Blue Marine Foundation

Using Baited Remote Underwater Video (BRUV) to assess species biodiversity in Berwickshire's Marine Protected Areas (MPAs) and the surrounding area

November 2022

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Introduction

Berwickshire, on the southeast coast of Scotland, provides some of the most biodiverse and productive seas in the UK. The coastline is known for its rocky reefs, kelp forests, soft corals, anemones and its association with marine mammals such as bottlenose dolphins, minke whales and breeding grey seals. The productive brown crab (*Cancer pagurus*) and European lobster (*Homarus gammarus*) fisheries within this area also support many local communities that depend on this industry for their livelihoods. Several designations have been established in Berwickshire to protect this biologically diverse marine environment. These designations are:

- **Berwickshire and North Northumberland Special Area of Conservation (SAC) -** Covering an area of 652 km^2, from Alnmouth in the south to Fast Castle Head in the north, this Marine Protected Area (MPA) protects rocky reefs, sea caves, and grey seals in the Berwickshire region, as listed in the EU Habitats Directive[1]. Other features within the SAC include mudflats and sandflats not covered by seawater at low tide and shallow inlets and bays.
- **The St Abbs and Eyemouth Static Gear Reserve (SGR)** - The Reserve covers 26 km^2 and extends one nautical mile offshore from St Abbs Head in the north to the Scotland - England Border in the south. The Reserve prohibits bottom trawling[2] and is protected under the Inshore Fishing Order 2004[3].
- **The Berwickshire Marine Reserve (BMR) -** Formally known as the St Abbs and Eyemouth Voluntary Marine Reserve, the BMR was established in 1984. It is a voluntary designation, managed locally, with the aim of conserving biodiversity by promoting responsible use and sustainable fisheries. It covers 10.3 km^2 from Pettico Wick Bay in the north to Eyemouth Fort in the south[4].

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[1] JNCC. (2022). Berwickshire and Northumberland Coast. Available at: https://sac.jncc.gov.uk/site/UK0017072
[2] Bottom trawling is defined in this report as 'weighted fishing gear that is dragged along the seabed (including nets and dredges)
[3] Legislation.gov.uk. (2022). The Inshore Fishing (Prohibition of Fishing and Fishing Methods) (Scotland) Order 2004. Available at: http://www.legislation.gov.uk/ssi/2004/276/article/5/made
[4] Marine Scotland. (2022). St Abbs and Eyemouth Voluntary Marine Reserve (VMR). Available at: http://marine.gov.scot/information/st-abbs-and-eyemouth-voluntary-marine-reserve-vmr

The Berwickshire Marine Reserve (BMR) sits inside the Static Gear Reserve (SGR), and the BMR and SGR sit within the Northumberland and Berwickshire Special Area of Conservation (SAC) (Figure 1). The SAC and SGR are statutory designations that contribute to the Scotland Marine Protected Area (MPA) network.

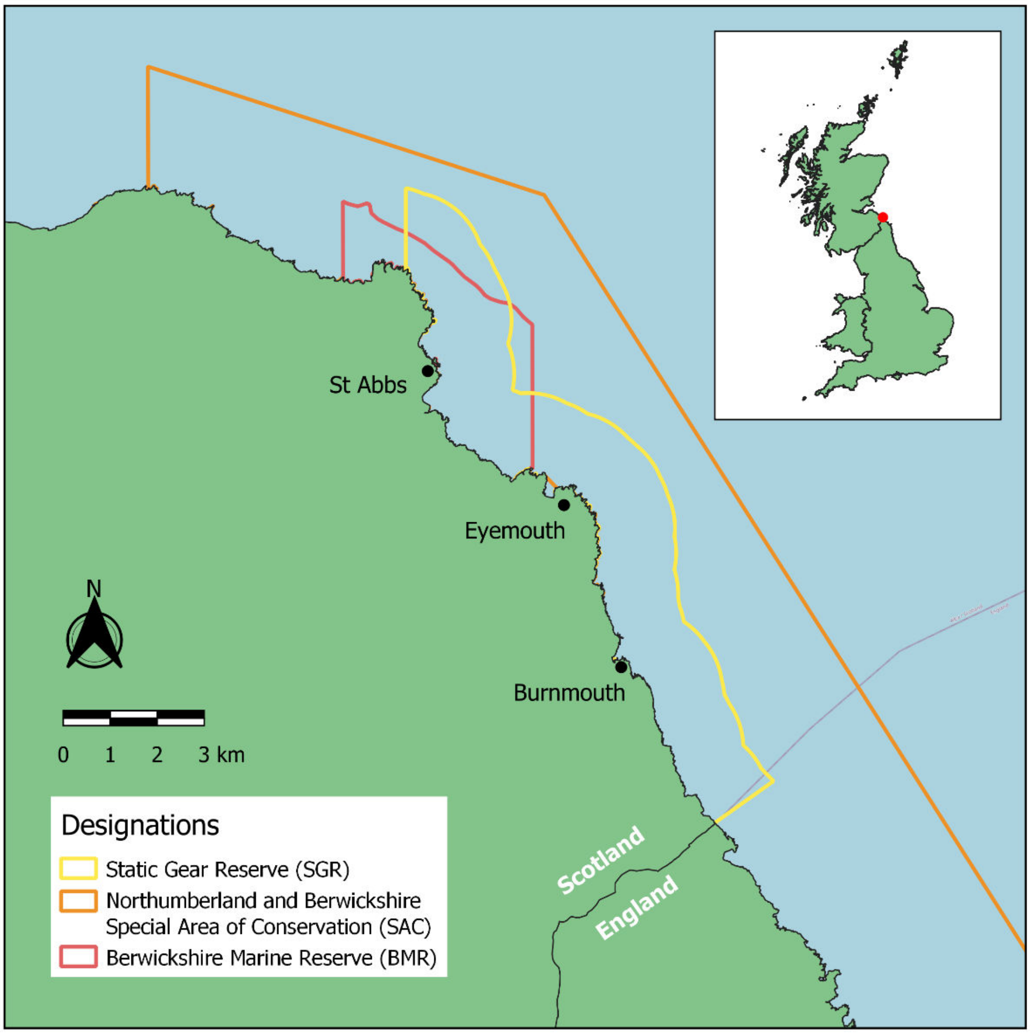

Figure 1: MPA designations in Berwickshire and the surrounding area.

Scotland's coastline equates to 10 % of Europe's coastline and its territorial waters account for around 63 % of the UK's as a whole. The Scottish Marine Act 2010 committed to not only protect but enhance the health of Scotland's seas. Despite this commitment, we have seen significant degradation of Scotland's marine ecosystems, habitats and fisheries. The decadal Scottish Marine Assessment released in 2020 (SMA 2020) revealed that there has been very little change since the 2010 assessment:

- 46 % of the fishing stocks surveyed remain overfished. This includes commercially important species such as mackerel, haddock, and cod[Error! Bookmark not defined.]. Many stocks are yet to be assessed or have catch limits put in place, for example scallops and all wrasse species.
- Important marine habitats, such as biogenic reefs, maerl and seagrass beds, have declined across all of Scotland's waters[Error! Bookmark not defined.]. Argyll has lost 53 % of its unique flame shell reefs and the Moray Firth has lost almost all of its blue mussel beds (99.5 %).
- Over half of the 31 Scottish MPAs assessed have no fisheries management measures. Seafloor habitats are predicted to be in poor condition across more than half of their area in nine out of 21 regions across Scotland. Some level of damage was found to be likely in all regions, caused by widespread bottom trawling fishing practices[Error! Bookmark not defined.].

The SMA 2020 assessment concluded that pressures associated with fishing have the biggest impact on the marine environment throughout Scotland. The assessment also stated that bottom trawling was widespread across most Scottish Marine Regions, with over 95 % of Scotland's inshore waters (0 -12nm) subject to this gear type. Reports of unlawful fishing are also common in the areas that are closed to bottom trawling practices.

Berwickshire is part of Blue Marine's fisheries and conservation partnership network in the UK. Blue Marine is applying a whole-site, ecosystem-based approach to management, working with low impact fishing communities and other stakeholders to deliver MPAs that work for people, nature and climate. As part of Blue Marine's aim to secure well protected and sustainably managed MPAs in Scotland, research programmes have been developed to better understand the marine environment and its users and determine whether the current statutory designations are effective for the recovery of biodiversity and fisheries. To do this, wide-scale ecological monitoring of mobile fish, crustaceans and invertebrates is necessary to give an indication of ecosystem health and is an important way of examining the direct and indirect effects of recreational and commercial activities on species communities[5] [6].

The research presented here used Baited Remote Underwater Videos (BRUVs), deployed in August 2022, to assess benthic and reef-associated nekton inside and outside of Berwickshire's MPAs and to spatially compare between the 'treatments' of the SAC, SGR, and the Open Control (OC) sites situated outside the designated zones. This data can provide evidence for further protection of marine species and habitats, as well as help to develop and promote sustainable fishing practices with relevant stakeholders. Abundance, diversity and assemblage composition data can help to establish an ecological baseline and be used to compare to future surveys to monitor changes over time. The results from this study will contribute towards a wider ecological assessment of Berwickshire's species and habitats adding to previous research including:

- Towed video to assess the abundance of sessile and sedentary benthic species and related habitat.
- Potting surveys to assess commercial crustacean species recovery.

Methods

To assess reef-associated nekton in Berwickshire, the response variables of abundance (total number of each species), species richness (number of species) and assemblages were compared between 'treatments'. The treatments are defined as:

- **Static Gear Reserve (SGR)** = sites inside the area closed to bottom trawling. These sites also fall within the SAC and two sites fall inside the Berwickshire Marine Reserve (BMR)
- **Special Area of Conservation (SAC)** = sites inside the Northumberland and Berwickshire Special Area of Conservation only

[5] Link, J. (2002). 'What does ecosystem-based fisheries management mean?', Fisheries, 27:18–21.
[6] Langlois, T., Harvey, E. and Meeuwig, J. (2022). 'Strong direct and inconsistent indirect effects of fishing found using stereo-video: Testing indicators from fisheries closures', Ecological Indicators, 23:524-534.

- **Open Controls (OC)** = control sites outside of both these designations which remain open to all fishing practices

2022 was the first year this survey was carried out.

Baited Remote Underwater Videos (BRUVs)

Baited Remote Underwater Videos were deployed in August 2022. The deployment of multiple static camera units increased sampling efficiency and statistical independence. Three BRUV units consisted of a 1.5 m pole with a bait canister on the end filled with 100 g of fresh mackerel (Figure 2). Bait was discarded and replaced after each deployment. Each unit hosted a battery-operated LED light and a Zerodis dive light (18000LM) secured to the frame. The unit also hosted three GoPro cameras; two cameras faced the bait cage and one camera faced backwards to assess the habitat. Only the front-facing right-hand camera was used in the analysis. At each site, the three BRUV units were deployed from the side of the boat with a marker buoy attached and placed 100 m away from each other to ensure samples were independent. Each unit was left to 'soak' for 1 hour before being brought back in. All BRUVs were undertaken across a 4-day period. At the deeper sites (70 m) (Appendix F Table 10), units were deployed around slack water to prevent the frame from tipping over and to improve visibility.

Figure 2: a) three Baited Remote Underwater video units; b) poles and bait canister

Sampling design

Survey sites were identified and selected pre-survey using local knowledge and geophysical survey data collected in 2018/19. All sites were of hard substrata, defined in the St Andrews geophysical survey report[7] as:

- 'Rocky reef'
- 'Hard base with sediment veneer'
- 'Exposed bedrock'
- 'Areas of boulders'

The sites were spatially replicated to allow for spatial variation within each treatment. Failed BRUVs were removed from the analysis, with the study totalling 10 sites and 30 replicates (Figure 3).

These sites are comparable to the March 2022 Berwickshire ecological monitoring report[8]. This research included Underwater Video, Towed Video and Baited Remote Underwater Video to set an ecological baseline for the area. The March BRUV data has not been compared to in this report due to the survey taking place at a different time of year and minimal species having been observed. The data from this August survey can, therefore, be added to the wider ecological data set.

[7] Bates (2019). Geophysical Survey at St Abbs Head. University of St Andrews. Unpublished report.
[8] Rees, A.G., Sheehan, E.V. (2022). Developing ecological long-term monitoring of the Berwickshire Marine Reserve and surrounding area. March 2022. Report to Blue Marine Foundation from University of Plymouth. DRAFT

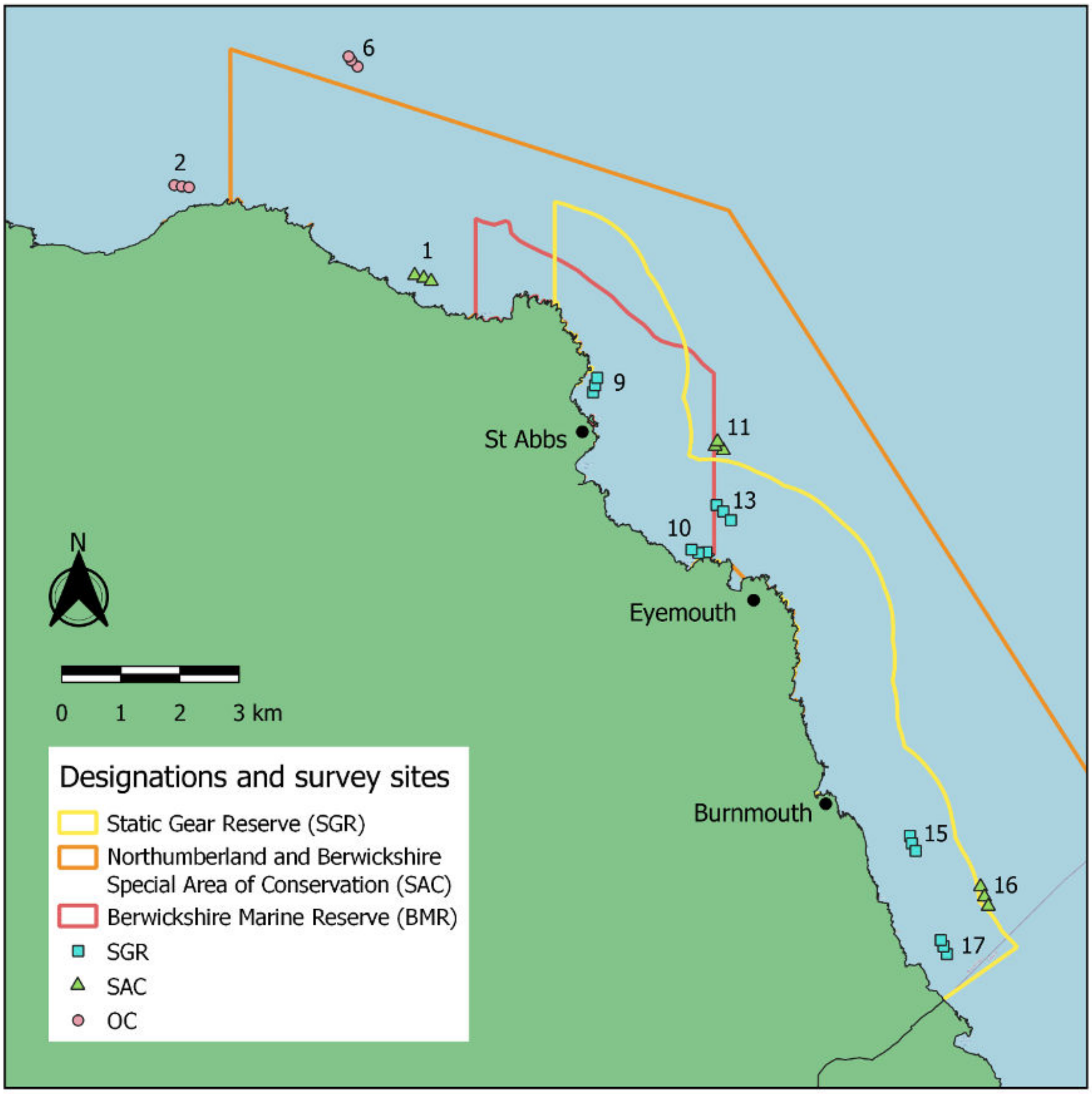

Figure 3: Baited Remote Underwater Video survey sites inside and outside Berwickshire's MPAs. Appendix E Figure 9 shows site 16 on a zoomed in map for clarity of the treatment.

Video analysis

Quantitative data was extracted from the BRUV footage by counting the maximum number of individuals on screen (MaxN) for each species that was recorded for a 45-minute period. MaxN is a conservative estimate of relative abundance of mobile species, due to decreasing the chance of an individual being counted more than once[9]. Organisms were identified to

[9] Cappo, M., Harvey, E. and Shortis, M. (2006). 'Counting and measuring fish with baited video techniques-an overview', in Australian Society for Fish Biology Workshop Proceedings. Tasmania: Australian Society for Fish Biology, 101-114.

the highest taxonomic level possible (Appendix A Table 2) and the counts were expressed as MaxN per species, per replicate. Examples of video footage are shown in Figure 4.

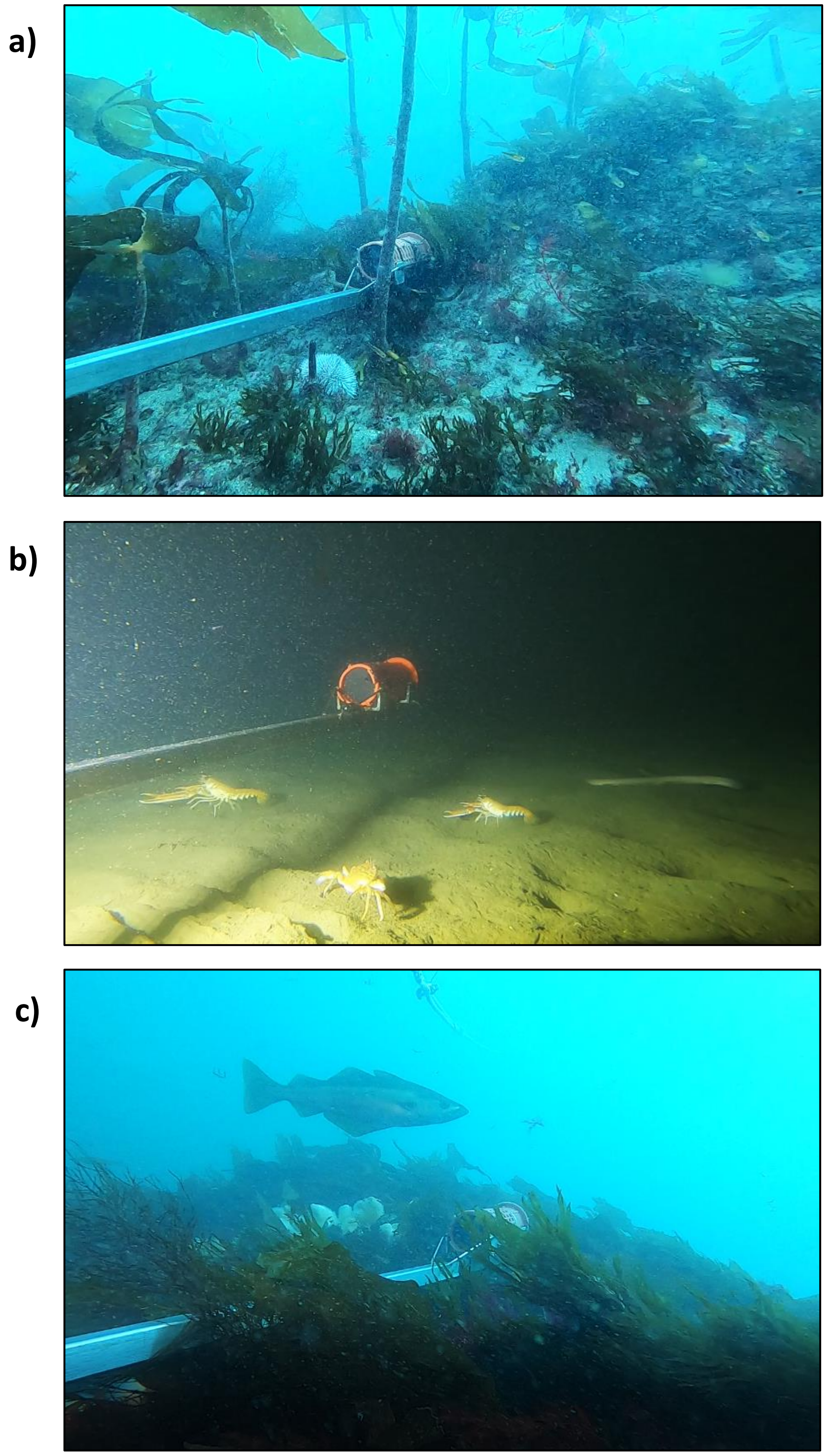

Figure 4: a) Shoal of two-spot gobies, b) nephrops, harbour crab and hagfish; c) adult pollack

Some groupings of organisms occurred due to between-species similarities:

- The brittlestar species *Ophiocomina nigra*, *Ophiothrix fragilis* and *Ophiura ophiura* were grouped and named as "Grouped brittlestars";
- *Pagarus bernhardus* and *Pagarus prideaux* were grouped as "*Pagurus* spp.";
- Indistinguishable goby species were grouped as "*Gobiidae* spp.";
- Indistinguishable wrasse species were grouped as "*Labridae* spp." And "*Juv. Labridae* spp.";
- Indistinguishable swimming crab species were grouped as "*Liocarcinus* spp.";
- Species that were not able to be identified or distinguished were assigned to a descriptive group that described their morphology, e.g. ray-finned bony fishes were grouped as "*Actinopterygii* spp."; juvenile fish were grouped as "*Juv. Actinopterygii* spp.*"* Juvenile fish were identified either by their size or identifiable features.

Data analysis

The response variables tested between treatments were Relative Abundance (Mean MaxN) and Species Richness (Mean). Mean MaxN and Richness values were used for analysis as counts were averaged across the replicates. The independent variable was Treatment which had three categories (SGR, SAC, OC). Each Treatment included a number of sites (SGR = 5, SAC = 3, OC = 2) with the three averaged replicates in each site. Kruskal-Wallis ANOVA tests were used in SPSS, followed by the comparison of mean ranks, to determine if there were significant differences in Relative Abundance and Species Richness between Treatments. Kruskal-Wallis tests are appropriate when analysing unbalanced sampling designs.

To further understand the test results, the percentage contribution of each species (SIMPER) was calculated to identify which species contributed to 80 % of the species in each treatment.

Species were also categorised into the pre-determined functional groups using information from the literature and expert knowledge, shown in the species list in Appendix A. The groups included predatory species and scavengers, to understand further how the functional composition of species communities can change following MPA designation, regarding their feeding behaviour.

Unconstrained Correspondence Analysis (CA) was used in Canoco 5 to examine assemblage composition and associations of species between treatments. Counts were log transformed [Y' = 100(y + 1)] to minimise the influence of extremely high or low values. Indicator taxa of juvenile fish and wrasse species (*Labridae spp*.) were selected to aid the identification of potential nursery areas and complex habitat. Wrasse exhibit extremely territorial behaviour and have small home ranges, choosing refuge areas that are formed of complex ground[10].

Results

From the Baited Remote Underwater Video surveys a total of 38 species (or groups of species) were recorded across all treatments across Berwickshire (Appendix A Table 2).

Abundance

Relative Abundance (Mean MaxN) was compared between treatments using a Kruskal-Wallis test. Distributions were not similar for all treatments, as shown by the boxplot in Figure 5 and so, mean ranks were used to identify the differences. The Kruskal-Wallis ANOVA results showed abundance was higher in the SGR than the OC and the SAC (Appendix B Table 5). However, these differences were not statistically significantly different, $\chi^2(3) = .131$, $p = .937$ (Appendix B Table 3).

[10] Aasen, N. L. (2019). The movement of five wrasse species (Labridae) on the Norwegian west coast. Master's Thesis. University of Oslo. Available at: https://www.duo.uio.no/handle/10852/71994

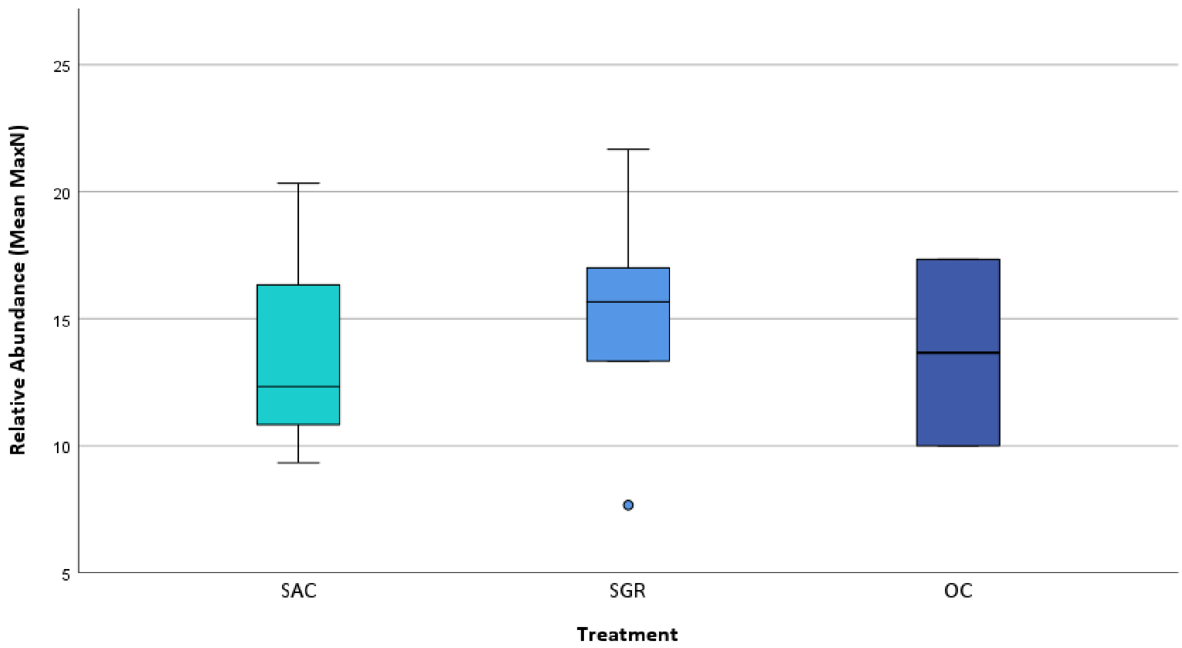

Figure 5: Distributions of Relative Abundance (Mean MaxN) between Treatments (SAC, SGR and OC) from a Kruskal-Wallis test.

Relative Abundance (Mean MaxN) of each individual species differed between treatments. The species recorded most across all treatments were squat lobster (*Galathea dispersa*), and common starfish (*Asterias rubens*). SIMPER analysis results showed that squat lobster was the principal contributor to the abundance of fish and invertebrates inside the SAC, contributing to 30 % of all species (Table 1). Common starfish was the principal contributor inside the SGR, contributing to 24 % of the species present (Table 1). Juvenile fish sp. (*Juv. Actinopterygii* spp.), two-spot goby (*Gobiusculus flavescens*) and edible sea-urchin (*Echinus esculentus*) were also the highest contributors to the total abundance inside the SGR. In the OC, no single species dominated, but juvenile whiting (*Juv. Merlangius merlangus*), harbour crab (*Liocarcinus depurator)* and squat lobster contributed to 42 % of the abundance of fish and invertebrates (Table 1).

Table 1: SIMPER analysis displaying the percentage contribution of each species to each treatment (up to 80 % of the total abundance).

SAC

Species	Relative Abundance	% Contribution	Cumulative Contribution
Squat lobster	4.44	31.75	31.75
Juvenile cod	1.89	13.49	45.24
Brown crab	1.22	8.73	53.97
Common starfish	1.11	7.94	61.90
Velvet swimming crab	1.00	7.14	69.05
Goldsinny wrasse	0.78	5.56	74.60
Dab	0.67	4.76	79.37

SGR

Species	Relative Abundance	% Contribution	Cumulative Contribution
Common starfish	3.67	24.34	24.34
Juvenile fish sp.	1.73	11.50	35.84
Two-spot goby	1.80	11.95	47.79
Edible sea urchin	1.27	8.41	56.19
Juvenile cod	0.80	5.31	61.50
Velvet swimming crab	0.67	4.42	65.93
Wrasse	0.67	4.42	70.35
Brown crab	0.47	3.10	73.45
Swimming crab sp.	0.40	2.65	76.11
Juvenile pollack	0.40	2.65	78.76

OC

Species	Relative Abundance	% Contribution	Cumulative Contribution
Juvenile whiting	2.17	15.85	15.85
Harbour crab	2.00	14.63	30.49
Squat lobster	1.67	12.20	42.68
Nephrops	1.17	8.54	51.22
Hagfish	1.17	8.54	59.76
Juvenile haddock	1.17	8.54	68.29
European lobster	0.83	6.10	74.39
Velvet swimming crab	0.83	6.10	80.49

The functional groupings (defined in Appendix A Table 2) show that the abundance of predatory species was much higher in the SAC compared to the SGR and OC (Figure 6a). The abundance of scavenging species was much higher in both the SAC and OC compared to the SGR (Figure 6b).

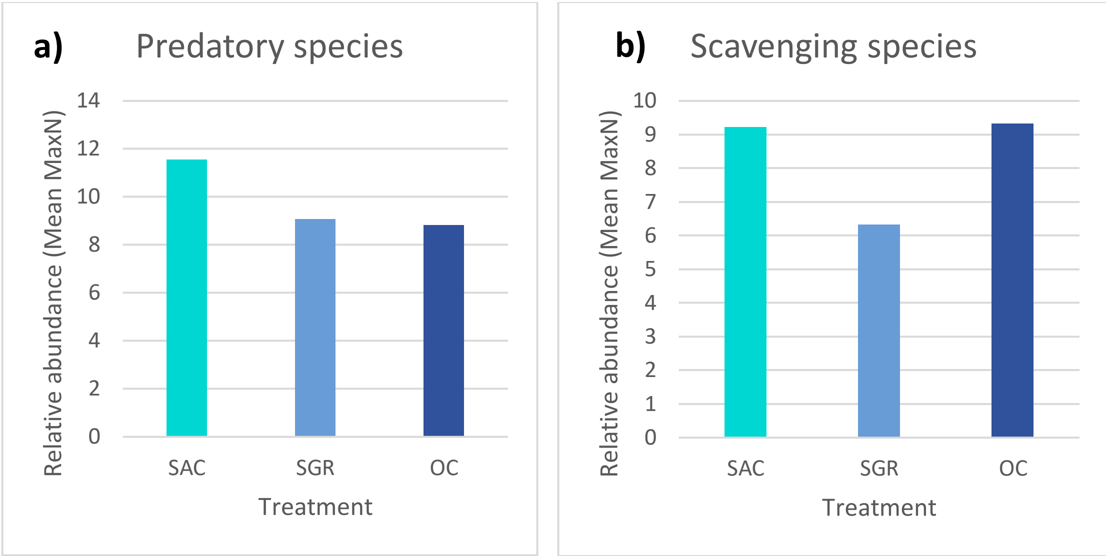

Figure 6: Relative Abundance of individuals in each treatment by species grouping a) predatory species, c) scavenging species.

Richness

Mean Species Richness was also compared between treatments using a Kruskal-Wallis ANOVA. Distributions of richness were not similar for all treatments, shown by the box plot in Figure 7, therefore, mean ranks were used to identify differences. The results of the test showed species richness was higher in the OC than in the SGR and the SAC (Appendix C Table 8), but this difference was not statistically significant, $\chi^2(3) = .206$, $p = .902$ (Appendix C Table 6).

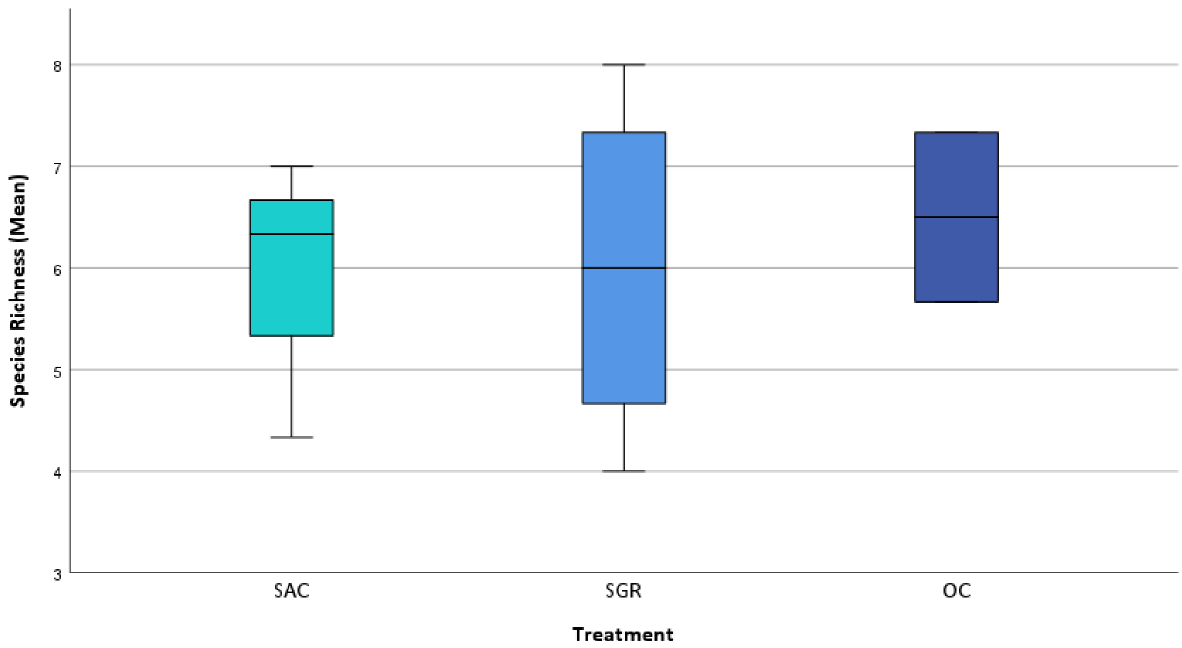

Figure 6: Distributions of Species Richness (Mean) between Treatments (SAC, SGR and OC) from a Kruskal-Wallis test.

Assemblage composition

Unconstrained Correspondence Analysis (CA) showed that there were considerable differences in species assemblages between treatments, showing a total variation of 0.64116 (Figure 8). The 1st CA axis (eigenvalue = 0.4082) determined 63.66% of the variation, and the 2nd axis (eigenvalue = 0.2330) explained 36.34% of variation (Appendix D Table 9). The SGR was highly associated with indicator species such as wrasse (*Labridae spp.*) and juveniles such as juvenile pollack (*Juv. Pollachius pollachius*) (Figure 8), although juvenile fish were present in all treatments. The SGR was also associated with adult pollack (*Pollachius pollachius*) and cod (*Gadus morhua*) but were not recorded in high enough numbers to make an accurate assessment. The OC was highly associated with species such as nephrops (*Nephrops norvegicus*) and hagfish (*Myxine glutinosa*) (Figure 8).

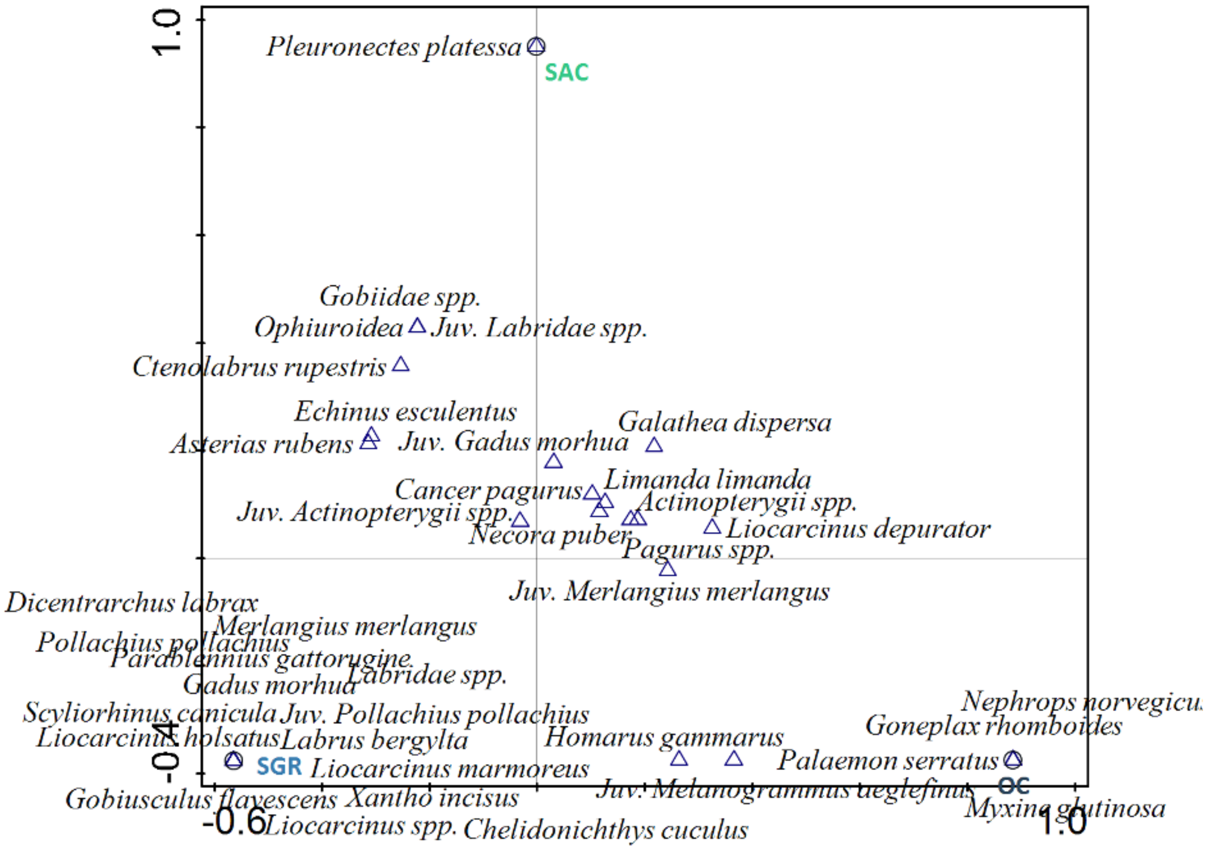

Figure 7: Ordination plot from Unconstrained Correspondence Analysis (CA) showing associations of species to Treatments (SGR, SAC, and OC).

Discussion

The Baited Remote Underwater Video surveys undertaken in August 2022 were the first summer surveys to assess reef-associated nekton across different treatments in Berwickshire. The data forms an ecological baseline which can be compared against in future years to provide a better understanding of the effect of management regimes of recreational and commercial activity on species health and recovery. The results form part of a wider comprehensive assessment of species and habitats throughout Berwickshire's MPAs, involving other methods such as towed underwater video and commercial potting surveys.

Relative abundance of species was higher in the SGR, than both the SAC and OC. This could suggest greater recovery inside the SGR and greater structural complexity of the seabed, which can create more dynamic habitats that are sought out by reef-associated nekton[11]. However, as the difference in species abundance between treatments was not significant, it could be suggested that both designations are not providing the function needed for considerable species recovery. Bottom trawling was removed from the SGR in 2004, allowing 18 years for recovery. When comparing to whole-site MPAs removing bottom trawling, such as in Lyme Bay[12] [13], it would be expected that the time since designation would be long enough to see increased abundance of sessile, sedentary and mobile species (including non-target species) through cascading trophic interactions[14] [15].

There are many potential reasons as to why the BRUV results suggest significant species recovery may not be occurring inside the designations. The effectiveness of the SGR and SAC could be impacted by increased potting levels within all treatments. Research in Lyme Bay has shown that high-intensity static gear fishing within MPAs can cause declines in commercial crustacean stocks and cause damage to benthic habitats due to abrasion from repeated hauling[16]. Further research inside Berwickshire's protected areas, beginning in 2023, will assess static fishing pressure to provide a better understanding of the impact of this activity on the MPAs. Furthermore, reports of unlawful fishing are also common in the areas that are closed to bottom trawling, which could hinder recovery inside the SGR and prevent it from achieving its conservation objectives.

The size of the SGR could also be compromising MPA effectiveness. Despite 38 species (or groups of species) occurring on the baited videos, the total number of larger adult nekton species, such as cod (*Gadus morhua*) or pollack (*Pollachius pollachius*), was low. No sharks, skates or rays were noted in any videos. These mobile species often range beyond MPA

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[11] Wilson, S. K., Fisher, R., Pratchett, M. S., Graham, N. A., Dulvy, N. K., Turner, R. A., Cakacaka, A. and Polunin, N. V. (2010). 'Habitat degradation and fishing effects on the size structure of coral reef fish communities', Ecological Applications, 20:442– 451.
[12] Sheehan, E.V., et al. (2021). Rewilding of protected areas enhances resilience of marine ecosystems to extreme climatic events. Front. Mar. Sci. 8:671427
[13] Davies, B.F.R., et al. (2022). Ecosystem benefits of adopting a whole–site approach to MPA management. Fish. Manag. Ecol. 1–16.
[14] Babcock, R. C., Shears, N. T., Alcala, A. C., Barrett, N. S. and Edgar, G. J. (2010). 'Decadal trends in marine reserves reveal differential rates of change in direct and indirect effects', Proceedings of the National Academy of Sciences, 107:18256–18261.
[15] Sheehan, E. V., Stevens, T. F., Gall, S. C., Cousens, S. L., Attrill, M. J. and Fulton, C. J. (2013b). 'Recovery of a temperate reef assemblage in a marine protected area following the exclusion of towed demersal fishing', PLoS ONE, 8:e83883.
[16] Rees, A. (2019). The Lyme Bay experimental potting study. Available at: https://pearl.plymouth.ac.uk/handle/10026.1/16833

boundaries and require much larger areas of protection for effective refuge, with some studies suggesting areas of at least 100 km^2 [17]. The small size of the SGR could also mean that high intensity fishing outside the MPA could impact inside the MPA through an 'edge effect'[18]. There have been reports of scallop dredgers dredging the SGR boundary each year. 'Fishing the line' along these boundaries can cause depletion of fish within the MPA through reducing habitat connectivity, removal of species that swim out of a protected area[19], or by changing individual movement or behaviour patterns of fish[20]. For mobile fish in particular, edge effects can extend to about 1,500 m into the MPA[18]. However, access to further effort data is necessary to understand the full impact of fishing pressure on the MPA boundaries. Furthermore, Berwickshire's MPAs may not be showing the same level of recovery as seen in whole-site MPAs such as Lyme Bay because of differences in geographical location, topography and structural complexity of habitats.

Similar to Lyme Bay, species richness was shown to be higher in areas open to bottom trawling than closed areas. To understand this further, it vital to look at changes in functional groups, individual species and their assemblages. There are many examples, such as in the North Sea, where bottom trawling has altered the functional composition of benthic communities on a sea-basin scale[21].

The greater abundances of echinoderms, such as common starfish (*Asterias rubens*) and edible sea urchin (*Echinus esculentus*) inside the SGR sites could be a sign of recovery following removal of bottom trawling disturbance[22]. However, because of a lack of ecological baselines, it is challenging to understand whether these species are representing traditional trophic cascade recovery or are a sign of ecological baseline shifts following

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[17] Chambers, P., de Gruchy, C., Morel, G., Binney, F., Jeffreys, G., Blampied, S. and McIlwee, K. (2020). 'Crossing jurisdictions: The implementation of offshore marine protected areas in an international fishery', in Marine Protected Areas. Jersey: Elsevier, 411-436.
[18] Ohayon, S., Granot, I. and Belmaker, J. (2021). A meta-analysis reveals edge effects within marine protected areas. Nat Ecol Evol 5:1301–1308.
[19] Kellner, J. B., Tetreault, I., Gaines, S. D. & Nisbet, R. M. (2007). Fishing the line near marine reserves in single and multispecies fsheries. Ecol. Appl. 17:1039–1054.
[20] Potts, J. R., Hillen, T. & Lewis, M. A. (2016). The 'edge effect' phenomenon: deriving population abundance patterns from individual animal movement decisions. Teor. Ecol. 9:233–247.
[21] Tillin, H.M., Hiddink, J.G., Jennings, S. and Kaiser, M.J. (2006). Chronic bottom trawling alters the functional composition of benthic invertebrate communities on a sea-basin scale. Marine Ecology Progress Series, 318:31-45.
[22] Rosellon-Druker, J. and Stokesbury, K.D.E. (2019). Quantification of echinoderms on Georges Bank, and the potential influence of marine protected areas on these populations. Invertebrate Biology, 138:12243.

disturbance. Spatial patterns of echinoderms are also strongly influenced by temperature and depth[22] which should be considered when understanding the effect of management. Research by Ramsay *et al.* also suggests that common starfish numbers can, in fact, increase with bottom trawling pressure until they reach a turning point, and their numbers start declining[23]. The high percentage contribution of two-spot gobies (*Gobiusculus flavescens*) shown inside the SGR is not surprising. Two-spot gobies are often found hovering in complex rocky reef habitats that have an abundance of kelp[24].

High abundances of scavenging species such as harbour crab (*Liocarcinus depurator*) at the OC could be linked to habitat disturbance by bottom trawling. Scavenging species are attracted to the damaged or dead marine life that are disrupted by the dragging movement along the seabed. There is an abundance of evidence that scavenging species can increase post-trawling[25] [26] [27], but this is varies depending on the species, habitat and locality. Squat lobsters (*Galathea* dispersa) are also considered a scavenging species[28] and were also highly present in the trawled areas of the SAC and OC. Whiting (*Merlangius merlangus*) also increases prey intake post-trawling, which could explain their high contribution to total abundance at the OC[29]. This is supported by the functional groupings analysis showcasing higher numbers of scavenging species in the OC and SAC than the SGR. This shift in functional composition of communities can have effects on the functioning of coastal ecosystems[21].

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[23] Ramsay, K., Kaiser, M.J., Rijnsdorp, A.D., Craeymeersch, J.A. and Ellis, J. (2000). Impact of trawling on populations of the invertebrate scavenger Asterias rubens. Iea, 6:2-7.
[24] Utne, A.C.W. and Aksnes, D.L. (1994). An experimental study on the influence of feeding versus predation risk in the habitat choice of juvenile and adult two-spotted goby Gobiusculus flavescens (Fabricius). Journal of experimental marine biology and ecology, 179:69-79.
[25] de Juan, S., Thrush, S.F. and Demestre, M. (2007). Functional changes as indicators of trawling disturbance on a benthic community located in a fishing ground (NW Mediterranean Sea). Marine Ecology Progress Series, 334:117-129.
[26] Groenewold, S. and Fonds, M. (2000). Effects on benthic scavengers of discards and damaged benthos produced by the beam-trawl fishery in the southern North Sea. ICES Journal of marine Science, 57:1395-1406.
[27] Dannheim, J., Brey, T., Schröder, A., Mintenbeck, K., Knust, R. and Arntz, W.E. (2014). Trophic look at soft-bottom communities—Short-term effects of trawling cessation on benthos. Journal of Sea Research, 85:18-28.
[28] Lovrich, G.A. and Thiel, M. (2011). Ecology, physiology, feeding and trophic role of squat lobsters. The biology of squat lobsters, 183:222.
[29] Ramsay, K., Kaiser, M.J. and Hughes, R.N., (1998). Responses of benthic scavengers to fishing disturbance by towed gears in different habitats. Journal of experimental marine biology and ecology, 224:73-89.

Species assemblages between treatments are often highly variable[30], and variation can depend on habitat complexity and availability, competition and predation, or larval and recruitment variability[31]. The unconstrained CA showed that scavenging species such as nephrops (*Nephrops norvegicus*), hagfish (*Myxine glutinosa*) and harbour crab (*Liocarcinus depurator*) were highly associated with the OC compared to other treatments. However, the lack of comparable rocky reef habitat within the OC could have an influence on this result, as these species are often found on the soft sediment substrate in deeper waters.

The SGR showed high association with indicator species, such as juvenile fish and wrasse (*Labridae* spp.), suggesting that some sites within this designation encompass complex and healthy reef habitat suitable for nursery grounds and species refuge[10]. However, juvenile fish species were noted in varying abundances in all treatments.

Future research and recommendations

It is recommended that these surveys are completed in future years to assess species recovery over time following changes in management. For more effective comparison between treatments, it may be useful to expand the surveys to include further sites and replicates. Further effort into identification and selection of sites with comparable reef habitats and depth in the OC and SAC may be useful to ensure analysis is consistent and directly comparable. Furthermore, considering environmental variables in the analysis, such as tidal range or climate, could lead to more specific conclusions on species patterns to identify management measures.

Unfortunately, using Baited Remote Underwater Video to assess species biodiversity has its limitations. For example, not all species observed in the study were attracted to the bait plumes and this can cause bias towards those species that eat mackerel or are scavengers. Interactions between species must also be recognised, with the presence of larger individuals leading to smaller individuals being driven away or eaten, not providing a representative sample of the entire population. Certain species may also be attracted to the

[30] Stige, L. C., Rogers, L. A., Neuheimer, A. B., Hunsicker, M. E., Yaragina, N. A., Ottersen, G., Ciannelli, L., Langangen, Ø. and Durant, J. M. (2019). 'Density- and size-dependent mortality in fish early life stages', Fish and Fisheries, 20:962–976.

[31] Harasti, D., Davis, T. R., Mitchell, E., Lindfield, S. and Smith, S. D. A. (2018). 'A tale of two islands: Decadal changes in rocky reef fish assemblages following implementation of no-take marine protected areas in New South Wales, Australia', Regional Studies in Marine Science, 18:229– 236.

lights on the BRUV units; therefore, it would be extremely useful to conduct further species-specific research to study these interactions. Any conclusions made from the results can only be speculated, as the survey is only sampling across a small area.

Data from these surveys can be used to help inform future marine protection across Berwickshire. Lack of significant differences between treatments shows the need for further effective management of these MPAs and supports the necessity for further research.

Appendix

Appendix A

Table 2: Species list and functional grouping defined by mobility and feeding behaviour

Common name	Latin name	Functional grouping
Ballan wrasse	*Labrus bergylta*	Free-swimming, predator[32]
Brown crab	*Cancer pagurus*	Predator, scavenger[33]
Lesser-spotted dogfish	*Scyliorhinus canicula*	Free-swimming, predator, scavenger[32]
Cod	*Gadus morhua*	Free-swimming, predator[32]
Common prawn	*Palaemon serratus*	Scavenger
Common starfish	*Asterias rubens*	Predator[34] but scavenger on trawl discards[35]
Dab	*Limanda limanda*	Free-swimming, scavenger
Edible sea urchin	*Echinus esculentus*	Neither predator nor scavenger, active or passive omnivore
Fish species	*Actinopterygii* spp.	-
Flying crab	*Liocarcinus holsatus*	Scavenger[36]
Furrowed crab	*Xantho incisus*	Scavenger
Goby	*Gobiidae* spp.	Free-swimming, predator
Goldsinny wrasse	*Ctenolabrus rupestris*	Free-swimming, predator[32]

[32] Henderson, P. (2014). Identification Guide to the Inshore Fish of the British Isles.
[33] Neal, K.J., Wilson, E. (2004). Cancer Pagurus. Edible Crab. Marine life information network: biology and sensitivity key information sub-programme. Mar Biol Assoc UK, Plymouth.
[34] Anger, K., Rogal, U., Schriever, G., Valentin, C. (1977). In-situ investigations on the echinoderm Asteria rubens as a predator of soft-bottom communities in the western Baltic Sea. Helgol wiss Meeresunters, 29:439-459.
[35] Ramsay, K., Kaiser, M.J. (1998). Demersal fishing disturbance increases predation risk for whelks (Buccinum undatum L.). J Sea Res, 39:299-304.
[36] Groenewold, S., Fronds, M. (2000). Effects on benthic scavengers of discards and damaged benthos produced by the beam-trawl fishery in the southern North Sea. ICES J Mar Sci, 57:1395-1406.

Grouped brittlestars	*Ophiuroidea*	Predator and scavenger[37]
Hagfish	*Myxine glutinosa*	Free-swimming, scavenger on dead or dying animals[32]
Harbour crab	*Liocarcinus depurator*	Scavenger[38]
Hermit crabs	*Pagurus spp.*	Scavenger[38]
Juvenile cod	*Juv. Gadus morhua*	Free-swimming, predatory, juvenile
Juvenile fish	*Juv. Actinopterygii* spp.	Free-swimming, juvenile
Juvenile haddock	*Juv. Melanogrammus aeglefinus*	Free-swimming, predator, scavenger, juvenile
Juvenile pollack	*Juv. Pollachius pollachius*	Free-swimming, predator[32], juvenile
Juvenile whiting	*Juv. Merlangius merlangus*	Free-swimming, predator[32], juvenile
Juvenile wrasse	*Juv. Labridae* spp.	Free-swimming, predator, juvenile
Lobster	*Homarus gammarus*	Predator
Marbled swimming crab	*Liocarcinus marmoreus*	Scavenger
Nephrops	*Nephrops norvegicus*	Predator, scavenger
Plaice	*Pleuronectes platessa*	Free-swimming, predator
Pollack	*Pollachius pollachius*	Free-swimming, predator
Red gurnard	*Chelidonichthys cuculus*	Free-swimming, predator[32]
Seabass	*Dicentrarchus labrax*	Free-swimming, predator
Square crab	*Goneplax rhomboides*	
Squat lobster	*Galathea dispersa*	Predator, scavenger
Swimming crab	*Liocarcinus* spp.	Scavenger
Tompot blenny	*Parablennius gattorugine*	Free-swimming, predator
Two-spot goby	*Gobiusculus flavescens*	Free-swimming
Velvet swimming crab	*Necora puber*	Predator[39], scavenger[40]
Whiting	*Merlangius merlangus*	Free-swimming, predator
Wrasse sp.	*Labridae* spp.	Free-swimming, predator

[37] Feder, H.M. (1981). Aspects of the feeding biology of the brittlestar Ophiura texturata. Ophelia, 20:215-235.
[38] Ramsay, K., Kaiser, M.J. and Hughes, R.N. (1998). Responses of benthic scavengers to fishing disturbance by towed gears in different habitats. Journal of experimental marine biology and ecology, 224:73-89.
[39] Norman, C.P., Jones, M.B. (1992). Influence of depth, season and moult stage on the diet of the velvet swimming crab Necora puber (Brachyura:Portunidae). Estuar Coast Shelf Sci, 34:71-83
[40] Moore, P.G., Howarth, J. (1996). Foraging by marine scavengers: effects of relatedness, bait damage and hunger. J Sea Res, 36:3-4.

Appendix B

Table 3: Hypothesis Test Summary of Kruskal-Wallis test for species abundance.

	Null Hypothesis	Test	Sig.[a,b]	Decision
1	The distribution of relative_abundance is the same across categories of treatment.	Independent-Samples Kruskal-Wallis Test	.937	Retain the null hypothesis.

a. The significance level is .050.
b. Asymptotic significance is displayed.

Table 4: Independent-Samples Kruskal-Wallis Test Summary for species abundance.

Total N	10
Test Statistic	.131[a]
Degree Of Freedom	2
Asymptotic Sig.(2-sided test)	.937

a. The test statistic is adjusted for ties.

Table 5: Ranks from Kruskal-Wallis test for species abundance.

	treatment	N	Mean Rank
relative_abundance	SAC	3	5.00
	SGR	5	5.80
	OC	2	5.50
	Total	10	

Appendix C

Table 6: Hypothesis Test Summary for species richness.

Hypothesis Test Summary

	Null Hypothesis	Test	Sig.[a,b]	Decision
1	The distribution of Mean_richness is the same across categories of treatment.	Independent-Samples Kruskal-Wallis Test	.902	Retain the null hypothesis.

a. The significance level is .050.
b. Asymptotic significance is displayed.

Table 7: Independent-Samples Kruskal-Wallis Test Summary for species richness.

Total N	10
Test Statistic	.206[a]
Degree Of Freedom	2
Asymptotic Sig.(2-sided test)	.902

a. The test statistic is adjusted for ties.

Table 8: Ranks from Kruskal-Wallis test for species richness.

	treatment	N	Mean Rank
Mean_richness	SAC	3	5.00
	SGR	5	5.50
	OC	2	6.25
	Total	10	

Appendix D

Table 9: Summary table of Unconstrained Correspondence Analysis (CA)

Statistic	Axis 1	Axis 2	Axis 3	Axis 4
Eigenvalues	0.4082	0.2330		
Explained variation (cumulative)	63.66	100.00		

Appendix E

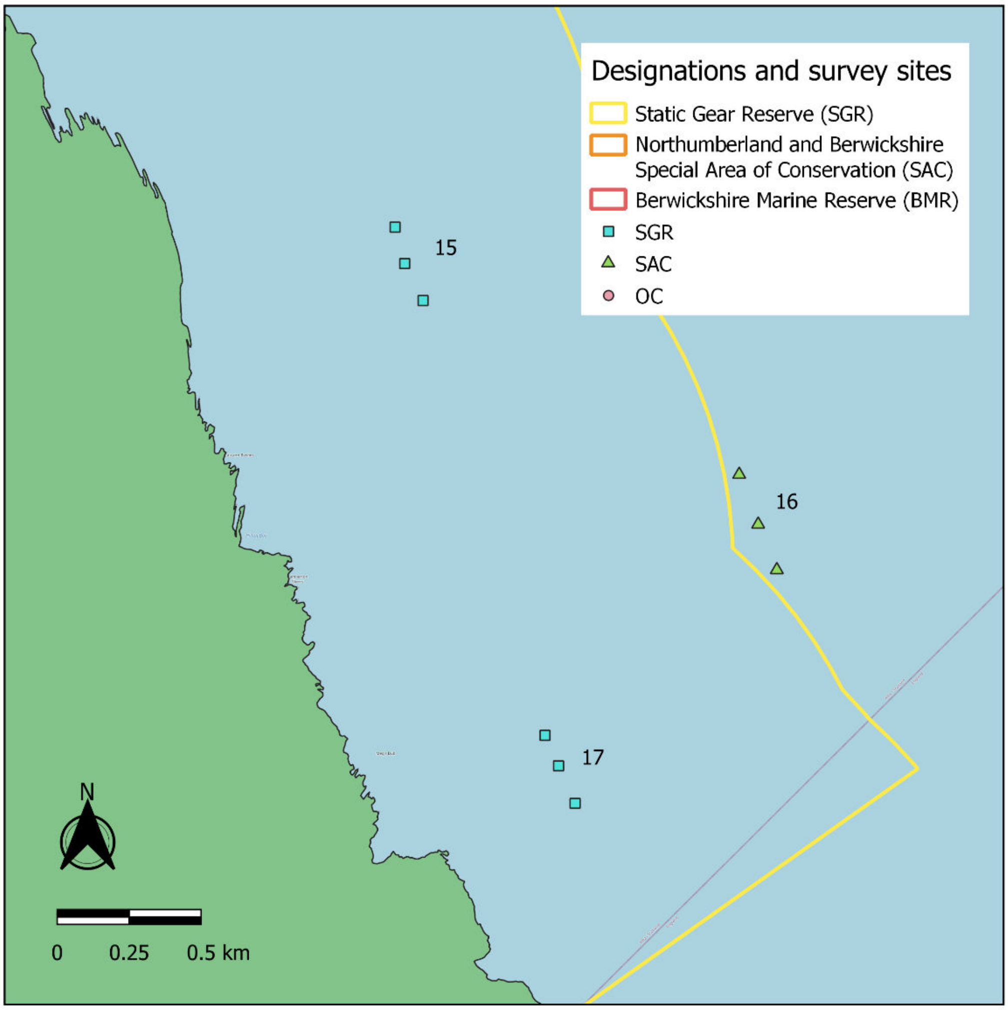

Figure 8: Close up map of sites 15, 16 and 17 to show different treatments.

Appendix F

Table 10: List of coordinates and depths of BRUV survey sites

Site Number	BRUV No.	Depth (m)	Latitude (DM.m)	Longitude (Dm.m)
1	1	23.2	55'55.288	002'10.561
1	2	23.7	55'55.263	002'10.414
1	3	24.4	55'55.235	002'10.294
2	1	25.9	55'56.088	002'14.214
2	2	25.8	55'56.096	002'14.330
2	3	26.2	55'56.107	002'14.450
6b	1	70.6	55'57.183	002'11.485
6b	2	70.2	55'57.240	002'11.589
6b	3	69.5	55'57.275	002'11.630
9	1	14.3	55'54.226	002'07.673
9	2	14.8	55'54.290	002'07.641
9	3	17.4	55'54.358	002'07.607
10	1	13.1	55'52.774	002'05.836
10	2	14.5	55'52.768	002'05.966
10	3	13.2	55'52.797	002'06.079
11b	1	38.3	55'53.699	002'05.565
11b	2	31.6	55'53.736	002'05.696
11b	3	36	55'53.778	002'05.759
13	1	22.5	55'53.202	002'05.673
13	2	18.5	55'53.144	002'05.565
13	3	16.7	55'53.065	002'05.442
15	1	23.4	55'50.194	002'02.543
15	2	20.8	55'50.126	002'02.511
15	3	22.1	55'50.057	002'02.450
16	1	28.7	55'49.732	002'01.400
16	2	26.7	55'49.639	002'01.337
16	3	30.4	55'49.554	002'01.275
17	1	18.9	55'49.245	002'02.045
17	2	17.3	55'49.188	002'02.000
17	3	17.5	55'49.118	002'01.945