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Assessing the impact of introduced infrastructure at sea with cameras: A case study for spatial scale, time and statistical power

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1 Assessing the impact of introduced infrastructure at sea with cameras: a case study for spatial
2 scale, time and statistical power

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24 **Abstract**

25 Detecting the effects of introduced artificial structures on the marine environment relies upon
26 research and monitoring programs that can provide baseline data and the necessary statistical
27 power to detect biological and/or ecological change over relevant spatial and temporal scales.
28 Here we report on, and assess the use of, Baited Remote Underwater Video (BRUV) systems
29 as a technique to monitor diversity, abundance and assemblage composition data to evaluate
30 the effects of marine renewable energy infrastructure on mobile epi-benthic species. The
31 results from our five-year study at a wave energy development facility demonstrate how
32 annual natural variation (time) and survey design (spatial scale and power) are important
33 factors in the ability to robustly detect change in common ecological metrics of benthic and
34 benthic-pelagic ecosystems of the northeast Atlantic. BRUV systems demonstrate their
35 capacity for use in temperate, high energy marine environments, but also how weather,
36 logistical and technical issues require increased sampling effort to ensure statistical power to
37 detect relevant change is achieved. These factors require consideration within environmental
38 impact assessments if such survey methods are to identify and contribute towards the
39 management of potential positive or negative effects on benthic systems.

40

41 **Keywords**

42 Marine monitoring, human impact, renewable energy, power analysis, BRUV

43

44

45 **Introduction**

46 The marine coastal environment provides a major, and disproportionate, contribution to
47 global ecosystem services (Costanza et al., 1997; Drakou et al., 2017; Liqueste et al., 2013).
48 Through human activity (e.g. commercial fishing, shipping, resource extraction, aquaculture,
49 dredging) it has, however, become the most impacted region of our global seas (Halpern et
50 al., 2008). A growing coastal population is expected to increase human pressure on coastal
51 regions with further potential adverse effects on natural systems (Heery et al., 2017;
52 Millennium Ecosystem Assessment, 2005; Neumann et al., 2015).

53 Coastal benthic habitat has been particularly altered by destructive fishing practices (Eigaard
54 et al., 2015; Hiddink et al., 2017) and the introduction of artificial infrastructure (Bulleri and
55 Chapman, 2010). The loss or disturbance of benthic habitats is concerning as they play a
56 pivotal role in the provision and support of key ecosystem services, e.g. food provision,
57 nutrient cycling, reproduction/nursery areas, water quality, biodiversity maintenance
58 (Galparsoro et al., 2014). The expanding marine renewable energy sector has led offshore
59 wind farms to become a prominent part of coastal and shelf waters of multiple countries
60 (GWEC, 2016; WindEurope, 2017). These installations have direct impact on benthic
61 habitats (i.e. monopile drilling and foundations), and subsequent direct or indirect effects on
62 the associated epi-benthic faunal communities (Bailey et al., 2014; Gill, 2005; Inger et al.,
63 2009; Pearce et al., 2014; Stenberg et al., 2015). The promise of wave and tidal energy
64 conversion is still to be fully realised, but has the potential to further modify benthic habitats
65 around our coastlines (Langhamer and Wilhelmsson, 2009; Witt et al., 2012). Unfortunately,
66 the monitoring programs and environmental impact assessments (EIA) that have considered
67 interactions between marine renewable energy installations (MREIs) and benthic habitats or
68 species have, so far, lacked the necessary baseline characterisation or survey rigour (spatially
69 or temporally) to robustly assess impact (positive or negative) (Boehlert and Gill, 2010; Fox

70 et al., 2018; Wilding et al., 2017), and led authors to term this situation as ‘data rich,
71 information poor (DRIP)’ (Fox et al., 2018; Ward et al., 1986; Wilding et al., 2017).

72 The high energy environment needed for marine energy convertors to operate means they are
73 well suited to mid to high latitude seas where these conditions regularly occur. The required
74 strong winds, wave action and/or large tides provide physical challenges to access
75 development sites and conduct surveys to assess impact on benthic communities, which are in
76 addition to the inherent difficulties of underwater research. Remote camera imagery is a
77 technique that can overcome some of these challenges, and has already proven a valuable tool
78 for studying the impact of human activities on the marine environment (Bicknell et al., 2016;
79 Mallet and Pelletier, 2014; Sheehan et al., 2014). Baited Remote Underwater Video systems
80 (known as ‘BRUV’) are a method that uses either one (mono) or two (stereo) cameras to film
81 the area surrounding a bait attractant held a short distance from a video camera and close to
82 the seabed (also modified for mid-water; Heagney et al., 2007). The technique has been used
83 extensively in the southern-hemisphere (tropics to temperate) to evaluate changes in demersal
84 fish populations (e.g. Denny et al., 2004; Malcolm et al., 2007; Watson et al., 2009; Watson
85 et al., 2007), and has demonstrated its value in sampling fishes and invertebrates in high
86 latitude turbid coastal waters in the northern-hemisphere (Elliott et al., 2017; Unsworth et al.,
87 2014). Recently, the method has also shown its application in assessing the mobile epi-
88 benthic fauna at an offshore wind farm in the Irish Sea (Griffin et al., 2016), but as yet not in
89 multi- season or multi- year impact studies. The method has bias (e.g. differentially attracting
90 carnivores or omnivores, bait type and plume effects, restricted view, light
91 attraction/repulsion), as do most survey methods, and these have been investigated and
92 detailed elsewhere (Dorman et al., 2012; Harvey et al., 2018; Harvey et al., 2012; Harvey et
93 al., 2007; Stobart et al., 2007). However, it has advantages on many traditional methods, such
94 as being non-destructive, having no or limited observer bias, allowing re-analysis or review

95 of video (data) and is unrestricted by depth (cost-dependent) (Cappo et al., 2004; Lowry et
96 al., 2012; Whitmarsh et al., 2016; Zintzen et al., 2012). When BRUVs are used to provide an
97 estimate of species abundance, a number of metrics have been considered (Stobart et al.,
98 2015), but in the vast majority of cases (81% of reviewed studies, Whitmarsh et al., 2016)
99 N_{max} (or MaxN) is used. This represents the maximum number of a particular species seen in
100 any one video frame across the duration of the video footage. It is a useful metric to assess
101 the relative abundance of species and considered a conservative estimate as there may be
102 uncounted individuals around the BRUV that did not enter the field of view (Whitmarsh et
103 al., 2016).

104 For monitoring programs to effectively assess environmental impact they require baseline
105 data that characterise the natural spatial and temporal variability of the focal system or
106 component (Judd, 2012). The challenges are then to detect the potential effect of the
107 introduced impact from the natural ‘background noise’ (variability) (Osenberg et al., 1994),
108 and recognize whether any detected change is biologically, ecologically or functionally
109 meaningful (Wilding et al., 2017). In highly variable marine systems it is particularly
110 important to determine the level of sampling effort required to gather robust baseline data and
111 provide statistical power to detect a given degree of change (Franco et al., 2015; Osenberg et
112 al., 1994). Ideally, prior or pilot data would be available to provide knowledge on variability
113 within the system and enable a power analysis to be conducted, but these data are rarely
114 available or analyses conducted (Franco et al., 2015; Maclean et al., 2014). Surveys will often
115 be based on applying fundamental statistical principles to the design (Box 1), balanced with
116 time, costs and logistical or methodological constraints. Many environmental impact studies
117 focus on site characterisation during one or two years as opposed to deploying bespoke
118 survey strategies designed to identify putative effects with anticipated levels of change in
119 environmental receptor groups.

120

121

Effects of design and data on statistical power

The ability to detect patterns/change reduces as variability in the parameter being measured increases

↑variance = ↓power

Parameter estimates become more precise with larger samples, thus differences between estimates are easier to detect amongst the ‘noise’

↑sampling = ↑estimate precision

↑estimate precision = ↑power

As the effect of the impact increases the more likely it is to be detected. i.e. a 40% change in a parameter estimate is more likely to be detected than 20% change

↑effect size = ↑power

(Underwood and Chapman, 2003)

122

123 Here, we present a case study of BRUV use over five years in boreal latitude coastal waters
124 of the northeast Atlantic to provide baseline characterisation data and impact assessment on
125 mobile epi-benthic species at a MREI. We use these data to investigate the power to detect
126 change in conventional ecological metrics (species richness, abundance, and assemblage
127 composition), how well a survey design and sampling effort performs given there was little
128 prior ecological knowledge of the site, and whether any effects on the mobile epi-benthic
129 community could be detected. We consider the appropriateness of this technique for long-
130 term impact monitoring at MREI, and, more generally, discuss the results in relation to future
131 EIAs of mobile epi-benthic communities.

132

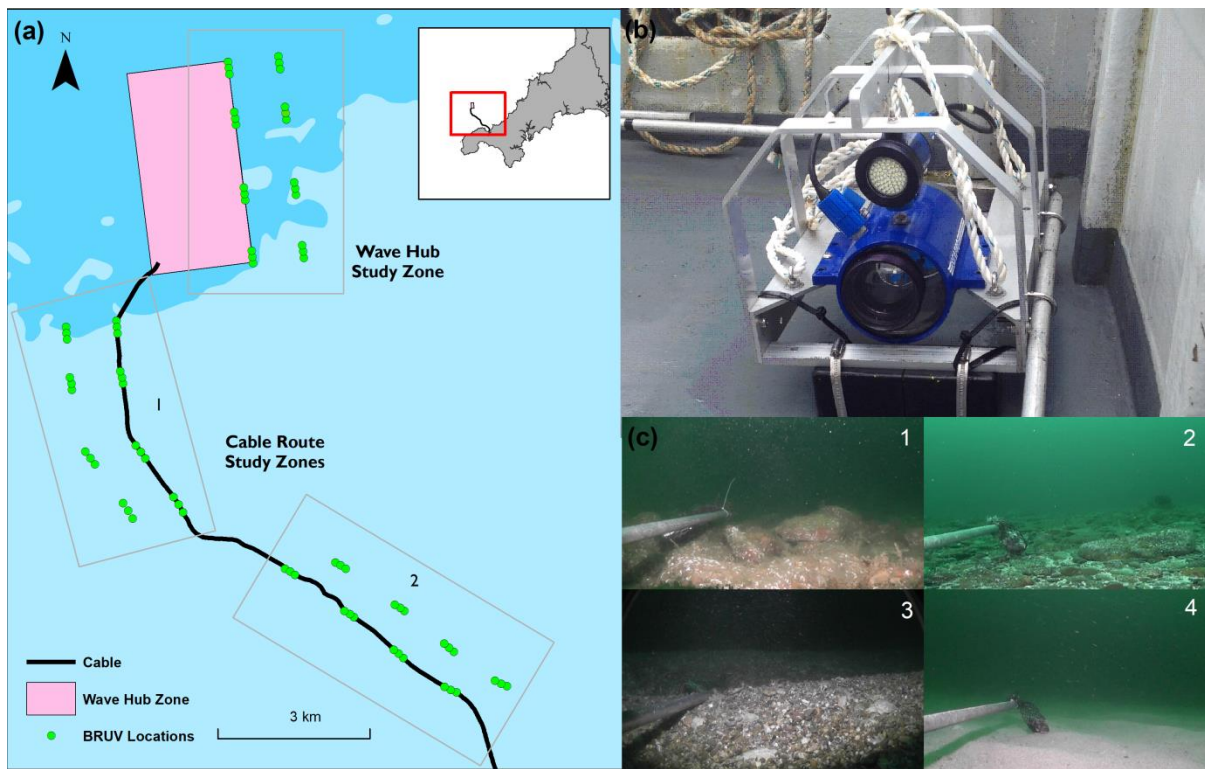
133 Materials and Methods

134

135 Study location

136 Baited remote underwater video (BRUV) surveys took place off the north coast of Cornwall
137 (UK) between 2011 and 2015. The study was located within and adjacent to a MREI
138 development zone (Wave Hub) and the associated seabed cable (Fig. 1a). Study zones ranged
139 between 3 and 10.5 nautical miles (5.6 and 19.5 km) offshore in water depths of 20 to 53 m

140 (at Lowest Astronomical Tide). The seabed cable and marker buoys (6 in total) for the
141 development zone were installed in autumn 2010. The cable was buried when on sand (near
142 shore) and covered with 80,000 tonnes of rock (creating a berm of 0.3 metres minimum
143 height) and concrete matressing every 120 metres when laid on hard substrate (deeper water).
144 Access within 500 metres of the electrical seabed hub (plug) was prohibited, but access was
145 permitted elsewhere. The presence of marker buoys has prevented commercial mid-water or
146 bottom trawling for fish. The only commercial fishing that has taken place in the
147 development zone or close to the seabed cable since installation is crustacean potting. There
148 were no connected deployments of operating marine energy devices during the period of the
149 study.



151 **Fig. 1.** Study zones and sampling locations (green filled circles) for BRUV surveys (a), BRUV
152 housing, frame and LED light (b), and example habitat types from the video footage (c) 1 = rocky
153 reef, 2 = large (course) sediment, 3 = medium (mixed) sediment, 4 = fine sediment.

154

155 **Sampling equipment**

156 Each BRUV consisted of an aluminium frame, wide-angle lens housing and white light LED
157 lighting system (Fig. 1b). An aluminium pole was attached to each BRUV to support bait
158 (fixed 1 metre from lens); lead weights (45 kg) were fastened to the frame for deployment
159 and stability over a wide range of tidal conditions (0.02 to 0.53 m^{-s}). Panasonic HDC-SD60
160 and HDC-SD80 camcorders were used to gather video data. A temperature depth recorder
161 (RBR, Nova Scotia, Canada) was attached to one BRUV in each sampling location during a
162 sampling campaign. Local small commercial leisure or fishing vessels were used for BRUV
163 deployment.

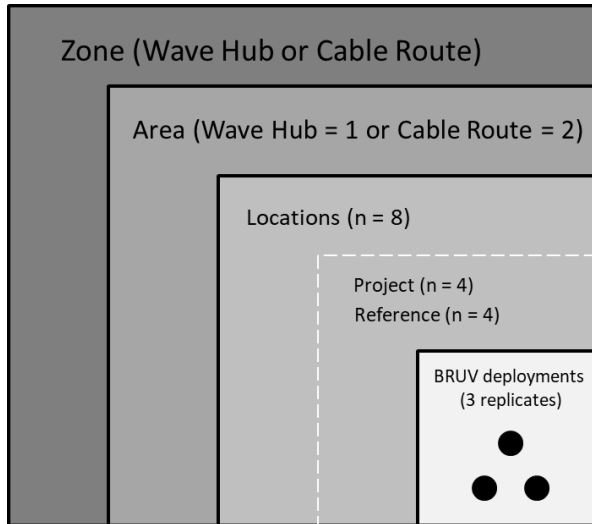
164

165 **Survey and design**

166 BRUV surveys were scheduled twice per year, in spring and autumn, commencing in autumn
167 2011 until 2015, and each survey campaign took three days to complete. The sampling design
168 consisted of: two study *zones* (Wave Hub = WHSZ and Cable Route = CRSZ); three *areas* (1
169 x Wave Hub, 2 x Cable Route); each area comprised 8 *locations* (four **project** and four
170 **reference** 1 km apart; Fig 1a). Three replicate BRUVs were deployed between 105-180 m
171 (mean ~130 m) apart in each location (Fig 2). **Project** locations were either inside the Wave
172 Hub exclusion zone or on/next to the seabed cable infrastructure (Fig 1a). *Treatment* will be
173 used as the term to describe the comparison between **project** and **reference** locations in the
174 subsequent analyses and models. BRUVs were deployed for up to 60 minutes during daylight
175 hours. Bait used was a single Atlantic mackerel *Scomber scombrus* for each deployment, cut
176 into three piece and held in a net bag (~100 g).

177 In order to investigate the BRUVs greatest distance of attraction (in metres) for teleosts, we
178 calculated the ‘effective range of attraction’ (AR) (formulised in Cappo et al., 2004; see
179 Appendix S3) for increasing soak time (i.e. time cameras are in the water, or video footage
180 analysed from the start). The average seabed current speeds (data from POLPRED seabed

181 CS20 models, NOC; <http://www.pol.ac.uk/>) during BRUV deployments (0.23 m s^{-1}), and a
182 maximum fish (endurance) swimming speed of 0.6 m s^{-1} (~200-300 mm fish length) were
183 used as the AR parameters (V_c = current speed, V_f = maximum fish speed).



184
185 **Fig. 2.** Nested design schematic for baited remote underwater video surveys.

186

187 **Image analysis**

188 Gathered video datasets were analysed (using a large monitor) to quantify species observed,
189 the number of mobile epi-benthic species (richness; S), the maximum number of individuals
190 of each species observed at the same time (N_{max}) and the time of each increment in N_{max}
191 (recorded in excel spreadsheets). The use of N_{max} as an estimator of relative abundance has
192 been assessed (Cappo et al., 2003; Ellis and DeMartini, 1995; Priede et al., 1994; Willis and
193 Babcock, 2000), and is considered a conservative estimate of abundance especially when
194 species occur at high density. Each video dataset was assigned a habitat type (1-rocky reef, 2-
195 large sediment [small boulders], 3-medium sediment, 4-fine sediment; see Appendix S2 for
196 habitat assignment details), visibility (good [can see beyond end of pole] or poor [could not
197 see beyond end of pole]) and camera frame position (vertical or horizontal).

198

199 **Data analysis**

200 Data on all mobile species were used in the following S analyses but only teleosts were used
201 in the N_{max} analyses. To remove the influence of extremely high abundance values, the teleost
202 N_{max} data were trimmed at the 95th percentile (eliminating 3 data points). Extreme abundance
203 values were related to large fish shoals that were rare and not informative for the purpose of
204 determining subtle and consistent change in species abundance.

205

206 *Species accumulation*

207 To examine the effect of BRUV deployment time on S and N_{max} , species and abundance
208 accumulation curves were created for the Wave Hub and Cable Route (project and reference
209 location data pooled). To investigate the potential effect of habitat, species and abundance
210 accumulation curves were created with all data pooled for habitat types; rocky reef and large
211 sediment (termed RRLS), and medium (gravel) and fine sediment (sand) (termed GS). These
212 curves were used to determine an optimum balance between the number of available
213 comparative video datasets and their recording duration (minutes) to be included in
214 subsequent analyses. All conducted in R version 3.4.4 (R Development Core Team, 2011)

215

216 *Statistical power, effect and sample size*

217 Power analyses were conducted using the SIMR package (Green and MacLeod, 2016) in R
218 version 3.4.4 (R Development Core Team, 2011) to investigate the relationships between the
219 number of samples (sample size), the size of change (effect size) and the probability to detect
220 change (power) in S or N_{max} .

221 Generalised linear mixed effect models were fitted to data (pooled data with years combined)
222 collected in spring at the WHSZ and CRSZ (separate models) using the lme4 package (Bates
223 et al., 2015). Only spring data were used in the models to remove seasonal effects (evidenced
224 in PERMANOVA models; Table 3a & 3b), which may increase variability in the data and

225 influence the power to detect change. (Step 1) The Poisson error models contained *year* and
226 *treatment* as fixed effects and *treatment* within *location* as a nested random effect. The effect
227 parameters from these models were used to build new simulation models with the same
228 structure (i.e. effects and error) but replicated a complete annual survey design for each study
229 zone (i.e. no loss of samples). (Step 2) Monte Carlo simulation was then utilised to generate
230 values for the response variable (S or N_{max}) of each model (1000 runs & seed = 1234). (Step
231 3) Power curves were generated for a range of effect sizes to explore the trade-off between
232 sample size and power. (Step 4). To examine inter-annual variation the steps detailed above
233 (1-3) were repeated for separate years (WHSZ = 4 years, CRSZ = 3 years) for effect sizes
234 where the pooled data model reached or exceeded 0.8 power. All analyses were conducted
235 using an (α) <0.05 threshold significance level, and results related to 0.8 power, the
236 commonly accepted level of confidence.

237

238 *Species richness, abundance and assemblage composition*

239 Permutational multivariate and univariate mixed effect models (PERMANOVA+) were used
240 in the software package PRIMERv6 (Anderson, 2001; Clarke and Warwick, 2001) to test
241 potential effects of the Wave Hub exclusion zone and the cable rock armouring (Cable Route)
242 on assemblage composition (a community structure measure incorporating both diversity and
243 abundance) , species richness (S) and abundance (N_{max}).

244

245 Between study zones

246 Models using the complete dataset were run to compare assemblage composition and S
247 between study zones (WHSZ & CRSZ). Factor *zone* was fixed, with *year*, *season*, *location*,
248 and *treatment* nested in *location* as random.

249

250 Within study zones

251 For each study zone (WHSZ & CRSZ), models for S and assemblage composition were
252 performed on the complete species dataset, and all three response variables for two defined
253 species guilds; teleost and crustacean. The factors *year* and *treatment* were fixed, and *season*,
254 *habitat*, *zone* (Cable Route only; 2 zones) and *treatment* nested in *location* were random. The
255 factor *year* had four and five levels for the CRSZ and the WHSZ respectively. The factor
256 *treatment* had two levels (WHSZ or CRSZ project, and reference). Depth and current speed
257 (tide indicator) were considered as environmental co-variates in initial models for each study
258 zone but were removed when found not to influence models.

259 Species N_{max} models were performed for four indicator taxa, family and species:
260 elasmobranchs, echinoderms, and *Pollachius spp.* (*Pollack & Saithe*) and *Cancer pagurus*
261 (*edible crab*) (Jackson et al., 2009). The same model structure was used as described above.

262

263 Between habitats

264 Models were also run using the complete dataset to test whether S , N_{max} and assemblage
265 composition differed with habitat type. Due to the low number of sampling events on fine
266 sediment habitats (n=9), making the model unbalanced, the final model contained only 3
267 levels (rocky reef [n=84], large [n=47] and medium [n=58] sediment) for the fixed factor
268 *habitat*, with *year* as random.

269 Prior to calculation of the Bray–Curtis (Bray and Curtis, 1957) similarity index, multivariate
270 data (assemblage composition) were dispersion weighted and square root transformed to
271 down weight taxa with erratic abundances and/or high abundances (Clarke et al., 2006a). As
272 joint species absences were important to consider between treatments, data were ‘zero-
273 adjusted by adding a dummy value of 1 (Clarke et al., 2006b). Without the dummy value,
274 Bray-Curtis would not consider samples similarly devoid of species as similar. Euclidean

275 distance indices were calculated for univariate data (S and N_{max}) that were $\text{Log}(x + 1)$
276 transformed (Anderson and Millar, 2004). Each term in the analyses used 9999 permutations
277 of the appropriate units. Significant interactions of fixed terms were tested using
278 PERMANOVA pairwise tests. Assemblage composition was visualised using nonmetric
279 Multi-Dimensional Scaling (nMDS).

280

281 **Results**

282 *Sampling and image quality*

283 The sampling regime was designed to gather data in both spring and autumn of each year
284 (2011-2015; full survey design = 720 deployments) to examine the potential effects of
285 season, but due to poor weather conditions in optimal tidal survey periods, the collection of
286 data in both seasons was only possible in 2013. Autumn sampling took place in 2011 and
287 2013; spring sampling took place in 2012, 2013, 2014 and 2015. Due to time and budgetary
288 constraints, no surveys could be completed in 2015 at the CRSZ, and the original survey
289 extent was reduced by two locations (most southerly project and reference locations in each
290 area) in 2013-2015 at the WHSZ and 2012-2014 for the CRSZ. The reduced survey effort
291 owing to weather, time and money resulted in a potential maximum of 312 BRUV
292 deployments, a considerable reduction (57%) from the original sampling design.

293 Further adverse weather during survey activity caused 15 (5%) deployments to be cancelled.

294 Across study areas and years, 297 BRUV deployments were successfully undertaken, of
295 which 38 (12.8%) failed due to technical reasons (e.g. battery or camera failure). Of the
296 remaining 259 (83% of total); 141 video datasets were 60 minutes or longer (45% of total),
297 199 video datasets were 45 minutes or longer (63% of total), and 247 were 30 minutes or
298 longer (79% of total). Once filtered for visibility (good visibility with an unobstructed view
299 of the seabed) the number of available videos for analysis were 116 with 60 minutes or more

300 (37% of total), 161 datasets with 45 minutes or more (51% of total), and 198 datasets with 30
301 minutes or more (63% of total) (Fig. S2).

302 The BRUVs effective range of attraction (AR) for a teleost of ~200-300 mm in length was
303 ~127 m for 30 minutes soak time, ~287 m for 45 minutes and ~510 m for 60 minutes for our
304 study (Fig. S3). This would indicate the replicate BRUV deployments in our study (mean
305 ~130 m apart) may not be independent (i.e. the same individual could attend and be recorded
306 on more than one camera) if more than 30 minutes video data were used. The AR calculation
307 does not consider current or plume direction, which could influence the range of attraction
308 shape (e.g. not cylindrical around the location, but elongated (ellipsoid/triangular) in the
309 direction of current). This in turn could affect the ability of animals to detect bait from a
310 second location (reduce the AR estimate in that direction) while visiting the first, if the bait
311 plume is directed away from the first. The BRUV replicates in our study were placed in a
312 linear manner across the prevalent NE↔SW tide, mainly N↔S and NW↔SE (Fig. 1). The
313 current direction and speed data for each BRUV deployment (Fig. S4) indicates the prevalent
314 seabed currents would advect bait plume away from other replicate locations rather than
315 towards them, suggesting it would be more difficult for an animal to detect bait from a
316 second location and subsequently move to it. When 30 minutes video data were analysed for
317 spatial auto-correlation (Moran's I) using teleost and elasmobranch abundance for each study
318 zone, year and season, the only significant auto-correlation was found for teleost in 2012 (2
319 out of 17 tests; Table S1). When the data for 2012 were investigated further, the effect was
320 likely caused by large aggregations of *Trisopterus minutus* counted on all three replicate
321 BRUVs at locations in each study zone in this year. This is a small (~100-200 mm) benthic
322 species, so their ability to attend two or three BRUVs (linearly ~130 or 260 m apart) would
323 be unlikely given the AR and current data, and the clustering (auto-correlation) would seem
324 more likely a representation of high abundance at all these locations during that time. Given

325 these analyses it was considered that species and abundance data gathered from 30 minutes
326 video data would be independent for teleost and the (relatively) small elasmobranch species
327 in our study.

328

329 *Species accumulation*

330 The maximum number of species identified during video datasets lasting 60 minutes was 12.

331 The mean number of species occurring on RRLS habitats was 6.8, while for GS it was 5.3

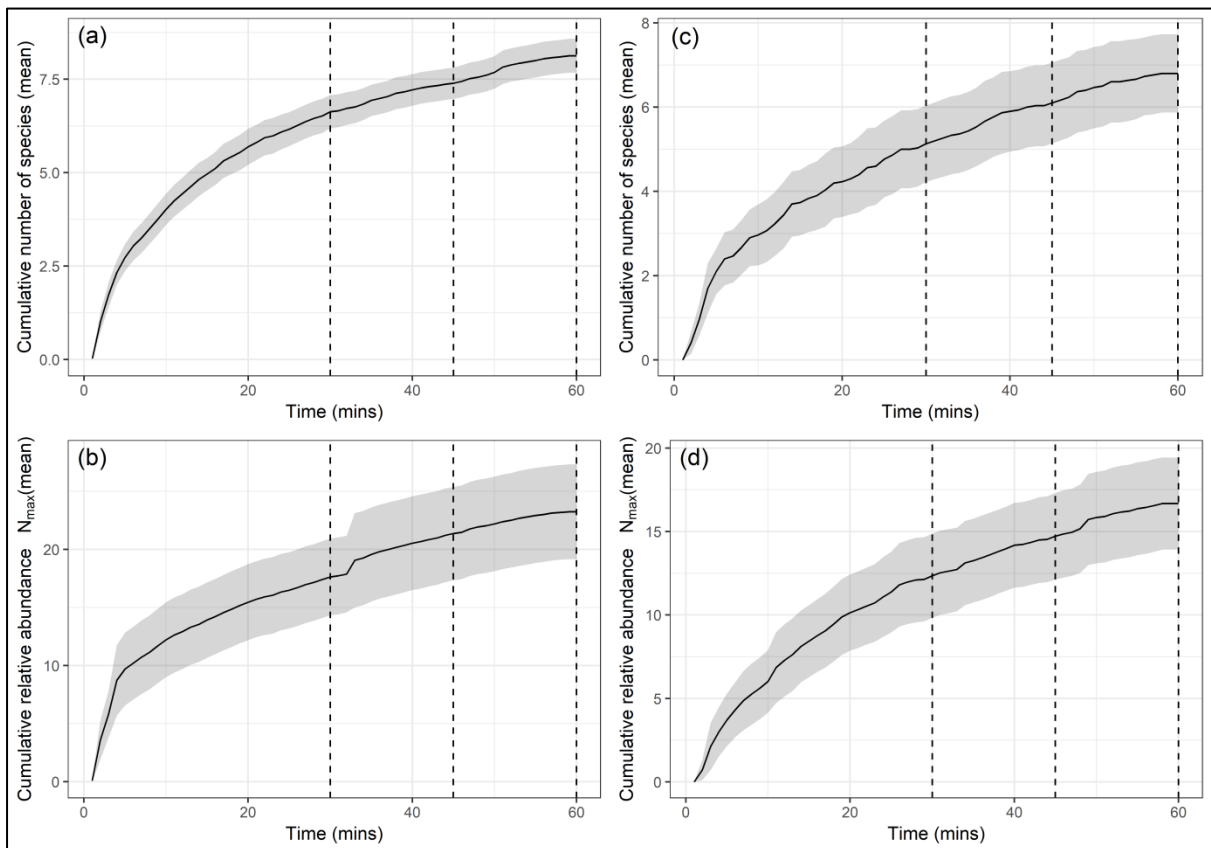
332 (Fig. 3a & c). Mean N_{max} was greater for RRLS habitats (maximum of 132 individuals)

333 compared to GS habitats (maximum 38 individuals) (Fig. 3b & d). In both habitats, 75% or

334 more of the total S and N_{max} (at 60 minutes) was achieved after 30 minutes, with the greatest

335 increase in number of species and abundance (rate of change) occurring in the first 20

336 minutes of video datasets (Fig S1).



337

338 **Fig. 3.** Mean species (a & c) and relative abundance (N_{max}) (b & d) accumulation curves from 60
339 minute BRUV footage for rocky reef/large sediment habitat (a & b) and medium/fine sediment
340 habitats (c & d). Grey shading = 95% confidence interval. Dashed lines = 30, 45 and 60 minutes.

341

342 *Optimal data*

343 Data from the first 30 minutes of BRUV footage were considered optimal for use in further
344 analyses as this soak period balanced sample size, coverage (study locations and years),
345 effective range of attraction, with available species and abundance information (accumulation
346 curves).

347

348 *Species diversity and taxonomic composition*

349 A total of 67 species from 46 families and 6 phyla were observed on the BRUV footage
350 across the two study zones for the survey period. This equated to 5,440 individual animals,
351 the vast majority of which were fishes (teleost ~73%). The WHSZ was more speciose (49
352 species) compared to the CRSZ (42 species), but total abundance was higher in the latter with
353 >3400 individuals (Table 1).

354

355 *Statistical power, effect and sample size*

356 Models with pooled spring data (years combined) revealed the minimum change that could
357 be detected with 0.8 (or more) power was ~10% in richness for both WHSZ and CRSZ,
358 which would require ≥ 200 samples (~100 within each project and reference). The samples
359 required to detect 20% change in S reduce to ~40-50 (~20-25 within each project and
360 reference) for both WHSZ and CRSZ (Fig 4a & 4c sub plots). For N_{max} , the minimum change
361 detectable was ~50% at the WHSZ (Fig 4b sub plot), which required ≥ 100 samples (~50
362 within each project and reference). The 0.8 power threshold was not reached for any level of

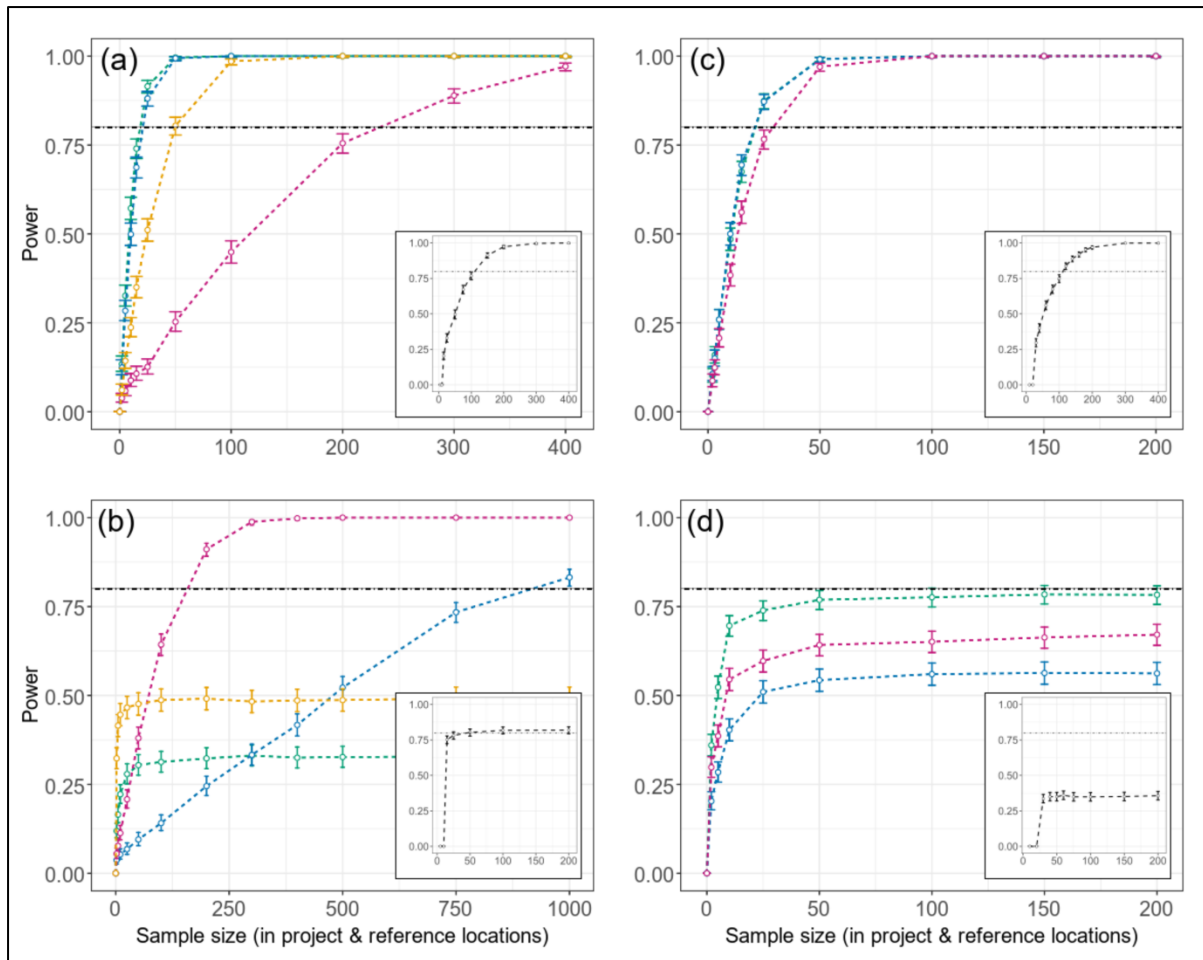
363 change in N_{max} using pooled spring data from the CRSZ, and the power to detect 50% change
364 did not exceed 0.4, no matter the sample size (Fig 4d sub plot).

365 **Table 1** Number of species and individuals by taxonomic phylum on BRUV footage at the Wave Hub and Cable Route study zones. The
 366 percentage of total species and individuals is shown in italics.

Phylum	Wave Hub Study Zone				Cable Route Study Zone			
	No. of species	<i>% of total species</i>	No. of individuals	<i>% of total individuals</i>	No. of species	<i>% of total species</i>	No. of individuals	<i>% of total individuals</i>
Arthropod	1	<i>2.00</i>	2	<i>0.10</i>	0	<i>0.00</i>	0	<i>0.00</i>
Crustacean	13	<i>26.53</i>	331	<i>16.62</i>	8	<i>19.05</i>	165	<i>4.79</i>
Echinoderm	6	<i>12.25</i>	261	<i>13.10</i>	6	<i>14.29</i>	390	<i>11.31</i>
Elasmobranch	3	<i>6.12</i>	162	<i>8.13</i>	2	<i>4.76</i>	152	<i>4.41</i>
Mollusc	5	<i>10.20</i>	18	<i>0.90</i>	0	<i>0.00</i>	0	<i>0.00</i>
Teleost	21	<i>42.86</i>	1218	<i>61.14</i>	26	<i>61.90</i>	2741	<i>79.50</i>
Total	49		1992		42		3448	

367

368



369

370 **Fig. 4.** Minimum effect size SIMR model outputs ($\pm 95\%$ CI; <0.05 significance) for pooled spring
 371 species richness (S) and relative teleost abundance (N_{max}) data (inserted plots) separated by year. (a) =
 372 10% change in S at WHSZ, (b) = 10% change in S at CRSZ, (c) = 50% change in N_{max} at WHSZ, and
 373 (d) = 50% change in N_{max} at CRSZ. Green dashed line = 2012, blue dashed line = 2013, purple dashed
 374 line = 2014, and orange dashed line = 2015. Dot-dash line on all plots indicates 0.8 power.

375

376 There was considerable variation among years when data were analysed separately for 20%
 377 (S) and 50% (N_{max}) effect size models (detailed above; Fig. 4). The sample size required to
 378 detect a 20% change in S at WHSZ with 0.8 power ranged from ~ 80 to >1000 samples (Fig.
 379 4a) and ~ 40 to 60 samples for the CRSZ (Fig. 4b). Teleost N_{max} model outputs revealed the
 380 majority of years (2 of 4 at WHSZ; 2 of 3 at CRSZ) had low power (<0.7) to detect 50%
 381 change, independent of sample size. At the WHSZ, 2014 data were modelled to achieve >0.8

382 power with ~150 samples (~ 75 within project and reference), and 2013 data would achieve
383 this level with ~1700 samples (~ 850 within project and reference) (Fig. 4c). The only data to
384 achieve 0.8 power to detect 50% change in N_{max} at CRSZ was for 2012, which would require
385 ~200 samples (100 within project and reference) (Fig. 4d).

386

387 *Assemblage, species richness and abundance analyses*

388

389 Comparison between study zones

390 Assemblage composition and species richness (S) models revealed significant differences
391 between study zones ($P(\text{perm}) = <0.001$; Table 2). The species contributing most (top 4
392 species in SIMPER table) to the dissimilarity in composition were Poor cod *Trisopterus*
393 *minutus*, common hermit crab *Pagurus bernhardus*, spiny starfish *Marthasterias glacialis*
394 and cuckoo wrasse *Labrus mixtus*, with higher abundance of all except hermit crab in the
395 CRSZ (Table 2). Higher average species richness was also observed within the CRSZ (CRSZ
396 = mean 6.8 ± 2.2 SD, $n=107$; WHSZ = mean 5.3 ± 2.1 SD, $n=91$). Yearly variation in overall
397 species richness and associated variance was observed in both study zones (Fig. 5), but was
398 particularly evident in teleost abundance data (Fig. 6).

399

400

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402

403

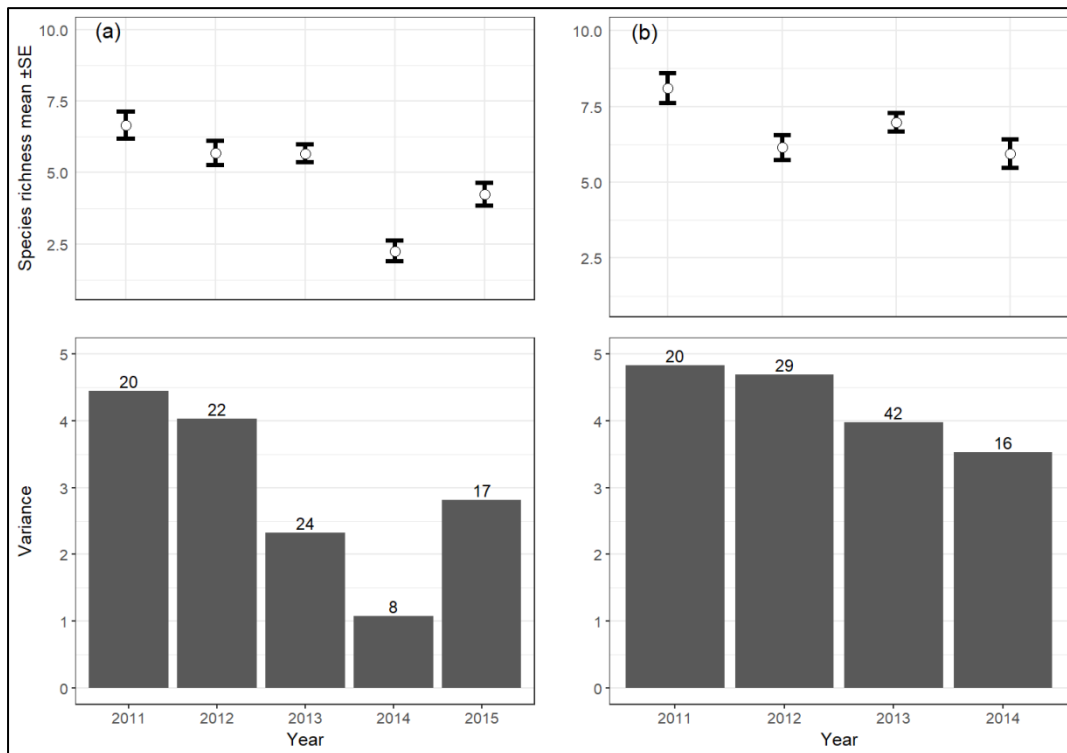
404

405

406 **Table 2.** PERMANOVA results for species assemblage and richness (S) models by study zones.
 407 Fixed effects are non-italicized and random effects italicized. Significant permutation p-values below
 408 the 0.01 level for fixed effects are shown in bold.

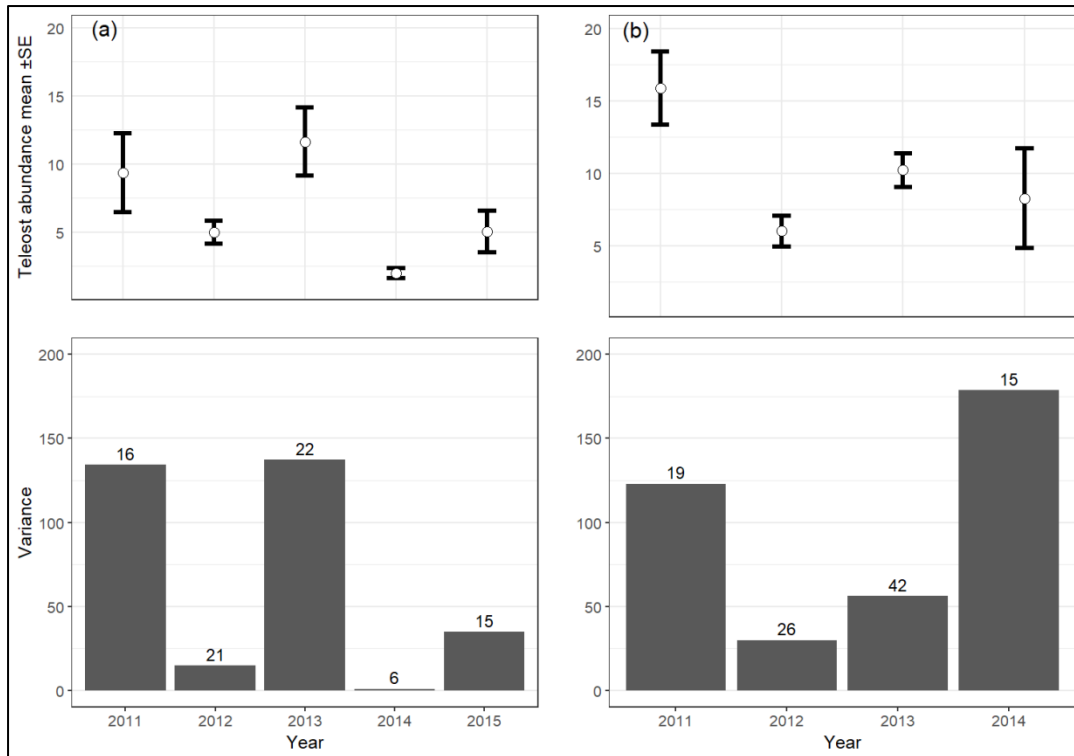
Wave Hub vs Cable Route Zones							
Source	d.f.	<i>Assemblage</i>			S		
		MS	Pseudo-F	P(perm)	MS	Pseudo-F	P(perm)
Zone	1	67326	13.159	0.0001	2.6850	8.0528	0.0029
<i>Treatment</i>	4	10121	3.2071	0.0017	1.0221	6.6190	0.0095
<i>Season</i>	1	4128.7	2.6529	0.0060	0.1888	2.0100	0.1602
<i>Location</i>	1	1144.5	0.3634	0.9417	0.2575	1.1326	0.3050
<i>Location(Treatment)</i>	21	3318.0	2.3059	0.0001	0.1961	2.2496	0.0019
Pooled terms	169	1439.0			0.0871		
Pair-wise test for Zone:				Pair-wise test for Zone:			
Group		t		P(perm)	Group	t	P(perm)
WH vs CR		3.6276		0.0001	WH vs CR	2.8378	0.0040
SIMPER output for Zone (>5% contribution):							
Species/family		WH av. abundance		CR av. abundance		% cont.	
<i>Trisopterus minutus</i>		1.25		1.44		11.4	
<i>Pagurus bernhardus</i>		1.18		0.16		8.47	
<i>Marthasterias glacialis</i>		0.28		1.18		8.23	
<i>Labrus mixtus</i>		0.25		1.24		8.07	
<i>Ophiurodea</i>		0.91		0.33		7.48	
<i>Ctenolabrus rupestris</i>		0.02		0.85		6.25	
<i>Scyliorhinus canicula</i>		1.08		0.91		5.87	
<i>Gobiidae</i>		0.52		0.43		5.05	

409
 410 **Fig. 5.** Species richness (mean \pm standard error) and variance for each year at the (a) Wave Hub and
 411 (b) Cable Route study zones. Number above bar = sample size.



412
 21

413 **Fig. 6.** Teleost relative abundance (mean \pm standard error) and variance for each year at the (a) Wave
 414 Hub and (b) Cable Route study zones. Number above bar = sample size.



415

416

417 Within Wave Hub study zone

418 No significant fixed effect or interaction term was found in the multivariate assemblage
 419 composition models for all species (all $P(\text{perm})$ values >0.05 ; Table 3a; Fig. S5), teleosts (all
 420 $P(\text{perm})$ values >0.05 ; Table S2a; Fig. S6) or crustaceans (all $P(\text{perm})$ values >0.05 ; Table
 421 S3a; Fig. S6) for the WHSZ.

422 Species richness (S) had a significant interaction $Year \times Treatment$ term ($P(\text{perm}) = 0.0136$;
 423 Table 3a), as did teleosts ($P(\text{perm}) = 0.0367$; Table S2a) and crustacean models ($P(\text{perm}) =$
 424 0.001 ; Table S3a). Pairwise tests revealed that these were mainly driven by significant
 425 differences between years within the Wave Hub project and reference locations (Tables 3a,
 426 S2a & S3a), with only a significant effect of $Treatment$ for all species in 2012 ($P = <0.05$;
 427 Table 3a) and for crustaceans in 2014 ($P = <0.001$; Table S3a). Relative abundance (N_{max})
 428 models had no significant fixed effect or interaction term for teleosts, crustaceans,

429 echinoderms, elasmobranchs or *Pollachius spp.* (all $P(\text{perm})$ values = >0.05 ; Tables 3a, S2a,
430 S3a, S5a & S5a), but there was a significant interaction term ($\text{Year} \times \text{Treatment}$) for *Cancer*
431 *pagurus* ($P(\text{perm}) = <0.05$; Table S5a). Pairwise tests showed significant differences
432 between years within Wave Hub project (2011-2013 & 2012-2013) or reference locations
433 (2011-2012), and for locations within one year (2011), however some pairwise tests could not
434 be conducted indicating a limitation of the data.

435

436 Within Cable Route study zone

437 Multivariate assemblage composition analysis revealed a significant interaction term, $\text{Year} \times$
438 Treatment , for all species ($P(\text{perm}) = 0.0421$; Table 3b; Fig. S5) and teleost models ($P(\text{perm})$
439 = 0.0118 ; Table S2b; Fig. S6). Pairwise tests revealed both had significant differences
440 between years within Cable Route project and reference locations, but not between Cable
441 Route project and reference locations within years (Table 3b & Table S2b). The crustacean
442 model had a significant Year term ($P(\text{perm}) = 0.0137$; Table S3b; Fig. S6), and 3 out of the 6
443 subsequent pairwise tests were significant (2011-13, 2011-14 & 2013-14; Table S3b).
444 Species richness (S) models had no significant fixed terms for all species or teleosts (Table 3b
445 & Table S2b), but the crustacean model revealed Year as significant ($P = <0.001$; Table S3b).
446 Four out of the 6 subsequent pairwise tests were significant (2011-13, 2012-13, 2012-14 &
447 2013-14; Table S3b). Relative abundance (N_{max}) models had no significant fixed terms for
448 teleosts, echinoderms or elasmobranchs (all $P(\text{perm})$ values >0.05 ; Table S2b and S4b). Year
449 was found to be significant in the crustacean ($P(\text{perm}) = 0.0002$; Table S3a) and *Cancer*
450 *pagurus* ($P(\text{perm}) = 0.0002$; Table S5b) models, with 4 from 6 pairwise test significant for
451 crustaceans (2011-14, 2012-13, 2012-14, 2013-14; Table S3a) and 3 from 6 pairwise test
452 significant for *Cancer pagurus* (2011-13, 2012-13, 2013-14; Table S5b). The only model to
453 have a significant fixed Treatment term was for relative abundance of *Pollachius spp.*

454 ($P(\text{perm}) = 0.01$; Table S5b), with significantly greater abundance in Cable Route project
455 than reference locations (Cable Route: mean 1.4 ± 0.8 SD, $n=23$; reference = mean 1 ± 0 SD,
456 $n=7$).

457

458 Habitats

459 Assemblage composition and species richness (S) models revealed significant differences
460 between habitat types (Types 1-3: $P(\text{perm}) = <0.001$; Table 4 ; Fig. S7.). No significant
461 difference in N_{max} was found between habitats.

462 **Table 3.** PERMANOVA results for all species assemblage, richness (S) and teleost species relative abundance (N_{max}) models for the Wave hub (a) and Cable
 463 Route (b) study zones. Fixed effects are non-italicized and random effects italicized. Significant permutation p-values below the 0.05 level for fixed effects
 464 and interactions are shown in bold.

(a)		Wave Hub study zone								
		<i>Assemblage (all species)</i>			<i>S (all species)</i>			<i>N_{max} (teleost species)</i>		
Source	d.f.	MS	Pseudo- <i>F</i>	<i>P</i> (perm)	MS	Pseudo- <i>F</i>	<i>P</i> (perm)	MS	Pseudo- <i>F</i>	<i>P</i> (perm)
Year	4	9525.1	2.6593	0.0838	1.26910	18.190	0.1155	4.6522	16.404	0.1225
Treatment	1	3424.5	0.7086	0.6604	0.00532	0.1248	0.7077	2.2460	1.5607	0.2674
<i>Season</i>	<i>1</i>	<i>3532.5</i>	<i>2.2758</i>	0.0207	<i>0.06245</i>	<i>0.7359</i>	<i>0.4073</i>	<i>0.2483</i>	<i>0.2947</i>	<i>0.5889</i>
<i>Habitat</i>	<i>3</i>	<i>5229.8</i>	<i>2.9543</i>	0.0001	<i>0.54909</i>	<i>6.9326</i>	0.0007	<i>1.8946</i>	<i>2.2091</i>	<i>0.0929</i>
<i>Location(Treatment)</i>	<i>6</i>	<i>4125.6</i>	<i>3.0025</i>	0.0001	<i>0.14502</i>	<i>2.1285</i>	<i>0.0588</i>	<i>1.2010</i>	<i>1.5015</i>	<i>0.1880</i>
Year x Treatment	4	1930.3	1.4048	0.0759	0.23181	3.4023	0.0136	1.2459	1.5576	0.1913
Pooled terms	71	1374.1			0.06813			0.7999		
Pair-wise test for Year x Treatment:										
<u>Project</u>										
Year						t	<i>P</i> (perm)			
						2011-2012	2.7615	0.0155		
						2011-2013	1.6244	0.1254		
						2011-2014	3.9172	0.0375		
						2011-2015	2.4403	0.0292		
						2012-2013	10.346	0.0380		
						2012-2014	2.7358	0.0278		
						2012-2015	0.6682	0.5179		
						2013-2014	7.1531	0.0003		
						2013-2015	15.652	0.0481		
						2014-2015	1.8411	0.0999		
<u>Reference</u>										
						2011-2012	0.7015	0.5037		
						2011-2013	0.9376	0.3630		
						2011-2014	5.2210	0.0003		
						2011-2015	2.5880	0.0211		
						2012-2013	1.2550	0.4944		
						2012-2014	5.4939	0.0002		
						2012-2015	3.5548	0.0019		
						2013-2014	3.7698	0.0489		
						2013-2015	1.2859	0.4656		
						2014-2015	3.7541	0.0046		
Pair-wise test for Year x Treatment :										
Year						t	<i>P</i> (perm)			
						2011	1.4630	0.2122		

2012	2.9688	0.0181
2013	1.5870	0.2158
2014	0.6848	0.5935
2015	0.5256	0.6315

(b)

Cable Route zone										
Source	d.f.	Assemblage (all species)			S (all species)			N _{max} (teleost species)		
		MS	Pseudo-F	P(perm)	MS	Pseudo-F	P(perm)	MS	Pseudo-F	P(perm)
Year	3	6536.5	2.5807	0.0079	0.33179	1.0787	0.4573	3.7770	1.1392	0.4471
Treatment	1	2513.6	1.1766	0.3133	0.38619	1.7755	0.2085	2.6261	2.5411	0.1362
Season	1	3245.0	2.0852	0.0369	0.46417	6.6699	0.0126	5.3069	7.8565	0.0060
Zone	1	3532.7	1.8065	0.0743	0.04206	0.2403	0.6855	0.4412	0.5121	0.5018
Habitat	3	3532.2	2.6270	0.0002	0.07837	0.9400	0.4225	1.4894	2.3797	0.0794
Location(Treatment)	13	1823.3	1.4602	0.0038	0.18391	2.9089	0.0027	0.8614	1.4885	0.1439
Year x Treatment	3	1953.6	1.5646	0.0421	0.10769	1.7034	0.1742	1.1476	1.9832	0.1243
Pooled terms	81	1248.7			0.06322			0.5787		
Pair-wise test for Year x Treatment:										
<u>Project</u>										
Year			t	P(perm)						
2011-2012			2.1595	0.0001						
2011-2013			1.4024	0.0473						
2011-2014			1.8228	0.0046						
2012-2013			2.9543	0.0007						
2012-2014			1.2262	0.2050						
2013-2014			1.8894	0.0062						
<u>Reference</u>										
2011-2012			1.2666	0.1783						
2011-2013			1.2847	0.1559						
2011-2014			2.1578	0.0007						
2012-2013			0.8882	0.6195						
2012-2014			1.1486	0.2702						
2013-2014			1.4463	0.1088						
Pair-wise test for Year x Treatment :										
Year			t	P(perm)						
2011			0.9143	0.5935						
2012			0.8401	0.7480						
2013			1.2993	0.1492						
2014			0.8659	0.5963						

465

466

467 **Table 4.** PERMANOVA results for species assemblage, richness (S) and relative abundance (N_{max}) models by habitat types; 1-3. Fixed effects are non-
 468 italicized and random effects italicized. Significant permutation p-values below the 0.01 level for fixed effects are shown in bold.

Habitats (1 – 3)										
Source	d.f.	<i>Assemblage</i>			S			N_{max}		
		MS	Pseudo-F	P(perm)	MS	Pseudo-F	P(perm)	MS	Pseudo-F	P(perm)
Habitat	2	30502	15.549	0.0001	1.2972	11.763	0.0001	1.1099	1.8523	0.1561
<i>Year</i>	4	<i>8813.1</i>	<i>5.2586</i>	<i>0.0001</i>	<i>0.6820</i>	<i>7.8907</i>	<i>0.0001</i>	<i>4.2311</i>	<i>9.4497</i>	<i>0.0001</i>
Pooled terms	182	1675.9			0.0864			0.4477		
Pair-wise test for Habitat				Pair-wise test for Habitat						
Habitats		t	P(perm)	Habitats		t	P(perm)			
1-3		5.2528	0.0001	1-3		4.7223	0.0001			
1-2		2.5931	0.0001	1-2		2.5438	0.0123			
3-2		3.0235	0.0001	3-2		1.9184	0.0582			

469 **Discussion**

470 Weather and sea state are major considerations when working in mid latitude offshore marine
471 environments, and can be a serious limitation to survey time and sampling effort when using
472 small vessels. In this study, poor weather conditions and logistical constraints reduced the bi-
473 annual survey to one season a year, with the exception of 2013. Access to larger vessels may
474 have improved this situation but would have appreciably increased costs and reduced cost
475 effectiveness of the BRUV technique. The study period encompassed some noteworthy bad
476 weather events across the British Isles, and the extreme 2014 storms (Masselink et al., 2016)
477 resulted in a reduction in samples due to bad visibility rendering footage unusable (Fig. S2).
478 With storm events predicted to increase with climate change (Coumou and Rahmstorf, 2012;
479 Zappa et al., 2013) the potential effects on sampling success need to be considered within
480 survey design and sampling effort plans, i.e. over-estimate required sampling for planned
481 redundancy. The coastal areas optimal for wind or wave energy generation will be, by their
482 nature, open to weather systems that will influence access to locations (sampling
483 opportunities) and the quality of data gathered (visibility). The sampling loss due to camera
484 and battery failures was relatively high and likely related to the repeated demands placed on
485 the equipment and experience of maintaining and deploying the equipment on/off small boats
486 in challenging conditions with small research teams (2 people). These losses were minimised
487 with experience and improved equipment, but could be further mitigated if funds and time are
488 available for repeat sampling efforts and/or campaigns.

489 The loss of survey periods or samples due to weather, technical problems or logistical issues
490 reduces the precision in the characterisation of spatial and temporal variability at a site, and
491 will ultimately affect the ability to detect impact (Underwood and Chapman, 2003).
492 Deploying BRUV systems for an optimal time can help reduce avoidable loss by limiting
493 time in the water. Accumulation curves derived from data gathered at different locations

494 around the world, for fish and invertebrate species, suggests BRUV deployment time (or
495 footage time) ranging from 30 minutes to 2 hours in order to capture a significant proportion
496 of the number of species or individuals attracted to the bait (Bernard and Gotz, 2012;
497 Unsworth et al., 2014). BRUVs were deployed for 60 minutes in our study, but only the first
498 30 minutes of the footage were used in the analyses, capturing on average 75% or more of the
499 total richness or relative abundance observed over the whole 60 minute footage (Fig. 3). Our
500 decision is worth briefly elaborating as it considered ecological and statistical elements of
501 BRUV sampling and, therefore, could be informative for future studies using this technique.
502 Firstly, in our study a BRUV system was used to provide a rapid assessment of species and
503 relative abundance for particular, small benthic areas. If longer time periods were used, the
504 effective range of attraction (AR) would increase so mobile species (teleost and
505 elasmobranch) could be attracted from much further afield (>500 m; not our objective), and
506 as a consequence, the animals could potentially attend multiple cameras, leading to double
507 counting of individuals (pseudo-replication in our design). Secondly, due to technical
508 (camera/battery failure) and condition (visibility or field of view) related issues, the number
509 of useable videos significantly reduced (~40%) when applying 60 minute (116 videos)
510 compared to a 30 minute footage requirement (198 videos) (Fig. S1). The decrease in sample
511 size reduces the power of these data to detect statistically significant change and, for 2011,
512 would have translated into no data being available and removal of this year from analyses.
513 The statistical benefit conferred by an increased sample size and inclusion of all surveyed
514 years (inter-annual variance), combined with the ecological reasoning behind using a shorter
515 time period, resulted in the decision to use 30 minute duration video data. However, a
516 limitation of only deploying, or using data, for a short period (30 minutes in our case), is the
517 chance of missing the presence of rare and/or cautious species. Given renewable energy
518 development sites (wave, wind or tidal) could become *de facto* Marine Protection Areas

519 (MPAs) by excluding damaging seabed fishing activities, the areas could become refuges for
520 species with small or recovering populations. Deploying a subset of BRUVs for longer could
521 help detect such species and be incorporated into study designs, specifically for this purpose.
522 In this study, no new species were detected when 60 minutes footage was compared to 30
523 minutes, but longer periods may be necessary to detect rare species. The specifics detailed
524 here are particular to our study but highlight, more generally, the need to understand the
525 survey technique being used and the data it is expected to capture. Moreover, how the
526 precautionary approach to sampling effort previously recommended (over-estimation) is
527 necessary to allow redundancy in sampling campaigns when working in highly dynamic
528 environments.

529 The analyses conducted using the BRUV data provided little consistent evidence of
530 differences in metrics (across taxa) between the locations influenced by either the trawling
531 exclusion (Wave Hub) or cable infrastructure (Cable Route), and reference locations within
532 study zones. The only consistent change across years was an increase in the relative
533 abundance of pollack and saithe (*Pollachius spp.*) around the cable infrastructure. These are
534 commercially important coastal species understood to associate with rocky reef and hard
535 substrate habitat as nursery areas (Seitz et al., 2014), suggesting the addition of rock and
536 concrete matressing on the cable may be providing suitable conditions. Greater abundance of
537 cuckoo wrasse *Labrus mixtus* observed during towed camera surveys in the same area has
538 also been attributed to the presence of the hard substrate for cable protection (Sheehan et al.,
539 2013). Both observations are consistent with other studies of fish abundance around MREI
540 structures (Wilhelmsson et al., 2006) and add to the evidence these introduced structures and
541 associated infrastructure are created habitat for species to utilise (Inger et al., 2009; Miller et
542 al., 2013; Sheehan et al., 2018; Witt et al., 2012). The lack of evidence towards an impact of
543 trawling exclusion in the Wave Hub zone may relate to low fishing effort in the area before

544 (Campbell et al., 2013) and during the study, creating equivalent disturbance (or lack of) in
545 project and reference locations. Alternatively, it could be an artefact of the survey design and
546 power to detect change. The power analyses revealed how the ability to detect change in
547 species richness and teleost relative abundance differed considerably between zones (spatial)
548 and among survey years (temporal), producing a range of sampling schedules to detect the
549 same effect size and varying levels of power (Fig. 4). Low power was particularly apparent in
550 teleost abundance indicating these data could only confidently detect large changes (>50%)
551 with high probability ($\alpha=0.05$), much less for the CRSZ. This could explain the lack of
552 significant results found for this metric in our univariate analyses (Table 2 & Table S2), with
553 potential influence on the outcome of the multivariate assemblage composition analysis (Fig.
554 S5). Low probability to detect directional change in abundance data due to high variability
555 has been found for other survey techniques and species (e.g. fish, cetaceans & seabirds; Al-
556 Chokhachy et al., 2009; Forney, 2000; Maclean et al., 2013). Our results re-iterate caution in
557 accepting that no change is taking place when analyses fail to reveal statistically significant
558 patterns, when a lack of statistical power may be the contributing factor (Al-Chokhachy et al.,
559 2009; Maclean et al., 2013). Despite good statistical power (~0.8) to detect relatively small
560 changes (20%) in species richness, the majority of statistically significant effects found in our
561 analyses (uni- and multivariate) were associated with yearly differences (Tables 2 and Tables
562 S2-5), and only a single year difference found for the exclusion zone (*'Treatment'*) in
563 richness of all species combined (2012; Table 2a) and crustaceans (2014; Table S3a) at the
564 WHSZ. Natural yearly variation was the main effect identified at the study zones suggesting
565 high levels of 'noise' in the system, presenting potential difficulty in distinguishing impacts
566 related to the Wave Hub exclusion zone or cable route infrastructure. Moreover, it
567 demonstrates how multi-year data are essential to capture site variability and provide accurate

568 baseline characterisation, from which single site or cumulative impact from renewable energy
569 convertors or manufactured infrastructure could be robustly assessed (Maclean et al., 2014).

570 The high natural inter-annual variation found for each metric (at both study zones; Figs. 4-6)
571 presents evidence for the difficulty in using pilot (e.g. single survey or one year) site data to
572 inform the design and effort (e.g. power analysis) required for robust impact assessment
573 surveys. If the power analysis was to rely on data from only one year of our study there
574 would be a risk of either over- or under-estimating the number of samples needed to obtain
575 acceptable power to detect a chosen effect size (Fig. 4). The consequence could either be a
576 lack of power in the subsequent impact analyses (under-estimate) or collecting samples that
577 are potentially unnecessary (over-estimate). The latter has the benefits of providing
578 redundancy in the sampling effort, as has been previously recommended, but requires
579 additional time, effort and cost, which would need to be factored into the assessment
580 program.

581 The difference in assemblage composition, species richness and variance found between
582 study zones (Table 2; Fig. 4 & 5) might be expected given the dominant habitat types found
583 at the Wave Hub (medium to fine sediment) and Cable Route (mix of rocky reef and large
584 boulders) study zones (e.g. Fig. S7), and the significant difference in composition and
585 richness found among them (Table 4). The site specific nature of ecosystem components,
586 even over a relatively small spatial area (~130 km²), is effectively demonstrated using these
587 data. The characterisations these data offer also illustrate how useful consistent data
588 collection methods (and survey design) can be in allowing comparison between locations.
589 Baseline characterisation and impact studies of benthic habitats and species are required at
590 MREIs located in coastal locations (inshore and offshore) with various physical (e.g. size,
591 bathymetry) and ecological characteristics. While the energy convertor design may differ
592 between locations the ability to understand more general or cumulative effects caused by

593 developments would be enhanced by multi-site data, collected using the same survey
594 techniques. Adopting standard practices and guidelines for pre- and post-development
595 benthic survey methods, design and analysis would help optimise costs associated with EIAs
596 and, if adopted across multi-sites and multi-years, could ultimately lead to impact models
597 with predictive power for species, communities or ecosystems (e.g. Butenschön et al., 2016).

598 Given there were no prior ecological data for the study zones monitored during this study, the
599 survey design performed well in providing adequate samples to detect change in species
600 richness but not so well for relative abundance. Caution is required when interpreting the
601 abundance results (detailed above), but overall it indicates the survey would be able to detect
602 differences between project and reference locations that could be considered large (e.g.
603 >50%). Whether changes of this size are biologically or ecologically significant is unknown
604 and would be highly dependent on the resilience of the ecosystem. For example, if a
605 reduction in species richness included the loss of a functional group (e.g. predators or
606 herbivores) within the local ecosystem, then it is likely this could lead to a significant
607 ecological impact (Micheli and Halpern, 2005).

608 As part of this study the BRUV system demonstrated its value as a tool for collecting
609 assemblage composition, species richness and relative abundance data for epi-benthic mobile
610 species in highly dynamic conditions, and a good candidate for use as part of marine EIAs
611 across latitudes. The system offers a cost effective and flexible method that can provide the
612 spatial and temporal coverage that is difficult to obtain using other methods (i.e. divers,
613 remote underwater vehicles). When used with stereo cameras, BRUV can also offer size data
614 that could help elucidate more detailed age related effects cause by introduced or altered
615 habitat (Elliott et al., 2017), or converted into biomass estimates (standing or relative)
616 providing another metric to assess impact. Although other traditional methods of sampling
617 these communities, e.g. trawling, potting, bottom lines or nets, can provide these metrics and

618 work in similar or worse sea conditions, they can be destructive or taxa specific, increasing
619 the cost of survey effort and to the ecosystem of study. With further development, BRUV
620 systems also have the potential to help address data collection gaps surveys often suffer from,
621 e.g. diurnal variation. With the integration of movement sensors (i.e. infra-red) and/or
622 artificial intelligent (AI) algorithms to activate equipment on species presence, 24 hour
623 deployments could be possible. BRUV biodiversity data has also been found to be
624 complimentary to environmental DNA (Stat et al., 2018), suggesting a combined approach
625 using these non-invasive methods could further enhance the effectiveness of monitoring
626 surveys.

627

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636

637 **References**

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