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Benthic impact assessment of a dredge sediment disposal in a dynamic resilient environment

Jean-Claude Dauvin, Noémie Baux, Sandric Lesourd

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31 to improve assessment and minimise the impact of dumping sediment on this benthic
32 habitat.

33

34 **Keywords:** Dredge sediment disposal management; sampling strategy; resilient ecosystem;
35 Benthic communities; Bay of Seine; English Channel

36

37 **1. Introduction**

38

39 The dredging and the disposal of dredge sediment in the coastal marine environment
40 is known to be among the human activities having a significant impact on the structure and
41 functioning of shallow benthic habitats (Bolam et al., 2006a). Nevertheless, the impact is
42 often spatially limited and remains local, being sometimes restricted to the sediment
43 disposal zone itself (Smith and Rule, 2001; Bolam et al., 2006a,b; Marmin et al., 2016).
44 However, among the numerous studies carried out to assess the impact of dredge sediment
45 deposition, three main areas can be defined: a zone corresponding to the area most
46 impacted by disposal, surrounded by a second zone corresponding to the area influenced by
47 sediment disposal where sediment and macrofaunal characteristics are slightly modified,
48 and a third zone more or less distant from the sediment disposal sites where the impact
49 cannot be adequately evaluated (Pezy et al., 2018; Baux et al., 2019, 2020).

50 As with other assessments of the impact of human activities in marine ecosystems,
51 the challenge of research on dredge sediment deposition is to validate a sampling strategy
52 able to identify the real effect of this activity, i.e. the evolution of an area influenced by
53 dredged sediment deposition is significantly different from the control zone. In fact, the
54 sampling of benthic habitats which are the most impacted by anthropic activities should be
55 adequate to distinguish their effect from other pressures on marine environments. This is
56 especially the case in the coastal zone where there are cumulative human activities, and
57 temporal changes of benthic communities are occurring in relation to Climate Change
58 (Hiscock et al., 2004; Schiel et al., 2004; Dauvin, 2008, 2019; Hewitt et al., 2016). Many
59 impact studies on these anthropogenic pressures nowadays adopt a BACI approach (Before
60 After Control Impact)(Underwood, 1994; Simonini et al., 2005; Ware et al., 2009; Pezy et al.,
61 2017, 2018). It is often difficult to apply this approach strictly, notably because the duration
62 of deposition in marine environments often exceeds several decades. The 'Before' state is

63 often available, because it is currently required to carry out a reference ecological study of
64 an area before authorizing a new dumping site; conversely, the 'After' state is often lacking
65 because dumping in the study site is still active (Baux, 2018; Baux et al., 2019). For dredge
66 sediment disposal, the experimental Machu dumping site in the eastern part of the Bay of
67 Seine (Fig. 1) is a textbook case, with a real and robust BACI approach (Marmin et al., 2014,
68 2016, Pezy et al., 2017, 2018; Dauvin et al., 2018; Baux et al., 2020).

69 In most cases, to evaluate the impact of sediment deposition on benthic habitats, the
70 sampling strategy is currently established by comparing the macrofauna collected at stations
71 located within the dumping site with stations located at a sufficient distance away to be
72 unaffected by sediment deposition (Olenin, 1992; Smith and Rule, 2001; Ware et al., 2009;
73 Bolam et al., 2006a,b). Moreover, the administrative authorities charged with authorizing
74 disposal at sea must also approve the sampling strategy, but their opinion is guided by
75 compliance with National and European regulations and statements a. These authorities do
76 not take into account scientific ecosystem approaches (Marmin et al., 2014; GEODE, 2014).

77 The eastern part of the Bay of Seine is known to be one of the areas more impacted
78 by human activities in the English Channel as well as in the North Sea – English Channel
79 system (Dauvin, 2019; Dauvin et al., 2020). The maritime traffic of both Rouen and Le Havre
80 harbours and their access to basins and channels means that it is necessary to dredge large
81 volumes of sediment mainly composed of mud and sand (Marmin et al., 2014, 2016).
82 Sediments dredged from Le Havre Harbour have been disposed on the Octeville disposal site
83 since 1947 (Fig. 1)(Baux et al., 2019, 2020).

84 During an experimental period from August 2016 to September 2017, a study was set up
85 to identify the spatio-temporal changes of the macrobenthos collected at ten stations and
86 on six dates: three impacted stations with different volumes of dumped sediments, two
87 influenced stations located within the Octeville site and five stations outside the disposal site
88 (north and south zones).

89 This study aims to compare the spatio-temporal changes of benthic habitats (in
90 impacted, influenced and unimpacted zones) during the 2016-2017 experimental period and
91 assess the impact of sediment dumping. We also propose recommendations on the future
92 long-term strategy to improve assessment and minimise the impact of sediment disposal on
93 benthic habitats.

94

95 **2. Materials and methods**

96

97 2.1. *Study site and main characteristics of the deposit*

98

99 The study area is located along the French coast of the English Channel, in the
100 eastern part of the Bay of Seine, between the mouth of the Seine estuary and Antifer
101 harbour (Fig. 1). The site is characterized by a macrotidal regime, with a tidal range of 7.5 m
102 during spring tide and a mean water depth of 15 m (with a maximum of 30 m). Tidal currents
103 flows (mean value of current speed: $1\text{m}\cdot\text{s}^{-1}$) mainly to the northeast (NE) during flood tide
104 and to the southwest (SW) during ebb tide (Baux et al., 2019). The Seine estuary is a major
105 source of freshwater and fine particles in the eastern part of the Bay of Seine. The average
106 annual discharge of the Seine is estimated at $435\text{ m}^3\text{ s}^{-1}$, rising to $534\text{ m}^3\text{ s}^{-1}$ over the last
107 decade (highest average annual discharge: $903\text{ m}^3\text{ s}^{-1}$ in 2001. The average input of fine
108 particles supplied by the Seine is $6\text{ }10^5\text{ t y}^{-1}$ (ranging between $1.3\times 10^5\text{ t y}^{-1}$ and $1.7\times 10^6\text{ t y}^{-1}$;
109 Lesourd et al., 2016). The disposed mud is unconsolidated and thus easily resuspended by
110 tidal currents (Lesourd et al., 2003, 2016).

111 The Octeville dumping site was used by the Le Havre port authority (GPMH) just after
112 the end of the Second World War in 1947 to dispose of heterogeneous materials resulting
113 from the destruction of the Le Have city during Operation Overlord in 1944 (Fig. 1). At that
114 time, the site was chosen for its proximity to the harbour (located 9 km from the harbour
115 entrance and 2 km offshore for the south-eastern part of the dumping site) and its strong
116 hydrodynamic regime (maximum speed of current of $2\text{ m}\cdot\text{s}^{-1}$), which leads to the natural
117 dispersal of fine particles. Since 2006, following the harbour extension works (Port 2000) the
118 dredged sediment represents approximately $2\text{--}2.5\text{ }10^6\text{ m}^3\text{ y}^{-1}$ (Baux et al., 2019). Before the
119 construction of Port 2000, the dredged volumes were lower: $1.3\text{ }10^6\text{ m}^3\text{ y}^{-1}$. Nowadays, mud
120 ($< 63\text{ }\mu\text{m}$) represents 80% of the disposed sediment, and the strong hydrodynamic regime
121 causes the resuspension of finer particles which are dispersed outside the dumping area
122 over large distances mainly towards the north-east, but also with an lower flux directed
123 towards the south (Baux et al., 2019). In the dumping site, which covers a surface-area of 14
124 km^2 , the seabed is covered by muddy sand and sandy mud in the northern and western
125 sectors, while gravel and pebbles occur in the south-eastern sector (Baux et al., 2019). The
126 SE sector of the site corresponds to a historical deposition area associated with a shoal (-5 m

127 with respect to minimum sea level in Le Havre harbour= Cote Marine du Havre, CMH). The
128 dumping strategy has changed over time to minimize the impact of sediment deposition on
129 the benthic habitats (Baux, 2018; Baux et al., 2019). At present, sediments are disposed in
130 the N, NW, and SW parts of the dumping site, according to a system of “500x500 m boxes”,
131 in which the sediment is distributed in a thin layer (between 0.2 and 0.6m thick per box per
132 year) (Baux et al., 2019).

133 From 2010 to 2017, the mean water depth measured by multibeam echosounder at
134 the dumping site was 13.9 m.

135 Outside the dumping site in the studied zone (Fig. 1), the seafloor is mainly composed
136 of sandy mud and muddy sand, with clean sand offshore and mixing with gravel, pebbles and
137 sands nearer the coast (Baux, 2018). Baux et al. (2019) carried out a geochemical study to
138 highlight the sedimentary dynamics of the disposed dredged sediment and concluded the
139 dispersal zone of dumped sediment was lower than 10 km. The NE part of the studied zone
140 is characterized by a state of sedimentary dynamic equilibrium, unaffected by sediment
141 dumping.

142

143 *2.2. Sampling strategy*

144

145 During the experimental period August 2016 to September 2017, benthic samples
146 were collected at ten stations: five in the dumping site (impacted and influenced zones) and
147 five outside the dumping site (northern and southern zones) and on six dates: T0: 17 August
148 2016, T1: 11 October 2016, T2: 19 December 2016, T3: 3 April 2017, T4: 19 June 2017 and
149 T5: 18 September 2017 (Fig. 2). In the dumping site, three stations were located in boxes
150 where dredge sediment deposition corresponded to impacted stations (HK, GJ and IJ); low
151 volumes were disposed inadvertently within the box at station ND102 at the beginning of
152 the survey, but dumping ceased in April 2017. Two stations (ND102 and IL) within the
153 dumping site are considered as being within the influenced zone (Fig. 1); moreover, during
154 the survey period, no sediment was disposed in the box at station IL, but sediment was
155 disposed on neighbouring boxes (Fig. 2). The volume of sediment disposed in the boxes of
156 three other stations within the dumping site varied between 41,000 to 186,000 m³ (Fig. 2).
157 Three stations were located in the northern zone of the dumping site (R21, R23 and R21),

158 while two other stations were located in the south zone (NP101 and P3); all these five
159 stations were situated outside the dumping site.

160 At all ten stations and at each date, the macrofauna was sampled using a 0.1 m² Van
161 Veen grab. Four replicates were taken from each station, one for the sediment
162 characteristics and three for the macrofauna which were sieved on a 1 mm circular mesh.
163 The sediment was fixed in a 10% solution of formaldehyde in seawater neutralized with
164 sodium tetraborate.

165

166 2.3. Laboratory analysis

167

168 In the laboratory, organisms were sorted, identified to species level when possible, and
169 then counted. The World Register of Marine Species consulted in January 2020 was used to
170 harmonize species names (<http://www.marinespecies.org/>).

171 To study the differences in abundance in relation to sediment deposition pressure and
172 recruitment periods at the ten stations during the six sampling periods, size measurements
173 were carried out on the four dominant species, i.e.: two bivalves [*Abra alba* (W. Wood,
174 1802) and *Phaxas pellucidus* (Pennant, 1777)] and two polychaetes (*Lagis koreni* Malmgren,
175 1866 and *Owenia fusiformis* Delle Chiaje, 1844).

176 For the bivalves *Abra alba* and *Phaxas pellucidus*, the total length of the antero-
177 posterior axis was measured using a calliper (precision of 0.1 mm) (Dauvin, 1986; Baux,
178 2018). For *A. alba*, the maximum total length was 20 mm and individuals with a total length
179 <5mm were considered as juveniles, while for *P. pellucidus*, the maximum total length was
180 30 mm and individuals with a total length < 8 mm were considered as juveniles.

181 For *Lagis koreni*, the cephalic diameter (CD) was chosen as an allometric variable
182 according to Elkaim and Irlinger (1987). The measurements were made using a binocular
183 microscope with an ocular micrometre (precision of 0.08 mm). The maximum CD was 7.5
184 mm and individuals with a CD <2 mm were considered as juveniles. For *Owenia fusiformis*,
185 the length of the fourth segment (I4) was chosen as an allometric variable according to
186 Ménard et al. (1989). The measurements were made using a binocular microscope with an
187 ocular micrometer (precision of 0.08 mm). The maximum CD was 20.4 mm and individuals
188 with a length I4 < 4mm were considered as juveniles.

189 Analyses of 100-500 g of sediment were performed using an AFNOR sieve column
190 with seven sieving mesh size to identify seven size classes: gravel >2 mm; very coarse sand
191 >1-2 mm; coarse sand >0.500-1 mm; medium sand >0.250-0.500 mm; fine sand >0.125-
192 0.250 mm; very fine sand >0.063-0.125 mm and mud < 63 μ m.

193 Total Organic Carbon (TOC) was analysed on superficial sediment collected by the
194 grab. Frozen samples were freeze-dried. The coarse fraction was removed with a 2-mm
195 sieve. Then, sediment samples were acidified by H₃PO₄ (1 M) to remove carbonates, dried
196 on a hot plate at 40 °C and measured for TOC content by combustion in a LECO CS 744
197 carbon sulphur analyser. Two or three replicates of dried and homogenized sediment (50
198 mg) were analysed per sample. The results of organic carbon measurements are expressed
199 in percentage of dry sediment. The precision determined from replicate subsampling was \pm
200 0.05% organic carbon (Baux et al., 2019).

201

202 *2.5. Statistical analysis*

203

204 The Shannon-Wiener diversity index H' (\log_2) and Pielou's evenness index J' were
205 calculated for the 10 stations. The biotic indices AMBI and BO2A were also calculated to
206 assess the ecological status of the macrofauna according to the thresholds proposed by
207 Borja et al., 2000; Muxika et al., 2007 and Dauvin, 2018).

208 To validate differences and associated significance levels, several statistical methods
209 were used after testing the normality of the data using the Shapiro-Wilk test (Shapiro and
210 Wilk, 1965). Kruskal-Wallis (KW) tests were performed to compare differences between
211 stations, campaigns and areas in terms of abundance and taxonomic richness. This KW
212 analysis, which is based on median comparisons, also tests the null hypothesis (H₀)
213 according to which the samples belong to the same population. When hypothesis H₀ is
214 rejected, statistically significant differences exist between the samples. In this case,
215 statistical tests of multiple comparisons (KWmc) were carried out with the R 3.3.3 software
216 (package 'pgirmess'). In addition, Spearman correlation coefficients (r) were calculated, using
217 R 3.3.3 software, between dumped volumes and abundances as well as taxonomic richness
218 at stations impacted by sediment deposition.

219 Moreover, a two-way ANOVA with interaction was used to test temporal changes on the
220 Taxonomic and the taxa abundances with a transformation of $\text{Log}(x+1)$ between zones and

221 dates while a one-way ANOVA was used for stations. Prior to each ANOVA, a Shapiro-Wilk
222 test and a Bartlett test for homogeneity of variances were performed. The Tukey Honestly
223 Significant Difference test was applied when ANOVA showed significant differences. The R
224 software package was used to perform ANOVA as well as the Shapiro, Bartlett and Shapiro
225 tests.

226 Data analysis was performed by non-metric Multidimensional Scaling ordination
227 (nMDS), and Hierarchical Ascendant Classification (HAC) created using group averages linking
228 with the Bray-Curtis similarity measure using the PRIMER-6 software package (Plymouth
229 Routines in Multivariate Ecological Research). $\log_{10}(x + 1)$ transformed abundances (0.3 m^{-2})
230 were used to down-weight the importance of very abundant species. All the taxa were taken
231 into account in the analyses and a unique matrix was created (mean average of the six
232 campaigns). To identify those species within different groups that primarily account for the
233 observed assemblage differences, SIMPER (SIMilarity PERcentage) routines were performed
234 using a decomposition of Bray-Curtis similarity on log-transformed abundance data (Clarke
235 and Gorley, 2006).

236

237 **3. Results**

238

239 *3.1. Changes of sediment composition and Total Organic Carbon (TOC)*

240

241 In the study area covering the dumping zone, there were mixed sediments composed
242 of mud, sand and coarse deposits including coarse sand and gravels. There were few
243 temporal changes at the unimpacted stations, both to the North and the South of the
244 dumping zone (Table 1). At these stations, fine particles ($< 63 \mu\text{m}$) made up between 17 %
245 (R43) and 31 % (R21) of the sediment, with a low Standard Deviation (SD). Similarly, in the
246 Influenced zone, the relative percentage of fine particles showed moderate values (16 and
247 24%) with low SD (Table 1; Fig. 3). Conversely, the percentage of fine particles was higher in
248 the impacted stations (39-42%) with a high SD (10-19%) reflecting the disposal of fine
249 particles at these stations as well as the temporal changes. The statistical analysis (Kruskal-
250 Wallis test) reveals significant differences in percentage of fine particles between stations (p -
251 value < 0.001) but not between different campaigns (p -value = 0.99).

252 Temporal changes were highly variable between the three impacted stations of the
253 dumping site (Fig. 3). Station GJ showed an increase in the percentage of fine particles
254 settling out after the T4 campaign (June 2017) reaching more than 60%, while the low levels
255 observed at the beginning of the study continued through T1 to T5. For station IJ, where
256 dredge sediment disposal were dumped only once at the beginning of the experiment, the
257 proportion of fine particles remained relatively stable throughout the monitoring (between
258 45 and 51%), except at T2 (December 2016) when the proportion of fines was lower (25%).
259 The third impacted station HK showed a high percentage of fine particles at T0 (>50%), after
260 which it remained stable (around 30 %) during the rest of the study (T1 to T5).

261 The percentage of TOC was low at unimpacted and influenced stations (0.40 to
262 0.80%), but higher at the three impacted stations (1.19 to 1.33%, i.e. 1.21% TOC on average
263 for these stations). The statistical analysis (Kruskal-Wallis test) revealed significant
264 differences in TOC between stations (p -value < 0.001) but not between different campaigns
265 (p -value = 0.99). The TOC was relatively high at the disposal site stations (GJ, HK and IJ)
266 regardless of the campaigns (Table 1). At influenced stations (ND102 and IL), TOC was much
267 lower, with an average of 0.48%. In the area north and south of the disposal site, the TOC
268 value was intermediate, with an average of 0.62% in the northern zone and 0.74% in the
269 southern zone.

270

271 3.2. Faunal composition

272

273 A total of 139,029 individuals and 171 taxa were collected at the 10 stations during
274 the 2016-2017 sampling period. The polychaetes dominated the Taxonomic Richness with 73
275 taxa (43 % of the total taxa), then the crustaceans with 60 taxa (35%), including 23
276 Amphipoda and 24 Decapoda taxa, as well as 21 Mollusca taxa (12%), and 7 Echinodermata
277 taxa, along with 10 miscellaneous taxa.

278 The ten most abundant species were the polychaetes *Owenia fusiformis* (19% of the
279 collected individuals), *Ampharete baltica* Eliason, 1955 (6.7%), *Kirkegaardia dorsobranchialis*
280 (Kirkegaard, 1959) (4.8%), *Melinna palmata* Grube, 1870 (4.8%), *Nephtys hombergii* Savigny
281 in Lamarck, 1818 (3.5%), *Pholoe baltica* Örsted, 1843 (2.9%), the bivalves *Abra alba* (15.6%),
282 *Phaxas pellucidus* (12.7%) and *Nucula nitidosa* Winckworth, 1930 (4.7%), and the amphipod

283 *Ampelisca tenuicornis* Liljeborg, 1856 (3.4%). These ten dominant species represented 78.1%
284 of the collected individuals.

285

286 3.3. Variations of Taxonomic Richness

287

288 The TR for 0.3m² varied from a minimum of 19 taxa at the impacted station HK in
289 June 2017 to 63 taxa at the influenced station ND 102 in October 2016 and September 2017
290 (Fig. 4). The TR at the five reference stations remained higher 40 all along the study except
291 for the south station NP101 in August 2016, December 2016 and April 2017, the station R43
292 in August 2016 and R21 in December 2016 where TR was included between 20 and 40. The
293 three impacted stations (HK, GJ and IJ) showed lower TR values than the other stations
294 throughout the 2016-2017 survey, except in September 2017 when the TR at both impacted
295 stations GT and IJ was of the same order of magnitude as at the other seven stations. In the
296 dumping zone, where dredged sediment was disposed every three months at station GJ, the
297 taxa remained particularly poor throughout the experiment (between 19 and 38 taxa); this
298 was also the case for station IJ, where dumping took place at the beginning of the
299 experiment (between 28 and 40 taxa). The taxa at station HK was a little richer (between 21
300 and 50 taxa), especially in June (50 taxa). The influenced stations in the dumping zone
301 (ND102 and IL) showed higher TR values, with between 41 and 63 taxa recorded (63 taxa
302 being the maximum).

303 The Kruskal-Wallis (KW) tests showed significant differences between stations (p -
304 value < 0.001) and between zones (impacted stations, influenced stations, North Stations and
305 South stations; p -values < 0.001). The differences between campaigns were small (p -value=
306 0.079). The multiple comparison statistical test (KWmc) carried out on the four zones
307 revealed significant differences between stations receiving experimentally dumped sediment
308 and the influenced stations, as well with stations in the northern zone and southern zones
309 (p -value < 0.05).

310 ANOVA analyses confirmed the significant differences between the dates of the
311 campaigns, the zones but neither between the station nor the interaction between dates
312 and zones (Table 2).

313

314

315 *3.4. Variations in abundance*

316

317 Very low abundances were observed at T0 (August 2016), contrasting with values
318 found during the same summer period in September 2017 (T5). As expected, the winter and
319 early spring periods (T2 and T3) were associated with lower abundances than during the
320 other periods (Fig. 5). The impacted stations were poorer in abundance than the other
321 stations, with station GJ having the lowest abundances (from T0 to T5: 820, 2,050, 2,026,
322 930, 1,070 and 420 ind.0.3m⁻²). High abundances were found in late summer, October 2016
323 and September 2017, with an increase as early as June 2017. Moreover, there were large
324 temporal variations. For example, station GJ showed higher abundances at T1 and T2,
325 compared to stations HK and IJ (i.e. 615 and 608 ind.0.3 m⁻² at GJ; 434 and 561 ind.0.3m⁻² at
326 HK; 362 and 339 ind.0.3m⁻² at IJ), while this trend was reversed in September 2017 (T5), with
327 marked differences in abundance between stations and very low values at GJ (126 ind.0.3m⁻²).
328

329 The KW statistical tests showed significant differences between stations (p-value=
330 0.016), between campaigns (p-value= 0.001) and between zones (p-value= 0.007), with
331 differences (KWmc multiple comparison test) between the impacted and influenced stations
332 in the dumping site and between the impacted stations and stations in the northern zone.

333 ANOVA analyses confirmed the significant differences between the dates of the
334 campaigns, the zones and the stations but not for the interaction between dates and zones
335 (Table 2).

336

337 *3.5. Ecological Quality Status*

338

339 AMBI was calculated for each station in each campaign (Table 3). Overall, ecological
340 status was defined as high or good. The AMBI values were low (1.73 on average), indicating a
341 very slightly disturbed environment. The proportion of group IV species (first-order
342 opportunists) was low (on average, about 10%) and very few group V species were observed
343 (second-order opportunists; 0.035% on average). The taxa were mainly distributed between
344 ecological groups I, II and III (on average 31, 34 and 26%, respectively), i.e. between taxa that
345 were sensitive, indifferent or tolerant to enrichment in Organic Matter.

346 The BO2A index yielded an ecological diagnosis defined as Excellent or Good for all
347 stations and campaigns (Table 3). Station NP101 (southern zone), showed higher values, due
348 to high abundances of opportunistic polychaetes compared to the other stations (averages
349 per campaign, 217 ind.0.3m⁻² on NP101 as against 21 ind.0.3m⁻² to 81 ind.m⁻² at the other
350 stations). In addition, amphipods were not very abundant on this station (on average, 15
351 ind.0.3m⁻²).

352

353 3.6. Spatial patterns of the macrofauna

354

355 At a similarity of 76% (which was very high, reflecting the close similarities between
356 the ten stations indicating that they all belong to the same benthic community), four
357 assemblages could be identified (Fig. 6). The first assemblage corresponded to the three
358 impacted stations (GJ, HK and IJ), the second assemblage to the influenced stations (IL and
359 ND102), while the two other assemblages included four stations, three in the northern zone
360 (R21, R23 and R 43) and one in the southern zone (P3), and the last assemblage
361 corresponded to a single station (NP01) located in the southern zone outside the dumping
362 zone.

363 A SIMPER analysis was carried out to identify the species responsible for the
364 separation of the assemblages and their contribution to the faunal composition. Table 3
365 reported the results for the ten most dominant species (contribution in percentage; and
366 mean abundances calculated for the four ecological groups). The polychaetes *O. fusiformis*,
367 *M. palmata* and *P. baltica* and the bivalve *P. pellucidus*, were present in all four assemblages,
368 while three other species, two polychaetes *N. hombergii* and *K. dorsobranchialis* and the
369 bivalve *S. subtruncata*, were present in three of the four assemblages.

370 *O. fusiformis* was largely dominant in the first assemblage on dumping stations, while
371 *A. alba* was the second most abundant species. *P. pellucidus*, *O. fusiformis* and *A. alba* were
372 dominant species in the second assemblage. Similar species dominated in the third
373 assemblage as observed in the second assemblage, but in a different order: *A. alba* was the
374 first *O. fusiformis* the second and *P. pellucidus* the third, while the amphipod *A. tenuicornis*
375 showed a noticeable abundance (Table 4). Lastly, the fourth assemblage (station NP101)
376 showed a very high abundance of the bivalve *P. pellucidus* and the polychaete *K.*
377 *dorsobranchialis*, while *O. fusiformis* was also highly abundant.

378

379 4.6. Relations between volume of disposed sediment, Taxonomic Richness and Abundance

380

381 Fig. 7A and 7B showed the Spearman correlations with abundance and taxonomic
382 richness, for the four stations with deposition of dredge sediment and the six other stations
383 without sediment deposition (denoted NC). The correlations were negative (respectively $\rho = -$
384 0.24 and $\rho = -0.46$). The stations impacted by sediment deposition were located in the lower
385 right part of the two graphs (A and B), except for station ND102 which was only slightly
386 affected by dumping during the experiment. These results indicated that the taxonomic
387 richness and abundance were negatively correlated with the disposed volumes.

388 Fig. 7 C and 7D presented the same dataset but in terms of average values, which
389 allowed a visualization of the impact gradient according to the amount of sediment
390 disposed. Station ND102 was the least heavily affected in terms of disposed volume and
391 showed the highest taxonomic richness and abundance; conversely, station GJ (the most
392 heavily impacted), showed the lowest values of taxonomic richness and abundance.

393

394 4.7. Size-distribution of dominant species collected during the experiment

395

396 *Abra alba* was a very abundant species at all stations (6 337 individuals collected),
397 except for NP01; the abundances at both impacted stations GJ and HK were among the
398 lowest (Table 5). For the ten stations, the abundances were minimal from December 2016 to
399 April 2017 and remained at the same order of magnitude for the four other dates (Table 5).
400 The KW statistical tests showed significant differences between dates (p -value= <0.01), but
401 no differences between zones (p -value= 0.507). There was synchronous spring recruitment
402 throughout the area, with numerous juveniles in April and June 2017; some juveniles were
403 also present in October-December, indicating a minor settlement period in autumn.

404 A total of 3 629 *Phaxas pellucidus* individuals were sampled, with a minimum in June
405 2017 and a maximum in September 2017 (Table 4). *P. pellucidus* had a low abundance at the
406 three impacted stations; this species preferred sandy sediment habitats such as found to the
407 south of the deposition zone (ND101 and ND102), and showed moderate abundance at the
408 three northern stations. The recruitment began at the beginning of the summer in June to
409 reach a peak at the end of the summer (October 2016 and September 2017). The KW

410 statistical tests showed significant differences between dates (p-value= 0.023), but no
411 differences between zones (p-value= 0.133).

412 *Lagis koreni* showed moderate abundance (1091 individuals collected) with a
413 minimum in April 2017 (40 individuals) and maxima in August 2016 (247) and September
414 2017 (236) (Table 5). Station NP101 in the south of the dumping zone showed the highest
415 abundance, while the nearby station NP102 in the influenced zone showed the lowest
416 abundance (Table 5). There was spring recruitment in 2017 at all stations except both
417 impacted stations GJ and IJ. The KW statistical tests revealed significant differences between
418 dates (T2-T4 and T3-T4; p-value< 0.001), but no differences between zones (p-value= 0.832).

419 *Owenia fusiformis* was abundant (4 987 individuals collected) at most of the stations
420 except at NP101 in the south of the dumping zone, and at the impacted station GJ (Table 5).
421 The total abundance in the ten stations was low at T0 and reached a maximum in September
422 2017, with a second peak in December 2016. The settlement period occurred in autumn and
423 at the end of the summer with a high percentage of juveniles in October-December 2016
424 and in September 2017. The KW statistical tests showed significant differences between
425 dates (T0-versus other dates; p-value< 0.001), as well as between zones (north/south,) and
426 between impacted and influenced stations/south zone (p-value< 0.001).

427

428 **4. Discussion**

429

430 The *Abra alba-Lagis koreni* muddy fine-sand community is associated with the faunal
431 composition of the four assemblages (with a high similarity >60% between assemblages in
432 the Figure 6) and occurs on a heterogeneous silty substrate present throughout the study
433 area. According to the EUNIS classification, the presence of *Nephtys hombergii*, *Owenia*
434 *fusiformis* and *Spisula subtruncata* corresponds to the A5.244 habitat, which is widely
435 described in the eastern part of the Bay of Seine in the English Channel (Thiébaud et al.,
436 1997; Baffreau et al., 2017; Dauvin et al., 2017; Bacouillard et al., 2021). This area has been
437 exposed to numerous anthropogenic impacts since the 19th century related to the
438 development of industrial activities and the continued need to ensure and improve maritime
439 traffic with an increase of activities since the 1950s: harbour extension work and an increase
440 in dredging and dumping of dredged sediment, Normandy bridge construction, aggregate
441 extraction and reduction of the intertidal area (Dauvin et al., 2020).

442 Moreover, in spite of its exposure to multiple stressors, the *Abra alba-Lagis koreni*
443 muddy fine sand community and its assemblages have been persistent in terms of species
444 composition, relative densities of the dominant species and spatial distribution over the last
445 30 years (Thiébaud et al., 1997; Bacouillard et al., 2020). The community is dominated by
446 highly resilient benthic-pelagic species capable of rapidly rebuilding their populations after a
447 stress: i.e. large number of larvae, long planktonic larval stage ensuring a wide dispersal due
448 to currents, retention near the Seine estuary-river system, drifting of juveniles and adults
449 favouring a return to shallow coastal waters (Olivier et al., 1996; Pezy et al., 2017, 2018).
450 Under these environmental conditions, after a one-year experiment of dumping sediment at
451 the Machu site, Pezy et al. (2017, 2018) have shown that the impact remains local and the
452 benthic habitats exhibit a high degree of resilience with the rapid recovery of the community
453 observed just after cessation of the disturbance. Moreover, during the deposition phase,
454 Impacted and Influenced zones can be distinguished from a Non-impacted zone (Pezy et al.,
455 2018). Similarly, the Octeville dumping site located in the eastern part of the Bay of Seine
456 appears to behave as a very resilient system on a 30-year time scale (Bacouillard et al.,
457 2020). This explains the relatively low impact of dumping sediment in spite of the large
458 volumes disposed in this type of dynamic environment with benthic species that are able to
459 rapidly colonize stressed local zones. Moreover, the dumped sediment is rapidly dispersed
460 since it is mainly composed of 80% of fine particles (<63 µm) dredged from the harbour
461 basin of the Havre (Baux, 2018; Baux et al., 2019). At the GJ station, which was regularly
462 impacted by dumping throughout the study, the percentage of fine particles reaches only
463 just over 60% at the end of the study from T4 to T5. This hydrodynamic regime prevents the
464 accumulation of mud on the dumping site and leads to the dispersal of fine particles around
465 the site mainly towards the North-east (Baux et al., 2019), in an area naturally enriched in
466 fine particles coming from the Seine estuary (Lesourd et al., 2016). Méar et al. (2018)
467 suggested that sediment coming from the Octeville dumping site, particularly the fine
468 fraction, is dispersed over long distances towards the north-northeast and then settles out
469 to form a thin layer. Moreover, the Antifer terminal breakwater is a key element in the
470 sedimentary dynamics of the eastern part of the Bay of Seine, favouring the trapping of fine
471 material.

472 Three main zones are recognized in our present study (Fig. 6): an impacted zone, an
473 influenced zone and an unimpacted zone. There is a negative correlation between the

474 volume of dumped sediment and the Taxonomic Richness as well as the Abundance of the
475 macrofauna, illustrating the negative impact of sediment dumping on the benthic habitat.
476 Dominant species such as the bivalves *Abra alba* and *Phaxas pellucidus* show their lowest
477 abundance at dumping site stations, while the polychaete *Lagis koreni* show the same order
478 of magnitude of abundance at all ten sampling stations, and *Owenia fusiformis* shows a
479 significant low abundance in the dumped zone. A seasonal pattern of settlement is observed
480 for the four species and appears synchronous at all the stations including dumping stations,
481 being concentrated from the end of spring to the beginning of autumn. The input of fine
482 particles to the dumping zone does not prevent the arrival of recruits of these species with a
483 benthic-pelagic type of development. However, differences are observed spatially, at the
484 scale of the dumping zone and around this zone. Over the dumping area, there are variations
485 within the community between stations within the dumping zone and those within the
486 influenced zone, with two distinct assemblages, namely a and b. The stations in the
487 influenced zone (assemblage b) are very rich in the bivalve *Phaxas pellucidus*. Nevertheless,
488 in the southern zone, one station shows a species composition that is different from the
489 others (NP101, assemblage d of Fig. 6). This station yields low abundances of the bivalve
490 *Abra alba*, but shows abundant individuals of the bivalve *Phaxas pellucidus*.

491 More generally, it remains a challenge to assess the impact of human activities on the
492 condition of marine ecosystems and various indices and tools have been developed for that
493 purpose. It is clear that in such hydrodynamic conditions and in a system highly resilient,
494 with a small impact of the deposit on the benthic communities, the benthic indices such as
495 AMBI and BOPA are not appropriate to assess the impact of dumping dredge sediment on
496 benthic communities. The dispersive hydrodynamics regime hinders the accumulation of fine
497 particles and the associated Total Organic Carbon; in the study area, the TOC is low (< 1.5%)
498 and the abundance of benthic opportunist species remains negligible. Our results differ from
499 those obtained by Katsiaras et al. (2015) in the Saronikos Gulf, Greece in an enclosed system
500 with low exchange with the surrounding coastal area, on a deeper sediment dumping zone
501 (50-80 m) with a monthly disposed volume of 33,000 m³ for a total of 700,000 m³ of
502 contaminated sediment disposed over a period of 21 months. In the dumping zone, soft-
503 bottom habitats are so degraded that they become azoic, and benthic communities at
504 locations up to 3.2 km away from the sediment deposit -ground show a significant decrease
505 of species number and abundance. In this case, benthic indices are significantly correlated

506 with hydrocarbon concentrations and sediment toxicity levels. After sediment dumping, the
507 macrofauna communities of the sediment deposit l-ground remain significantly degraded,
508 but the surrounding areas show patterns of recovery.

509 Several studies showed that the hydrodynamic conditions were determinant in the
510 resilience of dredged harbour sediment deposit sites. The impact of a deep-ocean deposit
511 offshore California on a high diversified benthic community was not detected at a regional
512 scale; Blake et al. (2009) observed no degradation of benthic infauna due to dredged
513 material disposal, and the reference stations and stations on the site boundary maintaining
514 high species richness and diversity during all the study. Blake et al. (2009) suggested that
515 colonization of fresh dredged material was rapid due to larval dispersal and settlement
516 account and adult organism migrations from adjacent areas to the disturbed sites via
517 boundary layer currents. The effects of dredge material disposal on benthic macrofauna was
518 studied offshore Santa Catarina in the south of the Brazil by Vivan et al. (2009) who showed
519 that abundance, taxa richness and Shannon–Weaver diversity were significantly smaller after
520 the disposal, and recovered the initial mean values after the disposal ends. The high volume
521 and frequency of sediment disposal were the main factors of environmental modifications,
522 since the amount of deposited sediment on the seabed exceeded the natural sedimentation
523 rates. Deeper areas and high speed currents decreased the damage to macrofauna
524 organisms, due to the sedimentation reduction and transport of the sediments by the water
525 column reducing the volume on the seabed (Vivan et al., 2009). Simonini et al. (2005)
526 showed that the communities of the dumping areas in the Northern Adriatic Sea offshore
527 Emilia-Romagna were well adapted to an unstable environment, characterized by high
528 hydrodynamic and high rates of sedimentation and re-suspension that cause marked
529 sediment instability, especially in winter, when the bottom surface was continuously
530 disturbed. Ware et al. (2010) underlined from an assessment of ecological consequences of
531 the disposal of dredged material at sites around the England and Wales coast that a number
532 of factors contributed to a site-specific nature of impacts such as hydrodynamic regimes,
533 habitat type in the site disposal before deposition, natural variability of associated faunal
534 communities and design of the disposal activity: material type, frequency, volume, timing,
535 severity and longevity of impacts.

536 In addition to structural changes of the benthic communities due to sediment
537 disposal, Bolam (2012) compared macrofaunal structural and functional responses based on

538 secondary production as the proxy for a functional attribute on 14 dredged material disposal
539 sites along England and Wales coast. Disposal resulted in significant reductions of total
540 secondary production at seven sites; no sites exhibited significant increases in production
541 estimates. These seven sites were generally those which displayed significant structural
542 impacts. Later Bolam et al (2016) had analysed the impact of deposit sediment on a site off
543 the northeast coast of England through changes of the biological traits of life of the recorded
544 species on impact and unimpacted stations. In this site, changes in physical structure of the
545 sediment with a continued smothering constant disposal resulted in an assemblage with
546 reduced number of species and total abundance, and a reduced capability to provide energy
547 to the next trophic level due to a reduced numerical part of highly-productive species.
548 Moreover, they discovered that the assemblages characterising the disposal site possess a
549 greater bioturbative capability compared to those not directly impacted by disposal.

550 Cole and McBrick (2004) have proposed to assess the impacts of dredge sediment
551 disposal using equivalence tests. More recently, Baux et al. (2020) have proposed an
552 integrative approach to assess the environmental impacts of dumping operations, via the
553 development of a Dredge Disposal Sediment Index (D2SI). The initial development of D2SI
554 was conducted on the experimental dumping site of Machu (Bay of Seine, by applying a BACI
555 (Before-After-Control Impact) approach. The D2SI was also tested on two other dumping
556 sites of the Bay of Seine (Kannik and Octeville) (Baux et al., 2020). Two types of stations are
557 required (impacted and unimpacted) to determine D2SI. In fact, D2SI takes account of six
558 sub-indices considering the biological and physical changes generated by dredge disposal on
559 the seabed: macrofauna, demersal fish, percentage of fine particles (percentage < 63 μm),
560 bathymetry and two traits of macrofauna species, i.e. resistance of the dumping and
561 availability of drifting near the sea bed. An additional implementation (D2SI2) is applicable in
562 the case of absence of data on fish or biological traits of life of macrobenthic fauna. Another
563 way to assess the impact of dumping harbour dredged material on ecosystem functioning is
564 to use trophic modelling along with Ecopath models associated with Ecological Network
565 Analysis indices (Raoux et al., 2020).. A similar modelling approach was proposed by de
566 Jonge and Schückel (2019) who explored the effects of dredging and organic waste on the
567 functioning and quantitative biomass structure of the Ems estuary food web by applying
568 Input Method balancing in Ecological Network Analysis. These last authors recommend
569 promoting an integrated approach for the assessment and management of such dumping

570 zones. Fettweis et al. (2011) studied the effects of the disposal of fine sediments on the
571 concentration of suspended particulate matter (SPM) in shallow waters off the Belgian coast
572 near Zeebrugge harbour (southern North Sea) and suggested that SPM concentrations could
573 be used as an indicator of environmental changes when a long time-series is available.
574 Virtasalo et al. (2018) have assessed the influence of dredge sediment dumping on the
575 geological integrity of the seafloor at the Vuosaari and Uusikaupunki-D dumping sites in
576 Finland (northern Baltic Sea and proposed two methods, i.e. a seabed seismo-acoustic
577 survey and sediment core analysis for the assessment of the geological integrity of the
578 seafloor.

579 Baux et al. (2019) conducted geochemical and sedimentological studies to highlight
580 the sedimentary dynamics of the Octeville deposit site to determine the area influenced by
581 dumped sediment and the sediment transport directions. The results showed that the fine
582 sediment disposed on the Octeville dumping site is dispersed by currents in three main
583 directions. The first component is directed mainly towards the North-Eastern sector, the
584 second to the north and the third appears to be dispersed southwards.

585 In summary, several tools and methods including modelling are needed to assess the actual
586 impact of dumping sediment in shallow coastal ecosystems. This assessment should be
587 carried out in terms of sediment accumulation and dispersion, taking into account the
588 impact of dumping on the benthic habitat and the ecological functioning of such ecosystems
589 influenced by anthropogenic stressors.

590

591 **5. Recommendations**

592

593 The off-shore discharge of dredged sediment disposal from the Le Havre harbour basins
594 onto the Octeville site in the eastern part of the Bay of Seine appears to be a sustainable
595 strategy for the management of uncontaminated dredged sediments. To study the
596 modifications of sediment composition due to the input of fine particles onto the Octeville
597 dumping site study in this paper, Méar et al. (2018) suggested using regular grid sampling for
598 the mapping of surficial sediment distribution. This is merely a strategy allowing the drawing
599 up of maps based on currently used interpolation methods.

600 Considering the results of our present study, we can make several recommendations for
601 the sustainable management of this dumping area. The dumping method used today by the

602 Le Havre harbour consists of spreading the deposits over the entire site (dumping by
603 alternating zones), which leaves a rest time for each zone (no dumping for six months). This
604 appears to an adequate technique to promote the rapid colonization of macrobenthic
605 species (Baux, 2018). The impact could be also diminished by a decrease or cessation of
606 dumping during the summer and early autumn, which corresponds to the settlement period
607 for the dominant benthic species. In the future, winter sampling campaigns should be
608 recommended because this would be outside the settlement period of the dominant species
609 (Dauvin et al., 2017).

610 For a long-term study of the Octeville dumping site, a single sampling campaign seems
611 sufficient to assess the evolution of the macrobenthic fauna. Sampling on the influenced
612 zone alone does not provide important information on the state of the system and is not
613 important to study its change for a long-term survey. Sampling must be carried out on the
614 impacted and unimpacted zones, to establish a comparison between these two zones and
615 monitor the evolution of the ecological status while using the same number of stations on
616 each of these two zones (impacted and non-impacted). Thus, according to the results of this
617 study, it is recommended to establish monitoring protocol for benthic communities and their
618 associated sediments, by defining the same number of stations located within the dumping
619 zone (for example, four stations) as defined in zones not impacted by the dumping located in
620 the south-east (two stations) and in the north-west (two stations).

621 Finally, our study illustrates a special case with high resilience in a dynamic environment.
622 This is in agreement with the review of Bolam et al. (2006), who carried out a holistic
623 assessment of activities at 18 different intertidal and subtidal disposal sites around the coast
624 of England and Wales. They pointed out that impacts are specific to each disposal site, and
625 disposal-site communities are generally impoverished to varying degrees while site-specific
626 variations depend mainly on prevailing hydrographic regimes.

627

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629

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638

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781

782 **Figures captions**

783

784 Fig. 1. Location of benthos sampling stations on the Octeville dumping site and in the
785 southern and northern part of the dredge sediment deposit in the eastern part of the Bay of
786 Seine.

787

788 Fig. 2. Experimental sampling strategy on the Octeville dumping site from May 2016 to
789 September 2017.

790

791 Fig. 3. Sediment composition (mean proportion % of the granulometric classes), and Total
792 Organic Carbon (% of dry sediment) in the five impacted and influenced stations at the six
793 dates of the study.

794

795 Fig. 4. Variation of taxonomic richness during experimental sampling on the Octeville
796 dumping site from May 2016 to September 2017.

797

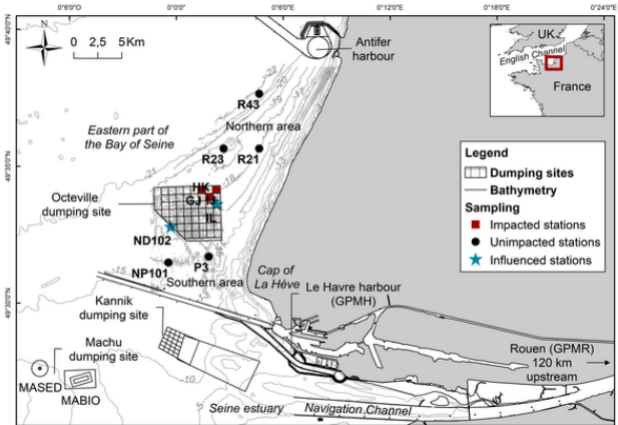
798 Fig. 5. Variation of abundances during experimental sampling on the Octeville dumping site
799 from May 2016 to September 2017.

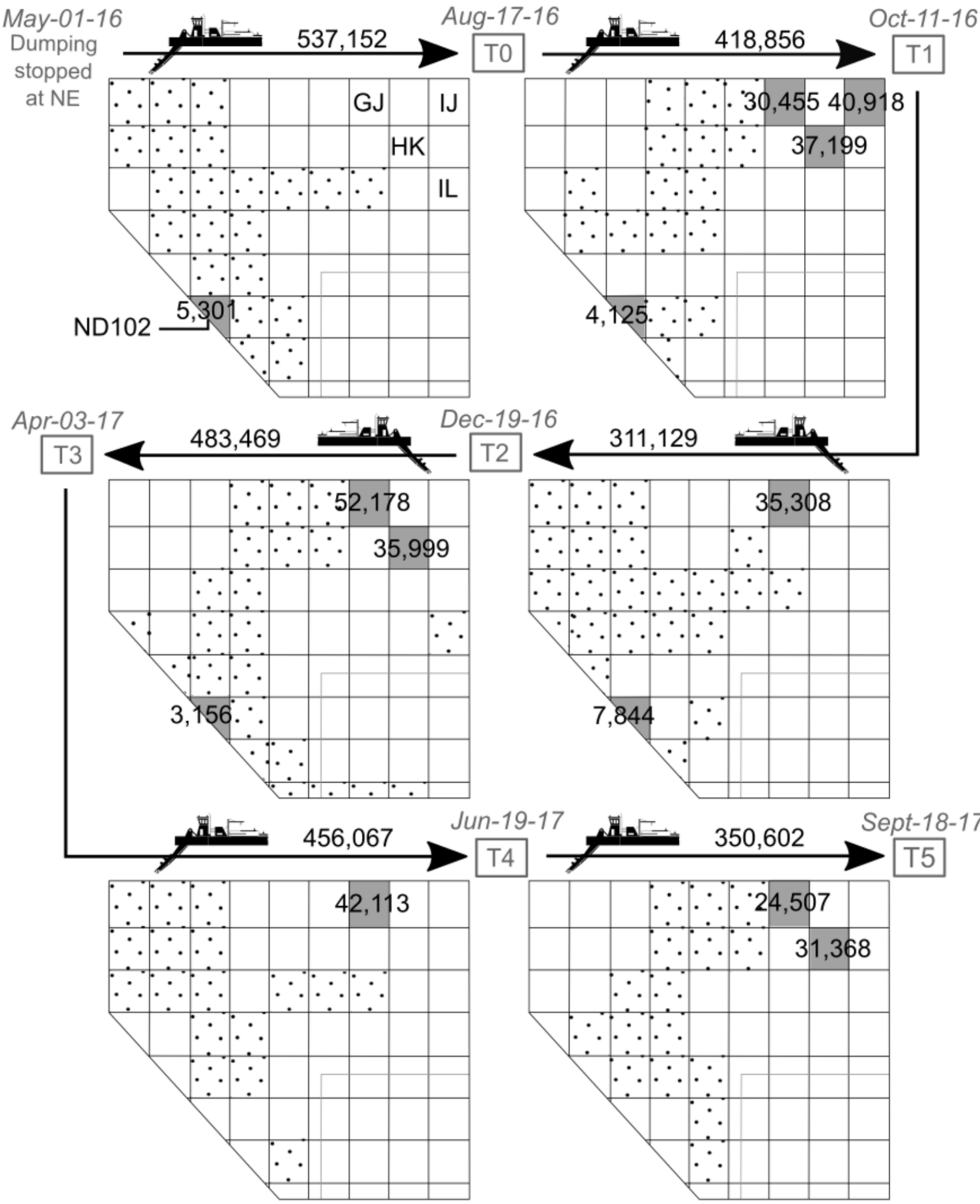
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801 Fig. 6. Results of cluster analysis using group average linking with the Bray-Curtis similarity
802 measure on the mean $\text{Log}_{10}(x + 1)$ transformed abundances (0.3 m^{-2}) for the 10 stations.

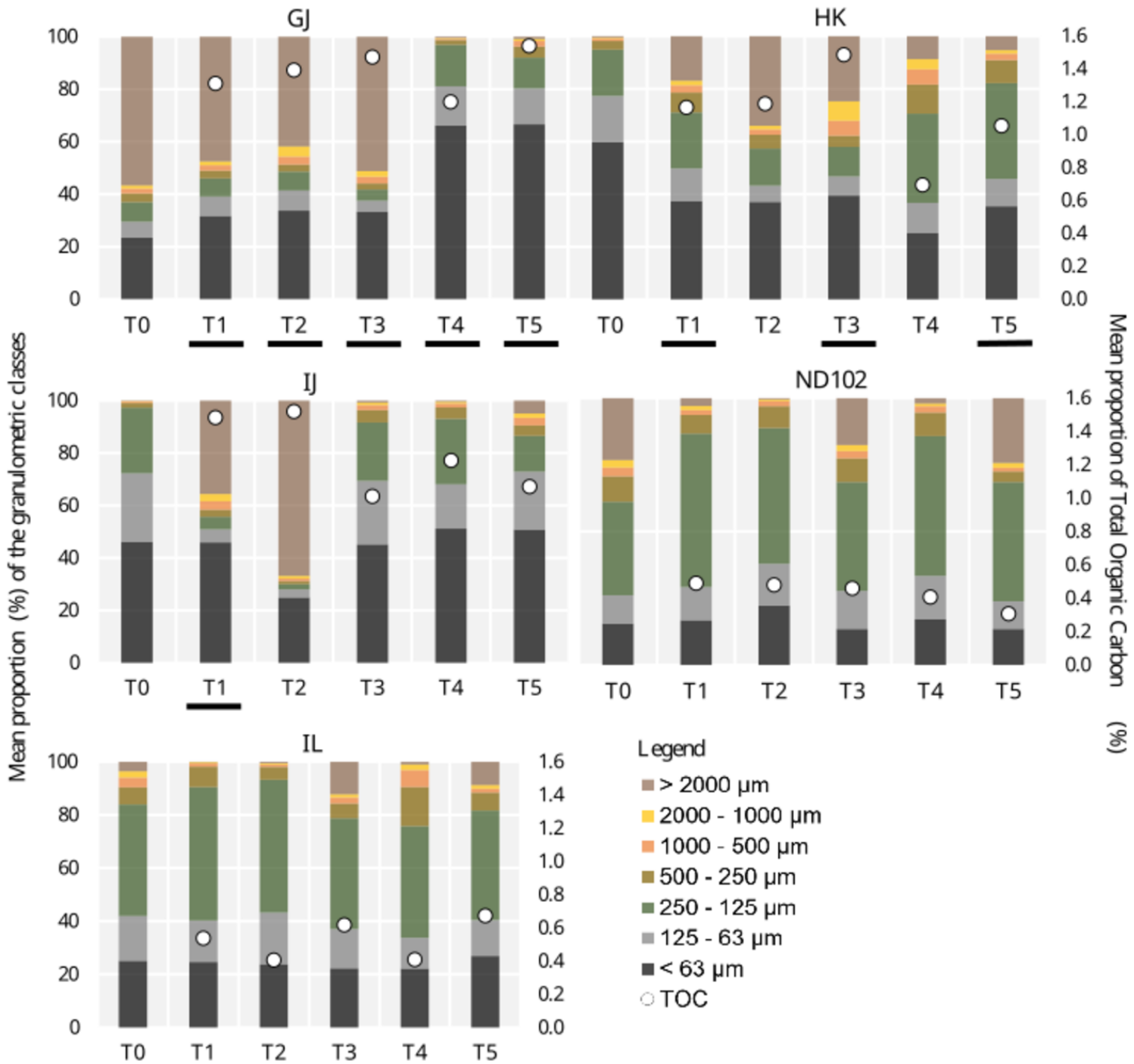
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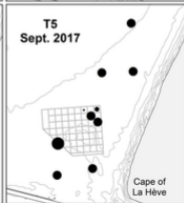
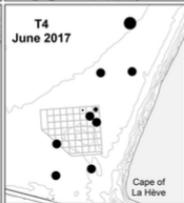
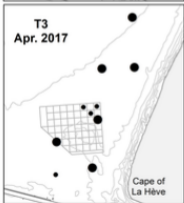
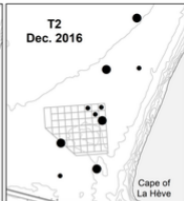
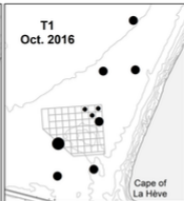
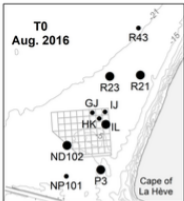
804 Fig. 7. Correlations between the dumped volumes in m^3 and the Taxonomic Richness as well
805 as the Abundance of the macrofauna during the six campaigns (T0 to T5) for stations GJ
806 (sediment disposed every three months at T1, T2, T3, T4 and T5); HK (sediment disposed
807 every six months at T1, T3 and T5); IJ (sediment disposed once at the start of the experiment
808 at T1) and ND102 (regular sediment deposition, but always in very small quantities). Natural
809 variability: cloud of values at other non-dumped stations. Spearman correlation coefficient
810 (r) and associated significance (*) calculated by considering only the dumped stations. See
811 Figure 1 for the location of the stations.





- Unexperimental dumping
- Experimental dumping and dumped volumes (m³) on each box
- Dumping period and total dumped volumes (m³) between each benthic surveys





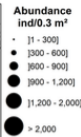
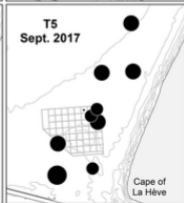
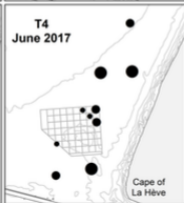
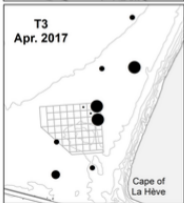
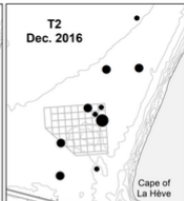
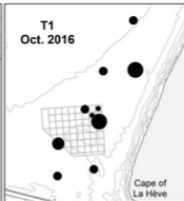
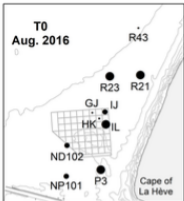
Taxonomic richness

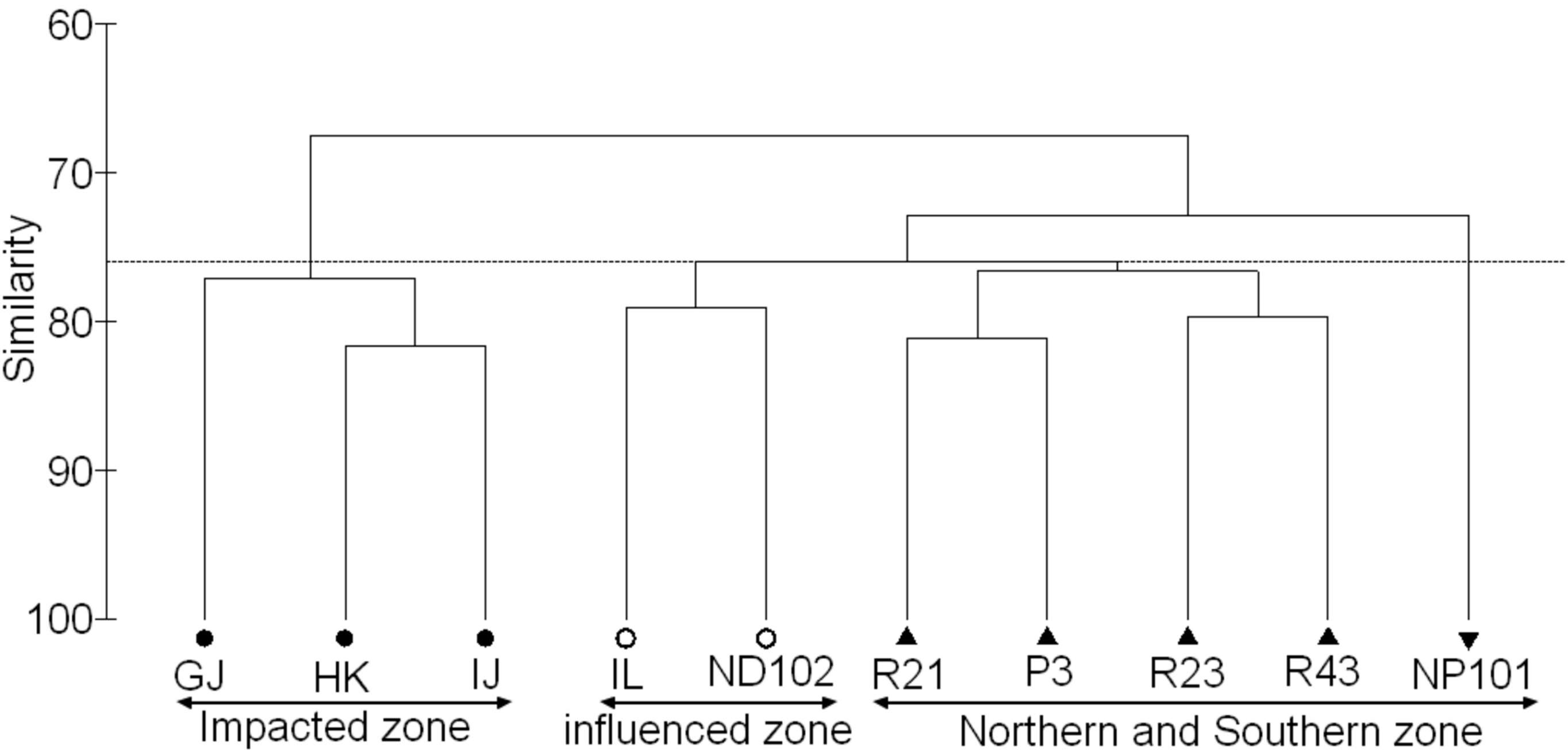
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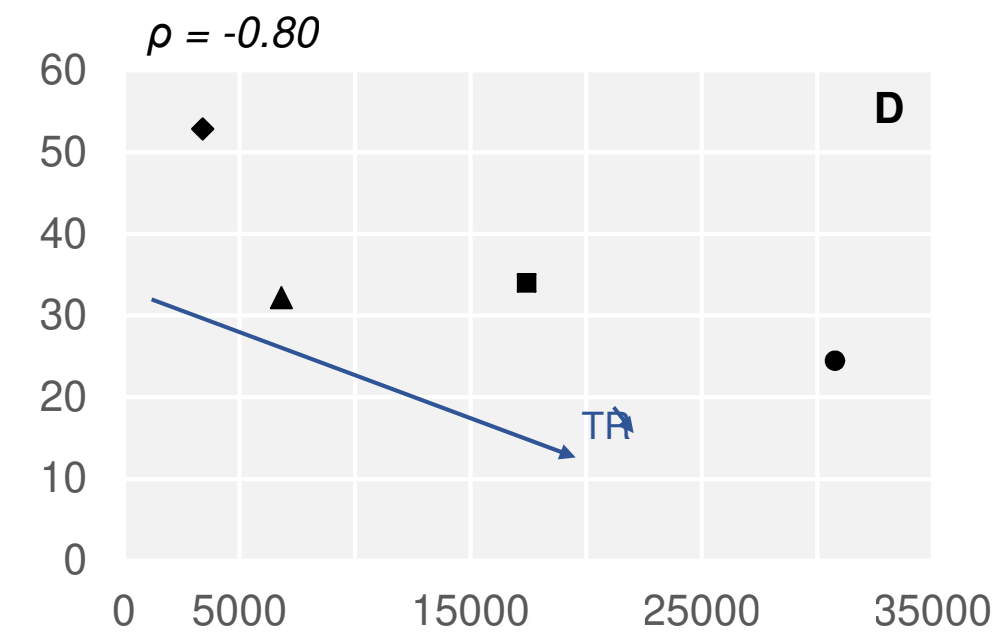
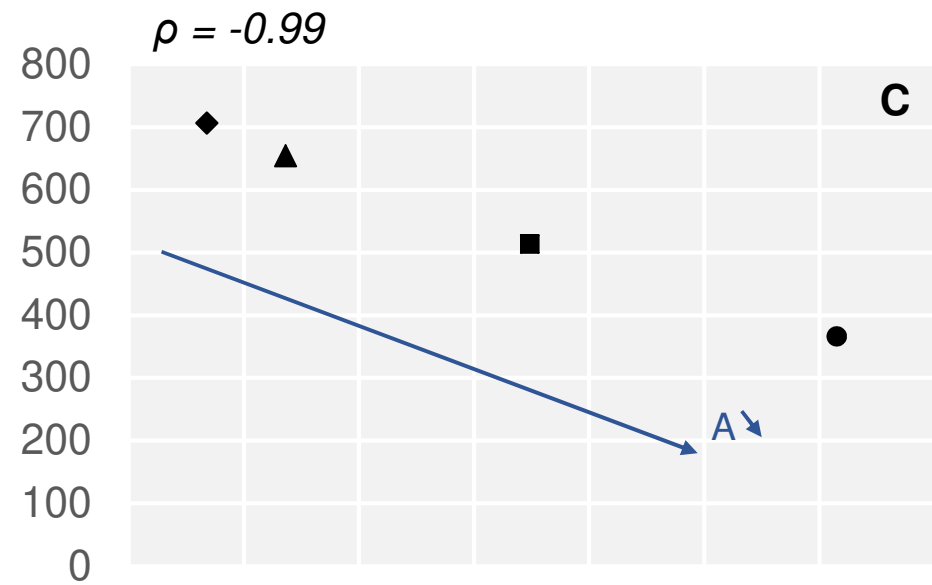
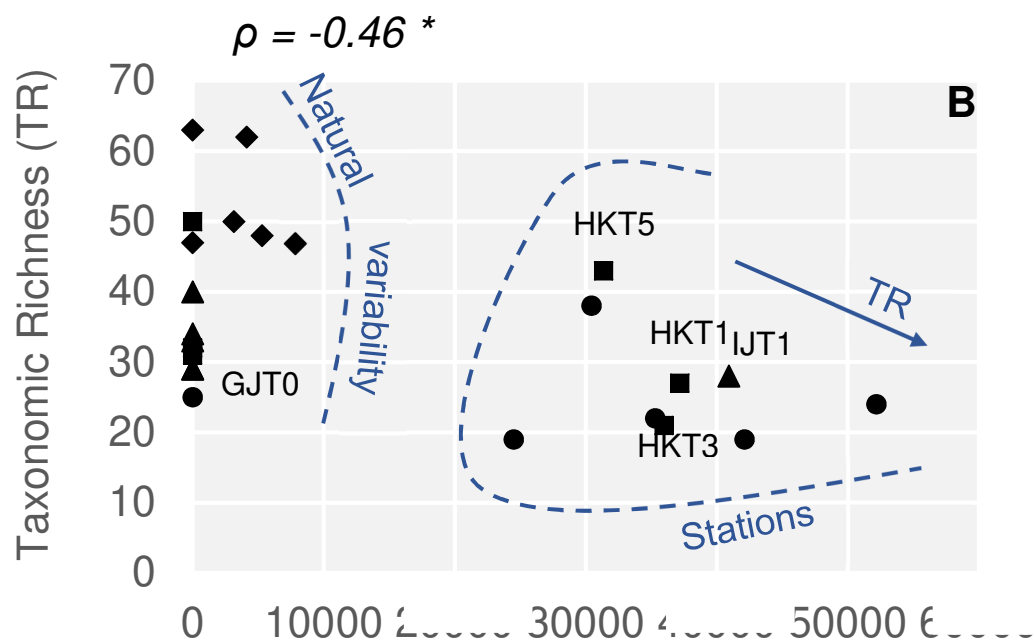
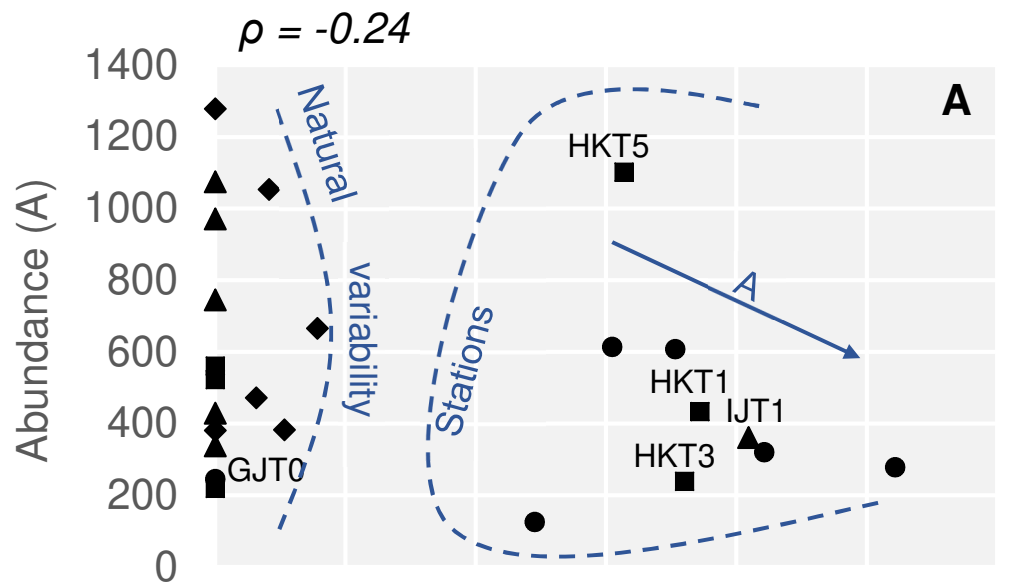
Octeville
dumping
site



0 1 2 Km







Volume of dumped sediment in m3

● GJ ■ HK ▲ IJ ◆ ND102

Table 1. Mean values of the percentage of the fine particles (< 63 μm) and Total Organic Carbon (TOC) with Standard Deviation (SD) in the 10 stations for the Impacted, Influenced, South and North Zones.

	Station	<63 μm (%)	SD (%)	TOC (%)	SD (%)
Impacted zone	GJ	42	19	1.33	0.17
	HK	39	11	1.12	0.29
	IJ	44	10	1.19	0.30
Influenced zone	IL	24	2	0.53	0.12
	ND102	16	3	0.43	0.07
South zone	NP101	27	3	0.73	0.07
	P3	22	6	0.75	0.15
North zone	R21	31	2	0.80	0.10
	R23	28	5	0.67	0.12
	R43	17	3	0.40	0.05

Table 2. Two-way ANOVA with interaction of the dates and the zones and one-way ANOVA for stations on the Taxonomic Richness and Abundances, and one-way ANOVA

Variables	Factors	Df	F	p
Taxonomic Richness	date	5	4.375	< 0.001
	zone	3	37.366	< 0.001
	station	6	2.012	0.09
	date x zone	15	0.566	0.88
Abundances	date	5	5.883	< 0.001
	zone	3	7.689	< 0.001
	station	6	2.526	0.04
	date x zone	15	0.541	0.89

Table 3. Values of the diversity indices of the 10 stations. Shannon-Wiener: H'; Pielou evenness: J'; AMBI and. BO2A. H': < 1 red; 1-1.99: orange; 2-2.99: yellow; 3-3.99: green and > 4 blue. J': < 0.20 red; 0.20–0.39 orange; 0.40–0.59: yellow; 0.60–0.79 green and > 0.79 blue. For threshold see Dauvin et al. (2017b).

	Zone	Stations	T0	T1	T2	T3	T4	T5
H'	Impacted	GJ	3.92	3.88	3.02	3.72	3.22	3.40
	Impacted	HK	3.99	3.30	3.49	2.69	4.19	3.15
	Impacted	IJ	3.64	3.37	3.48	3.50	3.23	2.68
	Influenced	IL	3.54	4.16	3.41	3.88	4.52	3.82
	Influenced	ND102	3.92	3.95	3.52	4.20	4.05	3.35
	North	R21	4.03	3.88	3.72	4.42	3.78	3.79
	North	R23	3.39	3.71	3.67	4.12	3.63	3.40
	North	R43	4.07	4.46	4.29	4.21	4.38	4.19
	South	NP101	4.20	3.61	3.57	3.96	4.23	3.26
	South	P3	3.47	3.81	3.98	4.05	4.11	3.58
J'	Impacted	GJ	0.84	0.74	0.68	0.81	0.76	0.80
	Impacted	HK	0.81	0.69	0.70	0.61	0.74	0.58
	Impacted	IJ	0.72	0.70	0.72	0.66	0.64	0.55
	Influenced	IL	0.66	0.71	0.63	0.69	0.77	0.66
	Influenced	ND102	0.70	0.66	0.63	0.74	0.73	0.56
	North	R21	0.74	0.67	0.70	0.78	0.65	0.69
	North	R23	0.61	0.68	0.69	0.74	0.63	0.60
	North	R43	0.78	0.77	0.79	0.78	0.74	0.73
	South	NP101	0.79	0.66	0.69	0.74	0.77	0.59
	South	P3	0.63	0.67	0.72	0.72	0.71	0.63
AMBI	Impacted	GJ	1.73	1.52	1.92	1.71	1.78	1.89
	Impacted	HK	1.85	1.57	1.89	2.10	1.57	1.80
	Impacted	IJ	1.79	1.79	1.68	1.82	1.74	2.02
	Influenced	IL	1.92	1.52	1.73	1.89	1.71	1.81
	Influenced	ND102	1.48	1.65	1.45	1.82	1.60	1.65
	North	R21	1.65	1.61	1.57	1.39	1.61	1.72
	North	R23	1.61	1.75	1.43	1.65	1.90	1.60
	North	R43	1.59	1.89	1.68	1.42	1.95	1.75
	South	NP101	2.02	1.78	2.17	1.89	1.57	2.45
	South	P3	1.75	2.02	1.90	2.17	1.65	2.05
BO2A	Impacted	GJ	0.020	0.017	0.019	0.021	0.023	0.014
	Impacted	HK	0.032	0.045	0.078	0.051	0.028	0.048
	Impacted	IJ	0.047	0.076	0.026	0.052	0.078	0.024
	Influenced	IL	0.025	0.24	0.017	0.023	0.020	0.019
	Influenced	ND102	0.014	0.021	0.018	0.016	0.019	0.027
	North	R21	0.040	0.034	0.045	0.042	0.057	0.060
	North	R23	0.012	0.007	0.018	0.015	0.021	0.023

	North	R43	0.014	0.019	0.021	0.015	0.023	0.014
	South	NP101	0.012	0.020	0.023	0.021	0.020	0.029
	South	P3	0.078	0.084	0.117	0.102	0.045	0.120

Table 4. Average abundance (A in ind.0.3 m⁻²) and contribution (C in %) of the ten main species for each group formed with the CAH and n-MDS analysis carried out on the average abundance data by station. The similarity value is indicated for each group in parentheses. The average abundances shown for group d correspond to the ten main species of NP101 station, since this group is formed only by this single station.

Impacted zone (78.61%)			Influence Zone (79.08%)			North zone plus P3 (77.88)			NP101	
Species	A	C (%)	Species	A	C (%)	Species	A	C (%)	Species	A
<i>Owenia fusiformis</i>	141.1	6.32	<i>Phaxas pellucidus</i>	164.7	4.04	<i>Abra alba</i>	163.8	4.24	<i>Phaxas pellucidus</i>	210.7
<i>Abra alba</i>	67.2	5.38	<i>Owenia fusiformis</i>	138.0	3.78	<i>Owenia fusiformis</i>	138.7	4.02	<i>Kirkegaardia dorsobranchialis</i>	154.5
<i>Phaxas pellucidus</i>	29.2	4.78	<i>Abra alba</i>	103.8	3.76	<i>Phaxas pellucidus</i>	64.6	3.72	<i>Owenia fusiformis</i>	100.2
<i>Nephtys hombergii</i>	22.3	4.55	<i>Ampharete baltica</i>	87.3	3.62	<i>Nucula nitidosa</i>	43.6	3.38	<i>Ampharete baltica</i>	61.0
<i>Nucula nitidosa</i>	26.5	4.41	<i>Melinna palmata</i>	68.8	3.04	<i>Ampharete baltica</i>	49.1	3.24	<i>Melinna palmata</i>	39.3
<i>Lagis koreni</i>	18.4	4.29	<i>Nephtys hombergii</i>	23.6	2.81	<i>Nephtys hombergii</i>	30.3	3.04	<i>Lagis koreni</i>	37.7
<i>Spisula subtruncata</i>	18.9	3.85	<i>Spisula subtruncata</i>	15.5	2.61	<i>Kirkegaardia dorsobranchialis</i>	31.5	2.87	<i>Spisula subtruncata</i>	35.2
<i>Acrocnida brachiata</i>	22.2	3.84	<i>Ophiura ophiura</i>	18.0	2.56	<i>Pholoe baltica</i>	19.8	2.78	<i>Pholoe baltica</i>	25.8
<i>Melinna palmata</i>	19.2	3.79	<i>Kirkegaardia dorsobranchialis</i>	10.9	2.36	<i>Ampelisca tenuicornis</i>	50.9	2.73	<i>Phoronis psammophila</i>	22.8
<i>Pholoe baltica</i>	16.0	3.76	<i>Pholoe baltica</i>	23.8	2.3	<i>Melinna palmata</i>	25.2	2.73	<i>Acrocnida brachiata</i>	22.3

Table 5. Total numbers of individuals measured at each date and in each stations and proportion of juveniles (%juv, line below) for *Abra alba* (Lt<5 mm), *Phaxas pellucidus*

	<i>Abra alba</i>										Total	<i>Phaxas pellucidus</i>										Total	
	GJ	HK	IJ	IL	ND102	R43	R23	R21	P3	NP101		GJ	HK	IJ	IL	ND102	R43	R23	R21	P3	NP101		
T0-August2016	30	22	102	181	130	38	371	178	244	11	1307	4	5	10	50	40	0	9	8	50	30	206	
%juv	0	0	0	2	0	0	0	0	0	0		25	0	0	0	5	0	0	0	0	0	50	
T1-October2016	59	41	123	250	195	71	183	185	148	35	1290	74	44	7	167	275	47	27	28	76	168	913	
%juv	1	2	12	7	0	6	1	0	0	0		0	0	0	0	0	0	0	0	0	0	0	
T2-December2016	123	25	35	84	68	29	75	149	37	11	636	29	60	9	126	176	21	22	19	57	215	734	
%juv	1	10	4	6	0	11	1	0	0	0		0	0	0	0	0	0	0	0	0	0	0	
T3-April2017	26	8	119	101	34	35	94	95	95	25	632	8	3	13	80	79	35	8	20	36	156	438	
%juv	30	12	15	11	13	57	32	14	45	41		0	0	0	0	0	0	0	0	0	0	0	
T4-June2017	62	98	124	62	44	214	217	392	262	18	1493	0	7	5	34	9	6	4	2	7	17	91	
%juv	4	13	10	28	7	2	27	24	6	15		0	0	0	5	20	0	0	0	0	0	50	
T5-September2017	11	29	128	49	27	97	212	309	111	6	979	14	34	64	118	285	112	151	94	94	281	1247	
%juv	0	2	0	0	0	2	8	2	0	0		0	0	0	0	0	0	0	0	0	0	0	
Total	311	223	631	727	498	484	1152	1308	897	106	6337	129	153	108	575	864	221	221	171	320	867	3629	
	<i>Lagis koreni</i>										Total	<i>Owenia fusiformis</i>										Total	
	GJ	HK	IJ	IL	ND102	R43	R23	R21	P3	NP101		GJ	HK	IJ	IL	ND102	R43	R23	R21	P3	NP101		
T0-August2016	28	52	32	24	8	16	26	17	14	30	247	24	9	35	44	18	28	51	24	3	1	237	
%juv	20	1	0	0	0	0	0	0	0	2		2	0	25	10	0	2	1	0	0	100		
T1-October2016	34	9	8	45	2	47	9	8	9	12	183	24	27	27	109	48	63	185	164	11	1	659	
%juv	2	0	0	1	0	4	0	0	0	17		90	90	20	50	60	40	32	5	55	100		
T2-December2016	22	52	9	21	9	17	11	8	2	4	155	26	132	119	332	31	84	207	88	11	29	1059	
%juv	0	0	0	0	0	0	0	0	0	0		90	42	30	41	77	41	41	35	80	80		
T3-April2017	5	0	11	9	2	6	3	4	0	0	40	1	143	40	291	18	77	111	131	6	1	819	
%juv	0	0	0	0	0	0	0	0	0	0		100	22	70	24	74	31	33	20	89	100		
T4-June2017	2	15	6	50	16	8	4	10	30	89	230	54	105	105	61	76	59	122	64	29	1	676	
%juv	0	55	0	52	50	50	50	50	45	10		10	21	17	35	28	7	2	9	40	100		
T5-September2017	0	17	27	30	14	23	25	10	4	86	236	1	119	223	414	257	81	276	111	54	1	1537	
%juv	0	0	20	10	0	4	7	70	0	7		100	87	72	9	14	85	31	80	87	100		
Total	91	145	93	179	51	117	78	57	59	221	1091	130	535	549	1251	448	392	952	582	114	34	4987	

axas pellucidus (Lt< 8 mm), Lagis koreni (Cl< 2 mm) and *Owenia fusiformis* (l4< 4 mm).