since 2000 and it is also a well-known example of
116 “mangrove afforestation and reforestation area” (Blasco et al. 2001). The Can Gio Mangrove is
117 home to more than 20 mangrove species with two dominant ones: *Rhizophora apiculata* and
118 *Avicennia alba* (Luong et al. 2015), and 200 species of fauna (e.g. amphibians, fishes, benthic
119 organisms, etc.). The main economic activities of the local people are forest management,
120 aquaculture, fishing and salt production (Kuenzer and Tuan 2013). The Can Gio Mangrove is
121 subject to the typical tropical monsoon climate, with two distinct seasons. The dry season lasts from
122 November to April and the monsoon season from May to October. The annual mean precipitation
123 is about 1,300 to 1,400 mm, with ~ 90 % of the precipitation falling during the monsoon season
124 and the annual mean temperature ranges from 26.5 °C to 30 °C. It is also subject to a semi-diurnal
125 tidal regime. The topography of the Can Gio Mangrove is generally low-lying. Average vertical
126 accretion rates in the Can Gio Mangrove are similar between fringe (1.06 ± 0.12 cm year⁻¹) and
127 interior mangrove (0.99 ± 0.09 cm year⁻¹) (MacKenzie et al. 2016). The sediment is composed of
128 alluvial deposits and lateritic soils (Luong 2011) from the Sai Gon and Dong Nai Rivers. The
129 lateritic soils originate from the physico-chemical weathering of basaltic rocks in the Central
130 Highland of Vietnam (Egawa and Ooba 1963).



131

132 *Fig. 1. Location of the Can Gio Mangrove and the three cores (90 cm depth) collected in the mudflat, beneath the*
 133 *Avicennia stand and the Rhizophora stand.*

134

135 **2.2. Sample collections and preservations**

136 Sampling was carried out in the center area of the Can Gio Mangrove during the monsoon
 137 season (October 2016). Three cores (90 cm depth) were collected with an Eijkelkamp gouge auger
 138 at low tide in the mudflat (lower elevation than the mangrove stands), beneath an *Avicennia alba*

139 stand and a *Rhizophora apiculata* stand (Fig. 1). Cores were immediately sectioned into 10 samples:
140 every 5 cm from the surface to 30 cm depth, every 10 cm from 30 to 50 cm depth, and every 20 cm
141 from 50 to 90 cm depth. Samples were stored in polythene bags, preserved in a cooler box for their
142 transfer to the laboratory and were then stored frozen (-20 °C) until drying. Dried samples were
143 grinded using an agate pestle and mortar, and sieved using 100 µm pore size for sequential
144 extraction of metals fractions and for total organic carbon (TOC) analysis.

145 For dissolved metal analysis, pore-waters were extracted on the day of coring with soil moisture
146 sampler Rhizon[®], which were directly inserted into the center of the cores (Marchand et al. 2012).
147 All samples were then filtered through 0.45 µm Sartorius[®] filter membranes and acidified to pH <
148 2 by Suprapur[®] concentrated HNO₃ (Merck). The samples were then preserved in cleaned 14 ml
149 polypropylene (PP) tubes at 4 °C until analysis.

150 *2.3. Analytical methods and calculations*

151 *2.3.1. Salinity, pH and redox measurements*

152 Additional cores beneath each mangrove stand and in the mudflat were collected to measure
153 salinity, pH and redox. These parameters were measured in-situ. Salinities were determined using
154 an ATAGO refractometer (S-10, Japan) after extracting a drop of pore-water from each sediment
155 layer. pH was measured using a glass electrode (pH 3110-WTW), which was pre-calibrated using
156 pH 4, 7 and 10 standard buffer solutions (NIST scale). Redox potential was measured using digital
157 voltmeter with Pt and Ag/AgCl (reference) electrode connected to pH/mV/T meter (pH100-YSI),
158 which was periodically checked using 0.43V standard solution and deionized water. Redox data
159 are reported relative to a standard hydrogen electrode i.e. after adding 194 mV to the original mV
160 values obtained with an Ag/AgCl (reference electrode) at 30 °C. The redox scale was fully
161 described by Marchand et al. (2011a), as follows: (i) oxic > 400 mV, containing measurable
162 dissolved oxygen; (ii) 100 mV < suboxic < 400 mV, with a lack of measurable oxygen or sulfide,
163 containing dissolved iron or manganese, with no reduction of sulfate; (iii) anoxic < 100 mV, with
164 sulfate reduction.

165 *2.3.2. Metal concentrations in pore-waters*

166 Dissolved Fe, Mn, Ni, Cr, Cu, Co and As concentrations were directly measured by Thermo
 167 Scientific iCAPQ ICP-MS with a Kinetic Energy Discrimination-Argon Gas Dilution module
 168 (KED - AGD mode) (Kutscher et al. 2014) using internal standard calibration (AETE-ISO platform,
 169 OSU-OREME/Université de Montpellier). Accuracy and precision were controlled using the
 170 certified reference material SLEW-3 (Table 1a).

171 *Table 1. Quality control of analytical methods applied for dissolved and total metals concentrations analysis*
 172 *respectively: a) accuracy, precision and detection limit using estuarine water SLEW-3; b) total metal concentrations*
 173 *were calculated by summing the four single fraction (F1 + F2 + F3 + F4) from CRM (BCR-277R).*

174 *a) Dissolved metal concentration analysis.*

<i>Element</i>	<i>Detection limit ($\mu\text{g L}^{-1}$)</i>	<i>Certificated values of SLEW-3 ($\mu\text{g L}^{-1}$)</i>	<i>Measured values ($\mu\text{g L}^{-1}$)</i>	<i>Recovery (%)</i>	<i>Relative standard deviation (%)</i>
Fe	0.031	0.568 ± 0.059	0.661 ± 0.073	116	11
Mn	0.013	1.61 ± 0.22	1.469 ± 0.016	91	1
Ni	0.011	1.23 ± 0.07	1.230 ± 0.014	100	1
Cr	0.022	0.183 ± 0.019	0.0182 ± 0.020	99	11
Cu	0.011	1.55 ± 0.12	1.491 ± 0.041	96	3
Co	0.0092	0.042 ± 0.010	0.0464 ± 0.0025	110	5
As	0.0062	1.36 ± 0.09	1.569 ± 0.013	115	1

175 *b) Total metal concentration analysis*

<i>Element</i>	<i>Certificated values of BCR-277R (mg kg^{-1})</i>	<i>Measured values (mg kg^{-1}, n = 9)</i>	<i>Recovery (%)</i>	<i>Relative standard deviation (%)</i>	<i>Analytical method</i>
Fe	NA	50,843 ± 3,029	-	5.8	FAAS
Mn	NA	897 ± 37	-	4.2	FAAS
Ni	130 ± 8	125.9 ± 6.1	96.9	4.9	ICP-MS
Cr	188 ± 14	179.9 ± 16.1	95.7	9	ICP-MS
Cu	63 ± 7	58.9 ± 4.6	93.6	7.9	ICP-MS
Co	22.5 ± 1.4	23.1 ± 1.7	102.6	7.5	ICP-MS
As	18.3 ± 1.8	18.1 ± 1.5	98.9	8.2	ICP-MS

176 *2.3.3. Sequential extraction: metal fractions*

177 To evaluate the geochemistry and availability of metals (Fe, Mn, Ni, Cr, Cu, Co and As) in the
 178 sediment cores, a sequential extraction procedure was carried out based on the method developed
 179 by Tessier et al. (1979) and Ure et al. (1993). Each element was divided into four operationally-
 180 defined geochemical fractions: exchangeable/carbonate fraction (acid-soluble phase), Fe – Mn
 181 oxides fraction (reducible phase), organic fraction (oxidizable phase) and a residual fraction.
 182 Briefly, the various single extractions were performed as following: 1 g of fine dry sediments were

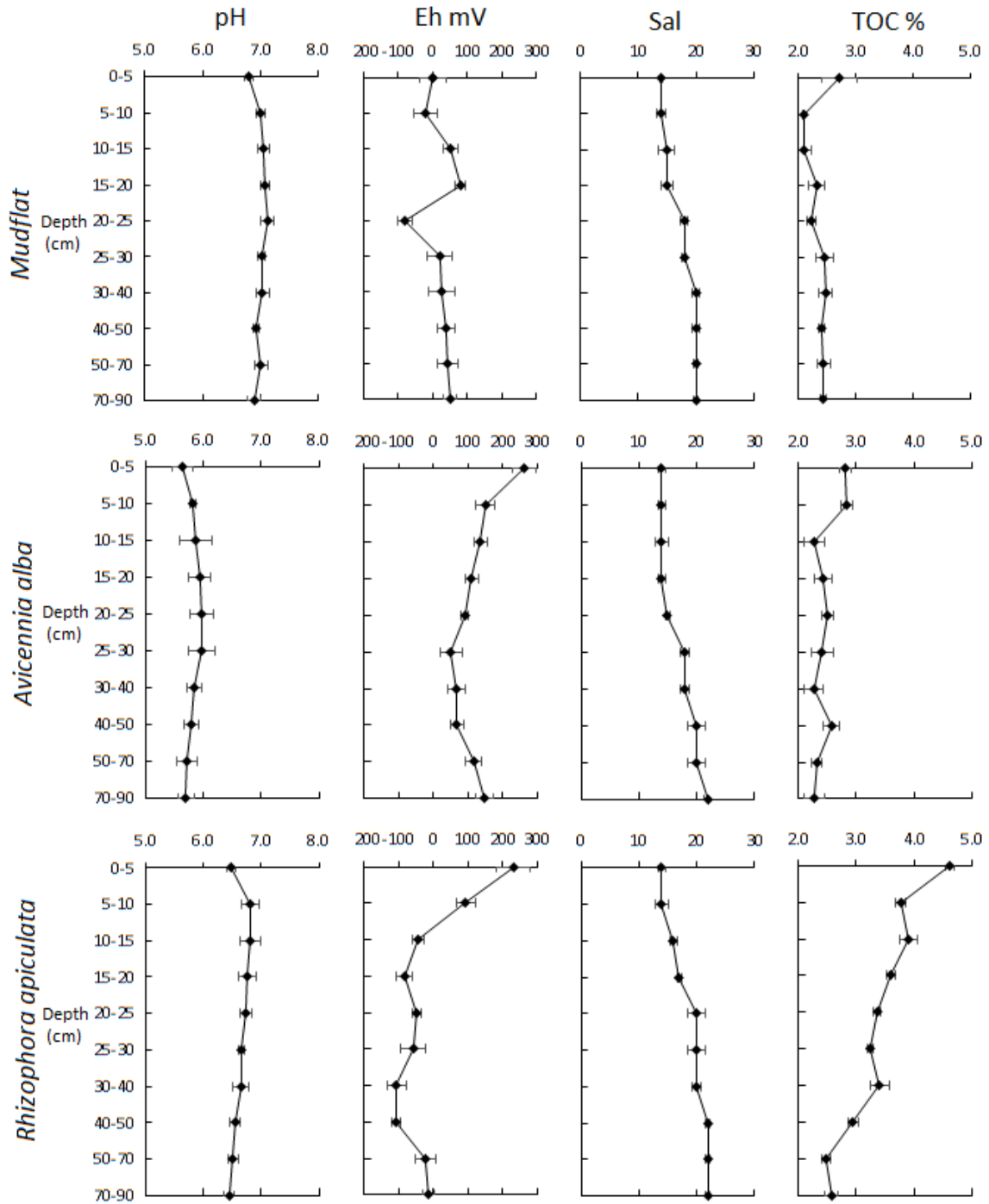
183 put into 50 mL PP tubes with caps, which were also used for shaking time and centrifugation to
184 minimize the possible loss of the centrifuge – washing step. For the determination of the acid-
185 soluble fraction (F1), we used 8 mL of 1 M buffer acidic solution ($\text{CH}_3\text{COOH}/\text{CH}_3\text{COONH}_4$, pH
186 = 5) at room temperature during 5 h; for the reducible fraction (F2), we used 20 mL of 0.04 M
187 $\text{NH}_2\text{OH}\cdot\text{HCl}$ in 25 % CH_3COOH (m/v) at 96 °C during 6 h; for the oxidizable fraction (F3), we
188 used 3 mL of 0.02 M HNO_3 and 8 mL of 30 % H_2O_2 . 5 mL of 3.2 M $\text{CH}_3\text{COONH}_4$ was added into
189 the solution to prevent reabsorb of these metals onto particles surface; for the total metals
190 concentration in the residual fraction (F4), the samples was digested using 10 mL of concentrated
191 $\text{HNO}_3/\text{HCl}/\text{HF}$ (3:1:1 = v/v) in polytetrafluoroethylene (PTFE) vessel at 110 °C for 48 h. After
192 cooling, 2 mL of concentrated HNO_3 ($n = 2$) was added into these samples for eliminating residual
193 HF. For each adding, the solution were evaporated near to dryness at 160 °C. The samples were
194 filtrated and adjusted using deionized water into 25 mL, and were then stored in pre-cleaned
195 polypropylene tubes at 4 °C until analysis.

196 2.3.4. Total metal concentration

197 Total metal concentrations were obtained by summing the four single fractions (F1 + F2 + F3
198 + F4). The quantifications of Fe and Mn concentrations were performed by Flame Atomic
199 Absorption (Shimadzu AA-6650) while the other metals (Co, Ni, Cu, As and Cr) were analyzed by
200 ICP-MS (Agilent 7700x) using spiked ^{103}Rh and ^{197}Au as internal standard (Table 1b). The
201 analytical precision and accuracy were insured by analyzing certified reference material estuarine
202 sediment (BCR-277R) intercalated in each batch of sample digestion. All reagents were analysis
203 grade (Merck) and all containers were de-contaminated by soaking in 5 % nitric acid for 24 h and
204 rinsed in deionized water.

205 2.3.5. Total organic carbon (TOC)

206 TOC analysis was carried out on a Shimadzu® TOC-L series analyzer combined with a solid
207 sample module (SSM-5000A) heating at 900 °C. Glucose standard (40 %, Sigma Aldrich) was used
208 for calibrations. Repeated measurements of the standards at different concentrations indicated a
209 measurement deviation < 2 %.



236

237 *Fig. 2. Depth distributions of pH, Eh, Salinity and TOC in the mudflat, the Avicennia and the Rhizophora stands*
 238 *in the Can Gio Mangrove.*

239 **3. Results**

240 *3.1. Physico-chemical parameters in the sediment cores*

241 The depth evolutions of pH, redox potential (Eh), salinity and total organic carbon (TOC) in
 242 the mudflat and the mangrove stands are presented in Fig. 2. pH was stable with depth whatever
 243 the sites, being lower in the *Avicennia* stand (5.6 to 6.0) than in the *Rhizophora* stand (6.5 to 6.8)
 244 and in the mudflat (6.8 to 7.1). Eh distributions and values differed in the three environments. The

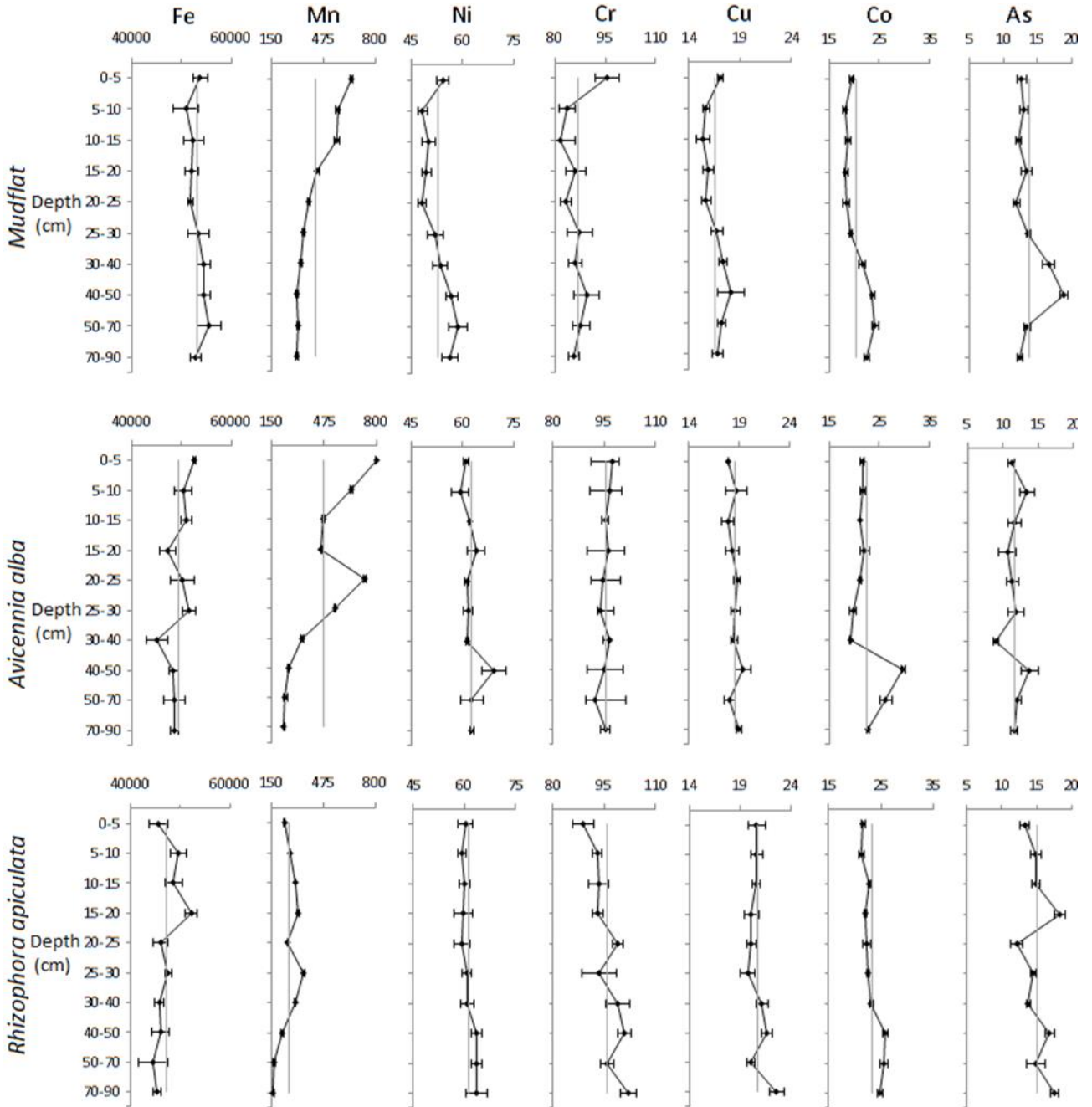
245 mudflat was characterized by anoxic condition in whole core, ranging between 73 mV and -88 mV.
246 The sediments beneath mangroves stands were characterized from suboxic to anoxic conditions
247 toward the bottom of the cores. In the *Avicennia* stand, Eh values decreased from 254 mV at the
248 top of the core to 43 mV at 30 cm depth, and then slightly increased without being higher than 140
249 mV. In the *Rhizophora* stand, redox values decreased from the top of the core (224 mV) to 20 cm
250 depth (-90 mV) and then fluctuated between of -20 mV to -115 mV. Salinity increased with depth
251 in the three cores with similar distribution patterns, from 14 to 22. TOC contents varied with depth
252 in the mudflat (from 2.7 % to 2.1 %) and in the *Avicennia* stand (from 2.8 % to 2.3 %) without
253 specific distributions patterns, while in the *Rhizophora* stand TOC concentrations were higher than
254 in the other sites, and decreased with depth (from 4.62 % to 2.1 %).

255 3.2. Total metal concentrations in the sediment cores

256 The total metal concentrations (Fe, Mn, Ni, Cr, Cu, Co and As) in the three cores were obtained
257 by summing the four single fraction (i.e. F1 + F2 + F3 + F4). The depth's distribution of each metal
258 in the mudflat, the *Avicennia* stand, the *Rhizophora* stand and their mean concentrations are
259 presented in Fig. 3 and supplementary data (Table SD).

260 Total Fe concentrations did not exhibit specific distribution with depth in the three
261 environments. The mean values of total Fe concentration decreased from the mudflat ($53,146 \pm$
262 $1,422 \text{ mg kg}^{-1}$) to the mangrove stands ($49,305 \pm 2,198 \text{ mg kg}^{-1}$ and $47,135 \pm 2,370 \text{ mg kg}^{-1}$ in the
263 *Avicennia* stand and *Rhizophora* stand, respectively). The total Mn concentrations decreased with
264 depth in each environment, from 653 mg kg^{-1} to 310 mg kg^{-1} in the mudflat, from 800 mg kg^{-1} to
265 227 mg kg^{-1} in the *Avicennia* stand and from 350 mg kg^{-1} to 160 mg kg^{-1} in the *Rhizophora* stand.
266 Total Ni and Cr concentrations presented small amplitude with depth in the three environments but
267 their mean concentrations in each core evidenced higher mean values in the *Avicennia* and
268 *Rhizophora* stands ($62.5 \pm 2.6 \text{ mg kg}^{-1}$ and $61.1 \pm 1.7 \text{ mg kg}^{-1}$ for Ni; $95.3 \pm 1.4 \text{ mg kg}^{-1}$ and 95.8
269 $\pm 4.2 \text{ mg kg}^{-1}$ for Cr, respectively) than in the mudflat ($52.7 \pm 3.7 \text{ mg kg}^{-1}$ for Ni and $86.6 \pm 3.8 \text{ mg}$
270 kg^{-1} for Cr). Total Co concentrations showed similar distributions between environments, being
271 stable values in the upper horizons from 0 to 40 cm depth and then increasing to reach a maximum

272 values of 24.2 mg kg⁻¹, 29.7 mg kg⁻¹ and 25.8 mg kg⁻¹ in the mudflat, the *Avicennia* stand and the
 273 *Rhizophora* stand, respectively. Finally, total As and Cu concentrations did not present any specific
 274 vertical distribution patterns, and their mean concentrations were similar to all environments
 275 (supplementary data, Table SD).



276
 277 *Fig. 3. Vertical distributions of metals concentrations (square dots) in the mudflat, the Avicennia stand and the*
 278 *Rhizophora stand (expressed in mg kg⁻¹), with gray line presenting mean metal concentrations of each core.*

279 **3.3. Metal stock estimation**

280 The metal stocks were estimated over the 50 cm depth profile of the cores. Interestingly, the
 281 metal stocks differed between the mangrove stands and the mudflat (Table 2). The Fe stock

282 presented decreasing values from the mudflat ($169.2 \pm 4.2 \text{ t ha}^{-1}$) to the inner mangrove (i.e. the
 283 *Rhizophora* stand, $126.4 \pm 6.2 \text{ t ha}^{-1}$). Conversely, the stock of Mn was higher beneath the *Avicennia*
 284 stand ($1.62 \pm 0.57 \text{ t ha}^{-1}$) than in the mudflat ($1.44 \pm 0.41 \text{ t ha}^{-1}$) and was even lower in the
 285 *Rhizophora* stand ($0.74 \pm 0.12 \text{ t ha}^{-1}$). Except As stock, which was higher in the mudflat, the stocks
 286 of Co, Ni, Cr and Cu showed higher values in the *Avicennia* stand than in the *Rhizophora* zone and
 287 in the mudflat. They varied (expressed in t ha^{-1}) from 0.16 ± 0.004 to 0.19 ± 0.01 for Ni, from 0.25
 288 ± 0.01 to 0.29 ± 0.003 for Cr, and from 0.053 ± 0.003 to 0.057 ± 0.002 for Cu, from 0.060 ± 0.004
 289 to 0.067 ± 0.010 for Co, from 0.035 ± 0.005 to 0.045 ± 0.008 for As.

290 *Table 2. Metal stocks in the sediments of the mudflat, the Avicennia stand and the Rhizophora stand in the Can*
 291 *Gio Mangrove (expressed in t ha^{-1} , values were obtained based on the average total metal concentration and average*
 292 *bulk density of each core on 50 cm depth). The metal stocks beneath Avicennia and Rhizophora stands were*
 293 *compared with those measured in the mudflat (%).*

Stand	Bulk density (g cm^{-3})	Fe	Mn	Ni	Cr	Cu	Co	As
<i>Mud flat</i>	0.64	169.2 ± 4.2	1.44 ± 0.41	0.16 ± 0.01	0.28 ± 0.01	0.053 ± 0.003	0.063 ± 0.006	0.045 ± 0.008
<i>Avicennia</i>	0.61	150.9 ± 7.5	1.62 ± 0.57	0.19 ± 0.01	0.29 ± 0.003	0.057 ± 0.002	0.067 ± 0.010	0.035 ± 0.005
(% mud flat)		89	112	116	105	107	106	79
<i>Rhizophora</i>	0.53	126.4 ± 6.2	0.74 ± 0.12	0.16 ± 0.004	0.25 ± 0.01	0.055 ± 0.002	0.060 ± 0.004	0.039 ± 0.005
(% mud flat)		75	51	97	91	103	95	87

294 3.4. Metal partitioning in the sediment cores

295 The metals (Fe, Mn, Ni, Cr, Cu, Co and As) partitioning in each layer of sediment core collected
 296 in the mudflat, the *Avicennia* stand and the *Rhizophora* stand was determined based on their
 297 concentrations in the exchangeable/carbonate fraction (F1), the oxides fraction (F2), the organic
 298 fraction (F3) and the residual fraction (F4). Their respective percentages/ proportions in each phase
 299 were evaluated based on their total concentrations (i.e. summing of metal concentrations in four
 300 fractions), being presented in Fig. 4.

301 3.4.1. Fe and Mn

302 The Fe and Mn partitioning exhibited different vertical distributions according to environments.
 303 In the mudflat and in the *Avicennia* stand, the Fe partitioning was $F4 > F2 > F3 \sim F1$ (Fig. 4a)
 304 whereas for Mn it was: $F4 \sim F1 > F2 > F3$ (Fig. 4b). Fe in F4 was stable with depth, representing
 305 72 % in the mudflat and 78 % in the *Avicennia* stand, whereas Mn in F4 increased with depth, from
 306 22 % to 38 % in the mudflat and from 18 % to 62 % in the *Avicennia* stand. Mn was characterized

307 by an important exchangeable/carbonate fraction, which decreased from 42 % to 27 % in the
308 mudflat and from 51 % to 22 % in the *Avicennia* stand. F2 of both Fe and Mn decreased with depth
309 in the mudflat and in the *Avicennia* stand, respectively from 20 % to 12 % and 20 % to 12 % for
310 Fe, and respectively from 42 % to 27 % and 51 % to 22 % for Mn. F3 of Fe and Mn presented
311 increasing values with depth in the mudflat, from 8 % to 15 % for Fe and from 6 % to 13 % for
312 Mn, while it is stable in the *Avicennia* stand with a mean value of 3 % for Fe and values ranging
313 from 2 % to 6 % for Mn. F1 of Fe was stable with depth with mean values of 2 % and 3% in the
314 mudflat and in the *Avicennia* stand, respectively.

315 In the *Rhizophora* stand, Fe and Mn exhibited the same partitioning: $F4 > F3 > F2 > F1$. The
316 residual fraction was stable from the top to the 40 cm depth, and then gradually increased with
317 depth from 62 % to 78 % for Fe and from 35 % to 66 % for Mn. The organic fraction of both Fe
318 and Mn was higher in the *Rhizophora* stand than in the *Avicennia* stand and the mudflat. Conversely
319 to F4, the proportion of Fe and Mn in F3 increased from the top to 40 cm depth (i.e. ranging from
320 18 to 36 % for Fe and 21 to 54 % for Mn), then decreased to the bottom of the core (i.e. ranged
321 from 35 to 16 % and from 52 to 17 % for Fe and Mn, respectively). Fe and Mn in F2 decreased
322 toward the bottom of the core, from 15 % to 3 % for Fe and from 23 % to 8 % for Mn. Eventually,
323 the exchangeable/carbonate fraction of Fe represented less than 1 % of total Fe concentrations,
324 while F1 of Mn decreases with depth from 16 % to 3 %.

325 3.4.2. *Ni, Cr, Cu, Co and As*

326 The Ni partitioning was characterized by $F4 > F2 \sim F3 > F1$ in the mudflat and the *Avicennia*
327 stand and by $F4 > F3 > F2 > F1$ in the *Rhizophora* stand (Fig. 4c). F4 was in the same range (from
328 76 % to 84 %) in the mudflat and in the *Rhizophora* stand, but higher (from 83 % to 91 %) in the
329 *Avicennia* stand, without clear vertical distributions in all cores. F3 was stable with depth and
330 presented a higher proportion in the *Rhizophora* stand (11 %) than in the mudflat (8 %) and in
331 *Avicennia* stand (5 %). F2 and F1 were stable with depth at all sites, and represented less than 8 %
332 for F2 and less than 3 % for F1.

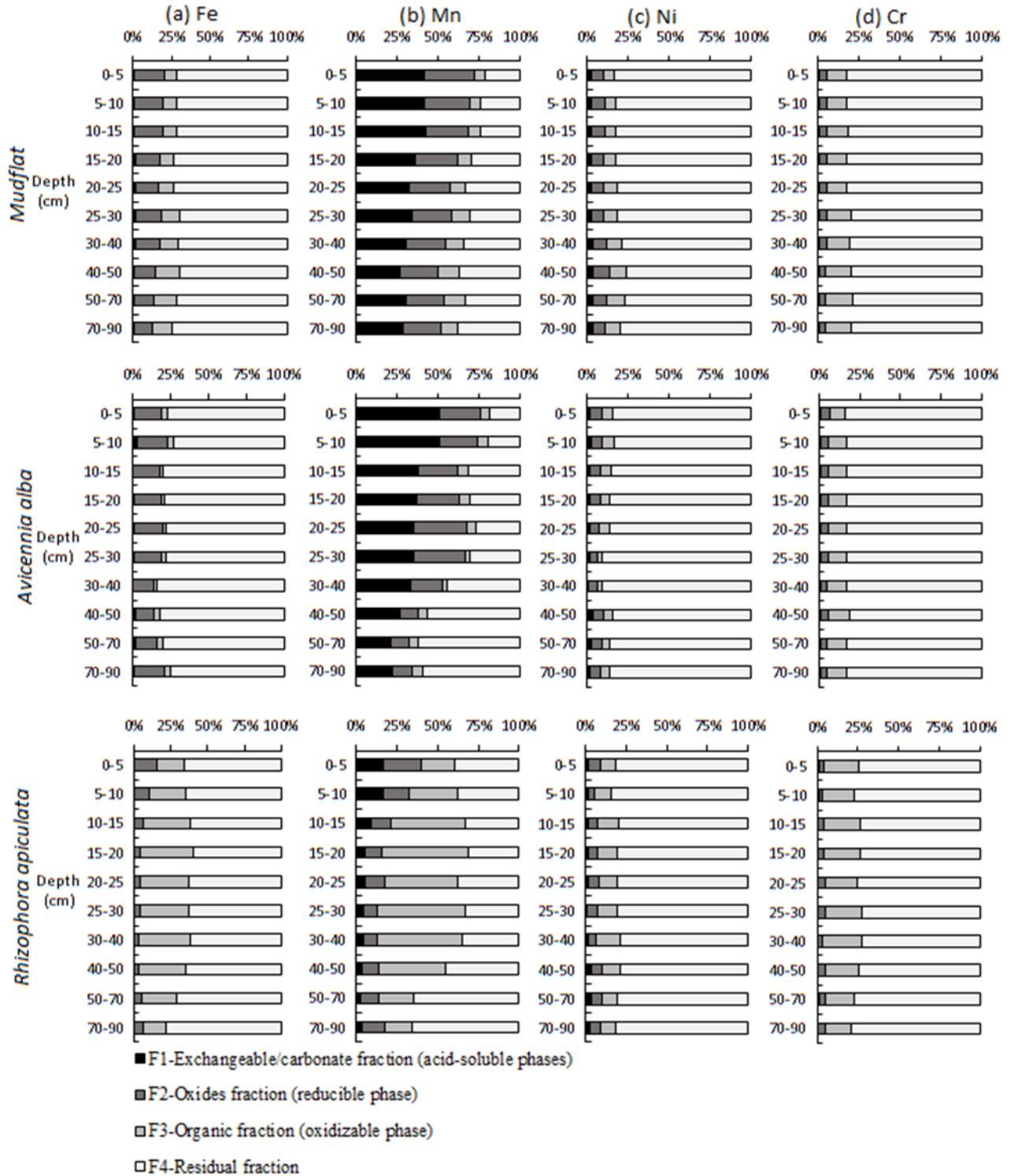
333 The Cr partitioning was $F4 > F3 > F2 > F1$, without any clear depth evolution (Fig. 4d). The
334 residual fraction ranged from 78 % and 84 % in the mudflat and the *Avicennia* stand and from 72
335 % to 79 % in the *Rhizophora* stand. The organic fraction was characterized by higher proportion in
336 the *Rhizophora* stand (16 % to 24 %) than in the mudflat (11 % to 17 %) and in the *Avicennia* stand
337 (10 % to 13 %). The oxides fraction and the exchangeable/carbonate fraction presented low
338 proportion (less than 5 %) and same range of values in all environments.

339 The Cu partitioning differed between sites: $F4 > F3 > F1 > F2$ in the mudflat, $F4 > F3 > F2 \sim$
340 $F1$ in the *Avicennia* stand, and $F4 > F3 > F2 > F1$ in the *Rhizophora* stand (Fig. 4e). For the F4, the
341 proportion of Cu was higher in the *Avicennia* stand (85 % to 94 %) than in the *Rhizophora* stand
342 (44 % to 73 %) and in the mudflat (57 % to 75 %). In the mudflat, the proportion of Cu in F4
343 decreased with depth while no specific distribution was observed beneath the mangrove stands. F3
344 ranged from 24 % to 41 % in the mudflat and *Rhizophora* stand, and from 6 % to 14 % in the
345 *Avicennia* stand. F1 and F2 fractions were characterized by low proportion (< 2 %) in all
346 environments except in the *Rhizophora* stand where the F2 increased from 20 cm depth until the
347 bottom of the core.

348 The Co partitioning also differed between cores: $F4 > F2 \sim F3 > F1$ in the mudflat, $F4 > F2 >$
349 $F1 \sim F3$ in the *Avicennia* stand and $F4 > F3 > F2 > F1$ in the *Rhizophora* stand (Fig. 4f). The residual
350 fraction was closed to 50 % in both the mudflat and the *Rhizophora* stand, but ranging from 50 %
351 to 76 % in the *Avicennia* stand with lower proportion at depth. F3 presented opposite pattern than
352 F4 in the mudflat and the mangrove stands. F2 and the F1 were stable in all environments with
353 lower proportion in the *Rhizophora* stand for F2 and increasing F1 proportion from 40 cm depth in
354 the *Avicennia* stand.

355 The As partitioning was $F4 > F2 > F3 > F1$ in the mudflat, $F4 > F2 > F3 \sim F1$ in the *Avicennia*
356 stand and $F4 > F2 \sim F3 > F1$ in the *Rhizophora* stand (Fig. 4g). In the upper part of the core in the
357 mudflat (to 25 cm depth), all fractions presented stable proportions. Then, As in F3 and F4
358 increased with depth while As in F1 and F2 decreased. In the core beneath the *Avicennia* stand, As
359 proportion in F4 gradually decreased from the top to the 40 cm depth and dramatically increased

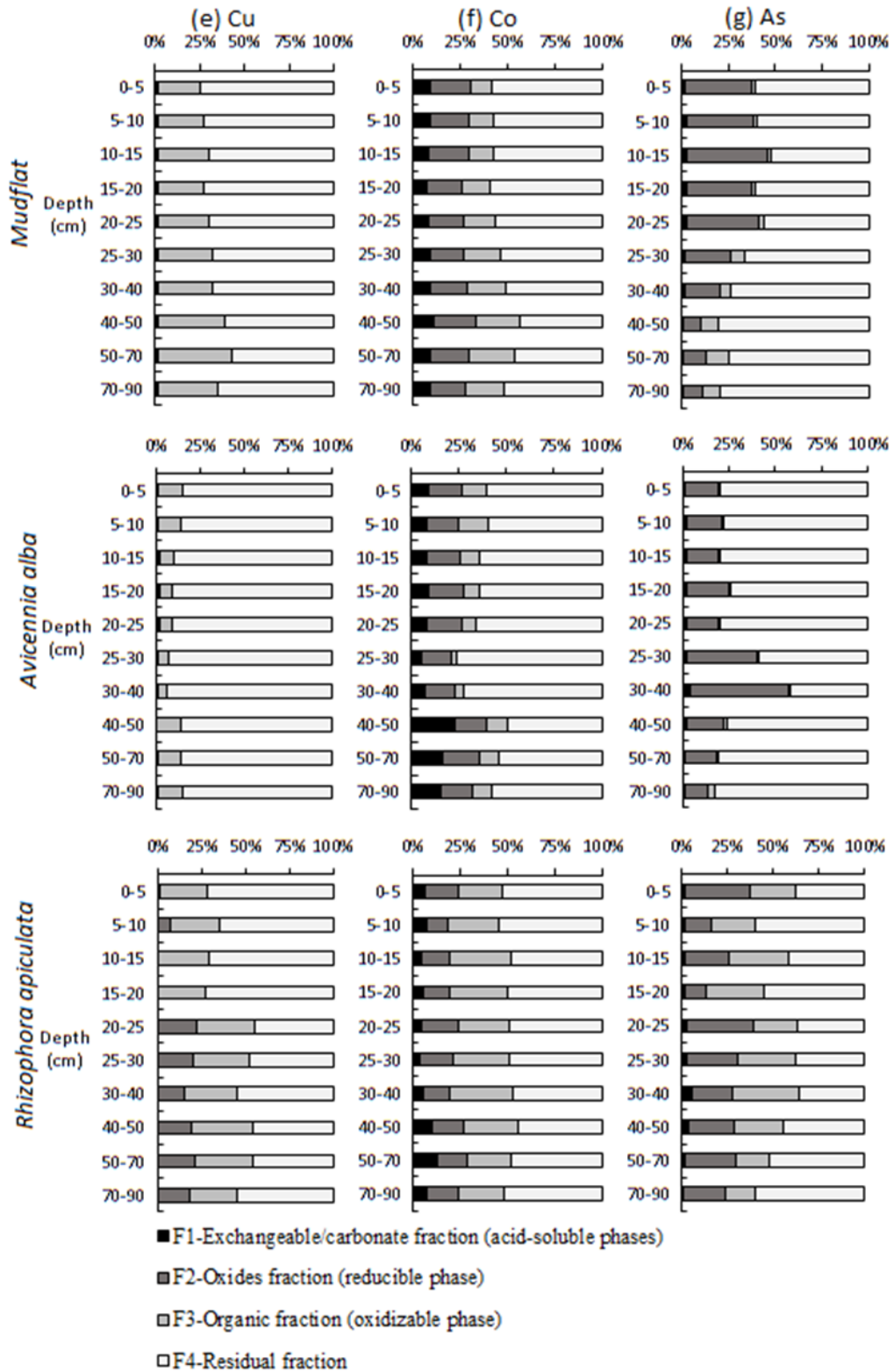
360 in deeper layers while the opposite distribution was observed for F3. F2 and F1 did not exhibit any
 361 specific distribution along the core. In the *Rhizophora* stand, the residual and the organic fractions
 362 were stable from the top of the core to 40 cm depth, then F4 increased toward the bottom while F3
 363 decreased. The F2 and F1 fraction did not exhibit any specific distribution along the core.



364
 365 Fig. 4. Depth profile of sequential extraction of (a) Fe, (b) Mn, (c) Ni, (d) Cr, (e) Cu, (f) Co and (g) As in the
 366 mudflat, the *Avicennia* stand and the *Rhizophora* stand in the Can Gio Mangrove (to be continued).

367

368



369

370

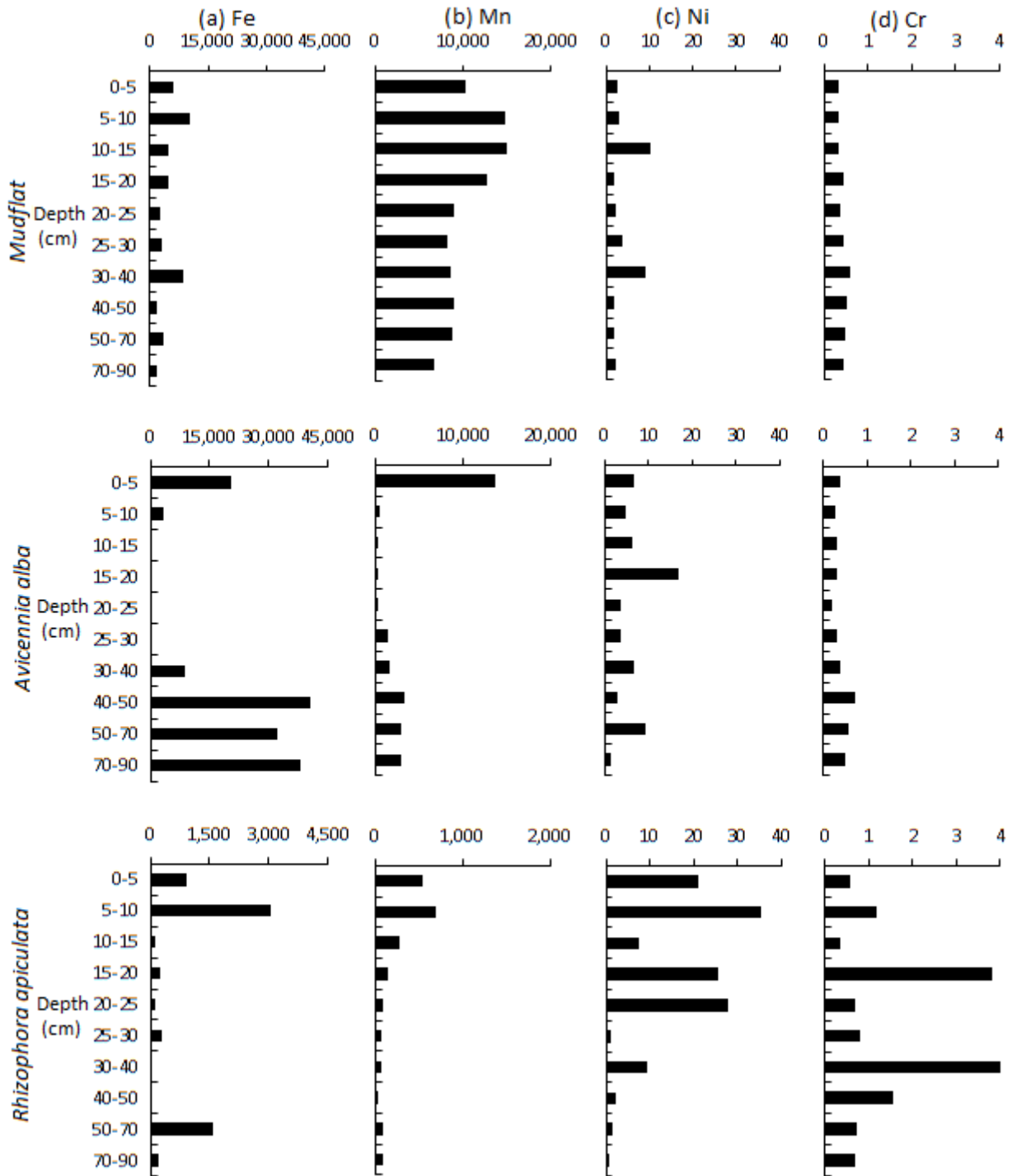
Fig. 4. (Continued)

371

3.5. Dissolved metal concentrations in the core's pore-waters

372 Dissolved Fe concentrations (Fe_D) in pore-waters presented lower level of concentration in the
373 *Rhizophora* stand (14 to 3,055 $\mu\text{g L}^{-1}$) and the mudflat (1,621 to 10,303 $\mu\text{g L}^{-1}$) than in the *Avicennia*
374 stand (333 to 40,771 $\mu\text{g L}^{-1}$). The Fe_D distributions were characterized by decreasing Fe_D from the
375 surface to 15 cm depth and then increasing values with depth in the stands (Fig. 5a). Dissolved Mn
376 concentrations (Mn_D) presented higher levels in the mudflat (from 6,812 to 15,106 $\mu\text{g L}^{-1}$) and in
377 the *Avicennia* stand (25 to 13,648 $\mu\text{g L}^{-1}$) than in the *Rhizophora* stand (from 542 to 643 $\mu\text{g L}^{-1}$)
378 (Fig. 5b). The Mn_D concentrations decreased from the top to 10 cm depth (in *Avicennia* stand) and
379 to 40 cm depth (in *Rhizophora* stand) and then increased again with depth.

380 The dissolved Ni (Ni_D), Cr (Cr_D) and Cu (Cu_D) concentrations in pore-waters presented similar
381 patterns in the three cores (Fig. 5c, 5d and 5e). In the mudflat and the *Avicennia* stand, Ni_D , Cr_D
382 and Cu_D were stable with depth with increasing concentrations at 30 cm depth in the mudflat and
383 at 15 cm depth in the *Avicennia* stand. In the *Rhizophora* stand, Ni_D , Cr_D and Cu_D presented higher
384 concentrations with a decreasing pattern from the top to the surface for Ni_D and Cu_D and a stable
385 pattern with two peaks at 15 cm and 30 cm depth for Cr_D . The dissolved Co (Co_D) and As (As_D)
386 concentrations in pore-waters (Fig. 5f, 5g) presented a stable vertical distribution with a peak at 30
387 cm in the mudflat core while in the *Avicennia* stand, they presented decreasing concentrations from
388 the top to 10 cm depth, increasing concentrations until reaching a peak at 30 cm and 40 cm depth
389 for Co_D and As_D respectively, and then decreasing concentrations towards the bottom. Finally in
390 the *Rhizophora* stand, Co_D decreased with depth with a peak at 15 cm depth while As_D is stable
391 with two peaks at 25 cm and 50 cm depth.



392

393

394

395

396

397

Fig. 5. Depth profile of dissolved metals concentrations (µg L⁻¹) in the pore-waters of the mudflat, *Avicennia* and *Rhizophora* stands. Asterisks highlight extreme values that could not be shown in the graph even with cut axis (i.e. values of 14.4 for Cr and 13.9 for Cu) (to be continued).

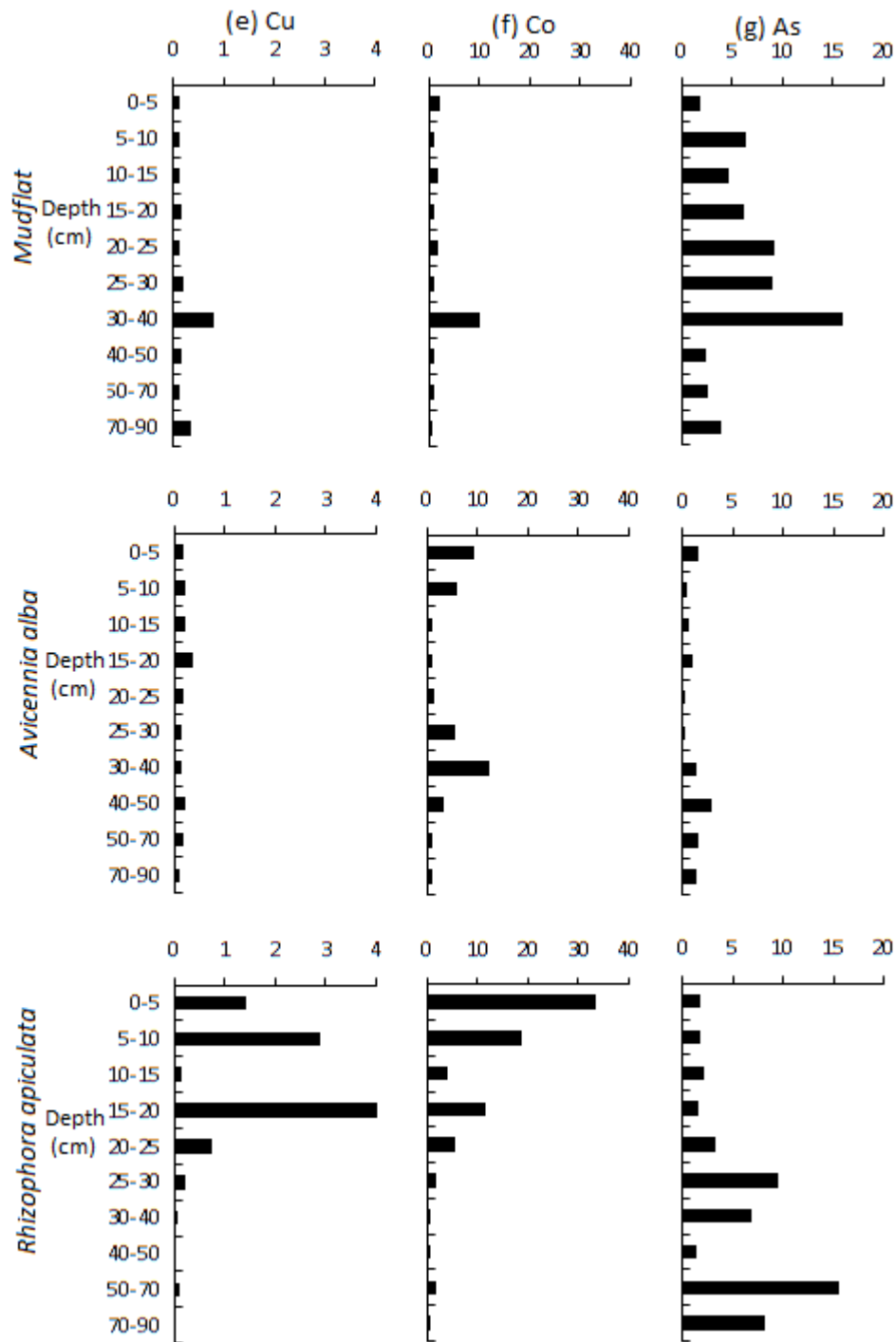


Fig. 5. (Continued)

398

399

400 4. Discussions

401 4.1. Mangrove sediments characteristics

402 Mangrove forests are known to be highly productive ecosystems (Bouillon et al. 2008), storing
 403 huge quantity of carbon in their sediments. Kristensen et al. (2008a) showed that TOC values in
 404 mangrove sediments usually range between 0.5 % and 15 %, with a median value around 2.2 %. In
 405 the Can Gio Mangrove, TOC reached up to 4.6 % with increasing values from the tidal creek (the

406 mudflat core) to the inner mangrove (the *Rhizophora* stand core) (Fig. 2). The higher values
407 measured beneath the *Rhizophora* stand than beneath the *Avicennia* stand may result from a higher
408 productivity of the first species, including a more developed root system as it was observed in
409 Australia (Alongi et al. 2000) or in New Caledonia (Marchand et al. 2011b). It may also be related
410 to the elevation of the sediment: the *Avicennia* trees in the Can Gio Mangrove developing at lower
411 elevation than the *Rhizophora* ones may thus be subject to a more intense tidal export of leaf litter,
412 which limits organic matter accumulation in the sediment. Additionally, the oxygen released by the
413 roots of *Avicennia* trees may induce more efficient organic matter decomposition (Marchand et al.
414 2004). Beneath the two mangrove stands, the redox values decreased with depth probably as a result
415 of organic matter decay and the lack of electron acceptors renewal at depth (Otero et al. 2009). In
416 the upper part of the sediments, the suboxic condition ($100 < Eh < 260$ mV) may be explained by
417 biological or physical factors (i.e. length of emersion, crab activity and bioturbation by mangrove
418 roots) (Marchand et al. 2012). The radial cable roots of *Avicennia* trees can also be important factor
419 inducing higher Eh values than in the *Rhizophora* stand (Marchand et al. 2004). Whereas the
420 mudflat is flooded almost all the time, the absence of air diffusion into the sediment can result to
421 anoxic condition despite its low organic content. We suggest that more intense organic
422 decomposition and possible sulfide oxidation may cause lower pH in the *Avicennia* stand than in
423 the *Rhizophora* stand and in the mudflat (Marchand et al. 2004). Finally, the upper sediment of
424 every core was characterized by lower salinity values than deeper sediment, which can be related
425 to the period of coring, i.e. the end of the wet season, the rainwater inducing a dilution of the saline
426 pore-waters. The rainwater may also be responsible of an enhanced renewal of electron acceptors,
427 influencing OM diagenesis.

428 4.2. Lateritic soil as a main source of trace metal in the mangrove sediments.

429 Mean metal concentrations (Fe, Mn, Ni, Cr, Co, Cu and As) in the Can Gio Mangrove sediments
430 (supplementary data, Fig. 3) are in the range of other mangroves around the world (see review
431 papers of Lewis et al. (2011) and Bayen (2012)) but are lower than those measured in mangroves
432 subject to strong anthropogenic pressure as in India (Fernandes and Nayak 2012). Furthermore,

433 these metal concentrations were also close to those measured in total suspended matter (TSM) in
 434 the Can Gio Mangrove Estuary (Thanh-Nho et al. 2018). These authors stated that TSM acted as
 435 main carrier for metals during their transport to the ocean, and that metal distribution changed due
 436 to the physical mixing by the seawater and/or organic matter decay processes. Consequently, most
 437 metal concentrations in TSM collected in the estuary were lower than upstream, specifically during
 438 the monsoon. We suggest that despite being downstream of a developing megacity characterized
 439 by low urban wastewater treatments (only 10 %, (FAO 2014)), metal accumulation inside the
 440 mangrove forest was relatively limited. Thus, we consider that the main source of metals in the Can
 441 Gio Mangrove is natural and originates from the rivers' watershed in the Central Highland of
 442 Vietnam, composed of lateritic soil originating from the physico-chemical weathering processes of
 443 basaltic rocks (Egawa and Ooba 1963) rich in hematite and goethite minerals. This hypothesis may
 444 be supported by the high proportion of metals in the residual fraction (Fig. 4) and interrelationships
 445 of total metals concentrations in the mudflat, which reflect sediment inputs without the influences
 446 of mangrove trees (Table 3).

447 *Table 3. Pearson correlation matrix of total metal concentrations together in the mangrove mudflat. Metal*
 448 *concentrations were expressed in mg kg⁻¹.*

	Fe	Mn	Ni	Cr	Cu	Co	As
Fe	1						
Mn	-.476	1					
Ni	.875**	-.483	1				
Cr	.538	.155	.544	1			
Cu	.842**	-.495	.854**	.674*	1		
Co	.831**	-.673*	.936**	.300	.828**	1	
As	.537	-.447	.424	.271	.757*	.562	1

** . Correlation is significant at the 0.01 level (2-tailed).

* . Correlation is significant at the 0.05 level (2-tailed).

449 4.3. Trace metal geochemistry in the sediment cores.

450 4.3.1. Redox sensitive elements (Fe and Mn)

451 Iron geochemistry

452 As suggested earlier, the erosion of lateritic soils upstream the Can Gio Mangrove may induce
453 the transfer of these Fe-oxyhydroxides towards the estuary and then their deposition in the
454 mangrove. In the main channel of the Can Gio Estuary, we measured iron concentration up to
455 55,302 mg kg⁻¹ during the monsoon (Thanh-Nho et al. 2018), which comfort our hypothesis.
456 However, in mangrove sediments, the oxide fraction was not the dominant one. This may be
457 attributed first to the limitation of the selective extraction method offered by Tessier et al. (1979),
458 which is inefficient to extract iron from highly crystallized oxides and oxyhydroxides, like iron
459 crystallized forms e.g. ferrihydrite or lepidocrocite (Ferreira et al. 2007), and which may explain the
460 dominance of the residual fraction. In the studied mangrove sediments, Fe partitioning in sediments
461 and Fe_D concentrations varied with depth and between stands (Fig. 4a and 5a), which could result
462 from the different redox conditions and driven by OM decomposition, bioturbation or root system
463 activities. Iron partitioning beneath the *Avicennia* stand and the mudflat was similar, but the one
464 beneath the *Rhizophora* stand was different and showed higher Fe concentrations in the oxidizable
465 fraction but lower dissolved iron concentrations and lower Fe concentrations in residual, reducible
466 and exchangeable/carbonate bound fractions. Thus, as Fe can form stable chelate complexes with
467 organic matter (Thamdrup 2000), the increase of TOC in anoxic condition which were observed
468 toward the landside of the mangrove could be responsible of the increasing Fe binding with organic
469 phases in the mangrove sediment (Chakraborty et al. 2016). This phenomenon could be affirmed
470 by negative correlation of Fe proportion in the oxidizable fraction with those in the reducible and
471 the residual phases ($r = - 0.67$ and $- 0.85$, Fig. 6a and 6b respectively). We also suggest that Fe-
472 oxihydroxydes originating from the lateritic soil of the watersheds were dissolved during diagenetic
473 processes because of increased organic content (Froelich et al. 1979) and that the dissolved iron
474 was subsequently complexed with OM as observed in other mangroves (Marchand et al. 2012). In
475 anoxic conditions, Fe_D can also precipitate as sulphides, which can be reflected by the increased
476 concentrations of Fe in the residual fraction at depth in the *Rhizophora* stand. Unfortunately, we
477 were not able to measure neither total sulfur (TS) nor sulphide in the studied mangrove sediments.
478 However, it is known that mangrove sediments are characterized by high rate of sulfate reduction

479 (Kristensen 2008b) and elevated TS content and pyrite are commonly observed in mangrove forest,
480 especially at depth (Marchand et al. 2011a, Marchand et al. 2006). Eventually and in addition to
481 these reduction processes, the re-oxidation of aqueous Fe(II) and pyrite could lead to the formation
482 of poorly ordered ferrihydrite, lepidocrocite (c-FeOOH) and likely goethite (Noël et al. 2014) in
483 the upper suboxic sediments, but cannot be confirmed with the selective extraction used.

484 *Manganese geochemistry*

485 Manganese is known as a highly redox sensitive element (Lacerda et al. 1999). Like for Fe, the
486 presence of Mn_D in the pore-waters may result from the reductive dissolution of Mn-oxyhydroxides
487 during the diagenetic processes (Froelich et al. 1979). This process can be responsible for the
488 presence of high Mn_D concentrations in pore-waters, reaching more than 10,000 µg L⁻¹ in the
489 mudflat (Fig. 5b). The lower Mn_D observed in the *Avicennia* and *Rhizophora* stands than in the
490 mudflat sediments may be related (i) to the loss of Mn_D by tidal drainage (Lacerda et al. 1999),
491 Mn_D release from mangrove pore-waters being as a significant component of Mn oceanic budget
492 (Holloway et al. 2016), (ii) to an uptake by mangrove plants (Wang et al. 2002), or iii) to more
493 intense reprecipitation of Mn with other bearing phases after oxyhydroxides dissolution. We
494 suggest that those processes were more pronounced for the *Rhizophora* stand, with 5-fold lower in
495 mean concentration of dissolved Mn than those in the *Avicennia* stand (Fig. 5b). However and
496 conversely to Fe, the dominant Mn fraction was the carbonate one in the two mangrove stands.
497 Unlike Fe, Mn sulphides are unstable ($pK_{sp} = 1.3$) and the precipitation of Mn as sulphides may be
498 severely limited by the presence of dissolved Fe²⁺ and carbonate. Because of similar ionic radii, Ca
499 can be substituted by Mn in carbonate minerals (Costa-Boddeker et al. 2017, Rath et al. 2009).
500 Previous studies, e.g. concerning mangrove sediments in the West coast of India (Noronha-D'Mello
501 et al. 2015) or in Brazilia (Otero et al. 2009), reported that a considerable quantity of the free
502 manganese was associated with carbonate (i.e. up to 2.5 µmol g⁻¹). Marchand et al. (2008) suggested
503 that the decomposition of organic matter in mangrove sediments leads to the production of
504 dissolved inorganic carbon (DIC) that can migrate at depth, where carbonate minerals can
505 precipitate because of the marked anoxic conditions that prevail there compared to the upper

506 suboxic layers. This carbonate precipitation can be a sink for some elements, including Mn. In the
507 *Rhizophora* stand, within the organic rich-layers, oxidizable fraction was the predominant one for
508 Mn, suggesting that Mn was adsorbed onto OM (Thamdrup 2000). This hypothesis may be
509 supported by negative correlation between Mn concentrations in the oxidizable fraction with those
510 in the reducible and in the residual phase (i.e. $r = -0.69$ and -0.79 , Fig. 6c and 6d respectively).

511 4.3.2. Geochemistry of Cu, Ni, Cr, Co, and As

512 Copper geochemistry

513 Copper may be released in pore-waters upon OM decay processes and/or reductive dissolution
514 of Fe-Mn oxyhydroxides under suboxic conditions. Cu is a chalcophile element, being easily
515 chemisorbed on or incorporated in several minerals such as chalcopyrite (CuFeS_2), covellite (CuS)
516 and malachite $\text{Cu}_2\text{CO}_3(\text{OH})_2$ (Pickering 1986). However, chalcopyrite and malachite are not stable
517 under acidic conditions, especially at pH values ranging from 5 to 6. Therefore, the high Cu
518 concentrations in the residual fraction beneath the *Avicennia* stand (where pH values < 6),
519 compared to the mudflat and the *Rhizophora* stand, may result from the intense precipitation of Cu
520 as sulphides, mainly CuS and/or Cu_2S (Fernandes 1997, Morse and Luther 1999). In the Can Gio
521 Mangrove sediment, the negative correlation observed between the organic fraction and the residual
522 one whatever the sites (i.e. $r = -0.89$, Fig. 6e), evidenced that the control of organic matter on Cu
523 partitioning (Chakraborty et al. 2015, Marchand et al. 2016, Silva et al. 2014) subsequently induces
524 the opposite trend of Cu concentrations in the other fractions (i.e. exchangeable/carbonate and
525 oxides bound, Fig. 4e). We note that beneath the *Rhizophora* stand, Cu concentrations in the organic
526 fraction were higher than in the two other sites (Fig. 4e). Interestingly, the Cu_D in the three cores
527 are in the same range than those measured in the Can Gio Mangrove Estuary surface water (Thanh-
528 Nho et al. 2018). Cu is known to be an essential element for plant growth (Yruela 2009) and to
529 bioaccumulate up to a bioconcentration factors of 9 in leaves and 5 in roots of mangrove trees
530 (MacFarlane et al. 2003, Usman et al. 2013). Therefore, its low contents in pore-waters in addition
531 to its precipitation with OM and sulphides may be related to its uptake by the mangroves root
532 systems (Chakraborty et al. 2014, MacFarlane and Burchett 2002). However, some peaks of

533 increasing Cu_D are observed at 5 cm, 10 cm and 20 cm depth ($1.4 \mu\text{g L}^{-1}$, $2.9 \mu\text{g L}^{-1}$ and $13.8 \mu\text{g L}^{-1}$, respectively) in the *Rhizophora* stand (Fig. 5e), which may be attributed to a limited re-
534 precipitation with oxides-bearing and supported by the Cu decreasing in the oxides fraction at these
535 layers (Fig. 4e).

537 *Nickel and Chromium geochemistry*

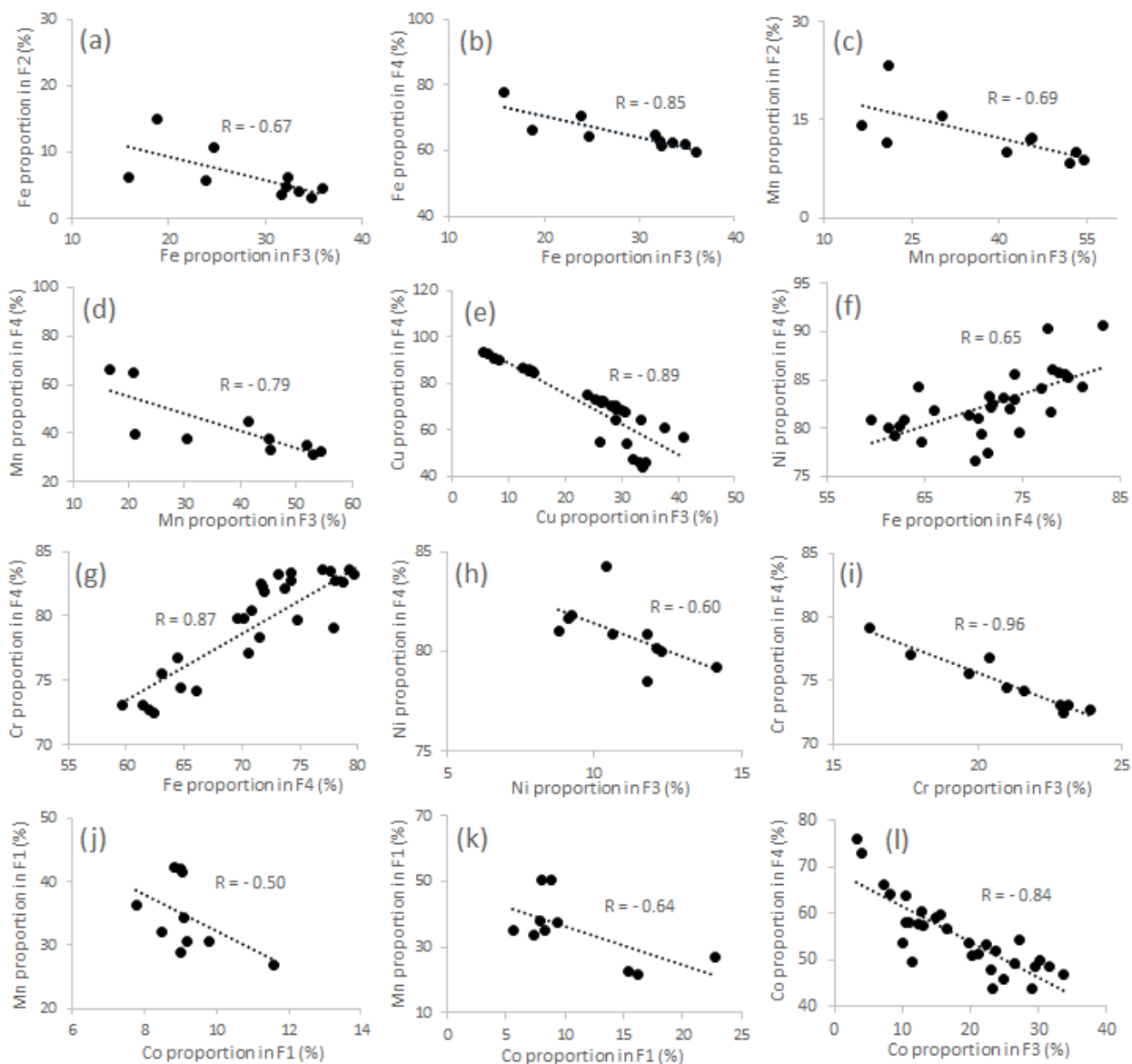
538 The high proportions of Ni and Cr in the residual fractions from the mudflat to the inner
539 mangrove stands may be attributed to more intense precipitation of Ni pyrite and/or Ni sulphide
540 (Clark et al. 1998, Noël et al. 2015), reflecting their natural sources linked to iron minerals. This
541 hypothesis could be supported by positive correlation between Ni, Cr and Fe proportions in the
542 residual fractions respectively ($r = 0.65$ and 0.87 , Fig. 6f and 6g). In the Can Gio Mangrove, the
543 OM increase in the *Rhizophora* stand's sediment implies intense diagenetic processes which could
544 lead to the formation of organocomplexes with dissolved Ni and Cr at the adequate pH (i.e. 3.4 %
545 of mean TOC concentrations and pH ranging from 6.4 to 6.8). Doig and Liber (2006) demonstrated
546 that the organonickel complexes may be formed at pH 6 to 8, which may explain the higher
547 concentrations of Ni in the organic fraction beneath *Rhizophora* than beneath the two other stands
548 (i.e. 11.4 % in the *Rhizophora* stand vs. 5.4 % in the *Avicennia* stand and 8.2 % in the mudflat, Fig.
549 4c). In fact, the variation of organic matter with depth in the *Rhizophora* stand induces the
550 modification of Ni partitioning between bearing phases (e.g. a negative correlation was observed
551 between Ni proportion in F3 and F4, $r = -0.6$, Fig. 6h). Similar phenomenon was observed for Cr
552 partitioning, being supported by a negative correlation between Cr proportion in F3 and F4 beneath
553 the *Rhizophora* stand, $r = -0.96$, Fig. 6i) like demonstrated in New Caledonia (Marchand et al.
554 2012). Lacerda et al. (1991) also observed in a Brazilian mangrove that Cr was immobilized in
555 sediment as organochromium complexes.

556 *Cobalt and Arsenic geochemistry*

557 Conversely to Cu, Ni, and Cr, the concentrations of Co and As in the exchangeable/carbonate
558 bound fraction were elevated, reaching up to 23 % and 5.4 % respectively. As explained earlier for
559 Mn partitioning and the presence of Mn-carbonates, we suggest that the decomposition of organic

560 matter can lead to DIC production, which can precipitate in specific redox-pH values, incorporating
561 some elements, mainly Mn but also Co and As. Co and Mn proportions in the
562 exchangeable/carbonate fraction showed the opposite distribution patterns with depth whatever the
563 sites (Fig. 4f) which implies considerable competition of dissolved Mn on the formation of
564 carbonate cobalt (i.e. negative correlation observed in the mudflat, $r = - 0.50$, Fig. 6j; in the
565 *Avicennia stand*, $r = - 0.64$, Fig. 6k). In fact, Co (III) can be substituted to Mn in Mn (III/IV) oxides
566 because of their similar ionic radii, (Manceau et al. 1997). This process can occur in the same Eh-
567 pH space in which Mn (II) oxidation occurs (Murray and Dillard 1979). However, it was not the
568 case for Co behavior in the *Rhizophora* stand which may be attributed to the influence of OM. As
569 a consequence, a negative correlation between the Co proportion in the residual fraction and the
570 organic one was observed ($r = - 0.83$, Fig. 6l), which evidenced one more time the key role of
571 organic matter in trace metal partitioning in mangrove sediments. Beneath the *Avicennia* stand, Co
572 concentrations in the organic fraction were the lowest of the 3 stands despite similar range of TOC
573 in the mudflat, which may result the precipitation of dissolved Co into sulphides forms (Charriau
574 et al. 2011). In the case of As, we suggest that the increasing concentrations of As in the residual
575 fraction with depth in every sites may result from the incorporation of As into pyrite. In fact, the
576 most common iron sulphide minerals have strong affinity for, and can incorporate large amounts
577 of arsenic in its structure, up to 10 wt % (Abraitis et al. 2004, Qiu et al. 2017). Noël et al. (2014)
578 stated that pyrite was the predominant Fe bearing phase at depth in mangrove sediments. Notably,
579 dissolved As concentrations were relatively high in the mudflat and at depth in the *Rhizophora*
580 stand, reaching almost $20 \mu\text{g L}^{-1}$ (Fig. 5g). We suggest that the presence of As_D in pore-waters
581 results from the reductive dissolution of As bound to oxides under suboxic/anoxic conditions
582 (Masscheleyn et al. 1991, Nickson et al. 2000). Recent studies also supported that the reduction of
583 Fe-oxyhydroxides was a cause of As release into ground water in Bangladesh (Anawar et al. 2003,
584 Zheng et al. 2004).

585



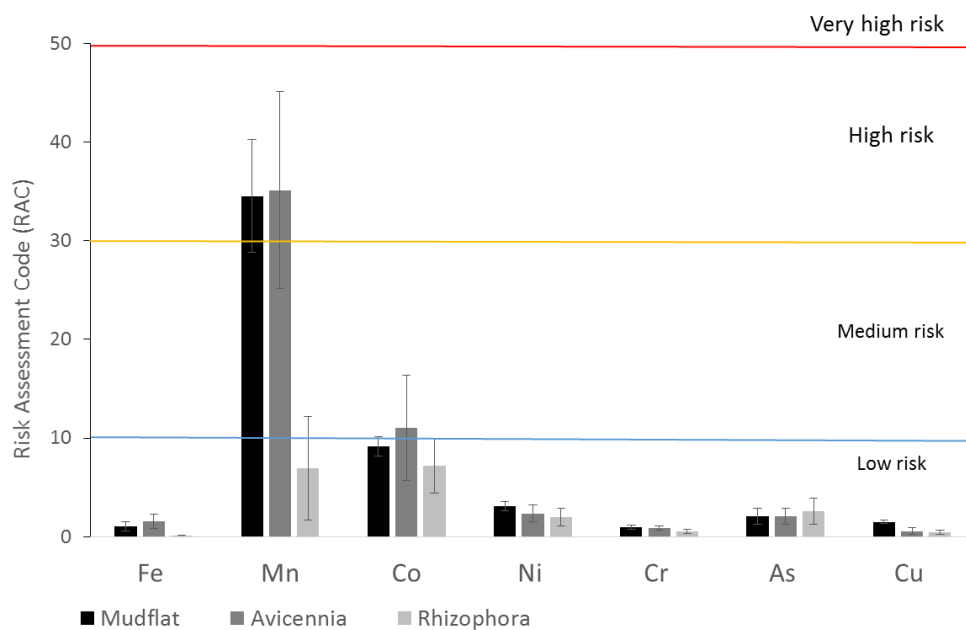
586 Fig. 6. Interrelationships of metals proportions in the different fractions of the sediment cores in Can Gio Mangrove:
 587 (a and b) Fe proportions beneath the *Rhizophora* stand; (c and d) Mn proportions beneath the *Rhizophora* stand; (e)
 588 data of Cu proportion beneath mangrove stands and mudflat; (f and g) Fe proportion in F4 correlated to Ni and Cr
 589 proportion in F4 of all sites from the mudflat to inner the *Avicennia* and *Rhizophora* stands, respectively; (h and i) Ni
 590 and Cr proportions beneath the *Rhizophora* stand; (j and k) Co and Mn proportions selected from the mudflat and
 591 the *Avicennia* stand, respectively; (l) Co proportions from all cores.

592 4.4. Assessment of metal stocks and ecological potential risks

593 The estimated metal stocks in the upper 50 cm sediment were lower in the *Rhizophora* stand
 594 compared to the *Avicennia* one and the mudflat and could be related to two processes. Firstly, the
 595 increase organic matter content in the sediment could induce a decrease in the bulk density of the
 596 sediments (0.55 g cm^{-3} in the *Rhizophora* vs. 0.61 g cm^{-3} in the *Avicennia* and 0.64 g cm^{-3} in the

597 mud flat). Secondly, it may be related to a more reactive substrate like described in the previous
 598 discussions, which can induce the dissolution of some bearing phases and the export of dissolved
 599 metals through pore-water seepage, or the uptake of dissolved metals by mangrove plants.
 600 Compared to mangrove of New Caledonia (Marchand et al. 2016), the estimated metal stocks of
 601 the Can Gio Mangrove sediments were lower confirming the moderate metal inputs in this system.

602 The bioavailability and potential toxicity of metals in the different geochemical fractions are
 603 expected to decrease in the following order: exchangeable/carbonate fraction > oxides fraction >
 604 organic fraction > residual fraction (Ma and Rao 1997). To assess the metal availability and
 605 associated potential ecological risks of the Can Gio Mangrove sediments, we used the Risk
 606 Assessment Code - RAC (Perin et al. 1985) and guideline (Benson et al. 2017, Passos et al. 2010)
 607 (Fig. 7).



608

609 *Fig. 7. Risk Assessment Code of metals in the sediment cores of the mudflat, the Avicennia stand and the*
 610 *Rhizophora stand in the Can Gio Mangrove.*

611 The RAC is evaluated based on the percentage of metal concentrations that is representative in
 612 the bioavailable fraction (exchangeable/carbonate bound) to the total trace metal concentrations.
 613 According to the RAC guideline, metal with less than 1 % of exchangeable/carbonate fraction
 614 would be at no risk to the environment while higher ratio suggest: 1 % to 10 %: low risk; 11 % to
 615 30 %: medium risk; 31 % to 50 %: high risk and > 50 %: very high risk. Consequently, in the Can

616 Gio Mangrove sediments, only Mn presented a high risk in the mudflat and in the *Avicennia* stand
617 (i.e. RAC > 33) and a low risk in the *Rhizophora* stand (i.e. RAC < 7). We note that in plants, Mn
618 excess can damage the photosynthesis apparatus (Millaleo et al. 2010) and thus the plants
619 productivity (Nguyen et al. 2018). Arsenic presented a low risk but its high dissolved concentrations
620 in pore-waters, reaching up to $\sim 20 \mu\text{g L}^{-1}$, may potentially be an additional ecological risk to
621 organisms. Ni, Co, Fe, and Cu showed low risks to the ecosystem while Cr was in the no risk
622 category whatever the sites. Along the mangrove, the RAC was lower in the *Rhizophora* stand than
623 in the *Avicennia* stand and the mudflat, which may result from the competition of organometallic
624 complexation (see earlier discussion), reflecting the important role of organic matter in scavenging
625 metals in mangrove sediments. Eventually, as a result of mangrove conversion for agriculture
626 and/or aquaculture (i.e. salt production and shrimp farming, etc.), these sediments may be subject
627 to an oxidation resulting in enhanced OM decomposition (Grellier et al. 2017), sulphide oxidation,
628 and thus sediments acidification. Noël et al. (2017) showed that increasing anthropogenic pressure
629 on coastal areas altered the redox state of mangrove sediments from reducing condition to oxidizing
630 condition, affecting the stability of Ni-accumulating Fe-sulfides and releasing significant dissolved
631 metals at the redox boundary. Consequently, metals which were associated with OM and sulphides
632 will be released in pore-waters, but also those associated with the exchangeable/carbonate bound
633 due to the acidification, conducting all metals to be potentially at high ecological risks in the Can
634 Gio Mangrove sediments.

635 **5. Conclusions**

636 The Can Gio Mangrove sediments did not present high metals concentrations. Their contents,
637 close to those of the crust, associated with their high proportion in residual fractions suggested that
638 studied metals originated from the lateritic soils of the Sai Gon and Dong Nai River watersheds.
639 These metals were deposited mainly as Fe-Mn oxyhydroxides in the sediment, which were
640 subsequently dissolved by bacteria in suboxic to anoxic conditions during diagenetic processes,
641 releasing them in pore-waters. The OM enrichment in the sediments from the mudflat to the
642 *Rhizophora* stand played a key role in metals partitioning, either because its decay modifies the

643 redox conditions (inducing oxyhydroxides dissolution and sulphides precipitation) or because of
644 the formation of organometallic compounds. Most metals presented low ecological risks to the
645 mangrove ecosystem except Mn, and possibly As due to its elevated dissolved concentrations. We
646 would like to underscore that any anthropogenic perturbation in the redox state of those mangrove
647 sediments may result in a release of metal contents in the adjacent ecosystems i.e. mangrove uptake
648 or export to surface water. We suggest that a further detailed investigation on dissolved metal
649 transfer and their accumulation into mangrove biota should be carried out to get a better
650 understanding of metal dynamic in the Can Gio Mangrove ecosystem.

651

652 **Acknowledgements**

653 The authors would like to thank the students Nguyen Ngoc Hon and Nguyen Truong Giang for
654 their help during sampling. This research is funded by Vietnam National University Ho Chi Minh
655 City (VNU-HCM) under grand number C2016-18-07.

656

657 **References**

- 658 Abraitis P., Pattrick R. and Vaughan D. (2004). Variations in the compositional, textural and electrical
659 properties of natural pyrite: a review. *International Journal of Mineral Processing* 74(1-4), 41-59.
- 660 Alongi D., Tirendi F. and Clough B. (2000). Below-ground decomposition of organic matter in forests of the
661 mangroves *Rhizophorastylosa* and *Avicenniamarina* along the arid coast of Western Australia. *Aquatic*
662 *Botany* 68(2), 97-122.
- 663 Anawar H. M., Akai J., Komaki K., Terao H., Yoshioka T., Ishizuka T., Safiullah S. and Kato K. (2003).
664 Geochemical occurrence of arsenic in groundwater of Bangladesh: sources and mobilization processes.
665 *Journal of Geochemical Exploration* 77(2-3), 109-131.
- 666 Babut M., Mourier B., Desmet M., Simonnet-Laprade C., Labadie P., Budzinski H., De Alencastro L. F., Tu T.
667 A., Strady E. and Gratiot N. (2019). Where has the pollution gone? A survey of organic contaminants in
668 Ho Chi Minh city / Saigon River (Vietnam) bed sediments. *Chemosphere* 217, 261-269.
- 669 Bayen S. (2012). Occurrence, bioavailability and toxic effects of trace metals and organic contaminants in
670 mangrove ecosystems: a review. *Environment International* 48, 84-101.
- 671 Benson N. U., Udosen E. D., Essien J. P., Anake W. U., Adedapo A. E., Akintokun O. A., Fred-Ahmadu O. H.
672 and Olajire A. A. (2017). Geochemical fractionation and ecological risks assessment of benthic
673 sediment-bound heavy metals from coastal ecosystems off the Equatorial Atlantic Ocean. *International*
674 *Journal of Sediment Research* 32(3), 410-420.
- 675 Blasco F., Aizpuru M. and Gers C. (2001). Depletion of the mangroves of Continental Asia. *Wetlands Ecology*
676 *and Management* 9, 245-256.
- 677 Bouillon S., Borges A. V., Castañeda-Moya E., Diele K., Dittmar T., Duke N. C., Kristensen E., Lee S. Y.,
678 Marchand C. and Middelburg J. J. (2008). Mangrove production and carbon sinks: a revision of global
679 budget estimates. *Global Biogeochemical Cycles* 22(2).
- 680 Chakraborty P., Chakraborty S., Jayachandran S., Madan R., Sarkar A., Linsy P. and Nath B. N. (2016). Effects
681 of bottom water dissolved oxygen variability on copper and lead fractionation in the sediments across

682 the oxygen minimum zone, western continental margin of India. *Science of The Total Environment* 566,
683 1052-1061.

684 Chakraborty P., Chakraborty S., Ramteke D. and Chennuri K. (2014). Kinetic speciation and bioavailability
685 of copper and nickel in mangrove sediments. *Marine Pollution Bulletin* 88(1-2), 224-230.

686 Chakraborty P., Ramteke D. and Chakraborty S. (2015). Geochemical partitioning of Cu and Ni in mangrove
687 sediments: relationships with their bioavailability. *Marine Pollution Bulletin* 93(1-2), 194-201.

688 Charriau A., Lesven L., Gao Y., Leermakers M., Baeyens W., Ouddane B. and Billon G. (2011). Trace metal
689 behaviour in riverine sediments: Role of organic matter and sulfides. *Applied Geochemistry* 26(1), 80-
690 90.

691 Clark M. W., McConchie D., Lewis D. W. and Saenger P. (1998). Redox stratification and heavy metal
692 partitioning in *Avicennia*-dominated mangrove sediments: a geochemical model. *Chemical Geology*
693 149(3-4), 147-171.

694 Cormier-Salem M.-C., Van Trai N., Burgos A., Durand J.-D., Bettarel Y., Klein J., Duc Huy H. and Panfili J.
695 (2017). The mangrove's contribution to people: Interdisciplinary pilot study of the Can Gio Mangrove
696 Biosphere Reserve in Viet Nam. *Comptes Rendus Geoscience* 349(6-7), 341-350.

697 Costa-Boddeker S., Hoelzmann P., Thuyen L. X., Huy H. D., Nguyen H. A., Richter O. and Schwalb A. (2017).
698 Ecological risk assessment of a coastal zone in Southern Vietnam: Spatial distribution and content of
699 heavy metals in water and surface sediments of the Thi Vai Estuary and Can Gio Mangrove Forest.
700 *Marine Pollution Bulletin* 114(2), 1141-1151.

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

703 Dent D. (1986). Acid sulphate soils: a baseline for research and development, ILRI.

704 Doig L. E. and Liber K. (2006). Nickel partitioning in formulated and natural freshwater sediments.
705 *Chemosphere* 62(6), 968-979.

706 Duke N. C., Meynecke J.-O., Dittmann S., Ellison A. M., Anger K., Berger U., Cannicci S., Diele K., Ewel K. C.
707 and Field C. D. (2007). A world without mangroves? *Science* 317(5834), 41-42.

708 Egawa T. and Ooba Y. (1963). Mineralogical studies of some soils in the central highland of vietnam. *Soil*
709 *Science and Plant Nutrition* 9(6), 14-20.

710 FAO (2014). *Aquastat: Global information system on water and agriculture*.
711 www.fao.org/nr/water/aquastat/data/cf/readPdf.html?f=VNM-CF_eng.pdf.

712 Fernandes H. M. (1997). Heavy metal distribution in sediments and ecological risk assessment: the role of
713 diagenetic processes in reducing metal toxicity in bottom sediments. *Environmental Pollution* 97(3),
714 317-325.

715 Fernandes L. L. and Nayak G. N. (2012). Heavy metals contamination in mudflat and mangrove sediments
716 (Mumbai, India). *Chemistry and Ecology* 28(5), 435-455.

717 Ferreira T. O., Otero X. L., Vidal-Torrado P. and Macías F. (2007). Effects of bioturbation by root and crab
718 activity on iron and sulfur biogeochemistry in mangrove substrate. *Geoderma* 142(1-2), 36-46.

719 Froelich P. N., Klinkhammer G., Bender M. L., Luedtke N., Heath G. R., Cullen D., Dauphin P., Hammond D.,
720 Hartman B. and Maynard V. (1979). Early oxidation of organic matter in pelagic sediments of the
721 eastern equatorial Atlantic: suboxic diagenesis. *Geochimica et Cosmochimica Acta* 43(7), 1075-1090.

722 Grellier S., Janeau J. L., Dang Hoai N., Nguyen Thi Kim C., Le Thi Phuong Q., Pham Thi Thu T., Tran-Thi N. T.
723 and Marchand C. (2017). Changes in soil characteristics and C dynamics after mangrove clearing
724 (Vietnam). *Science of The Total Environment* 593-594, 654-663.

725 Holloway C. J., Santos I. R., Tait D. R., Sanders C. J., Rose A. L., Schnetger B., Brumsack H.-J., Macklin P. A.,
726 Sippo J. Z. and Maher D. T. (2016). Manganese and iron release from mangrove porewaters: A
727 significant component of oceanic budgets? *Marine Chemistry* 184, 43-52.

728 Hong P. N. (2001). Reforestation of mangroves after severe impacts of herbicides during the the Viet Nam
729 war: the case of Can Gio. *Unasylva* (FAO).

730 Kristensen E. (2008b). Mangrove crabs as ecosystem engineers; with emphasis on sediment processes.
731 *Journal of Sea Research* 59(1-2), 30-43.

732 Kristensen E., Bouillon S., Dittmar T. and Marchand C. (2008a). Organic carbon dynamics in mangrove
733 ecosystems: A review. *Aquatic Botany* 89(2), 201-219.

734 Kuenzer C. and Tuan V. Q. (2013). Assessing the ecosystem services value of Can Gio Mangrove Biosphere
735 Reserve: Combining earth-observation- and household-survey-based analyses. *Applied Geography* 45,
736 167-184.

737 Kutscher D., Wills J. D. and McSheehy D. S. (2014). Analysis of High Matrix Samples using Argon Gas Dilution
738 with the Thermo Scientific iCAP Q ICP-MS. Technical Note 43202, Thermo Fisher Scientific, Bremen,
739 Germany.

740 Lacerda L., Rezende C., Aragon G. and Ovalle A. (1991). Iron and chromium transport and accumulation in
741 a mangrove ecosystem. *Water, Air, and Soil Pollution* 57(1), 513-520.

742 Lacerda L. D., Ribeiro Jr M. G. and Gueiros B. B. (1999). Manganese dynamics in a mangrove mud flat tidal
743 creek in SE Brazil. *Mangroves and Salt Marshes* 3, 105-115.

744 Lee S. Y., Primavera J. H., Dahdouh-Guebas F., McKee K., Bosire J. O., Cannicci S., Diele K., Fromard F.,
745 Koedam N., Marchand C., Mendelssohn I., Mukherjee N. and Record S. (2014). Ecological role and
746 services of tropical mangrove ecosystems: a reassessment. *Global Ecology and Biogeography* 23(7),
747 726-743.

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

750 Luong N. V. (2011). Mangrove forest structure and coverage change analysis using remote sensing and
751 geographical information system technology: A case study of Can Gio Mangrove Biosphere Reserve, Hi
752 Chi Minh City, Vietnam. Final report submitted to Rufford Small Grants Foundation, 40pp.

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

755 M. Brander L., J. Wagtendonk A., S. Hussain S., McVittie A., Verburg P. H., de Groot R. S. and van der Ploeg
756 S. (2012). Ecosystem service values for mangroves in Southeast Asia: A meta-analysis and value
757 transfer application. *Ecosystem Services* 1(1), 62-69.

758 Ma L. Q. and Rao G. N. (1997). Chemical fractionation of cadmium, copper, nickel, and zinc in contaminated
759 soils. *Journal of Environmental Quality* 26(1), 259-264.

760 MacFarlane G. and Burchett M. (2002). Toxicity, growth and accumulation relationships of copper, lead
761 and zinc in the grey mangrove *Avicennia marina* (Forsk.) Vierh. *Marine Environmental Research* 54(1),
762 65-84.

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

766 MacKenzie R. A., Foulk P. B., Klump J. V., Weckerly K., Purbospito J., Murdiyarso D., Donato D. C. and Nam
767 V. N. (2016). Sedimentation and belowground carbon accumulation rates in mangrove forests that
768 differ in diversity and land use: a tale of two mangroves. *Wetlands Ecology and Management* 24(2),
769 245-261.

770 Manceau A., Silvester E., Bartoli C., Lanson B. and Drits V. A. (1997). Structural mechanism of Co²⁺
771 oxidation by the phylломanganate buserite. *American Mineralogist* 82(11-12), 1150-1175.

772 Marchand C., Allenbach M. and Lallier-Vergès E. (2011b). Relationships between heavy metals distribution
773 and organic matter cycling in mangrove sediments (Conception Bay, New Caledonia). *Geoderma* 160(3-
774 4), 444-456.

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

778 Marchand C., Fernandez J. M. and Moreton B. (2016). Trace metal geochemistry in mangrove sediments
779 and their transfer to mangrove plants (New Caledonia). *Science of The Total Environment* 562, 216 -
780 227.

781 Marchand C., Fernandez J. M., Moreton B., Landi L., Lallier-Vergès E. and Baltzer F. (2012). The partitioning
782 of transitional metals (Fe, Mn, Ni, Cr) in mangrove sediments downstream of a ferralitized ultramafic
783 watershed (New Caledonia). *Chemical Geology* 300-301, 70-80.

784 Marchand C., Lallier-Vergès E. and Allenbach M. (2011a). Redox conditions and heavy metals distribution
785 in mangrove forests receiving effluents from shrimp farms (Teremba Bay, New Caledonia). *Journal of*
786 *Soils and Sediments* 11(3), 529-541.

787 Marchand C., Lallier-Vergès E. and Baltzer F. (2003). The composition of sedimentary organic matter in
788 relation to the dynamic features of a mangrove-fringed coast in French Guiana. *Estuarine, Coastal and*
789 *Shelf Science* 56(1), 119-130.

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

792 Marchand C., Lallier-Vergès E., Disnar J. R. and Kérais D. (2008). Organic carbon sources and
793 transformations in mangrove sediments: A Rock-Eval pyrolysis approach. *Organic Geochemistry* 39(4),
794 408-421.

795 Masscheleyn P. H., Delaune R. D. and Patrick Jr W. H. (1991). Effect of redox potential and pH on arsenic
796 speciation and solubility in a contaminated soil. *Environmental Science and Technology* 25(8), 1414-
797 1419.

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

801 Morse J. and Luther G. (1999). Chemical influences on trace metal-sulfide interactions in anoxic sediments.
802 *Geochimica et Cosmochimica Acta* 63(19), 3373-3378.

803 Murray J. W. and Dillard J. G. (1979). The oxidation of cobalt (II) adsorbed on manganese dioxide.
804 *Geochimica et Cosmochimica Acta* 43(5), 781-787.

805 Nguyen B. T., Do T. K., Tran T. V., Dang M. K., Dell C. J., Luu P. V. and Vo Q. T. K. (2018). High soil Mn and
806 Al, as well as low leaf P concentration, may explain for low natural rubber productivity on a tropical
807 acid soil in Vietnam. *Journal of plant nutrition*, 1-12.

808 Nguyen T. T. N., Nemery J., Gratiot N., Strady E., Tran V. Q., Nguyen A. T., Aime J. and Payne A. (2018).
809 Nutrient dynamics and eutrophication assessment in the tropical river system of Saigon - Dongnai
810 (southern Vietnam). *Science of The Total Environment* 653, 370-383.

811 Nickson R., McArthur J., Ravenscroft P., Burgess W. and Ahmed K. (2000). Mechanism of arsenic release to
812 groundwater, Bangladesh and West Bengal. *Applied Geochemistry* 15(4), 403-413.

813 Noël V., Juillot F., Morin G., Marchand C., Ona-Nguema G., Viollier E., Prévot F., Dublet G., Maillot F., Delbes
814 L., Marakovic G., Bargar J. R. and Brown G. E. (2017). Oxidation of Ni-Rich Mangrove Sediments after
815 Isolation from the Sea (Dumbea Bay, New Caledonia): Fe and Ni Behavior and Environmental
816 Implications. *ACS Earth and Space Chemistry* 1(8), 455-464.

817 Noël V., Marchand C., Juillot F., Ona-Nguema G., Viollier E., Marakovic G., Olivi L., Delbes L., Gelebart F.
818 and Morin G. (2014). EXAFS analysis of iron cycling in mangrove sediments downstream a lateritized
819 ultramafic watershed (Vavouto Bay, New Caledonia). *Geochimica et Cosmochimica Acta* 136, 211-228.

820 Noël V., Morin G., Juillot F., Marchand C., Brest J., Bargar J. R., Muñoz M., Marakovic G., Ardo S. and Brown
821 G. E. (2015). Ni cycling in mangrove sediments from New Caledonia. *Geochimica et Cosmochimica Acta*
822 169, 82-98.

823 Noronha-D'Mello, A. C. and Nayak G. N. (2015). Geochemical characterization of mangrove sediments of
824 the Zuari estuarine system, West coast of India. *Estuarine, Coastal and Shelf Science* 167, 313-325.

825 Otero X. L., Ferreira T. O., Huerta-Díaz M. A., Partiti C. S. M., Souza V., Vidal-Torrado P. and Macías F. (2009).
826 Geochemistry of iron and manganese in soils and sediments of a mangrove system, Island of Pai Matos
827 (Cananeia — SP, Brazil). *Geoderma* 148(3-4), 318-335.

828 Parvaresh H., Abedi Z., Farshchi P., Karami M., Khorasani N. and Karbassi A. (2011). Bioavailability and
829 concentration of heavy metals in the sediments and leaves of grey mangrove, *Avicennia marina* (Forsk.)
830 Vierh, in Sirik Azini Creek, Iran. *Biological trace element research* 143(2), 1121-1130.

831 Passos E. d. A., Alves J. C., dos Santos I. S., Alves J. d. P. H., Garcia C. A. B. and Spinola Costa A. C. (2010).
832 Assessment of trace metals contamination in estuarine sediments using a sequential extraction
833 technique and principal component analysis. *Microchemical Journal* 96(1), 50-57.

834 Perin G., Craboledda L., Lucchese M., Cirillo R., Dotta L., Zanette M. and Orio A. (1985). Heavy metal
835 speciation in the sediments of northern Adriatic Sea. A new approach for environmental toxicity
836 determination. *Heavy metals in the environment* 2(1), 454-456.

837 Pickering W. (1986). Metal ion speciation—soils and sediments (a review). *Ore Geology Reviews* 1(1), 83-
838 146.

839 Qiu G., Gao T., Hong J., Tan W., Liu F. and Zheng L. (2017). Mechanisms of arsenic-containing pyrite
840 oxidation by aqueous arsenate under anoxic conditions. *Geochimica et Cosmochimica Acta* 217, 306-
841 319.

842 Rath P., Panda U. C., Bhatta D. and Sahu K. C. (2009). Use of sequential leaching, mineralogy, morphology
843 and multivariate statistical technique for quantifying metal pollution in highly polluted aquatic
844 sediments--a case study: Brahmani and Nandira Rivers, India. *Journal of Hazardous and Materials*
845 163(2-3), 632-644.

846 Ross P. (1975). The mangroves of South Vietnam: the impact of military use of herbicides. Proceedings of
847 the international symposium on biology and management of mangroves. Gainesville, Florida: Institute
848 of Food and Agricultural Sciences, University of Florida.

849 Sanders C. J., Santos I. R., Barcellos R. and Silva Filho E. V. (2012). Elevated concentrations of dissolved Ba,
850 Fe and Mn in a mangrove subterranean estuary: Consequence of sea level rise? *Continental Shelf*
851 *Research* 43, 86-94.

852 Silva G. S. d., Nascimento A. S. d., Sousa E. R. d., Marques E. P., Marques A. L. B., Corrêa L. B. and Silva G.
853 S. d. (2014). Distribution and Fractionation of Metals in Mangrove Sediment from the Tibiri River
854 Estuary on Maranhão Island. *Revista Virtual de Química* 6(2).

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

859 Tam N. and Wong Y. (1996). Retention and distribution of heavy metals in mangrove soils receiving
860 wastewater. *Environmental Pollution* 94(3), 283-291.

861 Tessier A., Campbell P. G. and Bisson M. (1979). Sequential extraction procedure for the speciation of
862 particulate trace metals. *Analytical Chemistry* 51(7), 844-851.

863 Thamdrup B. (2000). Bacterial manganese and iron reduction in aquatic sediments. *Advances in microbial*
864 *ecology*, Springer, 41-84.

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

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

870 Ure A. M., Quevauviller P., Muntau H. and Griepink B. (1993). Speciation of heavy metals in soils and
871 sediments - an account of the improvement and harmonization of extraction techniques undertaken
872 under the auspices of the BCR of the commission of the European communities. *International Journal*
873 *of Environmental Analytical Chemistry* 51(1-4), 135-151.

874 Usman A. R., Alkredaa R. S. and Al-Wabel M. I. (2013). Heavy metal contamination in sediments and
875 mangroves from the coast of Red Sea: *Avicennia marina* as potential metal bioaccumulator.
876 *Ecotoxicology and Environmental Safety* 97, 263-270.

877 Wang Z., Shan X.-q. and Zhang S. (2002). Comparison between fractionation and bioavailability of trace
878 elements in rhizosphere and bulk soils. *Chemosphere* 46(8), 1163-1171.

879 Yruela I. (2009). Copper in plants: acquisition, transport and interactions. *Functional Plant Biology* 36(5),
880 409-430.

881 Zhang C., Yu Z. G., Zeng G. M., Jiang M., Yang Z. Z., Cui F., Zhu M. Y., Shen L. Q. and Hu L. (2014). Effects of
882 sediment geochemical properties on heavy metal bioavailability. *Environment International* 73, 270-
883 281.

884 Zheng Y., Stute M., Van Geen A., Gavrieli I., Dhar R., Simpson H., Schlosser P. and Ahmed K. (2004). Redox
885 control of arsenic mobilization in Bangladesh groundwater. *Applied Geochemistry* 19(2), 201-214.

886