

1 **Impact of breakwater relocation on benthic biodiversity associated to seagrass**
2 **meadows of Northern Adriatic Sea**

3

4 Carugati L.^{1*}, Lo Martire M.¹, Gambi C.¹, Danovaro R.^{1,2}

5

6

7 ¹*Università Politecnica delle Marche, Dipartimento di Scienze della Vita e dell'Ambiente,*
8 *Ancona, Italy*

9 ²*Stazione Zoologica Anton Dohrn, Napoli, Italy*

10

11

12 *Corresponding author: tel: +39 0712204328; e mail: l.carugati@univpm.it

13

14

15 **ABSTRACT**

16 Breakwaters are man-made constructions utilised for preventing coastal erosion primarily
17 from wave action. At the same time defence structures, modifying currents and circulation,
18 can affect water quality and benthic assemblages. Assessing and minimizing the impact of
19 these structures is a priority in human-modified coastal ecosystems, such as the central
20 northern Adriatic where breakwaters extend for hundreds of km. We investigated the effects
21 of breakwater relocation on benthic features and meiofaunal diversity. To do this we
22 conducted a before - after comparison (2011 - 2017) the relocation, which occurred in 2015.
23 The analysis was conducted comparing a sheltered site, characterized by the presence of
24 seagrass meadows and a wave-exposed site. Sediment features and meiofaunal variables were
25 altered by the relocation, especially in the areas colonised by seagrass meadows. Results
26 presented here pointed out an enrichment in organic matter, the loss of two sensitive taxa
27 (Cumacea and Ostracoda) and a shift in the assemblage structure with the increase of the
28 relevance of Copepoda and Polychaeta. These results indicate that the careful management of
29 breakwater is crucial for planning adequate conservation practises and protecting seagrass
30 habitats and their biodiversity.

31

32 **Key-words:** breakwaters, relocation, seagrass, organic matter, meiofauna, Adriatic Sea

33

34

35 This contribution is the written, peer-reviewed version of a paper presented at the Conference
36 "Changes and Crises in the Mediterranean Sea" held at Accademia Nazionale dei Lincei in
37 Rome on October 17, 2017.

38

39 **Introduction**

40 Breakwaters are artificial structures planned to contrast shoreline erosion from wave actions
41 and currents (Bertasi et al. 2007; Relini et al. 2007). Since the seventies, breakwaters have
42 been installed along the Mediterranean coasts, and 35% of them deployed along the Italian
43 coastline of the northern Adriatic Sea (Fabi and Spagnolo 2011; Santelli et al. 2013). Sandy
44 beaches along the north-western coasts of the Adriatic Sea have been subjected to a significant
45 anthropogenic impact since historical times, and the deployment of breakwaters along with
46 uncontrolled freshwater inputs in highly urbanized littorals may greatly increase the impact
47 on coastal ecosystem (Cencini 1998; Colantoni et al. 2004). By reducing wave action, defence
48 structures can alter sedimentation rates, the quantity and quality of organic matter and
49 nutrients that are transported and deposited in the coastal area (Thomalla and Vincent 2003;
50 Cuadrado et al. 2005; Zyserman et al. 2005). An improper breakwaters design can cause
51 hypoxia and reduced water quality in these coastal areas, making their relocation a potential
52 solution to solve these critical issues. The minimization of impacts of breakwaters is a priority
53 especially in sensitive or ecologically vulnerable coastal ecosystems, such as seagrass
54 meadows that are important engineering species providing sediment stability and refuge for
55 highly diverse assemblages (Airoldi et al. 2005; Barbier et al. 2011).

56 Besides the well-known effects on fish populations, man-made structures can also have a
57 strong impact on benthic assemblages living in the adjacent soft-bottom sediments. Most of
58 the studies that have been published about the effects of the breakwaters concerns the
59 macrofauna (Martin et al. 2005; Bertasi et al. 2007), while the response of the smaller-sized
60 meiofauna has been poorly investigated (Danovaro et al. 2002; Covazzi-Harriague et al. 2013,
61 Semprucci et al. 2017). Meiofauna is well known to respond rapidly to changes of
62 environmental factors such as grain size and food availability (Bianchelli et al. 2016). The
63 latter is one of the main factor influencing benthic assemblages in sandy beaches, modulating

64 the trophic interactions between meiofauna and higher trophic levels (Covazzi et al. 2000,
65 2006).

66 Meiofaunal organisms, due to their high abundance, diversity, short generation time and
67 strong sensitivity to environmental disturbances, have been widely used to determine the
68 extent of various types of impacts in marine ecosystems (Heip et al. 1985; Danovaro et al.
69 2000; Austen and Widdicombe 2006; Semprucci et al. 2015a, b, c).

70 In this study we aim to investigate the effects of breakwater relocation on nearby sediments
71 and associated meiofauna in the case study area of Gabicce Mare (North-Western Adriatic
72 Sea). Sediment samples were collected before (2011) and after (2017) the relocation at the
73 protected side of the breakwaters characterized by the presence of seagrass meadows and at
74 the wave-exposed shore, with bare sediments, to test the following null hypothesis: grain size
75 and food resources along with meiofaunal abundance, higher taxa richness and taxonomic
76 composition are not significantly influenced by breakwater relocation, in the two sampling
77 sites.

78

79 **MATERIAL AND METHODS**

80 **Study area**

81 The investigated area, located in the northern part of the Marche region (Italy), is Gabicce
82 Mare (Northwestern coastline, 43° 57.967' N and 12° 45.859' E). The northern part is limited
83 by a touristic harbour and the Tavollo River, and the southern portion is confined by the rocky
84 cliff, Monte San Bartolo, which gives the name to the Regional Natural Park created in 1997,
85 to protect an area of great naturalistic importance (Colantoni et al. 2004). Gabicce Mare is
86 characterized by important hot-spots of benthic biodiversity and the presence of seagrass
87 meadows constituted by *Zostera marina*, *Nanozostera noltii* and *Cymodocea nodosa*
88 (Coccioni 2003; Balsamo et al. 2011).

89 Gabicce Mare presents eroded Messinian deposits, mainly composed of sand and pelites
90 (Tramontana et al. 2005 and references therein; Principi et al. 2011). Although this coastal
91 area is characterized by lower primary productivity than the North Adriatic Sea (Zavatarelli
92 et al. 2000), the contribution of local Apennine rivers and the nutrient-enriched coastal current
93 coming from the northern basin enhance its nutrient availability (Artegiani et al. 1997).

94 Along the coast of Marche region, important extractions of sediments were carried out in the
95 1960s and 1970s, and several harbours were built. As a result, a reduction of sediment flow
96 was reported and several breakwaters and seawalls were placed to protect the coast from
97 further erosional processes, typically occurring between the mouth of the Foglia river and that
98 of the Metauro river, adjacent to the Natural Regional Park of Monte San Bartolo (Colantoni
99 et al. 2004; Semprucci et al., 2010). In Gabicce Mare, the breakwaters are located ca. 100 m
100 from the shore and parallel to the coast. The seawater circulation is mainly related to wave
101 motion and its interaction with breakwaters, inducing local longshore and rip currents that
102 control the sediment transport (Semprucci et al. 2010).

103 The breakwaters resulted not highly effective in protecting the coastline of Gabicce Mare,
104 which continue to be exposed to erosion phenomena. In addition, the water stagnation caused
105 by the breakwaters, determined hypoxic conditions in summer, reduced water quality and
106 consequent low level of satisfaction of end users (i.e., tourists). Thus, in order to solve these
107 critical issues, breakwaters have been relocated in 2015.

108

109 **Sampling**

110 Sampling activities were performed in spring before (2011) and after (2017) the breakwater
111 relocation. In both sampling periods, sampling strategy was aimed at covering the sheltered
112 site, between the coastal line and breakwaters, characterized by the presence of seagrass
113 meadows (hereafter called internal site), and the adjacent wave-exposed shore, characterized

114 by bare sediments (external site). Along a transect parallel to the coast, twenty stations were
115 sampled, during each sampling year: nine stations were selected at internal site, and other
116 eleven stations were sampled at external site. Control stations were chosen at 50 m from the
117 breakwaters, which is considered a sufficient distance to exclude man-made structure effects
118 on sediments (Ambrose and Anderson 1990). Here, we report the preliminary results
119 analysing a first set of six stations, sampled between 1.6 and 3.5 m depth: two stations at
120 internal site and other two stations at external site. CTRL2 and CTRL3 were selected as
121 controls (Fig. 1; Table 1).

122 Sediment samples were collected by means of a Van Veen grab (sampling surface 0.15 m²),
123 on board of the *R/V Actea*. To cope with the possible bias raised by using the Van Veen grab,
124 which may produce leaking of interstitial water during recovery, we collected samples only
125 from deployments in which the grab resulted completely watertight. Sediment sub-samples
126 for the subsequent analyses of total organic matter (TOM) and meiofauna were collected from
127 three independent deployments of the grab by means of Plexiglas corers (internal diameter 3.6
128 cm). Once on board, each sediment core was kept at in situ temperature until brought to the
129 laboratory.

130 In the laboratory, sediment samples dedicated to the analysis of organic matter were stored at
131 -20°C until analysis (usually within two weeks), whereas samples for meiofauna analyses
132 were preserved with formalin (final concentration 4% in sea water filtered on a 20 µm mesh)
133 and stained with Rose Bengal (0.5 g L⁻¹) until further treatment as described below, according
134 to Danovaro (2010).

135

136 **Environmental analyses**

137 Grain-size distribution was analysed using a dry sieve technique (% sand=fraction >0.625
138 mm) (Giere et al., 1988). Total organic matter (TOM) content was analysed by measuring the

139 weight loss after ignition of dry sediment (Buchanan and Kain 1971). Sediment samples were
140 first dried at 608 °C for 6 h, weighed in a precision balance (accuracy 0.1 mg), then calcined
141 in a muffle furnace at 550 °C for 4 h and weighed again to determine the inorganic fraction.
142 The organic fraction content was calculated as the difference between the total weight and the
143 ash weight.

144

145 **Meiofaunal analysis**

146 In the laboratory sediment samples were sieved through a 1000- μm mesh, and a 20- μm mesh
147 was used to retain the smallest metazoan organisms (Danovaro, 2010). The fraction remaining
148 on the latter sieve was re-suspended and centrifuged three times with Ludox HS40 (diluted
149 with water to a final density of 1.18 g cm⁻³), according to Danovaro (2010). All animals
150 remaining in the supernatant were passed again through a 20 μm mesh net, stained with Rose
151 Bengal and sorted under a stereomicroscope at x 40 magnification (Heip et al. 1985; Danovaro
152 2010).

153 According to the literature and, therefore, for comparison purposes, meiofaunal abundance
154 was expressed as number of individuals 10 cm⁻², and their diversity expressed as richness of
155 higher taxa.

156

157 **Statistical analyses**

158 Differences in the investigated variables before and after breakwater relocation and among
159 sampling sites, were analysed using a sampling design with three factors as main sources of
160 variance: *year* (fixed, 2 levels), *site* (fixed, 3 levels), and *sampling stations* (random, nested
161 in *site*). All the analyses were carried out using the distance-based permutational analysis of
162 variance (PERMANOVA; Anderson 2001; McArdle and Anderson 2001) and the tests were
163 based on matrixes of Euclidean distances (for TOM) or Bray–Curtis similarity matrices (for

164 faunal data) of previously normalized (TOM) or untransformed (faunal) data, using 999
165 permutations of the residuals under a reduced model (Anderson 2001; McArdle and Anderson
166 2001). To down-weight the effects of abundant species, we also performed a PERMANOVA
167 analysis for faunal data based on Bray–Curtis similarity matrices of previously
168 presence/absence transformed data.

169 Significant terms were investigated using *a posteriori* pair-wise comparison with the
170 PERMANOVA t statistic and 999 permutations. Because of the restricted number of unique
171 permutations, P values in the PERMANOVA and pairwise tests were obtained from Monte
172 Carlo asymptotic distributions.

173 The meiofaunal taxa contributing to dissimilarities between factors were investigated through
174 the SIMPER analysis (Gray 2000). Ranked matrices of Bray–Curtis similarities based on
175 untransformed and presence/absence data, were used as input for the SIMPER tests.

176 The PERMANOVA and SIMPER analyses were performed using the routines included in the
177 software PRIMER 6+ (Clarke and Gorley 2006; Anderson et al. 2008).

178

179 **RESULTS**

180 **Environmental parameters**

181 The analysis of grain size distribution showed that sand represented the dominant fraction in
182 the wave-exposed area, at controls and external site (~90% of sand and ~10% of pelite, on
183 average), in both sampling years. The mud fraction (pelite, <63 µm) was significantly higher
184 at the stations sampled in the sheltered site (up to 40%), within the seagrass meadows, than at
185 control and external sites. In addition, after the breakwater relocation, we found that the
186 contribution of gravel significantly increased, from 0% in 2011, up to 5% in 2017 (Table 1,
187 2).

188 Total organic matter contents found after the breakwater relocation were significantly higher
189 ($17.2 \pm 4.2 \text{ mg g}^{-1}$) than those found before works ($11.9 \pm 2.1 \text{ mg g}^{-1}$; Fig. 2; Table 2). The results
190 of the PERMANOVA tests showed the presence of significant effects for the interaction *Year*
191 \times *Site* (Fig. 2; Table 2). The pairwise test for the factor *year* revealed that at internal site, values
192 were significantly higher in 2017 ($30.3 \pm 2.9 \text{ mg g}^{-1}$) than in 2011 ($18.7 \pm 1.9 \text{ mg g}^{-1}$). We also
193 found that in both 2011 and 2017, total organic matter contents were significantly higher at
194 stations sampled within seagrass meadows than at non-vegetated areas (both external site and
195 controls).

196

197 **Meiofauna**

198 The number of meiofaunal individuals were lower in 2017 ($868.7 \pm 141.4 \text{ ind. } 10 \text{ cm}^{-2}$) than in
199 2011 ($1381.5 \pm 321.2 \text{ ind. } 10 \text{ cm}^{-2}$), but PERMANOVA did not reveal any significant change
200 (Fig. 3; Table 3). Before the breakwater relocation, the number of meiofaunal individuals
201 ranged between $386.1 \pm 197.8 \text{ ind. } 10 \text{ cm}^{-2}$ (External - B) and $2559.1 \pm 792.6 \text{ ind. } 10 \text{ cm}^{-2}$
202 (CTRL2). In 2017, meiofaunal abundance ranged from $469.1 \pm 89.5 \text{ ind. } 10 \text{ cm}^{-2}$ (CTRL3) to
203 $1301.1 \pm 152.3 \text{ ind. } 10 \text{ cm}^{-2}$ (Internal - A). PERMANOVA tests revealed significant variations
204 in meiofaunal abundance only between the two control stations sampled in 2017 (Table 3).

205 Overall, a total of 14 meiofaunal taxa were encountered (Fig. 4): Nematoda, Kinorhyncha,
206 Polychaeta, Bivalvia, Ostracoda, Acarina, Copepoda, Amphipoda, Cumacea and Isopoda,
207 were found in both sampling years. Oligochaeta and Platyhelminthes were recovered only in
208 2011, whereas Tanaidacea and Tardigrada only 2017. The highest richness value (10) was
209 detected at station B in the internal site in both years, while the lowest (3) was found after the
210 relocation, at station B in the external site. At this station, we registered the loss of Cumacea
211 and Ostracoda, and the presence of Polychaeta, not recovered before the works. Values of

212 higher taxa richness were significantly different between sites (Table 3) and differences were
213 driven by the high values recorded in stations sampled within seagrass meadows.

214 Nematodes and copepods, were the numerically dominant groups of meiofaunal assemblages
215 in both sampling years (in total, >98 and >95 % of the assemblage in 2011 and 2017,
216 respectively) (Fig. 5). All others represented <1% of total assemblages, with the only
217 exception of Polychaetes representing 4% of assemblages after the breakwater relocation, at
218 stations sampled within the seagrass meadows.

219 PERMANOVA based on untransformed data revealed that meiofaunal taxonomic
220 composition significantly varied before and after the breakwater relocation (Table 3). The
221 SIMPER analysis revealed the 43% of dissimilarity in the meiofaunal taxonomic composition
222 between the two sampling years. Variations in nematode and copepod abundance were
223 responsible for the observed dissimilarity percentages. Each taxon contributed to a different
224 extend to the total meiofaunal assemblages in the two years. The contribution of nematodes
225 significantly decreased from 2011 to 2017, being, as average, 97% in 2011 and 81% in 2017,
226 whereas the proportion of copepods (adults and *nauplii*) increased from 2% to 15%. In
227 particular, in 2011, copepods and *nauplii* represented 2 and 0.08%, respectively, whereas their
228 contribution increased in 2017, being 8 and 6%. It is noteworthy to note that the statistical
229 analysis based on presence/absence transformed data, showed significant differences in
230 taxonomic composition between sampling sites. Stations sampled within seagrass meadows
231 significantly differed from those sampled at external site and controls (Table 3). SIMPER
232 analysis revealed that within each sampling year, the highest percentage of dissimilarity
233 occurred comparing external and internal sites (55% in 2011 and 43% in 2017) and the lowest
234 between controls and external site (26% in 2011 and 29% in 2017).

235

236 **DISCUSSION**

237 *Effects of breakwater relocation on sediment characteristics and food availability*

238 Our results showed that breakwater relocation altered sediment features (e.g., by increasing
239 total organic matter content), and that the effects were more evident in the sampling site
240 colonized by seagrass meadows, where total organic matter contents doubled in 2017 when
241 compared with the values determined in 2011. The highest values of total organic matter
242 registered at sheltered site, especially after the breakwater relocation, could be related to the
243 different breakwater design, that have determined an increase in the accumulation of organic
244 matter by reducing hydrodynamic export to the open sea. In addition, the presence of seagrass
245 meadows, whose sediments are known to be richer in organic matter when compared to bare
246 sediments, could have favoured the enrichment found at internal site (Miyajima et al. 1998;
247 Gacia and Duarte 2001; Kennedy et al. 2010).

248 Changes in grain size between sheltered and wave-exposed sites found in both sampling years,
249 are likely to be the result of the altered hydrodynamic conditions due to the presence of the
250 breakwaters (Table 2; Martin et al. 2005; Zyserman et al. 2005; Bertasi et al. 2007). Internal
251 site clearly revealed the predominantly sheltered conditions, with an increased percentage of
252 pelite. The highest amount of pelite may be due to a fine supply not yet completely
253 redistributed by waves and currents coming from mud flows and/or erosion of the pelitic rocks
254 characterizing the area (Semprucci et al., 2017). Conversely, at wave-exposed site and
255 controls, higher water energy favoured unstable and coarser sediments. Similarly, previous
256 studies reported that along the coast of the Adriatic Sea, the reduced sand fraction within the
257 reef suggested reduced turbulence as a direct consequence of the presence of the artificial reef
258 (Danovaro et al. 2002; Semprucci et al. 2010, 2017).

259

260 *Response of meiofauna to breakwater relocation*

261 Coastal defence structures might affect meiobenthic assemblages in different ways, and
262 primarily by: i) altering the hydrodynamic regime and the physical sedimentary
263 characteristics; ii) modifying the distribution and/or composition of the available food
264 sources; and iii) altering the interactions between different components of the foodweb
265 (Danovaro et al. 2002). Limited and contrasting information is available on the response of
266 meiobenthic assemblages to man-made structures. Some studies did not find any decrease in
267 meiofaunal abundance near the artificial reefs (Semprucci et al. 2017). Conversely, Danovaro
268 et al. (2002) documented a significant decrease of the total number of individuals and different
269 taxonomic composition close to the man-made structures.

270 Our results showed that the relocation of breakwaters did not cause any significant effect on
271 meiofaunal abundance and taxa richness, even if we found a lower number of individuals, and
272 the loss of two taxa at external site, after the works. The number of taxa found in our study
273 was comparable with those previously reported for the investigated area (Danovaro et al.
274 2002; Frontalini et al. 2011; Semprucci et al. 2013, 2017). After the breakwater relocation, at
275 external site, we reported the loss of Cumacea and Ostracoda and the presence of Polychaeta.
276 Cumacea are known to be sensitive to disturbance such as organic enrichment (De La Ossa et
277 al. 2011), as well as indicators of eutrophication on soft bottoms (Corbera and Cardell 1995).
278 Ostracods are generally sensitive to pollution and disturbance in marine habitats (Ruiz et al.
279 2005). They respond to pollution-induced environmental changes showing high sensitivity to
280 hypoxic and anoxic conditions (Ruiz et al. 2005). The enrichment in organic matter found
281 after the relocation could have negatively influenced more sensitive taxa, leading to their loss,
282 and favoured the presence of those taxa (i.e., polychaetes) preferring sediments characterized
283 by high organic matter contents (Mirto et al. 2012).

284 Meiofaunal taxonomic composition significantly changed before and after the relocation
285 works, as shown in the output of the MDS analysis (Fig. 6a). As sustained by the SIMPER
286 results, this pattern was primarily driven by the reduction in the relative contribution of

287 nematodes, and the greatly increased number of harpacticoid copepods and *nauplii* at all
288 stations sampled after the relocation. The strong dominance of nematodes as well as the low
289 number of copepods found in 2011 could be related to the generally finer grain size of the
290 sediments, which is considered, a ‘super factor’ in meiobenthic ecology (Semprucci et al.,
291 2010 and references therein). The higher abundances of copepods found after the relocation,
292 could represent a consequence of sediments with larger grain size, which represent a
293 favourable condition for this meiofaunal taxon (Murrell and Fleeger 1989; Semprucci et al.,
294 2010).

295 Our study also highlighted the important ecological role of seagrass meadows, being
296 characterized by higher taxa richness than non-vegetated external site and controls.
297 Multivariate analysis of meiofaunal taxonomic composition based on presence/absence
298 transformed data, showed significant differences between vegetated and non-vegetated sites
299 (Fig. 6b; Table 3). Amphipoda, Isopoda, Oligochaeta, Kynorincha and Tanaidacea were
300 exclusive of stations sampled within seagrass meadows. Some of these taxa, indeed, display
301 habitat preference for the vegetated systems and the colonization /utilization of vegetal debris
302 (Giere, 2009). This is not surprising considering that seagrass cover could have a positive
303 impact on meiofauna, as vegetation can stabilize sediments. The positive impact of seagrasses
304 on meiofauna is also related to their positive effects on organic matter quality and pigment
305 concentration especially in shallow-water ecosystems (Sommerfield 2002; De Troch et al.
306 2001).

307

308 Overall, our study showed that sediment characteristics and meiofaunal variables were altered
309 by the relocation. We found an enrichment in organic matter and the loss of some sensitive
310 taxa after the relocation works. In addition, we pinpointed out a shift in meiofaunal
311 assemblage structure, with an increase of copepods and polychaetes, the latter known to
312 increase in sediments with higher nutrient loads.

313 Our results reported also significantly higher organic matter contents, meiofaunal abundance
314 and richness at sheltered than at the wave-exposed sites. Differences were due to breakwaters
315 modulating wave motion, and the presence of seagrass meadows, attenuating hydrodynamic
316 energy from currents and waves, stabilizing the seabed sediments, and providing habitats,
317 nursing grounds and food for marine animals.

318 Seagrasses guarantee important ecological processes and ecosystem services, some of which
319 of relevant economic value. Therefore, an integrated approach based on careful management
320 and conservation practices should be applied to protect and conserve vulnerable coastal
321 ecosystems and to favour the provision of societal and economic benefits of coastal areas.

322

323 **ACKNOWLEDGEMENTS**

324 This study has been conducted in the framework of the Project MERCES funded from the
325 European Union's Horizon 2020 research and innovation programme under grant agreement
326 No 689518.

327

328 **COMPETING INTERESTS**

329 The authors declare that they have no competing interests.

330 **References**

- 331 Airoldi L, Abbiati M, Beck MW, Hawkins SJ, Jonsson PR, Martin D, et al (2005) An
332 ecological perspective on the deployment and design of low-crested and other hard
333 coastal defence structures. *Coastal Engineering* 52:1073–1087.
- 334 Ambrose RF, Anderson TW (1990) Influence of an artificial reef on the surrounding infaunal
335 community. *Marine Biology*, 107: 41–52.
- 336 Anderson MJ (2001) A new method for non-parametric multivariate analysis of variance.
337 *Austral Ecol.* 26, 32e46.
- 338 Anderson MJ, Gorley RN, Clarke KR (2008) PERMANOVA+ for PRIMER: guide to 484
339 software and statistical methods. PRIMER-E: Plymouth, UK.
- 340 Artegiani A, Bregant D, Paschini E, Pinardi N, Raicich F, Russo A (1997) The Adriatic Sea
341 general circulation part II: baroclinic circulation structure. *J Phys Oceanogr* 27:1515–
342 1532.
- 343 Austen MC, Widdicombe S (2006) Comparison of the response of meio- and macrobenthos
344 to disturbance and organic enrichment. *J. Exp. Mar. Biol. Ecol.* 330, 96e104.
- 345 Balsamo M, Penna A, Semprucci F, Coccioni R, Frontalini F, Scaravelli D, Manti A, Papa S
346 (2011) The coast of the S. Bartolo SCI (Marches, Italy) as a site in a network of Adriatic
347 areas of conservation interest. *Varstvo Narave, Supl.1*, 173-174.
- 348 Barbier EB, Hacker SD, Kennedy C, Koch EW, Stier AC, et al (2011) The value of estuarine
349 and coastal ecosystem services. *Ecological Monographs* 81:169–193.
- 350 Bertasi F, Colangelo MA, Abbiati M, Ceccherelli VU (2007) Effects of an artificial protection
351 structure on the sandy shore macrofaunal community: the special case of Lido di Dante
352 (Northern Adriatic Sea). *Hydrobiologia* 586:277–290.
- 353 Bianchelli S, Buschi E, Danovaro R, Pusceddu A (2016). Biodiversity loss and turnover in
354 alternative states in the Mediterranean Sea: a case study on meiofauna. *Sci. Rep.* 6.

- 355 Buchanan JB, Kain JM (1971) Measurement of the physical and chemical environment. In:
356 Holme NA, McIntyre AD (eds) *Methods for the study of marine benthos*. Oxford
357 Blackwell Scientific Publication, Oxford, pp 30–52
- 358 Cencini C (1998) Physical processes and human activities in the evolution of the Po delta,
359 Italy, *J. Coast. Res.* 14:774–793.
- 360 Clarke KR, Gorley RN (2006) *PRIMER V6: User Manual/Tutorial*. PRIMER-E, Plymouth,
361 UK.
- 362 Coccioni R (ed.) (2003) *Verso la gestione integrata della costa del Monte San Bartolo: risultati*
363 *di un progetto pilota. Quaderni del Centro di Geobiologia, Università degli Studi di*
364 *Urbino ‘Carlo Bo’, Arti Grafiche STIBU, Urbina.*
- 365 Colantoni P, Mencucci D, Nesci O (2004) Coastal processes and cliff recession between
366 Gabicce and Pesaro (northern Adriatic Sea): a case history. *Geomorphology* 62:257-
367 268.
- 368 Corbera J, Cardell MJ (1995) Cumaceans as indicators of eutrophication on soft
369 bottoms. *Scientia Marina* 59:63–69.
- 370 Covazzi A, Pusceddu A, Della Croce N, Danovaro R (2000) Spatial and temporal changes in
371 beach meiofaunal communities of the Ligurian Sea (NW Mediterranean), *Rev. Biol.*
372 *Mar. Oceanogr.* 35:57–64.
- 373 Covazzi Harriague A, Gaozza L, Montella A, Misic C (2006) Benthic communities on a sandy
374 Ligurian beach (NW Mediterranean), *Hydrobiologia* 571:383–394.
- 375 Covazzi-Harriague A, Misic C, Valentini I, Polidori E, Albertelli G, Pusceddu A (2013) Meio-
376 and macrofauna communities in three sandy beaches of the northern Adriatic Sea
377 protected by artificial reefs. *Chem Ecol* 29:181–195.
- 378 Cuadrado DG, Gómez EA, Ginsberg SS (2005) Tidal and longshore sediment transport
379 associated to a coastal structure. *Estuar Coast Shelf Sci* 62:291–300.

- 380 Danovaro R (2010) *Methods for the Study of Deep-sea Sediments, Their Functioning and*
381 *Biodiversity*. CRC Press Taylor & Francis Group.
- 382 Danovaro R, Gambi C, Manini E, Fabiano M (2000) Meiofauna response to a dynamic river
383 plume front. *Mar. Biol.* 137, 359-370.
- 384 Danovaro R, Gambi C, Mazzola A, Mirto S (2002) Influence of artificial reefs on the
385 surrounding infauna: analysis of meiofauna. *ICES JMar Sci* 59:356–362.
386 doi:10.1006/jmsc.2002.1223
- 387 De la Ossa Carretero JA, Del-Pilar-Ruso Y, Giménez-Casalduero F, Sánchez-Lizaso JL (2011)
388 Assessing reliable indicators to sewage pollution in coastal soft-bottom communities.
389 *Environmental Monitoring and Assessment* 184:2133–2149.
- 390 De Troch M, Gurdebeke S, Fiers F, Vincx M (2001) Zonation and structuring factors for
391 meiofauna communities in a tropical seagrass bed (Gazi Bay, Kenya). *J. Sea Res.* 45,
392 4561.
- 393 Fabi G, Spagnolo A (2011) Artificial reefs in the management of Mediterranean Sea fisheries.
394 In: Bortone SA, Brandini FP, Fabi G, Otake S (eds) *Artificial reefs in fisheries*
395 *management*. CRC Press, Boca Raton, pp 167–181.
- 396 Frontalini F, Semprucci F, Coccioni R, Balsamo M, Bittoni P, Covazzi-Harriague A (2011)
397 On the quantitative distribution and community structure of the meio and macrofaunal
398 communities in the coastal area of the Central Adriatic Sea (Italy). *Environ Monit*
399 *Assess* 180:325–344. doi:10.1007/s10661-010-1791-y
- 400 Gacia E, Duarte CM (2001) Sediment retention by a Mediterranean *Posidonia oceanica*
401 meadow: the balance between deposition and resuspension. *Estuar. Coast. Shelf Sci.*
402 52:505–514. <http://dx.doi.org/10.1006/ecss.2000.0753>
- 403 Giere O, (2009). *Meiobenthology: The Microscopic Motile Fauna in Aquatic Sediments*, 2nd
404 edn. Springer Verlag, Heidelberg

405 Giere O, Eleftheriou A, Murison J (1988) Abiotic factors. In: Higgins RP, Thiel H (eds)
406 Introduction to the study of meiofauna. Smithsonian Institution Press, Washington DC,
407 pp 61–78

408 Gray JS (2000). The measurement of marine species diversity, with an application to the
409 benthic fauna of the Norwegian continental shelf. *J. Exp. Mar. Biol. Ecol.* 250, 23-49.

410 Heip C, Vincx M, Vranken G (1985) The ecology of marine nematodes. *Oceanogr. Mar. Biol.*
411 *Annu. Rev.* 23, 399-489.

412 Kennedy H, Beggins J, Duarte CM, Fourqurean JW, Holmer M, Marba N, Middelburg JJ,
413 Marbà N (2010) Seagrass sediments as a global carbon sink: isotopic/ constraints. *Glob.*
414 *Biogeochem. Cycles* 24, GB4026. <http://dx.doi.org/10.1029/2010GB003848>

415 Martin D, Bertasi F, Colangelo MA, de Vries M, Frost M, Hawkins SJ, Macpherson E,
416 Moschella PS, Satta MP, Thompson RC, Ceccherelli VU (2005) Ecological impact of
417 coastal defence structures on sediment and mobile fauna: Evaluating and forecasting
418 consequences of unavoidable modifications of native habitats, *Coast. Eng.* 52:1027–
419 1051

420 McArdle BH, Anderson MJ (2001) Fitting multivariate models to community data: a comment
421 on distance-based redundancy analysis. *Ecology* 82, 290-297.

422 Mirto S, Gristina M, Sinopoli M, Maricchiolo G, Genovese L, Vizzini S, Mazzola A (2012)
423 Meiofauna as an indicator for assessing the impact of fish farming at an exposed marine
424 site. *Ecol Indic* 18:468–476.

425 Miyajima T, Koike I, Yamano H, Iizumi H (1998) Accumulation and transport of seagrass-
426 derived organic matter in reef flat sediment of Green Island, Great Barrier Reef. *Mar.*
427 *Ecol. Prog. Ser.* 175:251–259.

428 Murrell MC, Fleeger JW (1989) Meiofauna abundance on the Gulf of Mexico continental
429 shelf affected by hypoxia. *Cont Shelf Res* 9, 1049–1062

430 Principi M, Pignone R, Tramontana M, Trincardi F, Guerrera F, Martelli L et al (2011) Carta
431 Geologica d'Italia alla scala 1:50.000, Foglio 268, Pesaro. ISPRA, Roma

432 Relini G, Relini M, Palandri G, Merello S, Beccornia E (2007) History, ecology and trends
433 for artificial reefs of the Ligurian sea. Italy. *Hydrobiologia* 580:193–217.

434 Ruiz F, Abad M, Bodergat AM, Carbonel P, Rodríguez-Lázaro J, et al. (2005) Marine and
435 brackish-water ostracods as sentinels of anthropogenic impacts. *Earth-Science Reviews*
436 72:89–111.

437 Santelli A, Punzo E, Scarcella G, Strafella P, Spagnolo A, Fabi G (2013) Decapod crustaceans
438 associated with an artificial reef (Adriatic Sea). *Mediterr Mar Sci* 14:64–75.

439 Semprucci F, Boi P, Manti A, Covazzi-Harriague A et al (2010) Benthic communities along
440 a littoral of the Central Adriatic Sea (Italy). *Helgol Mar Res* 64:101–115.
441 doi:10.1007/s10152-009-0171-x

442 Semprucci F, Frontalini F, Covazzi-Harriague A, Coccioni R, Balsamo M (2013) Meio- and
443 macrofauna in the marine area of the Monte St. Bartolo Natural Park (Central Adriatic
444 Sea, Italy). *Sci Mar* 77:189–199. doi:10.3989/scimar.03647.26

445 Semprucci F, Frontalini F, Sbrocca C, Arminot du Chatelet E, Bout- Roumazelles V,
446 Coccioni R, Balsamo M (2015b) Meiobenthos and free-living nematodes as tools for
447 biomonitoring environments affected by riverine impact. *Environ. Monit. Assess.* 187
448 (5), 1-19.

449 Semprucci F, Losi V, Moreno M (2015a) A review of Italian research on free-living marine
450 nematodes and the future perspectives in their use as ecological indicators (EcoInd).
451 *Mediterr. Mar. Sci.* 16, 352-365.

452 Semprucci F, Sbrocca C, Baldelli G, Tramontana M, Balsamo M (2017). Is meiofauna a good
453 bioindicator of artificial reef impact? *Mar Biodiv* doi: 10.1007/s12526-016-0484-3

- 454 Semprucci F, Sbrocca C, Rocchi M, Balsamo M (2015c) Temporal changes of the meiofaunal
455 assemblage as a tool for the assessment of the ecological quality status. *J. Mar. Biol.*
456 *Assoc. U. K.* 95 (2), 247-254.
- 457 Somerfield PJ, Yodnarasri S, Aryuthaka Ch (2002) Relationships between seagrass
458 biodiversity and infaunal communities: implications for studies of biodiversity effects.
459 *Mar. Ecol. Prog. Ser.* 237, 97-109.
- 460 Thomalla F, Vincent CE (2003) Beach response to shore-parallel breakwaters at Sea Palling,
461 Norfolk, UK. *Estuar Coast Shelf Sci* 56:203–212
- 462 Tramontana M, Raffaelli G, Savelli D, Mattioli M, Ferri L (2005) Sedimentary petrography
463 of upper Messinian sandstones in the coastal area of Northern Marche (Italy). *Ital J*
464 *Geosci (Boll Soc Geol It)* 4:87–93.
- 465 Zavatarelli M, Baretta JW, Baretta-Bekker JG, Pinardi N (2000) The dynamics of the Adriatic
466 Sea ecosystem. An idealized model study. *Deep Sea Res I* 47:937–970.
- 467 Zyserman JA, Johnson HK, Zanuttigh B, Martinelli L (2005) Analysis of far-field erosion
468 induced by low-crested rubble-mound structures. *Coast Eng* 52:977–994.

469 **Figures**

470 **Fig. 1** Control (circles), external (squares) and internal (triangles) stations investigated in the
471 study area of Gabicce Mare, before (2011) and (after) the breakwater relocation.

472 **Fig. 2** Total organic matter in the sediments of the investigated stations sampled before (2011)
473 and after (2017) the breakwater relocation

474 **Fig. 3** Meiofaunal abundance in the sediments of the investigated stations sampled before
475 (2011) and after (2017) the breakwater relocation

476 **Fig. 4** Meiofaunal higher taxa richness in the sediments of the investigated stations sampled
477 before (2011) and after (2017) the breakwater relocation

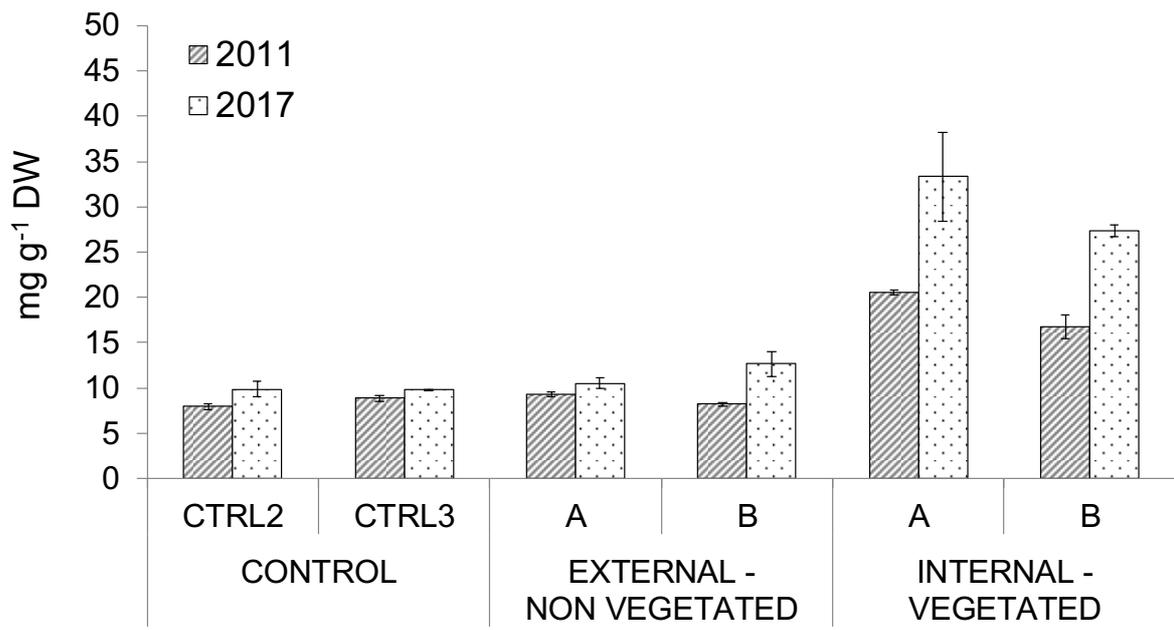
478 **Fig. 5** Meiofaunal taxonomic composition in the sediments of the investigated stations
479 sampled before (2011) and after (2017) the breakwater relocation

480 **Fig. 6** MDS ordination plots illustrating (a) the differences in the composition of meiofaunal
481 assemblages before (2011) and after (2017) the breakwater relocation (untransformed data)
482 and (b) the differences in the composition of meiofaunal assemblages between sampling sites
483 (presence/absence transformed data)



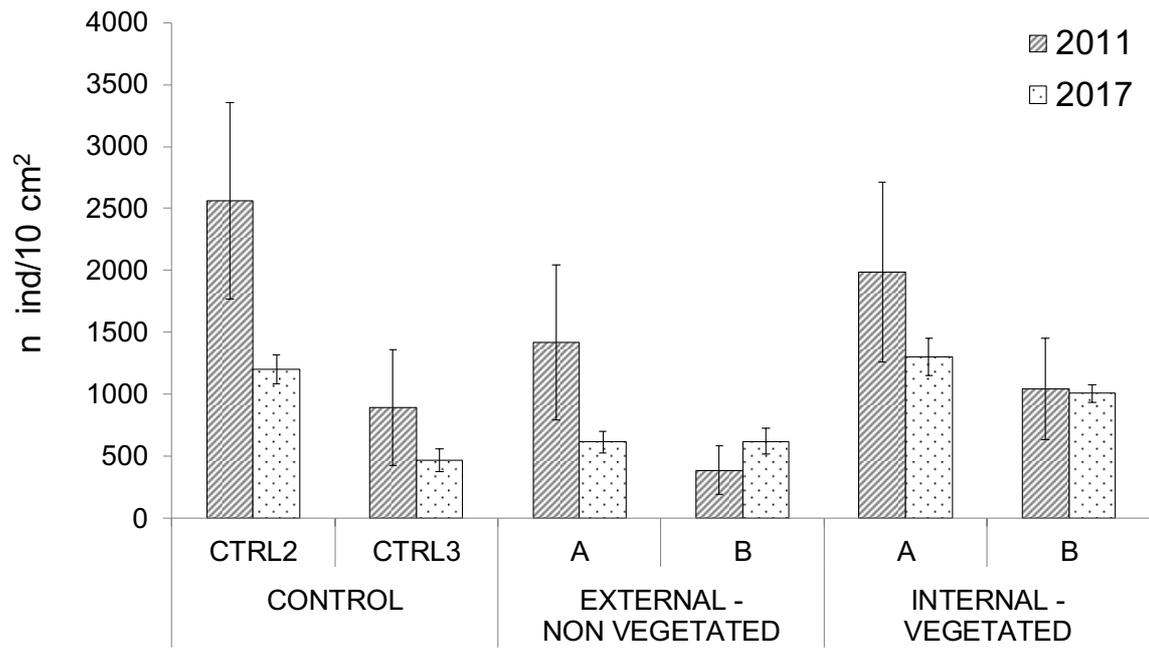
484

485 **Fig. 1**



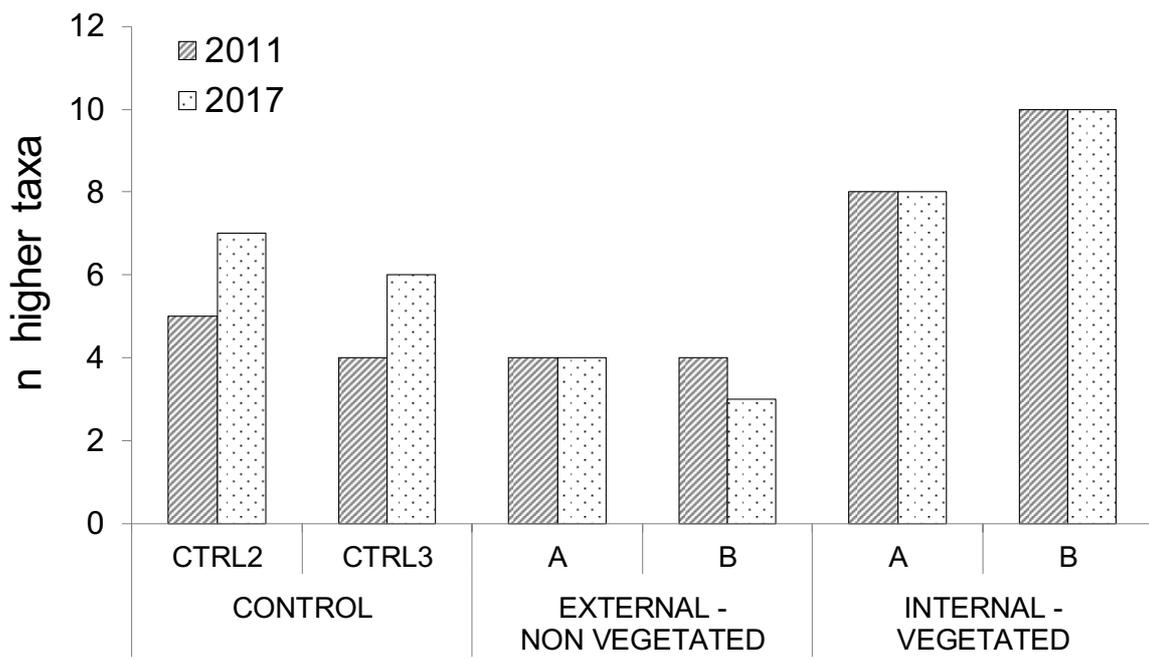
486
487

Fig. 2



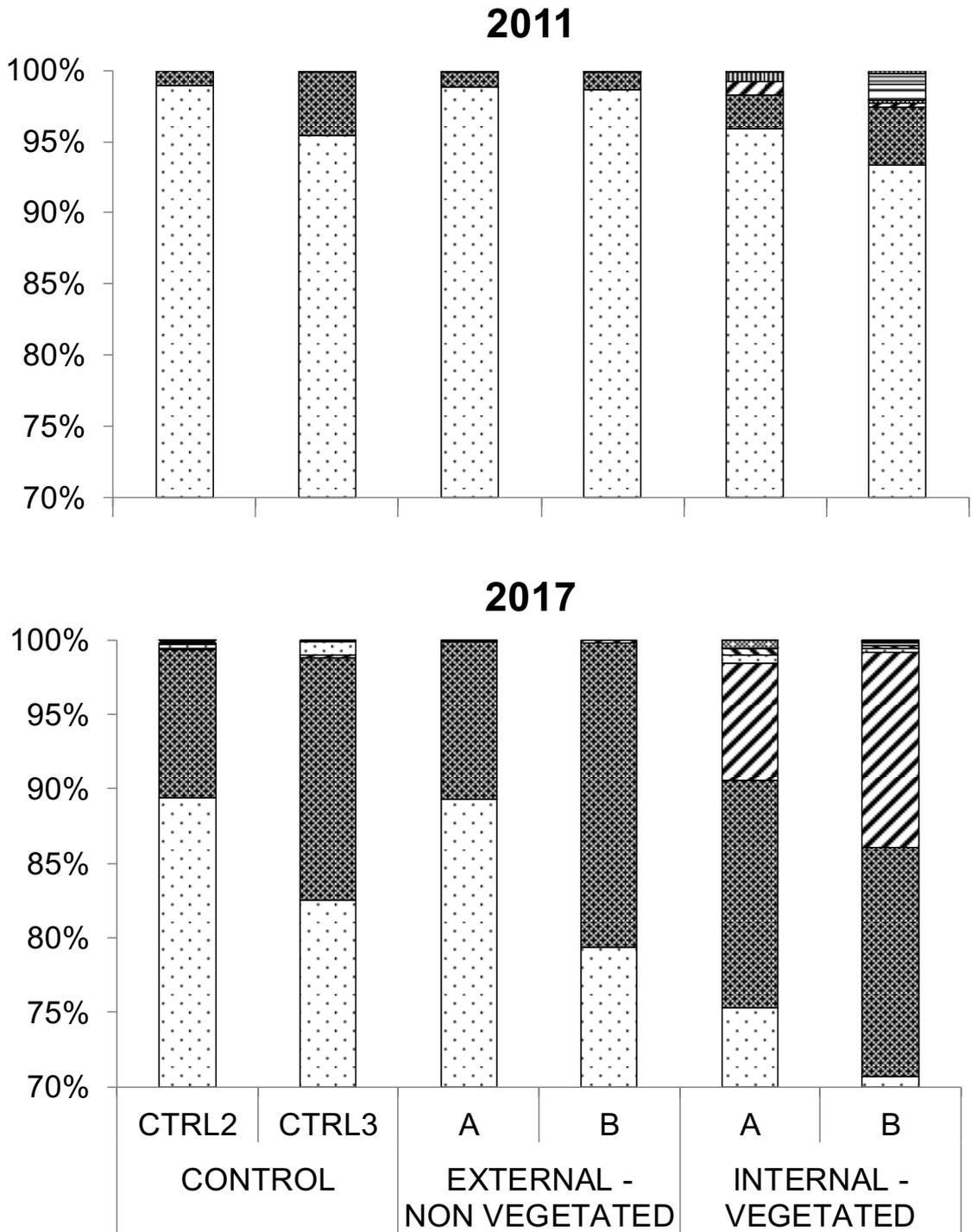
488

489 **Fig. 3**



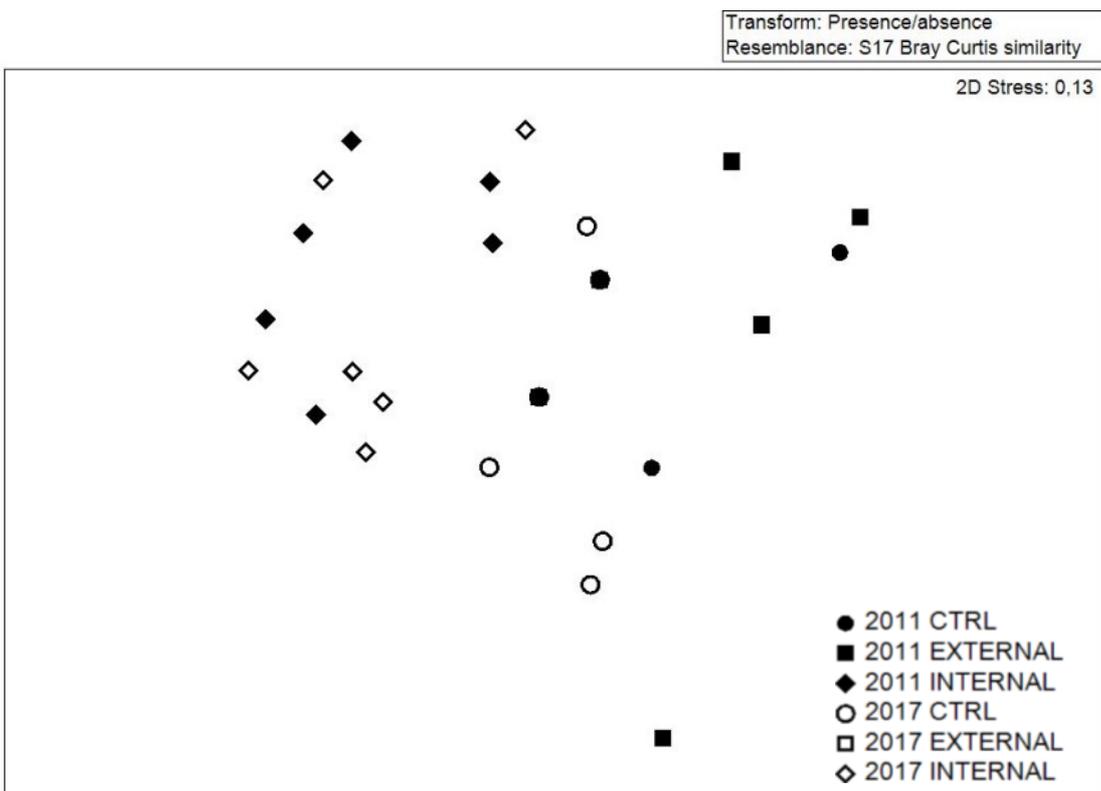
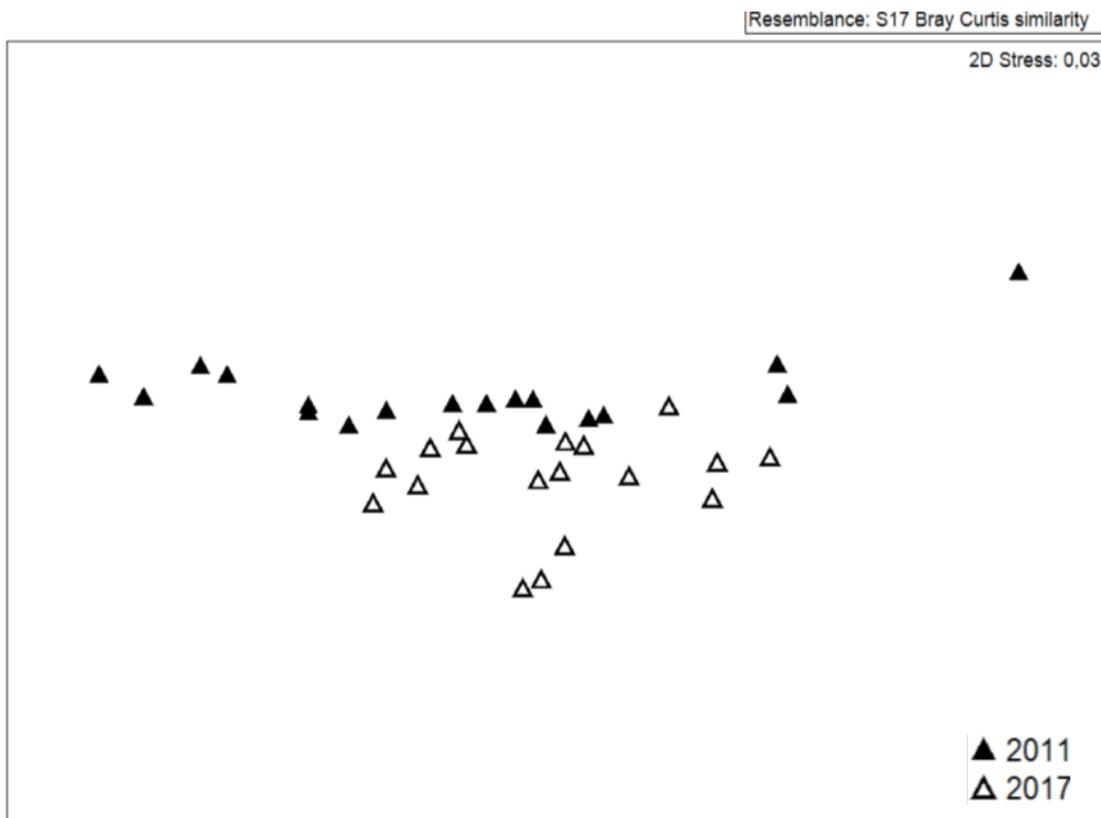
490

491 **Fig. 4**



492

493 **Fig. 5**



494

495 **Fig. 6**

Table 1 Sampling year, site, station, latitude, longitude, depth, grain size, total organic matter, and meiofaunal abundance and richness of taxa.

Year	Site	Station	Latitude	Longitude	Depth	Grain size			Total organic matter	Meiofaunal abundance	Meiofaunal higher taxa richness
						Gravel	Sand	Mud			
			°N	°E	m	%	%	%	mg g ⁻¹	n. ind. 10 cm ⁻²	n
2011	CONTROL	CTRL 2	43° 58.092'	12° 45.952'	3.5	0	96,64	3,36	7,9±0.5	2559,1±1372,8	5
		CTRL3	43° 58.071'	12° 46.132'	2.5	0	97,45	2,55	8,9±0.5	895,4±808.6	4
	EXTERNAL	A	43° 58.072'	12° 45.826'	3	0	97,83	2,17	9,3±0.3	1416,8±1084.5	4
		B	43° 58.059'	12° 46.071'	2.3	0	96,19	3,81	8,2±0.3	386,1±342.5	4
	INTERNAL	A	43° 58.046'	12° 45.831'	1.9	0	80,88	19,12	20,6±0.3	1987,1±1254.2	8
		B	43° 58.005'	12° 46.019'	1.6	0	53,7	46,3	16,8±1.8	1044,4±703.5	10
2017	CONTROL	CTRL 2	43° 58.090'	12° 45.952'	3.5	0,13	99,23	0,63	9,9±1.3	1199,2±201.2	7
		CTRL3	43° 58.071'	12° 46.132'	2.5	0,31	99,27	0,41	9,8±0.0	469,1±155.1	6
	EXTERNAL	A	43° 58.110'	12° 45.858'	2.3	0,06	99,62	0,33	10,5±0.8	617,1±151.7	4
		B	43° 58.073'	12° 46.062'	2.6	0,49	99,30	0,22	12,6±1.9	619,7±183.4	3
	INTERNAL	A	43° 58.046'	12° 45.831'	1.9	0,78	68,82	30,40	28,4±13.9	1301,1±263.7	8
		B	43° 58.005'	12° 46.019'	1.6	5,12	52,55	42,32	27,4±0.9	1006,1±122.2	10

498 **Table 2** Results of PERMANOVA and pairwise test for differences in grain size and sedimentary total organic matter. (df = degrees of freedom; MS
 499 = mean square; Pseudo-F = F statistic; P(MC) = probability levels obtained from Monte Carlo asymptotic distributions). ***=P<0.001; ** = P<0.01;
 500 * = P <0.05; ns = not significant.

Variable	Source	df	MS	Pseudo-F	P(MC)	Pairwise tests
Grain size	Year	1	11,412	0,18187	ns	n.a.
	Site	2	6005,1	10,462	*	Int≠CTRL,Ext
	Station (Site)	3	573,97	37576	***	CTRL2≠CTRL3; IntA≠IntB; ExtA≠ExtB
	Year*Site	2	84,12	1,3405	ns	n.a.
	Year*Station (Site)	3	62,75	4108	***	CTRL2 (2011≠2017); CTRL3 (2011≠2017); IntA and B and ExtA and B (2011≠2017); 2011 (CTRL2≠CTRL3); 2011 and 2017 (IntA≠IntB and ExtA≠ExtB)
	Residuals	12	1,53E-02			
Total organic matter	Year	1	2,4435	62,617	**	2017>2011
	Site	2	8,5446	3,64E+01	**	Int>Ext
	Station (Site)	3	2,35E-01	3,29E+00	ns	n.a.
	Year*Site	2	0,89521	2,29E+01	*	Int (2017>2011); 2011 and 2017(Int>Ext)
	Year*Station (Site)	3	3,90E-02	5,47E-01	ns	n.a.
	Residuals	12	7,13E-02			
	Total	23				

501

502 **Table 3** Results of PERMANOVA and pairwise test for differences in meiofaunal abundance, richness and taxonomic composition (df = degrees of
503 freedom; MS = mean square; Pseudo-F = F statistic; P(MC) = probability levels obtained from Monte Carlo asymptotic distributions). ** = P<0.01;
504 * = P <0.05; ns = not significant.

Variable	Source	df	MS	Pseudo-F	P(MC)	Pairwise tests
Meiofaunal abundance	Year	1	2,37E+06	4,0126	ns	n.a
	Site	2	1,21E+06	0,5813	ns	n.a
	Station (Site)	3	2,08E+06	4,0877	*	Ctrl(Ctrl2>Ctrl3)
	Year*Site	2	3,30E+05	0,55991	ns	n.a
	Year*Station (Site)	3	5,90E+05	1,1591	ns	n.a
	Residuals	24	5,09E+05			
Meiofaunal higher taxa richness	Year	1	3,3611	3,4571	ns	n.a
	Site	2	47,444	48,8	**	Int>Ext
	Station (Site)	3	0,97222	1,0606	ns	n.a
	Year*Site	2	1,7778	1,8286	ns	n.a
	Year*Station (Site)	3	0,97222	1,0606	ns	n.a
	Residuals	24	0,91667			
Meiofaunal taxonomic composition (untransformed data)	Year	1	4807,8	8,387	**	2011≠2017
	Site	2	2106,3	0,74744	ns	n.a
	Station (Site)	3	2818	4,1616	**	Ctrl(Ctrl2≠Ctrl3)
	Year*Site	2	440,55	0,76852	ns	n.a
	Year*Station (Site)	3	573,25	0,84658	ns	n.a
	Residuals	24	677,14			
Meiofaunal taxonomic composition (presence/absence)	Year	1	1187,4	2,376	ns	n.a
	Site	2	5451,2	9,4864	**	Ctrl=Ext; Ctrl and Ext ≠ Int
	Station (Site)	3	574,63	1,2802	ns	n.a
	Year*Site	2	1053,1	2,1075	ns	n.a
	Year*Station (Site)	3	499,72	1,1133	ns	n.a
	Residuals	24	448,86			
	Total	35				