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10 **Abstract**

11 Offshore marine renewable energy installations (MREI) introduce structure into the
12 marine environment and can locally exclude destructive, bottom trawl fishing. These
13 effects have the potential to aid restoration of degraded seabed habitats but may be
14 constrained by timescales of ecological succession following MREI construction, and the
15 removal of infrastructure during decommissioning. To inform managers about
16 appropriate decommissioning strategies, a 25 km cable and associated rock armouring
17 (Wave Hub, UK), installed on rocky reef, was monitored up to five years post-
18 deployment. The epibenthic Assemblage Composition, and Number of Taxa remained
19 significantly different from surrounding controls, while Abundance was similar in all
20 survey years between the cable and controls. Six morphotaxa showed four patterns of
21 colonisation on cable plots compared to the controls: 1) Early colonisation, which
22 remained in greater abundances (Porifera), 2) early colonisation, converging (Turf), 3)
23 slow colonisation, converging Anthozoa and Vertebrata), and 4) slow colonisation,

24 remaining lower in abundance (Tunicata and Echinoderms). The environmental
25 relevance of this MREI is considered relatively benign as it covers 0.01 % of the
26 surrounding bioregion, appears to be supporting similar assemblages to the surrounding
27 habitat, and exhibited minimal evidence of invasive species (three records of two non-
28 native species). Longer monitoring timescales are required to provide comprehensive,
29 site-specific decommissioning advice.

30

31 Keywords: Benthic, monitoring, underwater video, marine renewables, subsea cable,
32 marine non-native

33

34

35 **INTRODUCTION**

36 Energy generation from marine renewable sources is expected to contribute significantly
37 to the future worldwide energy mix as governments pursue renewable alternatives to
38 fossil fuels (Lewis *et al.*, 2011; Roche *et al.*, 2016). The European Commission's (EC) 2020
39 strategy set a target to increase the share of renewable energy sources in its collective
40 energy output to 20% by 2020 (European Commission, 2010). In line with this, the EC
41 Blue Growth agenda promotes the sustainable growth of "blue energy" and highlights
42 the potential for more than 130 TWh of energy to be produced annually, by offshore
43 renewable sources, by 2020 (European Commission, 2012). The goals of this Blue
44 Growth agenda are underpinned and supported by healthy, functioning marine
45 ecosystems (Lillebø *et al.*, 2017).

46

47 As marine renewable energy installations (MREIs) are likely to be located on habitats
48 previously impacted by bottom towed fishing gear, the introduction of structure and the
49 exclusion of the most destructive fishing activity could improve such ecosystems (Inger
50 *et al.*, 2009; Boehlert and Gill, 2010; Langhamer, 2012; Witt *et al.*, 2012; Sheehan *et al.*,
51 2013a). However, it has also been reported that MREIs, along with other artificial
52 structures placed in the marine environment, can act as “stepping stones” for non-
53 native and invasive species (Adams *et al.*, 2014; Airoidi *et al.*, 2015), potentially having a
54 detrimental effect on biodiversity.

55 Introducing hard substrate into the marine environment creates habitat for species to
56 colonise (Langhamer, 2012; Bishop *et al.*, 2017), with the potential to increase
57 biodiversity (Firth *et al.*, 2016), which can enhance the resilience of ecosystem functions
58 (Oliver *et al.*, 2015). As structures are gradually colonised they can have a positive
59 impact on the supply of ecosystem services; including the formation of biogenic habitat
60 to provide feeding and nursery resources for commercially important species
61 (Langhamer, 2012) and species of conservation importance (Langhamer and
62 Wilhelmsson, 2009). MREIs can act as fish aggregation devices (Broadhurst *et al.*, 2014),
63 and enhance conservation measures by creating *de facto* MPAs (Inger *et al.*, 2009;
64 Sheehan *et al.*, 2013b). However, the relative success of MREIs acting as artificial reefs
65 may be dependent on the receiving environment, since areas of existing low diversity
66 and minimal habitat structure stand to gain the most from the introduction of new hard
67 substrate (Inger *et al.*, 2009). Furthermore, community composition on artificial
68 structures does not necessarily reflect communities found in adjacent natural habitats
69 (Evans *et al.*, 2015), with the former often supporting lower diversity (Pister, 2009; Firth
70 *et al.*, 2013) . In some instances, increased provision of refuges for predators can

71 increase predation, negatively impact the surrounding prey species (Herrera *et al.*,
72 2002).

73 The species that colonise artificial infrastructure may depend on the type of material
74 used (Airoldi *et al.*, 2015; Firth *et al.*, 2016). Protective coverings such as boulders, that
75 are associated with MREI, can create complex communities with biomass increasing
76 over time (Wilson and Elliott, 2009). However, these components will be costly and
77 extremely difficult to remove without significant detriment to the marine environment
78 (Smyth *et al.*, 2015). Partial, rather than total, removal of subsea structures associated
79 with MREIs at the end of their operational life, has been advocated (Smyth *et al.*, 2015),
80 so that the marine habitat and associated ecosystem service provision can be sustained.
81 As the cables associated with offshore wind turbines may create habitat for species
82 recolonisation (Wilson and Elliott, 2009), cables associated with other MREI, such as
83 wave energy developments, may also provide habitat enhancement and associated
84 biodiversity and ecosystem services, although such data are not routinely collected for
85 subsea cable installations (Murray *et al.*, 2018). Furthermore, wave energy converters
86 (WECs) may be installed in habitats which differ from offshore wind installations, in their
87 seabed characteristics and hydrodynamic requirements, with the installation of wind
88 turbines typically taking place in sites with soft substratum and low hydrodynamic
89 forcing (Wilson and Elliott, 2009). While WECs are not constrained in the same manner
90 and may be installed on hard substratum, ideally in areas of greater hydrodynamic
91 forcing.

92 One aspect of MREIs that has received less attention is the impact of decommissioning
93 on the species and habitats that have formed over the operational life cycle of these
94 structures. As MREIs approach decommissioning, after 20-30 years for offshore wind
95 turbines (Smyth *et al.*, 2015), the question arises as to what course of action to take in

96 regards to how these structures can further be of use, if at all. Due to the renewables
97 sector still being a relatively infant industry (Copping, 2018), much of the guidance on
98 decommissioning comes from offshore oil and gas installations for which there is little
99 international consensus (Macreadie *et al.*, 2011). The international treaties that govern
100 the decommissioning of offshore oilrig platforms specify that full removal of these
101 structures must take place unless they can fulfil another valid purpose, such as the
102 creation of artificial reef habitat (Osmundsen and Tveterås, 2003). The decommissioning
103 of MREIs in the UK falls under the remit of the Department of Energy & Climate
104 Change's 2004 Energy Act. This stipulates full decommissioning, with some exemptions
105 such as if the installation can serve a new use by enhancing a living resource or if entire
106 removal would involve an unacceptable risk to the marine environment (DECC, 2004).

107 To inform future decommission strategies, epibenthic assemblage development was
108 assessed at a wave energy test site following the installation of a subsea cable and
109 associated rock armouring. This work aims to utilise an efficient video monitoring
110 approach to assess the development of epibenthic assemblages on the subsea cable
111 rock armouring for comparison to control areas. The outcomes of this work are of
112 relevance to MREI planning, management and decommissioning.

113 **METHODS**

114 *Study site and design*

115 Wave Hub is an 8 km² MREI located off the north coast of Cornwall, south west UK. A 25
116 km subsea power cable, 160 mm in diameter, connects a land-based electricity
117 substation to the hub, an electrical junction box (Figure 1). The seabed infrastructure
118 was deployed in 2010, with a life expectancy of 25 years (West *et al.*, 2009; Sheehan *et*
119 *al.*, 2013b). From the shore, the first 7 km of cable crosses sandy seabed and is buried.
120 The remaining cable crosses rocky reef with inter-reef sediments, reaching a maximum

121 depth between 50 and 60 metres. It is covered in rock armouring, at a minimum burial
122 depth of 300 mm, with concrete mattressing at 120 m intervals to provide additional
123 stabilisation as the substrate is not suitable for trenching. Overall, 80,000 tonnes of rock
124 was deployed on the seabed.

125 To assess the development of epibenthic assemblages on the cable rock armouring,
126 henceforth referred to as the “Cable”, relative to the surrounding environment, video
127 camera surveys were carried out two, four and five years after the cable infrastructure
128 deployment (June 2012, 2014 and 2015). Surveys were conducted over two days in each
129 year. Nine sites were haphazardly located along the extent of the cable from the hub, at
130 ~54 m to near the edge of the reef habitat at ~24 m (Figure 1). At each site, three
131 replicate ~200 m video transects were sampled, perpendicular to the cable route,
132 running east to west over the cable route or west to east, depending on the prevailing
133 tide/wind conditions. Replicate transects were spaced approximately 200 m apart, and
134 each one intersected the cable route at the transect mid-point. Each transect was later
135 split into three sections: Cable footage, and east and west control footage; a minimum
136 of 20 metres distance from the cable rock armouring (see Video analysis section). The
137 total distance sampled for each of the controls was therefore approximately 75 metres,
138 while the Cable varied in breadth between approximately two and five metres. Spatially
139 interspersing controls and impacted sites in this way ensures statistical comparisons can
140 be robustly attributed to the phenomenon of interest (in this instance, the cable
141 installation), rather than potentially being an artefact of an environmental gradient
142 (Underwood, 1997). It was not possible to survey each of the nine sites in each year,
143 owing to operational constraints. Seven sites were surveyed in 2012, and six in 2014. All
144 nine sites were surveyed during 2015.

145 *Benthic video survey*

146 Video transects were undertaken using High Definition (HD) video mounted on a flying
147 towed underwater video system (TUVS), described in detail elsewhere (Sheehan *et al.*,
148 2010, 2016). Briefly, the TUVS comprises a high definition camera, LED lamps, and laser
149 scaling, mounted on an aluminium frame which is suspended above the seabed, by the
150 counterbalance of weight and buoyancy, with a ground chain providing the only seabed
151 contact. The buoyancy control of this TUVS makes it an appropriate, relatively non-
152 destructive method for sampling epibenthos over variable seabed relief (Sheehan *et al.*,
153 2016). The live video feed is monitored and recorded in real-time to help avoid snagging
154 and to ensure video quality. The system was deployed from local commercial and fishing
155 vessels approximately 10 metres in length. Positioning was monitored and recorded
156 using differential GPS with the software package Hypack.

157 *Video analysis*

158 The video footage was analysed by extracting frame grabs at five-second intervals.
159 Where it was not possible to gather enough replicate frames, extraction took place at
160 one-second intervals. Frame grabs were then overlaid with a digital quadrat (Cybertronix
161 CXOverlay) (Figure 2). Unsuitable frames were rejected if they were out of focus,
162 overlapped the previous frame (to avoid replicate counts), or if the lasers were beyond
163 the margins of the digital overlay (see Sheehan *et al.*, 2010 for detail). Suitable frame
164 grabs for the Cable, Control W and Control E were subsequently identified and five were
165 randomly selected for analysis. All taxa within each frame grab were identified and
166 counted, or in the case of encrusting species or Turf assigned a percent cover score
167 using a gridded quadrat overlaid on the frame. Taxa were identified to the highest
168 taxonomic level possible and taxonomically alike species, which could not be
169 differentiated with confidence, were grouped (e.g., sponges grouped by life form i.e.

170 branching/massive/encrusting). Using this method, species of a broad range of sizes can
171 be enumerated, down to centimetre-scale. However, since the method relies on visual
172 identification, infauna and organisms inhabiting the underside of rocks will not be
173 detected. The quadrat area was calculated for every frame based on the position of the
174 lasers within the digital overlay, and subsequently used to derive density (individuals m⁻²).
175 Identified species were cross-referenced against a database of non-native species for
176 Great Britain (GB Non-native Species Secretariat, 2018).

177 *Response metrics*

178 Development of the epibenthic assemblage on the cable, relative to the controls, was
179 assessed using the following metrics: Assemblage Composition, Number of Taxa and
180 Abundance. To further examine the variation in assemblage development for the main
181 phyla observed, taxa were grouped by phylum and subdivided further by morphology
182 when members of the same phylum displayed clear functional differences (hereafter
183 “morphotaxa”).

184 Cnidaria were split into morphotaxa sub-group, Anthozoa and hydroids, since the latter
185 typically show fast rates of seasonal growth and provide a different level of biogenic
186 habitat complexity (Bradshaw *et al.*, 2003), compared to Anthozoans such as corals and
187 anemones. Chordata was further divided into Vertebrata and Tunicata to distinguish
188 their functional roles as; mobile predators and omnivores, and sessile habitat forming
189 filter feeders. Turf communities were categorised as one group as they consist of mixed
190 phyla, which cannot be separated using video analysis. Morphotaxa groups in this study
191 consisted of Porifera, Anthozoa, Echinodermata, Tunicata, Vertebrata and Turf.

192 *Environmental context*

193 To assess the impact scale of the cable relative to the surrounding environment, the
194 area of the cable, and the spatial extent of the habitat upon which it was laid were
195 calculated in Esri ArcMap 10.4. Broad-scale habitat data were obtained from the
196 European Marine Observation Data Network Seabed Habitats project (EMODnet, 2016),
197 and a bioregion was delineated based on; the aspect of the coastline, water depth and
198 extent of contiguous habitat, consistent with that upon which the cable was laid.

199 *Data analysis*

200 Permutational multivariate analysis of variance (PERMANOVA+ in the PRIMER v7
201 software package) (Anderson, 2001; Clarke and Gorley, 2006) was used to determine
202 whether the epibenthic assemblage was statistically significantly different between the
203 cable and east and west control sites over time. Three factors were used to examine
204 differences between the cable and nearby controls: Year (fixed; 2012, 2014, 2015),
205 Treatment (fixed; Cable, Control E, Control W), and Site (fixed 1-9), with three replicate
206 transects "Plots". Each Plot x Treatment combination constituted the average of five
207 random, replicate frame grabs. On occasion, five replicate frame grabs were not possible
208 due to the limited width of the cable cross section. In these circumstances, fewer were
209 averaged, rather than compromising the quality of the replicates. PERMANOVA was
210 utilised as it is robust to datasets with many zeros, allows the testing of interactions in
211 multivariate and univariate data, makes no assumptions about underlying data
212 distributions, and is robust to unbalanced sampling designs (Walters and Coen, 2006).

213 Number of Taxa was analysed without prior transformation, while all other response
214 metrics were fourth root transformed. Assemblage composition was based on Bray
215 Curtis similarity (Bray and Curtis, 1957) while all univariate response metrics were based
216 on Euclidean distance (Anderson and Millar, 2004). Significant differences between Year

217 x Treatment or just Treatment were further investigated using pairwise tests and for the
218 multivariate data, visualized using nonmetric multi-dimensional scaling (nMDS). Species
219 driving the nMDS were examined using SIMPER (similarity percentages) (Clarke and
220 Warwick, 2001) and the level of dispersion between sites for each year and treatment
221 combination was calculated using MVDISP (Clarke and Gorley, 2006).

222 **RESULTS**

223 *Wave Hub site*

224 The total area of the cable installation is approximately 0.036 km², and was installed
225 predominantly upon circalittoral rock and biogenic reef, a habitat which covers 395 km²
226 of the 914 km² bioregion (Figure 3), while 338 km² of the bioregion is circalittoral coarse
227 sediment. To the northeast of the bioregion is an extensive area of circalittoral coarse
228 sediment, with occasional circalittoral rock and biogenic reef.

229 To consider the installation and receiving environment in more detail, control plots
230 comprised cobbles and boulders on pebbly sand with the occasional rocky outcrop
231 (Figure 2), while the rock armouring used on the cable installation comprises boulders
232 which appear to be granitic, and in the size range of 100 to 200 mm diameter. Thus the
233 boulders used for the cable armouring appeared to be similar to the natural hard
234 substrate in the area. Concrete mattressing was not observed at any of the surveyed
235 cable sites.

236 *Assemblage composition, Taxa and Abundance*

237 Overall, 80 taxa from 11 phyla were identified. However, these were not uniformly
238 distributed across treatments. The epibenthic Assemblage composition on the cable was
239 distinct from that of the control sites throughout the period of the study, but was most

240 dissimilar in 2012. The Assemblage composition dispersion index within cable plots was
241 lowest in the first survey year (2012) and became more dispersed and more similar to
242 the controls over time (Table S1). Despite a Year x Treatment interaction ($p < 0.0001$)
243 (Figure 4a, Table 1), suggesting that the level of difference varied over time between
244 treatments, pairwise tests showed that the cable was significantly different to both
245 controls (E and W) in all three years (2012, 2014 and 2015), while the east and west
246 controls did not differ significantly from each other (Table 1). 29 taxa accounted for 90%
247 of the variation in the assemblages on the cable and in the controls across the years. The
248 main taxa driving these differences were predominantly *Alcyonidium diaphanum*,
249 *Cellepora pumicosa*, *Nemertesia antennina*, Hydroids spp. and Turf (SIMPER, Table S2).
250 Hydroids spp., Turf and the bryozoans *Alcyonidium diaphanum* and *Cellepora pumicosa*
251 were all more abundant on the cable than in the controls in 2012, collectively
252 contributing 60.5 % of the variation in Assemblage composition. However, by 2014 they
253 were all more abundant in the controls, apart from Turf, which remained more
254 abundant on the cable. In 2015, the group Hydroids spp. were more abundant on the
255 cable than the controls, while the hydroids *Nemertesia ramosa*, *Nemertesia antennia*
256 and *Halecium halecinum* were still in greater abundance in the controls. In 2012, 2 years
257 after deployment, the bryozoan *Pentapora foliacea* was not found on the cable, only in
258 the controls. However, in 2014 and 2015 this species was more abundant on the cable
259 than the controls (Table S2). Encrusting sponges were also more abundant on the cable
260 than the controls for all years, while branching sponges and species such as *Sycon*
261 *ciliatum* and *Polymastia boletiformis* were found to be more abundant in the controls in
262 2012 and 2014.

263 Many of the mobile fauna were observed in relatively low abundances, but still made a
264 small contribution to differences between treatments in the Assemblage composition;

265 including nudibranchs , the queen scallop *Aequipecten opercularis* (recorded in 2012 in
266 the east control only), the echinoderms *Echinus esculentus*, *Henricia oculata*,
267 *Marthasterias glacialis*, *Ophiocomina nigra*, *Ophiothrix fragilis* , and the goldsinny
268 wrasse *Ctenolabrus rupestris*, which in 2015 was more abundant on the Cable compared
269 to the controls (Table S2).

270 The Number of taxa was significantly greater in control sites than on the cable route ($p <$
271 0.001 ; Cable = 8.56 ± 0.44 SE ind. m^{-2} , Control E = 11.5 ± 0.50 SE ind. m^{-2} , Control W =
272 11.2 ± 0.54 SE ind. m^{-2}) (Figure 4b, Table 1). Conversely, Abundance was not significantly
273 different between treatments (Figure 4c, Table 1), and most variation was attributable
274 to spatial and temporal change.

275 The occurrence of non-natives at the Wave Hub site was low. Such species were only
276 identified on three separate occasions; the sea squirt *Styela clava* was recorded once on
277 the cable route and once in the controls in 2015 and another sea squirt *Molgula*
278 *manhattensis* was also recorded once on the cable route in 2015.

279 *Morphotaxa*

280 The six selected morphotaxa exhibited four general patterns of colonisation during the
281 five year study period; 1) Early colonisation, remaining in greater abundances on the
282 cable, 2) early colonisation, with assemblage convergence between cable and controls,
283 3) slow colonisation, with assemblage convergence between cable and controls and 4)
284 slow colonisation, remaining lower in abundance on cable.

285 Porifera exhibited the first colonisation pattern, being significantly more abundant on
286 the cable than the controls ($p < 0.001$; Cable = 1.67 ± 0.09 SE ind. m^{-2} , Control E = $1.33 \pm$
287 0.08 SE ind. m^{-2} , Control W = 1.19 ± 0.09 SE ind. m^{-2}) (Figure 5a; Table S3), and while the

288 abundance of cable Porifera appears to be converging with controls, there was no
289 significant effect of Year x Treatment.

290 The second colonisation pattern was exhibited by Turf. There was significantly more Turf
291 on the cable compared to the controls in 2012, two years after deployment, however by
292 2014 there was no significant difference between the cable and either controls (Figure
293 5b; Table S3). This similarity between cable route and controls was also apparent in
294 2015 (Year x Treatment $p < 0.01$; 2012: Cable = 2.71 ± 0.04 SE ind. m^{-2} , Control E = $1.84 \pm$
295 0.24 SE ind. m^{-2} , Control W = 1.48 ± 0.28 SE ind. m^{-2} ; 2015: Cable = 2.00 ± 0.13 SE ind. m^{-2} ,
296 Control E = 1.97 ± 0.11 SE ind. m^{-2} , Control W = 1.96 ± 0.12 SE ind. m^{-2}).

297 Anthozoa and Vertebrata displayed the third colonisation pattern. Anthozoa increased
298 on the cable from 2012-2015 (Figure 5c; Table S3), converging with the east control in
299 2014 and both controls by 2015 (Year x Treatment $p < 0.05$; 2012: Cable = 0.15 ± 0.1 SE
300 ind. m^{-2} , Control E = 1.61 ± 0.29 SE ind. m^{-2} , Control W = 1.01 ± 0.19 SE ind. m^{-2} ; 2015:
301 Cable = 0.89 ± 0.14 SE ind. m^{-2} , Control E = 1.26 ± 0.16 SE ind. m^{-2} , Control W = $0.93 \pm$
302 0.14 SE ind. m^{-2}). Vertebrata expressed a similar trend to Anthozoa, with increasing
303 abundance on the cable throughout the study (Figure 5d; Table S3), converging with
304 controls, however this trend was not significant.

305 The last colonisation pattern observed, was that of Tunicata and Echinodermata.
306 Tunicata were significantly more abundant in the controls compared to the cable ($p <$
307 0.05 ; Cable = 0.42 ± 0.12 SE ind. m^{-2} , Control E = 0.95 ± 0.21 SE ind. m^{-2} , Control W = 0.99
308 ± 0.22 SE ind. m^{-2}) and this trend was consistent throughout the study period (Figure 5e;
309 Table S3). Echinodermata (Figure 5f; Table S3) also exhibited this trend ($p < 0.05$; Cable =
310 0.58 ± 0.09 SE ind. m^{-2} , Control E = 0.98 ± 0.18 SE ind. m^{-2} , Control W = 1.05 ± 0.22 SE
311 ind. m^{-2}).

312

313 **DISCUSSION**

314 The aim of this study was to utilise an efficient monitoring method to assess
315 development in epibenthic assemblages and detect whether change was attributable to
316 the installation of a subsea cable and associated coverings. The cable and surrounding
317 seabed habitat supports a diverse range of epifaunal species (80 taxa across 11 phyla).
318 However, these are likely to be just a subsection of the entire community (Howarth *et*
319 *al.*, 2015), since only larger animals visible with the video array were recorded during
320 the survey, and infauna and some mobile species are not enumerated using this
321 method.

322 *Environmental context*

323 The cable was laid into a 395 km² area of circalittoral rock and biogenic reef (Figure 3)
324 within a wider bioregion of 914 km². The footprint of the cable infrastructure on the
325 seabed is approximately 0.036 km², or 0.009 % of this habitat area, but the overall
326 habitat space added to the system is likely to be orders of magnitude greater due to the
327 surface complexity of individual boulders and the installation as a whole. Small scale
328 habitat heterogeneity, such as the rugosity and surface roughness of introduced
329 substrate, increases habitat complexity and can promote species diversity (Luckhurst
330 and Luckhurst, 1978; Firth *et al.*, 2013) by providing a greater number of physical niches
331 for organisms to inhabit (Hixon and Beets, 1989; Langhamer and Wilhelmsson, 2009).
332 Decommissioning plans should acknowledge the relative contributions of each MREI
333 component to the habitat, alongside the spatial scale of the installation (Wilding *et al.*,
334 2017).

335 *Patterns of succession*

336 After five years deployment, Assemblage composition on the cable was still significantly
337 different to Assemblage composition in the controls. However, cable Assemblage
338 composition more closely resembled controls by 2014 and 2015 (Figure 4a). Owing to
339 the similarity in substrate between the cable rock armouring and surrounding habitat,
340 the colonising species on the cable were also present in the controls, despite significant
341 differences in Assemblage composition between treatments. Alternatively, when
342 artificial structures are introduced into the marine environment which are not
343 comparable with the surrounding natural habitat, a different species complex may result
344 (Evans *et al.*, 2015). Colonisation will also depend on the structure of the community in
345 the habitat surrounding the disturbed site (i.e. the cable route), as this supplies adults
346 and larvae for recolonization (Mazik and Smyth, 2013), and on the timing of the
347 disturbance in relation to larval supply (Osman, 1977).

348 Ecological succession was evident on the cable rock armouring from the first survey, two
349 years after its deployment. Turf was the dominant morphotaxa on the cable in 2012 and
350 was found in significantly greater abundance on the cable than in the east and west
351 controls (Figure 5b). By 2014 and 2015 the amount of Turf on the cable route was
352 becoming more similar to the control sites and was not found to differ significantly.
353 Sessile taxa such as Turf and Hydroids spp. are quick to establish (Jackson *et al.*, 2008;
354 Antoniadou *et al.*, 2009; Langhamer *et al.*, 2009), so early colonisation of the cable was
355 expected. Such early successional species then provide opportunities for larger
356 organisms, including mobile fauna, to colonise the habitat (Langhamer *et al.*, 2009).

357 The process of succession is slow and continuous and susceptible to local natural
358 disturbances (Antoniadou *et al.*, 2009). The 2014 surveys took place after a winter of

359 severe storms (Masselink *et al.*, 2016), which were responsible for the shifts in
360 abundance of benthic organisms in other areas of south west England (Sheehan *et al.*,
361 unpublished data). The effects of these storms may still have been present during the
362 2014 survey at the Wave Hub site and consequently affected the abundance of
363 organisms recorded that year.

364 The Number of taxa was found to be greater in the controls than on the cable (Figure
365 4b), yet it is not simply the Number of taxa which drives productivity or resilience within
366 a temperate reef system. Fauna which provide shelter from fish predation, greater
367 surface area, and structure for anchorage are also shown to be important factors
368 (Taylor, 1998). While encrusting species have been found to stabilise structures and act
369 as a substrate for the settlement of larvae that in turn create reef habitat for other
370 species (Bradshaw *et al.*, 2003; Bell, 2008), branching life forms create structurally
371 complex microhabitats and food sources for mobile macrofauna (Wulff, 2006). The
372 relative abundance of encrusting and low-lying organisms, and more structurally
373 complex taxa, suggests that the cable does not yet exhibit biogenic structures, typical of
374 an established temperate reef ecosystem. For example, while porifera were more
375 abundant on the cable route than in controls (Figure 5a), this was composed primarily of
376 encrusting species while the controls supported greater numbers of branching species.

377 Early sessile colonisers such as Hydroids spp., which were more abundant on the cable
378 than in controls in 2012 (Table S2), also provide biogenic structure (Turner *et al.*, 1999)
379 that increase the net three-dimensional habitat available for mobile species (Ferrari *et*
380 *al.*, 2016). The structurally complex bryozoan *Pentapora foliacea* was not found on the
381 cable route two years after deployment but by the fourth year was found across all
382 treatments. This species is typically slow growing and long lived (Jackson *et al.*, 2008)
383 and its presence on the cable rock armouring may indicate a trajectory towards a more

384 structurally complex habitat, that will in turn provide habitat for other species (Ferrari *et*
385 *al.*, 2016). However, there was no Treatment effect for Vertebrata indicating that the
386 cable route was not having an artificial reef effect within five years, and monitoring over
387 longer timescales would be needed to test this effect for mobile species.

388 While modelled and observational evidence exists for marine renewables infrastructure
389 acting to facilitate marine invasions (Adams *et al.*, 2014; De Mesel *et al.*, 2015), the
390 presence of hard substratum in the vicinity of the rock armouring on the cable route
391 suggests that this is unlikely to be of relevance at Wave Hub. This is supported by the
392 congruence of benthic assemblages on the rock armouring and control sites (Figure 4a),
393 and low number of records for non-native species over the period of the study.

394 *Monitoring approach*

395 To be sufficiently robust to inform the planning and decommissioning processes for
396 MREIs, data need to be gathered using an effective protocol at relevant spatial and
397 temporal scales, incorporating appropriate control sites, to allow detection of effects of
398 MREIs against a background of natural variability (Wilding *et al.*, 2017). Coastal
399 ecosystems are particularly vulnerable to natural environmental fluctuations as well as
400 anthropogenic pressures, both of which may make detection of temporal trends more
401 difficult (Elliott and Whitfield, 2011). It is therefore important to incorporate spatially
402 relevant control sites into survey design to facilitate detection of trends (Underwood,
403 1997).

404 It is also important to take any historical and present anthropogenic activity into account
405 when determining suitable control sites, as shifting baselines in the local environment
406 and varying recovery rates for the different response metrics can confound assessment
407 of assemblage development. This case study focused on the patterns of succession on

408 the cable rock armouring, by comparing response metrics to those in the surrounding
409 environment of the nearby control sites. The sea bed at the Wave Hub site has been
410 subjected to historical fishing pressure, predominantly from static gear (Campbell *et al.*,
411 2014), likely due to the rocky nature of the habitat being incompatible with most
412 bottom towed fishing methods. Therefore it is likely that these control sites would have
413 been exposed to past disturbance. Static fishing gear was observed on the cable itself
414 whilst survey work was ongoing (Sheehan, personal observation), hence any comparison
415 of the cable is to the existing state of the surrounding habitat, rather than a pristine
416 environment. However, control sites were considered to be suitable for comparing the
417 development of the community on the cable, as the local diversity appeared well
418 developed, with a diverse range of upright biogenic habitat forming organisms; typically
419 not present in areas that are subjected to bottom towed gear (Bradshaw *et al.*, 2003;
420 Sheehan *et al.*, 2013a).

421

422 *Impact of decommissioning*

423 During the lifetime of the Wave Hub cable installation, as with other MREIs, there is a
424 potential net habitat gain (Smyth *et al.*, 2015). Five years after deployment, the cable is
425 supporting an epibenthic community which is becoming congruent with the surrounding
426 ecosystem. The cable is not intended to be removed until 2037, after which time the
427 rock armouring will have been in place for 27 years and it is likely that any colonisation
428 will be advanced and productive after this time (Smyth *et al.*, 2015). Complete
429 decommissioning would effectively be removing this habitat and any associated
430 ecosystem services that it provides. Long lived species and complex habitats may take
431 decades to recover (Duarte *et al.*, 2015), so only long term monitoring will provide

432 conclusions as to how recovered the ecosystem can really be expected to become.
433 However, it must be borne in mind that the ecosystem will be returning to a pre-cable,
434 rather than a pristine state, acknowledging anthropogenic activities such as fishing
435 pressure in the region.

436 Despite statistically significant differences remaining between the cable and controls,
437 similarities between epibenthic assemblages at these sites suggests that the material
438 chosen for the rock armouring was appropriate for installation in this area. This pattern
439 may not be apparent in all marine renewable installations as colonisation varies with the
440 material of the artificial habitat (Firth *et al.*, 2016).

441 Our results indicate that the cable may eventually align with the controls but further
442 data collection would be required to ascertain time frames for these to converge, and
443 infer at which point it would no longer be ecologically justifiable to remove structures,
444 such as the cable and associated rock armouring from the marine environment. In some
445 instances partial decommissioning would be recommended, for example where the
446 assemblage on installed infrastructure differs significantly from that of the receiving
447 environment at the time of decommissioning, which may occur when hard substrate is
448 introduced to a soft sediment habitat. In such cases partial decommissioning may be
449 considered, as complete removal of infrastructure would remove biomass and habitat
450 from the system. Where non-native species are evident on infrastructure in elevated
451 abundance, complete decommissioning would be recommended. Where assemblages
452 on installed infrastructure are consistent with the surrounding system, partial, or no
453 decommissioning may also be favoured. Such decisions should take into account inter-
454 annual variability in assemblage, taxa and abundance since this may be considerable, as
455 seen in the controls at Wave Hub (Figure 4).

456 Complete decommissioning of MREI components on hard substrates would effectively
457 set back epibenthic assemblages by the length of the lifetime of long lived, habitat
458 forming organisms, which may be at least 30 years for some species (Jackson *et al.*,
459 2008). There is also an important distinction to be made between sites where rock
460 armouring was introduced to a natural hard substrate, and situations where the
461 receiving habitat is lacking prominent or hard substrates. Since hard substrates
462 introduced to soft substrates such as sand are likely to yield greater gains in terms of
463 biodiversity enhancement, the consequences of decommissioning in these scenarios
464 could be negative and permanent. However, where cables are laid on softer substrates it
465 is likely that trenching would negate the need for rock armouring in the first instance.

466 Similar conclusions were reached when considering the decommissioning of offshore oil
467 platforms, resulting in oilrigs being partially decommissioned and left as artificial reefs
468 rather than being completely removed at the end of their serviceable life (Kaiser and
469 Pulsipher, 2005). It is likely that partial decommissioning strategies for MREIs could offer
470 environmental and commercial benefits, as complete removal is not only expensive but
471 threatens to damage the seabed and any benthic communities that have developed
472 since installation.

473 **Conclusions**

474 Although thorough decommissioning planning is part of the consenting process for all
475 MREIs, there is little evidence as to what environmental impacts might occur. Case
476 studies such as this can help marine managers plan for an installation and
477 decommissioning programme that will least disturb any habitat development over the
478 useful lifetime of MREIs, maximising the ecological potential of installations. Using
479 appropriate materials for the receiving habitat will preserve ecosystem function and

480 associated services as well as protect marine habitats. Our results show that recovery on
481 the cable, which lies in a temperate reef ecosystem, is relatively fast compared to the
482 operational life of these structures, which are typically in the marine environment for 20
483 – 30 years. Initial findings from studying the assemblage development up to five years
484 after deployment of a subsea cable, indicate that the disturbed site is showing signs of
485 recovery and is becoming more comparable to the surrounding environment. Therefore,
486 a partial decommissioning strategy (over full removal) for this site and similar
487 installations would be supported. However, longer term monitoring would be required
488 to assess whether recovery trajectories continue as expected and determine the extent
489 of the consequences that decommissioning might have. Choosing appropriate materials
490 and methods when designing MREIs could allow such installations to contribute to the
491 wider national conservation aim of linking a coherent network of marine protected
492 areas around the UK, and allow for the continued expansion of existing reef-like
493 habitats. If left in the marine environment, these structures have the potential to
494 restore degraded habitats and could make a further contribution to conservation
495 measures, if the areas around MREIs were fully protected from anthropogenic pressures
496 such as the use of bottom towed fishing gear. However, the location, substrate, type
497 and composition of MREIs and their components make decommissioning guidelines for
498 these devices applicable on a site-by-site basis, and a rigorous, effective and replicable
499 monitoring approach that accounts for varying spatial and temporal scales must be
500 employed as future installations develop to commercial scale.

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Table 1. Summary of PERMANOVA results for Assemblage composition, Number of Taxa and Abundance, alongside results of pairwise tests for significant differences of interest. ** Term has one or more empty cells.

Source	df	SS	Pseudo-F	P(perm)	Pairwise comparison	t	P(perm)	t	P(perm)	t
Assemblage					Ye x Tr	2012		2014		2015
Year (Ye)	2	2.24x10 ⁴	9.54	0.0001	Control W, Control E	0.831	0.7212	0.729	0.8262	0.799
Treatment (Tr)	2	1.77x10 ⁴	7.55	0.0001	Control W, Cable Route	3.07	0.0001	1.90	0.0028	2.19
Site (Si)	8	5.60x10 ⁴	5.96	0.0001	Control E, Cable Route	3.20	0.0001	1.73	0.0082	2.63
Ye x Tr	4	1.03x10 ⁴	2.18	0.0001						
Ye x Si	11	2.56x10 ⁴	1.98	0.0001						
Tr x Si	16	2.75x10 ⁴	1.46	0.0004						
Ye x Tr x Si	19	2.40x10 ⁴	1.08	0.2611						
Residual	101	1.19x10 ⁵								
Total	163	3.02x10 ⁵								
Number of taxa					Tr					
Ye	2	91.178	4.0628	0.0216	Control W, Control E	0.51168	0.6106			
Tr	2	305.14	13.597	0.0001	Control W, Cable Route	4.1762	0.0005			
Si	8	288.36	3.2122	0.0038	Control E, Cable Route	4.8883	0.0001			
Ye x Tr	4	9.0166	0.20088	0.9396						
Ye x Si**	11	36.893	0.29889	0.9821						
Tr x Si	16	355.88	1.9822	0.0206						
Ye x Tr x Si**	19	209.39	0.98214	0.4792						
Res	101	1133.3								
Total	163	2429.2								
Overall abundance										
Ye	2	4.13	3.31	0.0401						
Tr	2	1.80	1.45	0.2428						
Si	8	27.4	5.49	0.0001						
Ye x Tr	4	3.42	1.37	0.2588						
Ye x Si**	11	19.9	2.90	0.003						
Tr x Si	16	19.1	1.92	0.0313						
Ye x Tr x Si**	19	11.1	0.936	0.5316						
Res	101	63.0								
Total	163	150								

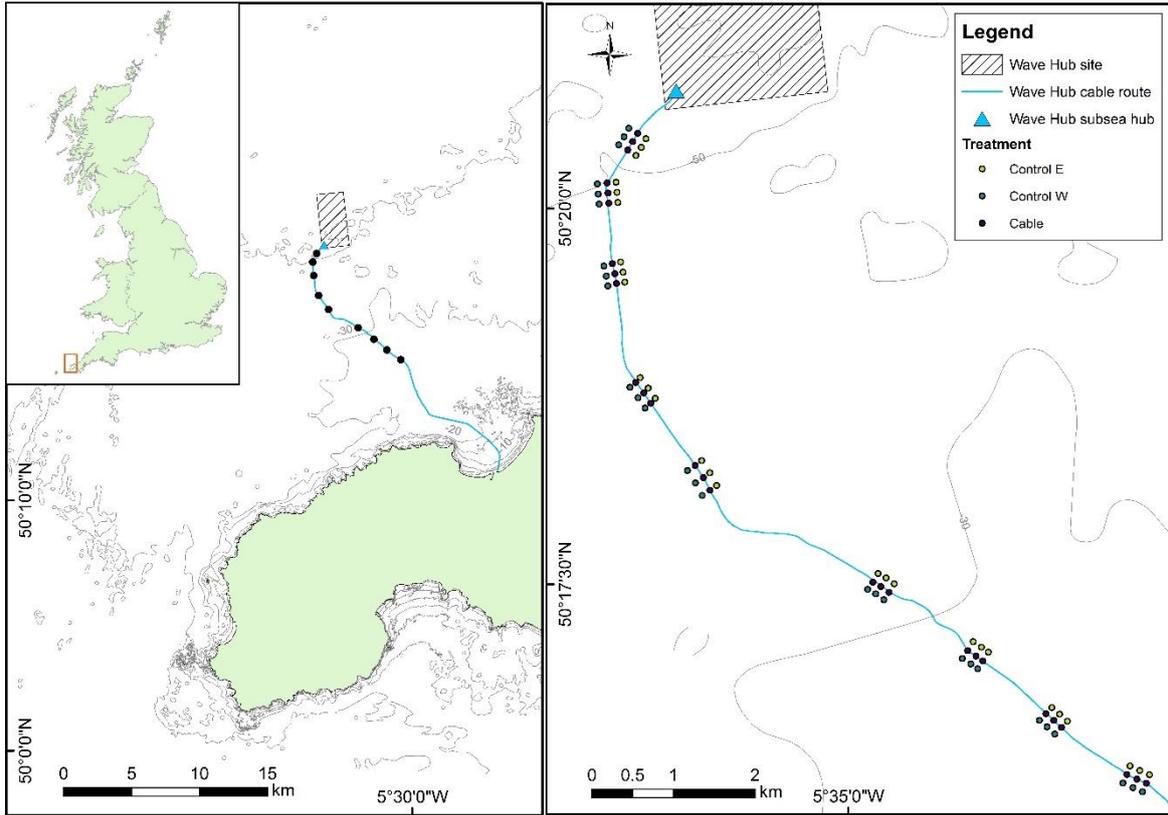


Figure 1: Site map of the Wave Hub site in the southwest of the UK, with detail of survey design along submarine cable route, with controls to the east and west.

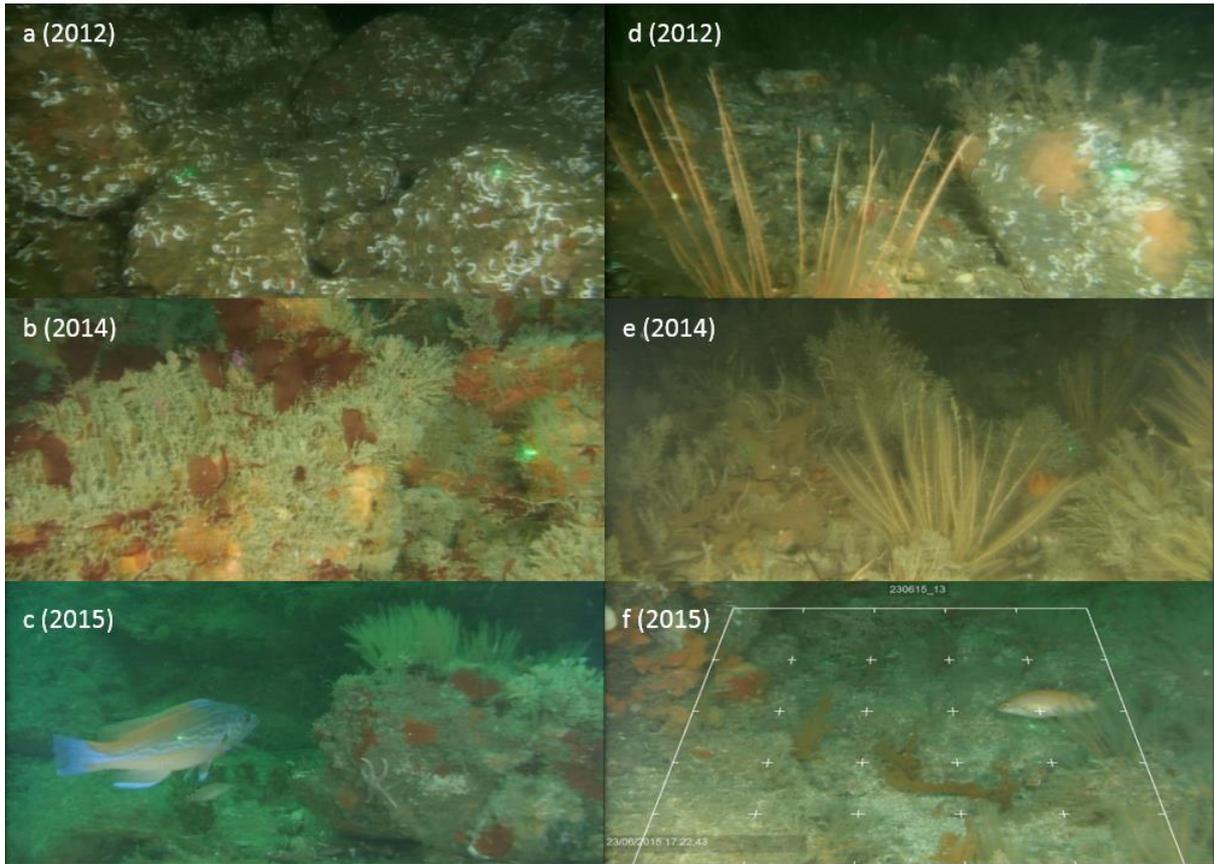


Figure 2: Example images from transect footage from cable rock armouring (a, b, c) and control sites adjacent to the cable route (d, e, f). Image f also depicts the digital quadrat overlay that was applied to each frame grab during the video analysis. Laser spacing (green dots) = 0.3 m.

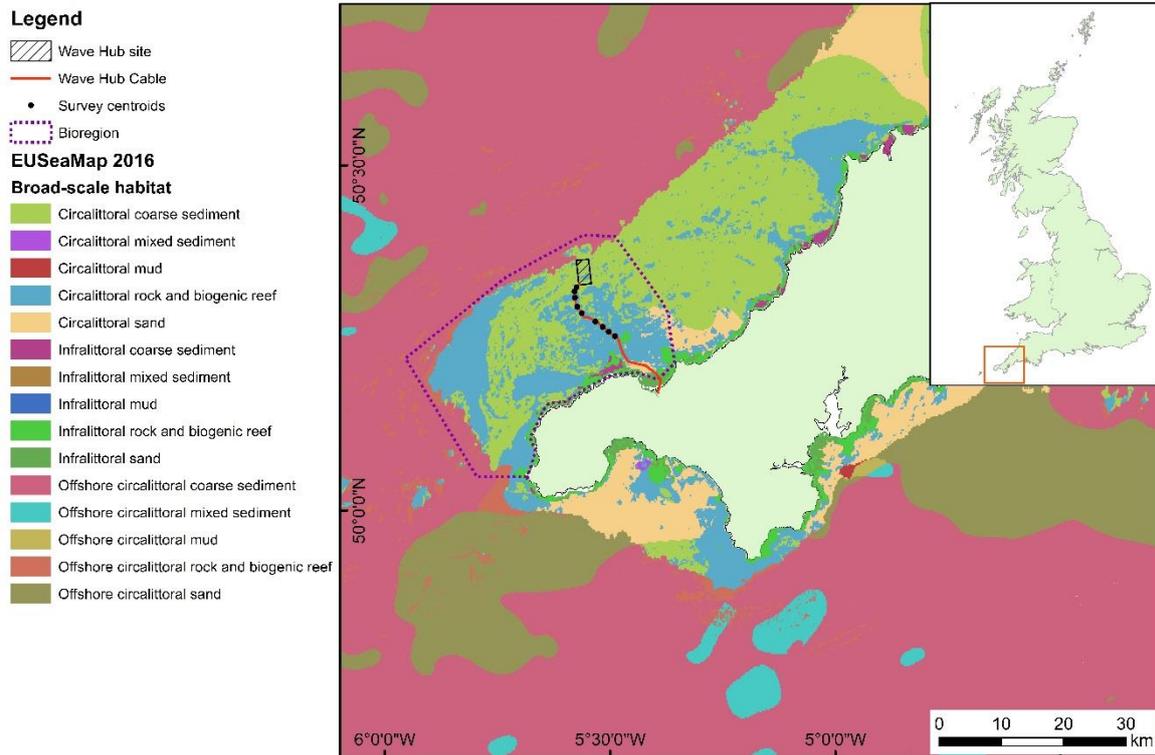


Figure 3: Broad scale habitat of the southwest coastal region of the UK (EMODnet, 2016). The bioregion is located on the northwest-facing coastline, extending out to the 50 metre contour. The eastern and western boundaries are located at the edge of the contiguous circalittoral rock and biogenic reef.

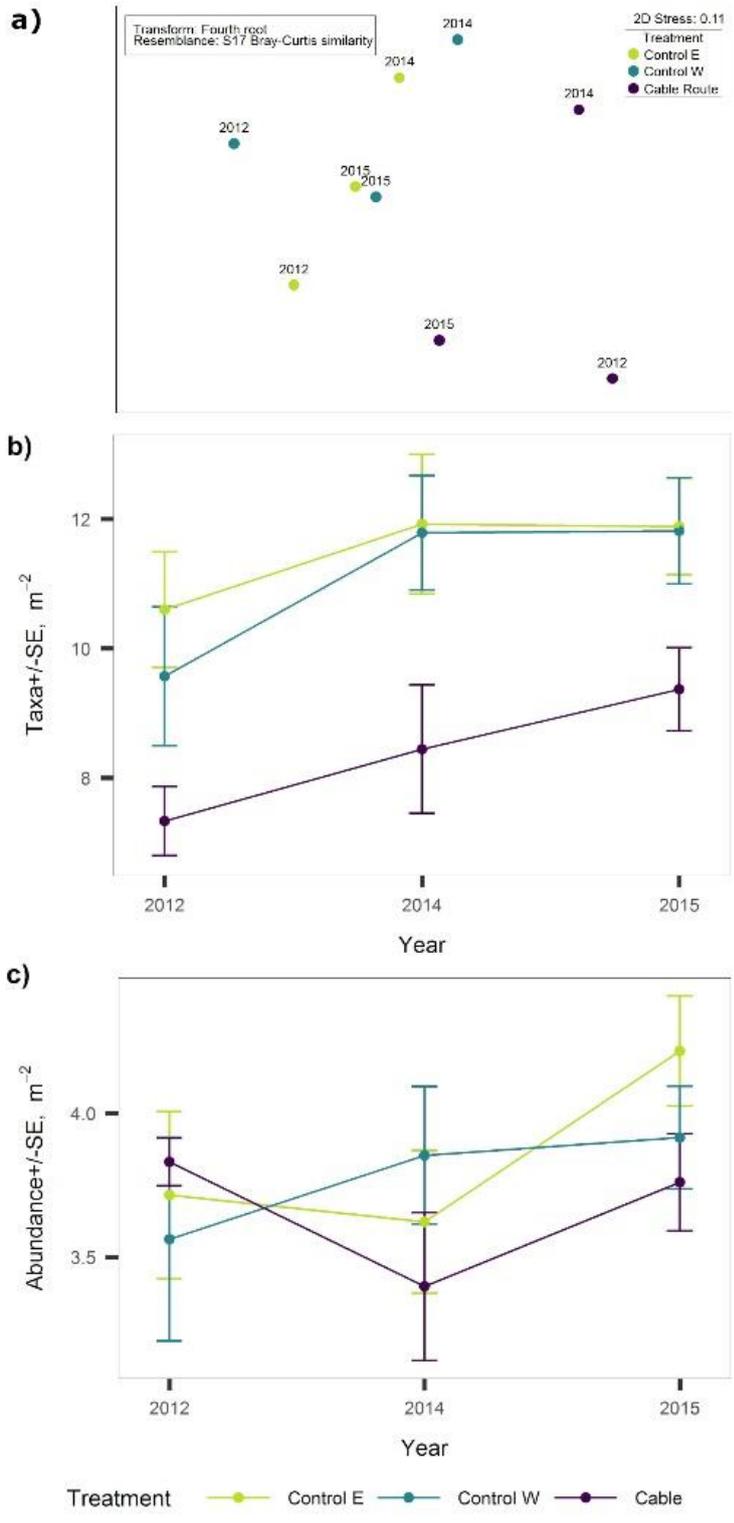
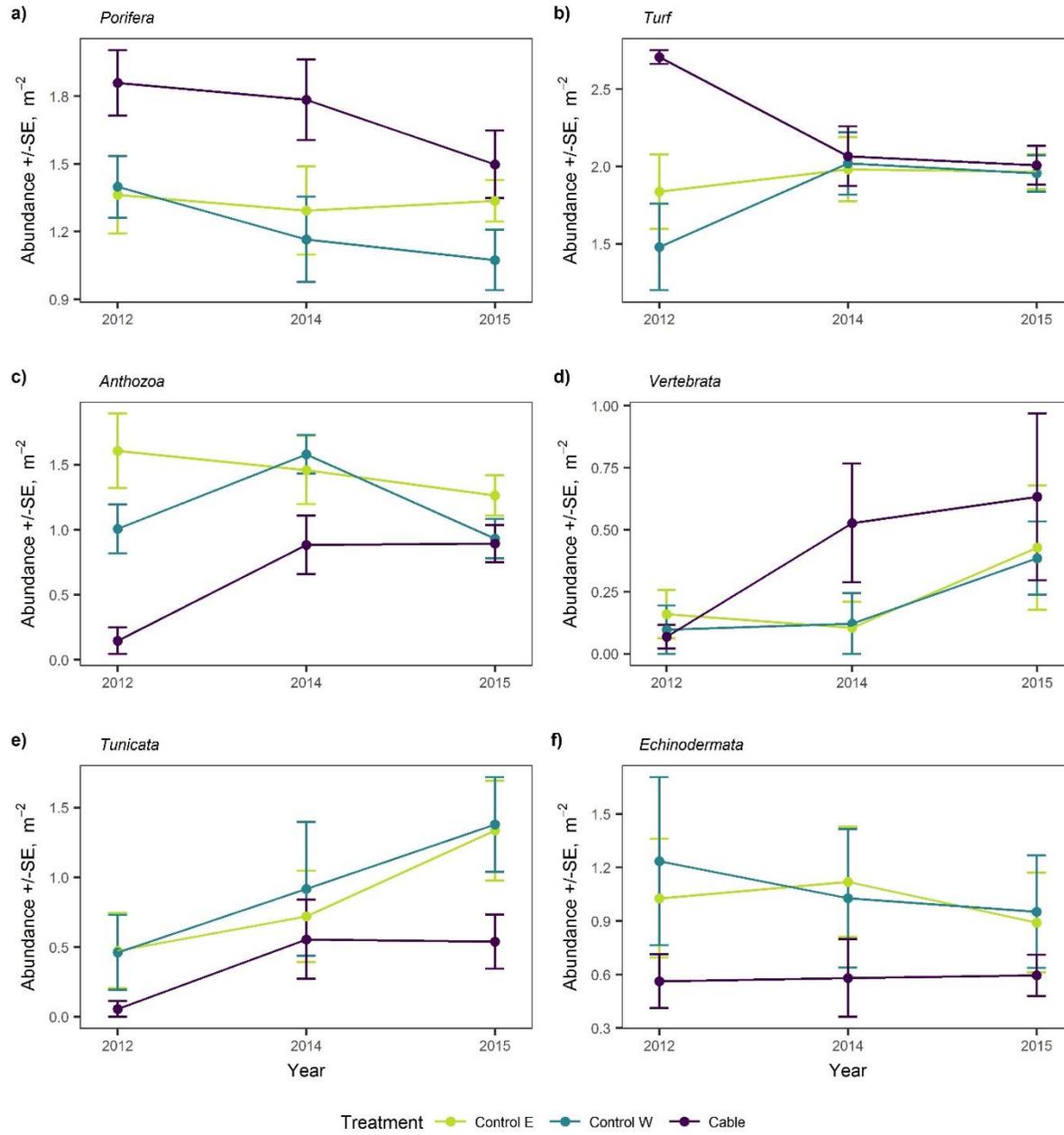


Figure 4: a) nMDS of benthic assemblage across the three treatments at the Wave Hub site over the period 2012 to 2015. Each point represents the Assemblage composition across all sites within each treatment in each year; b) Number of taxa; c) Overall abundance. Assemblage and Abundance data fourth-root transformed. Error bars denote Standard Error.



2 Figure 5: Mean abundance of phylomorphs within different treatments of the Wave Hub Cable over the three survey years, up to five years post-
3 deployment of submarine cable. Fourth-root transformed data are shown; error bars denote Standard Error.

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