Marine Environmental Research xxx (xxxx) xxx–xxx

Contents lists available at ScienceDirect

Marine Environmental Research

journal homepage: www.elsevier.com/locate/marenvrev

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

Anthony W.J. Bicknell[a,b,d,*], Emma V. Sheehan[c], Brendan J. Godley[b], Philip D. Doherty[b], Matthew J. Witt[a,b,d]

[a] Environmental Biology, University of Exeter, Hatherly Building, Prince of Wales Road, Exeter EX4 4PS, UK
[b] Centre for Ecology and Conservation, University of Exeter, Penryn, Cornwall TR10 9FE, UK
[c] School of Biological and Marine Sciences, University of Plymouth, Plymouth, Devon PL4 8AA, UK
[d] Environment and Sustainability Institute, University of Exeter, Penryn, Cornwall TR10 9FE, UK

ARTICLE INFO

Keywords:
Marine monitoring
Human impact
Renewable energy
Power analysis
BRUV

ABSTRACT

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

1. 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 et al., 2018; Wilding et al., 2017), and led authors to

---

* Corresponding author.
E-mail addresses: a.bicknell@exeter.ac.uk (A.W.J. Bicknell), emma.sheehan@plymouth.ac.uk (E.V. Sheehan), B.J.Godley@exeter.ac.uk (B.J. Godley), P.Doherty@exeter.ac.uk (P.D. Doherty), M.J.Witt@exeter.ac.uk (M.J. Witt).

https://doi.org/10.1016/j.marenvres.2019.04.007
Received 20 November 2018; Received in revised form 10 April 2019; Accepted 11 April 2019

Please cite this article as: Anthony W.J. Bicknell, et al., Marine Environmental Research, https://doi.org/10.1016/j.marenvres.2019.04.007

term this situation as 'data rich, information poor (DRIP)' (Fox et al., 2018; Ward et al., 1986; Wilding et al., 2017).

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

Effects of design and data on statistical power

The ability to detect patterns/change reduces as variability in the parameter being measured increases	**⇧variance = ⇩power**
Parameter estimates become more precise with larger samples, thus differences between estimates are easier to detect amongst the 'noise'	**⇧sampling = ⇧estimate precision** **⇧estimate precision = ⇧power**
As the effect of the impact increases the more likely it is to be detected. i.e. a 40% change in a parameter estimate is more likely to be detected than 20% change	**⇧effect size = ⇧power**

Underwood and Chapman (2003)

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

2. Materials and methods

2.1. Study location

Baited remote underwater video (BRUV) surveys took place off the north coast of Cornwall (UK) between 2011 and 2015. The study was located within and adjacent to a MREI development zone (Wave Hub) and the associated seabed cable (Fig. 1a). Study zones ranged between 3 and 10.5 nautical miles (5.6 and 19.5 km) offshore in water depths of 20–53 m (at Lowest Astronomical Tide). The seabed cable and marker buoys (6 in total) for the development zone were installed in autumn 2010. The cable was buried when on sand (near shore) and covered with 80,000 tonnes of rock (creating a berm of 0.3 m minimum height) and concrete matressing every 120 m when laid on hard substrate (deeper water). Access within 500 m of the electrical seabed hub (plug) was prohibited, but access was permitted elsewhere. The presence of marker buoys has prevented commercial mid-water or bottom trawling for fish. The only commercial fishing that has taken place in the development zone or close to the seabed cable since installation is crustacean potting. There were no connected deployments of operating marine energy devices during the period of the study.

2.2. Sampling equipment

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

2.3. Survey and design

BRUV surveys were scheduled twice per year, in spring and autumn,

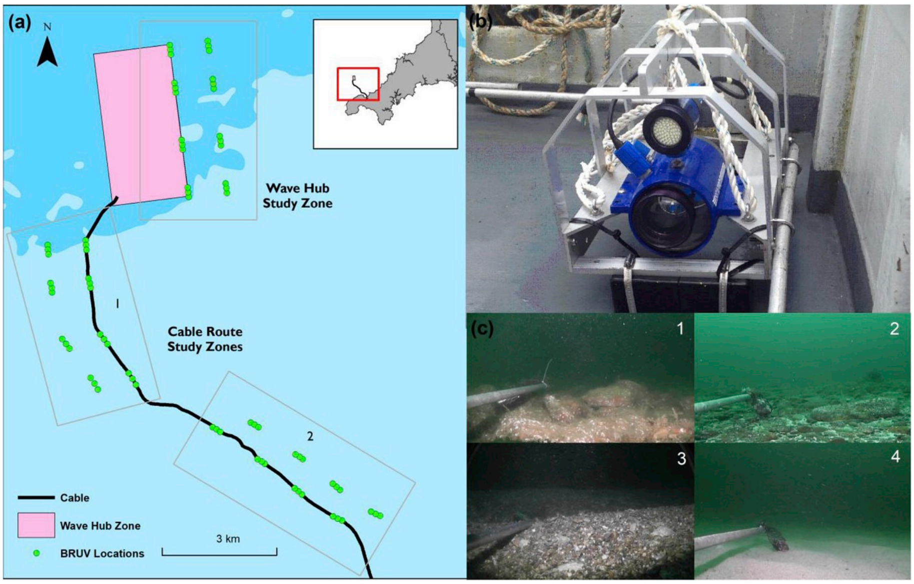

Fig. 1. Study zones and sampling locations (green filled circles) for BRUV surveys (a), BRUV housing, frame and LED light (b), and example habitat types from the video footage (c) 1 = rocky reef, 2 = large (course) sediment, 3 = medium (mixed) sediment, 4 = fine sediment. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.).

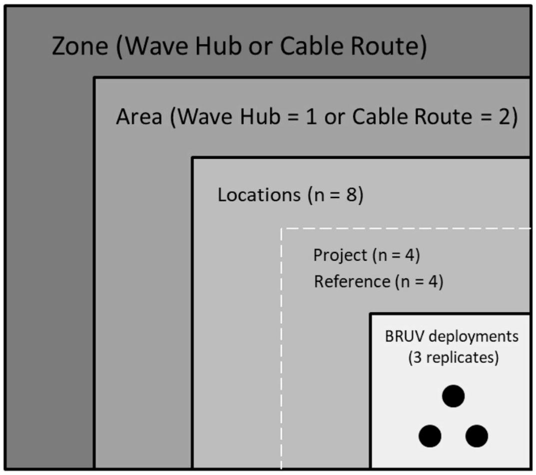

Fig. 2. Nested design schematic for baited remote underwater video surveys.

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

In order to investigate the BRUVs greatest distance of attraction (in metres) for teleosts, we calculated the ‘effective range of attraction’ (AR) (formulised in Cappo et al., 2004; see Appendix S3) for increasing soak time (i.e. time cameras are in the water, or video footage analysed from the start). The average seabed current speeds (data from POLPRED seabed CS20 models, NOC; http://www.pol.ac.uk/) during BRUV deployments ($0.23\,\mathrm{m\,s^{-1}}$), and a maximum fish (endurance) swimming speed of $0.6\,\mathrm{m\,s^{-1}}$ (~200–300 mm fish length) were used as the AR parameters (V_c = current speed, V_f = maximum fish speed).

2.4. Image analysis

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

2.5. Data analysis

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

2.5.1. Species accumulation

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

2.5.2. Statistical power, effect and sample size

Power analyses were conducted using the SIMR package (Green and MacLeod, 2016) in R version 3.4.4 (R Development Core Team, 2011)

Table 1
Number of species and individuals by taxonomic phylum on BRUV footage at the Wave Hub and Cable Route study zones. The 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**	

to investigate the relationships between the number of samples (sample size), the size of change (effect size) and the probability to detect change (power) in *S* or $N_{\max}$.

Generalised linear mixed effect models were fitted to data (pooled data with years combined) collected in spring at the WHSZ and CRSZ (separate models) using the lme4 package (Bates et al., 2015). Only spring data were used in the models to remove seasonal effects (evidenced in PERMANOVA models; Tables 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.

2.5.3. *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 ($N_{\max}$).

2.5.4. *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.

2.5.5. *Within study zones*

For each study zone (WHSZ & CRSZ), models for *S* and assemblage composition were performed on the complete species dataset, and all three response variables for two defined species guilds; teleost and crustacean. The factors *year* and *treatment* were fixed, and *season*, *habitat*, *zone* (Cable Route only; 2 zones) and *treatment* nested in *location* were random. The factor *year* had four and five levels for the CRSZ and the WHSZ respectively. The factor *treatment* had two levels (WHSZ or CRSZ project, and reference). Depth and current speed (tide indicator)

Table 2
PERMANOVA results for species assemblage and richness (S) models by study zones. Fixed effects are non-italicized and random effects italicized. Significant permutation p-values below 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)
WH vs CR	3.6276	**0.0001**

Pair-wise test for Zone:

Group	t	*P*(perm)
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

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 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 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)		Cable Route 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	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**						

(*continued on next page*)

Table 3 (*continued*)

(b)			Cable Route 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)
	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						

were considered as environmental co-variates in initial models for each study zone but where removed when found not to influence models.

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

2.5.6. Between habitats

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

Prior to calculation of the Bray–Curtis (Bray and Curtis, 1957) similarity index, multivariate data (assemblage composition) were dispersion weighted and square root transformed to down weight taxa with erratic abundances and/or high abundances (Clarke et al., 2006a). As joint species absences were important to consider between treatments, data were ‘zero-adjusted by adding a dummy value of 1 (Clarke et al., 2006b). Without the dummy value, 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).

3. Results

3.1. 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 min or longer (45% of total), 199 video datasets were 45 min or longer (63% of total), and 247 were 30 min 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 min or more (37% of total), 161 datasets with 45 min or more (51% of total), and 198 datasets with 30 min or more (63% of total) (Fig. S2).

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

3.2. Species accumulation

The maximum number of species identified during video datasets lasting 60 min was 12. The mean number of species occurring on RRLS habitats was 6.8, while for GS it was 5.3 (Fig. 3a and c). Mean N_{max} was greater for RRLS habitats (maximum of 132 individuals) compared to GS habitats (maximum 38 individuals) (Fig. 3b and d). In both habitats, 75% or more of the total *S* and N_{max} (at 60 min) was achieved after 30 min, with the greatest increase in number of species and abundance

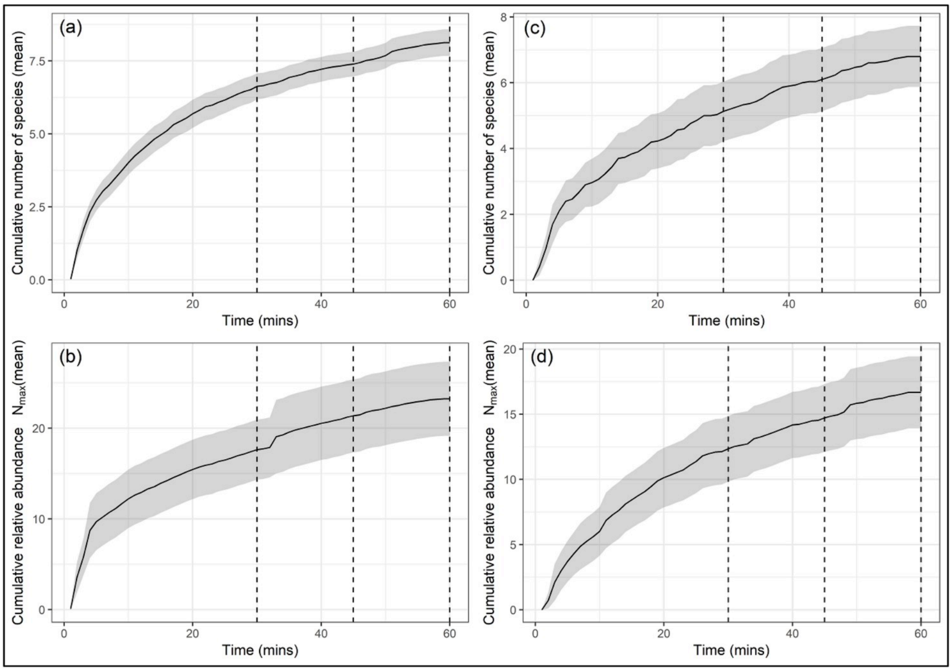

Fig. 3. Mean species (a & c) and relative abundance ($N_{\max}$) (b & d) accumulation curves from 60 min BRUV footage for rocky reef/large sediment habitat (a & b) and medium/fine sediment habitats (c & d). Grey shading = 95% confidence interval. Dashed lines = 30, 45 and 60 min.

(rate of change) occurring in the first 20 min of video datasets (Fig. S1).

3.3. Optimal data

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

3.4. Species diversity and taxonomic composition

A total of 67 species from 46 families and 6 phyla were observed on the BRUV footage across the two study zones for the survey period. This equated to 5440 individual animals, the vast majority of which were fishes (teleost ~73%). The WHSZ was more speciose (49 species) compared to the CRSZ (42 species), but total abundance was higher in the latter with > 3400 individuals (Table 1).

3.5. Statistical power, effect and sample size

Models with pooled spring data (years combined) revealed the minimum change that could be detected with 0.8 (or more) power was ~10% in richness for both WHSZ and CRSZ, which would require ≥200 samples (~100 within each project and reference). The samples required to detect 20% change in *S* reduce to ~40–50 (~20–25 within each project and reference) for both WHSZ and CRSZ (Fig. 4a and c sub plots). For $N_{\max}$, the minimum change detectable was ~50% at the WHSZ (Fig. 4b sub plot), which required ≥100 samples (~50 within each project and reference). The 0.8 power threshold was not reached for any level of change in $N_{\max}$ using pooled spring data from the CRSZ, and the power to detect 50% change did not exceed 0.4, no matter the sample size (Fig. 4d sub plot).

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 ~40 to > 400 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 power with ~150 samples (~75 within project and reference), and 2013 data would achieve this level with ~1700 samples (~850 within project and reference) (Fig. 4c). The only data to achieve 0.8 power to detect 50% change in $N_{\max}$ at CRSZ was for 2012, which would require ~200 samples (100 within project and reference) (Fig. 4d).

3.6. Assemblage, species richness and abundance analyses

3.6.1. Comparison between study zones

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

3.6.2. 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 ($N_{\max}$)

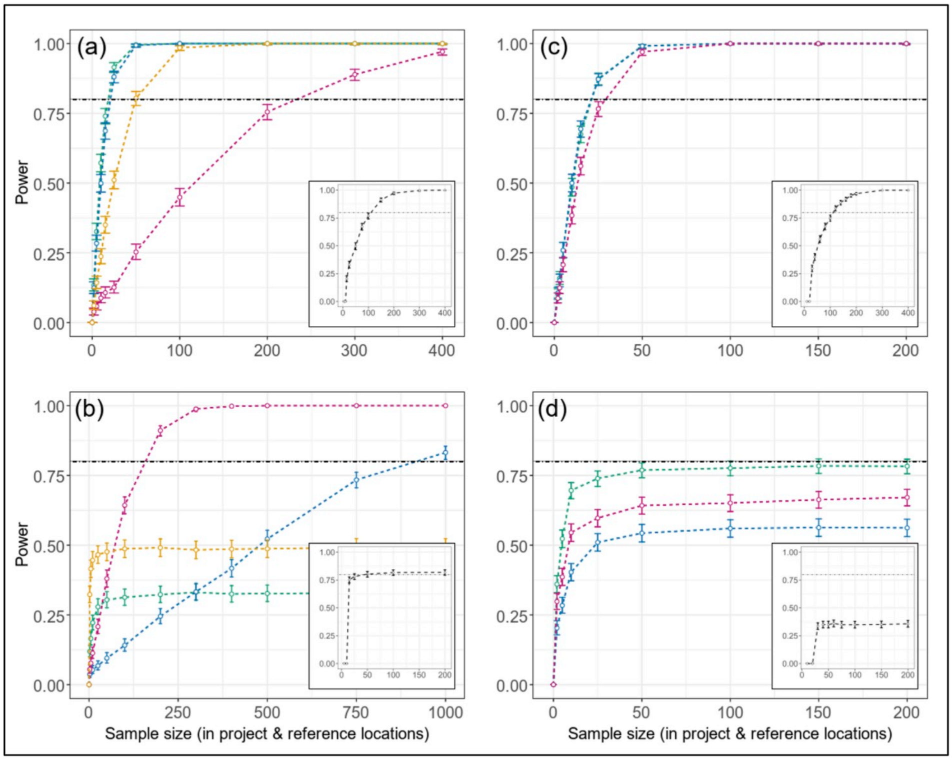

Fig. 4. Minimum effect size SIMR model outputs (± 95% CI; < 0.05 significance) for pooled spring species richness (*S*) and relative teleost abundance ($N_{\max}$) data (inserted plots) separated by year. (a) = 20% change in *S* at WHSZ, (b) = 20% change in *S* at CRSZ, (c) = 50% change in $N_{\max}$ 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. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

models had no significant fixed effect or interaction term for teleosts, crustaceans, echinoderms, elasmobranchs or *Pollachius spp*. (all *P(perm) values* = > 0.05; Tables 3a, S2a, S3a, S5a & S5a), but there was a significant interaction term (*Year x Treatment*) for *Cancer pagurus* (*P (perm)* = < 0.05; Table S5a). Pairwise tests showed significant differences between years within Wave Hub project (2011–2013 & 2012–213) or reference locations (2011–2012), and for locations within one year (2011), however some pairwise tests could not be conducted indicating a limitation of the data.

3.6.3. *Within cable route study zone*

Multivariate assemblage composition analysis revealed a significant interaction term, *Year x Treatment,* for all species (*P(perm)* = 0.0421; Table 3b; Fig. S5) and teleost models (*P(perm)* = 0.0118; Table S2b; Fig. S6). Pairwise tests revealed both had significant differences between years within Cable Route project and reference locations, but not between Cable Route project and reference locations within years (Table 3b & Table S2b). The crustacean model had a significant *Year* term (*P(perm)* = 0.0137; Table S3b; Figs. S6) and 3 out of the 6 subsequent pairwise tests were significant (2011–13, 2011–14 & 2013–14; Table S3b). Species richness (*S*) models had no significant fixed terms for all species or teleosts (Table 3b & Table S2b), but the crustacean model revealed *Year* as significant (*P* = < 0.001; Table S3b). Four out of the 6 subsequent pairwise tests were significant (2011–13, 2012–13, 2012–14 & 2013–14; Table S3b). Relative abundance ($N_{\max}$) models had no significant fixed terms for teleosts, echinoderms or

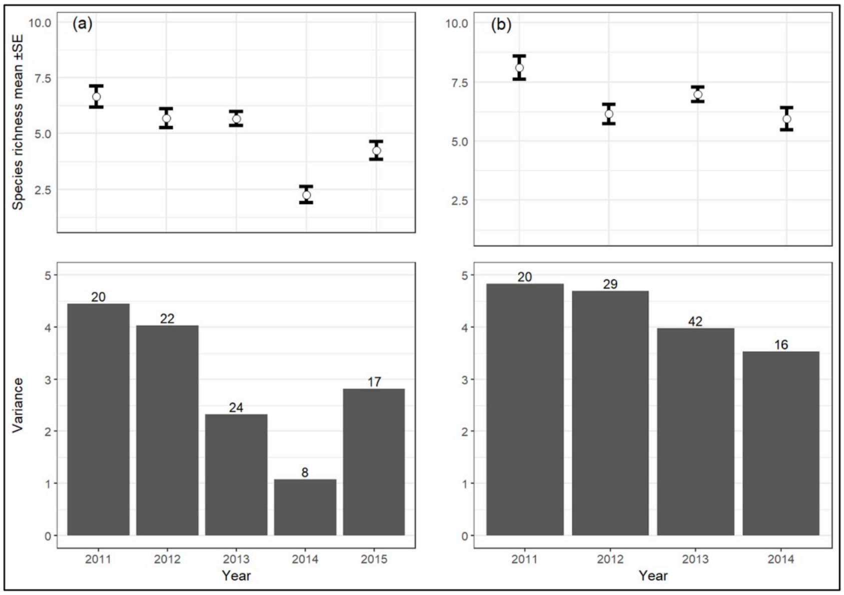

Fig. 5. Species richness (mean ± standard error) and variance for each year at the (a) Wave Hub and (b) Cable Route study zones. Number above bar = sample size.

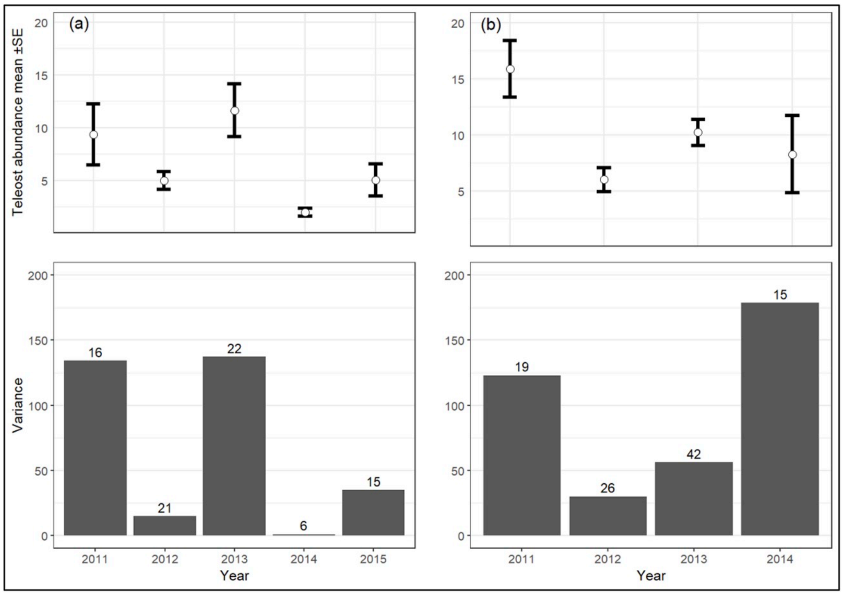

Fig. 6. Teleost relative abundance (mean ± standard error) and variance for each year at the (a) Wave Hub and (b) Cable Route study zones. Number above bar = sample size.

elasmobranchs (all *P*(perm) values > 0.05; Tables S2b and S4b). *Year* was found to be significant in the crustacean (*P(perm)* = 0.0002; Table S3a) and *Cancer pagurus* (*P(perm)* = 0.0002; Table S5b) models, with 4 from 6 pairwise test significant for crustaceans (2011–14, 2012–13, 2012–14, 2013–14; Tables S3a) and 3 from 6 pairwise test significant for *Cancer pagurus* (2011–13, 2012–13, 2013–14; Table S5b). The only model to have a significant fixed *Treatment* term was for relative abundance of *Pollachius spp.* (*P(perm)* = 0.01; Table S5b), with significantly greater abundance in Cable Route project than reference locations (Cable Route: mean 1.4 ± 0.8 SD, n = 23; reference = mean 1 ± 0 SD, n = 7).

3.6.4. Habitats

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

4. 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. overestimate 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

Table 4
PERMANOVA results for species assemblage, richness (S) and relative abundance ($N_{\max}$) models by habitat types; 1–3. Fixed effects are non-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		
	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			

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 min to 2 h 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 min in our study, but only the first 30 min 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 min 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 min (116 videos) compared to a 30 min 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 min duration video data. However, a limitation of only deploying, or using data, for a short period (30 min 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 (MPAs) by excluding damaging seabed fishing activities, the areas could become refuges for species with small or recovering populations. Deploying a subset of BRUVs for longer could help detect such species and be incorporated into study designs, specifically for this purpose. In this study, no new species were detected when 60 min footage was compared to 30 min, but longer periods may be necessary to detect rare species. The specifics detailed here are particular to our study but highlight, more generally, the need to understand the survey technique being used and the data it is expected to capture. Moreover, how the precautionary approach to sampling effort previously recommended (over-estimation) is necessary to allow redundancy in sampling campaigns when working in highly dynamic environments.

The analyses conducted using the BRUV data provided little consistent evidence of differences in metrics (across taxa) between the locations influenced by either the trawling exclusion (Wave Hub) or cable infrastructure (Cable Route), and reference locations within study zones. The only consistent change across years was an increase in the relative abundance of pollack and saithe (*Pollachius spp.)* around the cable infrastructure. These are commercially important coastal species understood to associate with rocky reef and hard substrate habitat as nursery areas (Seitz et al., 2014), suggesting the addition of rock and concrete matressing on the cable may be providing suitable conditions. Greater abundance of cuckoo wrasse *Labrus mixtus* observed during towed camera surveys in the same area has also been attributed to the presence of the hard substrate for cable protection (Sheehan et al., 2013). Both observations are consistent with other studies of fish abundance around MREI structures (Wilhelmsson et al., 2006) and add to the evidence these introduced structures and associated infrastructure are created habitat for species to utilise (Inger et al., 2009; Miller et al., 2013; Sheehan et al., 2018; Witt et al., 2012). The lack of evidence towards an impact of trawling exclusion in the Wave Hub zone may relate to low fishing effort in the area before (Campbell et al., 2013) and during the study, creating equivalent disturbance (or lack of) in project and reference locations. Alternatively, it could be an artefact of the survey design and power to detect change. The power analyses revealed how the ability to detect change in species richness and teleost relative abundance differed considerably between zones (spatial) and among survey years (temporal), producing a range of sampling schedules to detect the same effect size and varying levels of power (Fig. 4). Low power was particularly apparent in teleost abundance indicating these data could only confidently detect large changes (> 50%) with high probability ($\alpha = 0.05$), and much less for the CRSZ. This could explain the lack of significant results found for this metric in our univariate analyses (Table 2 & Table S2), with potential influence on the outcome of the multivariate assemblage composition analysis (Fig. S5). Low probability to detect directional change in abundance data due to high variability has been found for other survey techniques and species (e.g. fish, cetaceans & seabirds; Al-Chokhachy et al., 2009; Forney, 2000; Maclean et al., 2013). Our results re-iterate caution in accepting that no change is taking place when analyses fail to reveal statistically significant patterns, when a lack of statistical power may be the contributing factor (Al-Chokhachy et al., 2009; Maclean et al., 2013). Despite good statistical power (~0.8) to detect relatively small changes (20%) in species richness, the majority of statistically significant effects found in our analyses (uni- and multivariate) were associated with yearly differences (Table 2 and Tables S2–5). Only a single year difference was found for the exclusion zone ('*Treatment*') in richness of all species combined (2012; Table 2a) and crustaceans (2014; Table S3a) at the WHSZ. Natural yearly variation was the main effect identified at the study zones suggesting high levels of 'noise' in the system, presenting potential difficulty in distinguishing impacts related to the Wave Hub exclusion zone or cable route infrastructure. Moreover, it demonstrates how multi-year data are essential to capture site variability and provide accurate baseline characterisation, from which single site or cumulative impact from renewable energy convertors or manufactured infrastructure could be robustly assessed (Maclean et al., 2014).

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

The difference in assemblage composition, species richness and variance found between study zones (Table 2; Figs. 4 and 5) might be expected given the dominant habitat types found at the Wave Hub (medium to fine sediment) and Cable Route (mix of rocky reef and large boulders) study zones (e.g. Fig. S7), and the significant difference in composition and richness found among them (Table 4). The site specific nature of ecosystem components, even over a relatively small spatial area (~130 km^2), is effectively demonstrated using these data. The characterisations these data offer also illustrate how useful consistent data collection methods (and survey design) can be in allowing comparison between locations. Baseline characterisation and impact studies of benthic habitats and species are required at MREIs located in coastal locations (inshore and offshore) with various physical (e.g. size, bathymetry) and ecological characteristics. While the energy convertor design may differ between locations the ability to understand more general or cumulative effects caused by developments would be enhanced by multi-site data, collected using the same survey techniques. Adopting standard practices and guidelines for pre- and post-development benthic survey methods, design and analysis would help optimise costs associated with EIAs and, if adopted across multi-sites and multi-years, could ultimately lead to impact models with predictive power for species, communities or ecosystems (e.g. Butenschön et al., 2016).

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

As part of this study the BRUV system demonstrated its value as a tool for collecting assemblage composition, species richness and relative abundance data for epi-benthic mobile species in highly dynamic conditions, and a good candidate for use as part of marine EIAs across latitudes. The system offers a cost effective and flexible method that can provide the spatial and temporal coverage that is difficult to obtain using other methods (i.e. divers, remote underwater vehicles). When used with stereo cameras, BRUV can also offer size data that could help elucidate more detailed age related effects cause by introduced or altered habitat (Elliott et al., 2017), or converted into biomass estimates (standing or relative) providing another metric to assess impact. Although other traditional methods of sampling these communities, e.g. trawling, potting, bottom lines or nets, can provide these metrics and work in similar or worse sea conditions, they can be destructive or taxa specific, increasing the cost of survey effort and/or to the ecosystem of study. With further development, BRUV systems also have the potential to help address data collection gaps surveys often suffer from, e.g. diurnal variation. With the integration of movement sensors (i.e. infrared) and/or artificial intelligent (AI) algorithms to activate equipment on species presence, 24 h deployments could be possible. BRUV biodiversity data has also been found to be complimentary to environmental DNA (Stat et al., 2018), suggesting a combined approach using these non-invasive methods could further enhance the effectiveness of monitoring surveys.

Acknowledgments

This work was funded by a Natural Environment Research Grant (NE/J012319/1), an Engineering and Physical Sciences Research Council Grant (EP/P026109/1), an Intelligent Energy Europe (IEE) Grant (IEE/09/809/SI2.558291) and a European Cross-border Cooperation INTERREG IV Grant, co-founded by the European Regional Development Fund (ERDF). Thank you to Chris Lowe (Newquay Sea Safaris and Fishing) and Lech Kwiatkowski for safely skippering survey vessels, and two anonymous reviewers for their helpful comments and suggestions.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.marenvres.2019.04.007.

References

Al-Chokhachy, R., Budy, P., Conner, M., 2009. Detecting declines in the abundance of a bull trout (Salvelinus confluentus) population: understanding the accuracy, precision, and costs of our efforts. Can. J. Fish. Aquat. Sci. 66, 649–658.

Anderson, M.J., 2001. A new method for non-parametric multivariate analysis of variance. Austral Ecol. 26, 32–46.

Anderson, M.J., Millar, R.B., 2004. Spatial variation and effects of habitat on temperate reef fish assemblages in northeastern New Zealand. J. Exp. Mar. Biol. Ecol. 305, 191–221.

Bailey, H., Brookes, K.L., Thompson, P.M., 2014. Assessing environmental impacts of offshore wind farms: lessons learned and recommendations for the future. Aquat. Biosyst. 10, 8.

Bates, D., Mächler, M., Bolker, B., Walker, S., 2015. Fitting linear mixed-effects models using lme4. J. Stat. Softw. 67, 1–48.

Bernard, A.T.F., Gotz, A., 2012. Bait increases the precision in count data from remote underwater video for most subtidal reef fish in the warm-temperate Agulhas bioregion. Mar. Ecol. Prog. Ser. 471, 235–252.

Bicknell, A.W.J., Godley, B.J., Sheehan, E.V., Votier, S.C., Witt, M.J., 2016. Camera technology for monitoring marine biodiversity and human impact. Front. Ecol. Environ. 14, 424–432.

Boehlert, G.W., Gill, A.B., 2010. Environmental and ecological effects of ocean renewable energy development: a current synthesis. Oceanography 23, 68–81.

Bray, J.R., Curtis, J.T., 1957. An ordination of the upland forest communities of southern Wisconsin. Ecol. Monogr. 27, 325–349.

Bulleri, F., Chapman, M.G., 2010. The introduction of coastal infrastructure as a driver of change in marine environments. J. Appl. Ecol. 47, 26–35.

Butenschön, M., Clark, J., Aldridge, J.N., Allen, J.I., Artioli, Y., Blackford, J., Bruggeman, J., Cazenave, P., Ciavatta, S., Kay, S., Lessin, G., van Leeuwen, S., van der Molen, J., de Mora, L., Polimene, L., Sailley, S., Stephens, N., Torres, R., 2016. ERSEM 15.06: a generic model for marine biogeochemistry and the ecosystem dynamics of the lower trophic levels. Geosci. Model Dev. (GMD) 9, 1293–1339.

Campbell, M.S., Stehfest, K.M., Votier, S.C., Hall-Spencer, J.M., 2013. Mapping fisheries for marine spatial planning: gear-specific vessel monitoring system (VMS), marine conservation and offshore renewable energy. Mar. Pol. 45, 293–300.

Cappo, M., Harvey, E., Malcolm, H., Speare, P., 2003. Potential of video techniques to monitor diversity, abundance and size of fish in studies of marine protected areas. In: Beumer, J.P., Grant, A., Smith, D.C. (Eds.), Aquatic Protected Areas-What Works Best and How Do We Know? pp. 455–464 Cairns, Queensland, Australia.

Cappo, M., Speare, P., De'ath, G., 2004. Comparison of baited remote underwater video stations (BRUVS) and prawn (shrimp) trawls for assessments of fish biodiversity in inter-reefal areas of the Great Barrier Reef Marine Park. J. Exp. Mar. Biol. Ecol. 302, 123–152.

Clarke, K., Chapman, M., Somerfield, P., Needham, H., 2006a. Dispersion-based weighting of species counts in assemblage analyses. Mar. Ecol. Prog. Ser. 320, 11–27.

Clarke, K.R., Somerfield, P.J., Chapman, M.G., 2006b. On resemblance measures for ecological studies, including taxonomic dissimilarities and a zero-adjusted Bray–Curtis coefficient for denuded assemblages. J. Exp. Mar. Biol. Ecol. 330, 55–80.

Clarke, K.R., Warwick, R.M., 2001. Change in Marine Communities: an Approach to Statistical Analysis and Interpretation, second ed. PRIMER-E Ltd, Plymouth, United Kingdom.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. Nature 387, 253.

Coumou, D., Rahmstorf, S., 2012. A decade of weather extremes. Nat. Clim. Change 2, 491.

Denny, C.M., Willis, T.J., Babcock, R.C., 2004. Rapid recolonisation of snapper Pagrus auratus: sparidae within an offshore island marine reserve after implementation of no-take status. Mar. Ecol. Prog. Ser. 272, 183–190.

Dorman, S.R., Harvey, E.S., Newman, S.J., 2012. Bait effects in sampling coral reef fish assemblages with stereo-BRUVs. PLoS One 7, e41538.

Drakou, E.G., Pendleton, L., Effron, M., Ingram, J.C., Teneva, L., 2017. When ecosystems and their services are not co-located: oceans and coasts. ICES J. Mar. Sci. 74, 1531–1539.

Eigaard, O.R., Bastardie, F., Breen, M., Dinesen, G.E., Hintzen, N.T., Laffargue, P., Mortensen, L.O., Nielsen, J.R., Nilsson, H.C., O'Neill, F.G., Polet, H., Reid, D.G., Sala, A., Sköld, M., Smith, C., Sørensen, T.K., Tully, O., Zengin, M., Rijnsdorp, A.D., 2015. Estimating seabed pressure from demersal trawls, seines, and dredges based on gear design and dimensions. ICES J. Mar. Sci.: Journal du Conseil 73, i27–i43.

Elliott, S.A.M., Turrell, W.R., Heath, M.R., Bailey, D.M., 2017. Juvenile gadoid habitat and ontogenetic shift observations using stereo-video baited cameras. Mar. Ecol. Prog. Ser. 568, 123–135.

Ellis, D.M., DeMartini, E.E., 1995. Evaluation of a video camera technique for indexing abundances of juvenile pink snapper, Pristipomoides filamentosus, and other Hawaiian insular shelf fishes. Oceanogr. Lit. Rev. 42.

Forney, K.A., 2000. Environmental models of cetacean abundance: reducing uncertainty in population trends. Conserv. Biol. 14, 1271–1286.

Fox, C.J., Benjamins, S., Masden, E.A., Miller, R., 2018. Challenges and opportunities in monitoring the impacts of tidal-stream energy devices on marine vertebrates. Renew. Sustain. Energy Rev. 81, 1926–1938.

Franco, A., Quintino, V., Elliott, M., 2015. Benthic monitoring and sampling design and effort to detect spatial changes: a case study using data from offshore wind farm sites. Ecol. Indicat. 57, 298–304.

Galparsoro, I., Borja, A., Uyarra, M.C., 2014. Mapping ecosystem services provided by benthic habitats in the European north Atlantic ocean. Front. Mar. Sci. 1.

Gill, A.B., 2005. Offshore renewable energy: ecological implications of generating electricity in the coastal zone. J. Appl. Ecol. 42, 605–615.

Green, P., MacLeod, C.J., 2016. SIMR: an R package for power analysis of generalized linear mixed models by simulation. Methods in Ecol. Evol. 7, 493–498.

Griffin, R.A., Robinson, G.J., West, A., Gloyne-Phillips, I.T., Unsworth, R.K.F., 2016. Assessing fish and motile fauna around offshore windfarms using stereo baited video. PLoS One 11, e0149701.

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

Halpern, B.S., Walbridge, S., Selkoe, K.A., Kappel, C.V., Micheli, F., D'Agrosa, C., Bruno, J.F., Casey, K.S., Ebert, C., Fox, H.E., Fujita, R., Heinemann, D., Lenihan, H.S., Madin, E.M.P., Perry, M.T., Selig, E.R., Spalding, M., Steneck, R., Watson, R., 2008. A global map of human impact on marine ecosystems. Science 319, 948–952.

Harvey, E., Santana Garcon, J., Goetze, J., Saunders, B., Cappo, M., 2018. The use of stationary underwater video for sampling sharks. In: Carrier, J., Heithaus, M.R., Simpfendorfer, C.A. (Eds.), Shark Research: Emerging Technologies and Applications for the Field and Laboratory. CRC Press, pp. 111–132.

Harvey, E.S., Butler, J.J., McLean, D.L., Shand, J., 2012. Contrasting habitat use of diurnal and nocturnal fish assemblages in temperate Western Australia. J. Exp. Mar. Biol. Ecol. 426–427, 78–86.

Harvey, E.S., Cappo, M., Butler, J.J., Hall, N., Kendrick, G.A., 2007. Bait attraction affects the performance of remote underwater video stations in assessment of demersal fish community structure. Mar. Ecol. Prog. Ser. 350, 245–254.

Heagney, E.C., Lynch, T.P., Babcock, R.C., Suthers, I.M., 2007. Pelagic fish assemblages assessed using mid-water baited video: standardising fish counts using bait plume size. Mar. Ecol. Prog. Ser. 350, 255.

Heery, E.C., Bishop, M.J., Critchley, L.P., Bugnot, A.B., Airoldi, L., Mayer-Pinto, M., Sheehan, E.V., Coleman, R.A., Loke, L.H.L., Johnston, E.L., Komyakova, V., Morris, R.L., Strain, E.M.A., Naylor, L.A., Dafforn, K.A., 2017. Identifying the consequences of ocean sprawl for sedimentary habitats. J. Exp. Mar. Biol. Ecol. 492, 31–48.

Hiddink, J.G., Jennings, S., Sciberras, M., Szostek, C.L., Hughes, K.M., Ellis, N., Rijnsdorp, A.D., McConnaughey, R.A., Mazor, T., Hilborn, R., Collie, J.S., Pitcher, C.R., Amoroso, R.O., Parma, A.M., Suuronen, P., Kaiser, M.J., 2017. Global analysis of depletion and recovery of seabed biota after bottom trawling disturbance. Proc. Natl. Acad. Sci. Unit. States Am. 31, 8301–8306.

Inger, R., Attrill, M.J., Bearhop, S., Broderick, A.C., James Grecian, W., Hodgson, D.J., Mills, C., Sheehan, E., Votier, S.C., Witt, M.J., Godley, B.J., 2009. Marine renewable energy: potential benefits to biodiversity? An urgent call for research. J. Appl. Ecol. 46, 1145–1153.

Jackson, E.L., Langmead, O., Barnes, M., Tyler-Walters, H., Hiscock, K., 2009. Identification of Indicator Species to Represent the Full Range of Benthic Life History Strategies for Lyme Bay and the Consideration of the Wider Application for Monitoring of Marine Protected Areas.

Judd, A., 2012. Guidelines for Data Acquisition to Support Marine Environmental Assessments of Offshore Renewable Energy Projects. CEFAS, Lowestoft.

Langhamer, O., Wilhelmsson, D., 2009. Colonisation of fish and crabs of wave energy foundations and the effects of manufactured holes–a field experiment. Mar. Environ. Res. 68, 151–157.

Liquete, C., Piroddi, C., Drakou, E.G., Gurney, L., Katsanevakis, S., Charef, A., Egoh, B., 2013. Current status and future prospects for the assessment of marine and coastal ecosystem services: a systematic review. PLoS One 8, e67737.

Lowry, M., Folpp, H., Gregson, M., Suthers, I., 2012. Comparison of baited remote underwater video (BRUV) and underwater visual census (UVC) for assessment of artificial reefs in estuaries. J. Exp. Mar. Biol. Ecol. 416–417, 243–253.

Maclean, I.M.D., Inger, R., Benson, D., Booth, C.G., Embling, C.B., Grecian, W.J., Heymans, J.J., Plummer, K.E., Shackshaft, M., Sparling, C.E., Wilson, B., Wright, L.J., Bradbury, G., Christen, N., Godley, B.J., Jackson, A.C., McCluskie, A., Nicholls-Lee, R., Bearhop, S., 2014. Resolving issues with environmental impact assessment of marine renewable energy installations. Front. Mar. Sci. 1.

Maclean, I.M.D., Rehfisch, M.M., Skov, H., Thaxter, C.B., 2013. Evaluating the statistical power of detecting changes in the abundance of seabirds at sea. Ibis 155, 113–126.

Malcolm, H.A., Gladstone, W., Lindfield, S., Wraith, J., Lynch, T.P., 2007. Spatial and temporal variation in reef fish assemblages of marine parks in New South Wales, Australia-baited video observations. Mar. Ecol. Prog. Ser. 350, 277.

Mallet, D., Pelletier, D., 2014. Underwater video techniques for observing coastal marine biodiversity: a review of sixty years of publications. Fish. Res. 154 (1952–2012), 44–62.

Masselink, G., Castelle, B., Scott, T., Dodet, G., Suanez, S., Jackson, D., Floc'h, F., 2016. Extreme wave activity during 2013/2014 winter and morphological impacts along the Atlantic coast of Europe. Geophys. Res. Lett. 43, 2135–2143.

Micheli, F., Halpern, B.S., 2005. Low functional redundancy in coastal marine assemblages. Ecol. Lett. 8, 391–400.

Millennium Ecosystem Assessment, 2005. Ecosystems and Human Well-Being: Synthesis. World Resources Institute, Washington, DC.

Miller, R.G., Hutchison, Z.L., Macleod, A.K., Burrows, M.T., Cook, E.J., Last, K.S., Wilson, B., 2013. Marine renewable energy development: assessing the Benthic Footprint at multiple scales. Front. Ecol. Environ. 11, 433–440.

Neumann, B., Vafeidis, A.T., Zimmermann, J., Nicholls, R.J., 2015. Future coastal population growth and Exposure to sea-level rise and coastal flooding - a global assessment. PLoS One 10, e0118571.

Osenberg, C.W., Schmitt, R.J., Holbrook, S.J., Abu-Saba, K.E., Flegal, A.R., 1994. Detection of environmental impacts: natural variability, effect size, and power analysis. Ecol. Appl. 4, 16–30.

Pearce, B., Fariñas-Franco, J.M., Wilson, C., Pitts, J., deBurgh, A., Somerfield, P.J., 2014. Repeated mapping of reefs constructed by Sabellaria spinulosa Leuckart 1849 at an offshore wind farm site. Cont. Shelf Res. 83, 3–13.

Priede, I.G., Bagley, P.M., Smith, A., Creasey, S., Merrett, N.R., 1994. Scavenging deep demersal fishes of the Porcupine Seabight, North-east Atlantic: observations by baited camera, trap and trawl. J. Mar. Biol. Assoc. U. K. 74, 481–498.

R Development Core Team, 2011. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.

Seitz, R.D., Wennhage, H., Bergström, U., Lipcius, R.N., Ysebaert, T., 2014. Ecological value of coastal habitats for commercially and ecologically important species. ICES J. Mar. Sci. 71, 648–665.

Sheehan, E.V., Cartwright, A.Y., Witt, M.J., Attrill, M.J., Vural, M., Holmes, L.A., 2018. Development of epibenthic assemblages on artificial habitat associated with marine renewable infrastructure. ICES J. Mar. Sci fsy151.

Sheehan, E.V., Stevens, T.F., Gall, S.C., Cousens, S.L., Attrill, M.J., 2014. Recovery of a temperate reef assemblage in a marine protected area following the exclusion of towed demersal fishing. PLoS One 8, e83883.

Sheehan, E.V., Witt, M.J., Cousens, S.L., Gall, S.C., Attrill, M.J., 2013. Benthic interactions with renewable energy installations in a temperate ecosystem, the twenty-third international offshore and polar Engineering conference. Int. Soc. Offshore Polar Eng.

Stat, M., John, J., DiBattista, J.D., Newman, S.J., Bunce, M., Harvey, E.S., 2018. Combined use of eDNA metabarcoding and video surveillance for the assessment of fish biodiversity. Conserv. Biol. 0.

Stenberg, C., Støttrup, J.G., van Deurs, M., Berg, C.W., Dinesen, G.E., Mosegaard, H., Grome, T.M., Leonhard, S.B., 2015. Long-term effects of an offshore wind farm in the North Sea on fish communities. Mar. Ecol. Prog. Ser. 528, 257–265.

Stobart, B., Díaz, D., Álvarez, F., Alonso, C., Mallol, S., Goñi, R., 2015. Performance of baited underwater video: does it underestimate abundance at high population densities? PLoS One 10, e0127559.

Stobart, B., García-Charton, J.A., Espejo, C., Rochel, E., Goñi, R., Reñones, O., Herrero, A., Crec'hriou, R., Polti, S., Marcos, C., Planes, S., Pérez-Ruzafa, A., 2007. A baited underwater video technique to assess shallow-water Mediterranean fish assemblages: methodological evaluation. J. Exp. Mar. Biol. Ecol. 345, 158–174.

Underwood, A.J., Chapman, M.G., 2003. Power, precaution, Type II error and sampling design in assessment of environmental impacts. J. Exp. Mar. Biol. Ecol. 296, 49–70.

Unsworth, R.K.F., Peters, J.R., McCloskey, R.M., Hinder, S.L., 2014. Optimising stereo baited underwater video for sampling fish and invertebrates in temperate coastal habitats. Estuar. Coast Shelf Sci. 150, 281–287 Part B.

Ward, R.C., Loftis, J.C., McBride, G.B., 1986. The "data-rich but information-poor" syndrome in water quality monitoring. Environ. Manag. 10, 291–297.

Watson, D.L., Anderson, M.J., Kendrick, G.A., Nardi, K., Harvey, E.S., 2009. Effects of protection from fishing on the lengths of targeted and non-targeted fish species at the Houtman Abrolhos Islands, Western Australia. Mar. Ecol. Prog. Ser. 384, 241–249.

Watson, D.L., Harvey, E.S., Kendrick, G.A., Nardi, K., Anderson, M.J., 2007. Protection from fishing alters the species composition of fish assemblages in a temperate-tropical transition zone. Mar. Biol. 152, 1197–1206.

Whitmarsh, S.K., Fairweather, P.G., Huveneers, C., 2016. What is Big BRUVver up to? Methods and uses of baited underwater video. Rev. Fish Biol. Fish. 1–21.

Wilding, T.A., Gill, A.B., Boon, A., Sheehan, E., Dauvin, J.C., Pezy, J.-P., O'Beirn, F., Janas, U., Rostin, L., De Mesel, I., 2017. Turning off the DRIP ('Data-rich, information-poor') – rationalising monitoring with a focus on marine renewable energy developments and the benthos. Renew. Sustain. Energy Rev. 74, 848–859.

Wilhelmsson, D., Malm, T., Öhman, M.C., 2006. The influence of offshore windpower on demersal fish. ICES J. Mar. Sci.: Journal du Conseil 63, 775–784.

Willis, T.J., Babcock, R.C., 2000. A baited underwater video system for the determination of relative density of carnivorous reef fish. Mar. Freshw. Res. 51, 755–763.

WindEurope, 2017. The European Offshore Wind Industry: Key Trends and Statistics 2016. pp. 37.

Witt, M.J., Sheehan, E.V., Bearhop, S., Broderick, A.C., Conley, D.C., Cotterell, S.P., Crow, E., Grecian, W.J., Halsband, C., Hodgson, D.J., Hosegood, P., Inger, R., Miller, P.I., Sims, D.W., Thompson, R.C., Vanstaen, K., Votier, S.C., Attrill, M.J., Godley, B.J., 2012. Assessing wave energy effects on biodiversity: the Wave Hub experience. Phil. Trans. Math. Phys. Eng. Sci. 370, 502–529.

Zappa, G., Shaffrey, L.C., Hodges, K.I., Sansom, P.G., Stephenson, D.B., 2013. A multimodel assessment of future projections of north Atlantic and European Extratropical cyclones in the CMIP5 climate models. J. Clim. 26, 5846–5862.

Zintzen, V., Anderson, M.J., Roberts, C.D., Harvey, E.S., Stewart, A.L., Struthers, C.D., 2012. Diversity and composition of demersal fishes along a depth gradient assessed by baited remote underwater stereo-video. PLoS One 7, e48522.