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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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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   bentho-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.

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Introduction

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46   The marine coastal environment provides a major, and disproportionate, contribution to

47   global ecosystem services (Costanza et al., 1997; Drakou et al., 2017; Liquete 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

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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.

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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

6

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
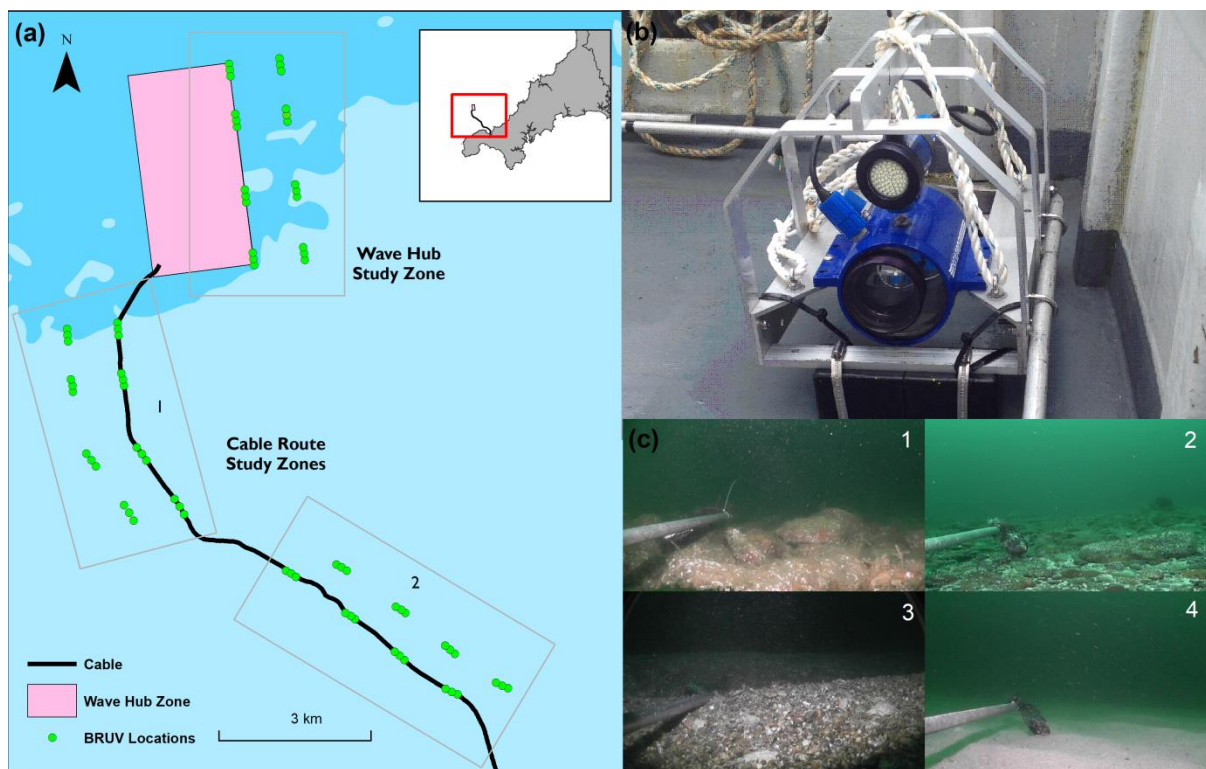
149   study.

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

8

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

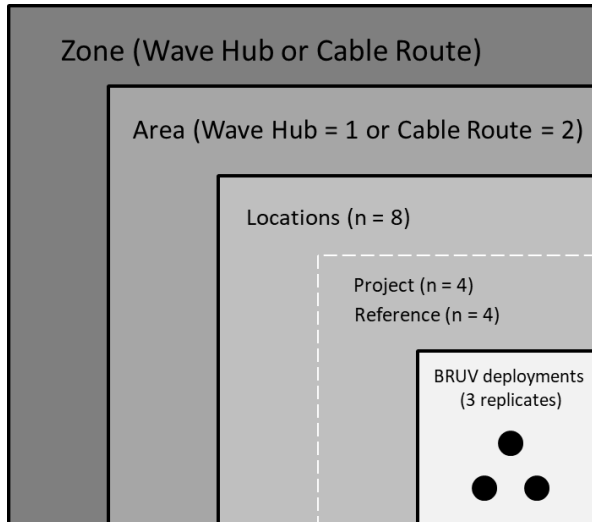183   used as the AR parameters (V_c = current speed, V_f = maximum fish speed).

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

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

Species richness, abundance and assemblage composition

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

Between study zones

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

<u>Within study zones</u>

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 <u>or</u> 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 where 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 <u>Between habitats</u>

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 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

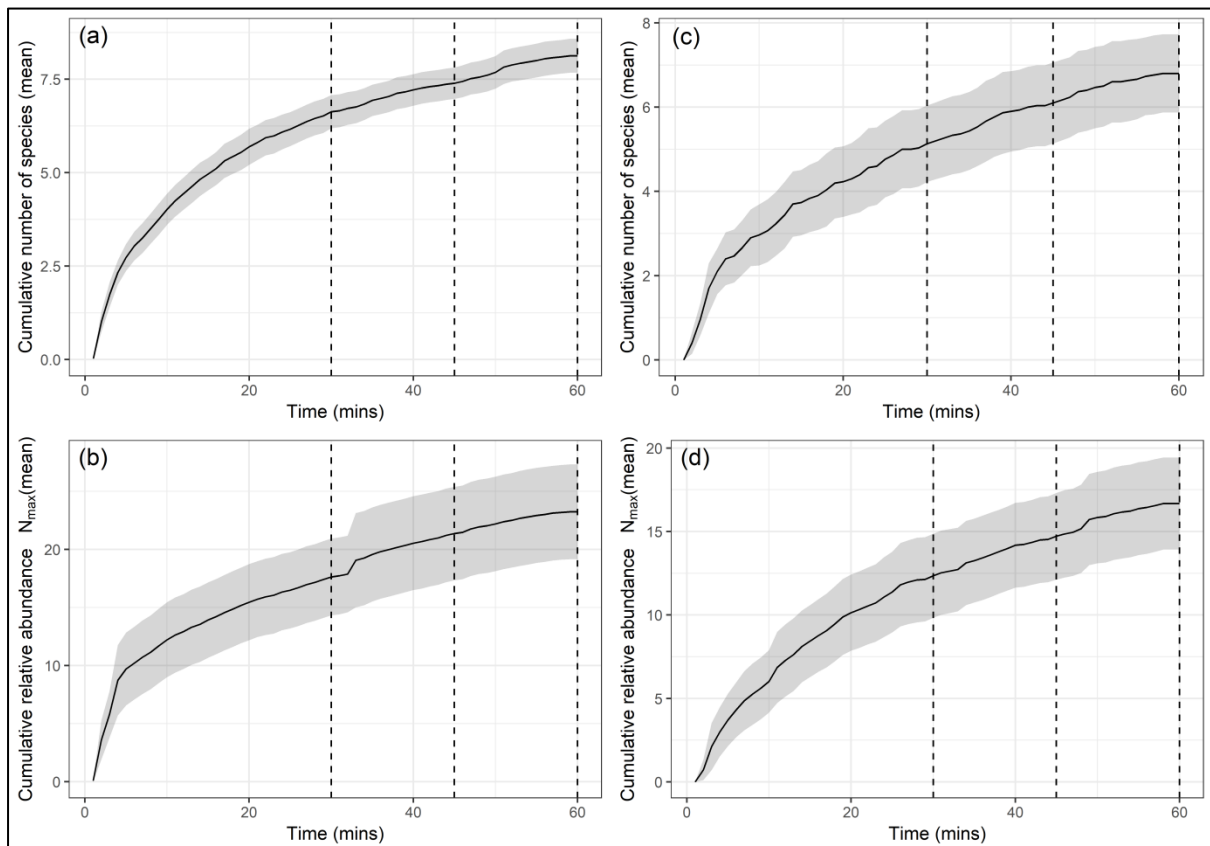336 minutes of video datasets (Fig S1).

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

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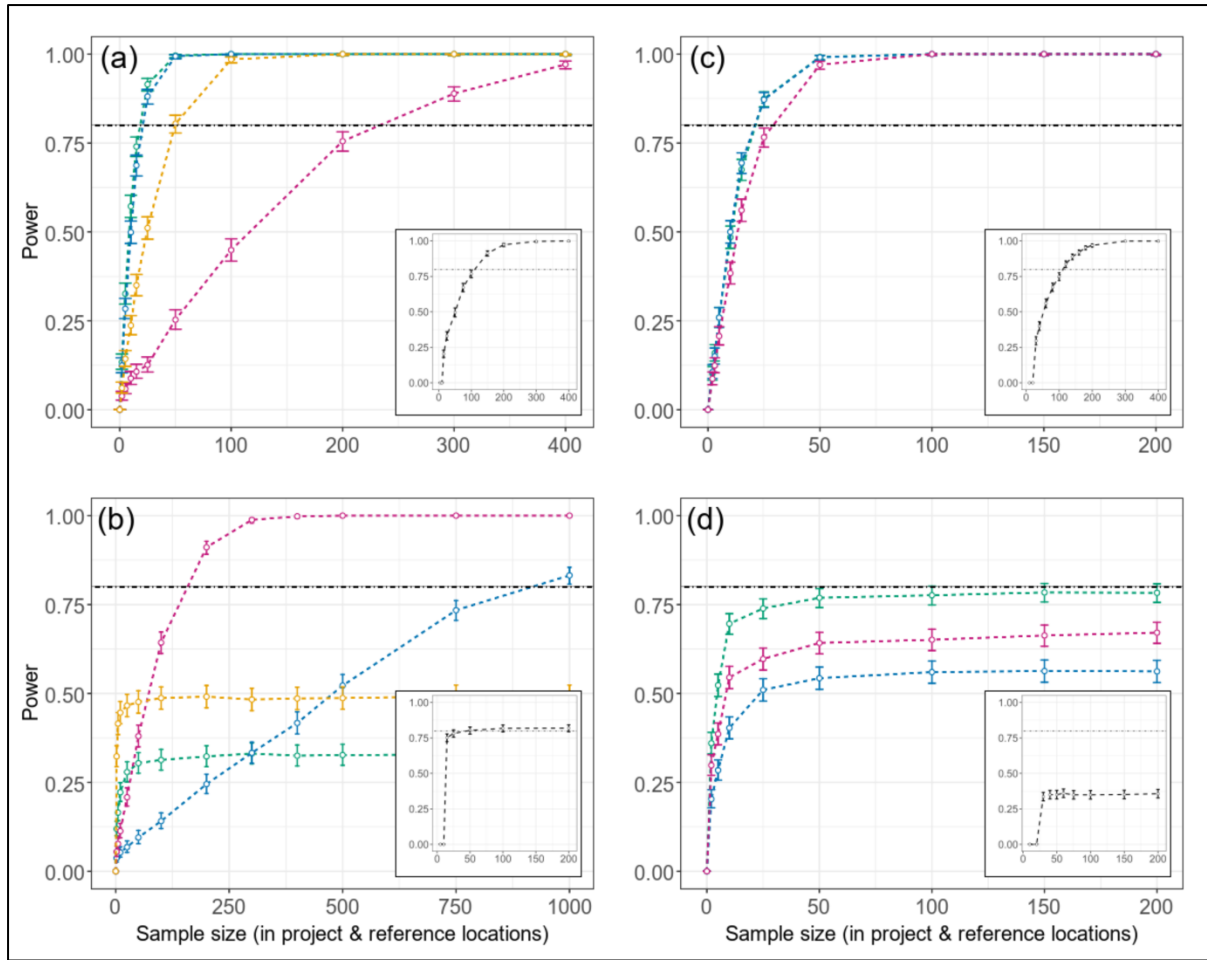
Fig. 4. Minimum effect size SIMR model outputs (±95% CI; <0.05 significance) for pooled spring species richness (*S*) and relative teleost abundance (N_{max}) data (inserted plots) separated by year. (a) = 10% change in *S* at WHSZ, (b) = 10% change in *S* at CRSZ, (c) = 50% change in N_{max} at WHSZ,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

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

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389   <u>Comparison between study zones</u>

390   Assemblage composition and species richness (*S*) models revealed significant differences

391   between study zones (*P*(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).

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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				
		Assemblage				*S*		
Source	d.f.	MS	Pseudo-*F*	*P*(perm)		MS	Pseudo-*F*	*P*(perm)
Zone	1	67326	13.159	**0.0001**		2.6850	8.0528	**0.0029**
Treatment	*4*	*10121*	*3.2071*	***0.0017***		*1.0221*	*6.6190*	***0.0095***
Season	*1*	*4128.7*	*2.6529*	*0.0060*		*0.1888*	*2.0100*	*0.1602*
Location	*1*	*1144.5*	*0.3634*	*0.9417*		*0.2575*	*1.1326*	*0.3050*
Location(Treatment)	*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:

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.
Trisopterus minutus	1.25	1.44	11.4
Pagurus bernhardus	1.18	0.16	8.47
Marthasterias glacialis	0.28	1.18	8.23
Labrus mixtus	0.25	1.24	8.07
Ophiurodea	0.91	0.33	7.48
Ctenolabrus rupestris	0.02	0.85	6.25
Scyliorhinus canicula	1.08	0.91	5.87
Gobiidae	0.52	0.43	5.05

409

410 **Fig. 5.** Species richness (mean ± standard error) and variance for each year at the (a) Wave Hub and

411 (b) Cable Route study zones. Number above bar = sample size.

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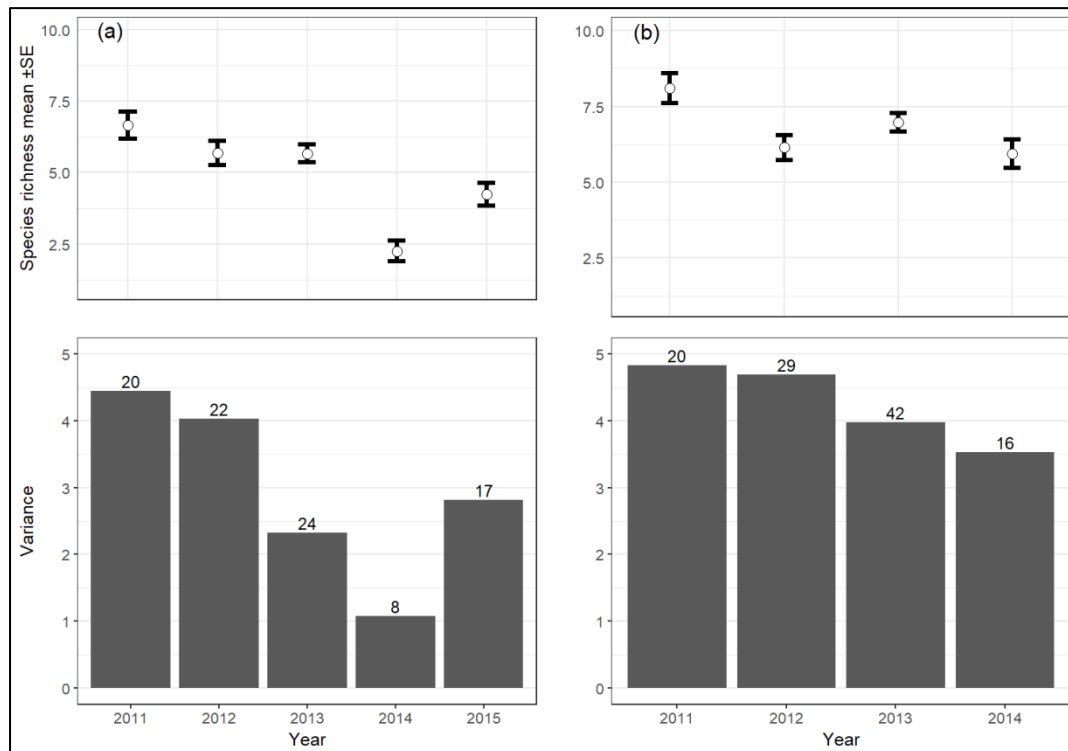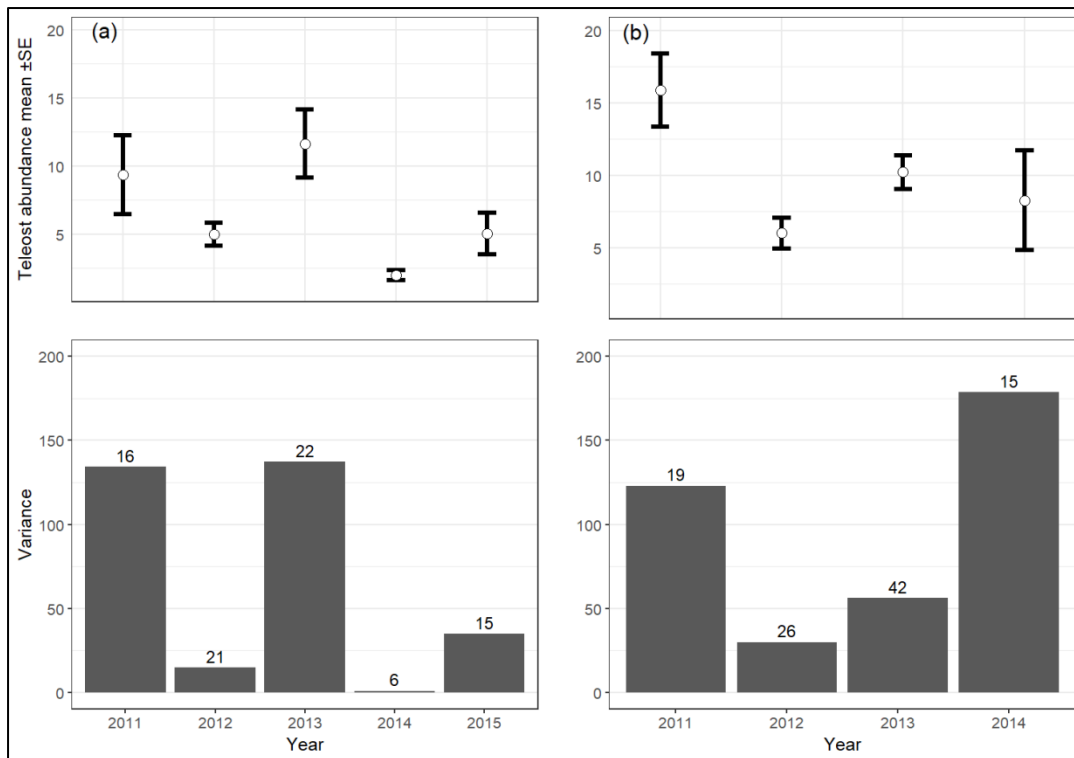419
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Fig. 6. Teleost relative abundance (mean ± standard error) and variance for each year at the (a) Wave Hub and (b) Cable Route study zones. Number above bar = sample size.

Within Wave Hub study zone

No significant fixed effect or interaction term was found in the multivariate assemblage composition models for all species (all *P(perm) values* >0.05; Table 3a; Fig. S5), teleosts (all *P(perm) values* >0.05; Table S2a; Fig. S6) or crustaceans (all *P(perm) values* >0.05; Table S3a; Fig. S6) for the WHSZ.

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

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429   echinoderms, elasmobranchs or *Pollachius spp*. (all *P(perm) values* = >0.05; Tables 3a, S2a,

430   S3a, S5a & S5a), but there was a significant interaction term (*Year x Treatment*) for *Cancer*

431   *pagurus* (*P(perm)* = <0.05; Table S5a). Pairwise tests showed significant differences

432   between years within Wave Hub project (2011-2013 & 2012-213) 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   <u>Within Cable Route study zone</u>

437   Multivariate assemblage composition analysis revealed a significant interaction term, *Year x*

438   *Treatment,* for all species (*P(perm)* = 0.0421; Table 3b; Fig. S5) and teleost models (*P(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(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*(perm) values >0.05; Table S2b and S4b). *Year*

449   was found to be significant in the crustacean (*P(perm)* = 0.0002; Table S3a) and *Cancer*

450   *pagurus* (*P(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(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   <u>Habitats</u>

459   Assemblage composition and species richness (*S*) models revealed significant differences

460   between habitat types (Types 1-3: *P*(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								
		Assemblage (all species)			*S (all species)*			*N_{max} (teleost species)*		
Source	d.f.	MS	Pseudo-*F*	*P*(perm)	MS	Pseudo-*F*	*P*(perm)	MS	Pseudo-*F*	*P*(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
Season	*1*	*3532.5*	*2.2758*	***0.0207***	*0.06245*	*0.7359*	*0.4073*	*0.2483*	*0.2947*	*0.5889*
Habitat	*3*	*5229.8*	*2.9543*	***0.0001***	*0.54909*	*6.9326*	***0.0007***	*1.8946*	*2.2091*	*0.0929*
Location(Treatment)	*6*	*4125.6*	*3.0025*	***0.0001***	*0.14502*	*2.1285*	*0.0588*	*1.2010*	*1.5015*	*0.1880*
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:

Project

Year	t	*P*(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

Reference

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	*P*(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

		Cable Route zone								
(b)		*Assemblage (all species)*			*S (all species)*			N_{max} *(teleost species)*		
Source	d.f.	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:

Project

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**

Reference

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

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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)					
		Assemblage			*S*			*N_{max}*		
Source	d.f.	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
Year	*4*	*8813.1*	*5.2586*	***0.0001***	*0.6820*	*7.8907*	***0.0001***	*4.2311*	*9.4497*	***0.0001***
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			

Discussion

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

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

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

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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.

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