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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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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

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

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

4

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

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

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.

150
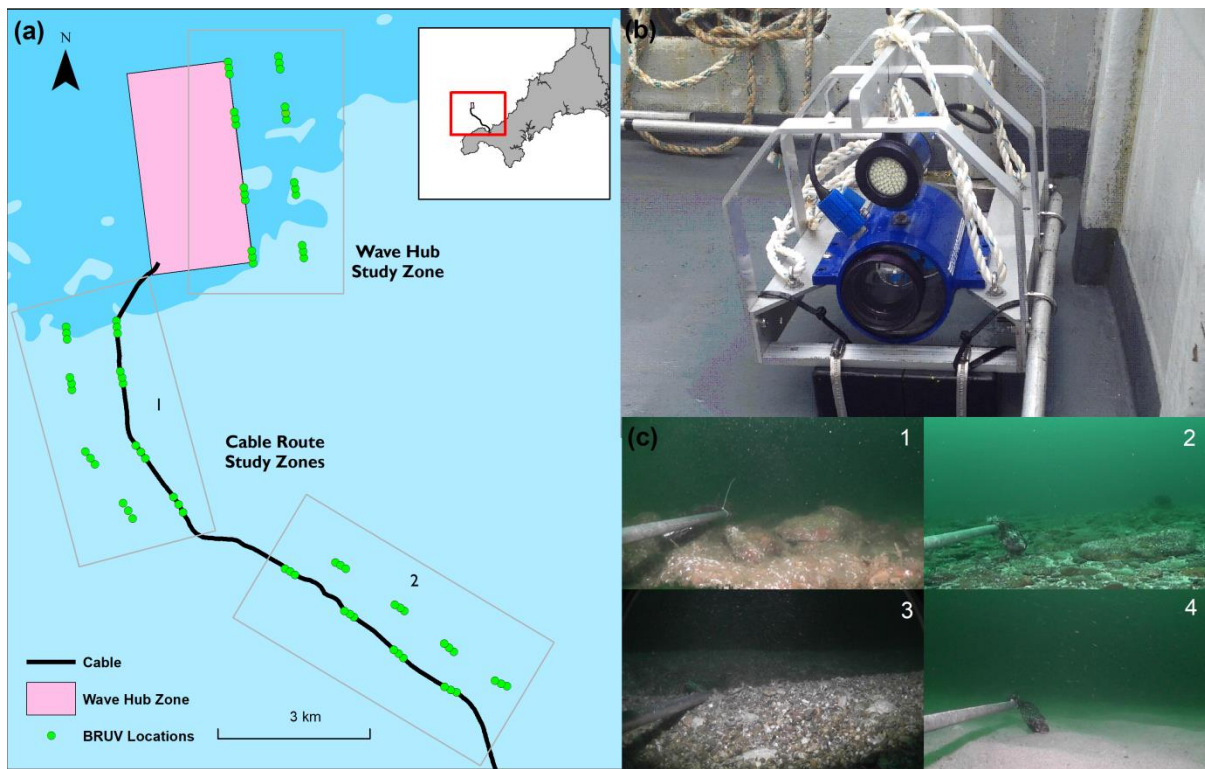
151 **Fig. 1.** Study zones and sampling locations (green filled circles) for BRUV surveys (a), BRUV

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).
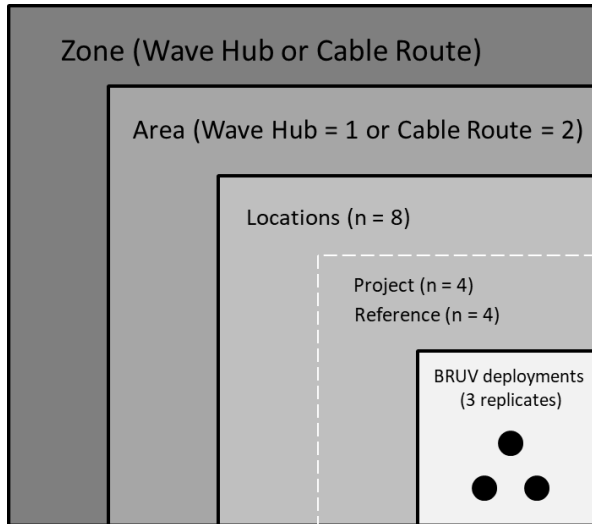
184

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   <u>Between study zones</u>

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

<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

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

Results

Sampling and image quality

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

Further adverse weather during survey activity caused 15 (5%) deployments to be cancelled. Across study areas and years, 297 BRUV deployments were successfully undertaken, of which 38 (12.8%) failed due to technical reasons (e.g. battery or camera failure). Of the remaining 259 (83% of total); 141 video datasets were 60 minutes or longer (45% of total), 199 video datasets were 45 minutes or longer (63% of total), and 247 were 30 minutes or longer (79% of total). Once filtered for visibility (good visibility with an unobstructed view 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).
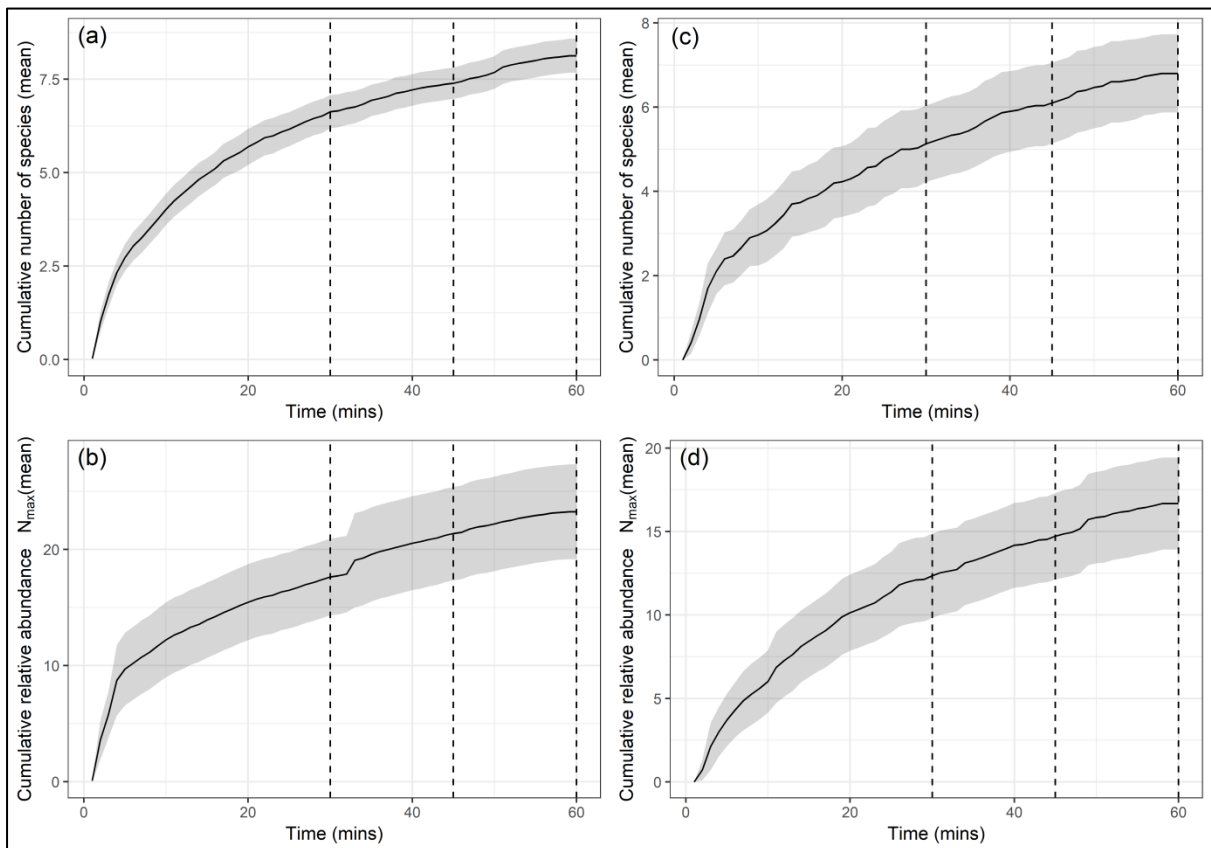
337

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.

	Wave Hub Study Zone				Cable Route Study Zone			
Phylum	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**	

367

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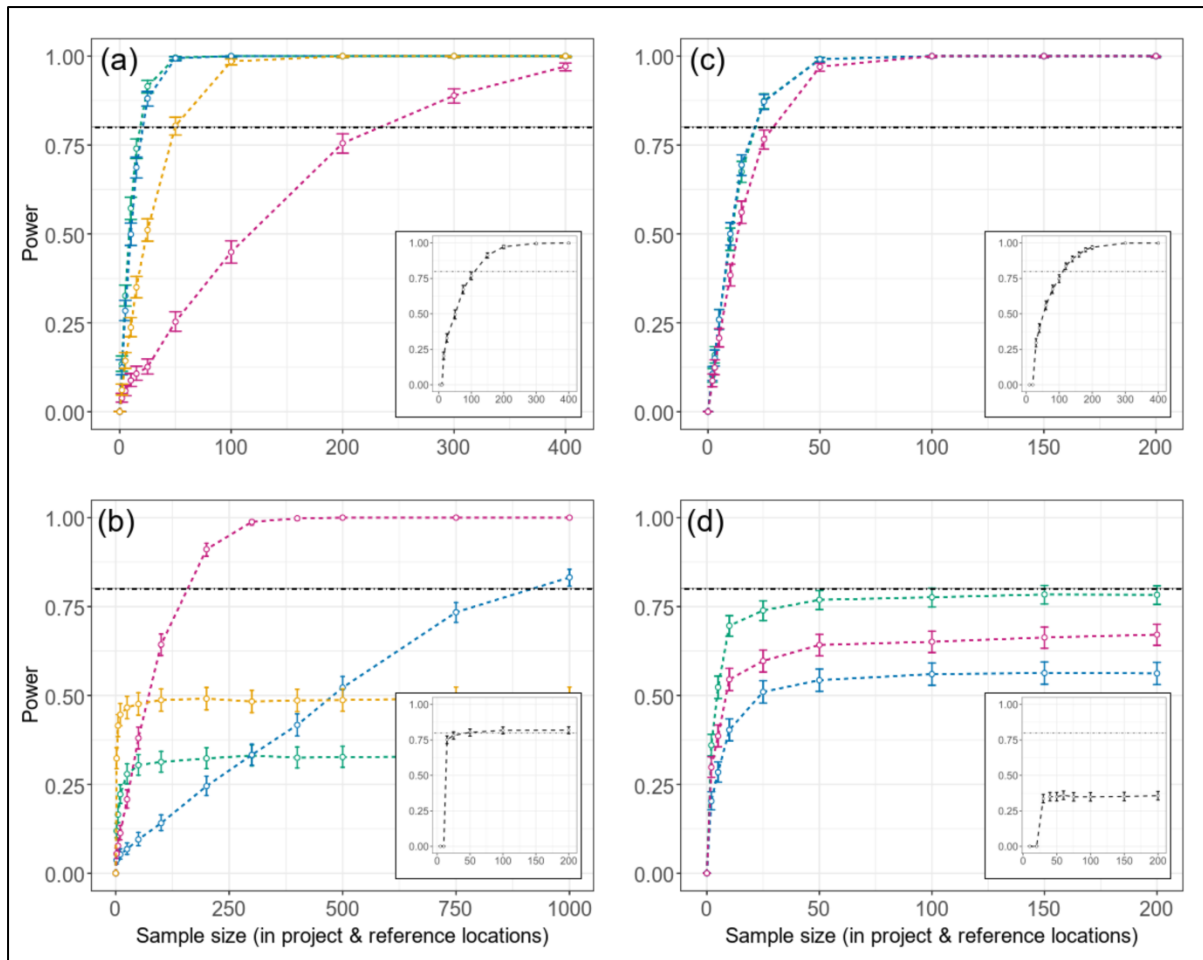
Fig. 4. Minimum effect size SIMR model outputs (±95% CI; <0.05 significance) for pooled spring species richness (S) and relative teleost abundance (N_{max}) data (inserted plots) separated by year. (a) = 10% change in S at WHSZ, (b) = 10% change in S at CRSZ, (c) = 50% change in N_{max} at WHSZ, and (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

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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:					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.

412
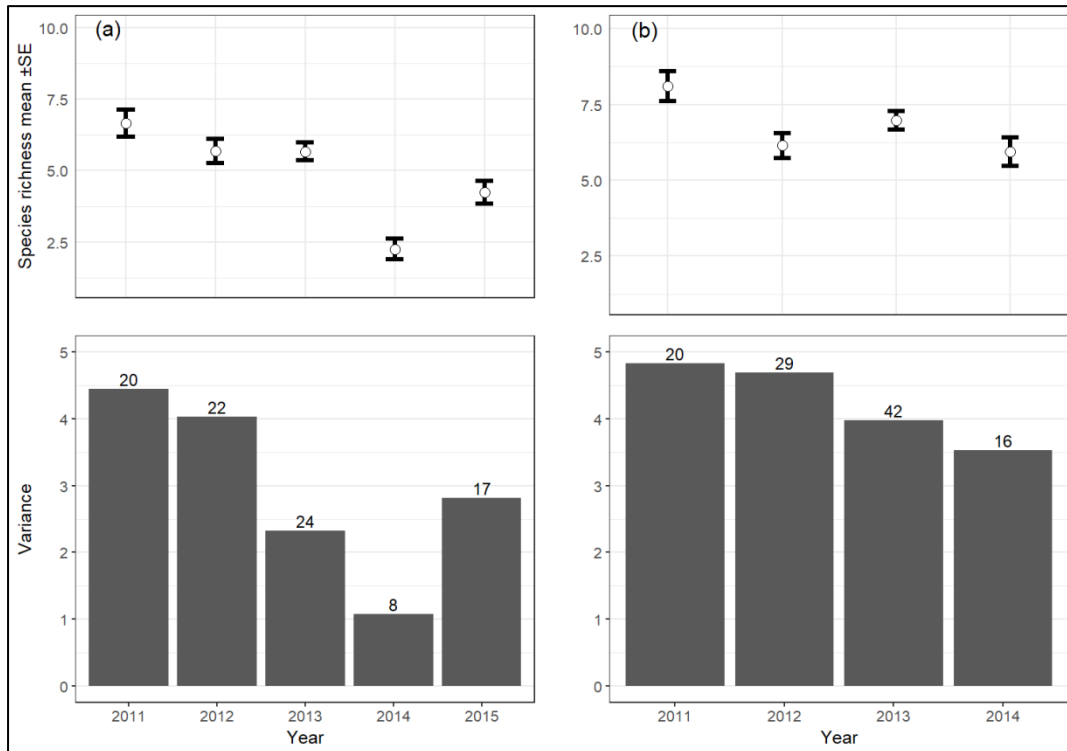
Fig. 6. Teleost relative abundance (mean ± standard error) and variance for each year at the (a) Wave

Hub and (b) Cable Route study zones. Number above bar = sample size.
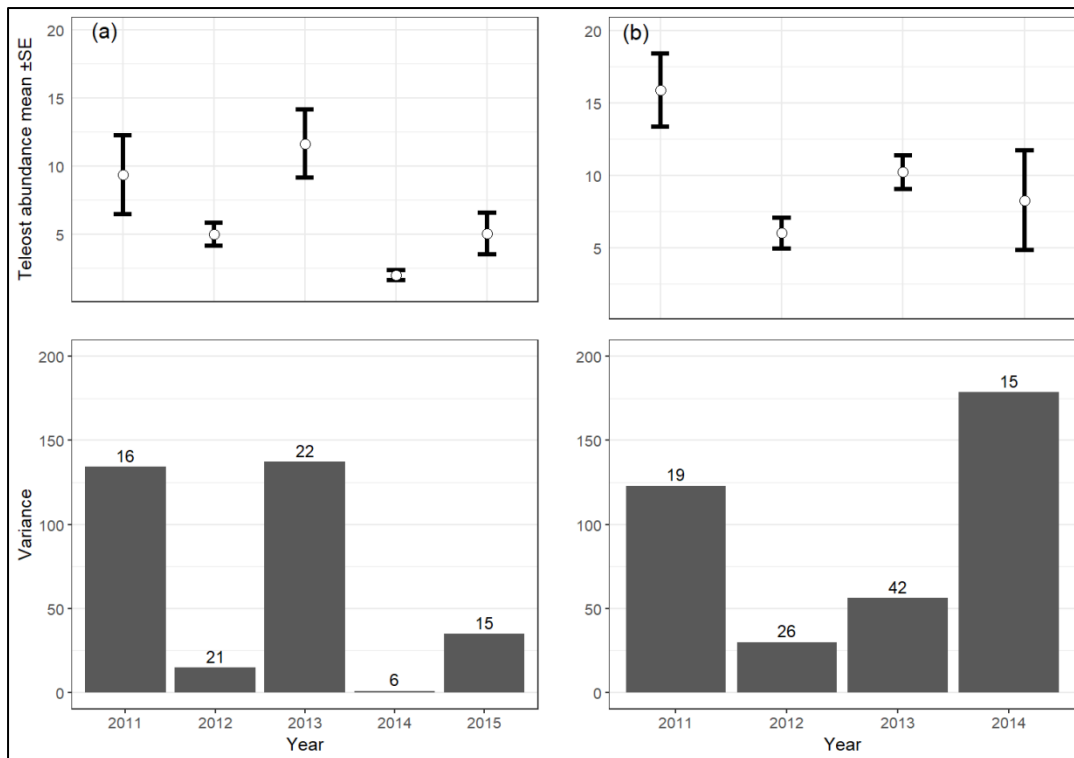
<u>Within Wave Hub study zone</u>

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,

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 Within Cable Route study zone

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   Habitats

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	

(b)

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:

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

Year	t	P(perm)
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 (α=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

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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637 **References**

638 Al-Chokhachy, R., Budy, P., Conner, M., 2009. Detecting declines in the abundance of a bull

639 trout (Salvelinus confluentus) population: understanding the accuracy, precision, and costs of

640 our efforts. Canadian Journal of Fisheries and Aquatic Sciences 66, 649-658.

641 Anderson, M.J., 2001. A new method for non-parametric multivariate analysis of variance.

642 Austral Ecology 26, 32-46.

35

643 Anderson, M.J., Millar, R.B., 2004. Spatial variation and effects of habitat on temperate reef

644 fish assemblages in northeastern New Zealand. Journal of Experimental Marine Biology and

645 Ecology 305, 191-221.

646 Bailey, H., Brookes, K.L., Thompson, P.M., 2014. Assessing environmental impacts of

647 offshore wind farms: lessons learned and recommendations for the future. Aquatic

648 biosystems 10, 8.

649 Bates, D., Mächler, M., Bolker, B., Walker, S., 2015. Fitting Linear Mixed-Effects Models

650 Using lme4. Journal of Statistical Software 67, 1-48.

651 Bernard, A.T.F., Gotz, A., 2012. Bait increases the precision in count data from remote

652 underwater video for most subtidal reef fish in the warm-temperate Agulhas bioregion.

653 Marine Ecology Progress Series 471, 235-252.

654 Bicknell, A.W.J., Godley, B.J., Sheehan, E.V., Votier, S.C., Witt, M.J., 2016. Camera

655 technology for monitoring marine biodiversity and human impact. Frontiers in Ecology and

656 the Environment 14, 424-432.

657 Boehlert, G.W., Gill, A.B., 2010. Environmental and ecological effects of ocean renewable

658 energy development: a current synthesis. Oceanography 23, 68-81.

659 Bray, J.R., Curtis, J.T., 1957. An ordination of the upland forest communities of southern

660 Wisconsin. Ecological monographs 27, 325-349.

661 Bulleri, F., Chapman, M.G., 2010. The introduction of coastal infrastructure as a driver of

662 change in marine environments. Journal of Applied Ecology 47, 26-35.

663 Butenschön, M., Clark, J., Aldridge, J.N., Allen, J.I., Artioli, Y., Blackford, J., Bruggeman,

664 J., Cazenave, P., Ciavatta, S., Kay, S., Lessin, G., van Leeuwen, S., van der Molen, J., de

665 Mora, L., Polimene, L., Sailley, S., Stephens, N., Torres, R., 2016. ERSEM 15.06: a generic

666 model for marine biogeochemistry and the ecosystem dynamics of the lower trophic levels.

667 Geosci. Model Dev. 9, 1293-1339.

668   Campbell, M.S., Stehfest, K.M., Votier, S.C., Hall-Spencer, J.M., 2013. Mapping fisheries

669   for marine spatial planning: Gear-specific vessel monitoring system (VMS), marine

670   conservation and offshore renewable energy. Marine Policy.

671   Cappo, M., Harvey, E., Malcolm, H., Speare, P., 2003. Potential of video techniques to

672   monitor diversity, abundance and size of fish in studies of marine protected areas, in:

673   Beumer, J.P., Grant, A., Smith, D.C. (Eds.), Aquatic Protected Areas-what works best and

674   how do we know?, Cairns, Queensland, Australia, pp. 455-464.

675   Cappo, M., Speare, P., De'ath, G., 2004. Comparison of baited remote underwater video

676   stations (BRUVS) and prawn (shrimp) trawls for assessments of fish biodiversity in inter-

677   reefal areas of the Great Barrier Reef Marine Park. Journal of Experimental Marine Biology

678   and Ecology 302, 123-152.

679   Clarke, K., Chapman, M., Somerfield, P., Needham, H., 2006a. Dispersion-based weighting

680   of species counts in assemblage analyses. Marine Ecology Progress Series 320, 11-27.

681   Clarke, K.R., Somerfield, P.J., Chapman, M.G., 2006b. On resemblance measures for

682   ecological studies, including taxonomic dissimilarities and a zero-adjusted Bray–Curtis

683   coefficient for denuded assemblages. Journal of Experimental Marine Biology and Ecology

684   330, 55-80.

685   Clarke, K.R., Warwick, R.M., 2001. Change in marine communities: an approach to

686   statistical analysis and interpretation. 2nd Edition. PRIMER-E Ltd: Plymouth, United

687   Kingdom.

688   Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K.,

689   Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1997. The

690   value of the world's ecosystem services and natural capital. Nature 387, 253.

691   Coumou, D., Rahmstorf, S., 2012. A decade of weather extremes. Nature Climate Change 2,

692   491.

37

693   Denny, C.M., Willis, T.J., Babcock, R.C., 2004. Rapid recolonisation of snapper Pagrus

694   auratus: Sparidae within an offshore island marine reserve after implementation of no-take

695   status. Marine Ecology Progress Series 272, 183-190.

696   Dorman, S.R., Harvey, E.S., Newman, S.J., 2012. Bait Effects in Sampling Coral Reef Fish

697   Assemblages with Stereo-BRUVs. PLoS ONE 7, e41538.

698   Drakou, E.G., Pendleton, L., Effron, M., Ingram, J.C., Teneva, L., 2017. When ecosystems

699   and their services are not co-located: oceans and coasts. ICES Journal of Marine Science 74,

700   1531-1539.

701   Eigaard, O.R., Bastardie, F., Breen, M., Dinesen, G.E., Hintzen, N.T., Laffargue, P.,

702   Mortensen, L.O., Nielsen, J.R., Nilsson, H.C., O'Neill, F.G., Polet, H., Reid, D.G., Sala, A.,

703   Sköld, M., Smith, C., Sørensen, T.K., Tully, O., Zengin, M., Rijnsdorp, A.D., 2015.

704   Estimating seabed pressure from demersal trawls, seines, and dredges based on gear design

705   and dimensions. ICES Journal of Marine Science: Journal du Conseil.

706   Elliott, S.A.M., Turrell, W.R., Heath, M.R., Bailey, D.M., 2017. Juvenile gadoid habitat and

707   ontogenetic shift observations using stereo-video baited cameras. Marine Ecology Progress

708   Series 568, 123-135.

709   Ellis, D.M., DeMartini, E.E., 1995. Evaluation of a video camera technique for indexing

710   abundances of juvenile pink snapper, Pristipomoides filamentosus, and other Hawaiian

711   insular shelf fishes. Oceanographic Literature Review 42.

712   Forney, K.A., 2000. Environmental Models of Cetacean Abundance: Reducing Uncertainty

713   in Population Trends. Conservation Biology 14, 1271-1286.

714   Fox, C.J., Benjamins, S., Masden, E.A., Miller, R., 2018. Challenges and opportunities in

715   monitoring the impacts of tidal-stream energy devices on marine vertebrates. Renewable and

716   Sustainable Energy Reviews 81, 1926-1938.

717   Franco, A., Quintino, V., Elliott, M., 2015. Benthic monitoring and sampling design and

718   effort to detect spatial changes: A case study using data from offshore wind farm sites.

719   Ecological Indicators 57, 298-304.

720   Galparsoro, I., Borja, A., Uyarra, M.C., 2014. Mapping ecosystem services provided by

721   benthic habitats in the European North Atlantic Ocean. Frontiers in Marine Science 1.

722   Gill, A.B., 2005. Offshore renewable energy: ecological implications of generating electricity

723   in the coastal zone. Journal of Applied Ecology 42, 605-615.

724   Green, P., MacLeod, C.J., 2016. SIMR: an R package for power analysis of generalized linear

725   mixed models by simulation. Methods in Ecology and Evolution 7, 493-498.

726   Griffin, R.A., Robinson, G.J., West, A., Gloyne-Phillips, I.T., Unsworth, R.K.F., 2016.

727   Assessing Fish and Motile Fauna around Offshore Windfarms Using Stereo Baited Video.

728   PLoS ONE 11, e0149701.

729   GWEC, 2016. Global Wind Report: Annual Market Update 2015, p. 76.

730   Halpern, B.S., Walbridge, S., Selkoe, K.A., Kappel, C.V., Micheli, F., D'Agrosa, C., Bruno,

731   J.F., Casey, K.S., Ebert, C., Fox, H.E., Fujita, R., Heinemann, D., Lenihan, H.S., Madin,

732   E.M.P., Perry, M.T., Selig, E.R., Spalding, M., Steneck, R., Watson, R., 2008. A Global Map

733   of Human Impact on Marine Ecosystems. Science 319, 948-952.

734   Harvey, E., Santana Garcon, J., Goetze, J., Saunders, B., Cappo, M., 2018. The Use of

735   Stationary Underwater Video for Sampling Sharks, in: Carrier, J., Heithaus, M.R.,

736   Simpfendorfer, C.A. (Eds.), Shark Research: Emerging Technologies and Applications for

737   the Field and Laboratory. CRC Press, pp. 111-132.

738   Harvey, E.S., Butler, J.J., McLean, D.L., Shand, J., 2012. Contrasting habitat use of diurnal

739   and nocturnal fish assemblages in temperate Western Australia. Journal of Experimental

740   Marine Biology and Ecology 426-427, 78-86.

741   Harvey, E.S., Cappo, M., Butler, J.J., Hall, N., Kendrick, G.A., 2007. Bait attraction affects

742   the performance of remote underwater video stations in assessment of demersal fish

743   community structure. Marine Ecology Progress Series 350, 245-254.

744   Heagney, E.C., Lynch, T.P., Babcock, R.C., Suthers, I.M., 2007. Pelagic fish assemblages

745   assessed using mid-water baited video: standardising fish counts using bait plume size.

746   Marine Ecology Progress Series 350, 255.

747   Heery, E.C., Bishop, M.J., Critchley, L.P., Bugnot, A.B., Airoldi, L., Mayer-Pinto, M.,

748   Sheehan, E.V., Coleman, R.A., Loke, L.H.L., Johnston, E.L., Komyakova, V., Morris, R.L.,

749   Strain, E.M.A., Naylor, L.A., Dafforn, K.A., 2017. Identifying the consequences of ocean

750   sprawl for sedimentary habitats. Journal of Experimental Marine Biology and Ecology 492,

751   31-48.

752   Hiddink, J.G., Jennings, S., Sciberras, M., Szostek, C.L., Hughes, K.M., Ellis, N., Rijnsdorp,

753   A.D., McConnaughey, R.A., Mazor, T., Hilborn, R., Collie, J.S., Pitcher, C.R., Amoroso,

754   R.O., Parma, A.M., Suuronen, P., Kaiser, M.J., 2017. Global analysis of depletion and

755   recovery of seabed biota after bottom trawling disturbance. Proceedings of the National

756   Academy of Sciences.

757   Inger, R., Attrill, M.J., Bearhop, S., Broderick, A.C., James Grecian, W., Hodgson, D.J.,

758   Mills, C., Sheehan, E., Votier, S.C., Witt, M.J., Godley, B.J., 2009. Marine renewable energy:

759   potential benefits to biodiversity? An urgent call for research. Journal of Applied Ecology 46,

760   1145-1153.

761   Jackson, E.L., Langmead, O., Barnes, M., Tyler-Walters, H., Hiscock, K., 2009.

762   Identification of indicator species to represent the full range of benthic life history strategies

763   for Lyme Bay and the consideration of the wider application for monitoring of Marine

764   Protected Areas.

765   Judd, A., 2012. Guidelines for data acquisition to support marine environmental assessments

766   of offshore renewable energy projects. CEFAS, Lowestoft.

767   Langhamer, O., Wilhelmsson, D., 2009. Colonisation of fish and crabs of wave energy

768   foundations and the effects of manufactured holes–a field experiment. Marine Environmental

769   Research 68, 151-157.

770   Liquete, C., Piroddi, C., Drakou, E.G., Gurney, L., Katsanevakis, S., Charef, A., Egoh, B.,

771   2013. Current Status and Future Prospects for the Assessment of Marine and Coastal

772   Ecosystem Services: A Systematic Review. PLOS ONE 8, e67737.

773   Lowry, M., Folpp, H., Gregson, M., Suthers, I., 2012. Comparison of baited remote

774   underwater video (BRUV) and underwater visual census (UVC) for assessment of artificial

775   reefs in estuaries. Journal of Experimental Marine Biology and Ecology 416–417, 243-253.

776   Maclean, I.M.D., Inger, R., Benson, D., Booth, C.G., Embling, C.B., Grecian, W.J.,

777   Heymans, J.J., Plummer, K.E., Shackshaft, M., Sparling, C.E., Wilson, B., Wright, L.J.,

778   Bradbury, G., Christen, N., Godley, B.J., Jackson, A.C., McCluskie, A., Nicholls-Lee, R.,

779   Bearhop, S., 2014. Resolving issues with environmental impact assessment of marine

780   renewable energy installations. Frontiers in Marine Science 1.

781   Maclean, I.M.D., Rehfisch, M.M., Skov, H., Thaxter, C.B., 2013. Evaluating the statistical

782   power of detecting changes in the abundance of seabirds at sea. Ibis 155, 113-126.

783   Malcolm, H.A., Gladstone, W., Lindfield, S., Wraith, J., Lynch, T.P., 2007. Spatial and

784   temporal variation in reef fish assemblages of marine parks in New South Wales, Australia-

785   baited video observations. Marine Ecology Progress Series 350, 277.

786   Mallet, D., Pelletier, D., 2014. Underwater video techniques for observing coastal marine

787   biodiversity: A review of sixty years of publications (1952–2012). Fisheries Research 154,

788   44-62.

789 Masselink, G., Castelle, B., Scott, T., Dodet, G., Suanez, S., Jackson, D., Floc'h, F., 2016.

790 Extreme wave activity during 2013/2014 winter and morphological impacts along the

791 Atlantic coast of Europe. Geophysical Research Letters 43, 2135-2143.

792 Micheli, F., Halpern, B.S., 2005. Low functional redundancy in coastal marine assemblages.

793 Ecology Letters 8, 391-400.

794 Millennium Ecosystem Assessment, 2005. Ecosystems and Human Well-Being: Synthesis.

795 World Resources Institute, Washington, DC.

796 Miller, R.G., Hutchison, Z.L., Macleod, A.K., Burrows, M.T., Cook, E.J., Last, K.S., Wilson,

797 B., 2013. Marine renewable energy development: assessing the Benthic Footprint at multiple

798 scales. Frontiers in Ecology and the Environment 11, 433-440.

799 Neumann, B., Vafeidis, A.T., Zimmermann, J., Nicholls, R.J., 2015. Future Coastal

800 Population Growth and Exposure to Sea-Level Rise and Coastal Flooding - A Global

801 Assessment. PLoS ONE 10, e0118571.

802 Osenberg, C.W., Schmitt, R.J., Holbrook, S.J., Abu-Saba, K.E., Flegal, A.R., 1994. Detection

803 of Environmental Impacts: Natural Variability, Effect Size, and Power Analysis. Ecological

804 Applications 4, 16-30.

805 Pearce, B., Fariñas-Franco, J.M., Wilson, C., Pitts, J., deBurgh, A., Somerfield, P.J., 2014.

806 Repeated mapping of reefs constructed by Sabellaria spinulosa Leuckart 1849 at an offshore

807 wind farm site. Continental Shelf Research 83, 3-13.

808 Priede, I.G., Bagley, P.M., Smith, A., Creasey, S., Merrett, N.R., 1994. Scavenging deep

809 demersal fishes of the Porcupine Seabight, North-east Atlantic: observations by baited

810 camera, trap and trawl. Journal of the Marine Biological Association of the United Kingdom

811 74, 481-498.

812 R Development Core Team, 2011. R: A Language and Environment for Statistical

813 Computing. R Foundation for Statistical Computing, Vienna, Austria.

814   Seitz, R.D., Wennhage, H., Bergström, U., Lipcius, R.N., Ysebaert, T., 2014. Ecological

815   value of coastal habitats for commercially and ecologically important species. ICES Journal

816   of Marine Science 71, 648-665.

817   Sheehan, E.V., Cartwright, A.Y., Witt, M.J., Attrill, M.J., Vural, M., Holmes, L.A., Handling

818   editor: Silvana, B., 2018. Development of epibenthic assemblages on artificial habitat

819   associated with marine renewable infrastructure. ICES Journal of Marine Science, fsy151-

820   fsy151.

821   Sheehan, E.V., Stevens, T.F., Gall, S.C., Cousens, S.L., Attrill, M.J., 2014. Recovery of a

822   Temperate Reef Assemblage in a Marine Protected Area following the Exclusion of Towed

823   Demersal Fishing. PLOS ONE 8, e83883.

824   Sheehan, E.V., Witt, M.J., Cousens, S.L., Gall, S.C., Attrill, M.J., 2013. Benthic Interactions

825   with Renewable Energy Installations in a Temperate Ecosystem, The Twenty-third

826   International Offshore and Polar Engineering Conference. International Society of Offshore

827   and Polar Engineers.

828   Stat, M., John, J., DiBattista, J.D., Newman, S.J., Bunce, M., Harvey, E.S., 2018. Combined

829   use of eDNA metabarcoding and video surveillance for the assessment of fish biodiversity.

830   Conservation Biology 0.

831   Stenberg, C., Støttrup, J.G., van Deurs, M., Berg, C.W., Dinesen, G.E., Mosegaard, H.,

832   Grome, T.M., Leonhard, S.B., 2015. Long-term effects of an offshore wind farm in the North

833   Sea on fish communities. Marine Ecology Progress Series 528, 257-265.

834   Stobart, B., Díaz, D., Álvarez, F., Alonso, C., Mallol, S., Goñi, R., 2015. Performance of

835   baited underwater video: does it underestimate abundance at high population densities? PloS

836   one 10, e0127559.

837   Stobart, B., García-Charton, J.A., Espejo, C., Rochel, E., Goñi, R., Reñones, O., Herrero, A.,

838   Crec'hriou, R., Polti, S., Marcos, C., Planes, S., Pérez-Ruzafa, A., 2007. A baited underwater

839   video technique to assess shallow-water Mediterranean fish assemblages: Methodological

840   evaluation. Journal of Experimental Marine Biology and Ecology 345, 158-174.

841   Underwood, A.J., Chapman, M.G., 2003. Power, precaution, Type II error and sampling

842   design in assessment of environmental impacts. Journal of Experimental Marine Biology and

843   Ecology 296, 49-70.

844   Unsworth, R.K.F., Peters, J.R., McCloskey, R.M., Hinder, S.L., 2014. Optimising stereo

845   baited underwater video for sampling fish and invertebrates in temperate coastal habitats.

846   Estuarine, Coastal and Shelf Science 150, Part B, 281-287.

847   Ward, R.C., Loftis, J.C., McBride, G.B., 1986. The "data-rich but information-poor"

848   syndrome in water quality monitoring. Environmental Management 10, 291-297.

849   Watson, D.L., Anderson, M.J., Kendrick, G.A., Nardi, K., Harvey, E.S., 2009. Effects of

850   protection from fishing on the lengths of targeted and non-targeted fish species at the

851   Houtman Abrolhos Islands, Western Australia. Marine Ecology Progress Series 384, 241-

852   249.

853   Watson, D.L., Harvey, E.S., Kendrick, G.A., Nardi, K., Anderson, M.J., 2007. Protection

854   from fishing alters the species composition of fish assemblages in a temperate-tropical

855   transition zone. Marine Biology 152, 1197-1206.

856   Whitmarsh, S.K., Fairweather, P.G., Huveneers, C., 2016. What is Big BRUVver up to?

857   Methods and uses of baited underwater video. Reviews in Fish Biology and Fisheries, 1-21.

858   Wilding, T.A., Gill, A.B., Boon, A., Sheehan, E., Dauvin, J.C., Pezy, J.-P., O'Beirn, F.,

859   Janas, U., Rostin, L., De Mesel, I., 2017. Turning off the DRIP ('Data-rich, information-

860   poor') – rationalising monitoring with a focus on marine renewable energy developments and

861   the benthos. Renewable and Sustainable Energy Reviews 74, 848-859.

862   Wilhelmsson, D., Malm, T., Öhman, M.C., 2006. The influence of offshore windpower on

863   demersal fish. ICES Journal of Marine Science: Journal du Conseil 63, 775-784.

864   Willis, T.J., Babcock, R.C., 2000. A baited underwater video system for the determination of

865   relative density of carnivorous reef fish. Marine and Freshwater Research 51, 755-763.

866   WindEurope, 2017. The European offshore wind industry: Key trends and statistics 2016, p.

867   37.

868   Witt, M.J., Sheehan, E.V., Bearhop, S., Broderick, A.C., Conley, D.C., Cotterell, S.P., Crow,

869   E., Grecian, W.J., Halsband, C., Hodgson, D.J., Hosegood, P., Inger, R., Miller, P.I., Sims,

870   D.W., Thompson, R.C., Vanstaen, K., Votier, S.C., Attrill, M.J., Godley, B.J., 2012.

871   Assessing wave energy effects on biodiversity: the Wave Hub experience. Philosophical

872   Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences 370,

873   502-529.

874   Zappa, G., Shaffrey, L.C., Hodges, K.I., Sansom, P.G., Stephenson, D.B., 2013. A

875   Multimodel Assessment of Future Projections of North Atlantic and European Extratropical

876   Cyclones in the CMIP5 Climate Models. Journal of Climate 26, 5846-5862.

877   Zintzen, V., Anderson, M.J., Roberts, C.D., Harvey, E.S., Stewart, A.L., Struthers, C.D.,

878   2012. Diversity and Composition of Demersal Fishes along a Depth Gradient Assessed by

879   Baited Remote Underwater Stereo-Video. PLoS ONE 7, e48522.