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Development of epibenthic assemblages on artificial habitat associated with marine renewable infrastructure.

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Abstract

Offshore marine renewable energy installations (MREI) introduce structure into the marine environment and can locally exclude destructive, bottom trawl fishing. These effects have the potential to aid restoration of degraded seabed habitats but may be constrained by timescales of ecological succession following MREI construction, and the removal of infrastructure during decommissioning. To inform managers about appropriate decommissioning strategies, a 25 km cable and associated rock armouring (Wave Hub, UK), installed on rocky reef, was monitored up to five years post-deployment. The epibenthic Assemblage Composition, and Number of Taxa remained significantly different from surrounding controls, while Abundance was similar in all survey years between the cable and controls. Six morphotaxa showed four patterns of colonisation on cable plots compared to the controls: 1) Early colonisation, which remained in greater abundances (Porifera), 2) early colonisation, converging (Turf), 3) slow colonisation, converging Anthozoa and Vertebrata), and 4) slow colonisation, remaining lower in abundance (Tunicata and Echinoderms). The environmental relevance of this MREI is considered relatively benign as it covers 0.01 % of the surrounding bioregion, appears to be supporting similar assemblages to the surrounding habitat, and exhibited minimal evidence of invasive species (three records of two non-native species). Longer monitoring timescales are required to provide comprehensive, site-specific decommissioning advice.

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

INTRODUCTION

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

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

Introducing hard substrate into the marine environment creates habitat for species to colonise (Langhamer, 2012; Bishop *et al.*, 2017), with the potential to increase biodiversity (Firth *et al.*, 2016), which can enhance the resilience of ecosystem functions (Oliver *et al.*, 2015). As structures are gradually colonised they can have a positive impact on the supply of ecosystem services; including the formation of biogenic habitat to provide feeding and nursery resources for commercially important species (Langhamer, 2012) and species of conservation importance (Langhamer and Wilhelmsson, 2009). MREIs can act as fish aggregation devices (Broadhurst *et al.*, 2014), and enhance conservation measures by creating *de facto* MPAs (Inger *et al.*, 2009; Sheehan *et al.*, 2013b). However, the relative success of MREIs acting as artificial reefs may be dependent on the receiving environment, since areas of existing low diversity and minimal habitat structure stand to gain the most from the introduction of new hard substrate (Inger *et al.*, 2009). Furthermore, community composition on artificial structures does not necessarily reflect communities found in adjacent natural habitats (Evans *et al.*, 2015), with the former often supporting lower diversity (Pister, 2009; Firth *et al.*, 2013) . In some instances, increased provision of refuges for predators can increase predation, negatively impact the surrounding prey species (Herrera *et al.*, 2002).

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

One aspect of MREIs that has received less attention is the impact of decommissioning on the species and habitats that have formed over the operational life cycle of these structures. As MREIs approach decommissioning, after 20-30 years for offshore wind turbines (Smyth *et al.*, 2015), the question arises as to what course of action to take in regards to how these structures can further be of use, if at all. Due to the renewables sector still being a relatively infant industry (Copping, 2018), much of the guidance on decommissioning comes from offshore oil and gas installations for which there is little international consensus (Macreadie *et al.*, 2011). The international treaties that govern the decommissioning of offshore oilrig platforms specify that full removal of these structures must take place unless they can fulfil another valid purpose, such as the creation of artificial reef habitat (Osmundsen and Tveterås, 2003). The decommissioning of MREIs in the UK falls under the remit of the Department of Energy & Climate Change's 2004 Energy Act. This stipulates full decommissioning, with some exemptions such as if the installation can serve a new use by enhancing a living resource or if entire removal would involve an unacceptable risk to the marine environment (DECC, 2004).

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

METHODS

Study site and design

Wave Hub is an 8 km^2 MREI located off the north coast of Cornwall, south west UK. A 25 km subsea power cable, 160 mm in diameter, connects a land-based electricity substation to the hub, an electrical junction box (Figure 1). The seabed infrastructure was deployed in 2010, with a life expectancy of 25 years (West *et al.*, 2009; Sheehan *et al.*, 2013b). From the shore, the first 7 km of cable crosses sandy seabed and is buried. The remaining cable crosses rocky reef with inter-reef sediments, reaching a maximum depth between 50 and 60 metres. It is covered in rock armouring, at a minimum burial depth of 300 mm, with concrete mattressing at 120 m intervals to provide additional stabilisation as the substrate is not suitable for trenching. Overall, 80,000 tonnes of rock was deployed on the seabed.

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

Benthic video survey

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

Video analysis

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

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

Response metrics

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

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

Environmental context

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

Data analysis

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

Number of Taxa was analysed without prior transformation, while all other response metrics were fourth root transformed. Assemblage composition was based on Bray Curtis similarity (Bray and Curtis, 1957) while all univariate response metrics were based on Euclidean distance (Anderson and Millar, 2004). Significant differences between Year x Treatment or just Treatment were further investigated using pairwise tests and for the multivariate data, visualized using nonmetric multi-dimensional scaling (nMDS). Species driving the nMDS were examined using SIMPER (similarity percentages) (Clarke and Warwick, 2001) and the level of dispersion between sites for each year and treatment combination was calculated using MVDISP (Clarke and Gorley, 2006).

RESULTS

Wave Hub site

The total area of the cable installation is approximately 0.036 km^2, and was installed predominantly upon circalittoral rock and biogenic reef, a habitat which covers 395 km^2 of the 914 km^2 bioregion (Figure 3), while 338 km^2 of the bioregion is circalittoral coarse sediment. To the northeast of the bioregion is an extensive area of circalittoral coarse sediment, with occasional circalittoral rock and biogenic reef.

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

Assemblage composition, Taxa and Abundance

Overall, 80 taxa from 11 phyla were identified. However, these were not uniformly distributed across treatments. The epibenthic Assemblage composition on the cable was distinct from that of the control sites throughout the period of the study, but was most dissimilar in 2012. The Assemblage composition dispersion index within cable plots was lowest in the first survey year (2012) and became more dispersed and more similar to the controls over time (Table S1). Despite a Year x Treatment interaction ($p < 0.0001$) (Figure 4a, Table 1), suggesting that the level of difference varied over time between treatments, pairwise tests showed that the cable was significantly different to both controls (E and W) in all three years (2012, 2014 and 2015), while the east and west controls did not differ significantly from each other (Table 1). 29 taxa accounted for 90% of the variation in the assemblages on the cable and in the controls across the years. The main taxa driving these differences were predominantly *Alcyonidium diaphanum*, *Cellepora pumicosa*, *Nemertesia antennina,* Hydroids spp. and Turf (SIMPER, Table S2). Hydroids spp., Turf and the bryozoans *Alcyonidium diaphanum* and *Cellepora pumicosa* were all more abundant on the cable than in the controls in 2012, collectively contributing 60.5 % of the variation in Assemblage composition. However, by 2014 they were all more abundant in the controls, apart from Turf, which remained more abundant on the cable. In 2015, the group Hydroids spp. were more abundant on the cable than the controls, while the hydroids *Nemertesia ramosa, Nemertesia antennia* and *Halecium halecinum* were still in greater abundance in the controls. In 2012, 2 years after deployment, the bryozoan *Pentapora foliacea* was not found on the cable, only in the controls. However, in 2014 and 2015 this species was more abundant on the cable than the controls (Table S2). Encrusting sponges were also more abundant on the cable than the controls for all years, while branching sponges and species such as *Sycon ciliatum* and *Polymastia boletiformis* were found to be more abundant in the controls in 2012 and 2014.

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

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

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

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

Morphotaxa

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

Porifera exhibited the first colonisation pattern, being significantly more abundant on the cable than the controls ($p < 0.001$; Cable = 1.67 ± 0.09 SE ind. m^{-2}, Control E = 1.33 ± 0.08 SE ind. m^{-2}, Control W = 1.19 ± 0.09 SE ind. m^{-2}) (Figure 5a; Table S3), and while the abundance of cable Porifera appears to be converging with controls, there was no significant effect of Year x Treatment.

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

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

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

DISCUSSION

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

Environmental context

The cable was laid into a 395 km^2 area of circalittoral rock and biogenic reef (Figure 3) within a wider bioregion of 914 km^2. The footprint of the cable infrastructure on the seabed is approximately 0.036 km^2, or 0.009 % of this habitat area, but the overall habitat space added to the system is likely to be orders of magnitude greater due to the surface complexity of individual boulders and the installation as a whole. Small scale habitat heterogeneity, such as the rugosity and surface roughness of introduced substrate, increases habitat complexity and can promote species diversity (Luckhurst and Luckhurst, 1978; Firth *et al.*, 2013) by providing a greater number of physical niches for organisms to inhabit (Hixon and Beets, 1989; Langhamer and Wilhelmsson, 2009). Decommissioning plans should acknowledge the relative contributions of each MREI component to the habitat, alongside the spatial scale of the installation (Wilding *et al.*, 2017).

Patterns of succession

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

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

The process of succession is slow and continuous and susceptible to local natural disturbances (Antoniadou *et al.*, 2009). The 2014 surveys took place after a winter of severe storms (Masselink *et al.*, 2016), which were responsible for the shifts in abundance of benthic organisms in other areas of south west England (Sheehan *et al.,* unpublished data). The effects of these storms may still have been present during the 2014 survey at the Wave Hub site and consequently affected the abundance of organisms recorded that year.

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

Early sessile colonisers such as Hydroids spp., which were more abundant on the cable than in controls in 2012 (Table S2), also provide biogenic structure (Turner *et al.*, 1999) that increase the net three-dimensional habitat available for mobile species (Ferrari *et al.*, 2016). The structurally complex bryozoan *Pentapora foliacea* was not found on the cable route two years after deployment but by the fourth year was found across all treatments. This species is typically slow growing and long lived (Jackson *et al.*, 2008) and its presence on the cable rock armouring may indicate a trajectory towards a more structurally complex habitat, that will in turn provide habitat for other species (Ferrari *et al.*, 2016). However, there was no Treatment effect for Vertebrata indicating that the cable route was not having an artificial reef effect within five years, and monitoring over longer timescales would be needed to test this effect for mobile species.

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

Monitoring approach

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

It is also important to take any historical and present anthropogenic activity into account when determining suitable control sites, as shifting baselines in the local environment and varying recovery rates for the different response metrics can confound assessment of assemblage development. This case study focused on the patterns of succession on the cable rock armouring, by comparing response metrics to those in the surrounding environment of the nearby control sites. The sea bed at the Wave Hub site has been subjected to historical fishing pressure, predominantly from static gear (Campbell *et al.*, 2014), likely due to the rocky nature of the habitat being incompatible with most bottom towed fishing methods. Therefore it is likely that these control sites would have been exposed to past disturbance. Static fishing gear was observed on the cable itself whilst survey work was ongoing (Sheehan, personal observation), hence any comparison of the cable is to the existing state of the surrounding habitat, rather than a pristine environment. However, control sites were considered to be suitable for comparing the development of the community on the cable, as the local diversity appeared well developed, with a diverse range of upright biogenic habitat forming organisms; typically not present in areas that are subjected to bottom towed gear (Bradshaw *et al.*, 2003; Sheehan *et al.*, 2013a).

Impact of decommissioning

During the lifetime of the Wave Hub cable installation, as with other MREIs, there is a potential net habitat gain (Smyth *et al.*, 2015). Five years after deployment, the cable is supporting an epibenthic community which is becoming congruent with the surrounding ecosystem. The cable is not intended to be removed until 2037, after which time the rock armouring will have been in place for 27 years and it is likely that any colonisation will be advanced and productive after this time (Smyth *et al.*, 2015). Complete decommissioning would effectively be removing this habitat and any associated ecosystem services that it provides. Long lived species and complex habitats may take decades to recover (Duarte *et al.*, 2015), so only long term monitoring will provide conclusions as to how recovered the ecosystem can really be expected to become. However, it must be borne in mind that the ecosystem will be returning to a pre-cable, rather than a pristine state, acknowledging anthropogenic activities such as fishing pressure in the region.

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

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

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

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

Conclusions

Although thorough decommissioning planning is part of the consenting process for all MREIs, there is little evidence as to what environmental impacts might occur. Case studies such as this can help marine managers plan for an installation and decommissioning programme that will least disturb any habitat development over the useful lifetime of MREIs, maximising the ecological potential of installations. Using appropriate materials for the receiving habitat will preserve ecosystem function and associated services as well as protect marine habitats. Our results show that recovery on the cable, which lies in a temperate reef ecosystem, is relatively fast compared to the operational life of these structures, which are typically in the marine environment for 20 – 30 years. Initial findings from studying the assemblage development up to five years after deployment of a subsea cable, indicate that the disturbed site is showing signs of recovery and is becoming more comparable to the surrounding environment. Therefore, a partial decommissioning strategy (over full removal) for this site and similar installations would be supported. However, longer term monitoring would be required to assess whether recovery trajectories continue as expected and determine the extent of the consequences that decommissioning might have. Choosing appropriate materials and methods when designing MREIs could allow such installations to contribute to the wider national conservation aim of linking a coherent network of marine protected areas around the UK, and allow for the continued expansion of existing reef-like habitats. If left in the marine environment, these structures have the potential to restore degraded habitats and could make a further contribution to conservation measures, if the areas around MREIs were fully protected from anthropogenic pressures such as the use of bottom towed fishing gear. However, the location, substrate, type and composition of MREIs and their components make decommissioning guidelines for these devices applicable on a site-by-site basis, and a rigorous, effective and replicable monitoring approach that accounts for varying spatial and temporal scales must be employed as future installations develop to commercial scale.

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

Source	df	SS	Pseudo-F	P(perm)	Pairwise comparison	t	P(perm)	t	P(perm)	t
Assemblage					Ye x Tr	*2012*		*2014*		*2015*
Year (Ye)	2	2.24×10^4	9.54	**0.0001**	Control W, Control E	0.831	0.7212	0.729	0.8262	0.799
Treatment (Tr)	2	1.77×10^4	7.55	**0.0001**	Control W, Cable Route	3.07	**0.0001**	1.90	**0.0028**	2.19
Site (Si)	8	5.60×10^4	5.96	**0.0001**	Control E, Cable Route	3.20	**0.0001**	1.73	**0.0082**	2.63
Ye x Tr	4	1.03×10^4	2.18	**0.0001**						
Ye x Si	11	2.56×10^4	1.98	**0.0001**						
Tr x Si	16	2.75×10^4	1.46	**0.0004**						
Ye x Tr x Si	19	2.40×10^4	1.08	0.2611						
Residual	101	1.19×10^5								
Total	163	3.02×10^5								
Number of taxa					Tr					
Ye	2	91.178	4.0628	0.0216	Control W, Control E	0.51168	0.6106			
Tr	2	305.14	13.597	**0.0001**	Control W, Cable Route	4.1762	0.0005			
Si	8	288.36	3.2122	**0.0038**	Control E, Cable Route	4.8883	0.0001			
Ye x Tr	4	9.0166	0.20088	0.9396						
Ye x Si**	11	36.893	0.29889	0.9821						
Tr x Si	16	355.88	1.9822	**0.0206**						
Ye x Tr x Si**	19	209.39	0.98214	0.4792						
Res	101	1133.3								
Total	163	2429.2								
Overall abundance										
Ye	2	4.13	3.31	**0.0401**						
Tr	2	1.80	1.45	0.2428						
Si	8	27.4	5.49	**0.0001**						
Ye x Tr	4	3.42	1.37	0.2588						
Ye x Si**	11	19.9	2.90	**0.003**						
Tr x Si	16	19.1	1.92	**0.0313**						
Ye x Tr x Si**	19	11.1	0.936	0.5316						
Res	101	63.0								
Total	163	150								

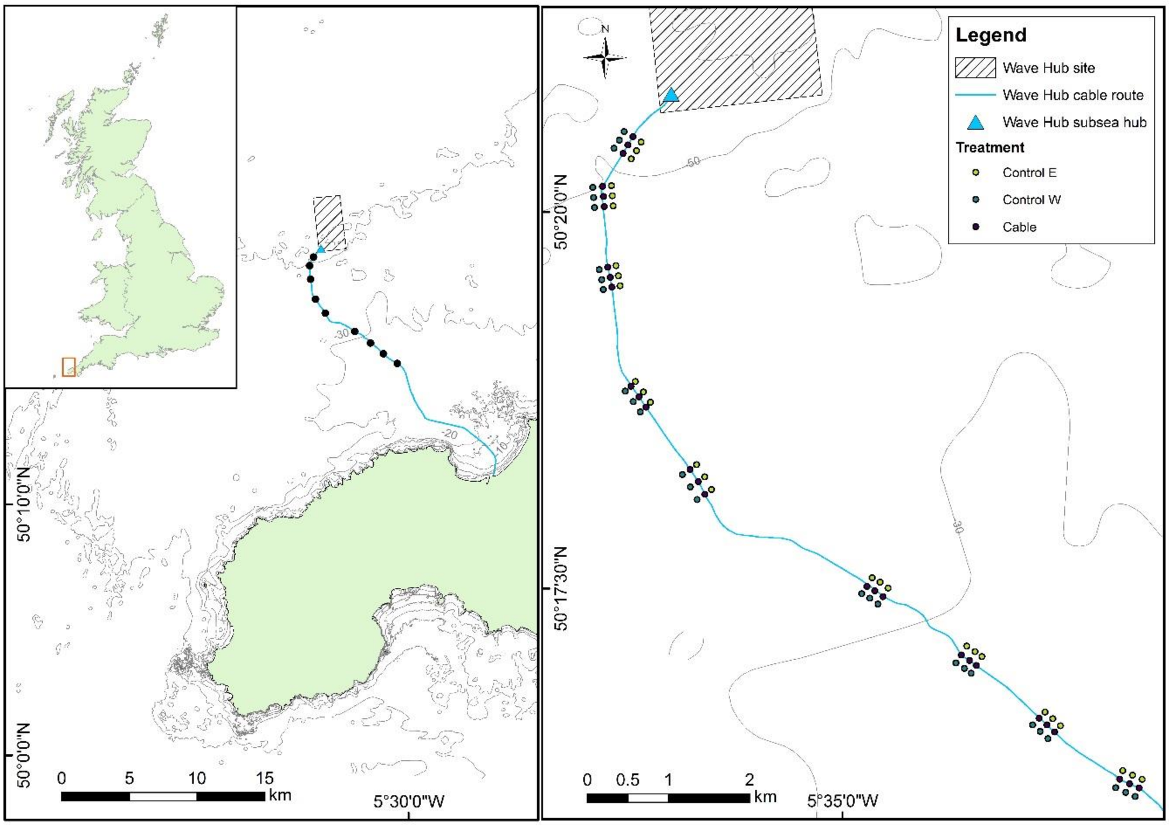


Figure 1: Site map of the Wave Hub site in the southwest of the UK, with detail of survey design along submarine cable route, with controls to the east and west.

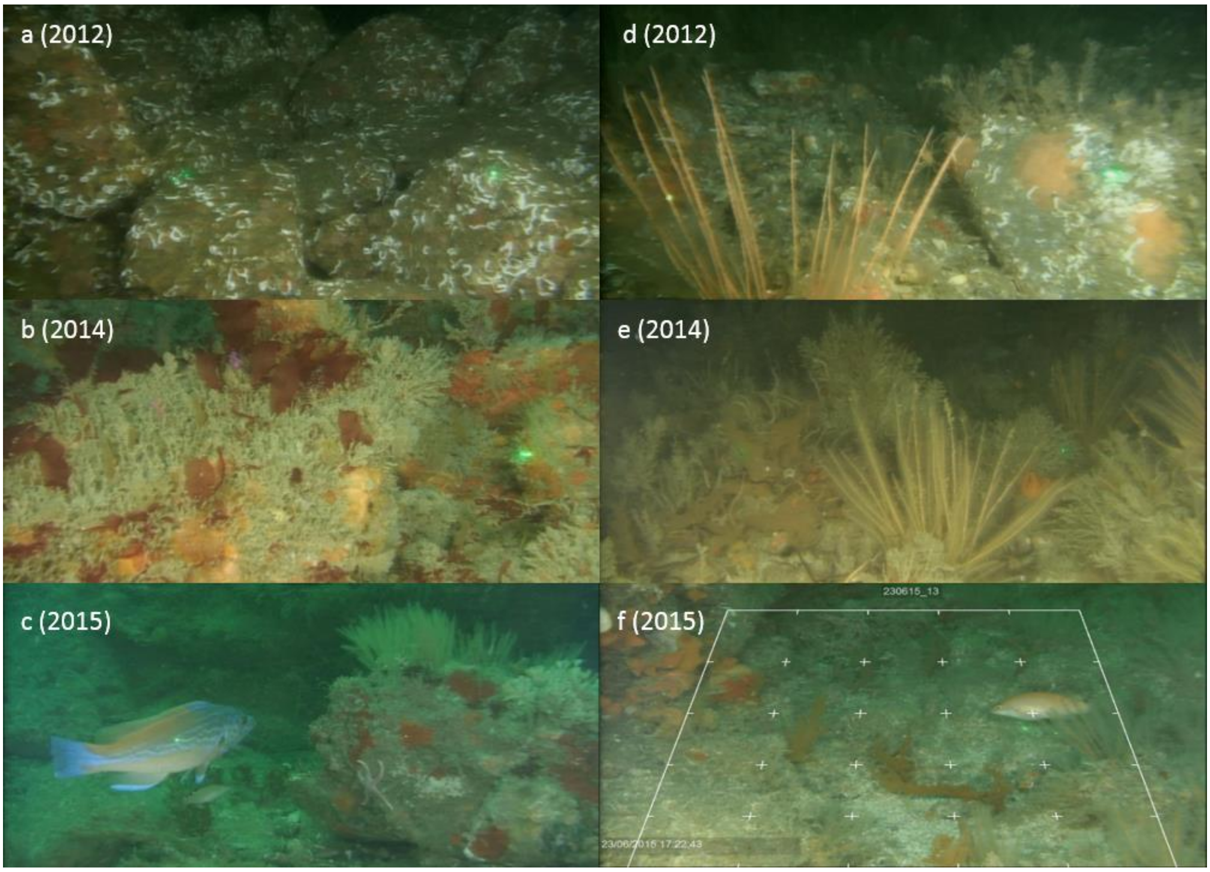


Figure 2: Example images from transect footage from cable rock armouring (a, b, c) and control sites adjacent to the cable route (d, e, f). Image f also depicts the digital quadrat overlay that was applied to each frame grab during the video analysis. Laser spacing (green dots) = 0.3 m.

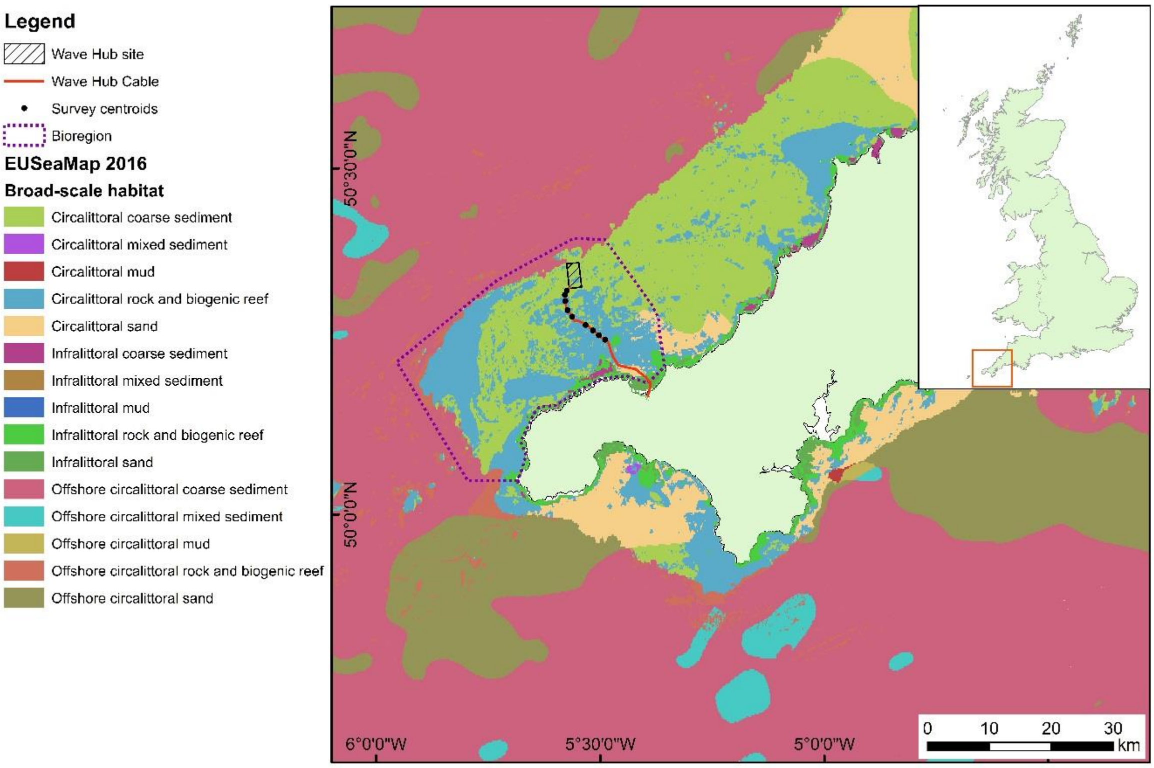


Figure 3: Broad scale habitat of the southwest coastal region of the UK (EMODnet, 2016). The bioregion is located on the northwest-facing coastline, extending out to the 50 metre contour. The eastern and western boundaries are located at the edge of the contiguous circalittoral rock and biogenic reef.

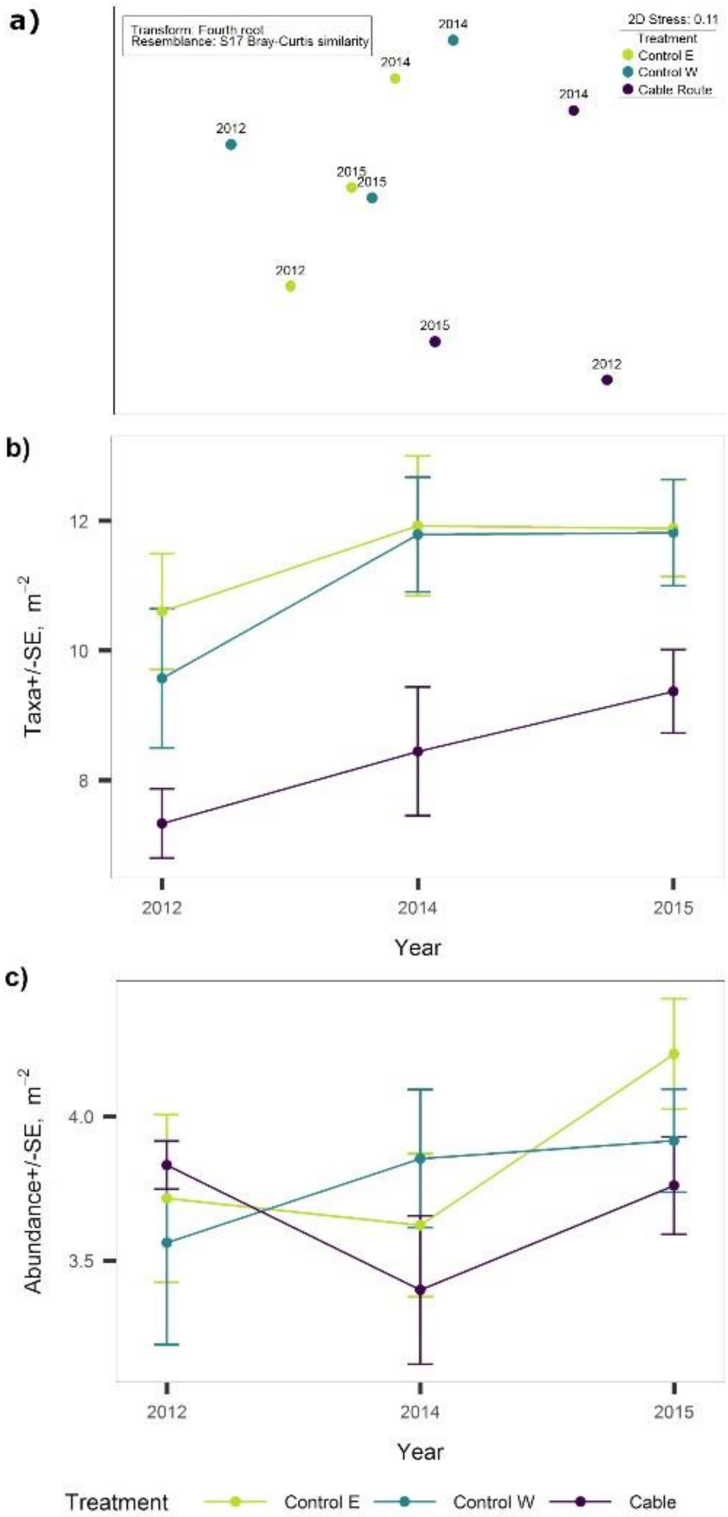
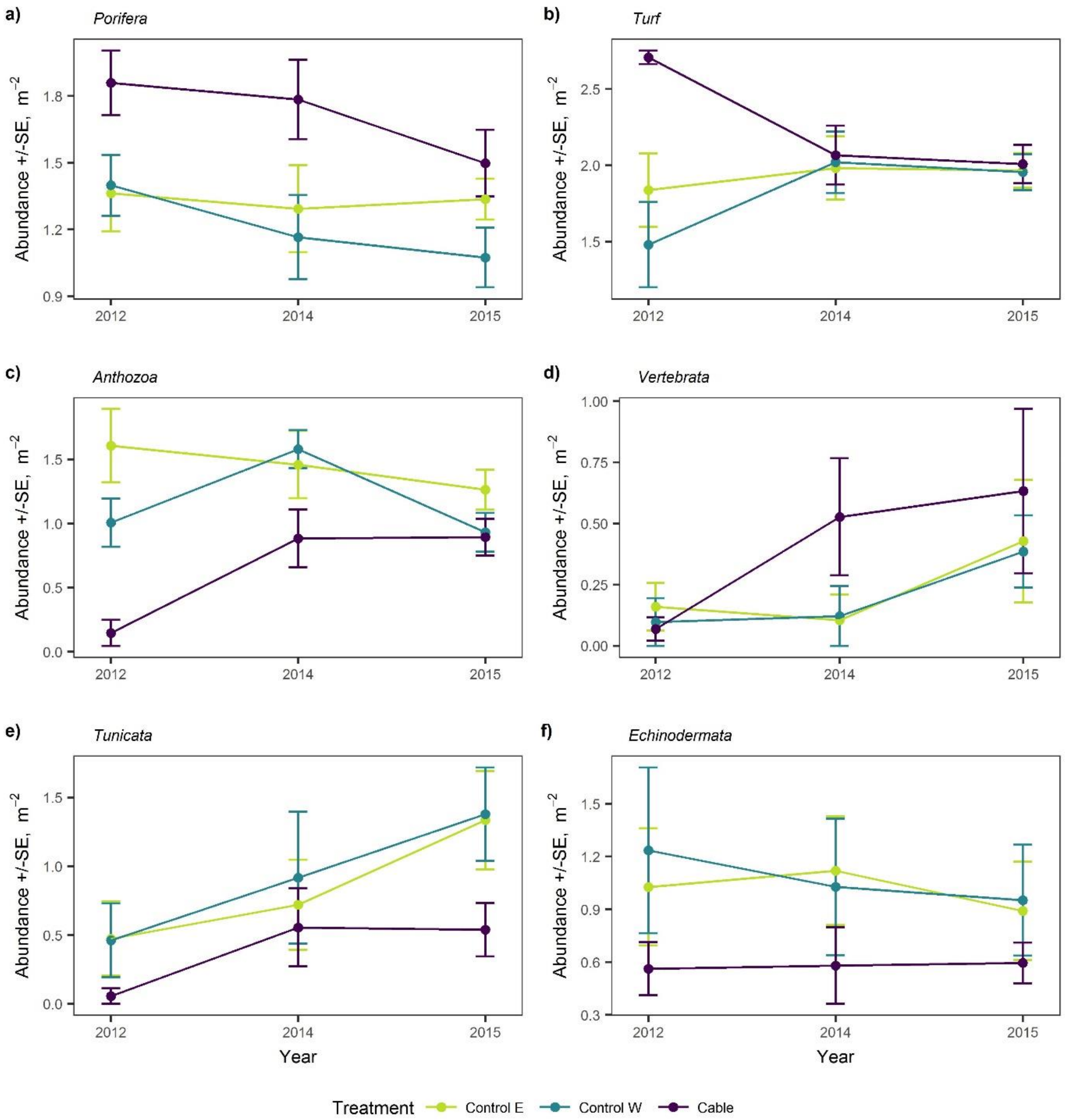


Figure 4: a) nMDS of benthic assemblage across the three treatments at the Wave Hub site over the period 2012 to 2015. Each point represents the Assemblage composition across all sites within each treatment in each year; b) Number of taxa; c) Overall abundance. Assemblage and Abundance data fourth-root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



a)
Porifera
b)
Turf
c)
Anthozoa
d)
Vertebrata
e)
Tunicata
f)
Echinodermata
Abundance +/-SE, m^{-2}
2012
2014
2015
Year
Treatment
Control E
Control W
Cable

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