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## **Gain in biodiversity but not in phytostabilization after 3 years of ecological restoration of contaminated Mediterranean soils**

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1 **Gain in biodiversity but not in phytostabilization after 3 years of ecological restoration**  
2 **of contaminated Mediterranean soils**

3  
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15 **Highlights**

16 - Metal and metalloid soil contamination slows down the ecological restoration dynamics

17 - Native plants transplanted in Mediterranean contaminated soils persist after 3 years

18 - Active ecological restoration allows the creation of favorable micro-niches for plants

19 - Metal and metalloid phytostabilization efficiency is not evidenced after 3 years

20

21 **Abstract**

22 Recent studies consider the native flora as a potential source of plant candidates for  
23 phytostabilization of metal and metalloid (MM) contaminated soils, but ecological restoration  
24 is not the main objective of these researches. However, in contaminated areas,  
25 phytostabilization should be considered as a useful tool for ecological restoration. The present  
26 study takes stock of 3 years of a Mediterranean pilot site implementation using native plant  
27 species to recover plant and microbial communities (diversity and functions) together with  
28 soil remediation in the Calanques National Park. To determine the success of this operation,  
29 three *in situ* treatments were compared: ecological restoration plots characterized by the  
30 handling of physical environment (creation of cultivation terraces) and plants, negative  
31 control plots without vegetation but with the same physical environment handling as  
32 ecological restoration plots, and positive control plots with natural vegetation and no

33 handling. The results suggest that an ecological restoration trajectory is initiated in the  
34 ecological restoration plots, characterized by a partial permanent plant cover. However, there  
35 is no evidence of a significant improvement of soil quality (evaluated by soil texture, pH,  
36 nutrients and organic carbon contents, cation exchange capacity, microbial biomass and  
37 activities) and phytostabilization efficiency after 3 years. Native plant communities and their  
38 associated microorganisms may need more time before improving soil quality and MM  
39 stabilization under the drastic Mediterranean conditions. Any amendment addition to  
40 accelerate restoration and MM immobilization was forbidden in this protected area. Under  
41 such conditions, an active restoration need to be carried out in this contaminated area even if  
42 resilient dynamics of the native plant communities may sporadically occur over a long period  
43 of time.

44

45 **Keywords:** active restoration, field experiment, Mediterranean protected area, metals and  
46 metalloids, native plant communities, resilient dynamics

47

## 48 **1. Introduction**

49 Human activities generate a significant degradation of ecosystems and specifically of soils.  
50 Indeed, in 2015, 33 % of soils were considered as moderately or severely degraded by  
51 erosion, nutrient depletion, acidification, salinization, compaction and chemical pollution  
52 (FAO, 2015). The European Strategy for Biodiversity expects at least 15% of degraded  
53 ecosystems to be restored by 2020 (EU, 2011). In this context, some scientists are currently  
54 working on ecological restoration of polluted environments (Wong, 2003; Li, 2006)  
55 particularly in the Mediterranean region (Heckenroth et al., 2016b). Ecological restoration is  
56 the intentional activity that assists ecosystem restoration in order to regain its structural and  
57 functional integrity, allowing it to be resistant and/or resilient to normal stress and  
58 environmental disturbance levels (SER, 2004). Ecological restoration can be active or passive  
59 depending on the level of ecosystem degradation (SER, 2016). Facing weak degradation,  
60 stopping the disturbance is often enough for spontaneous plant recolonization of the site (i.e.  
61 spontaneous regeneration through passive ecological restoration). When degradation is  
62 significant, active restoration operations are required by restoring abiotic components (i.e.  
63 assisted regeneration) or both abiotic and biotic components (i.e. reconstruction). In the case  
64 of contaminated soils, ecological restoration can be combined with phytoremediation  
65 techniques to decontaminate the environment or to avoid pollutants dispersion (Wong, 2003;  
66 Li, 2006). Phytostabilization consists of using plants and their associated microorganisms to

67 stabilize contaminants in the roots and/or rhizosphere to limit their transfer *via* erosion,  
68 leaching or runoff processes and their bioavailability (Pilon-Smits, 2005) and to promote  
69 restoration of contaminated ecosystem especially by metals and metalloids (MM) (Ma et al.,  
70 2011; Marchiol et al., 2013). The success of an ecological restoration combined with  
71 phytostabilization is based on several criteria. Plant species selection from the local floristic  
72 background is essential. The selection must consider plant adaptations to local conditions and  
73 their MM tolerance which promote plant survival and development (Baker et al., 2010; Bolan  
74 et al., 2011; Cortina et al., 2011). Moreover, permanent and large root systems and plant  
75 cover, as well as plant MM immobilization capacities, permit to reduce soil erosion and MM  
76 bioavailability, thus limiting contaminant dispersion and transfer into the environment (Pilon-  
77 Smits, 2005; Bochet et al., 2006; Bolan et al., 2011). In addition, consideration of biotic  
78 interactions is essential, in particular in harsh environments (Padilla and Pugnaire, 2006;  
79 Cortina, 2011). Facilitation by nurse plants, favoring micro-niche formation by improving  
80 microclimatic and edaphic conditions (Callaway, 2007), permits establishment of other plant  
81 species on unfavorable environments (Frérot et al., 2006). Plant associated microorganisms,  
82 besides potentially participate in MM immobilization, improve plant MM tolerance by  
83 enhancing nutrient and water absorption (Khan, 2005; Ma et al., 2011). Better consideration  
84 of soil quality, its contamination level and the contamination spatial heterogeneity are  
85 necessary to think how to improve soil properties involved in contaminant retention in order  
86 to reduce their bioavailability (Adriano et al., 2004; Bolan et al., 2011; Kumar Yadav et al.,  
87 2018). However, the use of ecological restoration tools combined with those of  
88 phytostabilization is still very recent and these success criteria are still to be assessed.  
89 Moreover, ecological restoration in Mediterranean dry habitats is a challenging topic and  
90 implies inventiveness in the developed processes to enhance vegetation establishment  
91 (Jaunatre et al., 2012; Rey and Burylo, 2014; Ballesteros et al., 2017). However, in addition to  
92 biogeoclimatic constraints framing the tools used, accessibility, pollution, topographical and  
93 regulatory constraints may restrict possible actions.

94  
95 Mining areas are relevant study cases of such approaches and literature on MM  
96 phytoremediation by using autochthonous plant species is becoming more important this last  
97 decade (Ilunga wa Ilunga et al., 2015; Marrugo-Negrete et al., 2015; Ghazaryan et al., 2019).  
98 However, fewer restoration trials are conducted in brownfields generating diffuse pollution of  
99 soils. One of those brownfields, the Escalette site, is located on the Mediterranean coast of  
100 Marseille (south-eastern France) at the outskirts of the city center. This site sheltered a lead-

101 smelting factory that operated from 1851 to 1925 using galena imported from Mediterranean  
102 countries (Daumalin and Raveux, 2016). This activity contributed to a diffuse MM  
103 contamination of surface soils of the surrounding area by slag deposits and by dispersion of  
104 contaminated ashes from the factory horizontal chimney *via* erosion and bioalteration  
105 processes (Testiati et al., 2013; Heckenroth et al., 2016a; Laffont-Schwob et al., 2016). The  
106 brownfield is now included in the protected area of the Calanques National Park (PNCal), the  
107 first French peri-urban National Park founded in 2012. This territory gathers several socio-  
108 environmental issues (environmental health, biodiversity conservation and management,  
109 regulatory issues) linked with anthropogenic disturbances (industrial and urban pollution, soil  
110 erosion, over-frequentation), affecting this urban and natural protected and biodiversity-rich  
111 area (Affre et al., 2015). At the Escalette site, the natural resilience capacity of the original  
112 ecosystem has been strongly affected by MM soil contamination, which is currently reflected  
113 by the presence of unvegetated scree which can constitute a source of contamination transfer  
114 by water erosion and/or wind (Heckenroth et al., 2016b). In November 2015, a joint active  
115 ecological restoration and phytostabilization trial has been implemented in this site  
116 (Heckenroth et al., 2016a). This pilot-scale experiment aims at stabilizing MM soil  
117 contamination sustainably by favoring a permanent native vegetation cover to limit its  
118 transfer, while considering restoration of plant and soil biodiversity and functionalities. Three  
119 years after the implementation of this experiment, two main questions remain: (i) At which  
120 step along an ecological restoration trajectory correspond the implemented plant  
121 communities? (ii) Do they ensure effective stabilization of MM soil contamination? To  
122 answer these questions, a comparative *in situ* monitoring of soil properties, MM  
123 concentrations in soils and in runoff water and characteristics of plant communities was  
124 performed. These parameters were monitored over three *in situ* treatments: ecological  
125 restoration plots characterized by the handling of physical environment variables (creation of  
126 cultivation terraces) and plants, negative control plots without vegetation but with a physical  
127 environment variable handling (terraces) such as ecological restoration plots, and positive  
128 control plots with natural vegetation and nor abiotic neither biotic variables handling.

129

## 130 **2. Material and methods**

131

### 132 **2.1. Study site**

133 The study site is located in the Marseilleveyre calcareous massif, characterized by significant  
134 karstic erosion and skeletal oligotrophic soils (Knoerr, 1959). The study site has a semi-arid

135 Mediterranean climate, characterized by hot and dry summers, mild and wet winters (Quézel  
136 and Médail, 2003) and by a prevailing cold, dry and often violent wind, called Mistral  
137 (Aillaud and Crouzet, 1988), accentuating the environment's aridity (average annual rainfall  
138 of 500 mm, average annual temperature of 16 °C). The study site is located at the Escalette  
139 industrial brownfield, at the outfall of the horizontal chimney of a former lead smelting  
140 factory, in the protected area of the PNCal (WGS84 co-ordinates: 43.22578°N, 5.35059°E,  
141 Fig. A). The site corresponds to a steep scree with an average slope of 23 %, located down  
142 below a ridge at 140 m of altitude. This site is defined as a hotspot of contamination mainly  
143 by lead (Pb) and arsenic (As) but also by copper (Cu) and zinc (Zn) (Testiati et al., 2013). The  
144 soil, whose depth varies between 10 and 30 cm, corresponds to a mixture of limestone, clayey  
145 limestone, marl (Knoerr, 1959) and contaminated ashes. A layer of ashes from a few to  
146 several centimeters covers the soil at the outfall of the horizontal chimney, which can then be  
147 defined as anthroposol.

148 In November 2015, an active ecological restoration has been carried out at the Escalette site  
149 by environmental handling and planting. This operation is in accordance with the PNCal's  
150 regulations, which prohibit the introduction of non-local plant material and the use of  
151 amendments to conserve local biodiversity. 20 plots of 1 m<sup>2</sup> were created on the unvegetated  
152 steep scree. These plots correspond to cultivation terraces made up of two strips, one  
153 upstream and one downstream. Limestones present on the site were removed from the soil  
154 surface and used to shape the terraces by installing a row of stones between upstream strip  
155 and downstream strip of cultivation terraces. Limited soil handling was targeted. Eight plant  
156 species from the local floristic background (constituted of calcareous xero-thermophilous  
157 shrubland: Mediterranean scrubland, grasslands and stands of Aleppo pine) and defined as  
158 pseudo-metallophytes according to the definitions given by Pauwels et al. (2006) and Faucon  
159 et al. (2009) were transplanted in each plot: *Brachypodium retusum* (Pers.) P.Beauv., 1812,  
160 *Cistus albidus* L., 1753, *Coronilla juncea* L., 1753, *Dianthus caryophyllus* L., 1753,  
161 *Globularia alypum* L., 1753, *Lobularia maritima* (L.) Desv., 1815, *Rosmarinus officinalis* L.,  
162 1753 and *Teucrium polium* L., 1753 (Heckenroth et al., 2017, Fig. B). Soil surface was  
163 covered with limestones to limit water stress. A unique watering was done to the plant's earth  
164 clod at the transplantation. Nor amendment adding neither watering was carried out after  
165 transplantation.

166

## 167 **2.2. Experimental design**

168 Three treatments (E: ecological restoration plots, Cn: negative control plots, Cp: positive  
169 control plots), each consisting of 5 replicates, were tested *in situ* (Fig. 1). E and Cn plots have  
170 been located on the unvegetated scree at the beginning of the study and correspond to active  
171 restoration with different levels of intervention. E plots are characterized by environmental  
172 and plant handling (i.e. creation of cultivation terraces and planting, respectively). E plots  
173 correspond to 5 plots selected from those implemented in November 2015 and contain planted  
174 and spontaneous plant species. These E plots were selected for their well-established  
175 perennial vegetation and the persistence of planted *C. juncea* individuals, assuming that their  
176 highly developed root systems act on soil contamination fixation. Cn plots are characterized  
177 by the same environmental handling as E plots, i.e. 1 m<sup>2</sup>-cultivation terraces located on the  
178 unvegetated scree, but without planting. E and Cn plots were equipped with runoff water  
179 collectors in November 2018. Runoff water collectors consist of a plastic chute placed at the  
180 level of the downstream band of the plots, closed at the end and connected to a can of water  
181 recovery by a funnel. A plastic tarpaulin was installed at ground level, under the stones of the  
182 downstream band of the terraces and, attached to the chute. Another tarpaulin was placed over  
183 the chute to recover only runoff water and not rainwater. This installed system can be easily  
184 dismantled as required by the PNCal regulation. Cp plots correspond to the reference plant  
185 community of this ecological restoration operation, i.e. a natural vegetation of young  
186 Mediterranean scrubland that has sporadically persisted during past industrial activities until  
187 today. No ecological restoration work was carried out on these plots, they have only been  
188 spatially delimited by limestones. Thus, Cp plots have not been structured in cultivation  
189 terraces or equipped with runoff water collectors to avoid disturbance and degradation due to  
190 abiotic handling. Cp are located close to E and Cn plots to maintain comparable soil  
191 contamination levels between the active restoration plots (E and Cn) and reference plots (Cp).  
192 This allows a realistic ecological restoration objective to be defined on MM contaminated  
193 soil.

194

### 195 **2.3. Soil analysis**

196 Soil samples were collected in 4 points at each plot corner in the top 15 cm-layer and pooled  
197 (mix of an equal weight of the 4 subsamples on each plot). The obtained composite soil  
198 samples for each E, Cp, and Cn plots were sieved at 2 mm and analyses of agronomic  
199 characteristics, MM contamination and microbial properties were performed.

200

#### 201 **2.3.1. Soil agronomic characteristics**

202 Soil agronomic analyses were carried out on 40 °C dried soil. pH was determined with the  
203 Orion 2 Star Thermo Scientific SM30B device, in a solution of soil and distilled water (1/5,  
204 V/V, NF ISO10390, 2005). Total nitrogen content (TKN) was estimated by the Kjeldahl  
205 method (NF ISO 11261, 1995). Total organic carbon (TOC) was determined by the difference  
206 between total carbon content and inorganic carbon content (NF ISO 10694, 1995).  
207 Granulometry and texture (NF X 31-107, 2003), available phosphorus (P, NF ISO 11263,  
208 1995) and cation exchange capacity (CEC, NF X 31-130, 1999) measurements were  
209 performed in the Laboratoire Développement Méditerranée (Alès, France).

210

### 211 **2.3.2. Soil contamination**

212 Soil samples were dried at 40 °C and ground at 0.2 mm (RETSCH zm 1000 with tungsten  
213 blades and titanium sieve) in order to evaluate soil contamination level and MM mobilization  
214 in soil. In order to determine pseudo-total MM concentrations ( $MM_{tot}$ ), samples were digested  
215 in a microwave mineralizer (Milestone Start D) using aqua regia (1/3  $HNO_3$  + 2/3 HCl), then  
216 filtered through a 0.45  $\mu m$  cellulose ester membrane filter (NF ISO 11466, 1995). For mobile  
217 MM concentrations ( $MM_{mob}$ ), 0.05 M EDTA (pH  $7.0 \pm 0.1$ ) solution was used as extractant  
218 following the method approved by the Community Bureau of Reference (CBR)  
219 (Quevauviller, 1998). A ratio of soil/EDTA solution corresponding to 1/10 w/V was used.  
220 The solution was stirred at room temperature for 1 h, then centrifuged during 10 min at  
221 8,000 rpm and filtered at 0.45  $\mu m$ .  $MM_{tot}$  and  $MM_{mob}$  concentrations were determined by ICP-  
222 AES (Jobin Yvon Horiba, Spectra 2000) for As, Cu, Pb and Zn. Quality controls and accuracy  
223 were checked using standard soil reference materials (CRM 049-050, from RTC-USA), with  
224 accuracies within  $100 \pm 10$  %. Contamination factors (CF) were calculated according to the  
225 formula:

226

227 Eq. 1.  $CF_{MM} = [MM]_{tot} / [MM]_{local\ background\ value}$ , where  $[MM]_{local\ background\ value}$  are 4.9, 7.5, 42.9,  
228 3.1 and 66  $mg.kg^{-1}$  for As, Cu, Pb, Sb and Zn, respectively, as previously reported with the  
229 same method used in the present study (Affholder et al., 2014).

230

231 Then, the multi-contamination level was estimated by calculating the pollution load index  
232 (PLI) according to the formula (Rashed, 2010; Affholder et al., 2014):

233

234 Eq. 2.  $PLI = \sqrt[4]{CF_{As} \times CF_{Cu} \times CF_{Pb} \times CF_{Zn}}$

235



236 Percentage of mobile MM in soil were calculated according to the formula (Affholder, 2013):

237

238 Eq. 3.  $\%MM_{\text{mob}} = ([MM]_{\text{mob}}/[MM]_{\text{tot}}) \times 100$

239

### 240 **2.3.3. Soil microbial analysis**

241 Soil microbial analyses were performed in order to assess the eco-physiological state of soil  
242 microbial communities.

243 Soil basal respiration (SBR) and substrate-induced respiration (SIR) were measured to assess  
244 global microbial activity and biomass (Anderson and Domsch, 1978; Anderson, 2003). Ten  
245 grams dry weight equivalent of fresh soil at 30 % of water holding capacity were placed in  
246 117 mL slightly open ajar glass jars and incubated for 48 h at 23 °C. For SBR measurement,  
247 each sample was oxygenated for 4 min and incubated for 4 h at 23 °C in tightly closed glass  
248 jars. Then, 1 mL of their atmosphere was sampled in the headspace with a syringe and  
249 injected into a gas chromatograph (Chrompack CHROM 3 – CP 9001) to analyse CO<sub>2</sub>  
250 production by microbial oxidation of the soil organic matter. SIR was estimated using a  
251 procedure from Anderson and Domsch (1978). Soil samples were amended with 450 mg of  
252 talc and 50 mg of powdered anhydrous glucose to maximize the respiration rate and incubated  
253 for 2 h at 23 °C in slightly open ajar glass jars. Then, each sample was oxygenated for 4 min  
254 and then incubated for 2 h at 23 °C in tightly closed glass jars. Then, 1 mL of their  
255 atmosphere was sampled in the head space with a syringe and injected into a gas  
256 chromatograph (Chrompack CHROM 3 – CP 9001) to analyse CO<sub>2</sub> production by microbial  
257 oxidation of glucose. The gas chromatograph was equipped with a thermal conductivity  
258 detector and a packed column (Porapack). The carrier gas helium flow was regulated at  
259 60 mL.h<sup>-1</sup>. Ambient CO<sub>2</sub> concentrations were subtracted from sampled CO<sub>2</sub> concentrations  
260 and resulting values were adjusted at 22 °C according to Ideal Gas Laws using a Q10 = 2. SIR  
261 was converted into microbial biomass (MB) using the relation established by Beare et al.  
262 (1990). Then, the metabolic quotient (qCO<sub>2</sub>), which is a sensitive eco-physiological indicator  
263 of soil stress induced by environmental conditions (Anderson, 2003), was obtained by  
264 calculating the SBR/MB ratio.

265 Glomalin-related soil protein (GRSP), a fungal glycoprotein synthesized by certain arbuscular  
266 mycorrhizal fungi involved in soil quality (Zhu and Miller, 2003; Rillig et al., 2004) and MM  
267 chelation (Cornejo et al., 2008; Vodnik et al., 2008), was quantified. Three analytical  
268 replicates were made for each sample. GRSP was extracted by repeated autoclaving  
269 extraction (Advantage-Lab ALO2-05) from 0.5 g of 40 °C dried soil in 4 mL of 50 mM

270 citrate pH 8.0 at 121 °C for 60 min. After each autoclaving cycle, the supernatant was  
271 removed by centrifugation (Hettich Zentrifugen Universal 320R) at 7000 g for 20 min. The  
272 extraction cycles were repeated until the supernatant had lost its dark brown-red color (8  
273 cycles). Supernatants of the different extraction cycles of the same analytical replica were  
274 pooled and then centrifuged at 7000 g for 10 min. GRSP content was measured in 1 mL  
275 supernatant by spectrophotometric dosage (595 nm) according to Bradford method, by using a  
276 Bovin Serum Albumin solution as standard. Finally, GRSP concentration was converted to  
277  $\text{g.kg}^{-1}$  of dry soil.

278

#### 279 **2.4. Runoff water analysis**

280 Runoff water were collected on each E and Cn plots over the period from 24 January 2019 to  
281 15 April 2019 during which there was 97.75 mm of rainfall. MM concentrations in runoff  
282 water ( $\text{MM}_{\text{water}}$ ) were determined by ICP-AES (Jobin Yvon Horiba, Spectra 2000) for As, Cu,  
283 Pb and Zn, for each E and Cn plot in order to evaluate MM transfer by hydric erosion. Before  
284 that, 20 mL of runoff water were filtered to 0.45  $\mu\text{m}$ , and 0.2 mL of nitric acid (Trace metal  
285 grade  $\text{HNO}_3$ ) were added to them. Quantity of each MM exported in runoff water ( $\text{MM}_{\text{water}}$ )  
286 was calculated according to the formula:

287

288 Eq. 4.  $\text{MM}_{\text{water}} = [\text{MM}]_{\text{water}} / \text{water volume of the collector}$

289

#### 290 **2.5. Vegetation analysis**

291 Vegetation analysis was performed in order to characterize the structure (plant cover), the  
292 composition (richness, diversity, equitability) and the sustainability (Raunkier plant life-form,  
293 Grime adaptative strategy) of each plant community. Cn plots, which were unvegetated at the  
294 beginning of this study, were considered in plant community analysis because plants  
295 established during the experiment.

296 Plant cover was estimated by different percentage class: < 10 %, 10-25 %, 25-50 %, 50-75 %,   
297 > 75 % (Braun-Blanquet et al., 1952). The following indices were estimated along with  
298 presence/absence and abundance field data: specific richness, Shannon's diversity index, and  
299 Pielou's equitability index (Wheater et al., 2011).

300 Plant functional traits (Garnier and Navas, 2013; Pérez-Harguindeguy et al., 2016) were  
301 studied from BASECO database (Gachet et al., 2005). Percentages of plant species in a plot  
302 per Raunkier plant life-form (geophyte, chamaephyte, hemicryptophyte, therophyte,  
303 phanerophyte) correspond to the ratio of the number of species in a plot with a given Raunkier

304 life-form on the total number of species in a plot. Percentages of plant species in a plot per  
305 Grime adaptative strategy (stress tolerant, competitive stress tolerant, ruderal stress tolerant,  
306 competitive ruderal stress tolerant) correspond to the ratio of the number of species in a plot  
307 with a given Grime adaptative strategy on the total number of species in a plot.

308

## 309 **2.6. Statistical analysis**

310 Statistical analyses were conducted with R (3.4.2 version). *P*-values were considered as  
311 significant when they were less than or equal to 0.05.

312 The two plant categories in E plots (i.e. planted and spontaneous) have been considered  
313 without distinction in statistical analysis because it was not possible to discriminate on field  
314 planted individuals from spontaneous individuals which came from vegetative propagation or  
315 seedlings from seeds of planted individuals or from the soil seed bank.

316 Kruskal-Wallis tests, followed by Dunn's post-hoc tests, were carried out to study the impact  
317 of experimental treatment (Cn, E, Cp) on soil (physicochemical and microbial) and plant  
318 community characteristics. Mann-Whitney tests were performed to study the impact of  
319 experimental treatment (Cn, E) on MM transfer in runoff water.

320 A path analysis was performed to evaluate the impacts of MM soil contamination on  
321 ecological restoration dynamic and the impacts of vegetal succession on agronomic and  
322 microbial soil quality. Path analysis is a particular case of structural equation modelling  
323 (SEM). Its permits to represent causal networks between several measured variables and to  
324 test model data consistency (Grace, 2006). Path analysis was performed by using the *lavaan*  
325 package (Rosseel, 2012). A conceptual model was developed from *a priori* knowledge  
326 consistent with our data and having biological sense (Fig. C). Five groups of measured  
327 variables were created by using axis 1 coordinates of principal component analysis (PCA):  
328 MM soil contamination ( $[As]_{tot}$ ,  $[Cu]_{tot}$ ,  $[Pb]_{tot}$ ,  $[Zn]_{tot}$ ; PCA axis 1 explaining 84.5 % of the  
329 variation), MM mobilization in soil and transfer by water erosion ( $[As]_{mob}$ ,  $[Cu]_{mob}$ ,  $[Pb]_{mob}$ ,  
330  $[Zn]_{mob}$ ,  $[As]_{H_2O}$ ,  $[Cu]_{H_2O}$ ,  $[Pb]_{H_2O}$ ,  $[Zn]_{H_2O}$ ; PCA axis 1 explaining 49.6 % of the variation),  
331 agronomic soil quality (pH, TOC, TKN, C/N, P, CEC, clay percentage; PCA axis 1  
332 explaining 55.9 % of the variation), microbial activities and biomass (SBR, MB, GRSP; PCA  
333 axis 1 explaining 88.2 % of the variation), plant community parameters (plant cover, specific  
334 richness, Shannon's diversity index, Pielou's equitability index, chamaephytes,  
335 hemicryptophytes and competitive stress tolerant species; PCA axis 1 explaining 74.2 % of  
336 the variation). The full model was simplified by stepwise exclusion of non-significant  
337 variables until a minimal adequate model was reached. The adequacy of the model was

338 determined by non-significant differences between predicted and observed covariance  
339 matrices (chisquared test,  $P > 0.05$ ), a low Root Mean Squared Error (RMSEA  $< 0.10$ ), a high  
340 Comparative Fit Index (CFI  $> 0.90$ ) and a low Standardized Root Mean Square Residual  
341 (SRMR  $< 0.08$ ) (Grace, 2006; Rosseel, 2012). The selected model does not consider MM  
342 mobilization in soil and transfer by hydric erosion because these variables do not permit to  
343 obtain an adequate model. Thus, Spearman correlation tests (correlation between two  
344 variables) were performed in order to test the relationship between MM mobilization in soil  
345 and transfer by hydric erosion and MM soil contamination, agronomic soil quality, microbial  
346 activities and biomass, plant cover, diversity and sustainability.

347

### 348 **3. Results**

#### 349 **3.1. Soil physicochemical characteristics**

350 Six parameters were significantly affected by the experimental treatment (Tables 1 and A).  
351 pH was 11 % higher for Cn than for Cp plots (Table 1). TOC content was 3.9 and 1.9 times  
352 higher in Cp than in Cn and E plots, respectively (Fig. 2a, Table 1). CEC was 2.1 and 1.4  
353 times higher in Cp than in Cn and E plots, respectively (Fig. 2b, Table 1). TKN was 2 times  
354 higher in Cp and E than in Cn plots (Fig. 2c, Table 1).  $As_{mob}$  percentage was 2.1 times higher  
355 in Cp than in Cn plots (Table 1).  $Zn_{mob}$  percentage was respectively 1.7 and 1.4 times higher  
356 in Cp and E than in Cn plots (Table 1).

357

#### 358 **3.2. Soil microbial properties**

359 Two parameters were significantly affected by the experimental treatments (Tables 1 and A).  
360 SBR was 4.2 and 5.2 times higher in Cp than in Cn and E plots, respectively (Table 1). GRSP  
361 content was 2.8 and 2.1 times higher in Cp than in Cn and E plots, respectively (Fig. 2d,  
362 Table 1).

363

#### 364 **3.3. Vegetation characteristics**

365 Ten parameters were significantly affected by the experimental treatments (Tables 1 and A).  
366 Plant cover was respectively 40.0 and 21.7 times higher in Cp and E plots than in Cn plots  
367 (Fig. 3a); specific richness was respectively 6.7 and 9.0 times higher in Cp and E plots than in  
368 Cn plots; Shannon's diversity index was respectively 8.3 and 9.6 times higher in Cp and E  
369 plots than in Cn plots (Fig. 3b); Pielou's equitability index was respectively 26.3 and 25.0  
370 times higher in Cp and E plots than for Cn plots; percentage of hemicryptophyte species  
371 respectively 18.1 and 9.0 times higher in Cp and E plots than for Cn plots (Fig. 3c);

372 percentage of competitive stress tolerant species respectively 83.3 and 29.8 times higher in Cp  
373 and E plots than for Cn plots (Fig. 3d); percentage of chamaephyte species respectively 28.7  
374 and 25.0 times higher in Cp and E plots than for Cn plots (Table 1). Cp plots presented a  
375 percentage of competitive stress tolerant species 2.8 times higher than in E plots (Fig. 3d,  
376 Table 1). E and Cn plots presented a percentage of therophyte species respectively 16.7 and  
377 14.7 times higher than in Cp plots (Fig. 3e, Table 1). E plots had a percentage of ruderal stress  
378 tolerant species 3.3 and 7.0 times higher than in Cp and Cn plots, respectively (Fig. 3f, Table  
379 1). In E plots, most of the transplanted native plant species remained and other spontaneous  
380 plant species appeared such as *Sonchus bulbosus* (L.) N. Kilian & Greuter, 2003, *Linaria*  
381 *supina* (L.) Chaz., 1790 or *Centranthus calcitrapae* (L.) Dufr., 1811. Spontaneous plant  
382 species also appeared in Cn plots (Table B).

383

### 384 **3.4. Correlations between soil, runoff water and plant community characteristics**

385 Path analysis (Fig. 4) showed that MM soil contamination negatively affected vegetation  
386 characteristics (i.e. plant cover, diversity and sustainability) and agronomic soil quality.  
387 Vegetation characteristics positively affected agronomic soil quality. Agronomic soil quality  
388 positively affected soil microbial properties (i.e. microbial biomass and activities). MM soil  
389 contamination and vegetation characteristics showed indirect effects on soil microbial  
390 properties mediated by degradation and improvement of agronomic soil quality, respectively.  
391 Spearman correlation tests showed that MM soil contamination was positively correlated to  
392 MM mobilization in soil and transfer by hydric erosion ( $r = 0.89$ ,  $P < 0.001$ ).

393

## 394 **4. Discussion**

395 The aim of this study was to evaluate the evolution of the ecological restoration trajectory and  
396 the effectiveness of phytostabilization at the peri-urban and protected site of the Escalette by a  
397 comparative *in situ* monitoring of soil properties, MM concentrations in soils and in runoff  
398 water and, characteristics of plant communities.

399

### 400 **4.1. Evaluation of ecological restoration success after three years**

401 Three attributes are mainly used to evaluate the success of an ecological restoration operation  
402 (Ruiz-Jaen and Mitchell Aide, 2005) out of the nine defined by the Society of Ecological  
403 Restoration (SER, 2016): diversity of the organism communities (richness, abundance),  
404 vegetation structure (cover, height, biomass) and ecological processes (biotic interactions,

405 nutrient and organic matter cycles). Compared to the reference plant community (i.e. a young  
406 Mediterranean scrubland for the present study), these features are milestones enabling to  
407 assess the ecosystem position on the restoration trajectory. When ecosystem disturbances are  
408 due to industrial pollution such as in this study site, the level of soil contamination is also an  
409 important parameter to consider in assessing the environmental physical conditions because it  
410 affects the composition, structure and functioning of the ecosystem (Wong, 2003; Li, 2006).

411 At the Escalette site, plant communities in the reference plots (Cp plots, young Mediterranean  
412 scrubland) and in the ecological restoration plots (E plots) had similar composition, relatively  
413 low diversity and high equitability during the study. However, they differed by their plant  
414 community structure and functions. In the reference plots, plant communities formed a  
415 completely permanent vegetation cover mainly shaped by hemicryptophytes and  
416 chamaephytes with competitive stress tolerant strategy. However, only a partial permanent  
417 plant cover occurred in ecological restoration plots (planted and spontaneous  
418 hemicryptophytes and chamaephytes with competitive stress tolerant strategy) with also  
419 annual species forming a more or less temporary vegetation cover (therophytes with ruderal  
420 and tolerant stress strategies) (Garnier and Navas, 2013; Pérez-Harguindeguy et al., 2016).  
421 Establishment of a dense and permanent vegetation cover is a central element for the success  
422 of an ecological restoration operation (Cortina et al., 2011). Indeed, this allows the creation of  
423 favorable micro-niches to the establishment of new local plant species by improving  
424 environmental conditions, notably edaphic ones, thus favoring plant succession and soil  
425 microbial functions (Padilla and Pugnaire, 2006; Padilla et al., 2009). At the Escalette site, the  
426 establishment of a large, diversified and sustainable plant cover is negatively affected by soil  
427 MM contamination. Indeed, soil contamination can constrain plant establishment and  
428 persistence because MM may cause toxic, oxidative and/or hydric stresses on plants leading  
429 to disruption of physiological processes (i.e. respiration, photosynthesis, water and nutrients  
430 absorption) essential for plant growth and development (Baryla et al., 2001; Yadav, 2010;  
431 Michalak, 2016). Moreover, MM contamination adds to intrinsic abiotic stress (i.e. scarce and  
432 scattered rain events, high summer temperatures, wind) which can strongly constraint plant  
433 establishment, growth and development (Bolan et al., 2011; Kumar Yadav et al., 2018).

434

435 Soil quality is also an important parameter to consider in an ecological restoration operation  
436 because soil functioning recovery is essential to pass the resilience threshold (Heneghan et al.,  
437 2008). Overall, the Escalette soils were characterized by poor agronomic quality (low organic  
438 matter, nutrient contents, high C/N ratio) (Thomas et al., 2006; Gobat et al., 2013). The high

439 MM soil contamination in this site negatively impacted the agronomic soil quality. A  
440 competition for adsorption onto soil particles can occur between toxic MM and essential  
441 cations for plant development (Adriano et al., 2004; Bolan et al., 2011; Kumar Yadav et al.,  
442 2018). This competition is governed by physicochemical soil parameters (MM concentrations,  
443 redox potential, cation exchange capacity, organic matter content and in particular pH) which  
444 influence MM distribution between solid soil phase and soil solution thus affecting MM  
445 mobility and bioavailability (Adriano et al., 2004; Bolan et al., 2011). Moreover, microbial  
446 communities were submitted to significant stress (high metabolic quotient whatever the  
447 treatment) due to the indirect negative effect of high MM contamination on microbial  
448 properties mediated by degradation of soil quality. This resulted in low microbial biomass as  
449 previously reported by Kızılkaya et al. (2004) and Gülser and Erdoğan (2008). This low  
450 biomass can be explained by a metabolic quotient increase. This indicates that sensitive  
451 microorganisms decrease whereas resistant microorganisms increase their respiration activity  
452 to maintain their basal metabolism at the expense of biosynthesis, resulting in their biomass  
453 decrease (Fließbach et al., 1994; Friedlová, 2010).

454

455 However, plant community differences between the reference plots, the ecological restoration  
456 plots and the unvegetated plots (Cn plots) induced soil properties variations. The completely  
457 permanent vegetation cover in the reference plots improved agronomic soil quality and  
458 therefore soil microbial activities and biomass compared to the ecological restoration and  
459 unvegetated plots. This may be due to regular litter input resulting in soil enrichment in  
460 organic matter and nutrients which increase microbial activities and biomass as reported by  
461 Kızılkaya et al. (2004), Gülser and Erdoğan (2008) and Friedlová (2010). Among these  
462 microbial activities, glomalin secretion by arbuscular mycorrhizal fungi was higher in the  
463 reference plots compared to the ecological restoration and unvegetated plots. This  
464 glycoprotein is known to enhance soil organic matter content (Rillig et al., 2001; Zhu and  
465 Miller, 2003), soil aggregation and structural stability (Rillig, 2004). In the ecological  
466 restoration plots, soil nitrogen content was higher than in the unvegetated ones. The  
467 occurrence of *C. juncea* individuals in the ecological restoration plots, a leguminous engineer  
468 plant in association with nitrogen-fixing bacteria and endomycorrhizal fungi (Carrasco et al.,  
469 2011), may facilitate other plant establishment by enhancing soil quality in particular nitrogen  
470 content (Padilla and Pugnaire, 2006).

471

472 Results of this study suggest that an ecological restoration trajectory was initiated at the  
473 ecological restoration plots (Fig. 5) according to the floristic composition and vegetation  
474 analysis processed in the Calanques National Park by Heckenroth et al. (2016b). Several  
475 hypotheses on ecological restoration trajectory evolution can be emitted. On the one hand,  
476 similarities of plant communities between the reference and the ecological restoration plots  
477 and presence of a partial annual plant cover at the ecological restoration plots may indicate  
478 that these plant communities are at an earlier stage of vegetation succession relative to the  
479 reference plots (*sensu* Aronson et al., 1993). Thus, ecological restoration trajectory may tend  
480 towards reference state, i.e. a young Mediterranean scrubland. On the other hand, differences  
481 in ecological niches between the reference plots and the ecological restoration plots (MM  
482 contamination heterogeneity, the occurrence of Aleppo pines close to Cp plots creating  
483 shading and litter inputs) can lead to the establishment of alternative stable states at the  
484 ecological restoration plots. Indeed, during an ecological restoration operation in  
485 Mediterranean area with limited human intervention, return to the reference state is rarely  
486 reached because aridity may keep the ecosystem below the irreversibility threshold (Aronson,  
487 1993; Young et al., 2001, 2005).

488

#### 489 **4.2. Evaluation of phytostabilization success after three years**

490 MM mobilization in soil and MM transfer by water erosion were not significantly different  
491 between the unvegetated and the ecological restoration plots. It was therefore not possible to  
492 demonstrate the stabilizing effect of soil contamination by plant communities and their  
493 associated microorganisms. This may be due to the temporary and removable runoff water  
494 collectors which were not really adapted to the scarce and scattered rain events which  
495 occurred during the study period. However, this system implementation was restricted by  
496 PNCal regulations in this protected area. Most of the studies used microcosms equipped with  
497 a downstream water collector to recover only runoff water from the plot (Bochet et al., 1998;  
498 Zhang et al., 2004; Martínez Raya et al., 2006). But, in this study, plots were not isolated from  
499 the local ecosystem. Thus, the runoff water recovered in collectors came from the plots  
500 themselves but also may come from upstream areas, generating a response more at the level  
501 of the ecosystem watershed than the plots.

502

503 After three years, no significant effect of soil agronomic quality, microbial and plant  
504 communities on MM immobilization and transfer were detected, although the literature  
505 indicates that improvement of soil quality, microbial functionalities, plant cover and



506 sustainability allow enhancing MM immobilization (Mendez et al., 2007; Mendez and Maier,  
507 2008; Valentín- Vargas et al., 2014). This result may be due to several parameters. Firstly, in  
508 this study, improvement of soil physicochemical properties was only related to the action of  
509 engineering native plant species and associated microorganisms, contrary to most  
510 phytostabilization studies using amendments (Rizzi et al., 2004; Radziemska et al., 2018).  
511 Thus, at the ecological restoration plots, improving the soil quality exclusively by plants and  
512 their impact on MM retention may take more time than when using amendments which hasten  
513 MM toxicity reduction and limit their bioavailability by adsorption and complexation  
514 processes (Rizzi et al., 2004; Mendez and Maier, 2008; Epelde et al., 2009; Radziemska et al.,  
515 2018). Then, MM immobilization by microorganisms by biosorption, bioaccumulation and  
516 transformation (Adriano et al., 2004; Khan, 2005; Ma et al., 2011) is limited by low biomass  
517 and microbial activities at the ecological restoration plots. Finally, the temporary vegetation  
518 cover of annual species with shallow roots may have not allowed a sustainable stabilization of  
519 MM contamination at the ecological restoration plots. Phytostabilization is enhanced by  
520 persistent and dense root systems and plant cover that reduce soil erosion by decreasing  
521 runoff water, protecting soil surface, increasing water infiltration and soil quality (Bochet et  
522 al., 1998; 2006; Durán Zuazo et al., 2006; 2008; Bolan et al., 2011) and immobilizing MM in  
523 roots and/or rhizosphere (Adriano et al., 2004; Pilon-Smits, 2005; Bolan et al., 2011).  
524 Therefore, without amendment inputs and under drastic Mediterranean environmental  
525 conditions, native plant communities and their associated microorganisms may need more  
526 time before improving MM stabilization by enhancing physicochemical soil parameters  
527 involved in MM retention and by biological mechanisms. Thus, a longer observation period  
528 may be necessary to conclude on a potential phytostabilization effect of these plant  
529 communities. It would also be interesting to take into account other aspects of soil microbial  
530 communities (litter decomposition, C and N mineralization, functional diversity) which are  
531 driving parameters for good management of soil MM contamination (Brookes, 1995;  
532 Friedlová, 2010, Colin et al., 2019).

533

## 534 **5. Conclusions**

535 The objectives of this study were to evaluate the evolution of restoration trajectory at the  
536 ecological restoration plots and the effectiveness of soil MM stabilization *via* soil, runoff  
537 water and plant community analyses. Three years after the implementation of the ecological  
538 restoration plots at the Escalette site, it appears that the mobilized ecological engineering tools  
539 have enabled the establishment of perennial and annual plant communities at these plots.

540 However, there is no evidence of a significant improvement of soil quality and  
541 phytostabilization efficiency. This may be due to the non-use of amendments and a too short  
542 observation period.

543 The active restoration actions carried out in this Mediterranean peri-urban and MM  
544 contaminated area, based on abiotic (creation of cultivation terraces) and biotic (native plant  
545 species planting) filter handling by less intrusive techniques on the environment adequate  
546 with regulations of a protected area, seem to allow the creation of micro-niches favorable to  
547 the establishment of resilient dynamics of the native plant communities. The obtained results  
548 support the fact that it is preferable to intervene in these environments rather than to leave in  
549 place unvegetated screes with risks of erosion of contaminated soil particles and transfer to  
550 the food web.

551

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566

## 567 **References**

- 568 Adriano, D.C., Wenzel, W.W., Vangronsveld, J., Bolan, N.S., 2004. Role of assisted natural  
569 remediation in environmental cleanup. *Geoderma*. 122, 121–142.
- 570 Affholder, M.C., 2013. Approche des mécanismes de tolérance du romarin aux éléments  
571 traces métalliques et métalloïdes : perspectives pour une phytostabilisation des sols  
572 méditerranéens pollués. Doctoral thesis, Aix-Marseille University, 325p.

- 573 Affholder, M.C., Pricop, A.D., Laffont-Schwob, I., Coulomb, B., Rabier, J., Borla, A.,  
574 Demelas, C., Prudent P., 2014. As, Pb, Sb, and Zn transfer from soil to root of wild  
575 rosemary: do native symbionts matter? *Plant Soil*. 382, 219–236.
- 576 Affre, L., Dumas, P.J., Dumas, E., Laffont-Schwob, I., Tatoni, T., 2015. Regard écologique  
577 sur le recul stratégique : atouts et risques pour la diversité végétale péri-urbaine  
578 marseillaise. *Vertigo*. Hors-série 21, Droit des risques littoraux et changement  
579 climatique : connaissance, anticipation et innovation. 14p.
- 580 Aillaud, G.J., Crouzet, A., 1988. Un milieu difficile, des plantes adaptées : la végétation  
581 littorale du Sud du golfe de Marseille. Centre Régional de Documentation  
582 Pédagogique, Marseille. 50p.
- 583 Anderson, J.P.E., Domsch, K.H., 1978. A physiological method for the quantitative  
584 measurement of microbial biomass in soils. *Soil Biol. Biochem.* 10, 215–221.
- 585 Anderson, T.H., 2003. Microbial eco-physiological indicators to assess soil quality. *Agr.*  
586 *Ecosyst. Environ.* 98, 285–293.
- 587 Aronson, J., Floret, C., Le Floc'h, E., Ovalle, C., Pontanier, R., 1993. Restoration and  
588 rehabilitation of degraded ecosystems in arid and semi-arid lands – View from the  
589 South. *Restor. Ecol.* 1, 8–17.
- 590 Baker, A.J.M., Ernst, W.H.O., van der Ent, A., Malaisse, F., Ginocchio, R., 2010.  
591 Metallophytes: the unique biological resource, its ecology and conservational status in  
592 Europe, central Africa and Latin America, in: Batty, L.C., Hallberg, K.B. (ed.),  
593 *Ecology of industrial pollution*. Cambridge University Press, pp. 7–40.
- 594 Ballesteros, M., Cañadas, E.M., Marrs, R.H., Foronda, A., Martín- Peinado, F.J., Lorite, J.,  
595 2017. Restoration of Gypsicolous Vegetation on Quarry Slopes: Guidance for  
596 Hydroseeding under Contrasting Inclination and Aspect. *Land Degrad. Develop.* 28,  
597 2146–2154.
- 598 Baryla, A., Carrier, P., Franck, F., Coulomb, C., Sahut, C., Havaux, M., 2001. Leaf chlorosis  
599 in oilseed rape plants (*Brassica napus*) grown on cadmium-polluted soil: causes and  
600 consequences for photosynthesis and growth. *Planta*. 212, 696–709.
- 601 Beare, M.H., Neely, C.L., Coleman, D.C., Hargrove, W.L., 1990. A substrate-induced  
602 respiration (SIR) method for measurement of fungal and bacterial biomass on plant  
603 residues. *Soil Biol. Biochem.* 22, 585–594.
- 604 Bochet, E., Rubio, J.L., Poesen, J., 1998. Relative efficiency of three representative matorral  
605 species in reducing water erosion at the microscale in a semi-arid climate (Valencia,  
606 Spain). *Geomorphology*. 23, 139–150.

607 Bochet, E., Poesen, J., Rubio, J.L., 2006. Runoff and soil loss under individual plants of a  
608 semi-arid Mediterranean shrubland: influence of plant morphology and rainfall  
609 intensity. *Earth Surf. Process. Landforms.* 31, 536–549.

610 Bolan, N.S., Park, J.H., Robinson, B., Naidu, R., Huh K.Y., 2011. Phytostabilization: a green  
611 approach to contaminant containment. *Adv. Agron.* 112, 145-204.

612 Braun-Blanquet J., Roussine N., Nègre R., 1952. Les groupements végétaux de la France  
613 méditerranéenne. *Dir. Carte Group. Vég. Afr. Nord, CNRS.* 297p.

614 Brookes, P.C., 1995. The use of microbial parameters in monitoring soil pollution by heavy  
615 metals. *Biol. Fertil. Soils.* 19, 269–279.

616 Callaway, R.M., 2007. Positive interactions and interdependence in plant communities, ed.  
617 Springer. 415p.

618 Carrasco, L., Azcón, R., Kohler, J., Roldán, A., Caravaca, F., 2011. Comparative effects of  
619 native filamentous and arbuscular mycorrhizal fungi in the establishment of an  
620 autochthonous, leguminous shrub growing in a metal-contaminated soil. *Sci. Total  
621 Environ.* 409, 1205–1209.

622 Colin, Y., Goberna, M., Verdú, M., Navarro-Cano, J.A., 2019. Successional trajectories of  
623 soil bacterial communities in mine tailings: The role of plant functional traits. *Journal  
624 of Environmental Management* 241, 284–292.

625 Cornejo, P., Meier, S., Borie, G., Rillig, M. C., Borie, F., 2008. Glomalin-related soil protein  
626 in a Mediterranean ecosystem affected by a copper smelter and its contribution to Cu  
627 and Zn sequestration. *Sci. Total Environ.* 406, 154–160.

628 Cortina, J., Amat, B., Castillo, V., Fuentes, D., Maestre, F.T., Padilla, F.M., Rojo, L., 2011.  
629 The restoration of vegetation cover in the semi-arid Iberian southeast. *J. Arid Environ.*  
630 75, 1377–1384.

631 Daumalin, X., Raveux, O., 2016. A region scarred by pollution, in: REF2C (ed.), *Pollution of  
632 Marseille's industrial Calanques: the impact of the past on the present.* pp. 131–187.

633 Durán Zuazo, V. H., Martínez, J. R. F., Pleguezuelo, C. R. R., Martínez Raya, A., Rodríguez,  
634 B. C., 2006. Soil-erosion and runoff prevention by plant covers in a mountainous area  
635 (SE Spain): implications for sustainable agriculture. *Environmentalist.* 26, 309–319.

636 Durán Zuazo, V. H., Rodríguez Pleguezuelo, C. R., Francia Martínez, J. R., Cárceles  
637 Rodríguez, B., Martínez Raya, A., Pérez Galindo, P., 2008. Harvest intensity of  
638 aromatic shrubs vs. soil erosion: an equilibrium for sustainable agriculture (SE Spain).  
639 *CATENA.* 73, 107–116.

640 Epelde, L., Becerril, J.M., Mijangos, I., and Garbisu, C., 2009. Evaluation of the efficiency of  
641 a phytostabilization process with biological indicators of soil health. *J. Environ. Qual.*  
642 38, 2041–2049.

643 European Commission, 2011. The EU biodiversity strategy to 2020. 27p.

644 FAO (Food and Agriculture Organization of the United Nations), 2015. Les sols sont une  
645 ressource non renouvelable. 4p.

646 Faucon, M. P., Colinet, G., Mahy, G., Ngongo Luhembwe, M., Verbruggen, N., Meerts, P.,  
647 2009. Soil influence on Cu and Co uptake and plant size in the cuprophytes  
648 *Crepidiorhopalon perennis* and *C. tenuis* (*Scrophulariaceae*) in SC Africa. *Plant Soil.*  
649 317, 201–212.

650 Fließbach, A., Martens, R., Reber, H., 1994. Soil microbial biomass and microbial activity in  
651 soils treated with heavy metal contaminated sewage sludge. *Soil Biol. Biochem.* 26,  
652 1201–1205.

653 Frérot, H., Lefèbvre, C., Gruber, W., Collin, C., Santos, A.D., Escarré, J., 2006. Specific  
654 interactions between local metallicolous plants improve the phytostabilization of mine  
655 soils. *Plant Soil.* 282, 53–65.

656 Friedlová, M., 2010. The influence of heavy metals on soil biological and chemical  
657 properties. *Soil Water Res.* 5, 21–27.

658 Gachet, S., Véla, E., Tatoni, T., 2005. BASECO: a floristic and ecological database of  
659 Mediterranean French flora. *Biodivers. Conserv.* 14, 1023–1034.

660 Garnier, E., Navas, M.L., 2013. Diversité fonctionnelle des plantes, ed. De Boeck Supérieur.

661 Ghazaryan, K.A., Movsesyan, H.S., Minkina, T.M., Sushkova, S.N., Rajput, V.D., 2019. The  
662 identification of phytoextraction potential of *Melilotus officinalis* and *Amaranthus*  
663 *retroflexus* growing on copper and molybdenum polluted soils. *Environ. Geochem.*  
664 *Health.* pp 1–9.

665 Gobat, J.M., Aragano, M., Matthey, W., 2013. Le sol vivant – Bases de pédologie – Biologie  
666 des sols, ed. Presses polytechniques et universitaires romandes. 817p.

667 Grace, J.B., 2006. Structural equation modeling and natural systems, ed. Cambridge  
668 University Press. 378p.

669 Gülser, F., Erdoğan, E., 2008. The effects of heavy metal pollution on enzyme activities and  
670 basal soil respiration of roadside soils. *Environ. Monit. Assess.* 145, 127–133.

671 Heckenroth, A., Prudent, P., Rabier, J., Monsara, P., Desrousseaux, M., Laffont-Schwob, I.,  
672 2016a. What are the potential environmental solutions for diffuse pollution?, in:

673 REF2C (ed.), Pollution of Marseille's industrial Calanques: the impact of the past on  
674 the present, pp. 291–328.

675 Heckenroth, A., Rabier, J., Dutoit, T., Torre, F., Prudent, P., Laffont-Schwob, I., 2016b.  
676 Selection of native plants with phytoremediation potential for highly contaminated  
677 Mediterranean soil restoration: tools for a non-destructive and integrative approach. *J.*  
678 *Environ. Manage.* 183, 850–863.

679 Heckenroth, A., Rabier, J., Monsara, P., Le Mire-Pecheux, L., Prudent, P., Sastre, C., Laffont-  
680 Schwob, I., 2017. Mise en place d'une approche intégrée pour la restauration  
681 écologique des milieux pollués du Parc national des Calanques. *Naturae.* 9, 1–14.

682 Heneghan, L., Miller, S.P., Baer, S., Callahan, M.A., Montgomery, J., Pavao-Zuckerman, M.,  
683 Rhoades, C.C., Richardson, S., 2008. Integrating soil ecological knowledge into  
684 restoration management. *Restor. Ecol.* 16, 608–617.

685 Ilunga wa Ilunga, E., Mahy, G., Piqueray, J., Séleck, M., Shutcha, M.N., Meerts, P., Faucon  
686 M.P., 2015. Plant functional traits as a promising tool for the ecological restoration of  
687 degraded tropical metal-rich habitats and revegetation of metal-rich bare soils: a case  
688 study in copper vegetation of Katanga, DRC. *Ecol. Eng.* 82, 214–221.

689 Jaunatre, R., Buisson, E., Dutoit, T., 2012. First-year results of a multi-treatment steppe  
690 restoration experiment in La Crau (Provence, France). *Plecevo* 145, 13–23.

691 Khan, A. G., 2005. Role of soil microbes in the rhizospheres of plants growing on trace metal  
692 contaminated soils in phytoremediation. *J. Trace. Elem. Med. Bio.* 18, 355–364.

693 Kızılkaya, R., Aşkın, T., Bayraklı, B., Sağlam, M., 2004. Microbiological characteristics of  
694 soils contaminated with heavy metals. *Eur. J. Soil Biol.* 40, 95–102.

695 Knoerr, A., 1959. Le milieu, la flore, la végétation, la biologie des halophytes, dans l'archipel  
696 de Riou et sur la côte Sud de Marseille. Thèse de doctorat, Aix-Marseille Université,  
697 France. 420p.

698 Kumar Yadav, K., Gupta, N., Kumar, A., Reece, L.M., Singh, N., Rezaia, S., Ahmad Khan,  
699 S., 2018. Mechanistic understanding and holistic approach of phytoremediation: a  
700 review on application and future prospects. *Ecol. Eng.* 120, 274–298.

701 Laffont-Schwob, I., Heckenroth, A., Rabier, J., Masotti, V., Oursel, B., Prudent, P., 2016.  
702 Diffuse and widespread modern-day pollution, in : REF2C (ed.), Pollution of  
703 Marseille's industrial Calanques: the impact of the past on the present. pp. 204–249.

704 Li, M.S., 2006. Ecological restoration of mineland with particular reference to the  
705 metalliferous mine wasteland in China: a review of research and practice. *Sci. Total*  
706 *Environ.* 357, 38–53.

707 Ma, Y., Prasad, M. N. V., Rajkumar, M., Freitas, H., 2011. Plant growth promoting  
708 rhizobacteria and endophytes accelerate phytoremediation of metalliferous soils.  
709 Biotechnol. Adv. 29, 248–258.

710 Marchiol, L., Fellet, G., Boscutti, F., Montella, C., Mozzi, R., Guarino, C., 2013. Gentle  
711 remediation at the former “Pertusola Sud” zinc smelter: evaluation of native species  
712 for phytoremediation purposes. Ecol. Eng. 53, 343–353.

713 Marrugo-Negrete, J., Durango-Hernández, J., Pinedo-Hernández, J., Olivero-Verbel, J., Díez,  
714 S., 2015. Phytoremediation of mercury-contaminated soils by *Jatropha curcas*.  
715 Chemosphere. 127, 58–63.

716 Martínez Raya, A., Durán Zuazo, V.H., Francia Martínez, J.R., 2006. Soil erosion and runoff  
717 response to plant-cover strips on semiarid slopes (SE Spain). Land Degrad. Develop.  
718 17, 1–11.

719 Mendez, M.O., Glenn, E.P., Maier, R.M., 2007. Phytostabilization potential of quailbush for  
720 mine tailings: growth, metal accumulation, and microbial community changes. J.  
721 Environ. Qual. 36, 245–253.

722 Mendez, M.O., Maier, R., 2008. Phytostabilization of mine tailings in arid and semiarid  
723 environments – an emerging remediation technology. Environ. Health Persp. 116, 278–  
724 283.

725 Michalak, A., 2016. Phenolic compounds and their antioxidant activity in plants growing  
726 under heavy metal stress. Pol. J. Environ. 15, 523–530.

727 NF ISO 11263, 1995. Qualité du sol - Dosage du phosphore - Dosage spectrométrique du  
728 phosphore soluble dans une solution d'hydrogénocarbonate de sodium. 6p.

729 NF ISO 11261, 1995. Qualité du sol - Dosage de l'azote total - Méthode de Kjeldahl modifiée.  
730 4p.

731 NF ISO 10694, 1995. Qualité du sol - Dosage du carbone organique et du carbone total après  
732 combustion sèche (analyse élémentaire). 7p.

733 NF ISO 11466, 1995. Qualité du sol - Extraction des éléments en traces solubles dans l'eau  
734 régale. 6p.

735 NF ISO 10390, 2005. Qualité du sol - Détermination du pH. 7p.

736 NF X 31-107, 2003. Qualité du sol - Détermination de la distribution granulométrique des  
737 particules du sol - Méthode à la pipette. 6p.

738 NF X 31-130, 1999. Qualité des sols - Méthodes chimiques - Détermination de la capacité  
739 d'échange cationique (CEC) et des cations extractibles. 7p.

740 Padilla, F.M., Ortega, R., Sánchez, J., Pugnaire, F.I., 2009. Rethinking species selection for  
741 restoration of arid shrublands. *Basic Appl. Ecol.* 10, 640–647.

742 Padilla, F.M., Pugnaire, F.I., 2006. The role of nurse plants in the restoration of degraded  
743 environments. *Front. Ecol. Environ.* 4, 196–202.

744 Pauwels, M., Frérot, H., Bonnin, I., Saumitou-Laprade, P., 2006. A broad-scale analysis of  
745 population differentiation for Zn tolerance in an emerging model species for tolerance  
746 study: *Arabidopsis halleri* (*Brassicaceae*). *J. Evolution. Biol.* 19, 1838–1850

747 Pérez-Harguindeguy, N., Díaz, S., Garnier, E., Lavorel, S., Poorter, H., Jaureguiberry, P.,  
748 Bret-Harte, M.S., Cornwell, W.K., Craine, J.M., Gurvich, D.E., Urcelay, C.,  
749 Veneklaas, E.J., Reich, P.B., Poorter, L., Wright, I.J., Ray, P., Enrico, L., Pausas, J.G.,  
750 de Vos, A.C., Buchmann, N., Funes, G., Quétier, F., Hodgson, J.G., Thompson, K.,  
751 Morgan, H.D., ter Steege, H., Sack, L., Blonder, B., Poschlod, P., Vaieretti, M.V.,  
752 Conti, G., Staver, A.C., Aquino, S., Cornelissen, J.H.C., 2016. New handbook for  
753 standardised measurement of plant functional traits worldwide. *Aust. J. Bot.* 61, 167–  
754 234.

755 Pilon-Smits, E., 2005. Phytoremediation. *Annu. Rev. Plant Biol.* 56, 15–39.

756 Quevauviller, P., 1998. Operationally defined extraction procedures for soil and sediment  
757 analysis I. Standardization. *Trends Anal. Chem.* 17, 289–298.

758 Quézel, P., Médail, F., 2003. *Ecologie et biogéographie des forêts du bassin méditerranéen*,  
759 ed. Elsevier, Paris. 576p.

760 Radziemska, M., Koda, E., Bilgin, A., Vaverková, M. D., 2018. Concept of aided  
761 phytostabilization of contaminated soil in postindustrial areas. *Int. J. Environ. Res.*  
762 *Public Health.* 24, 1–15.

763 Rashed, M.N., 2010. Monitoring of contaminated toxic and heavy metals, from mine tailings  
764 through age accumulation, in soil and some wild plants at Southeast Egypt. *J. Hazard.*  
765 *Mater.* 178, 739–746.

766 Rey, F., Burylo, M., 2014. Can bioengineering structures made of willow cuttings trap  
767 sediment in eroded marly gullies in a Mediterranean mountainous climate?  
768 *Geomorphology* 204, 564-572.

769 Rillig, M.C., 2004. Arbuscular mycorrhizae, glomalin, and soil aggregation. *Can. J. Soil Sci.*  
770 84, 355–363.

771 Rillig, M. C., Wright, S. F., Nichols, K. A., Schmidt, W. F., Torn, M. S., 2001. Large  
772 contribution of arbuscular mycorrhizal fungi to soil carbon pools in tropical forest  
773 soils. *Plant Soil.* 233, 167–177.



- 774 Rizzi, L., Petruzzelli, G., Poggio, G., Vigna Guidi, G., 2004. Soil physical changes and plant  
775 availability of Zn and Pb in a treatability test of phytostabilization. *Chemosphere*. 57,  
776 1039–1046.
- 777 Rosseel, Y., 2012. lavaan: an R package for structural equation modeling. *J. Stat. Soft.* 48, 1–  
778 36.
- 779 Ruiz-Jaen, M.C., Mitchell Aide, T., 2005. Restoration success: how is it being measured?  
780 *Restor. Ecol.* 13, 569–577.
- 781 Society for Ecological Restoration (SER), 2004. The SER international primer on ecological  
782 restoration. 13p.
- 783 Society for Ecological Restoration (SER), 2016. Standards internationaux pour la restauration  
784 écologique. 50p.
- 785 Testiati, E., Parinet, J., Massiani, C., Laffont-Schwob, I., Rabier, J., Pfeifer, H.R., Lenoble,  
786 V., Masotti, V., Prudent, P., 2013. Trace metal and metalloid contamination levels in  
787 soils and in two native plant species of a former industrial site: evaluation of the  
788 phytostabilization potential. *J. Hazard. Mater.* 248–249, 131–141.
- 789 Thomas, R.J., El-Dessougi, H., Tubeileh, A., 2006. Soil system management under arid and  
790 semiarid conditions, in: Uphoff, N. (ed.), *Biological approaches to sustainable soil*  
791 *systems*. CRC Press, pp. 41–55.
- 792 Valentín-Vargas, A., Root, R.A., Neilson, J.W., Chorover, J., Maier, R.M., 2014.  
793 Environmental factors influencing the structural dynamics of soil microbial  
794 communities during assisted phytostabilization of acid-generating mine tailings: a  
795 mesocosm experiment. *Sci. Total Environ.* 500–501, 314–324.
- 796 Vodnik, D., Grčman, H., Maček, I., van Elteren, J.T., Kovačević, M., 2008. The contribution  
797 of glomalin-related soil protein to Pb and Zn sequestration in polluted soil. *Sci. Total*  
798 *Environ.* 392, 130–136.
- 799 Wheeler, C.P., Bell, J.R., Cook, P.A., 2011. Analyzing and interpreting information, in: John  
800 Wiley & Sons Ltd (ed.), *Practical field ecology: a project guide*. Wiley-Blackwell,  
801 Oxford, pp. 235–294.
- 802 Wong, M.H., 2003. Ecological restoration of mine degraded soils, with emphasis on metal  
803 contaminated soils. *Chemosphere*. 50, 775–780.
- 804 Yadav, S.K., 2010. Heavy metals toxicity in plants: an overview on the role of glutathione  
805 and phytochelatins in heavy metal stress tolerance of plants. *S. Afr. J. Bot.* 76, 167–  
806 179.

- 807 Young, T.P., Chase, J.M., Huddleston, R.T., 2001. Comparing, contrasting and combining  
808 paradigms in the context of ecological restoration. *Ecol. Restor.* 19, 5–8.
- 809 Young, T.P., Petersen, D.A., Clary, J.J., 2005. The ecology of restoration: historical links,  
810 emerging issues and unexplored realms. *Ecol. Lett.* 662–673.
- 811 Zhang, B., Yang, Y., Zepp, H., 2004. Effect of vegetation restoration on soil and water  
812 erosion and nutrient losses of a severely eroded clayey Plinthudult in southeastern  
813 China. *Catena.* 57, 77–90.
- 814 Zhu, Y. G., Miller, R. M., 2003. Carbon cycling by arbuscular mycorrhizal fungi in soil-plant  
815 systems. *Trends Plant Sci.* 8, 407–409.
- 816

817 **Figure captions**

818

819 **Figure 1.** Location of (a) the Escalette site in the Calanques National Park (modified after  
820 Heckenroth et al., 2016b), (b) the Escalette lead smelting factory, horizontal chimney and  
821 study area (source: Google Earth), (c) Cn plots (red, negative control), E plots (yellow,  
822 ecological restoration plot) and Cp plots (green, positive control) plots in the study area  
823 (source: Géoportail) and (d) characteristics of Cn, E and Cp plots.

824

825 **Figure 2.** Total organic carbon (TOC) concentration, total Kjeldahl nitrogen (TKN)  
826 concentration, cation exchange capacity (CEC) and glomalin-related soil protein (GRSP)  
827 concentration in Cn plots (negative control, n = 5), E plots (ecological restoration plots, n = 5)  
828 and Cp plots (positive control, n = 5). Different letters above boxplots indicate a significant  
829 difference between plots for a given parameter ( $P \leq 0.05$ ).

830

831 **Figure 3.** Plant cover, Shannon's diversity index, percentage of hemicryptophyte species,  
832 percentage of therophyte species, percentage of competitive stress tolerant species and  
833 percentage of ruderal stress tolerant species in Cn plots (negative control, n = 5), E plots  
834 (ecological restoration plots, n = 5) and Cp plots (positive control, n = 5). Different letters  
835 above box plots indicate a significant difference between plots for a given parameter  
836 ( $P \leq 0.05$ ).

837

838 **Figure 4.** Path analysis performed between agronomic soil quality, MM soil contamination,  
839 microbial activities and biomass, plant cover, diversity and sustainability. The arrows  
840 represent the relationships between the different parameters. Red arrow indicates a negative  
841 relationship and green arrow a positive relationship. The thicker is the arrow, the greater is the  
842 correlation. Correlation coefficients and significance levels of the correlations (\*:  $P \leq 0.05$ ;  
843 \*\*:  $P \leq 0.01$ ; \*\*\*:  $P \leq 0.001$ ) are indicated at each arrow.

844

845 **Figure 5.** Ecological restoration process and perspectives after three years at the Escalette  
846 site.

847

848 **Table captions**

849 **Table 1.** Averages  $\pm$  standard errors ( $n = 5$ ) of soil (TOC: total organic carbon, TKN:  
850 Kjeldahl nitrogen, CEC: cation exchange capacity,  $[As]_{tot}$ ,  $[Cu]_{tot}$ ,  $[Pb]_{tot}$ ,  $[Zn]_{tot}$ : arsenic,  
851 copper, lead and zinc pseudo-total concentrations in soil, PLI: pollution load index,  $[As]_{mob}$ ,  
852  $[Cu]_{mob}$ ,  $[Pb]_{mob}$ ,  $[Zn]_{mob}$ : arsenic, copper, lead and zinc mobile fraction concentrations in soil,  
853 and % (in brackets) of mobile versus pseudo-total MM in soil, SBR: soil basal respiration,  
854 MB: microbial biomass,  $qCO_2$ : metabolic quotient, GRSP: glomalin-related soil protein),  
855 runoff water ( $As_{water}$ ,  $Cu_{water}$ ,  $Pb_{water}$ ,  $Zn_{water}$ : quantities of arsenic, copper, lead and zinc  
856 exported in runoff water) and plant communities parameters of Cn plots (negative control,  
857  $n = 5$ ), E plots (ecological restoration plots,  $n = 5$ ) and Cp plots (positive control,  $n = 5$ ).  
858 Different letters indicate a significant difference between plots for a given parameter  
859 ( $P \leq 0.05$ ).

860

861

862

863 **Appendix captions**

864

865 **Figure A.** Localization of the Escalette study site in Marseilleveyre massif (Calanques  
866 National Park).

867

868 **Figure B.** Schematic representation of Escalette's E plots organized in two terraces and  
869 planted with eight pseudo-metallophytes species from the local floristic background in  
870 November 2015 (modified from Heckenroth, 2017).

871

872 **Figure C.** Conceptual model of path analysis taking into account agronomic soil quality, MM  
873 soil contamination, microbial activities and biomass, plant cover, diversity and sustainability.  
874 The arrows represent the relationships between the different parameters. Red arrow indicates  
875 a negative relationship and green arrow a positive relationship.

876 **1** – MM soil contamination impacts negatively plant cover, diversity and sustainability. MM  
877 soil contamination may cause oxidative and hydric stresses disturbing essential physiological  
878 process of plant growth and development (respiration, photosynthesis, water and nutrients  
879 absorption) (Baryla et al., 2001; Yadav, 2010; Michalak, 2016).

880 **2** – MM soil contamination impacts negatively agronomic soil quality. MM soil  
881 contamination potentially created a competition between essential cations for plant  
882 development and toxic MM for adsorption to soil particles (Bolan et al., 2011).

883 **3** – MM soil contamination impacts negatively microbial activities and biomass. MM soil  
884 contamination may cause a reduction of microbial number, diversity and activities (Fließbach  
885 et al., 1994; Brookes, 1995; Kızılkaya et al., 2004; Gülser et Erdoğan, 2008).

886 **4** – Increase in plant cover, diversity and sustainability impacts positively agronomic soil  
887 quality. A permanent, dense and diversified plant cover improves soil organic matter, nitrogen  
888 and nutrient contents by regular litter inputs (Gobat et al., 2013).

889 **5** – Improvement of agronomic soil quality, in particular organic matter quantity and quality,  
890 impacts positively microbial activities and biomass (Kızılkaya et al., 2004; Gülser et Erdoğan,  
891 2008; Friedlová, 2010; Gobat et al., 2013).

892 **6** – The higher the MM soil contamination is, the more MM are mobilizable and transferable  
893 (Bolan et al., 2011).

894 **7, 8, 9** - Improvement of soil quality, microbial functionalities and plant cover, diversity and  
895 sustainability allows to enhance MM immobilization by physicochemical and biological

896 mechanisms. This reduces MM mobilization in soil and transfer by hydric erosion (Adriano et  
897 al., 2004; Pilon-Smits, 2005; Bolan et al., 2011).

898

899 The adequacy of the full model was not good ( $P$  of  $\chi^2 = 0.020$ ). Thus, we removed non-  
900 significant relationships ( $P$ -value of relationship 3 = 0.656;  $P$ -value of relationship 7 = 0.864;  
901  $P$ -value of relationship 8 = 0.306;  $P$ -value of relationship 9 = 0.696) from this model.  
902 Exclusion of non-significant relationships did not permit to obtain a good adequacy of the  
903 new model ( $P$  of  $\chi^2 = 0.010$ , CFI = 0.850, SRMR = 0.111). Finally, by removing the  
904 relationship 6, we obtained an adequate model ( $p$  of  $\chi^2 = 0.061$ , RMSEA = 0.067, CFI = 0.904  
905 and SRMR = 0.063) and this final model was selected.

906

907 **Table A.** (a)  $P$ -values of Kruskal-Wallis and Dunn tests performed for soil characteristics  
908 (TOC: total organic carbon, TKN: Kjeldahl nitrogen, CEC: cation exchange capacity,  $[\text{As}]_{\text{tot}}$ ,  
909  $[\text{Cu}]_{\text{tot}}$ ,  $[\text{Pb}]_{\text{tot}}$ ,  $[\text{Zn}]_{\text{tot}}$ : arsenic, copper, lead and zinc pseudo-total concentrations in soil, PLI:  
910 pollution load index,  $\text{As}_{\text{mob}}(\%)$ ,  $\text{Cu}_{\text{mob}}(\%)$ ,  $\text{Pb}_{\text{mob}}(\%)$ ,  $\text{Zn}_{\text{mob}}(\%)$ : percentages of mobile  
911 arsenic, copper, lead and zinc in soil, SBR: soil basal respiration, MB: microbial biomass,  
912  $q\text{CO}_2$ : metabolic quotient, GRSP: glomalin-related soil protein) and plant communities  
913 parameters of Cn plots (negative control,  $n = 5$ ), E plots (ecological restoration plots,  $n = 5$ )  
914 and Cp plots (positive control,  $n = 5$ ), (b)  $P$ -values of Mann-Whitney tests performed for  
915 runoff water parameters ( $\text{As}_{\text{water}}$ ,  $\text{Cu}_{\text{water}}$ ,  $\text{Pb}_{\text{water}}$ ,  $\text{Zn}_{\text{water}}$ : quantities of arsenic, copper, lead and  
916 zinc exported in runoff water) of Cn and E plots. Significant difference between treatments  
917 are indicated by stars (\*:  $P \leq 0.05$ ; \*\*:  $P \leq 0.01$ ; \*\*\*:  $P \leq 0.001$ ).

918

919 **Table B.** Floristic inventory on E plots (ecological restoration plots), Cp plots (positive  
920 control) and Cn plots (negative control).

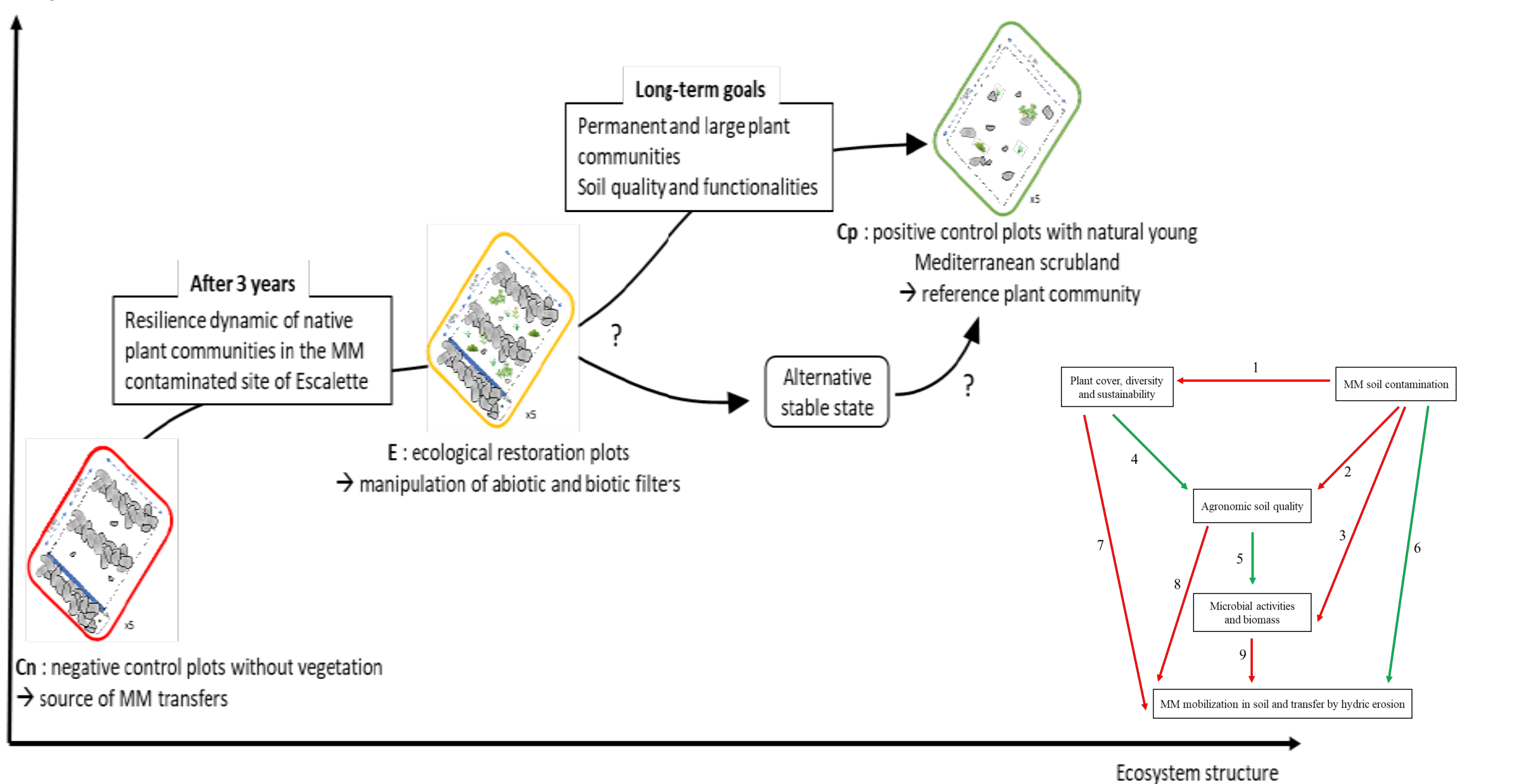


Figure1

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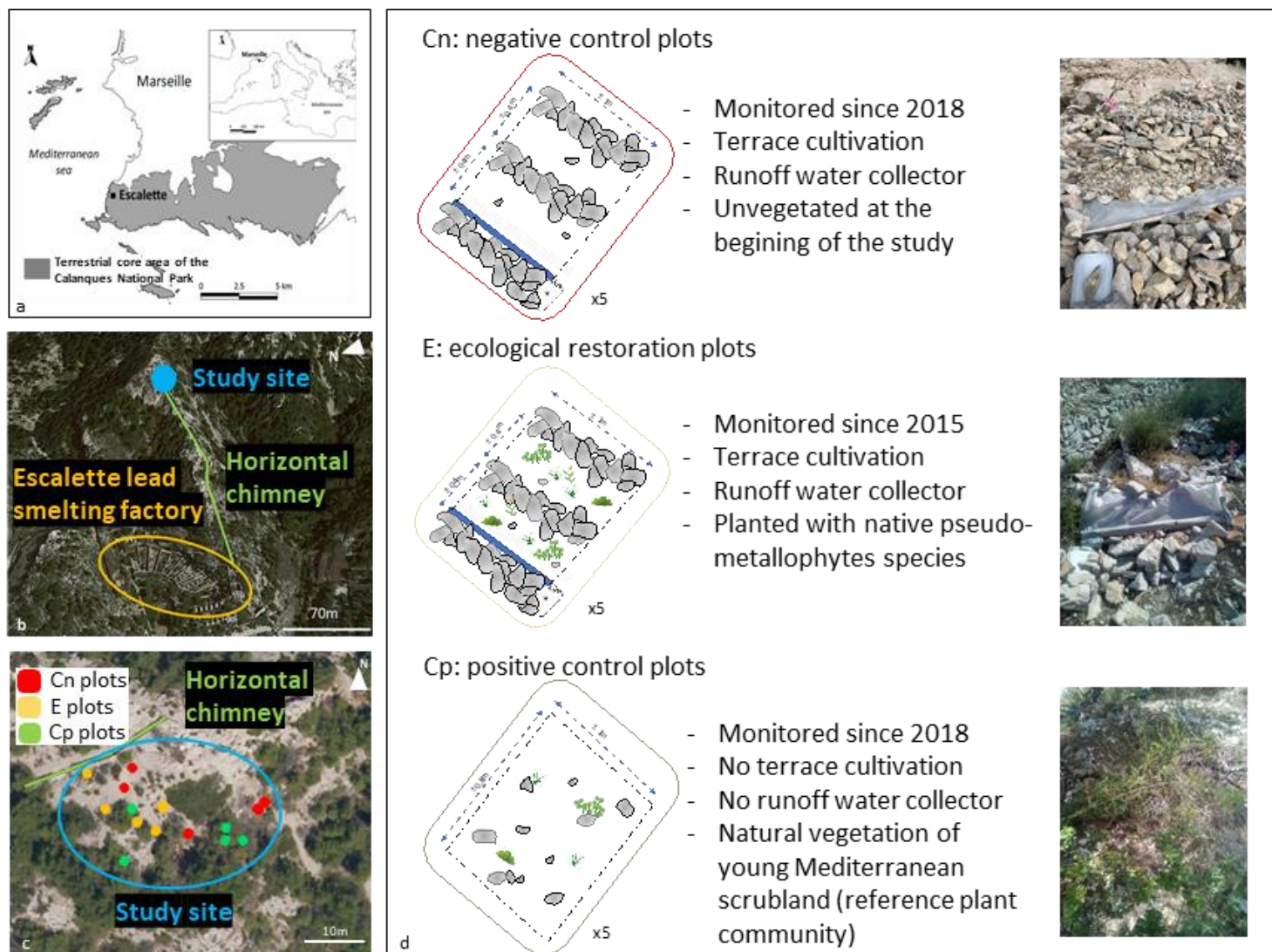
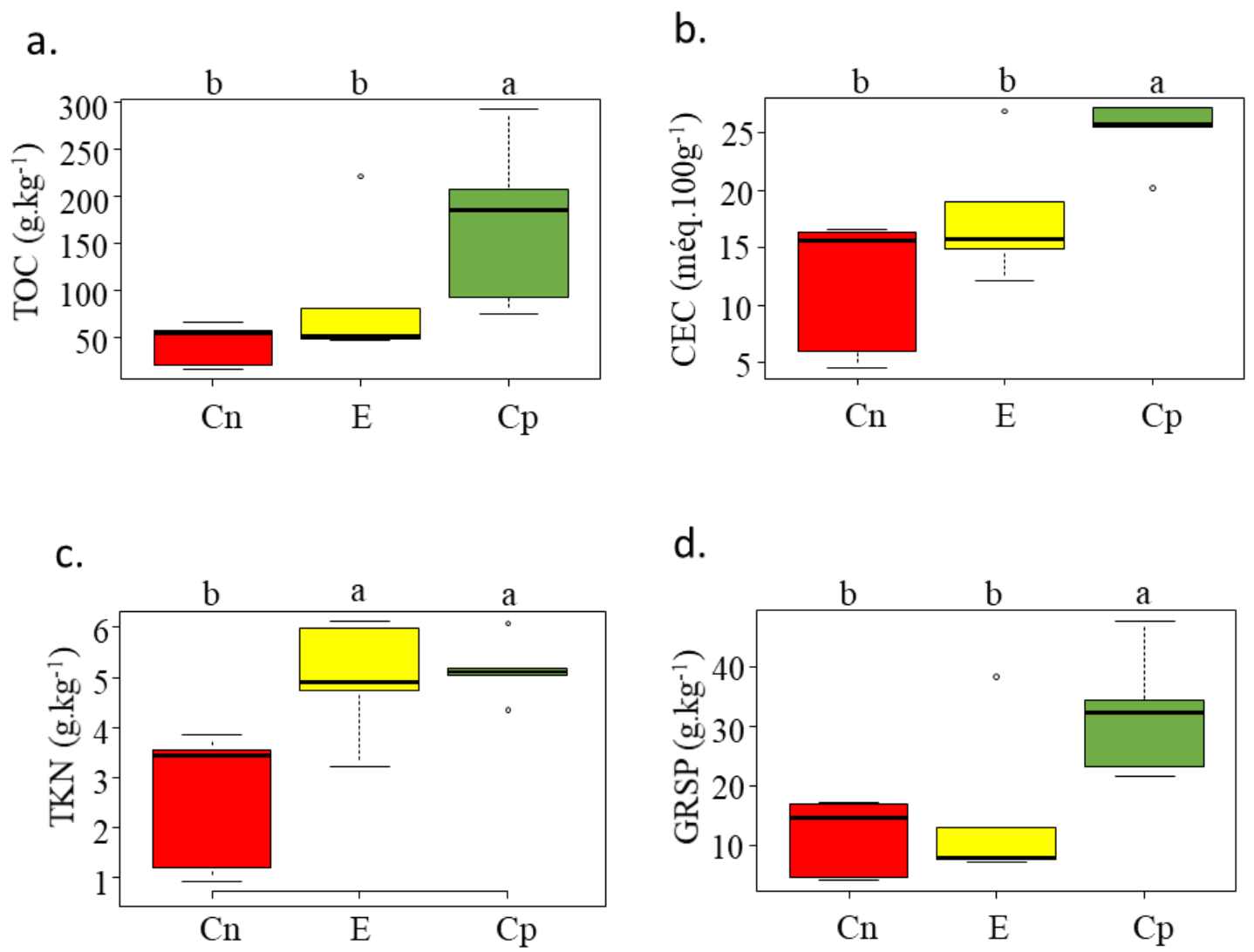




Figure2

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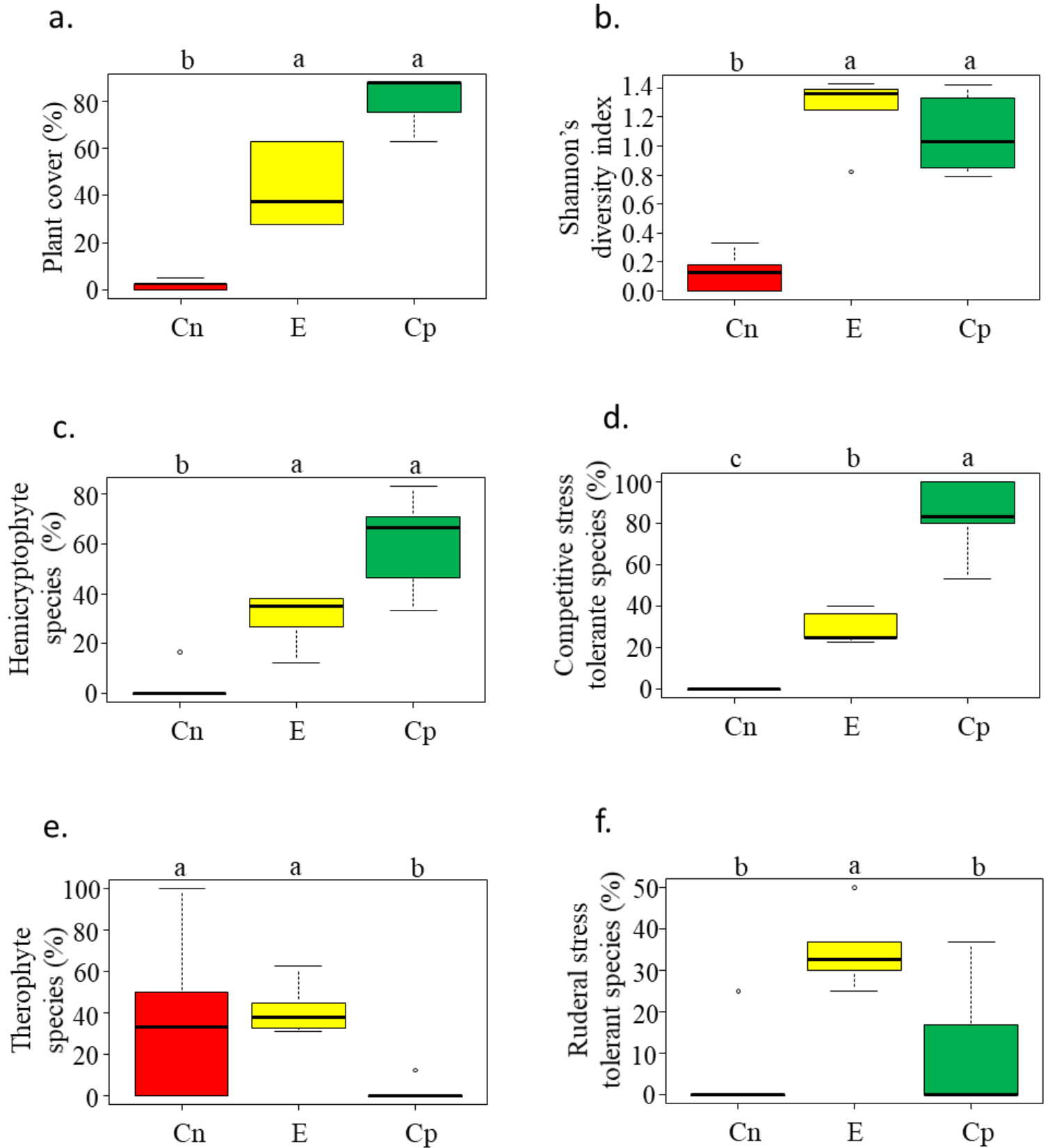


Figure4

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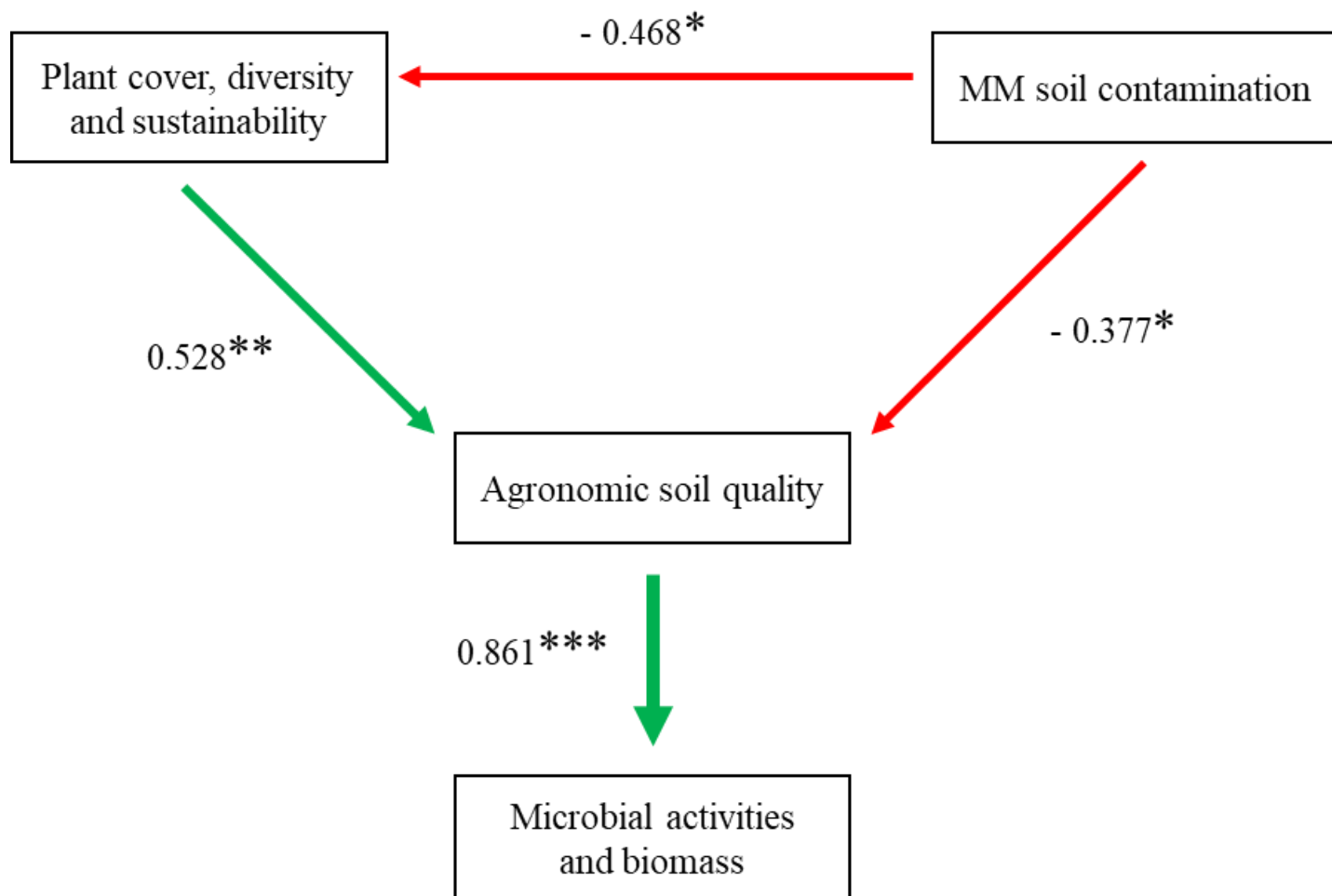
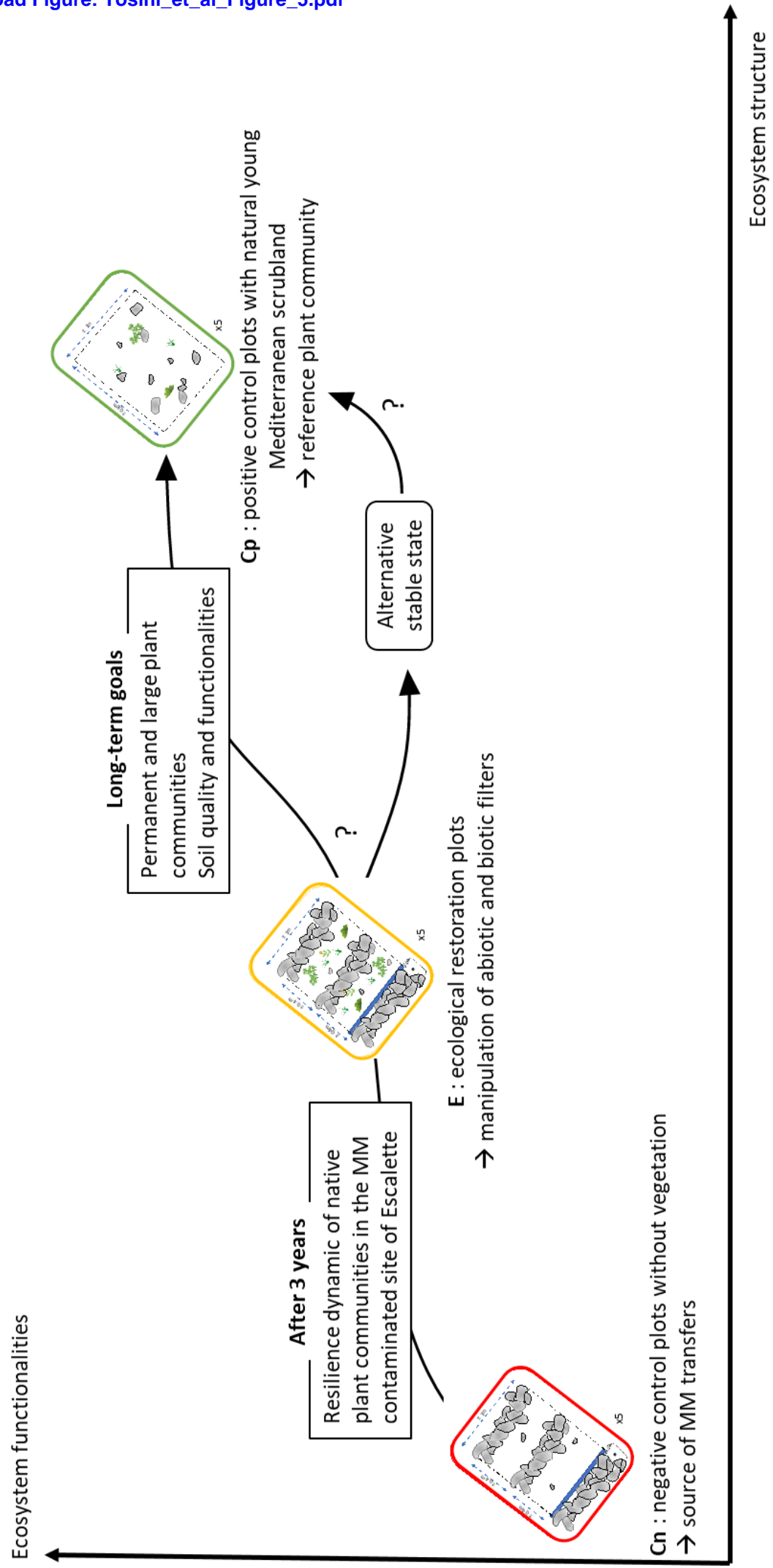


Figure5

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Ecosystem structure

Table1

	Parameters	Cn plots	E plots	Cp plots
Soil characteristics	pH	7.68 ± 0.06 <sup>a</sup>	7.41 ± 0.11 <sup>ab</sup>	6.93 ± 0.15 <sup>b</sup>
	TOC (g.kg <sup>-1</sup> )	43.58 ± 10.25 <sup>b</sup>	90.28 ± 33.25 <sup>b</sup>	171.36 ± 39.58 <sup>a</sup>
	TKN (g.kg <sup>-1</sup> )	2.59 ± 0.63 <sup>b</sup>	4.99 ± 0.52 <sup>a</sup>	5.15 ± 0.27 <sup>a</sup>
	C/N	35.86 ± 4.53 <sup>a</sup>	24.58 ± 6.78 <sup>a</sup>	36.65 ± 6.35 <sup>a</sup>
	P (g.kg <sup>-1</sup> )	0.04 ± 0.00 <sup>a</sup>	0.04 ± 0.00 <sup>a</sup>	0.05 ± 0.01 <sup>a</sup>
	CEC (méq.100g <sup>-1</sup> )	11.84 ± 2.66 <sup>b</sup>	17.70 ± 2.53 <sup>b</sup>	25.12 ± 1.28 <sup>a</sup>
	Clay (%)	14.96 ± 2.75 <sup>a</sup>	21.24 ± 2.48 <sup>a</sup>	21.18 ± 3.75 <sup>a</sup>
	Lime (%)	19.65 ± 2.57 <sup>a</sup>	21.37 ± 1.83 <sup>a</sup>	17.65 ± 1.04 <sup>a</sup>
	Sand (%)	22.86 ± 3.29 <sup>a</sup>	18.02 ± 2.62 <sup>a</sup>	21.76 ± 1.65 <sup>a</sup>
	[As] <sub>tot</sub> (mg.kg <sup>-1</sup> )	19770.04 ± 7082.15 <sup>a</sup>	13640.33 ± 1174.80 <sup>a</sup>	8579.00 ± 2250.43 <sup>a</sup>
	[Cu] <sub>tot</sub> (mg.kg <sup>-1</sup> )	75.54 ± 15.98 <sup>a</sup>	60.51 ± 3.28 <sup>a</sup>	51.40 ± 5.73 <sup>a</sup>
	[Pb] <sub>tot</sub> (mg.kg <sup>-1</sup> )	63470.68 ± 10902.03 <sup>a</sup>	75600.15 ± 3780.47 <sup>a</sup>	55685.76 ± 10653.43 <sup>a</sup>
	[Zn] <sub>tot</sub> (mg.kg <sup>-1</sup> )	33754.54 ± 11515.45 <sup>a</sup>	25944.16 ± 2664.65 <sup>a</sup>	14457.50 ± 4061.24 <sup>a</sup>
	PLI	406.54 ± 107.11 <sup>a</sup>	351.45 ± 23.02 <sup>a</sup>	239.33 ± 50.23 <sup>a</sup>
	[As] <sub>mob</sub> (mg.kg <sup>-1</sup> ) (%)	3306.47 ± 987.35 (18.65 ± 1.49) <sup>b</sup>	3568.09 ± 423.65 (27.29 ± 5.12) <sup>a</sup>	2821.20 ± 312.74 (39.35 ± 7.27) <sup>a</sup>
	[Cu] <sub>mob</sub> (mg.kg <sup>-1</sup> ) (%)	26.62 ± 5.34 (35.28 ± 2.30) <sup>a</sup>	22.17 ± 0.78 (36.98 ± 1.73) <sup>a</sup>	21.04 ± 1.64 (41.62 ± 1.87) <sup>a</sup>
	[Pb] <sub>mob</sub> (mg.kg <sup>-1</sup> )(%)	40357.01 ± 8421.02 (62.96 ± 6.78) <sup>a</sup>	45668.26 ± 2261.84 (60.67 ± 2.90) <sup>a</sup>	37579.61 ± 5272.05 (70.29 ± 3.97) <sup>a</sup>
	[Zn] <sub>mob</sub> (mg.kg <sup>-1</sup> ) (%)	11553.41 ± 3546.00 (36.17 ± 2.82) <sup>b</sup>	12398.93 ± 1075.45 (49.37 ± 5.10) <sup>a</sup>	8014.85 ± 1582.89 (63.34 ± 8.53) <sup>a</sup>
	SBR (µg CO <sub>2</sub> -C.g <sup>-1</sup> .h <sup>-1</sup> )	0.048 ± 0.024 <sup>b</sup>	0.042 ± 0.010 <sup>b</sup>	0.209 ± 0.024 <sup>a</sup>
	MB (µg C <sub>mic</sub> .g <sup>-1</sup> )	0.033 ± 0.015 <sup>a</sup>	0.033 ± 0.011 <sup>a</sup>	0.095 ± 0.009 <sup>a</sup>
qCO <sub>2</sub> (µg CO <sub>2</sub> -C.µg C <sub>mic</sub> <sup>-1</sup> .h <sup>-1</sup> )	2.237 ± 0.333 <sup>a</sup>	1.434 ± 0.147 <sup>a</sup>	2.224 ± 0.353 <sup>a</sup>	
GRSP (g.kg <sup>-1</sup> )	11.42 ± 2.95 <sup>b</sup>	14.73 ± 5.98 <sup>b</sup>	31.75 ± 4.69 <sup>a</sup>	
Water runoff characteristics	As <sub>water</sub> (µg)	16.09 ± 5.44 <sup>a</sup>	44.65 ± 20.80 <sup>a</sup>	NA
	Cu <sub>water</sub> (µg)	3.06 ± 0.91 <sup>a</sup>	3.52 ± 2.47 <sup>a</sup>	NA
	Pb <sub>water</sub> (µg)	47.79 ± 30.05 <sup>a</sup>	103.89 ± 65.44 <sup>a</sup>	NA
	Zn <sub>water</sub> (µg)	489.72 ± 188.40 <sup>a</sup>	252.77 ± 40.16 <sup>a</sup>	NA
Plant communities characteristics	Plant cover (%)	2.00 ± 0.94 <sup>b</sup>	43.50 ± 7.87 <sup>a</sup>	80.00 ± 5.00 <sup>a</sup>
	Specific richness	0.60 ± 0.29 <sup>b</sup>	5.40 ± 0.51 <sup>a</sup>	4.00 ± 0.45 <sup>a</sup>
	Shannon's diversity index	0.13 ± 0.06 <sup>b</sup>	1.25 ± 0.11 <sup>a</sup>	1.08 ± 0.13 <sup>a</sup>
	Pielou equitability index	0.03 ± 0.03 <sup>b</sup>	0.75 ± 0.04 <sup>a</sup>	0.79 ± 0.03 <sup>a</sup>
	Geophyte (%)	0.00 ± 0.00 <sup>a</sup>	3.10 ± 3.10 <sup>a</sup>	7.00 ± 5.17 <sup>a</sup>
	Chamaephyte (%)	0.00 ± 0.00 <sup>b</sup>	25.02 ± 4.36 <sup>a</sup>	28.67 ± 5.13 <sup>a</sup>
	Hemicryptophyte (%)	3.33 ± 3.33 <sup>b</sup>	30.07 ± 4.87 <sup>a</sup>	60.17 ± 8.93 <sup>a</sup>
	Therophyte (%)	36.67 ± 18.56 <sup>a</sup>	41.81 ± 5.73 <sup>a</sup>	2.50 ± 2.50 <sup>b</sup>
	Phanerophyte (%)	0.00 ± 0.00 <sup>a</sup>	0.00 ± 0.00 <sup>a</sup>	1.67 ± 1.67 <sup>a</sup>
	Stress tolerant (%)	35.00 ± 18.71 <sup>a</sup>	28.00 ± 2.76 <sup>a</sup>	4.00 ± 2.45 <sup>a</sup>
	Competitive Stress tolerant (%)	0.00 ± 0.00 <sup>c</sup>	29.83 ± 3.53 <sup>b</sup>	83.33 ± 8.56 <sup>a</sup>
	Ruderal Stress tolerant (%)	5.00 ± 5.00 <sup>b</sup>	34.83 ± 4.24 <sup>a</sup>	10.67 ± 7.26 <sup>b</sup>
Competitive Ruderal Stress tolerant (%)	0.00 ± 0.00 <sup>a</sup>	5.33 ± 3.43 <sup>a</sup>	2.00 ± 2.00 <sup>a</sup>	