



HAL
open science

Contrasting organic matter composition in pristine and eutrophicated mangroves revealed by fatty acids and stable isotopes (Rio de Janeiro, Brazil)

Mathias Chynel, Sofia Rockomanovic, Gwenaël Abril, Glenda Barroso, Humberto Marotta, Wilson Machado, Christian Sanders, Najet Thiney, Tarik Meziane

► To cite this version:

Mathias Chynel, Sofia Rockomanovic, Gwenaël Abril, Glenda Barroso, Humberto Marotta, et al.. Contrasting organic matter composition in pristine and eutrophicated mangroves revealed by fatty acids and stable isotopes (Rio de Janeiro, Brazil). *Estuarine, Coastal and Shelf Science*, 2022, 277, pp.108061. 10.1016/j.ecss.2022.108061 . hal-03797455

HAL Id: hal-03797455

<https://cnrs.hal.science/hal-03797455>

Submitted on 4 Oct 2022

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 **Title**

2 Contrasting organic matter composition in pristine and eutrophicated mangroves
3 revealed by fatty acids and stable isotopes (Rio de Janeiro, Brazil)

4

5 **Authors.**

6 Mathias Chynel^{1,*}, Sofia Rockomanovic¹, Gwenaël Abril^{1,2}, Glenda Barroso², Humberto
7 Marotta^{2,3}, Wilson Machado², Christian J. Sanders⁴, Najet Thiney¹, Tarik Meziane¹

8

9 **Affiliations**

10 ¹Muséum National d'Histoire Naturelle, Laboratoire Biologie des Organismes et
11 Ecosystème Aquatiques (UMR 8067 BOREA), Sorbonne Université, CNRS, IRD 207,
12 Université de Caen Normandie, Université des Antilles, 61 rue Buffon, 75005 Paris,
13 France

14 ²Programa de Pós-Graduação em Geociências (Geoquímica Ambiental),
15 Universidade Federal Fluminense, Niterói, RJ, Brazil

16 ³Programa de Pós-Graduação em Geografia, Universidade Federal Fluminense,
17 Niterói, RJ, Brazil

18 ⁴National Marine Science Centre, Faculty of Science and Engineering, Southern Cross
19 University PO Box 4321, Coffs Harbour, 2450 NSW, Australia

20 *Corresponding author: mathias.chynel@edu.mnhn.fr

21

22

23 **Abstract**

24 Mangrove sediments have a high capacity of carbon storage, as the result of larger
25 organic matter (OM) inputs from mangrove trees (litter fall and fine roots production)
26 than OM microbial degradation and export to coastal waters. Mangrove sediments also
27 act as traps for suspended matter and particulate OM (POM) from surrounding water
28 masses. Fatty acids (FAs) markers, $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ signatures were used here to
29 characterize the OM composition in three mangroves located in three coastal
30 embayments of the Rio de Janeiro state (Brazil) with increasing urbanization from a
31 pristine mangrove M1 to a moderately impacted mangrove M2 and a highly impacted
32 mangrove M3. In these mangroves, the $\delta^{15}\text{N}$ signature of tree leaves and sediments
33 increases with anthropogenic influence, consistent with a large-scale eutrophication
34 gradient along the three regions. At mangrove M1, predominant OM inputs from
35 mangrove trees are highlighted by high proportions of long-chain fatty acids,
36 particularly in the inland station, where high organic carbon concentrations (126 ± 108
37 $\text{mg}\cdot\text{g}^{-1}$) indicate limited sedimentation of mineral particles and high carbon storage
38 capacities. The sedimentary OM of M3 mangrove was more labile as confirmed by the
39 higher proportions of algal fatty acids, enriched $\delta^{13}\text{C}$ signature and the C/N ratio of 1.6
40 times lower ($p < 0.001$) than in the pristine mangrove M1. At the M2 mangrove site,
41 high contribution of bacterial FAs (around 20%) to sedimentary OM and high proportion
42 of poorly biodegradable saturated fatty acids suggest that bacteria degrade algal labile
43 OM in surface sediments but do not mineralize the most refractory fraction of OM. At
44 the eutrophic M3 site, our findings suggest that deposition of labile POM induced an
45 increase of fungal biomass on the sediment, apparently enhancing the microbial loop,
46 and potentially leading to mineralization of refractory OM and carbon losses through a
47 priming effect.

48 **Keywords**

49 Blue carbon; Organic matter biodegradability; Cultural eutrophication; Microalgal
50 blooms; Fungi; Bacteria.

51 **1. Introduction**

52 Mangrove forests are coastal ecosystems characterized by the presence of woody
53 plants tolerant to extreme environmental conditions such as high salinity and sediment
54 anoxia. These forest systems are present in tropical and subtropical areas, and
55 colonize a worldwide surface area between 137 600 km² (Bunting et al., 2018) and 157
56 000 km² (FAO, 2007). Brazil is the country with the 3rd largest mangrove cover on the
57 globe, which representing 8.5% of the total surface area of mangroves worldwide
58 (Webber, 2016).

59 Mangrove forests have an average net primary production (NPP) between 1360 ± 450
60 g C m⁻² year⁻¹ (Bouillon et al., 2008a) and 1522 g C m⁻² year⁻¹ (Alongi, 2014), that is, 4
61 to 7 times higher than the NPP by coastal marine phytoplankton (Cloern et al., 2014).
62 An important fraction of the OM produced by mangrove NPP is transferred to their
63 sediments as litter fall and fine roots production, fuelling microbial respiration and CO₂
64 and CH₄ emission from the sediments to waters and the atmosphere (Alongi, 2020;
65 Maher et al., 2018). However, anoxic conditions in the sediments slow down the
66 degradation rates of OM and favour its preservation and burial over the long term. In
67 addition, due to tidal inundation, exchange of organic and inorganic C occurs between
68 mangrove sediments and the surrounding coastal waters (David et al., 2018; Maher et
69 al., 2018; Santos et al., 2019). Recent estimates indicate that C accumulation in
70 mangrove sediments may reach 41 Mt annually (Wang et al., 2021), representing a
71 long-term C burial with high impact on global carbon budget and global warming
72 mitigation (Kristensen et al., 2008a). Mangroves account for 15% of the "blue carbon"
73 actually buried among all coastal marine ecosystems (Anand et al., 2020).

74 The composition of mangrove sedimentary organic matter (SOM) is strongly influenced
75 by local sources of OM (Bouillon et al., 2003). It is generally assumed that litterfall and

76 roots from mangrove trees are the most abundant sources of SOM (Alongi, 2014),
77 largely predominant in pristine mangroves (Sanders et al., 2014). However,
78 microphytobenthos and phytoplankton as well as their detritus in the particulate organic
79 matter (POM) may be an additional source of SOM produced on the mangrove surface
80 sediment and/or brought from the surrounding coastal waters and deposited by the
81 tide (David et al., 2018). Increasing human densities along coastal zones and the
82 intensification of agricultural activities are increasing runoff of OM and nutrient to the
83 coastal zone (Kitsiou & Karydis, 2011; Sanders et al., 2014). This leads in highly
84 anthropized coastal areas to a proliferation of phytoplankton, microphytobenthos and
85 macroalgae that can enter the mangrove surface sediment (Pérez et al., 2018a). A
86 third source is the direct discharge of untreated sewage which is also common in
87 anthropized watersheds (Kitsiou & Karydis, 2011).

88 At low tide, direct exposure of highly surface sediments to the atmosphere leads to the
89 oxygenation of these sediments contributing to intense microbial remineralization of
90 SOM in surface sediments (Sun et al., 2002). Tides also drive mangrove POM exports
91 to adjacent creeks (Meziane & Tsuchiya, 2002). In mangrove sediments, deeper layers
92 are typically depleted in oxygen and SOM is degraded by anaerobic processes (e.g.
93 reductions of nitrate, manganese, iron or sulfides, and methanogenesis), which are
94 less efficient than aerobic remineralization (Sun et al., 1993). Bioturbating organisms
95 (e.g. crabs) and the presence of roots contribute to increased sediment oxidation and
96 subsequent remineralization of SOM (Kristensen & Alongi, 2006; Zhu et al., 2018;
97 Sarker et al., 2021). Indeed, mangrove sediments emit to the atmosphere at low tide
98 and export with tidal pumping, significant amounts of CO₂ and CH₄ coming from the
99 mineralisation of SOM (Kristensen et al., 2008b; Pongparn et al., 2009; Nóbrega et
100 al., 2016).

101 In addition to the quantity of OM deposited, the quality of SOM is a crucial factor that
102 control remineralization or preservation. Labile compounds (e.g. algal or urban
103 domestic OM) are more easily degraded than refractory compounds (e.g. lignin,
104 cellulose from the mangrove forest) (Fontaine et al., 2003; Guenet et al., 2010).
105 Previous evidence has indicated that the production of labile OM by microalgal blooms
106 can also potentially generate an intensification of the refractory OM remineralization
107 (Gontikaki et al., 2015), through a phenomenon called the priming effect (Bianchi,
108 2011; Gontikaki et al., 2015). Further research is needed to better understand the
109 changes of the OM quality under increasing eutrophic condition over time and
110 particularly the role of microorganisms in sediment OM degradation (Bianchi, 2011;
111 Bouillon et al., 2008a; Lovelock et al., 2017; Queiroz et al., 2020).

112 Fatty acid and stable isotope composition of organic substrates are useful to identify
113 different POM and SOM sources and are also proxies of the eutrophication process in
114 a wide variety of ecosystems (Bergamino et al., 2014). FAs contribute to between 0.1
115 and 5% in coastal marine SOM (Canuel, 2001), between 10 and 30% in POM
116 (Wakeham et al., 1997) and between 5 and 25% in DOM (Nebbioso & Piccolo, 2013).
117 Some FAs are synthesized by specific groups of organisms and others are synthesized
118 in most organisms and in specific proportions (Dalsgaard et al., 2003; Kaneda, 1991).
119 FAs are intensively produced and degraded by organisms by heterotrophic micro and
120 macrofauna, especially the polyunsaturated FAs (Wakeham et al., 1997; Camacho-
121 Ibar et al., 2003). FAs are also powerful taxonomic indicators (from the species to
122 class) of 1) the living biomass in algal blooms, 2) sewage OM degradability and 3)
123 bacterial and fungal components within sediments and biofilms in the ecosystem
124 (Meziane & Tsuchiya, 2002; Xu & Jaffé, 2007; Kopprio et al., 2018). Consequently,
125 FAs are an efficient tool to trace the more labile part of SOM and POM, especially

126 when combined with bulk isotope C and N signature. Moreover, FAs can provide semi-
127 quantitative information on various sources to the OM pool.

128 Despite research attesting to coastal eutrophication around the world (Breitburg et al.,
129 2018), little is known on its impact on SOM composition in mangrove sediments. The
130 coast of Rio de Janeiro state (Brazil) shows contrasting conserved and highly
131 eutrophicated areas due to the discharge of untreated urban sewage (Cotovicz et al.,
132 2018; Ribeiro & Kjerfve, 2002). The aim of this study is to characterize the source and
133 fate of OM in three mangroves within the state of Rio de Janeiro (Brazil) along a
134 eutrophication gradient. For these purposes, the present study uses fatty acids (FAs)
135 and the isotopic signatures of carbon and nitrogen ($\delta^{13}\text{C}$ and $\delta^{15}\text{N}$) to assess the origin
136 and transfers of OM in mangrove sediments located along a gradient of urbanization
137 and eutrophication.

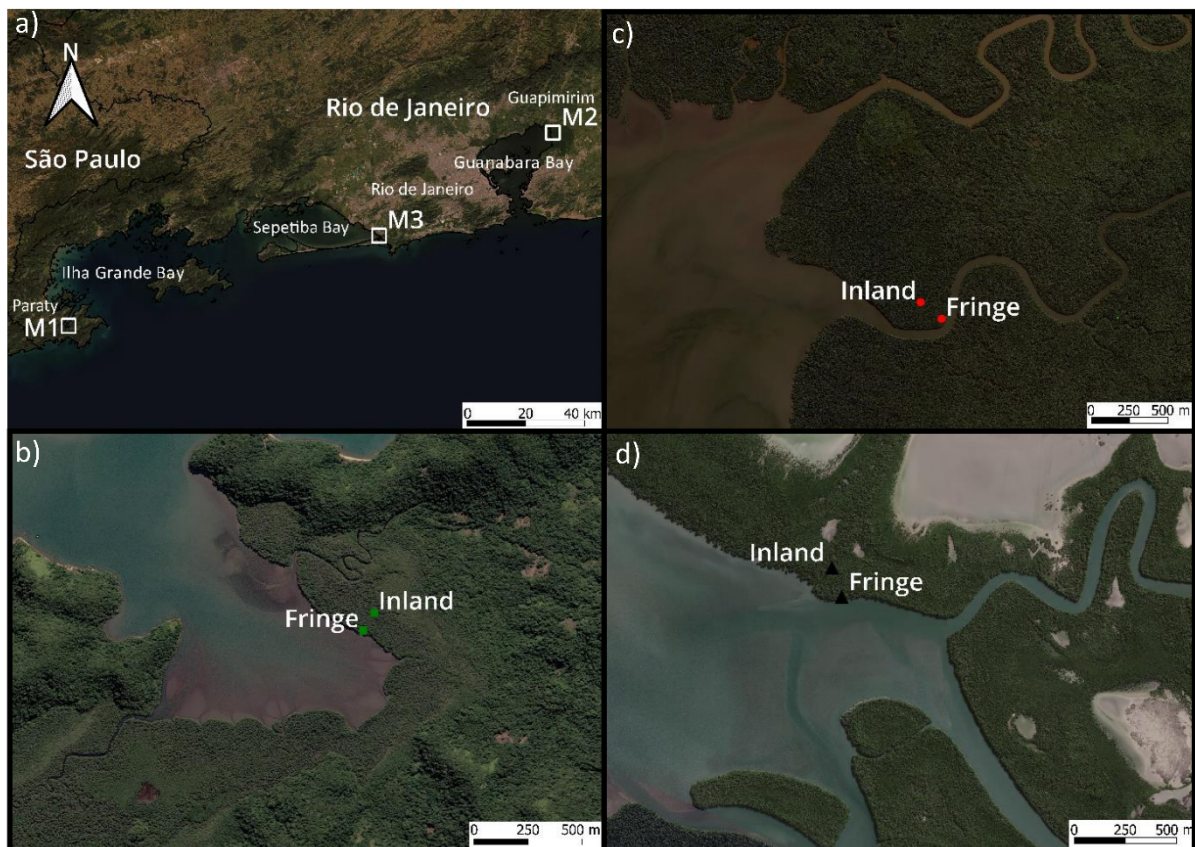
138

139 **2. Materials and Methods**

140 2.1 Study sites

141 The three studied mangroves in the State of Rio de Janeiro (Brazil) (Fig. 1) were
142 chosen because of their different levels of urban influence. Indeed, in the State of Rio
143 de Janeiro the small increase in wastewater treatment combined with the very large
144 increase in population leads to a wastewater discharge proportional to the watershed
145 population (Abril et al., 2022), thus highly polluted rivers discharge organic matter,
146 nitrogen and phosphorus especially in the bays of Guanabara and Sepetiba (Fistarol
147 et al., 2015; Rodrigues et al., 2009). In contrast, the southwestern region of the state
148 is not densely populated and the bay of Ilha Grande remains relatively pristine (Fig.
149 1A). The most pristine studied mangrove M1 is located at the back end of an 8km-long

150 elongated embayment called *Saco de Mamangua* with a very small watershed
151 containing a limited population (Table A1), and classified as an environmental reserve.
152 Although the entire drainage basin of Guanabara Bay is one of most urbanized in
153 Brazil, with an estimated population ~17 million compared to ~1.8 million in Sepetiba
154 Bay, the immediate watershed of the M2 mangrove in the region called *Guapi Mirim* is
155 larger but less populated than that of the M3 mangrove in the area called *Guaratiba*
156 (Table A1; Barroso et al. revised). Accordingly to the microtopography data of Barroso
157 et al. (revised), the estimated inundation times of these mangroves were relatively
158 similar. Finally, the $\delta^{15}\text{N}$ values of sediments and mangrove leaves attest to the
159 differences in eutrophication between the three sites (Table A1). The plant community
160 composition of each site is displayed in table A1.



161
162 *Figure 1 : Location of the tree studied mangroves at Rio de Janeiro (a), southeastern Brazil. (b) M1*
163 *in Ilha Grande Bay, (c) M2 in Guanabara Bay and M3 (d) in Sepetiba Bay. (2 column)*

164 2.2 Sampling

165 Samples from each of the study sites were collected during the rainy season between
166 October 11 and December 7, 2018. At each site, three zones were sampled: water
167 column (adjacent creek or channel), zone F corresponding to the mangrove fringe and
168 zone I corresponding to most inland location of the mangrove forest. The fringe
169 corresponds to the nearest part (about 20m) to the coast and the inland zone
170 corresponds to the part a furthest within the forest (between 100 and 200 meters).

171 Sediment samples were collected at low tide at the three study sites. Four 9cm-long
172 cores were taken at each of the 6 zones. Three sediment depths were sectioned: 0-2
173 cm (P0), 3-4 cm (P3) and 7-9 cm (P9). In order to characterize potential OM sources,
174 fresh leaves of *A.shaueriana*, *R.mangle* et *L.racemosa* were taken directly from the
175 trees. Samples from the water column were taken as 5 replicates. Water and material
176 collected with a 73 μm mesh plankton net were filtered on GF/F filters (0.7 μm). For
177 M3, because suspended material was very concentrated, the material from the
178 plankton nets was not filtered on GF/F filters but separated by decantation. For M2,
179 the water column samples were taken in two distinct areas: in the proper Guanabara
180 Bay (2B) at about 5km from the mangrove, and in the river channel (2C) during a rising
181 tide. All samples were frozen the day of sampling and were freeze-dried within three
182 days of collection.

183 2.3. Laboratory analysis

184 2.3.1. Fatty Acids

185 Prior to extraction, an internal standard (Tricosanoic acid: 23:0) was added in each
186 sample. Lipids were extracted according to the protocol of Bligh and Dyer as modified
187 by Meziane et al. (2007). Samples are extracted by sonication (20 minutes) a first time

188 in a mixture of chloroform, water and methanol (1:1:2; v:v:v). 1 mL chloroform and 1
189 mL water are added and the samples are vortexed and centrifuged 5 min at 3000 rpm.
190 The lipids are retained in the chloroform phase, which is collected. 2 mL chloroform is
191 added to the samples before a second extraction with the sonicator (20 min). The
192 samples are centrifuged again, then the chloroform is collected and the total 4 mL is
193 evaporated under nitrogen (N₂) flow. Lipids are saponified by adding a
194 methanol:sodium hydroxide (2N) mixture (2:1; v:v). Samples are put at 90°C for 1h30
195 then 500 µl of chloridric acid (37%) and 1.5 ml of CHCl₃ are added. The tubes are then
196 vortexed and centrifuged (3000 rpm). The chloroform phase is collected and
197 evaporated under nitrogen flow. Samples are methylated to form fatty acid methyl ester
198 (FAME) with 1 mL boron trifluoride and then placed in a dry bath for 10 min at 90°C. 1
199 mL H₂O and 1mL CHCl₃ are added. They are vortexed and centrifuged and then part
200 of the surface water is pipetted and discarded in order to rinse the solution well and
201 purify the final fatty acid extract. 1 mL H₂O and 1 mL CHCl₃ are added again and the
202 tubes are vortexed and centrifuged for 5 min (3000 rpm). All the chloroform containing
203 FAME is removed and placed in 2 mL vials. The vials are stored in the freezer at -
204 20°C. FAME are quantified by gas chromatography (Varian 3800-GC) with an ionizing
205 flame detector. The oven temperature is maintained at 60°C for 1 min, then it increases
206 to 150° (40°/min) for 3 min and then to 240° (3°/min) for 25 min. Fatty acids are
207 identified with a mass spectrometer (Varian 220-MS) and the comparison of fatty acid
208 retention times with the retention times of a commercial standard (Supelco® 37). We
209 report the values as % of total FA.

210 2.3.2. Stable isotopes

211 For $\delta^{13}\text{C}$ analysis, sediment and filter samples were acidified in order to eliminate
212 carbonates. For sediment samples, 10% HCl was directly added to the samples during

213 24 hours. Filter samples were fumigated by adding chlorhydric acid (37%). The filters
214 were exposed to the fumes generated by the acid for 6 hours.

215 Acidification can distort the values of $\delta^{15}\text{N}$, therefore the measurements of $\delta^{15}\text{N}$ were
216 performed separately for filters and sediments of which the 10% HCl was not applied.
217 About 15 mg of sediments and suspended material scratched from the filters were
218 weighted and stored in tin capsules.

219 The samples were analyzed at the University of California at the Davis Stable Isotopes
220 Facility (Department of Plant Sciences, University of California, Davis, California) with
221 an Elementar Vario EL Cube or Micro Cube Elemental analyzer (Elementar Analysen
222 systeme GmbH, Hanau, Germany) combined with a PDZ Europa 20-20 isotope ratio
223 mass spectrometer (Sercon Ltd., Cheshire, UK). Stable isotope results are reported in
224 parts per thousand (‰), using the standard delta notation ($\delta^{13}\text{C}$ and $\delta^{15}\text{N}$) relative to
225 international standards: VPDB (Vienna PeeDee Belemnite) for carbon and
226 atmospheric air for nitrogen.

227 2.4. Statistical analysis

228 Due to the too small number of replicates (4 replicates), statistical analyses were
229 performed using non-parametric tests. ANOSIMs (Clarke, 1993) with Bray-Curtis
230 distance were done to analyze the effect of depth, sampling site and sampling zone
231 within mangrove on fatty acids profiles (significance level $p < 0.05$). To characterize the
232 effect of depth, sampling site and sampling zone within mangrove in individual fatty
233 acid distributions, pairwise Wilcoxon tests (Bauer, 1972) were performed (significance
234 level $p < 0.05$).

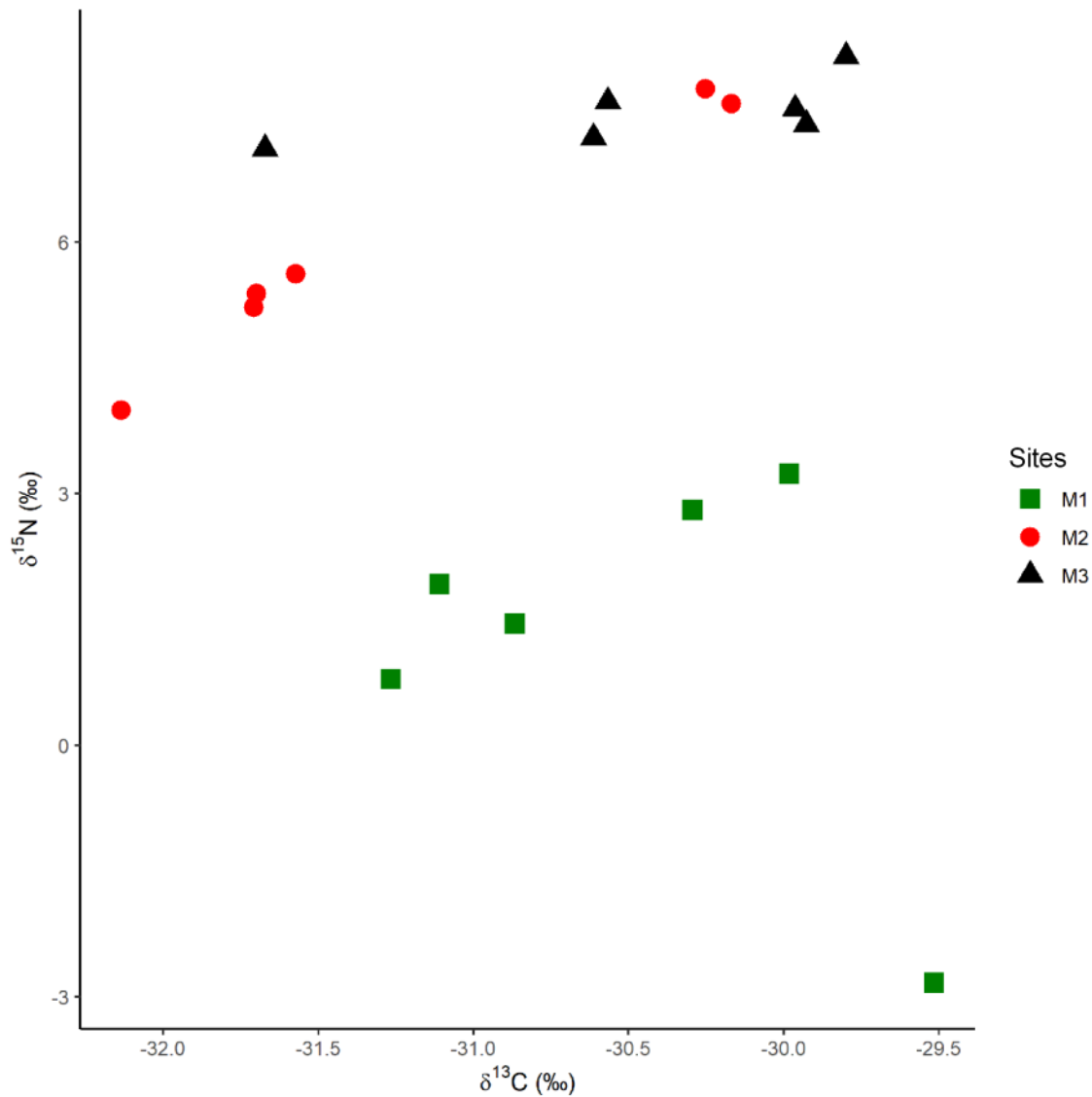
235 Data were analyzed with R software (version 4.0.5). Wilcoxon-Mann-Whitney
236 comparisons were performed with the "stat" package. ANOSIMs analysis were
237 performed with the package "vegan" (Oksanen et al., 2015).

238 **3. Results**

239 3.1. Isotopic composition and C/N ratio

240 3.1.1. Leaves

241 The $\delta^{13}\text{C}$ signature of mangrove leaves was relatively similar between the three sites
242 (mean \pm SD: $29.9 \pm 1.4\text{‰}$; Fig. 2). Conversely, $\delta^{15}\text{N}$ was higher in leaves from M3
243 ($7.3 \pm 0.4\text{‰}$) than in leaves of M2 ($5.6 \pm 1.4\text{‰}$; $p < 0.01$) and higher in leaves of M2 than
244 in those of M1 ($1.1 \pm 1.6\text{‰}$; $p < 0.001$).



246

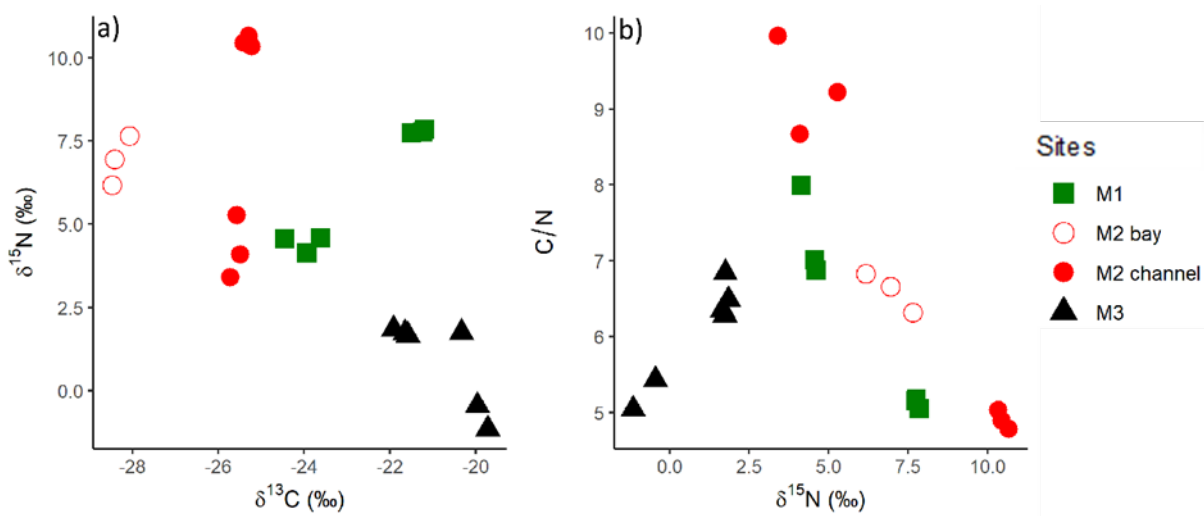
247 *Figure 2: Plot of (a) $\delta^{13}\text{C}$ vs $\delta^{15}\text{N}$ values of mangroves leaves in the three mangroves sites: M1 in*
 248 *Ilha Grande Bay, M2 in Guanabara Bay and M3 in Sepetiba Bay. (1 Column)*

249

250 3.1.2. Particulate organic matter (POM)

251 The characterization of isotopic signatures and C/N ratio of POM were performed from
 252 both plankton nets and GF/F filters for each zone. At the site M2, two areas were
 253 sampled: the channel and the bay (Fig. 3).

254 The $\delta^{13}\text{C}$, $\delta^{15}\text{N}$ signatures and C/N ratios of POM are presented in Figure 3. The $\delta^{13}\text{C}$
 255 values in M2 reached $-28.3\pm 0.2\text{‰}$. The $\delta^{13}\text{C}$ values were significantly depleted at M2
 256 channel ($-25.5\pm 0.2\text{‰}$) than in M1 and M3 ($-22.7\pm 1.5\text{‰}$ and $-20.9\pm 1.0\text{‰}$; $p < 0.01$). The
 257 $\delta^{13}\text{C}$ signature of POM was similar at M1 and M3 ($p > 0.05$). The $\delta^{15}\text{N}$ of POM was
 258 higher at M1 and M2 ($6.1\pm 1.9\text{‰}$ and $7.2\pm 2.8\text{‰}$) compared with M3 ($0.9\pm 1.3\text{‰}$; $p <$
 259 0.01 & $p < 0.001$). The C/N ratio values in the POM was 6.4 ± 1.5 on average and similar
 260 between sites.

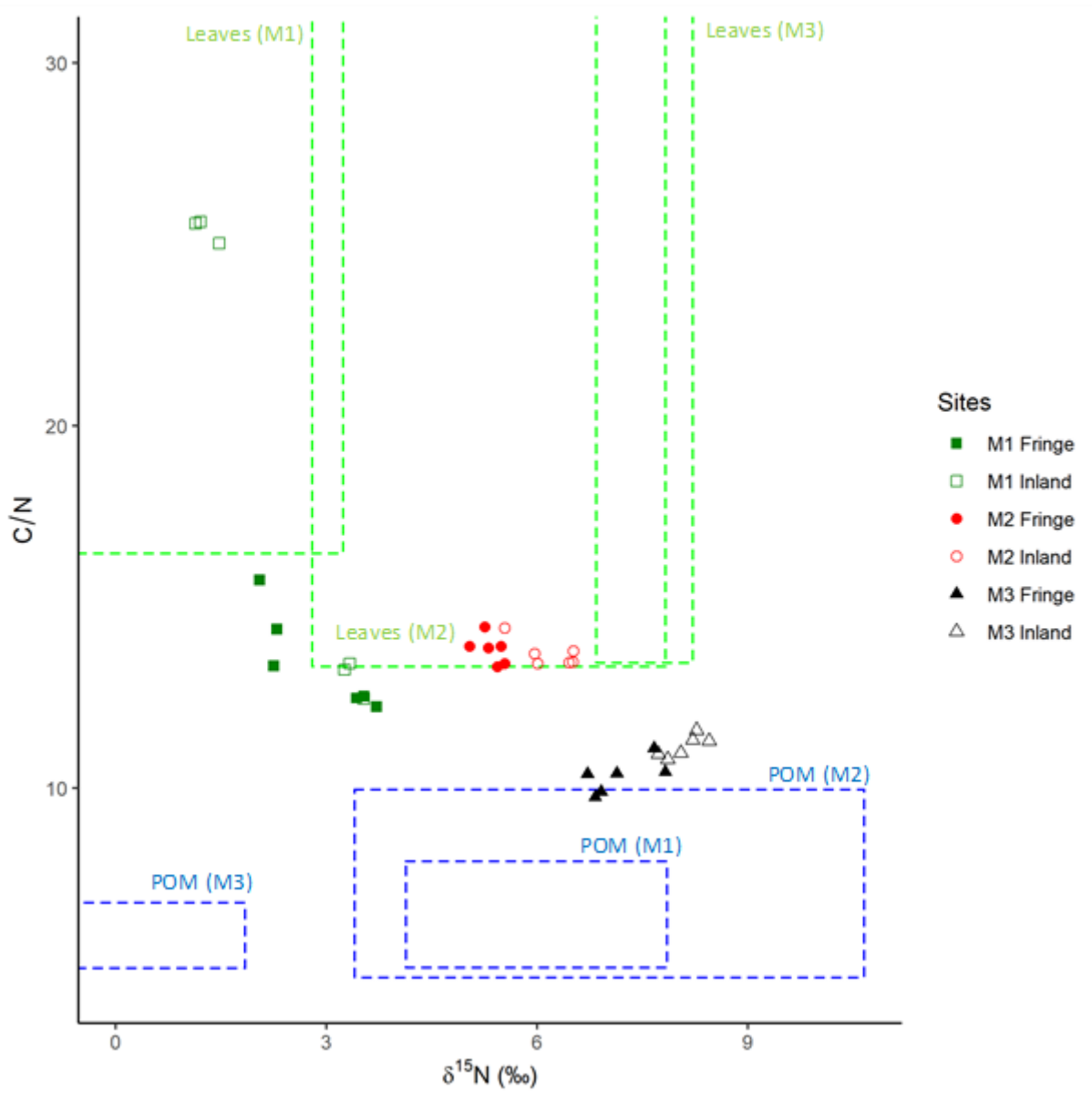


261
 262 *Figure 3: Plot of (a) $\delta^{13}\text{C}$ vs $\delta^{15}\text{N}$ and (b) $\delta^{15}\text{N}$ vs C/N values of particulate organic matter in the*
 263 *three mangroves sites: M1 in Ilha Grande Bay, M2 in Guanabara Bay and M3 in Sepetiba Bay. (2*
 264 *column)*

265 3.1.3. Sediments

266 There was no difference ($p > 0.05$) at M1 in $\delta^{13}\text{C}$, $\delta^{15}\text{N}$, and C/N values between F and
 267 I zone sediments (Fig. 4&5). At M2 I was significantly enriched in $\delta^{15}\text{N}$ compared to
 268 F (I: $6.2\pm 0.4\text{‰}$; F: $5.3\pm 0.2\text{‰}$; $p < 0.01$), while $\delta^{13}\text{C}$ and C/N were similar ($p < 0.05$)
 269 between the two zones (Fig. 4&5). At M3 the I zone was depleted in $\delta^{13}\text{C}$ ($-25.1\pm 0.1\text{‰}$;
 270 $-23.8\pm 0.4\text{‰}$; $p < 0.01$), enriched in $\delta^{15}\text{N}$ ($8.1\pm 0.3\text{‰}$; $7.2\pm 0.5\text{‰}$; $p < 0.01$) and had a
 271 higher C/N ratio (11.2 ± 0.3 ; 10.3 ± 0.5 ; $p < 0.01$) compared to the F sediments. No

272 differences in carbon or nitrogen isotopic signature or C/N ratio were observed
273 between surface and subsurface (3 and 9 cm) sediments at all sites.



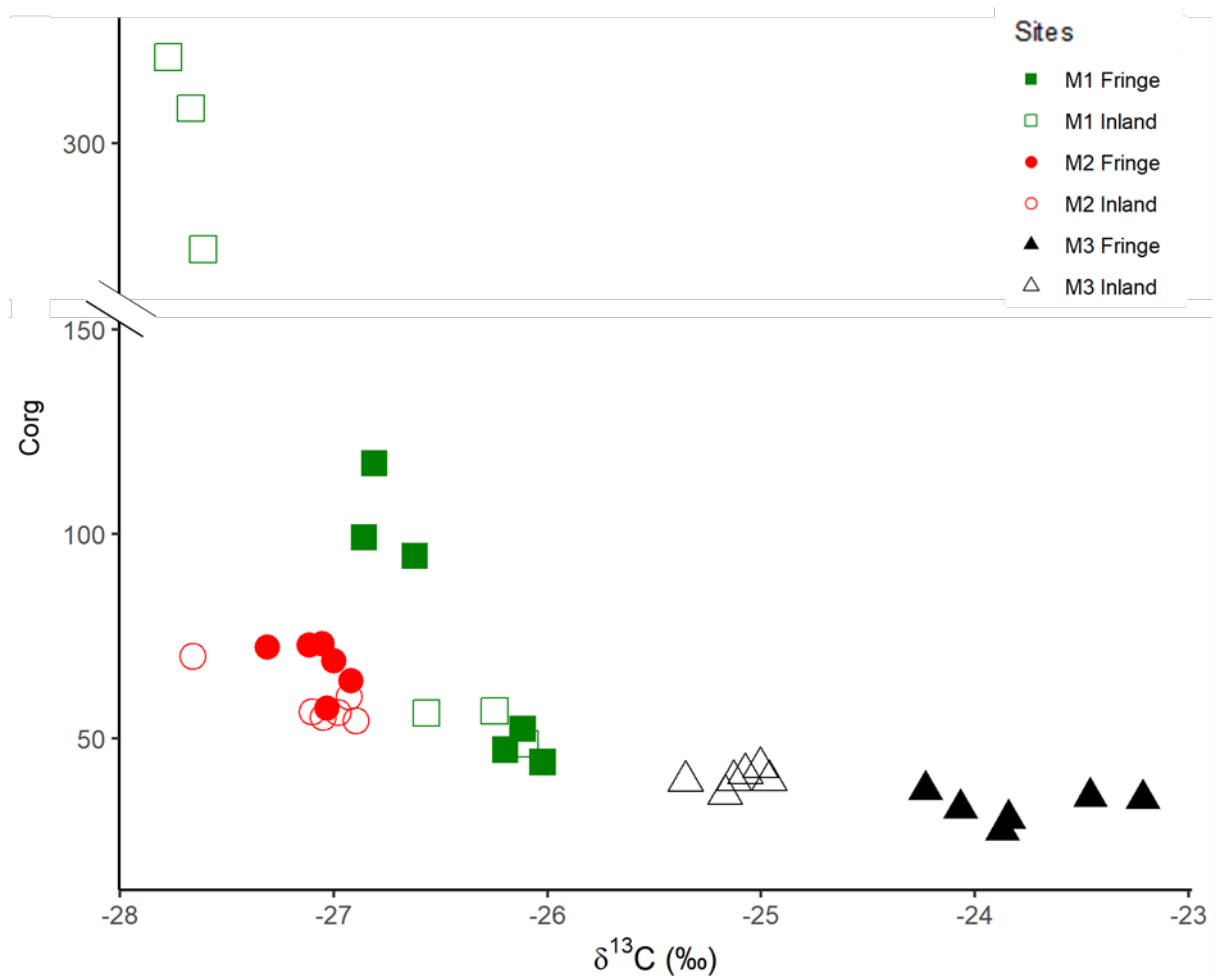
274

275 *Figure 4: Plot of $\delta^{15}\text{N}$ vs C/N in sediments mangroves of the three mangroves sites -: M1 in Ilha*
276 *Grande Bay, M2 in Guanabara Bay and M3 in Sepetiba Bay. The envelopes of leaves are*
277 *represented by green boxes and those of particulate organic matter in water are represented by*
278 *blue boxes. (1 column)*

279

280 Considering surface and subsurface sediments together as a pool, the $\delta^{13}\text{C}$ was
281 significantly depleted at M2 ($-27.1\text{‰} \pm 0.2\text{‰}$) than at the other two sites (M1 : $-26.8 \pm$

282 0.6‰, $p < 0.05$; M3 : $-24.4‰ \pm 0.7‰$, $p < 0.001$). Similarly, $\delta^{13}\text{C}$ was depleted at M1 (-
283 $26.8‰ \pm 0.6‰$) compared to M3 ($-24.4‰ \pm 0.7‰$; $p < 0.001$). M1 has the lowest $\delta^{15}\text{N}$,
284 followed by M2 and then M3 ($2.6 \pm 1.0‰ > 5.7 \pm 0.5‰ > 7.6 \pm 0.6‰$, $p < 0.001$). The C/N
285 ratio showed no significant difference between M1 and M2, but was significantly higher
286 in subsurface sediments of I site of M1 mangrove (Fig. 4; $p < 0.001$). The C/N values
287 of surface sediments at the F and the I zones in M1 were comparable to those at M2.
288 M1 and M2 had higher C/N values (16.3 ± 5.5 and 13.8 ± 0.4 respectively) than M3
289 (10.7 ± 0.6 ; $p < 0.001$). M1 and M2 had higher sedimentary organic carbon (C_{org})
290 concentrations than M3 ($126.6 \pm 108.3 \text{ mg.g}^{-1}$, $54.2 \pm 7.6 \text{ mg.g}^{-1}$ and $36.5 \pm 4.6 \text{ mg.g}^{-1}$; p
291 < 0.001). The high values and standard deviation of OC content in M1 sediments was
292 due to the very high organic carbon concentrations in the subsurface sediment of zone
293 I (Fig. 5).



294

295 *Figure 5: Plot of $\delta^{13}\text{C}$ vs. organic carbon in sediments mangroves of the three mangroves sites :*
 296 *M1 in Ilha Grande Bay, M2 in Guanabara Bay and M3 in Sepetiba Bay. (1 column)*

297

298 3.2. Fatty acids composition

299 3.2.1. Mangrove leaves

300 Long chain fatty acids (≥ 24 carbon) reached an average of 6.6% (Table 1). Among
 301 PUFAs, 18:3 ω 3 was the dominant fatty acid followed by 18:2 ω 6. The monounsaturated
 302 fatty acids (MUFAs) signature of mangrove leaves was almost entirely caused by the
 303 presence of 18:1 ω 9.

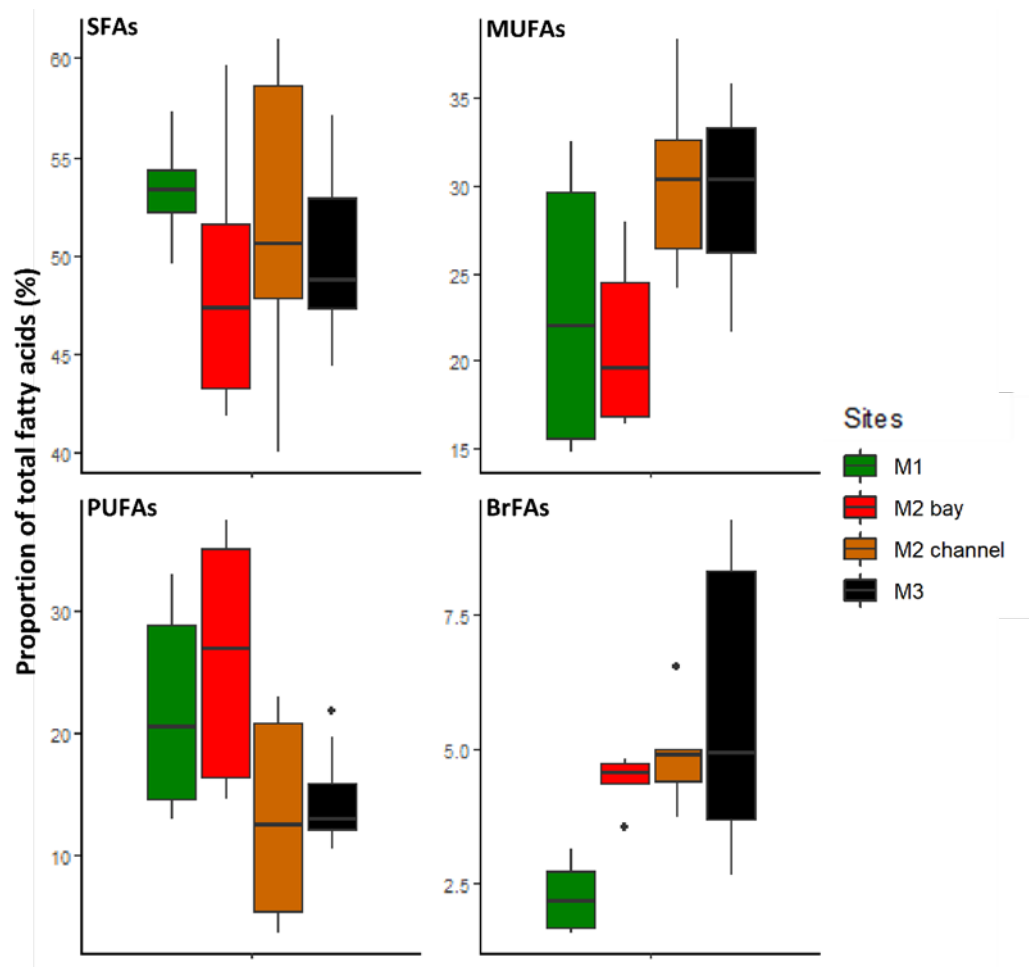
304 *Table 1 : Mean (\pm sd) of selected fatty acid composition of leaves in three sampling locations (M1*
 305 *in Ilha Grande Bay, M2 in Guanabara Bay and M3 in Sepetiba Bay). BD: Below Detection (1*
 306 *column)*

	Site M1		Site M2		Site M3	
	Fringe (n = 4)	Inland (n = 4)	Fringe (n = 4)	Inland (n = 4)	Fringe (n = 4)	Inland (n = 4)
16 :0	23.3 \pm 2.5	31.2 \pm 2.6	24.6 \pm 1.5	22.9 \pm 3.6	25.2 \pm 6.8	31.9 \pm 5.1
ΣLCFAs	2.6 \pm 2.8	4.5 \pm 2.9	3.1 \pm 2.6	2.8 \pm 1.5	0.4 \pm 0.3	9.0 \pm 11.3
Other SFAs	3.2 \pm 0.8	1.7 \pm 0.6	6.3 \pm 1.5	7.4 \pm 2.3	4.4 \pm 1.2	2.2 \pm 1.0
ΣSFAs	29.3 \pm 5.1	41.3 \pm 4.5	30.0 \pm 1.7	28.8 \pm 3.1	29.6 \pm 7.6	44.7 \pm 7.4
18:1ω9	13.3 \pm 1.1	9.2 \pm 0.4	11.3 \pm 1.3	12.7 \pm 2.0	13.8 \pm 2.4	12.9 \pm 2.5
Other MUFAs	6.0 \pm 2.6	10.1 \pm 3.5	5.4 \pm 2.3	5.9 \pm 1.5	4.4 \pm 0.9	12.8 \pm 8.9
ΣMUFAs	16.5 \pm 1.1	10.9 \pm 0.3	17.6 \pm 1.3	20.2 \pm 3.5	18.3 \pm 3.0	15.1 \pm 2.0
18:2ω6	11.2 \pm 1.9	12.0 \pm 4.0	13.3 \pm 2.0	13.6 \pm 3.3	13.5 \pm 2.9	15.0 \pm 2.8
18:3ω3	31.7 \pm 12.9	34.3 \pm 7.6	37.0 \pm 1.3	35.7 \pm 3.6	30.0 \pm 8.4	23.7 \pm 6.4
ΣPUFAs	43.0 \pm 12.2	46.9 \pm 4.6	50.4 \pm 1.7	49.3 \pm 2.8	43.5 \pm 6.8	38.9 \pm 9.1
ΣBrFAs	11.3 \pm 13.5	1.5 \pm 0.2	2.0 \pm 0.3	1.6 \pm 0.6	8.5 \pm 11.1	1.5 \pm 0.4

307

308 3.2.2. Particulate organic matter (POM)

309 The bay of M2 POM had significantly lower MUFA proportion than channel of M2 and
 310 M3 (Table 2; $p < 0.05$). At all sites, 16:1 ω 7 was the dominant fatty acid among MUFAs.
 311 PUFAs at M3 and M2 (bay) was significantly lower than at M1 and M2 (channel) ($p <$
 312 0.05 ; Table 2; Fig. 6). 20:5 ω 3 and 22:6 ω 3 were the dominant PUFAs at all 4 sites
 313 (Table 2). However, we observed a predominance of 20:5 ω 3 over 22:6 ω 3 at M3 ($p <$
 314 0.001). BrFAs contributed twofold more to the POM fatty acid pool at M3 and M2 than
 315 at M1 (Table 2; $p < 0.001$).



316

317 *Figure 6 : Contribution (% of total fatty acids) of SFAs, MUFAs, PUFAs and BrFAs to the total fatty*
 318 *acid pool in four particulate organic matter sampling locations (Ilha Grande Bay (M1), Guanabara*
 319 *Bay (M2 bay) and channel (M2 channel) and Sepetiba Bay (M3)). Median values (horizontal solid*
 320 *line inside the box), 25th and 75th percentiles (lower and upper ends of the boxes), 95% confidence*
 321 *intervals (whiskers) and outliers (circles) are shown. (1 column)*

322 Table 2: Mean (\pm sd) proportion (% of total fatty acids) of selected fatty acid composition of
 323 particulate organic matter in four sampling locations (M1 in Ilha Grande Bay, M2 in Guanabara Bay
 324 and M3 in Sepetiba Bay). BD: Below Detection (**1 column**)

	Site M1	Site M2 Channel	Site M2 Bay	Site M3
14:0	11.6 \pm 1.5	9.8 \pm 2.3	5.6 \pm 0.7	18.3 \pm 6.9
16:0	29.2 \pm 1.9	29.0 \pm 3.3	27.2 \pm 3.2	22.9 \pm 2.1
18:0	8.1 \pm 1.3	6.7 \pm 3.1	14.4 \pm 8.6	4.8 \pm 3.9
Σ Other SFAs	4.2 \pm 0.2	3.0 \pm 0.5	4.6 \pm 1.0	3.8 \pm 1.0
Σ SFAs	53.0 \pm 2.5	48.5 \pm 6.9	51.8 \pm 8.1	49.9 \pm 4.2
16:1ω7	7.2 \pm 1.0	12.5 \pm 2.8	7.5 \pm 2.8	17.9 \pm 2.8
18:1ω9	10.4 \pm 7.9	3.8 \pm 1.9	13.4 \pm 6.4	5.7 \pm 4.0
Σ Other MUFAs	4.2 \pm 1.5	4.6 \pm 0.6	9.4 \pm 3.6	6.9 \pm 3.3
Σ MUFAs	21.9 \pm 8.1	20.9 \pm 5.0	30.3 \pm 5.2	29.7 \pm 4.8
20:5ω3	6.3 \pm 3.6	9.1 \pm 5.2	3.3 \pm 2.7	6.2 \pm 3.2
22:6ω3	7.5 \pm 4.9	7.9 \pm 6.1	3.7 \pm 3.0	1.5 \pm 1.0
Σ Other PUFAs	9.1 \pm 0.7	9.1 \pm 1.2	6.1 \pm 3.1	6.9 \pm 1.8
Σ PUFAs	23.0 \pm 8.1	26.1 \pm 10.7	13.1 \pm 8.7	14.5 \pm 4.1
Σ BrFAs	2.1 \pm 0.6	4.4 \pm 0.5	4.9 \pm 0.9	5.8 \pm 2.7

325

326 3.2.3. Sediments

327 3.2.3.1. Variations with depth at each site

328 At M1, the percentages of LCFAs that were higher at 3-9 cm sediments of I zone than
 329 in F zone ($p < 0.01$). The fatty acid 16:1 ω 7 was significantly less abundant at the P3
 330 and P9 depths at I zone of M1 than in the F zone and surface sediments at I zone (p
 331 < 0.001). The BrFAs group showed the same trend with a lower percentage for P3 and
 332 P9 depth at the I zone of the M1 compared to the other samples ($p < 0.05$). PUFAs
 333 were the least abundant group at M1 and showed no significant difference between
 334 the different depths of the F and I zones ($p > 0.05$).

335 The FA composition of the mangrove I and F zones of M2 were not significantly
 336 different (ANOSIM, $R = 0.186$, $p > 0.05$). Thus, for M2 data from I and F zones were
 337 pooled and only tested according to depth. There was a significant difference in term
 338 of FA composition between the three depths ($R=0.62$; $p=0.001$; Table 3). LCFAs

339 contributed twice ($p < 0.05$) as much to the fatty acid pool at P9 depth as they did at P0
340 and P3 depths. The fatty acid 16:1 ω 7 was more abundant at the surface than below.
341 Conversely, 18:1 ω 9 was lower in surface sediments than at subsurface. The group of
342 BrFAs was similar between the three sampled depths. The group of PUFAs tend to
343 decreased from P0, to P9 (no significant difference).

344 At M3, the FA 16:1 ω 7 was more abundant in surface than in subsurface sediments at
345 both zones F and I ($p < 0.05$; Table 3). FA 18:1 ω 9 showed the opposite trend ($p <$
346 0.05). BrFAs were more abundant in I sediment profiles than in F ($p < 0.05$). BrFAs
347 were also more abundant in subsurface sediments (P3 and P9 depths) than in surface
348 sediments ($p < 0.05$). At M3, PUFAs did not differ significantly between I and F but
349 were more abundant at the surface than at depth in the I ($p < 0.05$) and more abundant
350 at P0 and P3 than at P9 in the F ($p < 0.05$).

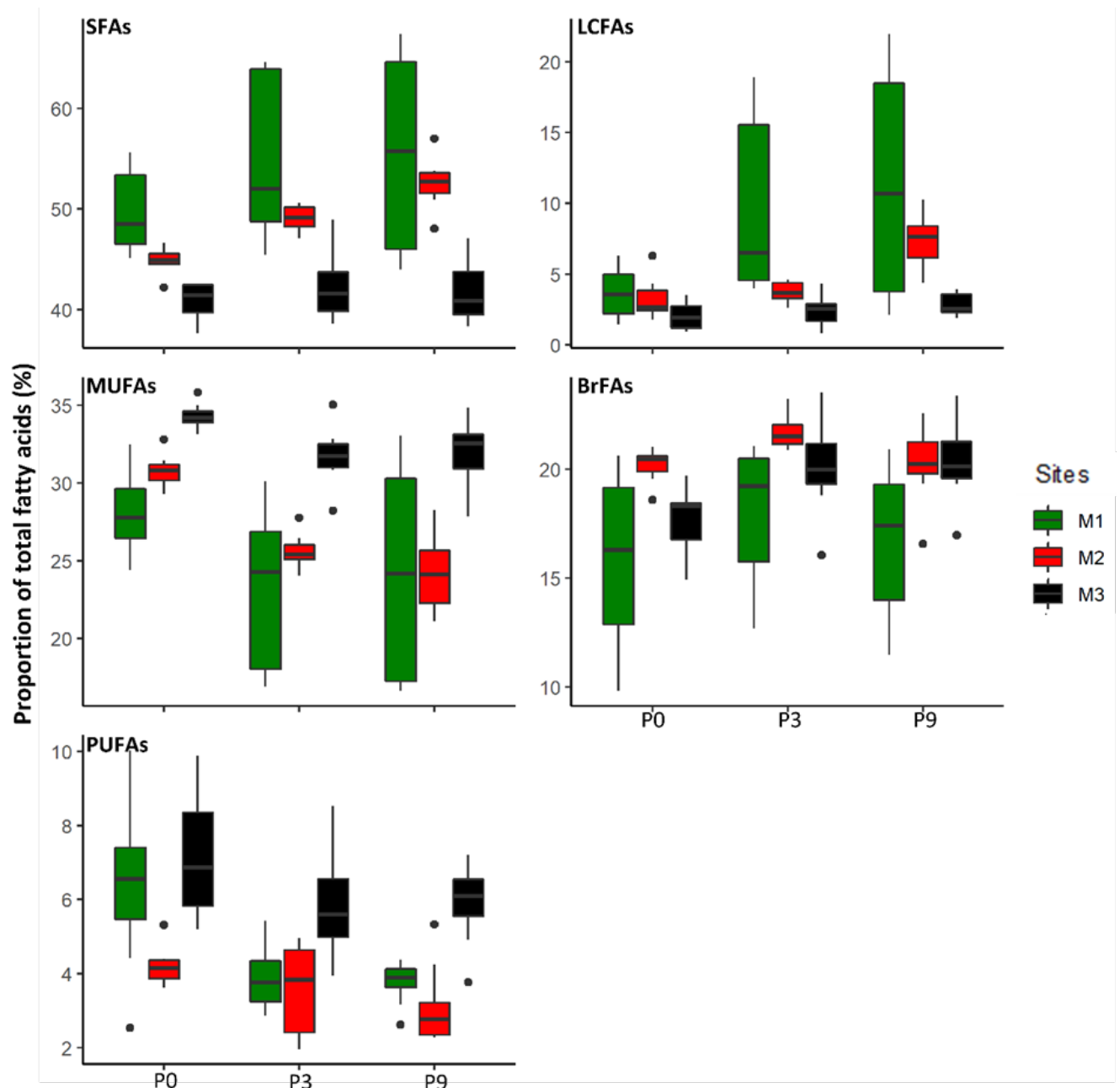
351 Table 3: Mean (\pm sd) proportion (% of total fatty acids) of selected fatty acid composition of sediments of inland and fringe mangrove at 0 cm, 3 cm and
 352 9cm in three sampling locations (M1 in Ilha Grande Bay, M2 in Guanabara Bay and M3 in Sepetiba Bay). BD: Below Detection (**2 column**)

Depth	Site M1						Site M2						Site M3					
	Fringe			Inland			Fringe			Inland			Fringe			Inland		
	0 cm	3 cm	9 cm	0 cm	3 cm	9 cm	0 cm	3 cm	9 cm	0 cm	3 cm	9 cm	0 cm	3 cm	9 cm	0 cm	3 cm	9 cm
16:0	20.9 \pm 1.8	22.1 \pm 0.2	20.1 \pm 0.6	20.7 \pm 1.4	23.9 \pm 1.8	23.1 \pm 3.0	20.4 \pm 0.7	22.6 \pm 1.5	21.2 \pm 0.4	22.4 \pm 0.9	22.4 \pm 0.6	22.4 \pm 1.1	22.0 \pm 0.7	21.1 \pm 1.5	22.1 \pm 1.7	21.1 \pm 0.8	21.3 \pm 0.4	20.8 \pm 0.6
ΣLCFAs	5.1 \pm 0.9	4.5 \pm 0.4	3.9 \pm 1.8	3.4 \pm 1.5	15.0 \pm 5.1	18.9 \pm 3.1	4.4 \pm 1.9	3.7 \pm 0.6	8.8 \pm 1.3	2.5 \pm 0.7	3.8 \pm 1.1	5.9 \pm 1.5	2.0 \pm 1.2	2.9 \pm 1.1	3.1 \pm 0.6	2.1 \pm 1.0	1.8 \pm 1.1	2.6 \pm 0.9
ΣSFAs	47.3 \pm 2.3	48.0 \pm 1.8	45.8 \pm 1.8	47.3 \pm 1.2	61.8 \pm 4.9	65.1 \pm 1.6	44.6 \pm 2.2	48.9 \pm 1.1	54.4 \pm 1.7	45.0 \pm 0.6	49.3 \pm 1.9	50.7 \pm 1.9	42.5 \pm 0.1	44.5 \pm 3.2	44.5 \pm 2.3	39.1 \pm 1.4	39.4 \pm 0.7	39.2 \pm 0.6
16:1ω7	9.3 \pm 0.7	7.3 \pm 1.6	8.4 \pm 0.7	6.0 \pm 1.2	2.9 \pm 0.8	2.1 \pm 0.6	9.0 \pm 1.9	5.2 \pm 1.2	4.0 \pm 0.9	8.4 \pm 0.4	6.6 \pm 1.1	5.7 \pm 1.4	9.4 \pm 1.9	4.3 \pm 1.5	3.2 \pm 0.6	6.8 \pm 2.4	4.3 \pm 0.8	4.3 \pm 2.6
16:1ω5	1.3 \pm 0.3	1.2 \pm 0.3	1.6 \pm 0.1	1.2 \pm 0.2	0.7 \pm 0.3	0.6 \pm 0.2	2.0 \pm 0.1	1.5 \pm 0.3	1.2 \pm 0.3	2.0 \pm 0.2	1.6 \pm 0.3	1.6 \pm 0.2	2.2 \pm 0.3	2.4 \pm 0.3	2.6 \pm 0.9	3.8 \pm 0.7	4.1 \pm 1.0	3.2 \pm 0.2
18:1ω9	5.2 \pm 0.3	6.7 \pm 1.4	6.7 \pm 0.7	8.2 \pm 2.4	6.5 \pm 1.0	6.7 \pm 1.9	5.9 \pm 0.2	8.9 \pm 1.4	7.9 \pm 1.1	7.3 \pm 0.3	7.5 \pm 0.3	8.3 \pm 1.1	7.4 \pm 0.3	10.3 \pm 2.5	11.1 \pm 0.7	8.2 \pm 1.4	9.6 \pm 0.6	10.9 \pm 0.8
18:1ω7	8.2 \pm 1.6	7.9 \pm 1.3	8.1 \pm 0.2	7.1 \pm 0.5	3.5 \pm 0.4	2.7 \pm 0.7	8.3 \pm 0.4	4.7 \pm 0.7	3.9 \pm 0.9	7.1 \pm 0.3	5.6 \pm 0.5	4.8 \pm 1.1	7.4 \pm 0.9	4.8 \pm 1.3	4.4 \pm 0.3	6.6 \pm 0.3	5.8 \pm 0.7	5.1 \pm 0.4
19:1ω9	1.5 \pm 0.2	1.8 \pm 0.3	1.0 \pm 0.1	1.0 \pm 0.2	2.7 \pm 0.4	2.5 \pm 0.4	1.8 \pm 0.1	2.5 \pm 0.3	2.6 \pm 0.5	1.7 \pm 0.2	2.1 \pm 0.1	2.3 \pm 0.2	2.2 \pm 0.8	4.3 \pm 1.1	5.4 \pm 0.6	4.8 \pm 0.9	6.3 \pm 0.5	6.7 \pm 0.6
ΣOther MUFAs	4.3 \pm 0.8	2.8 \pm 0.4	5.1 \pm 0.9	4.7 \pm 1.0	2.5 \pm 0.9	2.7 \pm 0.4	3.8 \pm 0.3	2.3 \pm 0.4	2.5 \pm 0.3	4.3 \pm 0.4	2.8 \pm 1.1	3.4 \pm 0.5	5.5 \pm 0.8	4.4 \pm 1.0	3.8 \pm 0.6	4.4 \pm 0.4	3.2 \pm 0.3	3.1 \pm 0.4
ΣMUFAs	29.8 \pm 2.9	27.7 \pm 2.1	31.0 \pm 1.5	28.2 \pm 1.8	18.9 \pm 2.8	17.3 \pm 0.8	30.8 \pm 1.8	25.2 \pm 1.0	22.2 \pm 1.0	30.8 \pm 0.5	26.2 \pm 1.3	26.1 \pm 1.5	34.1 \pm 0.8	30.5 \pm 1.5	30.6 \pm 2.2	34.5 \pm 1.0	33.3 \pm 1.5	33.3 \pm 1.1
18:2ω6	1.3 \pm 0.2	2.0 \pm 1.1	1.0 \pm 0.2	1.7 \pm 0.7	3.0 \pm 0.9	2.9 \pm 0.8	1.2 \pm 0.1	3.1 \pm 1.0	2.3 \pm 1.3	1.3 \pm 0.1	1.1 \pm 0.2	1.7 \pm 0.8	1.6 \pm 0.4	2.9 \pm 0.9	3.0 \pm 1.0	2.4 \pm 1.4	2.4 \pm 0.6	3.4 \pm 0.6
20:4ω6	1.4 \pm 0.3	0.6 \pm 0.2	0.8 \pm 0.2	0.9 \pm 0.5	0.4 \pm 0.2	BD	0.7 \pm 0.1	0.5 \pm 0.4	BD	0.7 \pm 0.1	0.4 \pm 0.1	0.3 \pm 0.1	1.7 \pm 1.1	1.5 \pm 1.5	1.1 \pm 0.3	1.8 \pm 0.5	1.4 \pm 0.4	1.0 \pm 0.2
20:5ω3	1.6 \pm 0.7	0.3 \pm 0.1	0.7 \pm 0.1	0.7 \pm 0.3	BD	BD	0.7 \pm 0.1	BD	BD	0.9 \pm 0.2	BD	BD	1.5 \pm 0.4	0.6 \pm 0.6	BD	1.2 \pm 0.6	BD	BD
ΣOther PUFAs	2.0 \pm 0.5	0.8 \pm 0.1	1.2 \pm 0.3	1.5 \pm 0.4	0.6 \pm 0.4	0.5 \pm 0.4	1.4 \pm 0.3	1.0 \pm 0.2	0.8 \pm 0.2	1.6 \pm 0.3	0.6 \pm 0.2	1.0 \pm 0.2	1.9 \pm 0.5	1.4 \pm 0.4	1.4 \pm 0.3	2.2 \pm 0.3	1.2 \pm 0.6	1.7 \pm 0.3
ΣPUFAs	6.3 \pm 1.3	3.8 \pm 0.8	3.7 \pm 0.3	4.8 \pm 1.8	4.0 \pm 1.1	3.8 \pm 0.8	4.0 \pm 0.4	4.5 \pm 0.5	3.3 \pm 1.4	4.4 \pm 0.6	2.3 \pm 0.3	2.9 \pm 0.9	6.8 \pm 1.9	6.3 \pm 1.7	5.7 \pm 1.4	7.6 \pm 1.7	5.3 \pm 1.2	6.2 \pm 1.0
ΣBrFAs	16.5 \pm 1.8	20.6 \pm 0.4	19.5 \pm 1.1	19.7 \pm 0.7	15.4 \pm 2.4	13.7 \pm 2.3	20.7 \pm 0.3	21.3 \pm 0.6	20.1 \pm 2.6	19.7 \pm 0.8	22.2 \pm 0.9	20.3 \pm 0.8	16.7 \pm 1.4	18.7 \pm 1.8	19.3 \pm 1.7	18.7 \pm 0.7	22.0 \pm 1.4	21.3 \pm 1.6

353

354 3.2.3.2. Inter-sites comparison

355 At surface sediments, there was a significant difference in FA composition between
356 the three mangrove sites (ANOSIM, $R = 0.475$; $p < 0.001$). LCFAs showed a higher
357 contribution at M1 compared to M2 ($p < 0.01$) and at M2 compared to M3 ($p < 0.01$;
358 Fig. 7). Among MUFAs, 16:1 ω 7 and 18:1 ω 9 had the largest contributions but did not
359 show significant differences in surface sediments of the three sites ($p > 0.05$; Table 3).
360 BrFAs were significantly more abundant in surface sediment from M2 than from M1
361 and M3 ($p < 0.01$; Fig. 7). PUFAs contribute more to the TFA pool in the surface
362 sediments of M1 and M3 than in M2 ($p < 0.01$, Fig. 7). In the surface sediments of M3
363 there was a higher contribution of 20:5 ω 3 compared to M2 ($p < 0.01$; Table 3).



364
 365 *Figure 7: Contribution (% of total fatty acids) of SFAs, LCFAs, MUFAs, BrFAs and PUFAs to the*
 366 *total fatty acid pool in the sediments of the three mangroves (M1 in Ilha Grande Bay, M2 in*
 367 *Guanabara Bay and M3 in Sepetiba Bay.). Median values (horizontal solid line inside the box), 25th*
 368 *and 75th percentiles (lower and upper ends of the boxes), 95% confidence intervals (whiskers) and*
 369 *outliers (circles) are shown. (1 column)*

370 At three centimetres depth (P3), FA composition of the sediment were different
 371 between the three sites (ANOSIM, $R = 0.476$; $p < 0.001$). The LCFAs were higher in M1
 372 than in M2 ($p < 0.01$) and in M2 than in M3 ($p < 0.05$; Fig. 7). The most contributing

373 MUFAs were 16:1 ω 7, 18:1 ω 9, and 18:1 ω 7 (Table 3). BrFAs showed significant
374 difference only between M1 and M2 ($p < 0.01$; Fig. 7). PUFAs were less abundant in
375 M1 and M2 than in M3 ($p < 0.01$ and $p < 0.05$ respectively; Fig. 7). Strong standard
376 deviations were observed at M1 (both zones) for P3 sediments in contrast to the other
377 two sites. These large standard deviations were caused by the strong difference in
378 profiles between the I and the F zones of M1 (Fig. 7).

379 In sediments at P9 depth, the FA profiles were significantly different between the three
380 sites (ANOSIM, $R = 0.639$, $p < 0.001$). LFCAs were particularly abundant in P9
381 sediments at M1 and M2 but not at M3. MUFAs were less abundant in the sediment
382 FA profiles of M1 and M2 compared to M3 ($p < 0.05$ and < 0.001 respectively; Fig. 7).
383 The 16:1 ω 5 ($p < 0.001$) was more abundant at M3 than at M1 and M2. BrFAs were
384 lower at M1 compared to M2 and M3 (Fig. 7). PUFAs were less abundant in the P9
385 sediments of M1 and M2 than in those of M3 ($p < 0.01$; Fig. 7). Among PUFAs 18:2 ω 6
386 was more abundant in the P9 sediments of M3 than those of M1 and M2 ($p < 0.05$).

387 **4. Discussion**

388 **4.1. Eutrophication gradient in the coast of Rio de Janeiro**

389 Stable isotopes and fatty acids analysed in POM, SOM and mangrove leaves clearly
390 indicate the existence of gradient of eutrophication in the three investigated mangroves
391 in the state of Rio de Janeiro (Brazil). This gradient from M1 to M3 sites is consistent
392 with urban occupation in the region (Table A1), as also analysed in Barroso et al. (in
393 revision).

394 The high proportions of BrFA found at M3 highlight a strong bacterial (Table A2)
395 contribution in the POM. Previous studies have shown that in the presence of
396 wastewater, heterotrophic bacteria generate a strong recycling of nitrogen which leads
397 to the excretion of ammonium with a depleted $\delta^{15}\text{N}$ signature in the water column that
398 can be further incorporated in the growth of marine phytoplankton (Ke et al., 2019).
399 The high contribution of microalgal material in M3 mangroves POM is demonstrated
400 by a high ratio of 16:1 ω 7/16:0 (Mortillaro et al., 2012; Napolitano et al., 1997) together
401 with an enriched $\delta^{13}\text{C}$ signature (Fig. 3). At site M3, these high microalgal and bacterial
402 contributions in POM, lead to a $\delta^{15}\text{N}$ signature near 0, therefore disconnected from
403 SOM which is characterized by enriched $\delta^{15}\text{N}$ signature of about 8 (sewage
404 characteristic) (Fig. 3 & 4). Isotopic fractionation during nitrogen uptake (Liu et al.,
405 2013) and the preferential use of ^{14}N -rich ammonium (generated by bacterial recycling)
406 as a nitrogen source by algae (Liu et al., 2013) could explain the $\delta^{15}\text{N}$ -depleted isotopic
407 signatures of the POM at M3. Our results reveal a greater influence of untreated
408 sewage discharge to the M3 mangrove sediments compared to M1 and M2 consistent
409 with Barroso et al. (in revision) based on the ratio of stanol biomarkers
410 (coprostanol/(coprostanol+colestanol)). Urban sewage loads, enriched with nutrients,

411 received by M3 mangroves could also trigger the growth of marine microalgae (Unger
412 et al., 2013) and generate important changes in the algal community (Smith, 2003). In
413 the POM of the water column of M3, the relative abundance of 20:5 ω 3 was 3 times
414 higher than that of 22:6 ω 3 (Table 3 and A2). This suggests that diatoms dominate
415 dinoflagellates within the planktonic biomass unlike in the waters of the M1 and M2
416 sites where the proportions of these two PUFAs were comparable (Antonio & Richoux,
417 2016; Bergamino et al., 2014). In general, it is accepted that eutrophication favours
418 dinoflagellates over diatoms (Dale, 2001). However, in eutrophic waters surrounding
419 mangroves, a dominance of diatoms over dinoflagellates was also observed
420 (Choudhury et al., 2015). In particular, high nitrogen concentrations are known to
421 promote planktonic diatom growth in mangroves waters (Hilaluddin et al., 2020). In the
422 waters of the Bay of Sepetiba, Rodrigues et al. (2009) showed a N/P ratio of 20 during
423 the wet season as well as high concentrations of silica. In general, diatoms are
424 favoured over dinoflagellates in coastal waters where nitrogen and silica are not limiting
425 (Svensen et al., 2007). Thus, the nature of eutrophication-related inputs as well as the
426 recycling of nitrogen by heterotrophs bacteria could explain the difference in the
427 phytoplankton derived fatty acids in the M3 mangrove compared to the other two sites.

428 In addition, the $\delta^{15}\text{N}$ signatures of sediments and leaves at the studied sites are
429 showing a clear gradient of eutrophication (Fig. 2, Fig. 4). Because mangrove trees
430 incorporate anthropogenic nitrogen present in the sediment porewaters, their $\delta^{15}\text{N}$
431 signatures can be used to identify a gradient of eutrophication locally or at a larger
432 scale (Dugdale & Wilkerson, 1986; Gritcan et al., 2016). Sedimentary $\delta^{15}\text{N}$ signatures
433 are also sensitive to anthropogenic nitrogen inputs (Pérez et al., 2020). Accordingly,
434 the mangrove surface sediments at M3 are the most eutrophic, followed by those of
435 mangrove sediments at M2 and with M1 showing the most pristine conditions (Fig. 2

436 and Fig. 4). Barroso et al. (revised) also found at the same three study sites an increase
437 in sedimentary inorganic phosphorus content from site M1 to M3. Several studies have
438 described eutrophication in waters of Guanabara and Sepetiba Bays, where high
439 concentrations of chlorophyll, inorganic nitrogen and phosphorus are found (Fistarol et
440 al., 2015; Rodrigues et al., 2009). The eutrophication induced by domestic wastewater
441 leads to an accumulation of exchangeable inorganic phosphorus in mangrove
442 sediments (Barcellos et al., 2019). Although Guanabara Bay is more populated (Table
443 A1) and eutrophic than Sepetiba Bay, because of local settings of the sampled
444 mangroves, the M3 mangrove sampled in Sepetiba appeared more impacted by inputs
445 of labile SOM from sewage and phytoplankton than the M2 mangrove in Guanabara
446 Bay.

447 The C/N values in sediments in the inland of M1 mangrove (mean value: 19.3) are in
448 the range of those reported in mangroves sediments under micro- to meso-tidal
449 influence (i.e. Bouillon et al., 2003). These high values indicate a predominant input
450 of mangrove tree material to the sediments. Indeed, sediments of inland M1 are
451 composed of almost pure litter and *R. mangle* fine roots. In contrast, the fringe
452 sediments at M1 and sediments of both fringe and inland zones at M2 mangrove have
453 lower C/N ratios, in the range of those documented for mangroves influenced by tidal
454 deposition and river inputs (Bouillon et al., 2003). The C/N values at M3 are even lower,
455 in the range of values described by Leng & Lewis (2017) for riverine POC and by
456 Jennerjahn and Ittekkot (1997) for sediments from another mangrove of the state of
457 Rio de Janeiro influenced by a river (Paraiba do sul). This could be due to a larger
458 influence of sewage inputs or marine phytoplankton in sediments of M3 than in the
459 mangroves of the two other sites. This is also corroborated by the significantly depleted
460 $\delta^{13}\text{C}$ signatures of superficial sediments in inland mangrove ($-25,12 \pm 0,14$) compared

461 to the coastal fringe mangrove ($-23,78 \pm 0,38$; more influenced by algal contributions)
462 could be caused by a greater influence of urban wastewater at the inland zone, and/or
463 of marine phytoplankton at the fringe zone.

464 **4.2. Organic matter sources and exchange of OM between waters and** 465 **mangroves sediments**

466 At the most pristine mangrove location (M1), concentrations of C_{org} at the inland site
467 were 3 times higher in subsurface sediments (3 and 9 cm depths) than at surface
468 sediments (Fig. 5). At that site, we observed the presence of a dense rhizosphere of
469 *R.mangle* in subsurface, in particular fine roots with very little silt. Root contribution
470 was characterized by more than 3-fold higher proportions of LCFAs and 18:2 ω 6 (FAs
471 characteristics of mangroves: Meziane et al., 2007; Table A2) and enriched $\delta^{13}C$
472 values in subsurface sediments compared to the surface ($-26.3 \pm 0.2\text{‰}$ > $-27.7 \pm 0.1\text{‰}$).
473 The presence of *R.mangle* fine roots explains the higher C_{org} concentrations in these
474 sediments at 3 and 9cm depth than in the surface sediments. Concerning the fringe
475 M1 site, the sediment organic carbon concentration was between 3 and 7 times lower
476 and the proportions of LCFAs and 18:2 ω 6 were 3 to 5-fold lower than at the inland site
477 (both surface and subsurface; Fig. 5). This was apparently due to lower root density of
478 *A. shaueriana* at the fringe compare to *R. mangle* at the inland site, as well as a larger
479 contribution of deposited clays and silts at the fringe. This corroborates the study of
480 Xiong et al. (2018) which shows a lower contribution of mangrove vegetation in
481 mangroves near the sea than in those located inland. The sediments of fringe M1
482 mangrove have $\delta^{13}C$ -enriched signature compared to the inland subsurface sediments
483 which reveals a greater influence of marine phytoplankton in the OM. The POM at M1
484 was characterized by a high relative contribution of 18:1 ω 9, abundant in mangrove leaf,

485 and $\delta^{13}\text{C}$ values close to those of mangrove leaves. This suggests an effective tidal
486 export of mangrove-derived POM at site M1.

487 Concentrations of C_{org} in subsurface sediments at the inland site of M1 mangrove were
488 4 to 6 times higher than M2 sediments, and between 7 and 10 times higher than in M3
489 sediments. These lower C_{org} concentrations in the two more eutrophic mangroves
490 when compared to the more pristine one can certainly be caused by the watersheds
491 that convey more mineral matter *via* a higher river flow, which leads to a lower OM
492 contribution to the sediments. In addition, the organic matter in a pristine mangrove in
493 Ilha Grande (about 40 km of our M1 mangrove) was composed almost exclusively of
494 mangrove material, with insignificant inputs of marine-derived organic matter (Sanders
495 et al. 2008) similarly to our M1 site. This is consistent with the high contribution of
496 LCFAs and 18:2 ω 6, the $\delta^{13}\text{C}$ values in our samples, as well as with lignin and sterol
497 biomarkers which revealed that mangrove's subsurface sediment at M1 and M2 are
498 richer in plant material than at M3 (Barroso et al., revised). In the most eutrophic M3
499 mangroves, the additional OM supply was due to a contribution of phytoplankton,
500 benthic algae, and urban sources (Pérez et al., 2020; Silva-Filho et al., 2011). Finally,
501 river inputs rich in mineral sediments has a strong influence on C_{org} concentration by
502 diluting it.

503 At high tide, as for other estuaries and bays (Bouillon et al., 2008b), the water columns
504 of the three studied bays contain some POM resuspended from the mangrove forest.
505 The presence of 18:2 ω 6 and 18:3 ω 3 FAs (David et al. 2019) in the POM samples in
506 the creeks and channels at all sites indicates an export of mangrove organic carbon to
507 waters (Table 2 and A2). At M2, the higher contribution of OM produced in the
508 mangrove forest and/or of terrestrial origin is also reflected by the signature of $\delta^{13}\text{C}$ of
509 the water column POM which is more depleted than the two other sites (M1 and M2).

510 At M1 (fringe) and M2, the $\delta^{13}\text{C}$ values (respectively 26.7‰ and 27.1‰ on average)
511 are typical of tidal sediments influenced by inputs from mangroves leaves (Kennedy et
512 al., 2004; Pessenda et al., 2012) and therefore emphasize the significant contribution
513 of mangroves carbon to the POM. The difference in $\delta^{13}\text{C}$ signature between mangrove
514 leaves and sediments at M1 (fringe) and M2 (fringe and inland) mangroves could be
515 the result of a mixing with marine OM but also of diagenesis of litter compounds which
516 leads to the enrichment in ^{13}C of the sedimentary OM (Kennedy et al., 2004). At M3
517 site, the low $\delta^{13}\text{C}$ and C/N values in sediment (Fig. 4 & 5) corroborate the significant
518 contribution of marine phytoplankton and/or microphytobenthic OM sources.

519 **4.3. Impact of eutrophication on organic carbon degradation in mangrove** 520 **sediments**

521 At M1, the sedimentary OM appears relatively refractory with a high proportion of
522 LCFAs and a low proportion of algal FAs, which are respectively the most refractory
523 and the most labile fatty acids (Camacho-Ibar et al., 2003). Conversely, in mangroves
524 of the more eutrophic bays (M2 and M3) the relative contributions of algal FAs as well
525 as the enriched ^{13}C signature suggest that SOM is more labile than at M1. Moreover,
526 C/N values of sediments and water at M2 (~13 and 6 respectively) and M3 (~10 and 6
527 respectively) indicate a higher microbial contribution than at M1 in view of their low
528 values in bacteria (C/N ~ 5; Leng & Lewis (2017)) and
529 phytoplankton/microphytobenthos (C/N ~ 6-7; Leng & Lewis (2017)) compared to
530 mangrove leaves (C/N~22; present study). The C/N consistently decreased from site
531 M1 to M2 and M3 and was 1.2 and 1.6 times lower at M2 and M3 respectively than at
532 M1 (Fig. 4) showing that when eutrophication occurs, nitrogen-rich compounds
533 become more abundant in SOM, relative to carbon-rich compounds. Also, along the
534 eutrophication gradient of our three study sites, FAs, ^{13}C signatures and C/N ratios

535 consistently reveal a larger contribution in the SOM of algal and microbial material,
536 which are more degradable, than the part derived from litters and roots in the mangrove
537 ecosystem. Thus, in our study, the C/N decreases with lability which is consistent with
538 the finding of Pérez et al. (2018b) in another mangrove of Guanabara Bay. This is also
539 consistent with Jennerjahn & Ittekkot (1997 and 1999) who show that the C/N ratio is
540 low in fresh planktonic biomass but high in plant material or material that has
541 undergone degradation because N-rich compounds such as amino acids are
542 preferentially degraded. However, the variability of the lability of N-rich compounds
543 showed by Nordhaus et al. (2011 and 2017) on total hydrolysable amino acids (THAA)
544 reveal that low C/N ratio is not necessarily an indicator of higher lability of OM. Indeed,
545 Nordhaus et al. (2017) found that during a litterbag experiments in an Indonesian
546 mangrove, THAA concentration was negatively correlated with the C/N but that the
547 basic THAA (more reactive than other amino acids groups) were preferentially
548 degraded over other amino acids and that their concentration was not correlated with
549 C/N ratio. Nordhaus et al. (2011) also found that crabs of various species feed
550 preferentially on mangrove leaves with high N bioavailability (shown by THAA-N% of
551 N) or high reactivity index (aromatic THAA/non-protein THAA) and that the food
552 preferences of these crabs is not correlated with C/N composition. Various parameters
553 can also influence the lability of SOM in a eutrophic mangrove such as, effluent type
554 (Queiroz et al., 2020), distance from the pollution source (Queiroz et al., 2020),
555 planktonic primary production (Cotovicz et al., 2018; Sanders et al., 2014) or
556 heterotrophic activity (Davis et al., 2003).

557 The M2 mangrove was found to be moderately impacted by eutrophication. However,
558 in both zones of M2 mangrove, SOM exhibits higher bacterial contribution than M1 and
559 M3 as evidenced by the higher proportions of BrFAs (Mfilinge et al., 2003). The

560 proliferation of these bacteria in the SOM at M2 is enhanced by the presence of labile
561 algal OM in sediments, as indicated by the high 16:1 ω 7/16:0 ratio, acting as a substrate
562 for their growth (Bouillon et al., 2004). These FAs data suggest a higher activity of
563 bacteria at M2 mangrove than at M1 mangrove, consistent with higher CO₂ emissions
564 to the atmosphere reported by Barroso et al. (revised). At the same time, M1 and M2
565 sediments have comparable contribution of LCFAs which indicate they have a similar
566 contribution of refractory OM. At M2, our data suggest that bacteria degrade algal labile
567 OM in surface sediments but do not mineralize the more refractory fraction of OM. This
568 corroborates studies that have shown that eutrophication enhances OM sedimentation
569 (Sanders et al. 2014) and at the same time enhances OM remineralization (Barroso et
570 al. revised; Chen et al., 2010; Jessen, 2016; Martin et al., 2020).

571 The M3 was the most eutrophised site, particularly because of the sewage effluent
572 inputs. High bacterial biomass in the POM is evidenced by the high proportion of BrFAs
573 at M3 (Table 2), which may be attributed to sewage organic matter degraded by
574 bacteria and fungi (Blaszczyk & Krzysko-Lupicka, 2013; Kaneda, 1991). Relative
575 contributions of fungal markers (16:1 ω 5 and 18:1 ω 9; Chen et al., 2001; Meziane et al.,
576 2006; Ngosong et al., 2012; Table A2) in the POM were not different at M1, M2 and
577 M3 sites. In contrast, in the surface and subsurface sediments, these fungal FAs were
578 1.2 to 2 times more abundant at site M3 than at sites M1 and M2 (Table 3). The fact
579 that at M3, high contribution of fungal biomarkers was measured in the SOM, but not
580 in the POM suggests that fungal growth occurred in the sediment. Fungi degrade
581 complex and refractory compounds into more labile OM and thereby make OM
582 bioavailable to bacteria (Chung & Suberkropp, 2008; Ferreira et al., 2006). The higher
583 CO₂ fluxes at M3 than at M1 and M2 (Barroso et al., revised) can therefore be caused
584 by the more efficient degradation of OM through the microbial loop stimulated by the

585 higher fungal contribution through the inputs of labile organic matter (Fenchel, 2008).
586 In addition, the increase of fungal biomass at subsurface sediments could lead to a
587 stronger degradation of low-quality OM at depth (LCFAs), contribute to increase the
588 CO₂ effluxes. Finally, even though C_{org} concentration is lower at M3 than at M1 and M2
589 mainly due to the higher inputs of mineral material, the enhanced microbial loop could
590 also to a lesser extent explain this measure. Our result reveal that coastal
591 eutrophication results in a mixing of refractory mangrove derived SOM with labile OM
592 from algal and sewage sources and this mixing enhances the fungal and bacterial
593 growth and CO₂ fluxes, potentially through a priming effect particularly at the M3 site.

594 **5. Conclusion**

595 This study provides combined data of fatty acid markers along with $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$
596 signatures that characterizes the source of OM in three mangroves within the state of
597 Rio de Janeiro. These mangroves are located along a eutrophication gradient as
598 assessed by $\delta^{15}\text{N}$, $\delta^{13}\text{C}$ signatures and C/N ratio in sediments, and particulate organic
599 matter. The organic matter at the more impacted site M3 sediments was found to be
600 supplied with greater algal or microphytobenthic biomasses than in the lesser impacted
601 site, M1 and M2, in which mangrove production is found to be the main organic input.
602 The SOM at M3 was more labile as indicated by the high contribution of PUFAs.
603 Concomitantly to the algal biomass increase, biomasses of fungi were higher within
604 the M3 sediments than at pristine M1 which suggests an enhancement of the
605 microbial loop. In the highly impacted M3 mangrove sediments, refractory LCFAs, were
606 in lower proportion than in the sediments of pristine site. Although the lower
607 concentration of C_{org} at M3 than at M2 and M1 may be preliminary due to a larger input
608 of mineral particles by rivers, it could also be promoted by a priming effect that
609 potentially decomposes the refractory OM from the mangrove vegetation. In view of

610 the preliminary data presented here, a better understanding of the biogeochemical
611 mechanisms involved in this important global storage of carbon could be gained by
612 implementing experiments under controlled conditions in which known amounts of
613 several OM sources can be added.

614 **Acknowledgments**

615 This work was supported by the French National programs LEFE (CNRS-INSU), the
616 France-Brazil International Research Project VELITROP (*Vulnérabilité des*
617 *Ecosystèmes Littoraux Tropicaux face à l'Eutrophisation*) funded by the French
618 National Centre for Scientific Research (CNRS-INEE), the French chair at the
619 University of the State of Rio de Janeiro (UERJ) *Multidisciplinary approaches to*
620 *describe, understand and monitor eutrophication in the tropical coastal ocean*, the
621 Brazilian National Council for Scientific and Technological Development (CNPq no.
622 314995/2020-0), Carlos Chagas Filho Foundation for Research Support of the State
623 of Rio de Janeiro (FAPERJ no. E-26/203.304/2017), the Coordination for the
624 Improvement of Higher Education Personnel (CAPES, Finance Code 001 and the
625 internalization program PRINTCAPES/UFF FEEDBACKS no. 772
626 99997.310301/2018-00). M.C. is funded by a Ph.D grant of the TOTAL Fondation

627

628 **Literature cited**

- 629 Abril, G., C. Cotovicz Jr., L., Nepomuceno, A., Erbas, T., Costa, S., V. Ramos, V.,
630 Moser, G., Fernandes, A., Negri, E., A. Knoppers, B., Brandini, N., Machado,
631 W., Bernardes, M., Vantrepotte, V., 2022. Spreading eutrophication and
632 changing CO₂ fluxes in the tropical coastal ocean: a few lessons from Rio de
633 Janeiro. *ACMAR* 55, 461–476.
634 <https://doi.org/10.32360/acmar.v55iEspecial.78518>
- 635 Alongi, D. M. (2014). Carbon Cycling and Storage in Mangrove Forests. *Annual Review*
636 *of Marine Science*, 6(1), 195-219. [https://doi.org/10.1146/annurev-marine-](https://doi.org/10.1146/annurev-marine-010213-135020)
637 [010213-135020](https://doi.org/10.1146/annurev-marine-010213-135020)
- 638 Alongi, D. M. (2020). Carbon Balance in Salt Marsh and Mangrove Ecosystems : A
639 Global Synthesis. *Journal of Marine Science and Engineering*, 8(10), 767.
640 <https://doi.org/10.3390/jmse8100767>
- 641 Anand, A., Pandey, P. C., Petropoulos, G. P., Pavlides, A., Srivastava, P. K., Sharma,
642 J. K., & Malhi, R. K. M. (2020). Use of Hyperion for Mangrove Forest Carbon
643 Stock Assessment in Bhitarkanika Forest Reserve : A Contribution Towards
644 Blue Carbon Initiative. *Remote Sensing*, 12(4), 597.
645 <https://doi.org/10.3390/rs12040597>
- 646 Antonio, E. S., & Richoux, N. B. (2016). Tide-Induced Variations in the Fatty Acid
647 Composition of Estuarine Particulate Organic Matter. *Estuaries and Coasts*,
648 39(4), 1072-1083. <https://doi.org/10.1007/s12237-015-0049-x>
- 649 Barcellos, D., Queiroz, H. M., Nóbrega, G. N., de Oliveira Filho, R. L., Santaella, S. T.,
650 Otero, X. L., & Ferreira, T. O. (2019). Phosphorus enriched effluents increase
651 eutrophication risks for mangrove systems in northeastern Brazil. *Marine*
652 *Pollution Bulletin*, 142, 58-63. <https://doi.org/10.1016/j.marpolbul.2019.03.031>

653 Barroso, G. C., Abril, G., Machado, W., Abuchacra, R. C., Peixoto R. B., Bernardes,
654 M., Marques, G. S., Sanders, C. J., Oliveira, G. B., de Oliveira Filho, S. R.,
655 Amora, L., Marotta, H. (Revised). Linking eutrophication to carbon dioxide and
656 methane emissions from exposed mangrove soils along an urban gradient.

657 Bauer, D. F. (1972). Constructing Confidence Sets Using Rank Statistics. *Journal of*
658 *the American Statistical Association*, 67(339), 687-690.
659 <https://doi.org/10.1080/01621459.1972.10481279>

660 Bergamino, L., Dalu, T., & Richoux, N. B. (2014). Evidence of spatial and temporal
661 changes in sources of organic matter in estuarine sediments : Stable isotope
662 and fatty acid analyses. *Hydrobiologia*, 732(1), 133-145.
663 <https://doi.org/10.1007/s10750-014-1853-1>

664 Bianchi, T. S. (2011). The role of terrestrially derived organic carbon in the coastal
665 ocean : A changing paradigm and the priming effect. *Proceedings of the*
666 *National Academy of Sciences*, 108(49), 19473-19481.
667 <https://doi.org/10.1073/pnas.1017982108>

668 Błaszczuk, K., & Krzyśko-Łupicka, T. (2013). Microbial diversity of sewage sludge.
669 *Proceedings of ECOpole*, Vol. 7, No. 2.
670 [https://doi.org/10.2429/proc.2013.7\(2\)059](https://doi.org/10.2429/proc.2013.7(2)059)

671 Bouillon, S., Borges, A. V., Castañeda-Moya, E., Diele, K., Dittmar, T., Duke, N. C.,
672 Kristensen, E., Lee, S. Y., Marchand, C., Middelburg, J. J., Rivera-Monroy, V.
673 H., Smith, T. J., & Twilley, R. R. (2008a). Mangrove production and carbon
674 sinks : A revision of global budget estimates. *Global Biogeochemical Cycles*,
675 22(2), Article 2. <https://doi.org/10.1029/2007GB003052>

676 Bouillon, S., Connolly, R. M., & Lee, S. Y. (2008b). Organic matter exchange and
677 cycling in mangrove ecosystems : Recent insights from stable isotope studies.

678 *Journal of Sea Research*, 59(1), 44-58.
679 <https://doi.org/10.1016/j.seares.2007.05.001>

680 Bouillon, S., Dahdouh-Guebas, F., Rao, A. V. V. S., Koedam, N., & Dehairs, F. (2003).
681 Sources of organic carbon in mangrove sediments: Variability and possible
682 ecological implications. *Hydrobiologia*, 495(1), 33-39.
683 <https://doi.org/10.1023/A:1025411506526>

684 Bouillon, S., Moens, T., Koedam, N., Dahdouh-Guebas, F., Baeyens, W., & Dehairs,
685 F. (2004). Variability in the origin of carbon substrates for bacterial communities
686 in mangrove sediments. *FEMS Microbiology Ecology*, 49(2), 171-179.
687 <https://doi.org/10.1016/j.femsec.2004.03.004>

688 Breitburg, D., Levin, L. A., Oschlies, A., Grégoire, M., Chavez, F. P., Conley, D. J.,
689 Garçon, V., Gilbert, D., Gutiérrez, D., Isensee, K., Jacinto, G. S., Limburg, K.
690 E., Montes, I., Naqvi, S. W. A., Pitcher, G. C., Rabalais, N. N., Roman, M. R.,
691 Rose, K. A., Seibel, B. A., Zhang, J. (2018). Declining oxygen in the global
692 ocean and coastal waters. *Science*, 359(6371), eaam7240.
693 <https://doi.org/10.1126/science.aam7240>

694 Bunting, P., Rosenqvist, A., Lucas, R. M., Rebelo, L.-M., Hilarides, L., Thomas, N.,
695 Hardy, A., Itoh, T., Shimada, M., & Finlayson, C. M. (2018). The Global
696 Mangrove Watch—A New 2010 Global Baseline of Mangrove Extent. *Remote
697 Sensing*, 10(10), 1669. <https://doi.org/10.3390/rs10101669>

698 Camacho-Ibar, V. F., Aveytua-Alcázar, L., & Carriquiry, J. D. (2003). Fatty acid
699 reactivities in sediment cores from the northern Gulf of California. *Organic
700 Geochemistry*, 34(3), 425-439. [https://doi.org/10.1016/S0146-6380\(02\)00211-5](https://doi.org/10.1016/S0146-6380(02)00211-5)

701 Canuel, E. A. (2001). Relations between river flow, primary production and fatty acid
702 composition of particulate organic matter in San Francisco and Chesapeake

703 Bays: A multivariate approach. *Organic Geochemistry*, 32(4), 563-583.
704 [https://doi.org/10.1016/S0146-6380\(00\)00195-9](https://doi.org/10.1016/S0146-6380(00)00195-9)

705 Chen, G. C., Tam, N. F. Y., & Ye, Y. (2010). Summer fluxes of atmospheric greenhouse
706 gases N₂O, CH₄ and CO₂ from mangrove soil in South China. *Science of The*
707 *Total Environment*, 408(13), 2761-2767.
708 <https://doi.org/10.1016/j.scitotenv.2010.03.007>

709 Chen, J., Ferris, H., Scow, K. M., & Graham, K. J. (2001). Fatty acid composition and
710 dynamics of selected fungal-feeding nematodes and fungi. *Comparative*
711 *Biochemistry and Physiology Part B: Biochemistry and Molecular Biology*,
712 130(2), 135-144. [https://doi.org/10.1016/S1096-4959\(01\)00414-6](https://doi.org/10.1016/S1096-4959(01)00414-6)

713 Choudhury, A. K., Das, M., Philip, P., & Bhadury, P. (2015). An Assessment of the
714 Implications of Seasonal Precipitation and Anthropogenic Influences on a
715 Mangrove Ecosystem Using Phytoplankton as Proxies. *Estuaries and Coasts*,
716 38(3), 854-872. <https://doi.org/10.1007/s12237-014-9854-x>

717 Chung, N., & Suberkropp, K. (2008). Influence of shredder feeding and nutrients on
718 fungal activity and community structure in headwater streams. *Fundamental*
719 *and Applied Limnology* 173(1):35-46, 173(1), 35-46.
720 <https://doi.org/10.1127/1863-9135/2008/0173-0035>

721 Clarke, K. R. (1993). Non-parametric multivariate analyses of changes in community
722 structure. *Austral Ecology*, 18(1), 117-143. [https://doi.org/10.1111/j.1442-](https://doi.org/10.1111/j.1442-9993.1993.tb00438.x)
723 [9993.1993.tb00438.x](https://doi.org/10.1111/j.1442-9993.1993.tb00438.x)

724 Cloern, J.E., Foster, S.Q., Kleckner, A.E., 2014. Phytoplankton primary production in
725 the world's estuarine-coastal ecosystems. *Biogeosciences* 11, 2477–2501.
726 <https://doi.org/10.5194/bg-11-2477-2014>

727 Cotovicz, L. C., Knoppers, B. A., Brandini, N., Poirier, D., Costa Santos, S. J., Cordeiro,
728 R. C., & Abril, G. (2018). Predominance of phytoplankton-derived dissolved and
729 particulate organic carbon in a highly eutrophic tropical coastal embayment
730 (Guanabara Bay, Rio de Janeiro, Brazil). *Biogeochemistry*, 137(1), 1-14.
731 <https://doi.org/10.1007/s10533-017-0405-y>

732 Dale, B. (2001). Marine dinoflagellate cysts as indicators of eutrophication and
733 industrial pollution : A discussion. *Science of The Total Environment*, 264(3),
734 235-240. [https://doi.org/10.1016/S0048-9697\(00\)00719-1](https://doi.org/10.1016/S0048-9697(00)00719-1)

735 Dalsgaard, J., St. John, M., Kattner, G., Müller-Navarra, D., & Hagen, W. (2003). Fatty
736 acid trophic markers in the pelagic marine environment. In *Advances in Marine*
737 *Biology* (Vol. 46, p. 225-340). Academic Press. [https://doi.org/10.1016/S0065-](https://doi.org/10.1016/S0065-2881(03)46005-7)
738 [2881\(03\)46005-7](https://doi.org/10.1016/S0065-2881(03)46005-7)

739 David, F., Marchand, C., Taillardat, P., Thành-Nho, N., & Meziane, T. (2018).
740 Nutritional composition of suspended particulate matter in a tropical mangrove
741 creek during a tidal cycle (Can Gio, Vietnam). *Estuarine, Coastal and Shelf*
742 *Science*, 200, 126-130. <https://doi.org/10.1016/j.ecss.2017.10.017>

743 David, F., Marchand, C., Thiney, N., Nhu-Trang, T.-T., & Meziane, T. (2019). Short-
744 term changes in the quality of suspended particulate matter in a human
745 impacted and mangrove dominated tropical estuary (Can Gio, Vietnam).
746 *Continental Shelf Research*, 178, 59-67.
747 <https://doi.org/10.1016/j.csr.2019.03.011>

748 Davis, S. E., Corronado-Molina, C., Childers, D. L., & Day, J. W. (2003). Temporally
749 dependent C, N, and P dynamics associated with the decay of *Rhizophora*
750 mangle L. leaf litter in oligotrophic mangrove wetlands of the Southern

751 Everglades. *Aquatic Botany*, 75(3), 199-215. <https://doi.org/10.1016/S0304->
752 3770(02)00176-6

753 Dugdale, R. C., & Wilkerson, F. P. (1986). The use of ¹⁵N to measure nitrogen uptake
754 in eutrophic oceans; experimental considerations^{1,2}. *Limnology and*
755 *Oceanography*, 31(4), 673-689. <https://doi.org/10.4319/lo.1986.31.4.0673FAO>.
756 (2007). *The world's mangroves 1980-2005*. Food and Agriculture Organization
757 of the United Nations. <http://www.fao.org/3/a1427e/a1427e00.htm>

758 Fenchel, T. (2008). The microbial loop – 25 years later. *Journal of Experimental Marine*
759 *Biology and Ecology*, 366(1), 99-103.
760 <https://doi.org/10.1016/j.jembe.2008.07.013>

761 Ferreira, V., Gulis, V., & Graça, M. A. S. (2006). Whole-stream nitrate addition affects
762 litter decomposition and associated fungi but not invertebrates. *Oecologia*,
763 149(4), 718-729. <https://doi.org/10.1007/s00442-006-0478-0>

764 Fistarol, G. O., Coutinho, F. H., Moreira, A. P. B., Venas, T., Cánovas, A., de Paula,
765 S. E. M. J., Coutinho, R., de Moura, R. L., Valentin, J. L., Tenenbaum, D. R.,
766 Paranhos, R., do Valle, R. de A. B., Vicente, A. C. P., Amado Filho, G. M.,
767 Pereira, R. C., Kruger, R., Rezende, C. E., Thompson, C. C., Salomon, P. S., &
768 Thompson, F. L. (2015). Environmental and Sanitary Conditions of Guanabara
769 Bay, Rio de Janeiro. *Frontiers in Microbiology*, 6.
770 <https://doi.org/10.3389/fmicb.2015.01232>

771 Fontaine, S., Mariotti, A., & Abbadie, L. (2003). The priming effect of organic matter :
772 A question of microbial competition? *Soil Biology and Biochemistry*, 35(6),
773 837-843. [https://doi.org/10.1016/S0038-0717\(03\)00123-8](https://doi.org/10.1016/S0038-0717(03)00123-8)

774 Gontikaki, E., Thornton, B., Cornulier, T., & Witte, U. (2015). Occurrence of Priming in
775 the Degradation of Lignocellulose in Marine Sediments. *PLOS ONE*, *10*(12),
776 e0143917. <https://doi.org/10.1371/journal.pone.0143917>

777 Gritcan, I., Duxbury, M., Leuzinger, S., & Alfaro, A. C. (2016). Leaf Stable Isotope and
778 Nutrient Status of Temperate Mangroves As Ecological Indicators to Assess
779 Anthropogenic Activity and Recovery from Eutrophication. *Frontiers in Plant*
780 *Science*, *7*. <https://doi.org/10.3389/fpls.2016.01922>

781 Guenet, B., Danger, M., Abbadie, L., & Lacroix, G. (2010). Priming effect : Bridging the
782 gap between terrestrial and aquatic ecology. *Ecology*, *91*(10), 2850-2861.
783 <https://doi.org/10.1890/09-1968.1>

784 Hilaluddin, F., Yusoff, F. M., Natrah, F. M. I., & Lim, P. T. (2020). Disturbance of
785 mangrove forests causes alterations in estuarine phytoplankton community
786 structure in Malaysian Matang mangrove forests. *Marine Environmental*
787 *Research*, 104935.

788 Jennerjahn, T., Ittekkot, V., 1997. Organic matter in sediments in the mangrove areas
789 and adjacent continental margins of Brazil .1. Amino acids and hexosamines.
790 *Oceanologica Acta* *20*, 359–369.

791 Jennerjahn, T.C., Ittekkot, V., 1999. Changes in organic matter from surface waters to
792 continental slope sediments off the São Francisco River, eastern Brazil. *Marine*
793 *Geology* *161*, 129–140. [https://doi.org/10.1016/S0025-3227\(99\)00045-6](https://doi.org/10.1016/S0025-3227(99)00045-6)

794 Jessen, B. (2016). Ecological Effects of Nutrient Enrichment on a Coastal Fringe
795 Mangrove Forest. *Open Access Dissertations*. [https://doi.org/10.23860/diss-](https://doi.org/10.23860/diss-jessen-brita-2016)
796 [jessen-brita-2016](https://doi.org/10.23860/diss-jessen-brita-2016)

797 Kaneda, T. (1991). Iso- and anteiso-fatty acids in bacteria : Biosynthesis, function, and
798 taxonomic significance. *Microbiology and Molecular Biology Reviews*, 55(2),
799 288-302.

800 Ke, Z., Tan, Y., Huang, L., Liu, J., Xiang, C., Zhao, C., & Zhang, J. (2019). Significantly
801 depleted 15N in suspended particulate organic matter indicating a strong
802 influence of sewage loading in Daya Bay, China. *Science of The Total
803 Environment*, 650, 759-768. <https://doi.org/10.1016/j.scitotenv.2018.09.076>

804 Kennedy, H., Gacia, E., Kennedy, D. P., Papadimitriou, S., & Duarte, C. M. (2004).
805 Organic carbon sources to SE Asian coastal sediments. *Estuarine, Coastal and
806 Shelf Science*, 60(1), 59-68. <https://doi.org/10.1016/j.ecss.2003.11.019>

807 Kitsiou, D., & Karydis, M. (2011). Coastal marine eutrophication assessment : A review
808 on data analysis. *Environment International*, 37(4), 778-801.
809 <https://doi.org/10.1016/j.envint.2011.02.004>

810 Kopprio, G. A., Dutto, M. S., Garzón Cardona, J. E., Gärdes, A., Lara, R. J., & Graeve,
811 M. (2018). Biogeochemical markers across a pollution gradient in a Patagonian
812 estuary : A multidimensional approach of fatty acids and stable isotopes. *Marine
813 Pollution Bulletin*, 137, 617-626.
814 <https://doi.org/10.1016/j.marpolbul.2018.10.059>

815 Kristensen, E., & Alongi, D. M. (2006). Control by fiddler crabs (*Uca vocans*) and plant
816 roots (*Avicennia marina*) on carbon, iron, and sulfur biogeochemistry in
817 mangrove sediment. *Limnology and Oceanography*, 51(4), 1557-1571.
818 <https://doi.org/10.4319/lo.2006.51.4.1557>

819 Kristensen, E., Bouillon, S., Dittmar, T., & Marchand, C. (2008a). Organic carbon
820 dynamics in mangrove ecosystems : A review. *Aquatic Botany*, 89(2), 201-219.
821 <https://doi.org/10.1016/j.aquabot.2007.12.005>

822 Kristensen, E., Flindt, M. R., Ulomi, S., Borges, A. V., Abril, G., & Bouillon, S. (2008b).
823 Emission of CO₂ and CH₄ to the atmosphere by sediments and open waters in
824 two Tanzanian mangrove forests. *Marine Ecology Progress Series*, 370, 53-67.
825 <https://doi.org/10.3354/meps07642>

826 Leng, M. J., & Lewis, J. P. (2017). C/N ratios and Carbon Isotope Composition of
827 Organic Matter in Estuarine Environments. In K. Weckström, K. M. Saunders,
828 P. A. Gell, & C. G. Skilbeck (Éds.), *Applications of Paleoenvironmental*
829 *Techniques in Estuarine Studies* (p. 213-237). Springer Netherlands.
830 https://doi.org/10.1007/978-94-024-0990-1_9

831 Liu, K.-K., Kao, S.-J., Chiang, K.-P., Gong, G.-C., Chang, J., Cheng, J.-S., & Lan, C.-
832 Y. (2013). Concentration dependent nitrogen isotope fractionation during
833 ammonium uptake by phytoplankton under an algal bloom condition in the
834 Danshuei estuary, northern Taiwan. *Marine Chemistry*, 157, 242-252.
835 <https://doi.org/10.1016/j.marchem.2013.10.005>

836 Lovelock, C. E., Fourqurean, J. W., & Morris, J. T. (2017). Modeled CO₂ Emissions
837 from Coastal Wetland Transitions to Other Land Uses: Tidal Marshes,
838 Mangrove Forests, and Seagrass Beds. *Frontiers in Marine Science*, 4.
839 <https://doi.org/10.3389/fmars.2017.00143>

840 Maher, D. T., Call, M., Santos, I. R., & Sanders, C. J. (2018). Beyond burial : Lateral
841 exchange is a significant atmospheric carbon sink in mangrove forests. *Biology*
842 *Letters*, 14(7), Article 7. <https://doi.org/10.1098/rsbl.2018.0200>

843 Martin, R. M., Wigand, C., Oczkowski, A., Hanson, A., Balogh, S., Branoff, B., Santos,
844 E., & Huertas, E. (2020). Greenhouse Gas Fluxes of Mangrove Soils and
845 Adjacent Coastal Waters in an Urban, Subtropical Estuary. *Wetlands*, 40(5),
846 1469-1480. <https://doi.org/10.1007/s13157-020-01300-w>

847 Meziane, T., d'Agata, F., & Lee, S. Y. (2006). Fate of mangrove organic matter along
848 a subtropical estuary : Small-scale exportation and contribution to the food of
849 crab communities. *Marine Ecology Progress Series*, 312, 15-27.
850 <https://doi.org/10.3354/meps312015>

851 Meziane, T., Lee, S. Y., Mfilinge, P. L., Shin, P. K. S., Lam, M. H. W., & Tsuchiya, M.
852 (2007). Inter-specific and geographical variations in the fatty acid composition
853 of mangrove leaves : Implications for using fatty acids as a taxonomic tool and
854 tracers of organic matter. *Marine Biology*, 150(6), 1103-1113.
855 <https://doi.org/10.1007/s00227-006-0424-z>

856 Meziane, T., & Tsuchiya, M. (2002). Organic matter in a subtropical mangrove-estuary
857 subjected to wastewater discharge : Origin and utilisation by two
858 macrozoobenthic species. *Journal of Sea Research*, 47(1), 1-11.
859 [https://doi.org/10.1016/S1385-1101\(01\)00092-2](https://doi.org/10.1016/S1385-1101(01)00092-2)

860 Mfilinge, P. L., Meziane, T., Bachok, Z., & Tsuchiya, M. (2003). Fatty acids in
861 decomposing mangrove leaves : Microbial activity, decay and nutritional quality.
862 *Marine Ecology Progress Series*, 265, 97-105.
863 <https://doi.org/10.3354/meps265097>

864 Mortillaro, J.-M., Rigal, F., Rybarczyk, H., Bernardes, M., Abril, G., Meziane, T., 2012.
865 Particulate Organic Matter Distribution along the Lower Amazon River:
866 Addressing Aquatic Ecology Concepts Using Fatty Acids. PLOS ONE 7,
867 e46141. <https://doi.org/10.1371/journal.pone.0046141>

868 Napolitano, G. E., Pollero, R. J., Gayoso, A. M., Macdonald, B. A., & Thompson, R. J.
869 (1997). Fatty acids as trophic markers of phytoplankton blooms in the Bahía
870 Blanca estuary (Buenos Aires, Argentina) and in Trinity Bay (Newfoundland,

871 Canada). *Biochemical Systematics and Ecology*, 25(8), 739-755.
872 [https://doi.org/10.1016/S0305-1978\(97\)00053-7](https://doi.org/10.1016/S0305-1978(97)00053-7)

873 Nebbioso, A., & Piccolo, A. (2013). Molecular characterization of dissolved organic
874 matter (DOM) : A critical review. *Analytical and Bioanalytical Chemistry*, 405(1),
875 109-124. <https://doi.org/10.1007/s00216-012-6363-2>

876 Ngosong, C., Gabriel, E., & Ruess, L. (2012). Use of the Signature Fatty Acid 16:1 ω 5
877 as a Tool to Determine the Distribution of Arbuscular Mycorrhizal Fungi in Soil.
878 *Journal of Lipids*, 2012, e236807. <https://doi.org/10.1155/2012/236807>

879 Nóbrega, G. N., Ferreira, T. O., Siqueira Neto, M., Queiroz, H. M., Artur, A. G.,
880 Mendonça, E. D. S., Silva, E. D. O., & Otero, X. L. (2016). Edaphic factors
881 controlling summer (rainy season) greenhouse gas emissions (CO₂ and CH₄)
882 from semiarid mangrove soils (NE-Brazil). *Science of The Total Environment*,
883 542, 685-693. <https://doi.org/10.1016/j.scitotenv.2015.10.108>

884 Nordhaus, I., Salewski, T., Jennerjahn, T.C., 2011. Food preferences of mangrove
885 crabs related to leaf nitrogen compounds in the Segara Anakan Lagoon, Java,
886 Indonesia. *Journal of Sea Research* 65, 414–426.
887 <https://doi.org/10.1016/j.seares.2011.03.006>

888 Nordhaus, I., Salewski, T., & Jennerjahn, T. C. (2017). Interspecific variations in
889 mangrove leaf litter decomposition are related to labile nitrogenous compounds.
890 *Estuarine, Coastal and Shelf Science*, 192, 137-148.
891 <https://doi.org/10.1016/j.ecss.2017.04.029>

892 Oksanen, J., Blanchet, F. G., Kindt, R., Legendre, P., Minchin, P., O'Hara, B., Simpson,
893 G., Solymos, P., Stevens, H., & Wagner, H. (2015). Vegan : Community Ecology
894 Package. *R Package Version 2.2-1*, 2, 1-2.

895 Pérez, A., Libardoni, B. G., & Sanders, C. J. (2018a). Factors influencing organic
896 carbon accumulation in mangrove ecosystems. *Biology Letters*, *14*(10),
897 20180237. <https://doi.org/10.1098/rsbl.2018.0237>

898 Pérez, A., Machado, W., Gutiérrez, D., Borges, A. C., Patchineelam, S. R., & Sanders,
899 C. J. (2018b). Carbon accumulation and storage capacity in mangrove
900 sediments three decades after deforestation within a eutrophic bay. *Marine*
901 *Pollution Bulletin*, *126*, 275-280.
902 <https://doi.org/10.1016/j.marpolbul.2017.11.018>

903 Pérez, A., Machado, W., Gutierrez, D., Smoak, J. M., Breithaupt, J. L., Saldarriaga, M.
904 S., Sanders, L., Marotta, H., & Sanders, C. J. (2020). Carbon and nutrient
905 accumulation in mangrove sediments affected by multiple environmental
906 changes. *Journal of Soils and Sediments*, *20*(5), 2504-2509.
907 <https://doi.org/10.1007/s11368-020-02612-4>

908 Pessenda, L. C. R., Vidotto, E., De Oliveira, P. E., Buso, A. A., Cohen, M. C. L.,
909 Rossetti, D. de F., Ricardi-Branco, F., & Bendassolli, J. A. (2012). Late
910 Quaternary vegetation and coastal environmental changes at Ilha do Cardoso
911 mangrove, southeastern Brazil. *Palaeogeography, Palaeoclimatology,*
912 *Palaeoecology*, *363-364*, 57-68. <https://doi.org/10.1016/j.palaeo.2012.08.014>

913 Pongpam, S., Komiyama, A., Tanaka, A., Sangtiew, T., Maknual, C., Kato, S.,
914 Tanapermpool, P., & Patanaponpaiboon, P. (2009). Carbon dioxide emission
915 through soil respiration in a secondary mangrove forest of eastern Thailand.
916 *Journal of Tropical Ecology*, *25*(4), 393-400.
917 <https://doi.org/10.1017/S0266467409006154>

918 Queiroz, H. M., Ferreira, T. O., Taniguchi, C. A. K., Barcellos, D., do Nascimento, J.
919 C., Nóbrega, G. N., Otero, X. L., & Artur, A. G. (2020). Nitrogen mineralization

920 and eutrophication risks in mangroves receiving shrimp farming effluents.
921 *Environmental Science and Pollution Research*, 27(28), 34941-34950.
922 <https://doi.org/10.1007/s11356-020-09720-1>

923 Ribeiro, C., & Kjerfve, B. (2002). Anthropogenic influence on the water quality in
924 Guanabara Bay, Rio de Janeiro, Brazil. *Regional Environmental Change*, 3,
925 13-19. <https://doi.org/10.1007/s10113-001-0037-5>

926 Rodrigues, R. P., Knoppers, B. A., Souza, W. F. L. de, & Santos, E. S. (2009).
927 Suspended matter and nutrient gradients of a small-scale river plume in
928 Sepetiba Bay, SE-Brazil. *Brazilian Archives of Biology and Technology*, 52,
929 503-512. <https://doi.org/10.1590/S1516-89132009000200030>

930 Sanders, C. J., Eyre, B. D., Santos, I. R., Machado, W., Luiz-Silva, W., Smoak, J. M.,
931 Breithaupt, J. L., Ketterer, M. E., Sanders, L., Marotta, H., & Silva-Filho, E.
932 (2014). Elevated rates of organic carbon, nitrogen, and phosphorus
933 accumulation in a highly impacted mangrove wetland. *Geophysical Research*
934 *Letters*, 41(7), 2475-2480. <https://doi.org/10.1002/2014GL059789>

935 Sanders, C.J., Smoak, J.M., Naidu, A.S., Patchineelam, S.R., 2008. Recent Sediment
936 Accumulation in a Mangrove Forest and Its Relevance to Local Sea-Level Rise
937 (Ilha Grande, Brazil). *Journal of Coastal Research* 24, 533–536.
938 <https://doi.org/10.2112/07-0872.1>

939 Santos, I. R., Maher, D. T., Larkin, R., Webb, J. R., & Sanders, C. J. (2019). Carbon
940 outwelling and outgassing vs. Burial in an estuarine tidal creek surrounded by
941 mangrove and saltmarsh wetlands. *Limnology and Oceanography*, 64(3),
942 996-1013. <https://doi.org/10.1002/lno.11090>

943 Sarker, S., Masud-UI-Alam, M., Hossain, M. S., Chowdhury, S. R., & Sharifuzzaman,
944 S. M. (2021). A review of bioturbation and sediment organic geochemistry in

945 mangroves. *Geological Journal*, 56(5), 2439-2450.
946 <https://doi.org/10.1002/gj.3808>

947 Silva-Filho, E. V., Sanders, C. J., Bernat, M., Figueiredo, A. M. G., Sella, S. M., &
948 Wasserman, J. (2011). Origin of rare earth element anomalies in mangrove
949 sediments, Sepetiba Bay, SE Brazil : Used as geochemical tracers of sediment
950 sources. *Environmental Earth Sciences*, 64(5), 1257-1267.
951 <https://doi.org/10.1007/s12665-011-0942-y>

952 Smith, V. H. (2003). Eutrophication of freshwater and coastal marine ecosystems a
953 global problem. *Environmental Science and Pollution Research*, 10(2), 126-139.
954 <https://doi.org/10.1065/espr2002.12.142>

955 Sun, M.-Y. i, Aller, R. C., Lee, C., & Wakeham, S. G. (2002). Effects of oxygen and
956 redox oscillation on degradation of cell-associated lipids in surficial marine
957 sediments. *Geochimica et Cosmochimica Acta*, 66(11), 2003-2012.
958 [https://doi.org/10.1016/S0016-7037\(02\)00830-X](https://doi.org/10.1016/S0016-7037(02)00830-X)

959 Sun, M.-Y. i, Lee, C., & Aller, R. C. (1993). Laboratory studies of oxic and anoxic
960 degradation of chlorophyll-a in Long Island Sound sediments. *Geochimica et*
961 *Cosmochimica Acta*, 57(1), 147-157. [https://doi.org/10.1016/0016-](https://doi.org/10.1016/0016-7037(93)90475-C)
962 [7037\(93\)90475-C](https://doi.org/10.1016/0016-7037(93)90475-C)

963 Svensen, C., Viličić, D., Wassmann, P., Arashkevich, E., & Ratkova, T. (2007).
964 Plankton distribution and vertical flux of biogenic matter during high summer
965 stratification in the Krka estuary (Eastern Adriatic). *Estuarine, Coastal and Shelf*
966 *Science*, 71(3), 381-390. <https://doi.org/10.1016/j.ecss.2006.07.022>

967 Unger, D., Herbeck, L.S., Li, M., Bao, H., Wu, Y., Zhang, J., Jennerjahn, T., 2013.
968 Sources, transformation and fate of particulate amino acids and hexosamines
969 under varying hydrological regimes in the tropical Wenchang/Wenjiao Rivers

970 and Estuary, Hainan, China. *Continental Shelf Research, Land – Sea*
971 *Interactions in Tropical Ecosystems of Hainan, China* 57, 44–58.
972 <https://doi.org/10.1016/j.csr.2012.02.014> Volkman, J. K. (2006). Lipid Markers
973 for Marine Organic Matter. In J. K. Volkman (Éd.), *Marine Organic Matter :
974 Biomarkers, Isotopes and DNA* (p. 27-70). Springer.
975 https://doi.org/10.1007/698_2_002

976 Wakeham, S. G., Hedges, J. I., Lee, C., Peterson, M. L., & Hernes, P. J. (1997).
977 Compositions and transport of lipid biomarkers through the water column and
978 surficial sediments of the equatorial Pacific Ocean. *Deep Sea Research Part II:
979 Topical Studies in Oceanography*, 44(9), 2131-2162.
980 [https://doi.org/10.1016/S0967-0645\(97\)00035-0](https://doi.org/10.1016/S0967-0645(97)00035-0)

981 Wang, F., Sanders, C.J., Santos, I.R., Tang, J., Schuerch, M., Kirwan, M.L., Kopp,
982 R.E., Zhu, K., Li, X., Yuan, J., Liu, W., Li, Z., 2021. Global blue carbon
983 accumulation in tidal wetlands increases with climate change. *National Science
984 Review* 8, nwaa296. <https://doi.org/10.1093/nsr/nwaa296>

985 Webber, M.K., 2016. Chapter 49- Mangroves, in: *First Global Integrated Marine
986 Assessment (First World Ocean Assessment)*. p. 19.

987 Xiong, Y., Liao, B., & Wang, F. (2018). Mangrove vegetation enhances soil carbon
988 storage primarily through in situ inputs rather than increasing allochthonous
989 sediments. *Marine Pollution Bulletin*, 131, 378-385.
990 <https://doi.org/10.1016/j.marpolbul.2018.04.043>

991 Xu, Y., & Jaffé, R. (2007). Lipid biomarkers in suspended particles from a subtropical
992 estuary : Assessment of seasonal changes in sources and transport of organic
993 matter. *Marine Environmental Research*, 64(5), 666-678.
994 <https://doi.org/10.1016/j.marenvres.2007.07.004>

995 Zhu, P., Wang, Y., Shi, T., Huang, G., & Gong, J. (2018). Genetic Diversity of Benthic
996 Microbial Eukaryotes in Response to Spatial Heterogeneity of Sediment
997 Geochemistry in a Mangrove Ecosystem. *Estuaries and Coasts*, 41(3), 751-764.
998 <https://doi.org/10.1007/s12237-017-0317-z>