



HAL
open science

How can interspecific interactions in freshwater benthic macroinvertebrates modify trace element availability from sediment?

Victoria Soledad Andrade, Claudia Wiegand, Alexandrine Pannard, Ana María Gagneten, Mathieu Pédrot, Martine Bouhnik-Le Coz, Christophe Piscart

► **To cite this version:**

Victoria Soledad Andrade, Claudia Wiegand, Alexandrine Pannard, Ana María Gagneten, Mathieu Pédrot, et al.. How can interspecific interactions in freshwater benthic macroinvertebrates modify trace element availability from sediment?. *Chemosphere*, 2020, 245, pp.125594. <10.1016/j.chemosphere.2019.125594>. <insu-02403881>

HAL Id: insu-02403881

<https://insu.hal.science/insu-02403881v1>

Submitted on 11 Dec 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.



HAL Authorization

Journal Pre-proof

How can interspecific interactions in freshwater benthic macroinvertebrates modify trace element availability from sediment?

Victoria Soledad Andrade, Claudia Wiegand, Alexandrine Pannard, Ana María Gagneten, Mathieu Pédrot, Martine Bouhnik-Le Coz, Christophe Piscart



PII: S0045-6535(19)32834-6

DOI: <https://doi.org/10.1016/j.chemosphere.2019.125594>

Reference: CHEM 125594

To appear in: *ECSN*

Received Date: 16 September 2019

Revised Date: 2 December 2019

Accepted Date: 8 December 2019

Please cite this article as: Andrade, V.S., Wiegand, C., Pannard, A., Gagneten, Ana.Mari., Pédrot, M., Bouhnik-Le Coz, M., Piscart, C., How can interspecific interactions in freshwater benthic macroinvertebrates modify trace element availability from sediment?, *Chemosphere* (2020), doi: <https://doi.org/10.1016/j.chemosphere.2019.125594>.

This is a PDF file of an article that has undergone enhancements after acceptance, such as the addition of a cover page and metadata, and formatting for readability, but it is not yet the definitive version of record. This version will undergo additional copyediting, typesetting and review before it is published in its final form, but we are providing this version to give early visibility of the article. Please note that, during the production process, errors may be discovered which could affect the content, and all legal disclaimers that apply to the journal pertain.

© 2019 Published by Elsevier Ltd.

Credit author statement

Andrade, Victoria Soledad: main investigation, and writing of the initial draft

Wiegand, Claudia: investigation, supervision, methodology, reviewing and editing of the manuscript

Pannard, Alexandrine: investigation, methodology, formal analysis, visualisation

Gagnetten, Ana María: funding acquisition, project administration, supervision,

Pédrot, Mathieu: investigation, methodology,

Bouhnik-Le Coz Martine: investigation, methodology,

Piscart, Christophe: conceptualization, funding acquisition, project administration, supervision, investigation, methodology, providing of resources for the study, co-writing of the initial manuscript.

1 **How can interspecific interactions in freshwater benthic**
2 **macroinvertebrates modify trace element availability from**
3 **sediment?**

4

5 Andrade, Victoria Soledad*^{1, 2,3}; Wiegand, Claudia³; Pannard, Alexandrine³; Gagnetten,
6 Ana María¹; Pédrot, Mathieu⁴; Bouhnik-Le Coz Martine⁴; Piscart, Christophe³

7

8 ¹*Present address:* Lab. de Ecotoxicología. Facultad de Humanidades y Ciencias.
9 Universidad Nacional del Litoral. Ciudad Universitaria, Ruta Nacional N° 168, Km 472.4,
10 3000. Santa Fe, Argentina.

11 ² Consejo Nacional de Investigaciones Científicas y Tecnológicas (CONICET), Godoy
12 Cruz 2290, C1425FQB, Buenos Aires, Argentina.

13 ³ Univ Rennes, CNRS, ECOBIO - UMR 6553, 263 Avenue Général Leclerc, Campus de
14 Beaulieu, Bâtiment 14A, F-35000, Rennes, France.

15 ⁴ Univ Rennes, CNRS, Géosciences Rennes - UMR 6118, 263 Avenue Général Leclerc,
16 Campus de Beaulieu, Bâtiment 15, F-35000, Rennes, France.

17 *victoria.s.andrade@gmail.com.

18 +54-0342-155139910. Sarmiento 2873 2B, 3000, Santa Fe, Argentina.

19

20

21 Abstract

22 This study aimed to assess how bioturbation by freshwater benthic macroinvertebrates with
23 different biological traits alone or in combination could modify trace elements (TE) fate
24 between sediment and water, and if water TE concentration and animal TE content impair
25 their body stores. Three macroinvertebrate species were exposed to TE contaminated
26 sediment for 7 days: the omnivorous *Echinogammarus berilloni* (Amphipoda), the
27 sediment feeding *Tubifex tubifex* (Oligochaeta) and the filter feeding *Pisidium* sp.
28 (Bivalvia). Treatments were one without invertebrates (control), two with amphipods or
29 mussels alone, and the combinations amphipod-mussel, and amphipod-mussel-worms.
30 Water TE concentration increased significantly in 2 or 3 species mesocosms, concerning
31 mainly Rare Earth Elements, Cr, U and Pb, known to be associated to the colloidal phase.
32 By contrast, water soluble TE were not affected by animals. For both, amphipods and
33 mussels, TE body content increased with the number of coexisting species. For amphipods,
34 this increase concerned both, soluble and colloid-associated TE, possibly due to intense
35 contact and feeding from sediment and predation on tubificids. TE bioaccumulation in
36 mussel was less important and characterized by soluble TE, with water filtration as most
37 plausible uptake route. Protein, triglyceride and Whole Body Energy Budget increased in
38 amphipods with the number of coexisting species (probably by feeding on mussels' feces
39 and tubificids) whereas triglycerides declined in mussels (presumably filtration was
40 disturbed by amphipods). This study highlights interspecific interactions as key drivers
41 explaining both: TE bioturbation, depending on their water solubility or colloidal
42 association, and the exposure/contamination of species through another species activity.

43

44 **Keywords:** Bioturbation, Trace element, Macroinvertebrate, Rare earth element.

45 **1 Introduction**

46 Benthic macroinvertebrates bioturbate the sediment-water interface and are considered as
47 ecosystem engineers because they modify this environment and influence the availability of
48 resources and contaminants (Berckenbusch and Rowden, 2003). As many anthropogenic
49 pollutants find their way into aquatic ecosystems and into the sediments as final sink
50 (Eggelton and Thomas, 2004), bioturbation can modify their fate and distribution, either
51 directly by mixing or resuspending subsurface and surface sediment, or indirectly by
52 altering biogeochemical conditions (stability, redox conditions, organic content) (Blankson
53 and Klerks, 2016).

54 Within the hydrographic network, trace elements (TE) are distributed as dissolved, colloid,
55 or particulate form (Stumm and Morgan, 1996). Mineral constituents of sediments,
56 particularly the clay fraction, exhibit large surface areas, thus huge sorption capacity
57 regarding TE and organic compounds (Morley and Gadd, 1995). Moreover, redox-related
58 biogeochemical processes facilitate the transition of the particle bound TE to the pore water
59 and then to the pelagic zone (Xie et al., 2018). Remobilization of TE from sediments
60 depends on many factors, including the physico-chemical characteristics of the TE itself,
61 the content of organic matter and clay in the sediment, the hydrological flow, and non-the
62 least the bioturbation by the benthic community (Blankson and Klerks, 2017; He et al.,
63 2017, 2019; Xie et al., 2018).

64 Between TE, rare earth elements' (REE) characteristics make them sensitive tracers of
65 water-rock interactions, groundwater mixing, past redox conditions and soil phases
66 contributing to TE sources (e.g. Davranche et al., 2011; Laveuf et al., 2012; Pédrot et al.,
67 2015). REE in aqueous solution have an organic speciation and complexation in the
68 presence of natural organic molecules (e.g. Marsac et al., 2010; Pédrot et al., 2008),

69 exhibiting specific REE signatures (Davranche et al., 2005; Tang and Johannesson, 2010).
70 Organic complexation of REE implies the development of a REE pattern in solution with a
71 middle rare earth element (MREE) downward concavity, associated to an absence of Ce
72 anomaly. Consequently, REE fingerprinting can allow to trace organic colloid
73 solubilization (Pédrot et al., 2010) and can be used to distinguish a differential
74 solubilization of organic colloids according to the activity of benthic communities.

75 Benthic macroinvertebrates bioturbate by feeding, locomotion, and using the sediment for
76 their habitat (Pearson, 2001; Mermillod-Blondin and Rosenberg, 2006). Consequently,
77 several functional groups have been proposed: biodiffusers, who randomly mix the
78 sediment (e.g. amphipods); upward and downward conveyors, who move the sediment
79 vertically through feeding activity; regenerators, burrowing organisms that relocate
80 sediment (e.g. bivalves); and gallery diffusors, burrowing organisms that generate extensive
81 tunnels (e.g. *Tubifex tubifex*) (François et al., 2002). Few bioturbation studies have focused
82 on interspecific interactions affecting exposure to contaminants. For instance, *Lumbriculus*
83 *variegatus* (Oligochaeta) bioturbation augmented zinc resuspension, but reduced its
84 bioavailability for *Chironomus tepperi* (Chironomidae) (Colombo et al., 2016). Remaili et
85 al. (2016) found an increase in metals by bioturbation with the number of coexisting
86 species and a higher tissue concentration of chromium in bivalve in the presence of
87 amphipods.

88 Benthic macroinvertebrates are good sentinels of pollution on freshwater ecosystems
89 because of their relatively small action radius sensitivity to contaminants (Kiffney and
90 Clements, 1994) and some with relatively long-life cycles (Rosenberg and Resh, 1993),
91 moreover diverse ecological functions, thus taking part in a multitude of biological
92 interactions (Gerino et al., 2003). Even though interspecific interactions are complex

93 (McAuliffe, 1984), it needs to be revealed how these diverse functional groups –
94 individually or grouped – impact on contaminants fate, bioaccumulation and physiological
95 effects.

96 To address these questions, three sediment-associated macroinvertebrates were selected: the
97 omnivorous biodiffusor *Echinogammarus berilloni* (Amphipoda), the tunnel generating and
98 sediment feeding *Tubifex tubifex* (Oligochaeta) and the burrowing and filter feeding
99 *Pisidium* sp. (Bivalvia). *Echinogammarus berilloni* inhabits streams and rivers with a
100 preference for running waters (Piscart et al., 2007, 2011). *Pisidium* sp. lives in a wide
101 variety of habitats and seems to be of low sensibility towards high organic matter and
102 decreased oxygen saturation (Herrington, 1962; Zeybek et al., 2012). *Tubifex tubifex*
103 inhabits fine sediment in freshwater lakes or lotic river sections, it resuspends the upper
104 sediment layer by feeding (Milbrink, 1987) and is relatively resistant to pollution
105 (Karikhoff and Morris, 1985).

106 The aim of this study was to assess how bioturbation by freshwater benthic
107 macroinvertebrates with different biological traits (i.e. omnivorous, filter feeder and deposit
108 feeder) alone or in combination (single to three species) could modify TE distribution
109 between sediments and water, using small mesocosms as exposure scenario. If bioturbation
110 enhances bioavailability it might also cause physiological effects in the organisms exposed.
111 Consequently, the specific goals were i) to analyze how macroinvertebrates bioturbation
112 and their interspecific interactions affect the water TE concentration and the TE content in
113 invertebrates; and ii) to analyze if water TE concentration and animal content impair body
114 stores of macroinvertebrates.

115

116 **2 Material and methods**

117 2.1 Sediment collection

118 Sediment was collected using a grab sampler in the Sélune River (Normandy, France) at the
119 confluence with the Yvrande River (48°34'24.0"N 1°11'04.8"W), a tributary known by its
120 historical TE pollution caused by industries (DREAL-Normandie, 2012). The sediment,
121 remaining wet in its water, was sieved (\emptyset 2mm), homogenized and stored at 4°C two days
122 before the experiment.

123

124 2.2 Organisms collection

125 *Echinogammarus berilloni* (Amphipoda) and *Pisidium* sp. (Bivalvia) were collected in the
126 Petit Hermitage River (48°29'16.1"N 1°34'15.8"W) located in the Villecartier forest,
127 considered as a reference site (Piscart et al., 2009). Specimens were maintained for one
128 day in water of their origin at $16.0 \pm 1^\circ\text{C}$ with aeration. *Tubifex tubifex* (Oligochaeta) were
129 bought from a pet shop (Grebil Co., France) and maintained in the same conditions.
130 Biofilm was collected in the Sélune River upstream of the confluence with the polluted
131 Yvrande River using artificial substrates (glass plates) exposed for one month at 1m depth.
132 Biofilm was then inoculated on glass gems (previously washed with HCl 1N and Milli-Q to
133 remove TE) by incubating them during one month in a climate-controlled incubator with
134 $30.0 \pm 1 \mu\text{mol m}^{-2}\text{s}^{-1}$ of light, 12:12 light:dark regime at $20.0 \pm 1^\circ\text{C}$.

135

136 2.3 Experimental design

137 Flat round glass bowls (\emptyset 14 cm) were filled with 100 mL of sediment and 200 mL of water
138 from the reference site where invertebrates were collected. The approximate depth for
139 sediment was 0.7 cm, and for water, 1.7 cm. The contact water/sediment surface was 154

140 cm². The water/sediment proportion was chosen according to the number of organisms and
141 their body sizes, and to ensure oxygen supply as no air bubbling was used to not add
142 additional water current. The bowls were left to settle in a climate-controlled incubator
143 (30±1 µmol m⁻²s⁻¹ light, 12:12 light:dark regime, 16±1 °C). After 24h, invertebrates and 5
144 biofilm gems were added. Five combinations of organisms were performed with 5
145 replicates each: one without invertebrates (control), two with 10 individuals of either
146 amphipods or mussels alone, one combination of amphipod-mussel (10 individuals of
147 each), and a combination amphipod (10 individuals) - mussel (10 individuals) - worms (20
148 individuals).

149 The exposure lasted seven days to avoid any starvation (Foucreau et al., 2013) or oxygen
150 depletion (Lawrence and Mason, 2001). From each bowl, one individual was frozen at -
151 25°C for body stores and another specimen was dried at 65°C for 72h for TE content; four
152 animals were frozen at -80°C for later enzyme analysis. Unfortunately none of the *T. tubifex*
153 could be recovered. Samples of water and sediment were collected for nutrient
154 concentration and TE content analysis.

155

156 *2.4 Analysis of nutrients in the water*

157 Ammonium, nitrate and phosphate concentrations were measured at the beginning and at
158 the end of the experiment from filtered water (GF/F) by colorimetric methods (Aminot and
159 Chaussepied, 1983) using a Gallery Discrete Autoanalyzer (Thermo Scientific). The pH
160 was measured with a combined Mettler InLab® electrode. Dissolved oxygen was measured
161 with an Odeon portable apparatus (Ponsel) equipped with an optod probe.

162

163 *2.5 Trace elements determination*

164 The elemental compositions of the sediment samples were determined at the CNRS Service
165 d'Analyse des Roches et des Minéraux (SARM), Centre de Recherches Pétrographiques et
166 Géochimiques (CRPG-CNRS, Nancy, France). After sample preparation (described in
167 Carignan et al., 2001), the major element concentrations were determined by inductively-
168 coupled plasma optical emission spectrometry (ICP-OES, Thermo ICap 6500) and the TE
169 concentrations by ICP-mass spectrometry (Thermo Elemental - X7 ICP-MS). The
170 bioavailable TE fraction in sediment was analyzed by leaching the water, acid soluble and
171 exchangeable fraction of sediment, following the first step of modified BCR extraction
172 (Pédrot et al., 2009). The first step was applied three times to 1 g of dried sediment sample,
173 in ultracleaned 50-ml polypropylene centrifugal tubes. Extractant and procedure used: 0.11
174 M CH_3COOH , 40 mL, room temperature, shake for 16 h, centrifuged 3000 g for 30 min.

175 All collected waters were filtered using 0.2- μm -pore-size filter capsules (polyether sulfone
176 membrane Sartorius Minisart). Major cation and TE concentrations were determined by
177 ICP-MS (Agilent 7700x), using rhenium and rhodium as internal standard. Calibration
178 curves and accuracy controls were performed in accordance with Yéghicheyan et al.
179 (2013), using river water reference material for cation and trace element with a large
180 compositional range (the international geostandard SLRS5, National Research Council of
181 Canada). The water samples were solubilized in 0.37 M HNO_3 with appropriate dilution(s)
182 regarding the ICP-MS quantifications limits. At the beginning of the experiment, TE
183 concentration were assessed in sediment (including the BCR), animals (supplementary data
184 S1 and S2) and water; and at the end of the experiment, in water and animals.

185 For animal tissues, samples were prepared in a clean room in acid-washed digestion vessels
186 (Savillex), respectively 50-mL tubes (24h in 1.5M HNO_3 at 45°C, 24h in deionized water at
187 45°C - repeated twice for the digestion vessels). Dry tissues were digested 5 times in sub-

188 boiled nitric acid (14.6 M HNO₃), with or without hydrogen peroxide (H₂O₂), followed by
189 8h evaporation (95°C) each time. Final solids were re-solubilized in 0.37 M HNO₃ with
190 appropriate dilution(s) regarding the ICP-MS quantifications limits. Typical uncertainties
191 including all error sources were below 5% for all TE, whereas for major cations, the
192 uncertainty lied between 2% and 5%.

193

194 *2.6 Body stores determination*

195 The concentrations in proteins, glycogen and triglycerides were measured *via*
196 colorimetric assays, as described by Foray et al. (2012). Each individual was lyophilized
197 and weighed (Balance XP2U Mettler Toledo, Columbus, OH, d=0.1 µg). A volume of 600
198 µL of phosphate buffer was added to each sample, then homogenized for 1 min 30 sec at
199 25Hz (bead-beating device, Retsch™ MM301, Retsch GbmH, Haan, Germany), and
200 centrifuged (500 g, for 5 min, 4 °C). 2.5 µL of the supernatant were collected for
201 quantifying protein content according to Bradford (1976) using bovine serum albumin as
202 standard.

203 The rest of the supernatant (597.5 µL) was mixed with 900 µL of a methanol-chloroform
204 solution (2/1, volume/volume). After separation of the phases by centrifugation at 4 °C at
205 180g for 15 min, 300 µL of chloroform was transferred into new microtubes for triglyceride
206 assays, and the pellets were kept for Glycogen content. Chloroform was evaporated using a
207 speedvac (MiVac DUO, Genevac, France) before redissolving the remains in 400 µL of
208 Triton-BSA buffer. The manufacturer's instructions were followed for the triglycerides
209 colorimetric assay (Triglycerides, kit reference CC02200, LTA srl, Italy).

210 The glycogen content was assayed from the pellets by adding 1200 µL of fresh anthrone
211 solution (1.42 g L⁻¹ anthrone in 70% sulfuric acid). Samples were heated at 90 °C for 15

212 min, and absorbance was measured at 625 nm after a 2-or 4-fold dilution. Glucose was used
213 as standard.

214 The Whole Body Energy Budget (WBEB, kJ.g^{-1}) was calculated as sum of combustion of
215 ($17 \text{ kJ.g}^{-1} \text{ glycogen} + 39.5 \text{ kJ.g}^{-1} \text{ triglycerides} + 24 \text{ kJ.g}^{-1} \text{ proteins}$) expressed per gram of
216 dry weight of the sample (Smolders et al., 2003).

217

218 *2.7 Statistical analysis*

219 PCA was performed on the TE concentrations in water and the body content of
220 invertebrates; dominant and physiologically relevant TE (Ca, K, Na, and Mg) were
221 removed for the second PCA because they could be modified by physiological activities.

222 A PCA was also performed on nutrients concentrations, after scaling and reducing the data.

223 To highlight similar controlling processes (covariation, not causal-effect link) on nutrients
224 and TE to treatments, the co-variations of nutrients and TE ordinations from the first two
225 axes of PCAs were analyzed through a co-inertia (Doledec and Chessel, 1994; Dray et al.,
226 2003). The expected link here is not a causal – effect one, but covariations, which would
227 indicate similar controlling processes. The costructure between nutrients and TE is shown
228 on the plot of samples scores, on which the corresponding sample scores from both PCA
229 ordinations are linked by an arrow. The strength of the costructure is thus given for each
230 sample by the length of the arrow, with smallest arrows indicating maximum strength. To
231 test statistically the global strength of the costructure, a permutation test of Monte Carlo has
232 been performed by randomly permuting rows and columns of one PCA and calculating the
233 new co-structure (1000 permutations). All the analyses were performed using ade4 and
234 FactoMineR libraries implemented in R freeware (R Core Team, 2013).

235 The differences in nutrient or TE concentrations in water between treatments were assessed
236 using one-way ANOVAs. The differences in TE body content and body stores were tested
237 using two-ways ANOVAs with invertebrate species and species combinations as fixed
238 factors. For all ANOVAs, Tukey's HSD tests were used to carry out post-hoc pairwise
239 comparisons. All ANOVAs were performed using procedures from Statistica 7.1 (StatSoft,
240 2004).

241

242 **3 Results**

243 In all mesocosms and treatments, the physico-chemical parameters remained suitable for
244 the animals. The oxygen concentration remained above 7.0 mg.L⁻¹ and pH ranged from 7.0
245 to 7.3 except for one mesocosm that increased until 7.7. For both, amphipods and mussels,
246 survival rates were >90% in each mesocosm, except for one with only 80% of survival.

247

248 *3.1 Nutrient and TE concentrations in Water*

249 Nitrate concentration increased significantly in the 2-species exposure ($p < 0.008$, Figure
250 1). Ammonium and phosphate concentrations both increased significantly in the 3-species
251 exposure ($p < 0.0029$). Moreover, phosphate concentrations were significantly higher in all
252 exposures containing amphipods ($p < 0.003$); a similar tendency was also observed for
253 ammonium.

254 The first component (PC1, Figure 2a) of the PCA of the TE concentrations in the water
255 represent 54.6% of the total inertia and is explained by most of TE, whereas the second
256 component (PC2) explains 16.3% variations of a group of 9 TE (B, Ba, Cd, Co, Li, Mn,
257 Mg, Sr, and Ca). Eigenvalues are shown in Figure 2b. The water in mesocosms without
258 invertebrates (controls) and the mesocosms with only one invertebrate species (mussels or

259 amphipods) had similar TE concentration, even though mesocosms with amphipod tended
260 to have a higher content in TE associated with the second component (Figure 2c). The
261 water of mesocosms with 2 or 3 invertebrate species had a significantly higher TE
262 concentration than mesocosms with only one species (P-values < 0.05); for 76.7% of
263 metals, these TE being associated with the first component. This group include REE, Fe,
264 Cr, U, and Pb, which increased significantly with number of species (Figure 3a). Moreover,
265 a MREE downward concavity was evidenced in all modalities, with an absence of
266 evolution of Ce anomaly (data not shown). The increase of TE concentration is particularly
267 strong for TE associated to organic colloids (e.g. Cr, Pb, REE and U), which are all on the
268 right part of the PCA, whereas true dissolved TE (Li, B, Ba, Ca, K, Mg, Mn, Na and Sr)
269 were not affected (Figures 2a and 3d,e). A co-inertia (Figure 4) has been performed from
270 the previous PCA on TE concentrations and from a PCA on nutrients concentrations. Both
271 ordinations showed a high percentage of variance on the first two axes, with 90 % for
272 nutrients and 70 % for TE. The first axis of the co-inertia separates the three species
273 treatments on the left part of the co-inertia from controls and mussels on the right part, with
274 Amphipods and two species in the middle (Figure 4). Compared to the initial PCA, some
275 TE are highlighted by the co-inertia, such as Rb for the first axis and Cu, Zn and As for the
276 second axis (Figure 4). The dispersion of the samples on the plot moreover increased with
277 the number of species.

278

279 *3.2 TE content in animal's body*

280 The mean TE content in non-exposed animals were relatively low. The ratios between the
281 TE content in mesocosms with one species VS non-exposed animals ranged from 1 to 10
282 for amphipods and from 0.4 to 2 for mussels.

283 The TE body contents for exposed animals are ranged between the two first components
284 (PC1 and PC2) representing a total of 73.6% of the inertia (Figure 5a). The first component
285 (PC1) represent 51% of the total inertia and is explained by most of TE, whereas the second
286 component (PC2) explains 12.6% variations of a group of 13 TE (As, B, Ba, Cd, Co, Cu,
287 Eu, Fe, Mn, Rb, Sb, Sr, and Zn). Eigenvalues are shown in Figure 5b. Apart from Ca,
288 which is higher in mussels ($F_{1,2,24} = 462.8$; $P < 0.001$), the overall body content of TE was
289 significantly higher in amphipods compared to mussels (Figure 5c). Concerning single TE,
290 apart from Ni ($P = 0.129$), and Cd ($P = 0.454$), all others were significantly more
291 concentrated by amphipods than by mussels (P -values < 0.003). The body content in
292 animals increased according to both, the species considered and the number of species in
293 mesocosms. For amphipods, increase in TE concerned mainly TE associated with the
294 second axis (e.g. Rb, Cu, Sr, and Mn), whereas for mussels this increase was mainly
295 associated with the TE depicted on the first component (e.g. Al, Ce, Nd, and La) (Figure
296 5c).

297 Amphipods' TE body contents were not correlated with TE concentration in water.
298 However, for mussels, significant correlations, most of them being negative were found
299 with 22 TE, including all REE, V Be, Cr, Fe, Ga, Pb, U, Y, and Zn.

300

301 *3.3 Body stores*

302 Protein content differed significantly between the species but not according to the
303 treatments, despite a slight increase in amphipods with number of coexposed species
304 (Figure 6a). Glycogen content did not differ significantly, neither between species nor
305 between combinations of species (Figure 6b, P -values > 0.165). Triglyceride contents did
306 not differ significantly between amphipods and mussels in mesocosms with one or two

307 species (P-values > 0.276) but became significant in mesocosms with three species (Figure
308 6c, c; P-values <0.018).

309 The WBEB was not significantly higher in amphipod than in mussels in mesocosms with
310 one species (Figure 6d) but was significantly higher in the two and three species exposures
311 (P = 0.004). TE concentrations in water did not significantly influence the WBEB, whereas
312 significant negative correlations between WBEB and TE body content were observed. For
313 amphipods, WBEB was negatively correlated with 22 TE (all REE, Fe, Ga, Pb, Sc, Si, U,
314 V, and Y). For mussels, negative correlations were only found with Li (P = 0.027), Sb (P =
315 0.036) and Sc (P = 0.043).

316

317 **4 Discussion**

318 *4.1 Nutrients and TE in Water*

319 Bioturbation of the upper sediment layer by amphipods and oligochaetes increased the
320 concentration of ammonium and phosphate in the water, confirming studies by Mermillod-
321 Blondin et al. (2008). Moreover, excretion by both organisms might have contributed,
322 especially because animal biomass increased with the number of species in mesocosms
323 (Fukuhara and Yasuda, 1989; Gardner et al., 1981; Nalepa et al., 1983). Amphipods are
324 moreover suspected to have benefited from *T. tubifex* as nutrient source, thus their
325 excretion and the decay of the oligochaetes contributed to the high values in these
326 exposures. Contrarily, mussels alone did not change any of their energy storage, probably
327 due to their lower activity in bioturbation.

328 Bioturbation similarly modified the TE content in the water, again in particular in
329 exposures including amphipods. Li, B, Ba, Ca, K, Mg, Mn, Na and Sr, were not affected by
330 the animals but crossed the benthic-pelagic border by diffusion. Indeed, these elements are

331 known to be not or slightly associated to the colloidal phase, resulting in a distribution
332 within the truly dissolved phase (< 1 nm) (Pédrot et al., 2008). Within the dissolved
333 fraction in river waters (< 0.22 μm), Fe-organic colloids are major carriers of TE with
334 lower solubility (e.g. Cr, U, Pb and REE) (Pokrovsky and Schott, 2002; Sholkovitz et al.,
335 1995). REE fingerprinting confirmed their organic complexation, and the increase of REE
336 concentrations with the number of species showed an increase of organic colloid
337 solubilization. Cr, U and Pb and REE were the most affected by the presence of organisms,
338 indicating that bioturbation was the main factor releasing TE associated to colloids (less
339 bioavailable) from the sediments. This study confirms that bioturbation contributes to TE
340 redistribution from the sediment into the pelagic phase thus increasing its bioavailability. It
341 moreover contributes to the understanding of the specificity of this process depending on
342 the water solubility or colloidal association of the TE.

343 The contribution of each bioturbator type however, varied considerably. The least
344 contributed the mussels, which even though living buried in the sediment seemed to have
345 used the overlaying water for their respiration and filtering activities without causing
346 disturbances of the sediment. The amphipods conversely, foraging by moving their
347 sideways compressed body with their legs, apparently swirled sediment particles.
348 Tubificids might have added to this bioturbation by their movements and moreover by
349 feeding from deeper sediment layers and rejecting the feces into the overlaying space (He et
350 al., 2019; Karlekhoff and Morris, 1985), but we could not prove it with our experimental
351 set up. Similarly, comparing a range of benthic species, *Gammarus pulex* bioturbation
352 activity was found to be higher than that of *T. tubifex*, which in turn liberated higher
353 quantities of Cu from the sediment and considerably increased the toxicity (van der Meer et
354 al., 2017).

355

356 *4.2 TE content in animal's body*

357 TE bioavailability to aquatic invertebrates is complex as both uptake routes (waterborne
358 and dietary) exist in parallel (Brix et al., 2011). The dominant pathway may vary depending
359 on species, permeable surfaces, feeding behaviour, metal, concentrations and available
360 binding sites (De Jonge et al., 2010; Marsden and Rainbow 2004). Generally, amphipods
361 seemed to accumulate more TE than mussels, confirming that they are physiologically good
362 bioaccumulators (Geffard et al., 2010; Schaller et al., 2011). Amphipods foraging
363 behaviour in intense contact to the sediment exposes their thin articular membranes and
364 gills as uptake routes for TE. Secondly, feeding activities, especially predation on
365 invertebrates such as oligochaetes (Piscart et al., 2011), tubificids in this scenario, also
366 promotes their contamination with TE through biomagnification. Contrarily,
367 bioaccumulation in mussels was lower and characterized by soluble TE (e.g. As and Zn),
368 whose main uptake route was presumably by water filtration.

369 For both, amphipods and mussels, the whole TE content increased with the number of
370 species in the mesocosms. TE affected by the experimental conditions were however not
371 the same. In mesocosms with amphipods and mussels, the water soluble TE increased in
372 amphipods which could be explained by the filtering activity of mussels, clearing the TE
373 from the water column but concentrated them with their faeces near the sediment (Vaughn
374 and Hakenkamp, 2001) where they are consumed by amphipods (Gergs and Rothaupt,
375 2008). In those mesocosms, for mussels the soluble TE content decreased whereas colloidal
376 TE content increased, according to its increase in the water caused by amphipods'
377 bioturbation. Finally for both, amphipods and mussels, the increase in TE content were
378 more important in the presence of tubificids. The tubificids' bioturbation activity might

379 have transferred TE from deeper layers of the sediment to the pelagic zone (He et al., 2019;
380 Mermillod-Blondin et al., 2007; Reible et al., 1996), or modified physical properties of
381 sediment, e.g. redox potential (Graf and Rosenberg, 1997), increasing their bioavailability.
382 Moreover, for the amphipods increase could have been via biomagnification (Mermillod-
383 Blondin et al., 2004). Our study confirms the increase of TE transport over the sediment
384 water interface by bioturbation (Remaili et al., 2016) and moreover demonstrates how
385 increasing interactions between invertebrates affect TE accumulation depending on species
386 and TE partitioning between colloidal and dissolved.

387 4.3 Body stores

388 In our exposure scenario with environmentally low TE concentrations, body stores of
389 neither amphipods nor mussels were strongly impacted. On the contrary, body stores tended
390 to increase with the number of species coexisting. This is consistent with previous studies
391 showing a low effect of heavy metals on triglycerides and protein in the freshwater
392 gastropod *Potamopyrgus antipodarum* exposed even at higher concentration of Cd (0.8
393 $\mu\text{g.L}^{-1}$) and Zn (50 $\mu\text{g.L}^{-1}$) (Gust et al., 2011) than in our experiment (highest mean
394 concentrations measured: Cd = 0.24 and Zn = 35.2 $\mu\text{g.L}^{-1}$). Moreover, amphipods and
395 mussels are known to be good metal bioaccumulators able to sequester metals by cellular
396 ligands such as metallothioneins, lysosomes, and mineralized organically based concretions
397 without being considerably harmed (Geffard et al., 2010; Langston et al., 1998).

398 The correlation between protein and triglyceride contents with number of species exposed
399 lead to a significant increase of WEBB in amphipods whereas it declined in mussels. This
400 pattern highlights better experimental conditions for amphipods as they benefitted from the
401 presence of tubificid preys (Mermillod-Blondin et al., 2004) and hence increased their
402 energy. Amphipods might also have benefitted from mussels which concentrate the food

403 items (e.g. bacteria) in the surface of the sediment (Vaughn and Hakenkamp, 2001) or
404 produce consumable feces (Gergs and Rothaupt, 2008). In turn, physical interactions
405 between amphipods and mussels might have disturbed the filtration rate of mussels and
406 lead to the observed decrease in their triglyceride content (and the low change in TE
407 concentration). The pattern observed for glycogen in amphipods is more complex with a
408 strong decrease in mesocosms with amphipods and mussels together. This might be due to
409 the increase of certain TE in the water of these mesocosms, especially for the toxic ones
410 such as As, Cd, Cu, and Zn (Borgmann et al., 2005). Glycogen is used by amphipods to
411 cope with harsh environmental conditions and may strongly decrease in presence of toxic
412 compounds (Dehedin et al., 2013; Maazouzi et al., 2011; Marmonier et al., 2013). The
413 physiological state of the organisms in terms of their energetic stores was not that strongly
414 impaired by the TE during the short exposure time (7 days) and the relatively low TE
415 concentrations in the sediment. Nevertheless, some of the TE were negatively correlated
416 with body stores of amphipods (Fe, Ga, almost all REE, Sc, Th, V, and Y) and mussels (B,
417 Sb, Sc and Th). However, further studies are needed to understand the underlying
418 mechanisms and the effects of TE over a longer term.

419

420 **5 Conclusion**

421 Our study confirmed the strong effect of bioturbation of sediments by aquatic invertebrates
422 on the transfer of colloidal and organic matter associated TE to the pelagic compartment,
423 whereas the soluble fraction of TE was not affected. We also highlighted that interspecific
424 interactions appear to be a key driver explaining both the resuspension of TE, depending on
425 their characteristic, and the exposure/contamination of species through another species
426 activity. Previous studies combining several species with different trophic positions focused

427 on the biomagnification of TE along the food web, but our study revealed that a
428 combination of several species would also control indirect contamination via increased
429 bioavailability of TE by bioturbation activities. Moreover, we could not demonstrate high
430 energy allocation in organisms exposed to TE but amphipods however, clearly benefitted
431 from the additional energetic resources provided by tubificids. Further studies are required
432 to measure the intensity of these interactions with more and or other functional feeding
433 groups.

434

435 **Acknowledgments:** This study was supported by Fond Français pour l'Environnement
436 Urbain (FFEM), Municipalidad de Santa Fe, Universidad Nacional del Litoral, Argentina
437 and Université Rennes 1 and 2, France. Moreover, this study was funded by the Selune
438 Project No. 1 053 864 (Seine-Normandy Water Agency) and the LTSER France Zone
439 Atelier Armorique. We thank the plateforme PLAY, ECOBIO Rennes, for the possibility to
440 conduct the bioassay and the body stores analyses. We also thank the reviewers for the time
441 devoted to its evaluation and comments.

442

443 **6 References**

444 Aminot, A., Chaussepied, M., 1983. Manuel des amuyses chimiques en milieu marino,
445 Conseil National pour l' Exploration des Oceans, Univ. de Brest, Francia. 395 p.

446 Berkenbusch, K., Rowden, A.A., 2003. Ecosystem engineering—moving away from 'just-
447 so' stories. *New. Zeal. J. Ecol.* 67-73. <https://www.jstor.org/stable/24058163>

- 448 Blankson, E.R., Klerks, P.L., 2016. The effect of bioturbation by *Lumbriculus variegatus*
449 on transport and distribution of lead in a freshwater microcosm. Environ. Toxicol.
450 Chem. 35(5), 1123-1129. <https://doi.org/10.1002/etc.3248>
- 451 Blankson, E.R., Klerks, P.L., 2017. The effect of sediment characteristics on bioturbation-
452 mediated transfer of lead, in freshwater laboratory microcosms with *Lumbriculus*
453 *variegatus*. Ecotoxicology, 26(2), 227-237. <https://doi.org/10.1007/s10646-016-1757-0>
- 454 Borgmann, U., Couillard, Y., Doyle, P., Dixon, D.G., 2005. Toxicity of sixty-three metals
455 and metalloids to *Hyaella azteca* at two levels of water hardness. Environ. Toxicol.
456 Chem. 24(3), 641-652. <https://doi.org/10.1897/04-177R.1>
- 457 Brix, K.V., DeForest, D.K., Adams, W.J., 2011. The sensitivity of aquatic insects to
458 divalent metals: A comparative analysis of laboratory and field data. Sci. Total
459 Environ. 409(20), 4187-4197. <https://doi.org/10.1016/j.scitotenv.2011.06.061>
- 460 Carignan, J., Hild, P., Mevelle, G., Morel, J., Yeghicheyan, D., 2001. Routine analyses of
461 trace elements in geological samples using flow injection and low pressure on-line
462 liquid chromatography coupled to ICP-MS: a study of geochemical reference materials
463 BR, DR-N, UB-N, AN-G and GH. Geostandard. Newslett. 25(2-3), 187-198.
464 <https://doi.org/10.1111/j.1751-908X.2001.tb00595.x>
- 465 Colombo, V., Pettigrove, V.J., Hoffmann, A.A., Golding, L.A., 2016. Effects of
466 *Lumbriculus variegatus* (Annelida, Oligochaete) bioturbation on zinc sediment
467 chemistry and toxicity to the epi-benthic invertebrate *Chironomus tepperi* (Diptera:

- 468 Chironomidae). *Environ. Pollut.* 216, 198-207.
469 <https://doi.org/10.1016/j.envpol.2016.05.063>
- 470 Davranche, M., Pourret, O., Gruau, G., Dia, A., Le Coz-Bouhnik, M., 2005. Adsorption of
471 REE (III)-humate complexes onto MnO₂: Experimental evidence for cerium anomaly
472 and lanthanide tetrad effect suppression. *Geochim. Cosmochim. Ac.* 69(20), 4825-4835.
473 <https://doi.org/10.1016/j.gca.2005.06.005>
- 474 Davranche, M., Grybos, M., Gruau, G., Pédrot, M., Dia, A., Marsac, R., 2011. Rare earth
475 element patterns: a tool for identifying trace metal sources during wetland soil reduction.
476 *Chem. Geol.* 284(1-2), 127-137. <https://doi.org/10.1016/j.chemgeo.2011.02.014>
- 477 De Jonge, M., Blust, R., Bervoets, L., 2010. The relation between acid volatile sulfides
478 (AVS) and metal accumulation in aquatic invertebrates: implications of feeding behavior
479 and ecology. *Environ. Pollut.* 158(5), 1381-1391.
480 <https://doi.org/10.1016/j.envpol.2010.01.001>
- 481 Dehedin, A., Maazouzi, C., Puijalon, S., Marmonier, P., Piscart, C., 2013. The combined
482 effects of water level reduction and an increase in ammonia concentration on organic
483 matter processing by key freshwater shredders in alluvial wetlands. *Glob. Change.*
484 *Biol.* 19(3), 763-774. <https://doi.org/10.1111/gcb.12084>
- 485 Dolédec, S., Chessel, D., 1994. Co-inertia analysis: an alternative method for studying
486 species–environment relationships. *Freshwater. Biol.* 31(3), 277-294.
487 <https://doi.org/10.1111/j.1365-2427.1994.tb01741.x>

- 488 Dray, S., Chessel, D., Thioulouse, J., 2003. Co-inertia analysis and the linking of
489 ecological data tables. *Ecology*. 84(11), 3078-3089. <https://doi.org/10.1890/03-0178>
- 490 Direction Regionale de l'Environnement, de l'Aménagement et du Logement de Basse-
491 Normandie (DREAL-Normandie), 2012. Demantelement des barrages de vezins et de la
492 roche qui boitsur la selune etat initial de l'etude d'impact. Artelia – Direction Regionale
493 Ouest.
- 494 Eggleton, J., Thomas, K.V. 2004. A review of factors affecting the release and
495 bioavailability of contaminants during sediment disturbance events. *Environ. Int.* 30(7),
496 973-980. <https://doi.org/10.1890/03-0178>
- 497 Foucreau, N., Puijalon, S., Hervant, F., Piscart, C. 2013. Effect of leaf litter characteristics
498 on leaf conditioning and on consumption by *Gammarus pulex*. *Freshwat. Biol.* 58, 1672-
499 1681. <https://doi.org/10.1111/fwb.121580.1>
- 500 Foray, V., Pelisson, P.F., Belvenner, M.C., Desouhant, E., Venner, S., Menu, F., Giron,
501 D., Rey, B., 2012. A handbook for uncovering the complete energetic budget in insects:
502 the van Handel's method (1985) revisited. *Physiol. Entomol.* 37(3), 295-302.
503 <https://doi.org/10.1111/j.1365-3032.2012.00831.x>
- 504 François, F., Gerino, M., Stora, G., Durbec, J.P., Poggiale, J.C., 2002. Functional approach
505 to sediment reworking by gallery-forming macrobenthic organisms: modeling and
506 application with the polychaete *Nereis diversicolor*. *Mar. Ecol. Prog. Ser.* 229, 127-136.
507 <https://doi.org/10.3354/meps229127>

- 508 Fukuhara, H., Yasuda, K., 1989. Ammonium excretion by some freshwater zoobenthos
509 from a eutrophic lake. *Hydrobiologia*. 173(1), 1-8. <https://doi.org/10.1007/BF00008595>
- 510 Gardner, W.S., Nalepa, T.F., Quigley, M.A., Malczyk, J.M., 1981. Release of phosphorus
511 by certain benthic invertebrates. *Can. J. Fish. Aquat. Sci.* 38(8), 978-981.
512 <https://doi.org/10.1139/f81-131>
- 513 Geffard, A., Sartelet, H., Garric, J., Biagianti-Risbourg, S., Delahaut, L., Geffard, O., 2010.
514 Subcellular compartmentalization of cadmium, nickel, and lead in *Gammarus fossarum*:
515 comparison of methods. *Chemosphere*, 78(7), 822-829.
516 <https://doi.org/10.1016/j.chemosphere.2009.11.051>
- 517 Gergs, R., Rothhaupt, K.O., 2008. Feeding rates, assimilation efficiencies and growth of
518 two amphipod species on biodeposited material from zebra mussels. *Freshwater*.
519 *Biol.*, 53(12), 2494-2503. <https://doi.org/10.1111/j.1365-2427.2008.02077.x>
- 520 Gérino, M., Stora, G., Francois, F., Gilbert, F., Poggiale, J.C., Mermillod-Blondin, F.,
521 Desrosiers, G., Vervier, P., 2003. Macro-invertebrate functional groups in freshwater
522 and marine sediments: a common mechanistic classification. *Vie. Milieu*. 53(4), 221-
523 232.
- 524 Graf, G., Rosenberg, R., 1997. Bioresuspension and biodeposition: a review. *Journal of*
525 *Marine Systems*, 11(3-4), 269-278. [https://doi.org/10.1016/S0924-7963\(96\)00126-1](https://doi.org/10.1016/S0924-7963(96)00126-1)
- 526 Gust, M., Buronfosse, T., Geffard, O., Coquery, M., Mons, R., Abbaci, K., Giamberini, L.,
527 Garric, J., 2011. Comprehensive biological effects of a complex field poly-metallic

- 528 pollution gradient on the New Zealand mudsnail *Potamopyrgus antipodarum*
529 (Gray). *Aquat. Toxicol.* 101(1), 100-108. <https://doi.org/10.1016/j.aquatox.2010.09.007>
- 530 He, Y., Men, B., Yang, X., Li, Y., Xu, H., Wang, D., 2017. Investigation of heavy metals
531 release from sediment with bioturbation/bioirrigation. *Chemosphere.* 184, 235-243.
532 <https://doi.org/10.1016/j.chemosphere.2017.05.177>
- 533 He, Y., Men, B., Yang, X., Li, Y., Xu, H., Wang, D. 2019. Relationship between heavy
534 metals and dissolved organic matter released from sediment by
535 bioturbation/bioirrigation. *J. Environ. Sci.* 75, 216-223.
536 <https://doi.org/10.1016/j.jes.2018.03.031>
- 537 Herrington, H.B., 1962. A revision of the Sphaeriidae of North America (Mollusca:
538 Pelecypoda). Museum of Zoology, University of Michigan, No. 118.
- 539 Karickhoff, S.W., Morris, K.R., 1985. Impact of tubificid oligochaetes on pollutant
540 transport in bottom sediments. *Environ. Sci. Technol.* 19(1), 51-56.
541 <https://pubs.acs.org/doi/pdf/10.1021/es00131a004>
- 542 Kiffney, P.M., Clements, W.H., 1994. Effects of heavy metals on a macroinvertebrate
543 assemblage from a Rocky Mountain stream in experimental microcosms. *J. N. Am.*
544 *Benthol. Soc.* 13(4), 511-523. <https://doi.org/10.2307/1467847>
- 545 Langston, W.J., Bebianno, M.J., Burt, G.R., 1998. Metal handling strategies in molluscs.
546 In *Metal metabolism in aquatic environments*. Springer, Boston, MA. pp. 219-283.
547 https://doi.org/10.1007/978-1-4757-2761-6_8

- 548 Lawrence, A.L., Mason, R.P., 2001. Factors controlling the bioaccumulation of mercury
549 and methylmercury by the estuarine amphipod *Leptocheirus plumulosus*. Environ.
550 Pollut. 111(2), 217-231. [https://doi.org/10.1016/S0269-7491\(00\)00072-5](https://doi.org/10.1016/S0269-7491(00)00072-5)
- 551 Laveuf, C., Cornu, S., Guilherme, L.R.G., Guerin, A., Juillot, F., 2012. The impact of redox
552 conditions on the rare earth element signature of redoximorphic features in a soil
553 sequence developed from limestone. Geoderma. 170, 25-38.
554 <https://doi.org/10.1016/j.geoderma.2011.10.014>
- 555 Maazouzi, C., Piscart, C., Legier, F., Hervant, F., 2011. Ecophysiological responses to
556 temperature of the “killer shrimp” *Dikerogammarus villosus*: is the invader really
557 stronger than the native *Gammarus pulex*? Comp. Biochem. Phys. A. 159(3), 268-274.
558 <https://doi.org/10.1016/j.cbpa.2011.03.019>
- 559 Marmonier, P., Maazouzi, C., Foulquier, A., Navel, S., François, C., Hervant, F.,
560 Mermillod-Blondin, F., Vieney, A., Barraud, S., Togola, A., Piscart, C., 2013. The use
561 of crustaceans as sentinel organisms to evaluate groundwater ecological quality. Ecol.
562 Eng. 57, 118-132. <https://doi.org/10.1016/j.ecoleng.2013.04.009>
- 563 Marsac, R., Davranche, M., Gruau, G., Dia, A., 2010. Metal loading effect on rare earth
564 element binding to humic acid: experimental and modelling evidence. Geochim.
565 Cosmochim. Ac., 74(6), 1749-1761. <https://doi.org/10.1016/j.gca.2009.12.006>
- 566 Marsden, I.D., Rainbow, P.S., 2004. Does the accumulation of trace metals in crustaceans
567 affect their ecology—the amphipod example? J Exp. Mar. Biol. Ecol. 300(1-2), 373-408.
568 <https://doi.org/10.1016/j.jembe.2003.12.009>

- 569 McAuliffe, J.R., 1984. Competition for space, disturbance, and the structure of a benthic
570 stream community. *Ecology*, 65(3), 894-908. <https://doi.org/10.2307/1938063>
- 571 Mermillod-Blondin, F., Gaudet, J.P., Gerino, M., Desrosiers, G., Jose, J., Chatelliers,
572 M.C.D., 2004. Relative influence of bioturbation and predation on organic matter
573 processing in river sediments: a microcosm experiment. *Freshwater. Biol.*, 49(7), 895-
574 912. <https://doi.org/10.1111/j.1365-2427.2004.01233.x>
- 575 Mermillod-Blondin, F., Nogaro, G., Gibert, J., 2007. Colmatage des bassins d'infiltration
576 par des sédiments urbains: influence de la bioturbation par les invertébrés. *Novatech*
577 2007.
- 578 Mermillod-Blondin, F., Nogaro, G., Vallier, F., Gibert, J., 2008. Laboratory study
579 highlights the key influences of stormwater sediment thickness and bioturbation by
580 tubificid worms on dynamics of nutrients and pollutants in stormwater retention
581 systems. *Chemosphere*. 72(2), 213-223.
582 <https://doi.org/10.1016/j.chemosphere.2008.01.052>
- 583 Mermillod-Blondin, F., Rosenberg, R., 2006. Ecosystem engineering: the impact of
584 bioturbation on biogeochemical processes in marine and freshwater benthic
585 habitats. *Aquat. Sci.* 68(4), 434-442. <https://doi.org/10.1007/s00027-006-0858-x>
- 586 Milbrink, G., 1987. Biological characterization of sediments by standardized tubificid
587 bioassays, in: *Aquatic Oligochaeta*. Springer, Dordrecht, pp. 267-275.
588 https://doi.org/10.1007/978-94-009-3091-9_34

- 589 Morley, G.F., Gadd, G.M., 1995. Sorption of toxic metals by fungi and clay
590 minerals. *Mycol. Res.* 99(12), 1429-1438. [https://doi.org/10.1016/S0953-](https://doi.org/10.1016/S0953-7562(09)80789-2)
591 [7562\(09\)80789-2](https://doi.org/10.1016/S0953-7562(09)80789-2)
- 592 Nalepa, T.F., Gardner, W.S., Malczyk, J.M., 1983. Phosphorus release by three kinds of
593 benthic invertebrates: effects of substrate and water medium. *Can. J. Fish. Aquat.*
594 *Sci.*, 40(6), 810-813. <https://doi.org/10.1139/f83-105>
- 595 Pearson, T.H., 2001. Functional group ecology in soft-sediment marine benthos: the role of
596 bioturbation, in: *Oceanography and Marine Biology: An Annual Review*. Aberdeen
597 University Press/Allen Unwin, London. ISSN 0078-3218, 39, pp. 233-267.
- 598 Pédrot, M., Dia, A., Davranche, M., Bouhnik-Le Coz, M., Henin, O., Gruau, G., 2008.
599 Insights into colloid-mediated trace element release at the soil/water interface. *J.*
600 *Colloid. Interf. Sci.* 325(1), 187-197. <https://doi.org/10.1016/j.jcis.2008.05.019>
- 601 Pédrot, M., Dia, A., Davranche, M., 2009. Double pH control on humic substance-borne
602 trace elements distribution in soil waters as inferred from ultrafiltration. *J. Colloid.*
603 *Interf. Sci.* 339(2), 390-403. <https://doi.org/10.1016/j.jcis.2009.07.046>
- 604 Pédrot, M., Dia, A., Davranche, M., 2010. Dynamic structure of humic substances: Rare
605 earth elements as a fingerprint. *J. Colloid. Interf. Sci.* 345(2), 206-213.
606 <https://doi.org/10.1016/j.jcis.2010.01.069>
- 607 Pédrot, M., Dia, A., Davranche, M., Gruau, G., 2015. Upper soil horizons control the rare
608 earth element patterns in shallow groundwater. *Geoderma.* 239, 84-96.
609 <https://doi.org/10.1016/j.geoderma.2014.09.023>

- 610 Piscart, C., Manach, A., Copp, G.H., Marmonier, P., 2007. Distribution and microhabitats
611 of native and non-native gammarids (Amphipoda, Crustacea) in Brittany, with
612 particular reference to the endangered endemic sub-species *Gammarus duebeni*
613 *celticus*. J. Biogeogr. 34(3), 524-533. <https://doi.org/10.1111/j.1365-2699.2006.01609.x>
- 614 Piscart, C., Genoel, R., Doledec, S., Chauvet, E., Marmonier, P., 2009. Effects of intense
615 agricultural practices on heterotrophic processes in streams. Environ. Pollut., 157(3),
616 1011-1018. <https://doi.org/10.1016/j.envpol.2008.10.010>
- 617 Piscart, C., Roussel, J.M., Dick, J.T., Grosbois, G., Marmonier, P., 2011. Effects of
618 coexistence on habitat use and trophic ecology of interacting native and invasive
619 amphipods. Freshwater Biol. 56(2), 325-334. <https://doi.org/10.1111/j.1365-2427.2010.02500.x>
- 620
- 621 Pokrovsky, O.S., Schott, J., 2002. Iron colloids/organic matter associated transport of major
622 and trace elements in small boreal rivers and their estuaries (NW Russia). Chem.
623 Geol. 190(1-4), 141-179. [https://doi.org/10.1016/S0009-2541\(02\)00115-8](https://doi.org/10.1016/S0009-2541(02)00115-8)
- 624 R Core Team, 2013. R: A language and environment for statistical computing.
625 <ftp://ftp.uvigo.es/CRAN/web/packages/dplR/vignettes/intro-dplR.pdf>
- 626 Reible, D.D., Popov, V., Valsaraj, K.T., Thibodeaux, L.J., Lin, F., Dikshit, M., Todaro,
627 M.A., Fleeger, J. W., 1996. Contaminant fluxes from sediment due to tubificid
628 oligochaete bioturbation. Water Res. 30(3), 704-714. [https://doi.org/10.1016/0043-1354\(95\)00187-5](https://doi.org/10.1016/0043-1354(95)00187-5)
- 629

- 630 Remaili, T.M., Simpson, S.L., Amato, E.D., Spadaro, D.A., Jarolimek, C.V., Jolley, D.F.,
631 2016. The impact of sediment bioturbation by secondary organisms on metal
632 bioavailability, bioaccumulation and toxicity to target organisms in benthic bioassays:
633 implications for sediment quality assessment. *Environ. Pollut.* 208, 590-599.
634 <https://doi.org/10.1016/j.envpol.2015.10.033>
- 635 Rosenberg, D.M., Resh, V.H., 1993. Introduction to freshwater biomonitoring and benthic
636 macroinvertebrates, in Rosenberg, D. M., Resh, V. H. (Eds.), *Freshwater bio-*
637 *monitoring and benthic macroinvertebrates*. Chapman and Hall, London, pp. 1-9.
- 638 Schaller, J., Dharamshi, J., Dudel, E.G., 2011. Enhanced metal and metalloid
639 concentrations in the gut system comparing to remaining tissues of *Gammarus pulex*
640 L. *Chemosphere*. 83(4), 627-631. <https://doi.org/10.1016/j.chemosphere.2010.11.063>
- 641 Sholkovitz, E.R., 1995. The aquatic chemistry of rare earth elements in rivers and
642 estuaries. *Aquat. Geochem.* 1(1), 1-34. <https://doi.org/10.1007/BF01025229>
- 643 Smolders, R., De Boeck, G., Blust, R., 2003. Changes in cellular energy budget as a
644 measure of whole effluent toxicity in zebrafish (*Danio rerio*). *Environ. Toxicol.*
645 *Chem.* 22(4), 890-899. <https://doi.org/10.1002/etc.5620220429>
- 646 StatSoft, 2004. *Statistica*. StatSoft Inc. (www.statsoftinc.com).
- 647 Stumm, W., Morgan, J.J., 1996. *Aquatic Chemistry An Introduction Emphasizing*
648 *Chemical Equilibria in Natural Waters*. Schnoor, J.L., Zehnder, A. (Eds.), 3rd ed. John
649 Wiley Intersciences, New York.

- 650 Tang, J., Johannesson, K.H., 2010. Ligand extraction of rare earth elements from aquifer
651 sediments: implications for rare earth element complexation with organic matter in
652 natural waters. *Geochim. Cosmochim. Ac.* 74(23), 6690-6705.
653 <https://doi.org/10.1016/j.gca.2010.08.028>
- 654 van der Meer, T.V., de Baat, M.L., Verdonschot, P.F., Kraak, M.H., 2017. Benthic
655 invertebrate bioturbation activity determines species specific sensitivity to sediment
656 contamination. *Front. Environ. Sci.* 5, 83. <https://doi.org/10.3389/fenvs.2017.00083>
- 657 Vaughn, C.C., Hakenkamp, C.C., 2001. The functional role of burrowing bivalves in
658 freshwater ecosystems. *Freshwater Biol.* 46(11), 1431-1446.
659 <https://doi.org/10.1046/j.1365-2427.2001.00771.x>
- 660 Xie, M., Wang, N., Gaillard, J.F., Packman, A.I., 2018. Interplay between flow and
661 bioturbation enhances metal efflux from low-permeability sediments. *J Hazard.*
662 *Mater.* 341, 304-312. <https://doi.org/10.1016/j.jhazmat.2017.08.002>
- 663 Yéghicheyan, D., Bossy, C., Bouhnik Le Coz, M., Douchet, C., Granier, G., Heimbürger,
664 A., Lacan, F., Lanzanova, A., Rousseau, T., Seidel, J., Tharaud, M., Candaudap, F.,
665 Chmeleff, J., Cloquet, C., Delpoux, S., Labatut, M., Losno, R., Pradoux, C., Sivry, Y.,
666 Sonke, J., 2013. A compilation of silicon, rare earth element and twenty-one other trace
667 element concentrations in the natural river water reference material SLRS-5 (NRC
668 CNRC). *Geostand. Geoanal. Res.* 37(4), 449-467. [https://doi.org/10.1111/j.1751-](https://doi.org/10.1111/j.1751-908X.2013.00232.x)
669 [908X.2013.00232.x](https://doi.org/10.1111/j.1751-908X.2013.00232.x)

- 670 Zeybek, M., Kalyoncu, H., Ertan, Ö.O., 2012. Species composition and distribution of
671 Mollusca in relation to water quality. Turkish Journal of Fisheries and Aquat. Sci. 12(3),
672 719-727. https://doi.org/10.4194/1303-2712-v12_3_21

Journal Pre-proof

673 **Figure legends**

674 **Figure 1.** Mean (\pm SD) concentrations of nitrate (NO₃), phosphate (PO₄) and ammonium
675 (NH₄) in water according to the five experimental conditions. Significant differences are
676 indicated by letters and numbers.

677

678 **Figure 2.** PCA on trace elements in water. (a) Correlation circle. (b) Eigenvalues. (c)
679 Distribution of experimental conditions on the PC1–PC2 axes. Mean position of each
680 experimental condition (white circles) are positioned at the weighted average of the five
681 replicates (black dots).

682

683 **Figure 3.** Mean (\pm SE) of selected TE concentrations in water according to the five
684 experimental conditions: (a) total content in Rare Earth Elements (REE), (b) iron
685 concentration, (c) Sum of Cr, U and Pb, (d) total content in highly Soluble Trace Element
686 (STE) including (Li, B, Ba, Ca, K, Mg, Mn, and Sr) , and (e) Sodium on its own.

687

688 **Figure 4.** Projection of two first axes of co-inertia analysis (COIA) after a double PCA: (a)
689 nutrients concentrations, (b) TE in water (REE are shown in the box), and (c) samples
690 scores by modalities and the arrows showing the change from one view to the other,
691 indicating the strength of the links.

692

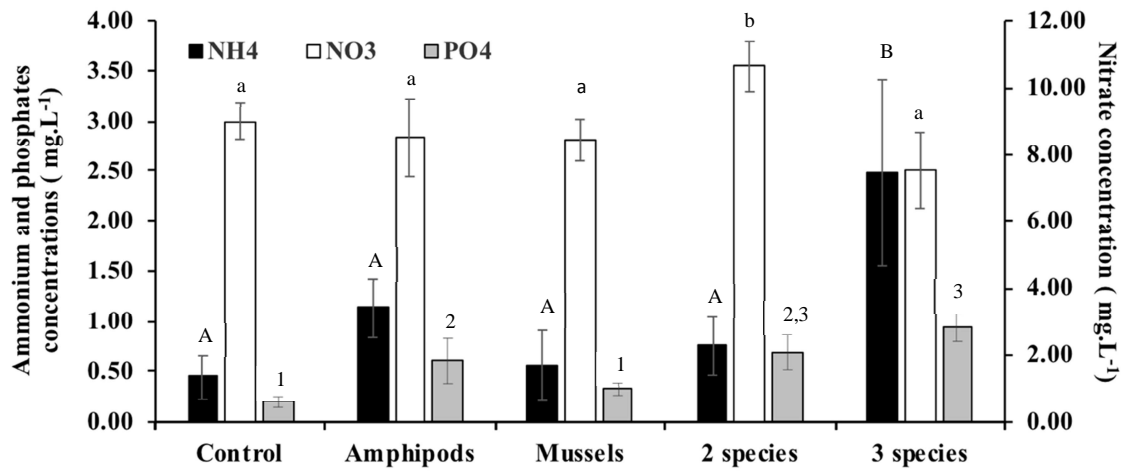
693 **Figure 5.** PCA on trace elements in animals: (a) Correlation circle, (b) Eigenvalues, (c)
694 Distribution of experimental conditions on the PC1–PC2 axes. Mean position of each

695 experimental condition (white circles) are positioned at the weighted average of the five
696 replicates (black dots).

697

698 **Figure 6.** Mean (\pm SE) content in (a) proteins, (b) triglycerides, (c) glycogen, (d) mean (\pm
699 SD) values for Whole Body Energy Budget (WBEB) according to experimental conditions
700 for amphipods (dark bars) and mussels (white bars). Significant differences are indicated by
701 letters.

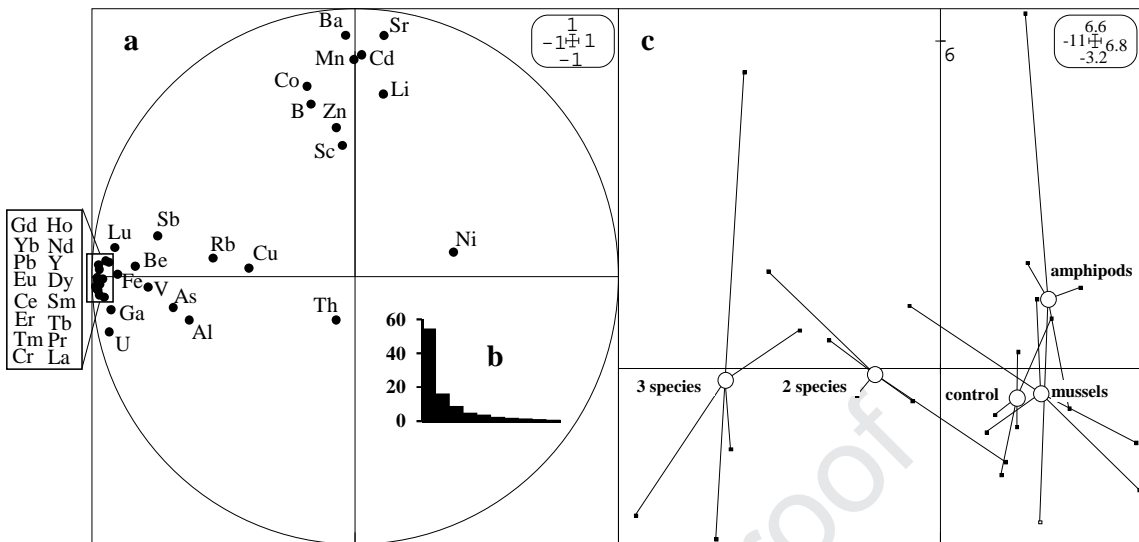
702



703

704 **Figure 1.**

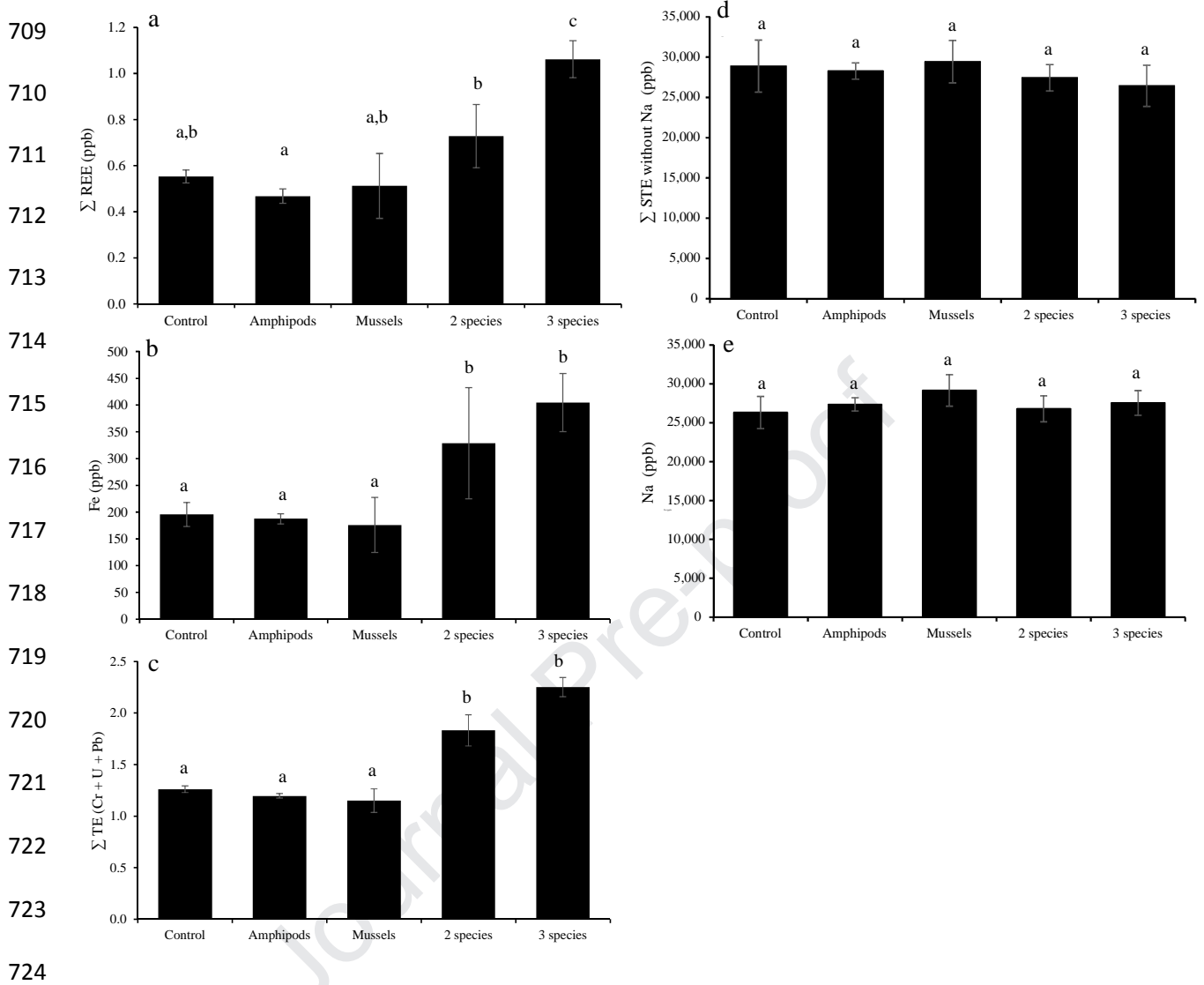
705

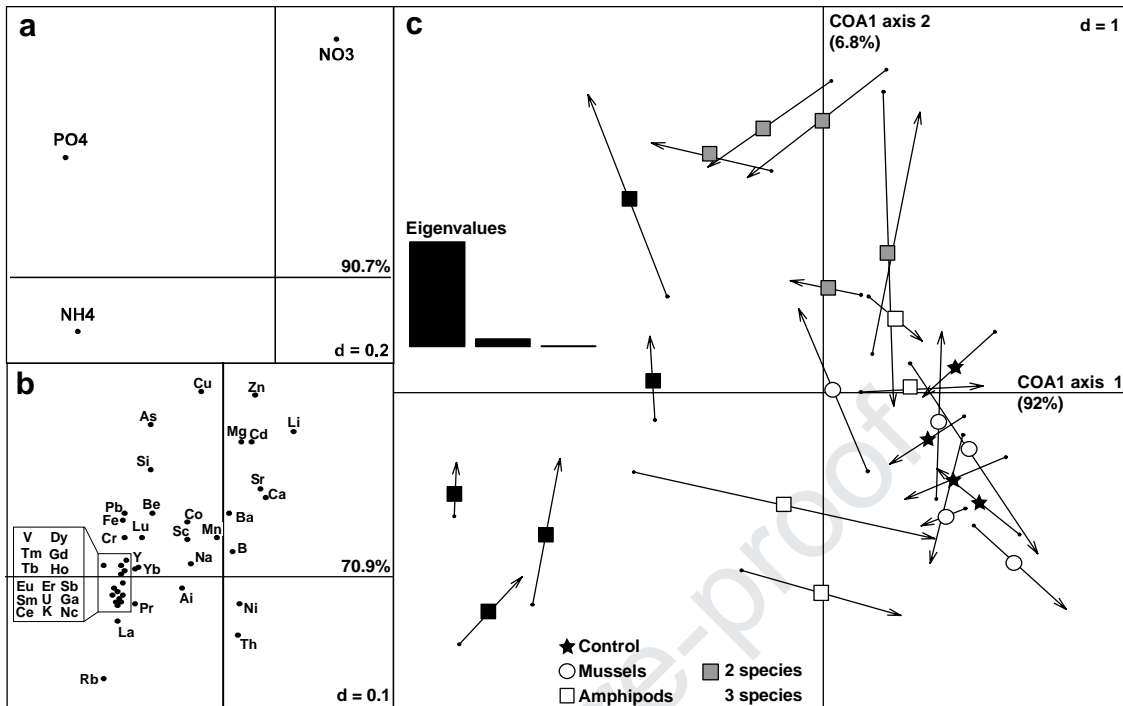


706

707 **Figure 2.**

708

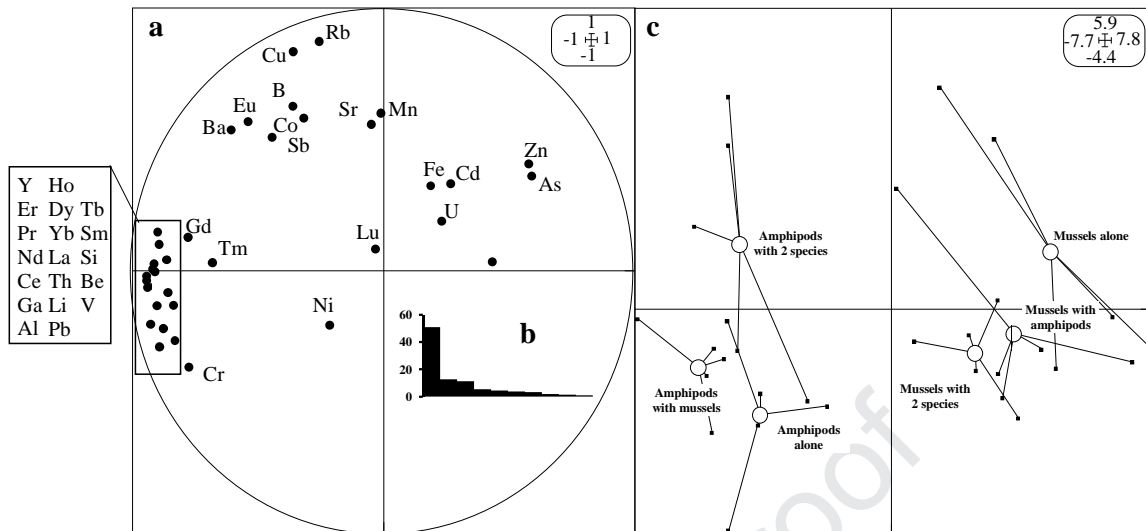
**Figure 3.**



727

728 **Figure 4.**

729



730

731 **Figure 5.**

732

733

734

735

736

737

738

739

740

741

742

743

744

745

746

747

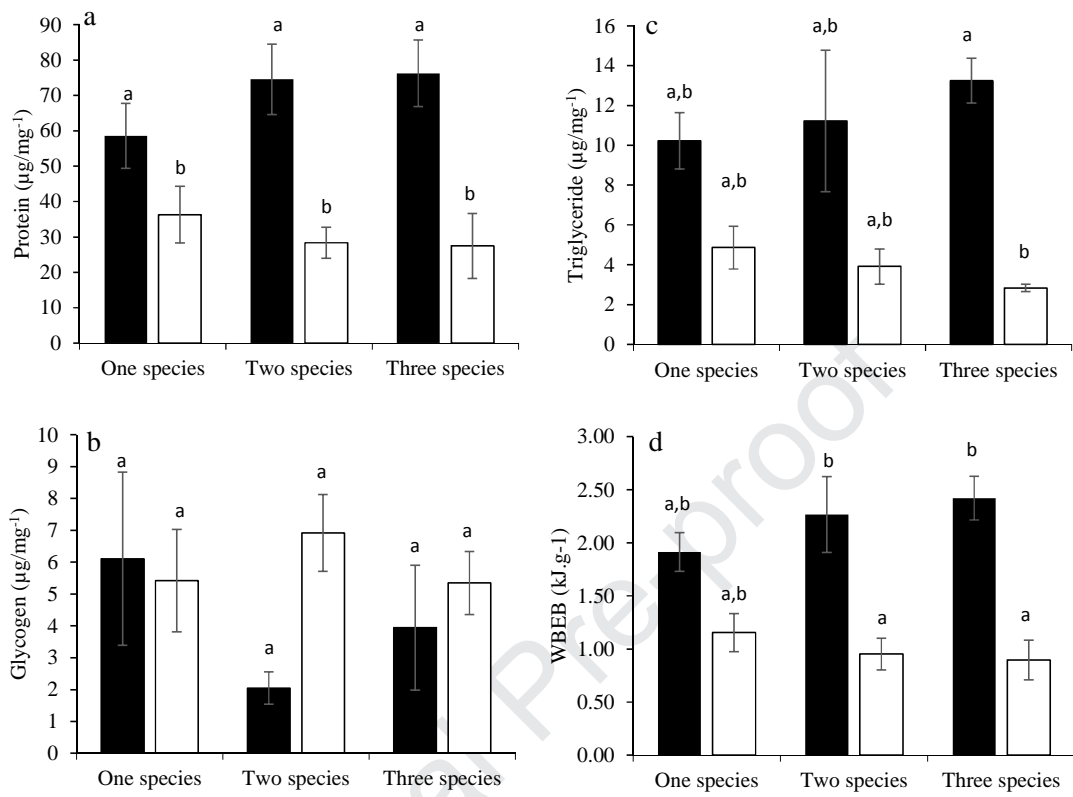


Figure 6.

1 Highlights

- 2 - Macroinvertebrates bioturbation affects trace elements fate among sediment and water
- 3 - Trace elements bioturbation depends on their solubility or colloidal association
- 4 - Trace elements bioturbation depends on macroinvertebrates biological traits
- 5 - Interspecific interactions are key drivers for trace elements bioaccumulation
- 6 - Bioturbators increase trace elements bioavailability for coexisting specie

Declaration of interests

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: