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Impacts of benthic trawling on sponge community composition around Western Iceland

Chloe Webster
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University College London (UCL)
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Ágrip:

The productive waters of western Iceland are home to an important commercial fish species, the Atlantic cod. Demersal trawling, the main form of fishing in this particular industry, is known to have adverse effects on surrounding habitats and organisms on the deep-sea floor. Sponges are important elements of benthic communities, providing habitats and protection for other epifauna. These sponges are highly vulnerable to intense fishing activity, due to their slow-growth and delicate body forms. In 2011, a scientific cruise collected benthic images of the seabed at depths between 175-360m from different sites around western Iceland, subject to varying fishing efforts. Here, we identify a range of sponge taxa from these images, and explore the impact of fishing effort on sponge community composition using multilinear regressions and multidimensional scaling. This study demonstrates that protected areas contain a higher diversity in community composition and larger sizes of individual taxa, but that overall abundance is similar between fished and unfished sites. This is due to one small, successful species, *Quasillina brevis*, which seems to thrive in heavily impacted areas. Environmental factors silicate and slope also show significance in affecting sponge distributions, although fishing effort is the overall driver. Nonetheless, further study is required to both increase identifications of sponge taxa, and to determine the threshold at which trawling effort becomes destructive to the long-term prevalence of sponge communities. Overall, this can be used to help inform future management and protection strategies.

Lykilorð:

Western Iceland, Sponges, Demersal Trawling, Benthic Organisms, Community Composition.

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Impacts of benthic trawling on sponge community composition around Western Iceland

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Abstract

The productive waters of western Iceland are home to an important commercial fish species, the Atlantic cod. Demersal trawling, the main form of fishing in this particular industry, is known to have adverse effects on surrounding habitats and organisms on the deep-sea floor. Sponges are important elements of benthic communities, providing habitats and protection for other epifauna. These sponges are highly vulnerable to intense fishing activity, due to their slow-growth and delicate body forms. In 2011, a scientific cruise collected benthic images of the seabed at depths between 175-360m from different sites around western Iceland, subject to varying fishing efforts. Here, we identify a range of sponge taxa from these images, and explore the impact of fishing effort on sponge community composition using multilinear regressions and multidimensional scaling. This study demonstrates that protected areas contain a higher diversity in community composition and larger sizes of individual taxa, but that overall abundance is similar between fished and unfished sites. This is due to one small, successful species, *Quasillina brevis*, which seems to thrive in heavily impacted areas. Environmental factors silicate and slope also show significance in affecting sponge distributions, although fishing effort is the overall driver. Nonetheless, further study is required to both increase identifications of sponge taxa, and to determine the threshold at which trawling effort becomes destructive to the long-term prevalence of sponge communities. Overall, this can be used to help inform future management and protection strategies.

Key words: Western Iceland, Sponges, Demersal Trawling, Benthic Organisms, Community Composition.

Introduction

Deep-sea ecosystems (>200m) collectively make up the largest biome on earth, covering over 60% of the earth's hard crust (Glover & Smith 2003). Due to the vastness of these ecosystems, and the time and resources required for sufficient research, they are unsurprisingly understudied both functionally and biologically (Megrey et al. 2009; Smith et al. 2009; Armstrong et al. 2010). However, they are known to contribute to a number of ecosystem services, including habitat, nutrient recycling and carbon storage (Thurber et al. 2014; Kahn et al. 2015). Since the 1970s, there has been a global movement towards fishing in deeper environments, driven by the over-exploitation of shallow, coastal habitats, as well as improved technological advances (Levin and Dayton 2009; Norse et al. 2012). However, negative impacts on the deep-sea can be long-lasting and local and regional recovery is slow (Pitcher et al. 2015). Negative impacts include the extensive removal of organisms as by-catch, estimated to contribute to 40% of the overall global catch per year (Chuenpagdee et al. 2003). Such habitats are also at risk of exploitative demersal (or bottom) trawling, often resulting in widespread, long-term degradation, both biologically and geologically (Danovaro et al. 2008; Norse et al. 2012; Keledjian et al. 2014). The current understanding of the impacts exerted on these deep-sea habitats and communities is limited by knowledge gaps in the literature, calling for an increase in research to harmful anthropogenic impacts like overfishing and by-catch. This is necessary in order to manage and conserve local habitats for sustainability purposes, for the sake of the overall environment as well as local economy (Danovaro et al. 2008).

Although positioned close to the Arctic Circle at 63-67°N and 18-23°W, Iceland's waters are rich with life. In particular, waters surrounding the Westfjords, where the northwards-flowing North Atlantic Drift splits into the Irminger Current and passes the southwards-flowing East Greenland Current, along the Denmark Strait, are continuously bathed in warm and saline Atlantic waters, causing primary production to be higher in these waters than anywhere else around Iceland (Astthorsson et al. 2007).

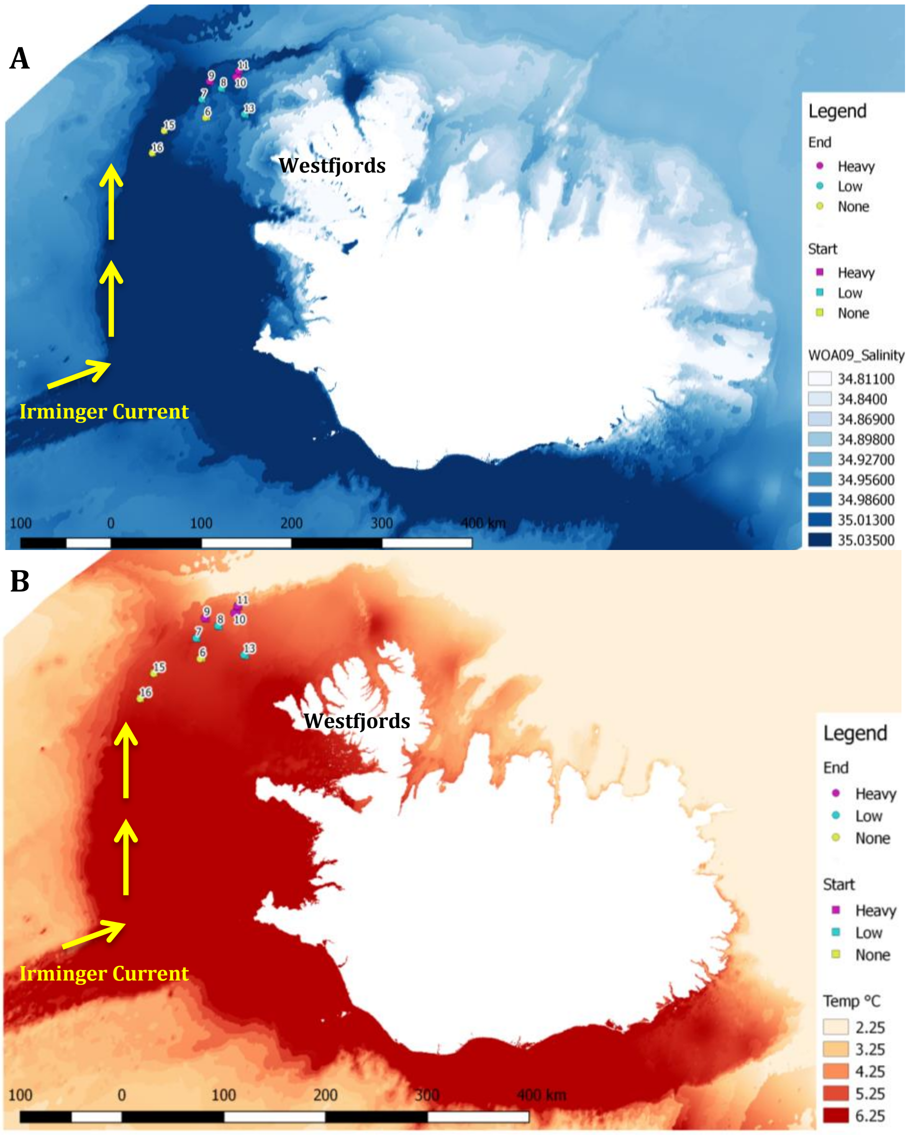


Figure 1, A and B Maps of Iceland, with map **A** showing salinity and map **B** showing temperature, as inferred from GIS data based on location and depth, using World Ocean Atlas. The pattern shown by the darker colours shows the northwards flow of the Irminger Current. Survey stations are situated to the North West of the Westfjords.

As a result, deep sea ecosystems are subject to varying degrees of fishing effort due to the abundance one of the country's most important species in the commercial fishing industry, the Atlantic cod (*Gadus morhua*), accounting for around 40% of the overall seafood export revenue in recent years (Ministry of Fishing and Agriculture). As a demersal fish species that feeds on benthic infauna and epifauna, the Atlantic cod is typically fished using a demersal trawl that uses weights to allow the net to drag along the seabed. These waters have been subject to intense cod fishing since the turn of the 20th Century, when the first Icelandic trawler started operating in 1905 (Schou 2000). In Iceland, the contemporary demersal trawl is known to catch the highest value of the total catch (40-50%) compared to other fishing gears, thus proving to be of great importance to the fishing industry (Knútsson et al. 2011). The industry as a whole is very important economically, making up 41% of merchandise exports, 26% total exports, and ~10.5% of the country's GDP (as of 2011): Iceland is therefore heavily dependent on its fisheries as a main income source (Knútsson et al. 2011). Therefore, as well as aiming to assure long term sustainability for the future of Iceland's economy, the Icelandic fishing industry is also under pressure to meet with a growing global demand. With this demand often comes the use of fishing methods that are not only exploitative of fish stocks if unsustainably managed, but also damaging to the surrounding deep-sea environments (Ólafsson et al. 2015).

Intense trawling has been documented to have severe impacts on large benthic, structural species like sponges and corals, due to their sedentary nature, slow-growth and longevity (Collie et al. 2000; Curtis et al. 2013). Sponges (phylum Porifera) are integral members of deep-sea benthic communities, rivaling reef-building corals in their abundance and diversity and forming large structural habitats that provide shelter for epifauna, including some fish species (Miller et al. 2012; Kersken et al. 2014). The slow-growth of deep-sea sponges is likely owed to the low temperatures and light availability below 200m, allowing them to live up to several hundred years (Leys and Lauzon 1998; Wagner and Kelley 2016). Certain sponge taxa, notably Hexactinellida and Demospongiae classes, form dense aggregations, referred to as "ostur" by local fishermen, along the North Atlantic seafloor (Klitgaarda and Tendala 2004; OSPAR Commission 2010). These dense aggregations are composed of 'high-risk' taxa due to their vulnerability to fragmentation and ease in being caught by demersal trawlers (Kersken et al. 2014; Jørgensen et al. 2016). The removal of sponges as by-catch may in turn result in a series of negative impacts to the ecosystem functions that these organisms supply to the deep-sea environment, such as water filtration and habitat provisioning (NOAA 2010). However, although the impacts of the removal of sponges are relatively unknown, the removal of other organisms has been seen to cause disruption in important ecosystem dynamics, such as the alteration of marine food webs (Steele and Schumacher 2000).

There are, however, certain protective measures in place around Icelandic waters. These include exclusions of certain fishing gears, temporary closures of areas to allow juvenile cod growth, and the establishment of marine protected areas (MPAs); often around vulnerable marine ecosystems (VMEs) (Ministry of Fishing and Agriculture). In Iceland, VMEs include cold-water corals, hydrothermal vents and large sponge aggregations. However, the Icelandic Government has not yet implemented any future plan for the protection of sponge beds, as it has for the other VME priority-habitats, owing mainly to the lack of research supporting the need for such protection (M.S.C.S.F. Assessment 2014). Overall, sponge communities and distributions remain poorly understood, with few studies attempting to describe their ecology. One successful example is the collaborative BIOICE project, which identified 2,000 marine invertebrate fauna around Iceland, many of which were sponges, with 41 new invertebrates described as new to science (Klitgaarda and Tendala 2004; Omarsdottir et al. 2013). In general however, deep-sea communities of the Iceland Sea have not been subject to much biological study to date, with focus mainly on physical and chemical oceanography (Palsson et al. 2012). This means that although there are a number of studies showing the negative impacts of fishing in similar sponge habitats (e.g. Van Dolah et al. 1987; McConnaughey et al. 2000), poor knowledge of the community compositions of these habitats around Iceland hinders further conservation and protection measures in response to anthropogenic activities.

In this study we aim to investigate how fishing effort may affect the abundance, diversity and individual size of benthic sponges and their communities, by analyzing the trawling impacts of 9 chosen sites around the western seas of Iceland. We test the hypothesis that trawling negatively impacts sponge communities, as well as the relative importance of fishing compared with environmental influences of community composition. By doing this we aim to test the value of protected areas, with the view of assisting potential management changes to the fisheries. We also aim to provide a general overview of the sponge taxa present in the area, to assist in the on-going benthic mapping of the Marine and Freshwater Research Institute (MFRI), Iceland.

Materials and Methods

Study Region and Data Collection

In 2011, benthic surveys were conducted upon research vessel RV Bjarni Sæmundsson, as part of a Benthic Habitat Mapping project carried out by the MFRI, Iceland. Sites were chosen to represent different areas of impact, from protected to heavily fished areas, (see Table 2 for details). At each site, the research vessel engine was switched off, and an underwater, digital, Sony “Campod” camera, mounted to a three-legged frame, was sunk to the seabed. The camera took images at an average of 3 seconds apart as the vessel drifted at a range between 0.76 to 2.02 knots (kn). As a result, transects ranged from 29 to 94 minutes covering a range of distances of 755 to 2778m (Appendix Supplementary Table (ST) 4). Although 23 transects were made, not all were suitable for analysis, and 9 were selected based on quality and fishing effort. From these, images (n = 20) were selected based on a standardized level of quality (the least marine snow or sediment cloud), scale (1.5 cm – 2.5 cm on image between the two laser beams located 10 cm apart on camera), and an unobscured view. Environmental data was inferred from GIS data based on location and depth, using data from World Ocean Atlas for temperature, salinity and silicate and GLODAP for carbonate chemistry (Garcia et al. 2013; Locarnini et al. 2013; Zweng et al. 2013).

Data Processing

Images from the photographic transects were uploaded to software analysis ‘Poseidon’ (Johns 2014), and sponge taxa were identified and tagged to lowest possible level. It was not possible to identify all sponges present due to the visibility difficulty in the images and lack of dry specimen for accurate species identification: therefore morphotypes were used. Often, even when physical samples are available, species identification is difficult without genetic information due to the little variation between spicules, so using morphotypes as appropriate measures is common across sponge studies (Solé-Cava & Thorpe 1986). In this investigation, morphotypes were used: for taxa that were recognisable around Iceland by previous visual analysis, but had not yet been identified; taxa that had not been previously identified in Icelandic waters, but were known elsewhere; and taxa seemingly absent from the literature, but consistent in size and shape e.g. white lattice vase/leaf. However, classifying morphotypes by shape, structure, colour and size may be unreliable, as some sponges are known to differ in morphology within one species, and some similar in morphology may be different species (Swierts et al. 2013).

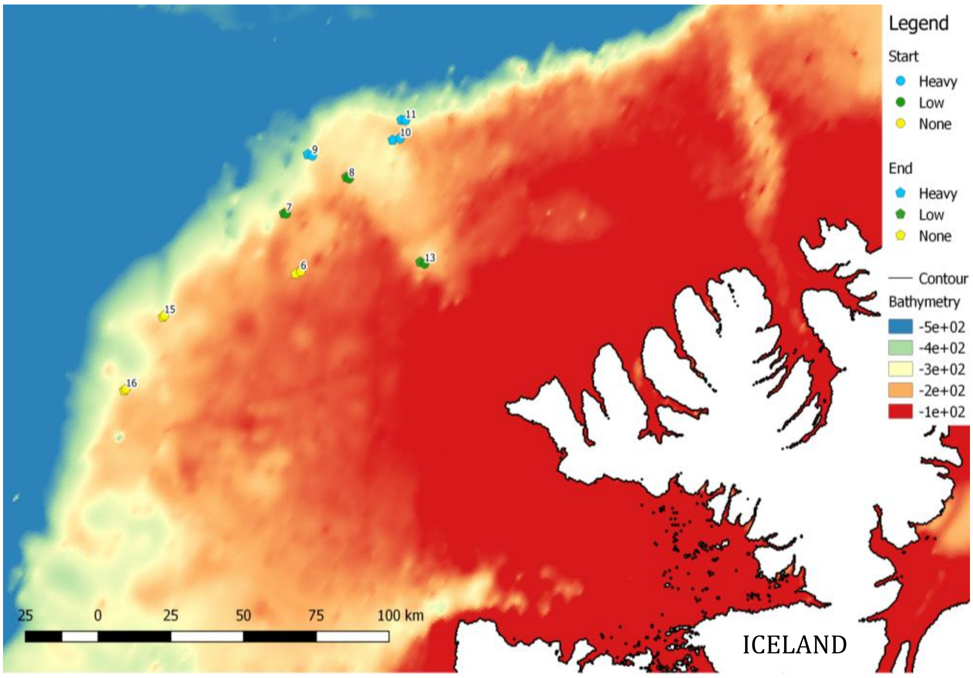


Figure 2 Location of surveyed stations in North Western Iceland, showing bathymetry depths between -100m (red) and -500m (blue).

As varied morphologies may be the result of local environmental influences, human error should be taken into account if using data from this study (Conway et al. 2007). Therefore, analysis was performed at the lowest taxon identification level, which was either species or morphotype. Fishing effort data were obtained through the Icelandic Fisheries Electronic Log Book (ELB) and Vessel Monitoring System (VMS), by averaging the numbers of trawls per year between 2008 and 2013 within a 1000m buffer around each transect.

Analysis

Statistical language and programming software "*R*", was used for data analysis (R Development Core Team 2015). Diversity indices (DIs) for sponge identifications at each station were obtained through the '*Vegan*' package in *R:* Abundance, Shannon, Simpson, number of taxa (NTaxa), Evenness and Richness (Oksanen et al. 2016). Abundance, as the count of all individuals per taxa, differs from the NTaxa (number of taxa present across all sites). Simpson diversity is a similarity measure that accounts for relative abundance of each species, and the number of those species present, weighted towards identifying common species. Shannon diversity takes into consideration Richness and Evenness, aiming to describe the certainty of predicting a random sample from the overall population. Evenness describes the spread of the taxa, determining how equal in abundance each species is. The Richness output is the result of the '*invsimpson*' DI, similar to rarefied species richness: the expected number of species for a given number of a random sample of individuals (Oksanen et al. 2016). Normality was tested using histograms and Shapiro-Wilk tests, and transformed accordingly to reduce skewness. Difference between index means for each fishing effort was tested with a one-way Analysis of Variance (ANOVA), and where significant, *post hoc* Tukey tests were performed to determine where the significance lied between groups. In order to determine the contribution of each individual species to Bray-Curtis dissimilarity between sites, SIMilarity PERcentage analysis (SIMPER) was performed using the fishing effort groups, none, low and heavy (Oksanen et al. 2016). This resulted in the percentage contribution of the top six taxa that best explain the similarities or dissimilarities between each paired combination. Sponge taxa were also categorized into 3 sizes based on calibrated images and general morphotypes: >10 cm in height, thin encrusting, <9 cm in height. ANOVA and Tukey tests were run to determine possible significance of sponge size between fishing effort types.

Multiple linear regression (MLR) predictive analyses were performed in order to explain the relationship between dependent variables from multiple explanatory variables (i.e. "diversity index ~ environment"), however not all environmental variables could be used due to high collinearity between certain pairs. Therefore, prior to modelling, a Pearson's correlation matrix was created, using the '*cor*' function, in order to determine variables with high collinearity (Becker et al. 1988). Variance inflation factors *(vif)* from package *'HH'* were used to determine the highest possible combination of variables with the least collinearity (Heiberger 2016). This iterative process removes variables with the highest *'vif'* value at each step until all variables are under the value of 5. The '*rel.imp'* factor from package *'relaimpo'* was then used to measure relative importance of each variable within the model, using metrics of lmg (R^2 partitioned by averaging over orders), (Grömping 2006).

Non-metric Multi-Dimensional Scaling (nMDS) analyses were used in order to provide ordination based on a dissimilarity matrix, identifying the composition of taxa between the 9 stations. Unlike other ordination techniques like Principal Coordinates Analysis, rank orders are used, making nMDS flexible enough to use on a wide variety of data. This also makes it appropriate to use for irregular distributions of marine communities (Yesson et al. 2015). In order to perform the analyses, the '*MetaMDS*' function in '*vegan*' package was used (Ramette 2007, Oksanen et al. 2016). nMDS constructs a 2-dimensional (2-D) configuration of the samples and regresses the differences between the configured and the predicted distances, termed 'stress'. Stress >0.05 shows excellent representation, > 0.1 is good, >0.2 is medium, stress >0.3 is poor. The '*enfit'* function in the '*vegan'* package was used to fit environmental variables onto the ordination in a complimentary analysis (Oksanen et al. 2016). This enabled points to be projected in correlation to one another, the influences of which are shown via direction and length of arrows on the plot.

Across all analyses, significance thresholds range from the traditional $p < 0.05$ to $p < 0.1$, that latter of which is usually dismissed due to the increased probability of a Type I error (falsely concluding the presence of a significant effect). However, Type II errors (falsely excluding the presence of a significant effect) should be accounted for, and are at least as important as Type I errors, especially when documenting impacts on the environment for resource management (Thrush et al. 1995).

Results

Sites of different fishing efforts were categorized by <1 trawl/year = No Effort (n = 3), 1-2 trawls/year = Low Effort (n = 3) and >2 trawls/year = Heavy Effort (n = 3), see Table 2. In total, 120 images were analyzed along 9 different transects (sites) and sponges were identified to three different taxa level, species (n = 16), genus (n = 6), family (n = 3) identifications. Each taxon identified, and their abundances, are shown in ST1. Those taxonomically unidentifiable were classified into morphotypes (n = 31), totaling at 56 (see ST1). DIs of the raw data at station level included the most abundant species, the fishing effort categorized for each station, and the top 3 most abundant species from each fishing effort, (Tables 1, 2 and 3 respectively).

Station	Abundance	NTaxa	Simpson	Shannon	Richness (invsimpson)	Evenness	Most Abundant
6	245	25	0.802	2.181	5.063	0.678	*Q. brevis*
7	146	21	0.635	1.731	2.736	0.569	*Q. brevis*
8	159	27	0.702	2.104	3.357	0.638	*Q. brevis*
9	153	16	0.548	1.425	2.214	0.514	*Q. brevis*
10	321	24	0.432	1.203	1.760	0.379	*Q. brevis*
11	234	24	0.663	1.851	2.965	0.582	Axinellidae
13	123	24	0.844	2.441	6.408	0.768	*Q. brevis*
15	280	31	0.797	2.377	4.915	0.692	*Q. brevis*
16	310	35	0.750	2.240	3.999	0.630	*Q. brevis*

Table 1 Abundances and diversity indices (DIs) for all sponge taxa at each station, with station labels.

Station	Category	Fishing Effort
6	None	0.021768707
15	None	0.031578947
16	None	0.045614035
7	Low	1.545901639
8	Low	1.195419847
13	Low	1.483116883
9	Heavy	12.6425
10	Heavy	3.5375
11	Heavy	9.334782609

Table 2 Average fishing effort per station measured in mean number of trawls/year for the years 2008-2013, and categorized by None = <1 trawl, Low = 1-2 trawls, Heavy = >2 trawls. A 1000m buffer was used around each transect.

Fishing Category	**Most Abundant Taxa**	**Overall Abundance**	**Image examples**
None	*Quasillina brevis*	335	
	cf *Antho dichotoma*	60	
	Axinellidae	53	
Low	*Quasillina brevis*	171	
	Axinellidae	47	
	Mycale lingua	23	
Heavy	*Quasillina brevis*	473	
	cf *Antho dichotoma*	27	
	Lobate massive	22	

Table 3 The top 3 most abundant taxa, their overall abundances, and image examples are displayed for each fishing category.

The total abundance of sponges at each station, grouped into fishing efforts, is shown in Figure 3. Average abundances and standard deviations for each level of fishing effort are shown in Table 4. Relationships between fishing effort types and other DIs at station level are shown on scatter plots in Figures 4; A-E. Due to the high abundance of *Q. brevis* across most sites, abundances of sponges minus (–) *Q. brevis* are also shown in Figure 3, to compare the difference in trends. When using a multiple linear regression (MLR), fishing effort is more significant in affecting abundance within the model for the transformed abundance minus *Q. brevis*, ($p = 0.03$), than for the overall transformed abundance, with ($p = 0.10$), (Table 6). In order to determine whether there is a significant difference between fishing efforts in terms of sponge abundance, the null hypothesis (H_0) states that there is no significant difference between extreme fishing efforts (i.e. stations of none and heavy fishing). ANOVA tests were run for both a) total abundance ($p = 0.03$, $F = 6.25$) and b) abundance – *Q. brevis* ($p = 0.04$, $F = 6.03$) and were both significant at $p < 0.05$. *Post-hoc* Tukey tests showed significant differences between none and low fishing ($p = 0.03$) for a) total abundance, and between none and heavy fishing ($p = 0.04$) for b) abundance – *Q. brevis.* Thus, the H_0 can be accepted for test a), but can be rejected for test b). Further ANOVA and Tukey tests were performed to determine any significant differences between the stations at fishing efforts concerning diversity, significance levels were set at $p < 0.05$. There was a significant difference between none and heavy fishing with Simpson diversity ($p = 0.05$, $F = 5.32$) and with Shannon diversity ($p = 0.04$, $F = 6.05$), thus rejecting the H_0, but no significance occurred with Evenness or Richness, accepting the H_0.

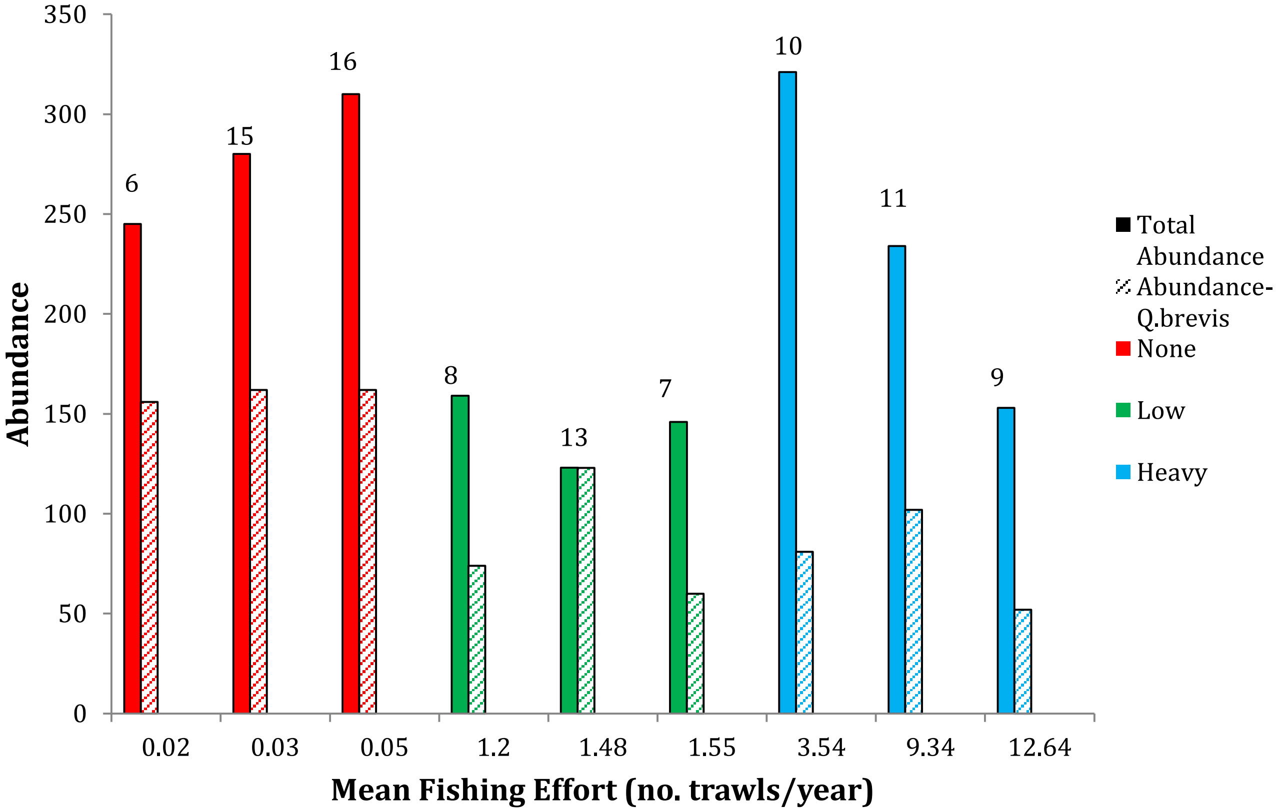
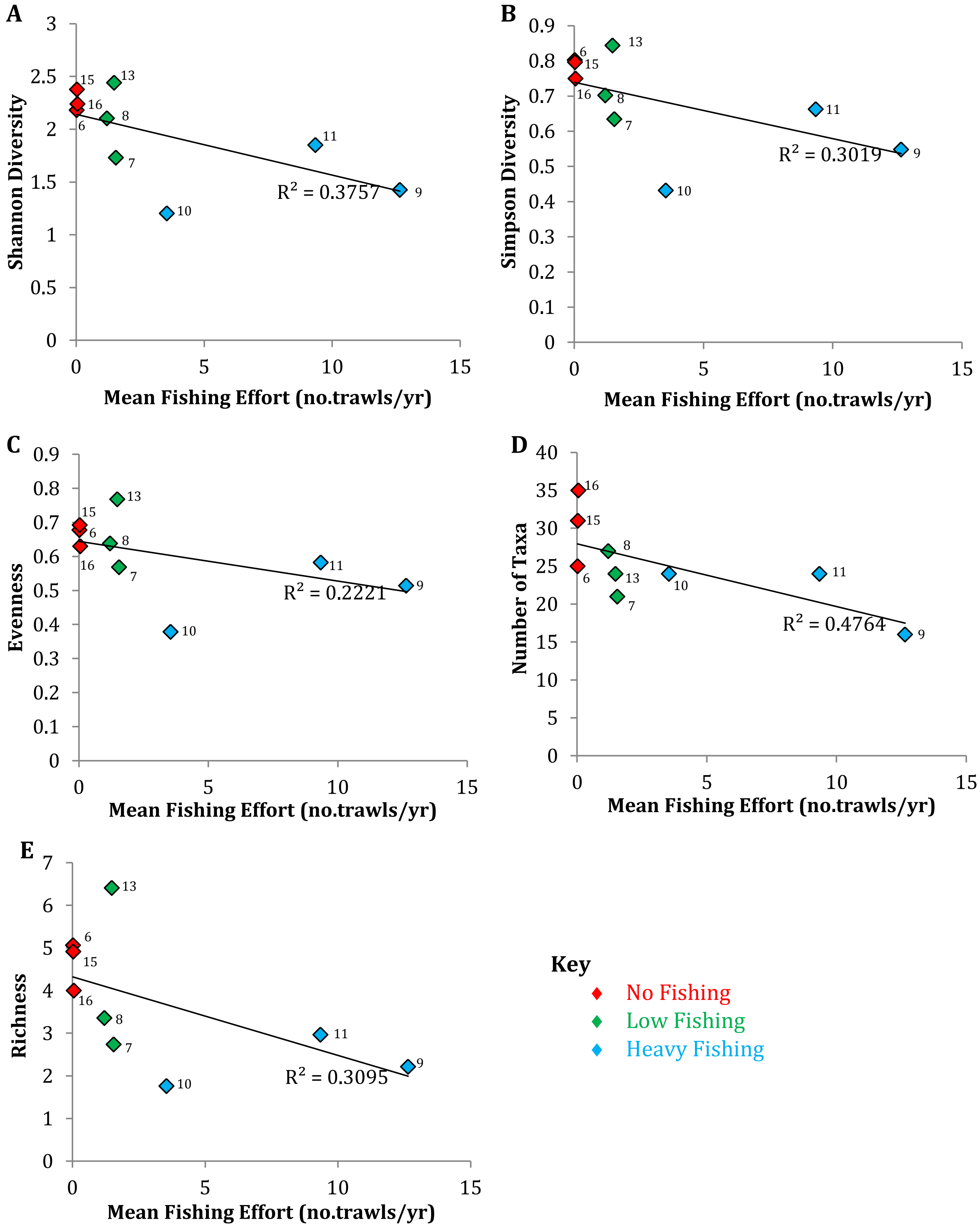


Figure 3 The total abundance of sponge taxa and the abundance of sponge taxa minus the most common sponge, *Q. brevis*, at each station. These stations are categorized into one of the three fishing effort types (None, Low and Heavy) based on average number of trawls/year between 2008 and 2013.

Fishing Effort	Mean Abundance	Standard Deviations
None	278.33	32.53
Low	142.66	18.23
Heavy	236.00	84.02

Table 4 The averages and standard deviations for overall abundance of sponges at each fishing effort type.



Figures 4; A-E The relationship for DIs Shannon (A), Simpson (B), Evenness (C), Number of Taxa (D) and Richness (E) at each station, against mean fishing effort, measured in average number of trawls/year across 2008-2013, classified into three types: None, Low and Heavy.

The chosen environmental variables inferred from GIS data are show in Table 5 for each station. As grab samples were not attainable, sediment types were categorized visually from the calibrated images, and based where possible on the Wentworth sediment scale (Wentworth 1922). Cobbles were categorized as 6-20 cm, and boulders as ≥30 cm (see image examples of sediments in ST3). ANOVA results showed a significant difference between the abundance of sponges in different sediments ($p = 0.07$, $F = 4.35$). Tukey tests imply low significance between Mixed, Cobbles & Boulders and Mixed, Cobbles ($p = 0.09$). Further ANOVA and *post hoc* analyses showed significant differences between these two sediment types with Simpson, ($p = 0.07$, $F = 4.41$) and Shannon ($p = 0.03$, $F = 6.65$) DIs. However, Evenness and Richness did not show any significance. Figure 6 shows the different sediment types for each station, categorized into fishing effort. In order to determine relationship between sediment type, fishing effort and sponge taxa, sponges were categorized into 3 size classes >10 cm in height, thin encrusting and <9 cm in height, also, <9 cm in height minus the abundant *Q. brevis*, showing that abundance of larger sponges are higher in areas of no fishing (see Figure 5 and ST2). These categories are based on the calibrated images, general morphotypes, and size estimates from online identification portals (e.g. Boxshall et al 2016). ANOVA and Tukey tests resulted in a significant difference seen only between none and heavy fished stations for taxa >10 cm, ($p = 0.07$, $F = 3.81$).

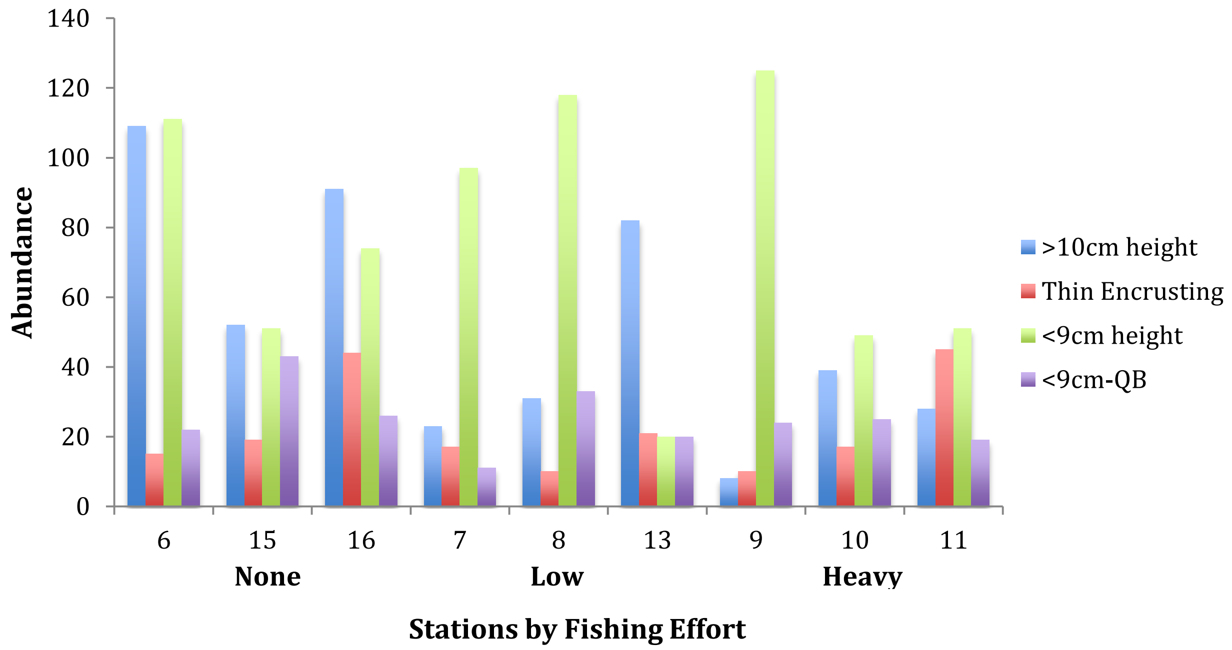


Figure 5 Abundances of sponge taxa classified into three groups, with a fourth group to demonstrate the abundance within the <9 cm group minus the hyper-abundant *Q. brevis*.

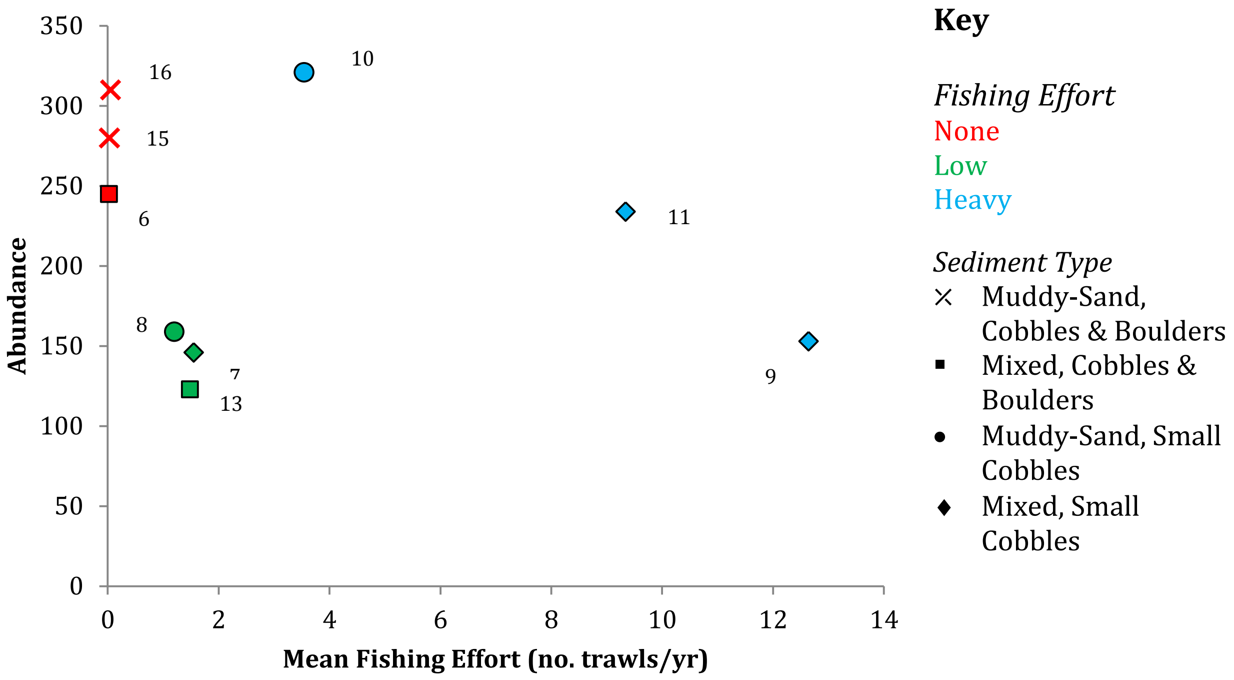


Figure 6 The total abundance of sponge taxa at each station, labeled with station number and categorized by colour for fishing effort types, (mean number of trawls/year between 2008 and 2013) and by shape for sediment type.

Station	Depth Range	Mean Depth	Slope (°)	Temp (°C)	Salinity	Silicate	Calcite
6	175-181	178	0.001486	5.58	34.99	7.48	3.01
7	294-358	326	0.002567	5.54	35.03	8.10	2.87
8	229-281	255	0.001371	5.24	35.01	7.60	2.94
9	201-242	221.5	0.002327	5.08	35.03	7.81	2.80
10	220-233	226.5	0.000609	5.08	34.98	7.36	3.01
11	237-264	250.5	0.003372	4.55	35.02	7.54	2.80
13	317-331	324	0.003243	5.43	35.03	7.94	2.95
15	227-234	230.5	0.001434	5.55	35.04	7.76	2.87
16	245-251	248	0.001939	5.74	35.05	7.74	2.87

Table 5 Environmental data at each station inferred from GIS data, using data from World Ocean Atlas and GLODAP based on location and depth.

Prior to MLR modelling, a Pearson’s correlation matrix was run using the ‘*cor*’ function, along with a test for multicollinearity using ‘*vif*’ function in order to choose the variables with the least correlation and collinearity to run MLR analyses through. Apart from fishing effort, resulting response variables most suitable according to these tests were slope, silicate and calcite. Stepwise MLR analyses performed for each DI are shown in Table 6, along with the calculated value of relative importance for each environmental variable.

An nMDS plot was then used to display the ordination of the community composition at the different stations, showing clustering of stations in fishing effort with good stress levels (0.132). The most significant environmental variable to determine this community composition is silicate (p = 0.013), (Figure 7). SIMPER output displays the percentage contribution of the top 3 taxa per fishing effort type when compared between each pair (Table 7). These taxa were then placed on the nMDS plot to show which is influenced the most by which environmental factors.

	Estimate	**Std. Error**	**T value**	**Pr (>|t|)**	**Rel.imp(lmg)**
Abundance					
Fishing Effort	-0.03	0.02	-2.18	0.09	0.20
Slope	-0.13	0.10	-1.31	0.26	0.20
Silicate	-0.39	0.20	-1.96	0.12	0.30
Calcite	-3.11	1.67	-1.86	0.13	0.08
Overall R^2 = 0.85		**F** = 3.31		**P** = 0.18	
Abundance - *Q. brevis*					
Fishing Effort	-0.07	0.02	-4.34	**0.007**	0.68
Slope	0.17	0.10	1.75	0.14	0.04
Silicate	-0.38	0.21	-1.80	0.13	0.07
Overall R^2 = 0.80		**F** = 6.80		**P** = 0.03	
NTaxa					
Fishing Effort	-1.64	0.57	-2.87	**0.02**	1.00
Overall R^2 = 0.54		**F** = 8.19		**P** = 0.02	
Simpson					
Fishing Effort	-0.04	0.01	-4.06	**0.009**	0.46
Slope	0.21	0.05	3.91	**0.01**	0.33
Calcite	1.65	1.04	1.59	0.17	0.06
Overall R^2 = 0.86		**F** = 10.56		**P** = 0.01	
Shannon					
Fishing Effort	-0.14	0.04	-3.56	**0.01**	0.52
Slope	0.43	0.18	2.43	**0.05**	0.20
Overall R^2 = 0.72		**F** = 7.71		**P** = 0.02	
Evenness					
Fishing Effort	-0.03	0.01	-2.68	**0.04**	0.31
Slope	0.19	0.06	3.40	**0.01**	0.40
Calcite	1.58	1.10	1.43	0.21	0.07
R^2 = 0.79		**F** = 6.20		**P** = 0.04	
Richness					
Fishing Effort	-0.35	0.14	-2.50	**0.05**	0.33
Slope	2.35	0.77	3.06	**0.03**	0.28
Calcite	32.51	15.15	2.15	*0.08*	0.16
Overall R^2 = 0.78		**F** = 5.76		**P** = 0.04	

Table 6 Multiple linear regressions with DIs: Abundance, Abundance minus the most abundant sponge, *Q.* brevis, NTaxa, Simpson, Shannon, Evenness and Richness. Results show the most significant the best model after the removal of insignificant variables via stepwise function, and the relative importance of the variables remaining, which are highlighted. The variable with the highest relative importance is highlighted in yellow, and significance levels are show at $p < 0.05$ = **bold** and at $p < 0.1$ = *italics*.

Taxa	**Actual Contribution %**	**Number of Taxa in Fishing Effort Pairs**	
		None	Low
White round massive	0.054	2	0
cf Theneidae	0.044	2	1
Tethya/Craniella	0.039	3	5
cf Sycon	0.038	0	1
Stylocordyla borealis	0.035	9	10
Orange encrusting	0.028	10	1
		None	Heavy
Quasillina brevis	0.045	145	157
cf Theneidae	0.040	2	1
Cream thin encrusting	0.033	0	1
Cream thick	0.032	2	3
Yellow round massive	0.030	4	2
Stylocordyla borealis	0.029	9	4
		Low	Heavy
Yellow irregular	0.037	5	0
White round massive	0.036	0	7
Tethya/Craniella	0.035	5	18
Quasillina brevis	0.034	171	157
cf Theneidae	0.031	1	1
Lobate massive	0.030	17	33
Stylocordyla borealis	0.028	10	4

Table 7 SIMPER analysis output showing the taxa that most contribute to each fishing effort combination: none/low, none/heavy and low/ heavy. The highlighted cells indicate the top three most contributable taxa for that fishing effort group, within the combination comparison.

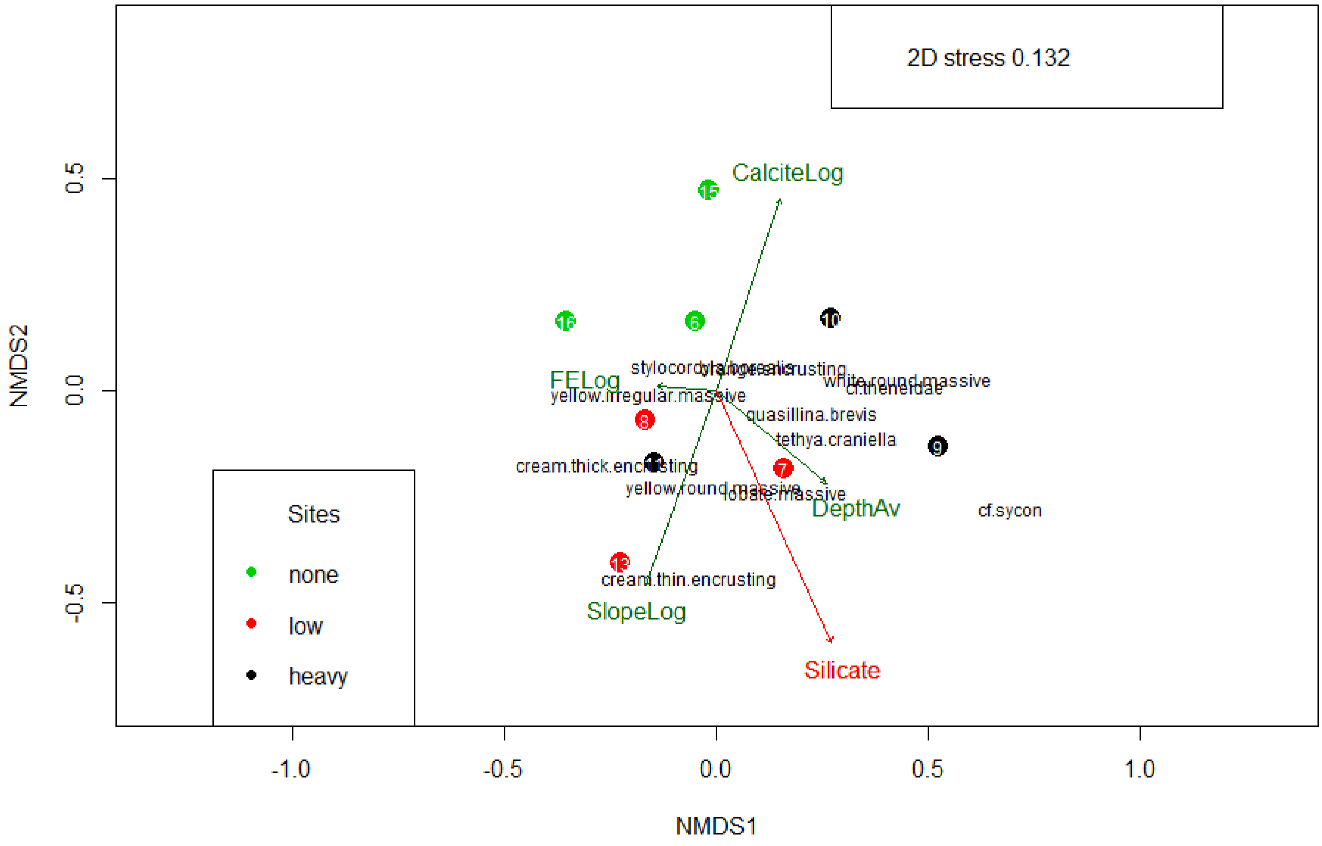


Figure 7 An nMDS ordination plot of all sponge taxa, using the most influential environmental variables that resulted from the stepwise regression (see Table 6). SIMPER analysis results of the top 6 taxa that most contribute to pairwise comparisons between fishing efforts are displayed on the plot. Stations (sites) are labelled in colour depending on different fishing efforts.

Discussion

This study presents the first attempt to describe the impacts of trawling on benthic sponge taxa in the Westfjord area of Iceland, as well reducing the knowledge gap for identification and description of sponge communities. It also adds to the extensive evidence in the literature suggesting negative associations between demersal trawling and benthic community abundance and diversity (e.g. Jennings et al. 2001).

Protective Measures

The sites selected for the 2011 benthic survey are situated in areas of varying protection and fishing effort. Such protection around Iceland currently exists not only to secure the sustainability of the fishing industry, but also to protect certain VMEs (Ministry of Fishing and Agriculture). Stations 15 and 16 are in one such MPA, where a ban on bottom and pelagic trawling has been established since 1978 (see Figure 8), in order to protect the juvenile red fish, *Sebastes sp.* (S. Olafsdottir, *pers. comms;* Jaworski et al. 2006). Such long-term protection measures appear to result in a higher abundance and diversity of sponge communities as well as size of individual taxa, compared to non-protected areas (Figures 3, 4; A-E, 5 and Table 4). Around Iceland, MPAs have not yet been established for the unique protection of sponge grounds; however sponge communities found in these areas seemingly thrive from their protection. This is supported by significant statistical differences between non- and heavily-fished sites, with a higher abundance, diversity and individual size in areas of low fishing. From the results, it could be concluded that sponges in fished areas may be inhibited from growing to larger sizes and have lower community diversity. The length of time an MPA has been established, and the intensity of its protection is suggested to affect the re-growth and re-colonisation period of the benthic organisms within (Freese 2001, Huvenne et al. 2016). In order to validate the success of MPAs in allowing recovery and re-growth of sponge communities around Iceland, examples showing results from sites varying in operation lengths would help to demonstrate the conservation status for individual areas (e.g. Syms & Carr 2001). Alternatively, the need for MPAs may not be necessary to reduce fishing effort, as station 6 and stations 7, 8 and 13 receive no protection although situated in non- and low-fished areas, respectively. The low cod abundance in these areas may have led to low fishing effort as fishers target larger populations in other areas (S. Olafsdottir, *pers. comms.*). Furthermore, fishermen often purposely avoid extensive sponge beds as they may cause damage to trawling gear; thus such ‘natural protection’ could be a result of the presence of the sponges themselves (Hogg et al. 2010). To understand further, fish stock comparisons of these areas must be considered.

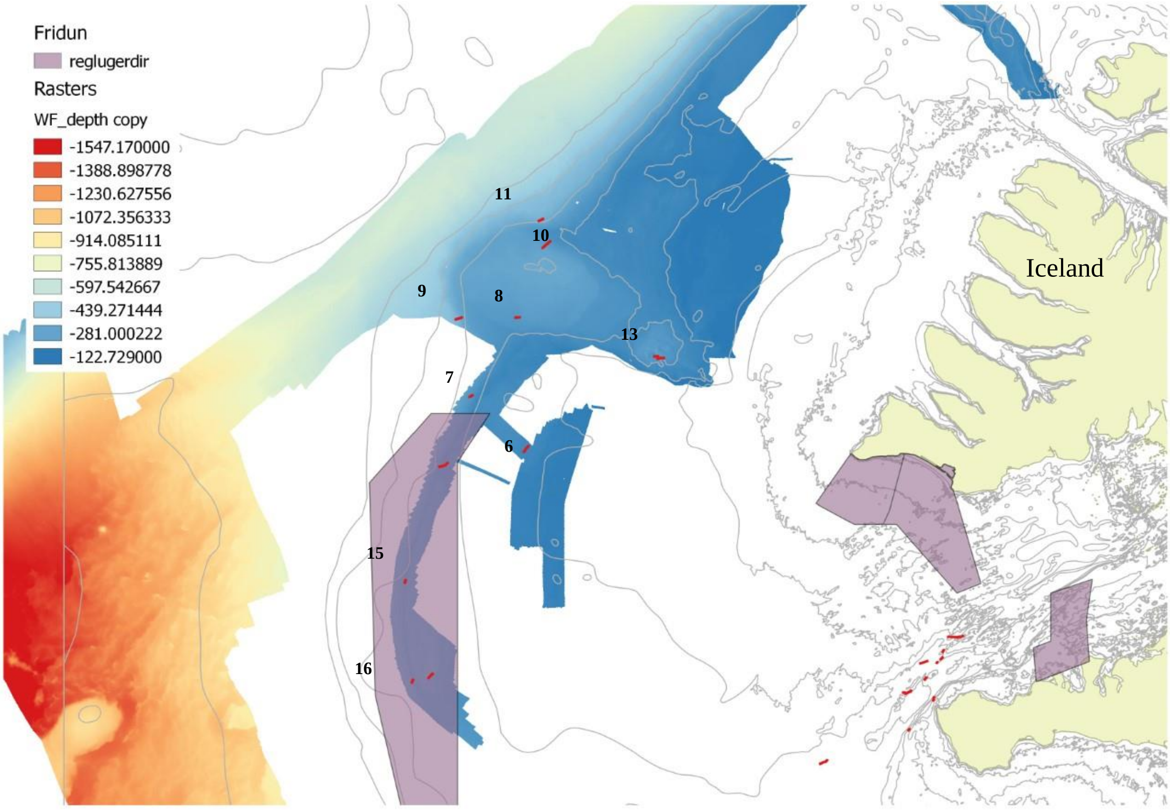


Figure 8 All 2011 benthic survey study sites off the coast of the Westfjords, Iceland. The stations chosen for analysis in this study are labelled with transect numbers. The purple polygon to the left shows long-term protection since 1978, for sites 15 and 16. Map courtesy of S. Olafsdottir and Dr J. Burgos, MFRI, Iceland, reproduced with permission.

Stations 9, 10 and 11 are situated along the shelf break towards the Denmark Strait, and although this area is subject to much trawling, the abundance of fish remains high enough that trawlers still persist (S. Olafsdottir, *pers. comms*, source: ELB data). This persistence of mature and spawning cod may be due to the warm Atlantic waters moving northwards, providing much productivity to the region (Begg & Marteinsdottir 2002). Temperature and salinity around the west of Iceland can be seen in Figures 1, A and B and in Figure 2, showing the Irminger Current flowing northwards from the Atlantic, along with environmental conditions that differ to the surrounding waters. The cold, polar waters of the East Greenland Current and the warmer, more-saline Irminger Current allow for high productivity around the shelf break where the heavily fished stations are situated (Palsson et al. 2012). These conditions provide suitable environments for species like cod, which require a mixture of conditions for their different life stages (Begg & Marteinsdottir 2002). However, the persistence of cod in areas lacking sustainable protection measures may result not only in the over-exploitation of fish stocks, but also of non-target benthic communities like sponges.

Sediment Types

Sponges are obligate, sessile organisms that thrive in areas where geological features allow sufficient anchorage to the seabed, for example, on boulders and cobbles in iceberg plough-marks along the seafloor (Conway et al. 2005; Freiwald & Roberts 2006). In this study, sediment types of the low- and non-fished sites were softer, and included features like cobbles and boulders, rather than the small cobbles and mixed gravel substrate seen in the heavy fished sites (see Figure 6 and ST3). When analysed, statistically significant differences were seen between the non- and heavy-fished sediment types. Large boulders and cobbles provide attachment to large taxa and those with a high surface area, for example, Axinellidae and cf *Clathria barleei.* This could substantiate the low abundance of these taxa in heavily-fished sites, as trawling can remove both large sponges and the stones they attach to (Freese 2001). Furthermore, SIMPER analysis shows that small, rounder sponges are more important in classifying areas of high fishing effort in comparison to both lower-fished areas (Figure 5, Table 7). For example, the presence of small cobbles in heavily fished areas suggests that flatter sponges like the “Lobate Massive” morphotype may be able to survive, whereas more upright sponges, like *S. borealis*, may not. In this study, ANOVA results allude to this species-specific relationship with sediment type, with significant differences for both abundance and diversity of sponges, the latter showing stronger significance (Figure 6). Although much research demonstrates the dependency of sponge distributions to certain sediments, critical gaps still exist in both the initial identification of sponge taxa and of the physical and adaptive responses to their different sediment types (Bell et al. 2015).

Anomalous Taxa

With more evidence on the cause-effect relationship between substrate type and fishing effort, it may be possible to determine the sediment preference for some sponge species, for example, *Q. brevis*, due to its high abundance in areas of high fishing. Few studies show the ecology and community dynamics of this sponge (e.g. Plotkin 2004). However, knowledge gaps still persist, particularly around Iceland, and therefore only assumptions can be made to explain its high abundance. Certain ecological characteristics may enable *Q. brevis* to be well suited to these areas. For example, its small height (up to 5.5 cm), and small, stalked holdfast could enable attachment to smaller sediment types like gravel (see Table 3 for image examples), (Bowerbank 1861c; Kenchington et al. 2015). The presence of this species in highly fished sites suggests *Q. brevis* may thrive in sites of intense trawling where disturbance may select for such successful species, leading to domination by communities of rapid colonisers (Dayton et al. 1995; Pisera et al. 2010). *Q. brevis* is highly abundant not just in heavily-fished sites, but across all fishing efforts, with station 13 (low-fished) being the only station where it is absent (Figure 3). Station 13 differs from other low-fished stations as it contains a higher abundance of larger taxa (>10 cm), and higher diversity of taxa overall (Figures 4; A-E, Figure 5). Thus reasons for the absence of *Q. brevis* may include visual obscuring of the smaller sponge by the larger taxa when analysing images, or a possible out-competing by the higher diversity of larger taxa present.

Analyses supports the importance of *Q. brevis* in influencing overall abundance, as instead of fitting trends observed in other research (i.e. where an increase in fishing effort is negatively correlated with species abundance, e.g. Althaus et al. 2009) a significant difference is observed between none and low fished sites instead. This observation is further supported when *Q. brevis* is removed from the overall abundance data, as results show a stronger significant difference between none and high-fishing effort (Table 6). Therefore, removing *Q. brevis* from total abundance data suggests that as fishing effort increases, the abundance of other sponge taxa decreases, fitting the hypothesis (see Figure 3). However, diversity measures Simpson and Shannon, which are based on a balance of abundance and taxon diversity, are not influenced by such abundant taxa and negatively correlate with increased fishing effort.

Environmental Impacts

Other factors apart from fishing effort may impact sponge community composition, as studies imply how abiotic and biotic factors are significant in dictating sponge distributions (Strekal and McDiffett 1974; Wulff 2012; Powell et al. 2014). Environmental data were integrated into analysis to observe how external factors may impact diversity of different morphology types. According to significant results from the MLR analysis, silicate seems to be an important environmental factor in influencing sponge total abundance and number of taxa. As 75% of sponges use dissolved silicon (DSi) to build up their siliceous skeleton, it is likely this variable is important (Maldonado et al. 2011). Results from nMDS analysis demonstrate this, indicating that silicate has a greater impact on sponge community composition than fishing effort does, more closely ordinated around trawled sites relative to non-trawled sites. This phenomenon may be due to the release of silica-containing sediment into the water column from trawling disturbance along the benthos (Cooper 1952). However, a mixture of currents resulting in the highly productive waters west and south of Iceland could also be an important factor. Studies suggest that some sponges, in particular *Axinellidae,* can take in levels of DSi much higher than oceanic concentrations (Maldonado et al. 2011). However, the same study suggests that one possible reason that they do not reach maximum uptake efficiency in marine ecosystems is due to the presence of silica-containing diatoms. Sponges are outcompeted by diatoms due to the persistence of a relatively inefficient, ancestral, uptake system in the sponge anatomy (Maldonado et al. 2011). On the west of Iceland, spring and autumn blooms bring in high abundances of phytoplankton, mainly dominated by such diatoms (DU CIEM 2008; Palsson et al. 2012). This primary productivity is highly correlated to the rich nutrient availability in the western and southern waters around Iceland, owing much to the currents flowing up from the Atlantic waters (Stefánsson & Ólafsson 1991). As the shelf break provides conditions promoting this productivity, there could be favourably high salinity levels causing high abundances of competitive silica-containing diatoms. Therefore, lack of successfully competitive, silica-containing sponges may explain the low diversity and abundance of sponges at the heavily fished sites. To test this argument, further analysis of organic compounds that are used in forming the skeletal spongin fibres within sponges could be performed across observed taxa.

Overall, when analysed with environmental predictor variables, fishing effort was the most important factor determining the overall abundance, NTaxa and diversity measures Simpson and Shannon. In addition, slope shows high significance as a predictor variable of sponge diversity (Table 6). Previous research shows that community diversity may be affected by slope due to substrate preferences, with those organisms requiring soft sediment attachments potentially not persisting on steeper gradients where bedrock is exposed (Consalvey et al. 2016). Environmental variables are therefore important in determining sponge community composition; however, fishing effort still has the greatest significant impact.

Future Work

Few studies suggest cause-effect relationships of fishing on sponge community composition, and although this study did not demonstrate this relationship, it aimed to explore any correlations between potential explanatory variables surrounding this question, to provide a better understanding for future work. Thus in order to improve this study, an increased number of samples across a wider range of fishing efforts would be necessary to further understand the impact of fishing effort on sponge community composition. Following this, research into substrate dependency by means of grab-sampling would enhance our understanding of species-specific choice of sediment type, as substrate may be acting as a confounding factor between sponge composition and fishing effort. There is also a need to measure biological and ecological interactions of individual sponge taxa and communities in areas of high fishing, to best inform ongoing benthic conservation work as well as sustainable fishery management. Finally, as sponges are largely ill-defined due to difficulties in taxa identification, the use of new techniques such as DNA barcoding would improve the taxonomic library for this phylum (Voigt and Wörheide 2016).

Conclusion

This study demonstrates that heavy trawling significantly impacts the sponge communities in the western seas around Iceland, showing a lower diversity of sponge taxa and smaller size of sponges in higher fished sites. However, in areas where sponge taxa are more diverse, fishing effort is minimal due to either protection measures or lack of fish stock, thus providing good controls for this study. Overall abundance, on the other hand, is similar in non-fished and heavily-fished sites. This is due to the high abundance of certain species like *Q. brevis,* which may be successful in these disturbed environments, and when removed from analyses, the impacts of heavy fishing effort are seen to be adverse to the abundance of other sponges. This study supports current knowledge of the impacts that bottom-trawling has on certain benthic species and communities, and highlights how taxa can respond in unexpected ways to heavy trawling. Our research also highlights the benefits of habitat protection, and encourages further research to consider the establishment of MPAs around diverse and abundant sponge beds in the Iceland Sea. However, it is important to stress the uncertainty of considering cause-effect relationships between sponge abundance and fishing effort, with further work required to understand the influences of other potentially impacting biotic and abiotic factors. This study also helps reduce a knowledge gap on the presence of sponge individuals and communities around Iceland, contributing to the on-going benthic habitat-mapping program run by the Marine and Freshwater Research Institute (MFRI) of Iceland.

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Appendices

Supplementary Tables (ST)

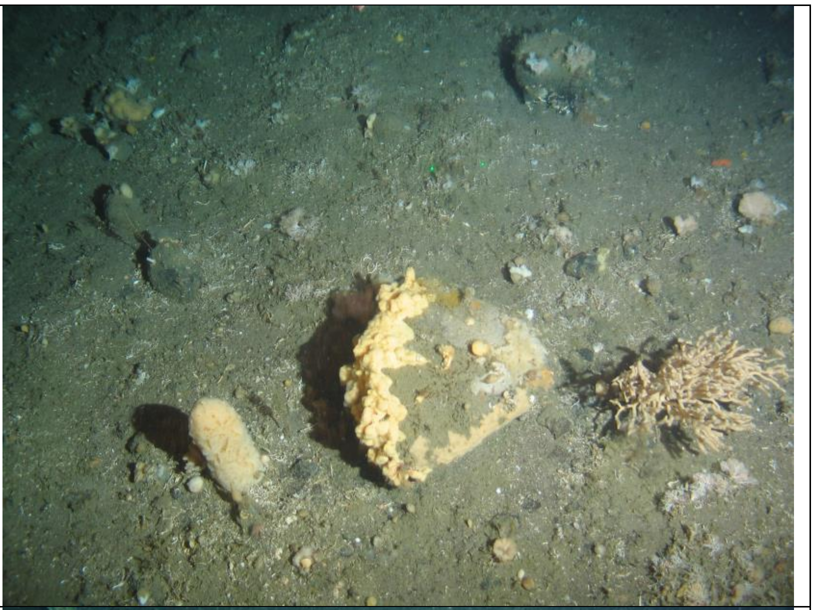
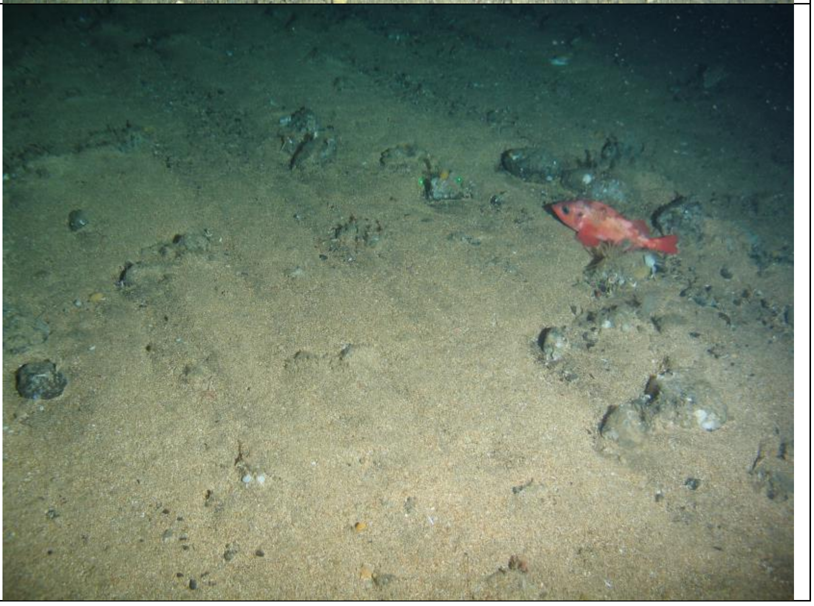
Station	**6**	**7**	**8**	**9**	**10**	**11**	**13**	**15**	**16**
Aplysilla sulfurea	0	0	0	0	0	0	0	1	0
Asbestopluma pennatula	8	1	6	0	1	5	0	6	3
Asconema foliata	0	0	0	0	0	0	0	0	11
Axinellidae	53	2	7	2	2	7	4	6	7
Barrel massive	0	0	0	0	0	0	1	0	0
Brown massive	0	0	0	0	0	0	2	0	0
cf *Antho dichotoma*	7	7	1	2	27	0	5	3	29
cf *Clathria barleei*	2	0	1	0	0	0	0	5	1
cf *Craniella zetlandica*	7	6	5	4	7	5	3	7	5
cf *Geodia barretti*	1	0	0	0	0	0	0	0	0
cf *Haliclona* encrusting	0	0	0	0	0	0	0	3	0
cf *Haliclona* massive	5	0	0	1	1	0	0	5	2
cf *Hamacantha papillata*	0	0	1	0	0	0	0	0	0
cf Polymastiidae green	0	0	0	0	0	0	0	13	0
cf *Spongionella pulchella*	0	0	2	0	0	0	0	0	1
cf *Suberites carnosus*	0	0	1	0	0	0	0	2	1
cf Suberites encrusting	2	1	1	0	1	5	1	3	9
cf Suberites massive	2	0	0	0	0	1	0	0	3
cf Theneidae	1	1	0	1	0	0	0	1	0
Clathrina lacunosa	0	0	0	0	0	0	0	1	0
Cream encrusting projections	0	0	0	0	0	0	0	2	0
Cream irregular massive	3	1	2	0	3	1	2	4	
Cream massive	6	0	1	0	0	1	2	2	4
Cream thick encrusting	0	0	3	0	1	2	3	0	2
Cream thin encrusting	0	2	0	0	0	1	2	0	0
Cream upright	0	0	0	0	0	0	0	1	4
Hymedesmia paupertas	2	4	0	0	1	3	0	2	26
Irregular massive vase	0	0	0	0	0	4	0	0	1
Lobate massive	0	8	3	4	7	22	6	0	0
Mycale lingua	26	2	4	0	0	6	23	7	6
Orange encrusting	1	1	0	2	1	7	0	6	3
Pink encrusting	0	0	0	0	2	0	0	0	0
Purple encrusting	0	0	0	0	0	1	1	0	1
Quasillina brevis	89	86	85	101	24	32	0	8	48
Semisuberites cribrosa	0	0	0	0	0	0	1	0	0
Stalked	0	0	0	0	0	0	0	0	1
Sycon sp	0	1	0	1	0	0	0	0	0
Stylocordyla borealis	0	4	5	0	4	0	1	5	4
Tethya/Craniella	1	0	3	10	5	3	2	0	2

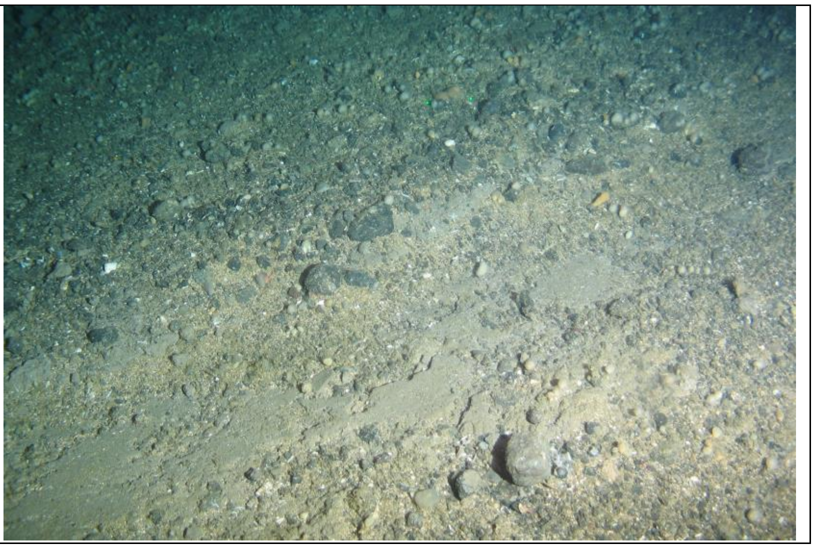
White branching massive	1	0	0	0	1	0	0	1	0
White full vase/leaf	0	0	1	0	0	0	0	0	4
White irregular massive	1	2	4	4	1	5	4	1	6
White irregular projections	0	0	0	0	0	0	1	0	0
White lattice vase/leaf	2	2	0	0	0	0	0	0	6
White massive	2	1	6	2	2	2	4	1	1
White round massive	0	0	0	4	2	1	0	1	1
White thick encrusting	0	0	1	0	2	0	0	0	0
White thin encrusting	0	1	5	0	0	5	2	0	1
Yellow irregular	0	0	0	0	0	0	0	11	2
Yellow irregular massive	3	1	0	0	0	0	4	3	1
Yellow massive	0	0	5	0	1	3	1	6	0
Yellow projections	2	1	0	0	0	0	0	0	10
Yellow round massive	3	2	2	1	0	1	5	0	1
Yellow spiked	0	0	1	0	4	0	0	0	0
Yellow thick encrusting	0	0	2	1	1	0	0	2	0
Yellow thin encrusting	5	0	1	3	4	1	7	3	2

ST1 Overall taxa abundances, including morphotypes, for each station.

>10cm in height	**Thin Encrusting**	**<9cm in height**
Asbestopluma pennatula	Brown massive	*Aplysilla sulfurea*
Asconema foliata	cf *Haliclona* massive	cf *Craniella zetlandica*
Axinellidae	cf *Suberites* encrusting	cf *Haliclona* encrusting
Barrel massive	Cream thin encrusting	cf *Hamacantha papillata*
cf *Antho dichotoma*	*Hymedesmia paupertas*	cf Polymastiidae green
cf *Clathria barleei*	Lobate massive	cf *Spongionella pulchella*
cf *Geodia barretti*	Orange encrusting	cf *Suberites carnosus*
Cream irregular massive	Pink encrusting	cf *Suberites* massive
Irregular massive vase	Purple encrusting	cf Theneidae
Mycale lingua	White thin encrusting	*Clathrina lacunosa*
Semisuberites cribrosa	Yellow thin encrusting	Cream encrusting projections
Stylocordyla borealis		Cream thick encrusting
White branching massive		Cream massive
White full vase/leaf		Cream upright
White irregular massive		*Quasillina brevis*
White irregular projections		Stalked
White lattice vase/leaf		*Sycon sp*
Yellow irregular		*Tethya/Craniella*
Yellow irregular massive		White massive
Yellow projections		White round massive
		White thick encrusting
		Yellow massive
		Yellow round massive
		Yellow spiked
		Yellow thick encrusting

ST2 Taxa categorized into three sizes based on calibrated images, general morphotypes, and online identification portals.

Muddy-Sand, Cobbles & Boulders	
Mixed, Cobbles & Boulders	
Muddy-Sand, Small Cobbles	

Mixed, Small Cobbles	

ST3 Example images of sediment samples.

Station	Start Time	End Time	Duration (min)	Distance (m)	Speed (m/s)	Start Coordinate	End Coordinate
6	19:35	20:44	69	2032	29.44927536	65.52160, -26.3653	65.50752, -26.3907
7	22:55	23:39	44	1148	26.09090909	65.64310, -26.68285	65.63925, -26.69845
8	01:52	02:23	35	2185	62.42857143	65.82848, -26.41528	65.82758, -26.43875
9	03:58	05:06	68	1592	23.41176471	65.82778, -26.74315	65.82370, -26.77690
10	08:03	09:06	63	2778	44.0952381	66.00777, -26.24217	65.99233, -26.28433
11	10:41	11:42	61	1407	23.06557377	66.06177, -26.27772	66.05588, -26.30548
13	17:15	18:49	94	1629	17.32978723	65.73197, -25.59723	65.73008, -25.63118
15	03:49	04:11	82	755	9.207317073	65.19820, -27.06190	65.19183, -27.06607
16	06:15	06:44	29	790	27.24137931	64.95498, -27.01998	64.94847, -27.02602

ST4 Transect information for on-board data collection at each analysed station.

HAFRANNSÓKNASTOFNUN

Rannsókna- og ráðgjafarstofnun hafs og vatna