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Response of Benthic Fauna to Merchant Vessel Anchoring in the Point Lynas Anchorage, Anglesey

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Response of Benthic Fauna to Merchant Vessel Anchoring in the Point Lynas Anchorage, Anglesey

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Declaration

I hereby declare that this thesis is the results of my own investigations, except where otherwise stated. All other sources are acknowledged by bibliographic references. This work has not previously been accepted in substance for any degree and is not being concurrently submitted in candidature for any degree unless, as agreed by the University, for approved dual awards.

1 ABSTRACT

Merchant vessel anchoring affects seabed habitats worldwide; however, impacts are not well understood. The effects of anchoring are unlikely to be uniform and likely to vary greatly with habitat type, natural disturbance regime, and spatial and temporal variations in anchoring pressure exerted across an area. It is concerning that studies of anthropogenic disturbance of seabed habitats neglect anchoring activity in their assessments, as abrasion of the seabed has been shown to result in negative community effects and altered biogeochemistry. Understanding the role of merchant vessel anchoring in sediment habitats is fundamental, because the frequency at which they anchor during routine operations, in combination with the size of the anchor gear used for these vessels, may result in significant disturbances in benthic communities.

Observations of abundance, biomass, and species richness of benthic invertebrates were collated from stations of differing anchoring pressure (0.0 – 45.38472 tons$^{2/3}$ hours/m$^{2)}$ using day grab samples, beam trawls, and seabed images from the study site at Point Lynas anchorage off the northeast coast of Anglesey, UK. Anchoring history of the seabed was determined using terrestrial Automatic Identification System (AIS) data purchased from Marine Traffic in conjunction with inclusion criteria to identify anchored vessels and distinguish separate anchoring events. Biodiversity metrics of abundance, biomass, and species richness were broadly consistent between replicates, regardless of anchoring history. Anchoring disturbance did not lead to statistically significant changes in the community composition of benthic infauna or epifauna indicating that any long-term chronic effects of anchoring disturbance were not detectable against the background of high levels of natural disturbance that are found in Red Wharf Bay or that the community is already adapted to disturbance. Therefore, the fragmented and sporadic nature of the anchoring activity at Point Lynas anchorage does not elicit a significant effect in the community. The results presented in this study highlight that the effects reported in experimental studies on anchoring and other abrasive activities, such as trawling cannot be easily extrapolated to explain the

ecosystem-level effects of merchant vessel anchoring. Discrete cumulative effects of anchoring in soft sediment communities may emerge only when anchoring disturbances are investigated over greater spatial and temporal scales and across a gradient of natural disturbance.

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3 KEY WORDS

Anchoring, benthic community, disturbance, infauna, epifauna

4 INTRODUCTION

There are 100,000 merchant vessels trading internationally (Statista 2021) transporting approximately 90% of global trade, the volume of which is estimated to double by 2030 (Lloyds Register 2019).

Anchoring is a routinely used practice for merchant vessels, for safety (United Nations 1982), maintenance, and supply chain management (Broad et al. 2020). Deployment and subsequent retrieval of the anchor are likely to cause significant disturbances. During anchoring, the ship moves such that the current provides a horizontal force. This provides the required drag on the shoulders of the anchor, causing the flukes to push downward into the seabed. This is assisted by operating the engine in an astern direction to lay out the chain as the flukes dig further into the seabed, as illustrated in figure 1 a & figure 1 b. The combined weight of a vessel's anchor, and curved section of chain hanging down from the vessel, known as the catenary, maintains the position of the vessel at anchor, as shown in figure 1 c, thus more laid out chain results in greater holding power (Alandia 2022). The scope is the length of chain deployed to allow the vessel's position to be maintained accounting for changes in wind, waves, and tides, the scope is generally 5-10 times the depth of the water as this allows a horizontal force to be maintained on the anchor (Cult of Sea 2022). In good weather the optimum position will be with the chain appearing to go straight down from the bow of the ship, as shown in figure 1 c.

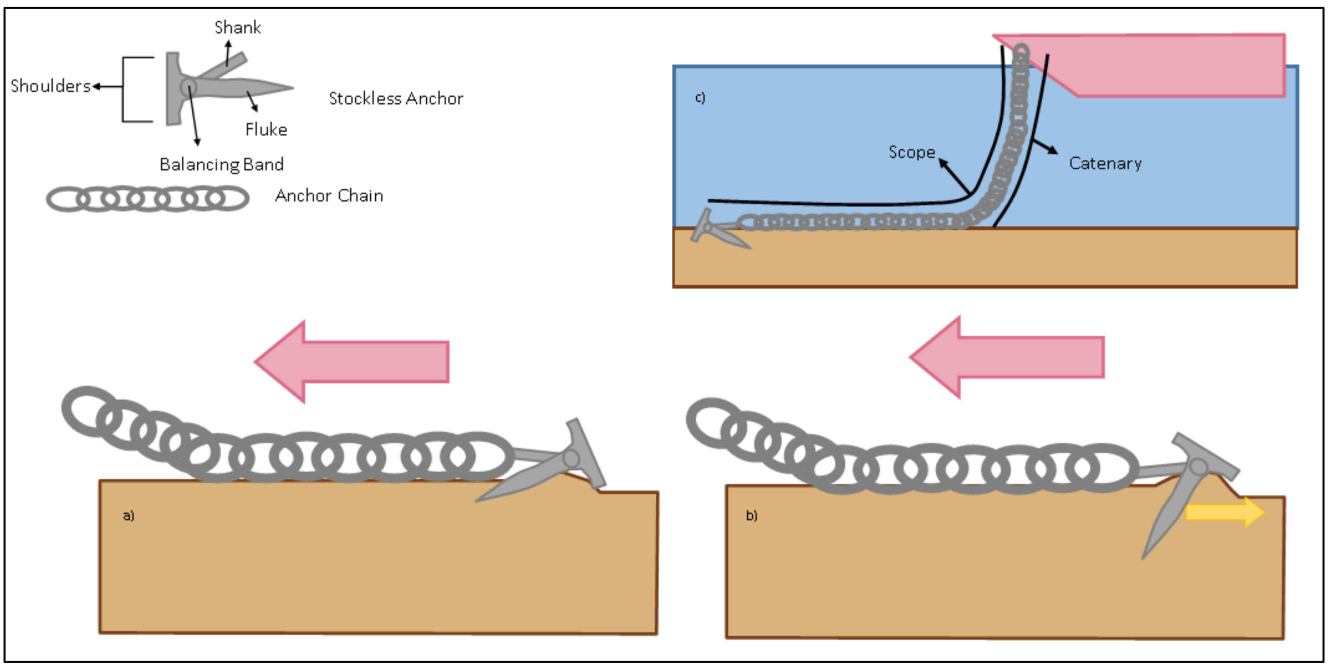


Figure 1. Schematic illustrating setting of anchor and the maintenance of vessel position by anchor (This infographic uses information gained from (Alandia 2022).

When at anchor, vessels do not remain stationary. The effects of the tides and shifting winds cause vessels to move in an arc around the anchor point, as illustrated in figure 2. As this action occurs, the anchor and chain swing around the anchor point, which creates furrows and ridges in the seabed and produces what is known as anchor scour (Davis et al. 2016; Tinsley 2021). These marks on the seafloor caused by vessels at anchor have been termed anchor scars and have been likened to trawl marks made by fishing trawlers (Abadie et al. 2016; Ganteaume et al. 2016; Broad et al. 2020). Increased anchoring of large vessels during the Covid-19 pandemic has shown that large vessels at anchor can leave significant anchor and chain scars (Tinsley 2021)

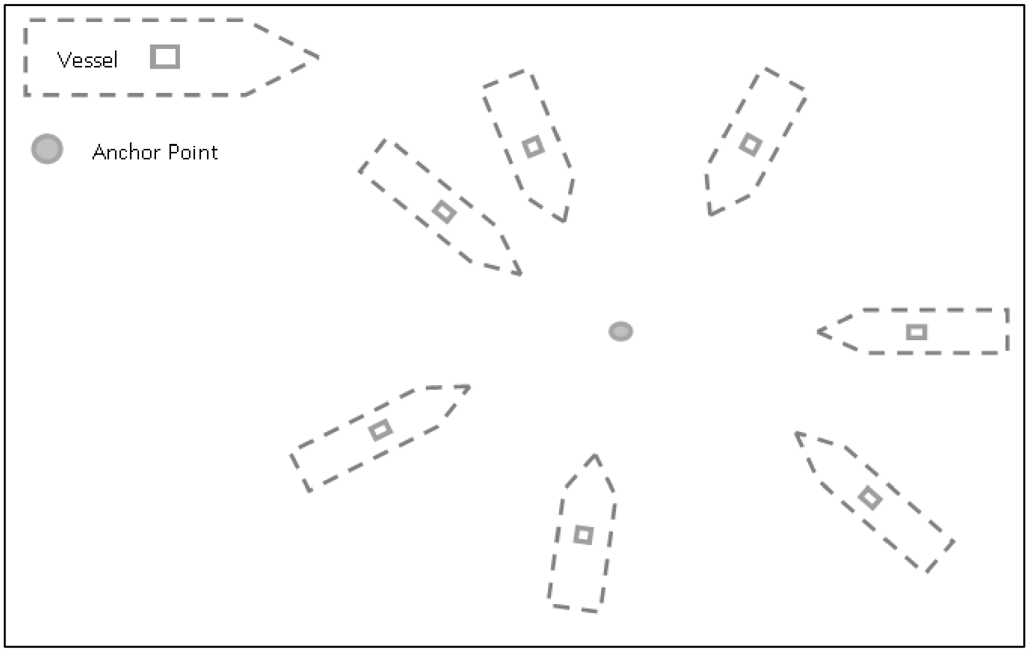


Figure 2 - schematic illustrating how a vessel at anchor may move in an arc around the central anchor with the wind and tides.

4.1 Effects of Seabed Disturbance

It is well understood that disturbance plays a significant role in shaping ecological communities (Pickett et al. 1999). Seabed habitats are the basis for marine life, with perhaps as much as 90% of the ocean's biodiversity closely associated with benthos (Broad et al. 2020). Exploitative human activity can have negative consequences for marine habitats (Rossi 2013); therefore, it is imperative to identify, monitor, and mitigate these impacts if marine habitats are able to remain healthy and productive.

Previous studies have found that mechanical disturbance of the seabed causes sediment resuspension (Churchill 1989), remineralisation of nutrients and contaminants (Kaiser et al. 2010) and particle size distribution changes (Palanques et al. 2014; O'Neill and Ivanović 2016). Disturbance can lead to reduction of the topographic complexity of seabed habitats and removal of biogenic structures resulting in community changes (Riesen and Reise 1982; George and Warwick 1985; Kaiser et al. 2010) with long recovery times (Beukema 1995; Hall-Spencer and Moore 2000). Community changes such as reduction of biomass (Collie et al. 2000b; Jennings et al. 2001; Duplisea et al. 2002), reduction of diversity (Dayton et al. 1995; Collie et al. 1997), and selection for communities dominated by fauna with rapid life histories (Kaiser et al. 2006; van Denderen et al. 2015). Cumulative effects of disturbances lead to changes in trophic structure, function, and production (Duplisea et al. 2002; Hiddink et al. 2016).

It has been shown that the extent of impacts of disturbance correlates with the penetration depth into the sediment. For example, the effect of bottom fishing on benthic communities is strongly gear specific (Sciberras et al. 2018) with one trawl pass leading to 19% reduction in species richness and 26% reduction in invertebrate abundance, all with a mean penetration depth of only ~5cm. This is a significant disturbance, with a reported recovery time of more than three years (Sciberras et al. 2018).

4.2 Disturbance by Anchoring Vessels

Commercial Vessels typically use stockless anchors, and the size of anchor and length of chain is determined by the vessel's equipment number. Equipment number is a function of the vessel displacement$^{2/3}$, the frontal, and lateral (wind exposed) areas of the ship (Luger and Harkes, 2013; Veritas, 2010a; DNV, 2017). Provided that the shape of the vessel does not change, the anchor mass will be proportional to the tonnage$^{2/3}$ (Luger and Harkes 2013). For example, a Suezmax tanker will have a length of roughly 275 metres stipulated as per the Suez Canal passage requirements (Marine Insight 2021) and a deadweight of 120,000 – 200,000 tons. The Equipment number for a Suezmax vessel was calculated to be 5444 (Oh et al. 2020) which would require an anchor of 16,100kg and an anchor chain length of 742.5 metres**,** with a weight of approximately 286kg per metre as per published equipment number tables (Pilot Fits Engineering 2020).

The penetration depth of the anchors and chains depends on the weight of the anchor gear and seabed characteristics. Recreational anchors have been shown to produce anchor pits of up to 18 cm from an anchor of only 20 kg (Backhurst and Cole 2000; Collins et al. 2010), with greater depths occurring in seagrass habitats. Anchors on the largest merchant vessels can weigh more than 25t and the individual links in their chain may weigh up to 200kg per meter (House 2002) and consequently the penetration of anchors and chains of merchant vessels is much deeper and are likely to cause significantly more damage. The penetration depth of merchant vessel anchors has been shown to reach 6m in unconsolidated sediments (Broad et al. 2020).

Empirical studies that explore the effects of anchoring by merchant vessels in sediment habitats are scarce, with most current research focusing on recreational vessels in seagrass and coral habitats. A systematic review undertaken by Broad et al (2020) showed that studies of merchant vessel anchor interactions with the seabed focussed almost exclusively on tropical coral reef habitats. Broad et al (2020) found just one study investigating temperate seagrass and none on the most wide-spread sedimentary habitats. This scarcity of research into the effects of merchant vessels anchoring on offshore sediments is a fundamental oversight. Merchant vessel anchoring will likely account a significant amount of disturbance, due to the scale of the global merchant trade, the size of vessels and the intensity of the disturbance (Deter et al., 2017; DNV, 2010b; Zhuang et al., 2016). Disturbance at this scale may perpetuate significant and cumulative ecological impacts on the seabed, such as that which occurs when trawling occurs repeatedly on the seabed (Poiner et al. 1998; Drabsch et al. 2001).

4.3 ESTIMATING SCALE OF ANCHORING DISTURBANCE

It is a substantial concern that studies aimed at estimating the cumulative impacts of human pressures on seabed habitats have not included anchoring activity in their assessments of activities that cause abrasion of the seabed. In 2011, it was estimated that 52.20% of the UK seabed was abraded by benthic fishing alone, in a study that only considered benthic fishing, wind farm scour pits, and submarine cable burial in its assessment of abrasive activity (Foden et al. 2011). Benthic fishing is the most prevalent human activity effecting continental shelves (Eigaard et al. 2017; Sciberras et al. 2018) and it has significant negative effects on benthic ecosystems (Dayton et al. 1995; Hiddink et al. 2016, 2017; Sciberras et al. 2018). The effects of bottom trawling are strongly gear specific (Sciberras et al. 2018), similarly the effects of anchoring activity depend significantly on the size of anchor gear used, the frequency of anchoring activity (Backhurst and Cole 2000; Collins et al. 2010) and the seabed substrate (Broad et al. 2020). Anchoring activity is limited to a shallower depth range (10-80m) than most bottom trawlers but takes place more frequently and has greater penetration into the seafloor (Broad et al. 2020; Moore et al. 2021; Watson et al. 2022). No UK-wide estimate of the size of the footprint of anchoring activity has been made with studies focusing on individual Marine Protected Areas (Griffiths et al. 2017). However, a New Zealand study produced the first estimate of the volume of seabed sediment displaced by one high-tonnage vessel at the anchor to be 2,733 m^3, which would more than fill an Olympic-sized swimming pool (Watson et al. 2022).

4.4 EFFECTS OF ANCHORING BY SEABED HABITAT

4.4.1 Anchoring on seagrass habitats

Scour by anchor gear directly damages seagrass fronds (Hastings et al. 1995), uproots seagrass shoots (Milazzo et al. 2004) and creates anchor pits (Collins et al. 2010). reducing seagrass cover (Okudan et al. 2011; Colomer et al. 2017; Mishra et al. 2020) and impacting associated fauna (Collins et al. 2010), reducing species richness (McCloskey and Unsworth 2015), reducing chlorophyll production and increasing susceptibility to disease (Williams 1988). Studies show anchor scour sites display reduced species diversity and relative abundance for infauna (Collins et al. 2010), epifauna (Hendriks et al. 2013; Deudero et al. 2015; Vázquez-Luis et al. 2015), and fishes (Lanham et al. 2018). However, it can be difficult to ascertain whether the changes observed between sites result directly from anchor gear damage to organisms or from subsequent environmental changes, such as changes in sediment particle size composition (Collins et al. 2010).

4.4.2 Anchoring on Coral Reefs

Anchor damage on coral reefs has been shown to severely affect associated communities by removing and damaging sessile biota, reducing topographical complexity, and reducing species diversity (Rogers and Garrison 2001a; Dinsdale and Harriott 2004; Beeden et al. 2014; Giglio et al. 2017; Flynn and Forrester 2019), with limited recovery over the subsequent decade (Rogers and Garrison 2001b; Forrester et al. 2015). The low survival of coral recruits and fragments indicates that anchor scar sites, which consist of unconsolidated rubble, mobile sediment, and high macroalgal cover (Rogers and Garrison 2001b), are unsuitable for recolonization (Highsmith 1982). (Highsmith, 1982). Frequently anchored sites have been shown to support half as many fish as low anchor areas (Flynn and Forrester 2019) and accidental cruise liner anchoring events have been shown to be related to a 30% reduction in fish diversity and 95% reduction in abundance (Forrester et al. 2015).

4.4.3 Anchoring & Mooring on Maerl Beds

Maerl beds disturbed by moorings are associated with reduced species richness and reduced abundance of macroalgae, invertebrates, fish, and invertebrates (Gabara et al. 2018). Intense crushing of rhodolith thalli, such as that caused by anchoring and mooring, leads to mortality and habitat degradation, reducing net productivity and increasing respiration rates (Dolinar et al. 2020; Kim et al. 2021)With continued disturbance, these habitats will be replaced by sand owing to extremely slow growth rates of < 1 mm/yr (Blake and Maggs 2003). The loss of these habitats is a concern because the biodiversity in these areas can be twice as high as that of nearby sediments (Steller et al. 2003). Therefore, they have been identified as non-renewable resources that are threatened by anthropogenic activities (Nelson 2009; Basso et al. 2016).

4.4.4 Anchoring on Unvegetated Sediments

The published literature presents a substantial lack of research being undertaken on unvegetated soft sediments which are favoured sites for anchoring or mooring as they represent the majority of estuarine and marine habitats (Wilson 1990), and provide greater holding than other kinds of seabed for anchoring vessels (Broad et al. 2020). Anchoring mobilises sediments and chain swing causes sediment winnowing and shift towards larger particle sizes (Herbert et al. 2009; Hedge et al. 2017; Macolino et al. 2019). Increased suspended sediment negatively affects filter-feeders, decreases pumping rates, and can cause direct mortality of bivalves and sponges (Wilber and Clarke 2001; Grant et al. 2019). Direct contact with anchor gear can crush or damage infaunal biota inhabiting the top layer of sediment (Byers and Grabowski 2014) causing mortality or increasing vulnerability to predation and disease (Backhurst and Cole 2000; Vázquez-Luis et al. 2015). The removal of these organisms can have significant negative impacts, as the ecological processes carried out by organisms are lost. Such as the contribution to sediment stability provided by soft-sediment organisms that ingest and excrete sediments and mucus to support their burrows (Rhoads 1974; Taghon 1982; Self and Jumars 1988). Unconsolidated sediments provide habitats for extensive infaunal assemblages, that naturally display high levels of temporal and spatial variation even over small scales. In addition, anchoring activity can be sporadic across areas, and consequently, the impacts of disturbances, such as those caused by anchor gear, can be difficult to detect and quantify (Thrush 1991; Drabsch et al. 2001). Therefore, using fine spatial scales can be important when investigating the impacts of anchoring and mooring on these communities (Macolino et al. 2019) to detect and quantify their effects on sediment communities.

It can be difficult to accurately assess the impacts of anchoring and recovery because there are very few suitable sites where anchoring has not already been going on for a considerable amount of time. Therefore, a shift in the ecological baseline has probably already occurred (Pauly 1995). Rapid growth rates of infaunal biota (Haig et al. 2012) and apparent short recovery times could indicate genuinely rapid recovery after disturbance (Backhurst and Cole 2000) however, whether this “recovery” is to the same pre-disturbance community composition can be difficult to ascertain without pre-disturbance assessment data.

4.5 RESEARCH AIMS

This research provides vital knowledge to fill gaps evident by focusing on the impacts of merchant vessel anchoring at depths of 25.5-35m in sedimentary habitats, rather than recreational vessels in coral or seagrass habitats in the shallows (<10m), which dominates the published literature in this area. It is important to investigate the impacts of anchoring by merchant vessels, as the results of published studies on recreational vessels suggest that anchoring by merchant vessels has a dramatic impact.

The fundamental aim of this research is to gain a comprehensive understanding of the cumulative ecological effects of repeated disturbances by the anchoring activity of merchant vessels on soft-sediment communities at Point Lynas anchorage. It aims to establish whether anchoring pressure affects biomass, species richness, and abundance of benthic infauna and epifauna. The hypothesis that will be tested is that repeated mechanical disturbance by anchor scour of merchant vessels disrupts benthic communities, resulting in significant simplification of communities at high anchoring intensity compared with communities that are not impacted by anchoring pressure.

5 METHODOLOGY

In June 2021, infauna and epifauna were sampled using day grab samples, seabed imagery, and beam trawls to assess the abundance, biomass, and species richness of the study area. Community data were compared with data related to anchoring activities within the study area. Methodology for the mapping of anchoring polygons is outlined in section 5.3.

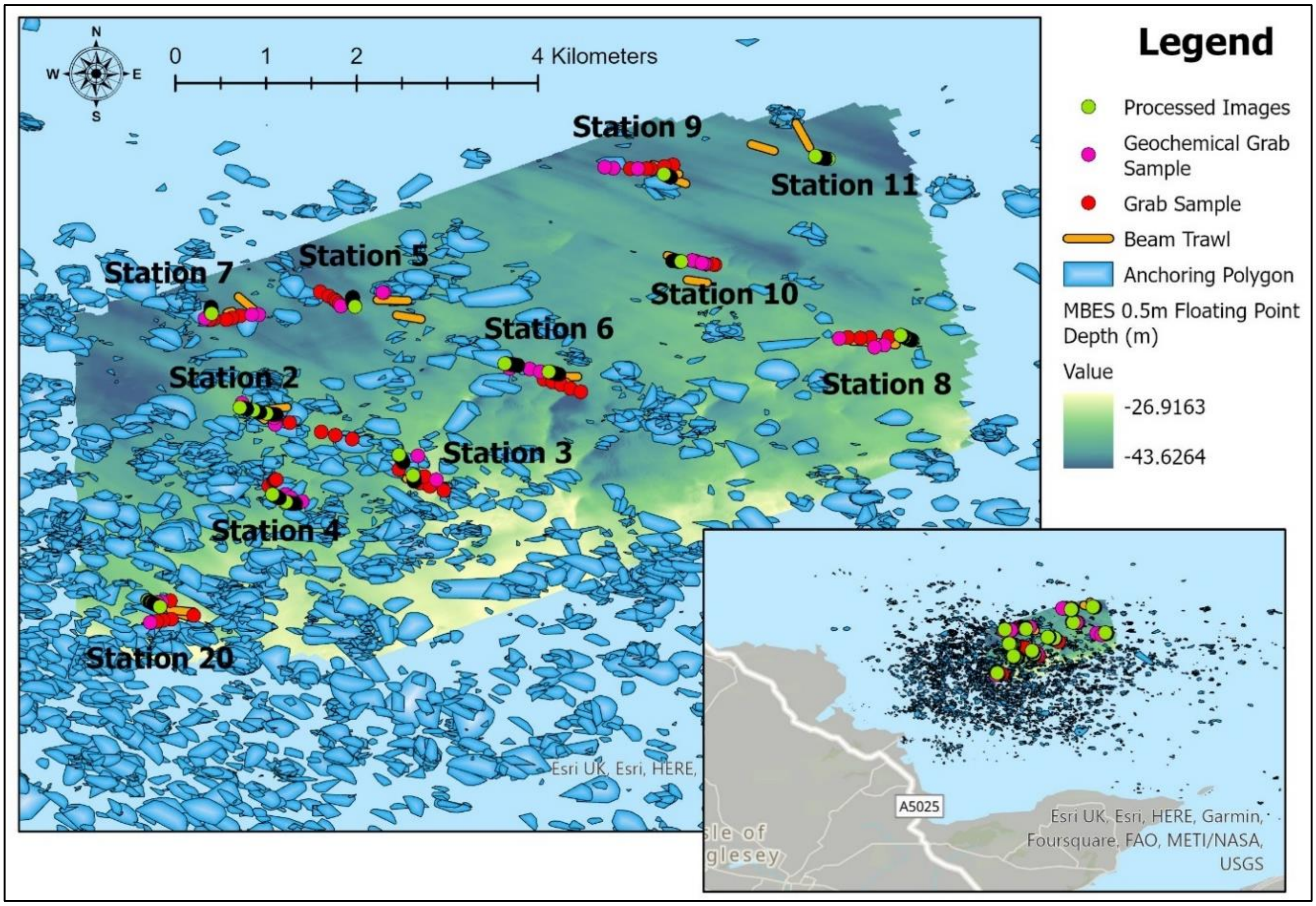


Figure 3. Location of the study site at the Point Lynas anchorage off the east coast of Sir Ynys Môn (Isle of Anglesey), UK. Multibeam (MBES) data shows broad bathymetry of the study area. Locations of samples taken are indicated, as well as anchoring polygons that document areas impacted by anchoring between '2017-01-01 00:00' and '2021-07-01 23:59' UTC.

5.1 STUDY AREA

The effects of vessel anchoring activity on benthic communities were investigated at the Point Lynas anchorage off the northeast coast of Anglesey, as shown in figure 3. This area is popular for anchoring merchant vessels sheltering from the prevailing south-westerly winds and vessels waiting to access the Mersey or dock at Liverpool, one of the busiest container ports, the largest west coast port, and the main gateway to the UK for transatlantic trade. The port of Liverpool handled 34.3 million tonnes of goods in 2019 (2020b)

This area was selected for this research to have minimal variation in environmental variables as it displays fairly homogenous local characteristics such as sediment type (coarse sediment category, gravelly sand), mean spring peak flow tidal velocities (0.78-1 m/s), wave energy, depth (25.5-35 m), and strong variation in anchoring activity (Whitton et al. 2022). The area is characterized by an OSPAR bottom fishing intensity of 0,0 – 1,5 swept area ratio (OSPAR Commission). The mean bed shear stress, as established by SEACAMS2, Bangor University, of the study area ranged from 0.5177 N m^{-2} to 0.7616 N m^{-2}.

5.2 SAMPLING DESIGN

In the sampling design provisional semi-quantitative data of AIS density from emodnet (Emodnet) was used to calculate the average hours per square kilometre of all vessel types for 2017 - 2019 for 1km^2 boxes in the study area. Ten of these 1km^2 sampling stations were identified to cover a range of different anchoring activities, as well as backup stations in case the preferred stations were anchored for the duration of the research cruise. Sampling of the benthic communities was carried out between 17-22nd June 2021 from the *RV Prince Madog*.

5.3 SAMPLING

5.3.1 Grab Samples

Within each 1 km^2 sampling station, five 0.1-m^2 day grab samples were taken to study infaunal communities, as well as three grab samples for geochemical analysis. Grab samples were taken haphazardly within the 1 km^2 box around the centre of the sampling station.

Samples for infauna were sieved through a 1 mm mesh sieve at sea and then again in the laboratory prior to sample sorting. Samples were fixed in 4% buffered formalin and stained with Rose Bengal to aid in sorting fauna. In the laboratory, species were identified to the lowest possible taxonomic level within the time allowed (Annelids to family, Arthropoda, and Mollusca to Genus), and blotted wet mass was recorded.

Geochemical samples were collected using a plastic spoon from the centre of each geochemical Day grab. The samples were immediately fixed in 4% formalin solution diluted with distilled water. These samples were then analysed by CGG (Compagnie Générale de Géophysique) at their laboratories in Llandudno.

5.3.2 Beam Trawl sampling

Two 5-minute beam trawls were performed at each station at a speed of ~1-knot, to sample the epifauna. The trawls were positioned as close as possible to the centre of the Station 1 km^2 box. The catches were sorted, identified, and weighed at sea. Unfortunately, due to Covid-19 restrictions, there was a limited number of crew on board the vessel, and therefore compromises were made in sorting the catch. Due to the high abundance of crabs and green sea urchins and the time constraints to sort each catch, green sea urchins were subsampled (2 litre bucket of trawl material from each trawl) and only hermit crabs were sorted and photographed. Hermit crabs were counted from the trawl photographs.

5.3.3 Seabed Camera Tow sampling

One camera tow of ~16 min at ≤ 1-knot was undertaken at each station to sample the epibenthos. The camera tow moved across the seabed in short bursts of quicker and slower movement and therefore when the auto shutter setting was used, and the tow was moving quickly the resulting images were out of focus, which would have made identification of fauna difficult in analysis. Thereafter, photographs were taken manually whenever the camera sled slowed down, approximately every 5 seconds, resulting in approximately 200 photographs taken at each sampling station. Nearly 500 photos were taken at the first station as the equipment and settings were adjusted. Photographs taken using the auto shutter setting were excluded from the analysis.

This method resulted in 2700 images from 11 different sampling stations that covered a range of different anchoring pressures. This was far more than what could reasonably be processed to extract species identification and abundance information from the images. Therefore, the transects were used to identify 800 images for processing. Subsampling was achieved by taking an initial transect of 50 images from the start of each camera tow, as well as five transects of 50 images that were identified, to increase the range of anchoring pressures sampled at each station. This can be seen in figure 2, where there are transects of the processed images at either end of the camera tow. Seabed images were processed using the Biigle image annotation software (Langenkämper et al. 2017). Epifauna were identified to the lowest possible taxonomic level allowed by the image quality, counted, and recorded. The identification levels of the species included in the analysis are shown in Table 2. For organisms such as hydroids and dead man's fingers, polygons were used to estimate coverage in square pixels (sqpx) by drawing around the visible edges of organisms. The images from station 2 were much darker than those from other stations; therefore, it was much harder to distinguish fibrous brown hydroids from each other. As a result, for all stations *Vesicularia spinosa*, *Hydrallmania falcata*, and *Sertularia* species were grouped together and included within the category that was already being used for indistinguishable brown hydroids. Fishes and more mobile epifauna were excluded from the analysis, as they are unlikely to remain within specific areas of different anchoring pressures.

5.4 CUMULATIVE ANCHORING PRESSURE

After sampling, it was found that the anchoring activity was patchy, as seen in Figure 3, 4, and 6, within the previously identified 1 km^2 stations, , with many areas with no anchoring over the 3-year period, as well as other areas where many anchoring events had occurred and the anchoring polygons overlapped, as shown in figures 2 and 4. It was therefore inappropriate to assign the same anchoring intensity to all samples from a station. It was more appropriate to calculate the cumulative anchoring pressure of each sample using more recent Automatic Identification system (AIS) data. The AIS transmits navigational information about the vessel, including GPS location, speed, course, and heading, as well as information about the vessel itself, including the ship name, MMSI ID, ship type, and vessel length. As a result of this recalculation, each sample (seabed image, grab, and trawl) was assigned a corresponding cumulative anchoring pressure that most precisely described the cumulative anchoring pressure of the sample site over the last three years. The assigned cumulative anchoring pressure values varied within and between the sampling stations, as illustrated in Figure 4. For seabed images and grab samples, the exact GPS location of the sample was available, the total anchoring pressures of all overlapping anchoring polygons at that sample point were extracted, and the sum of those was assigned to the sample.

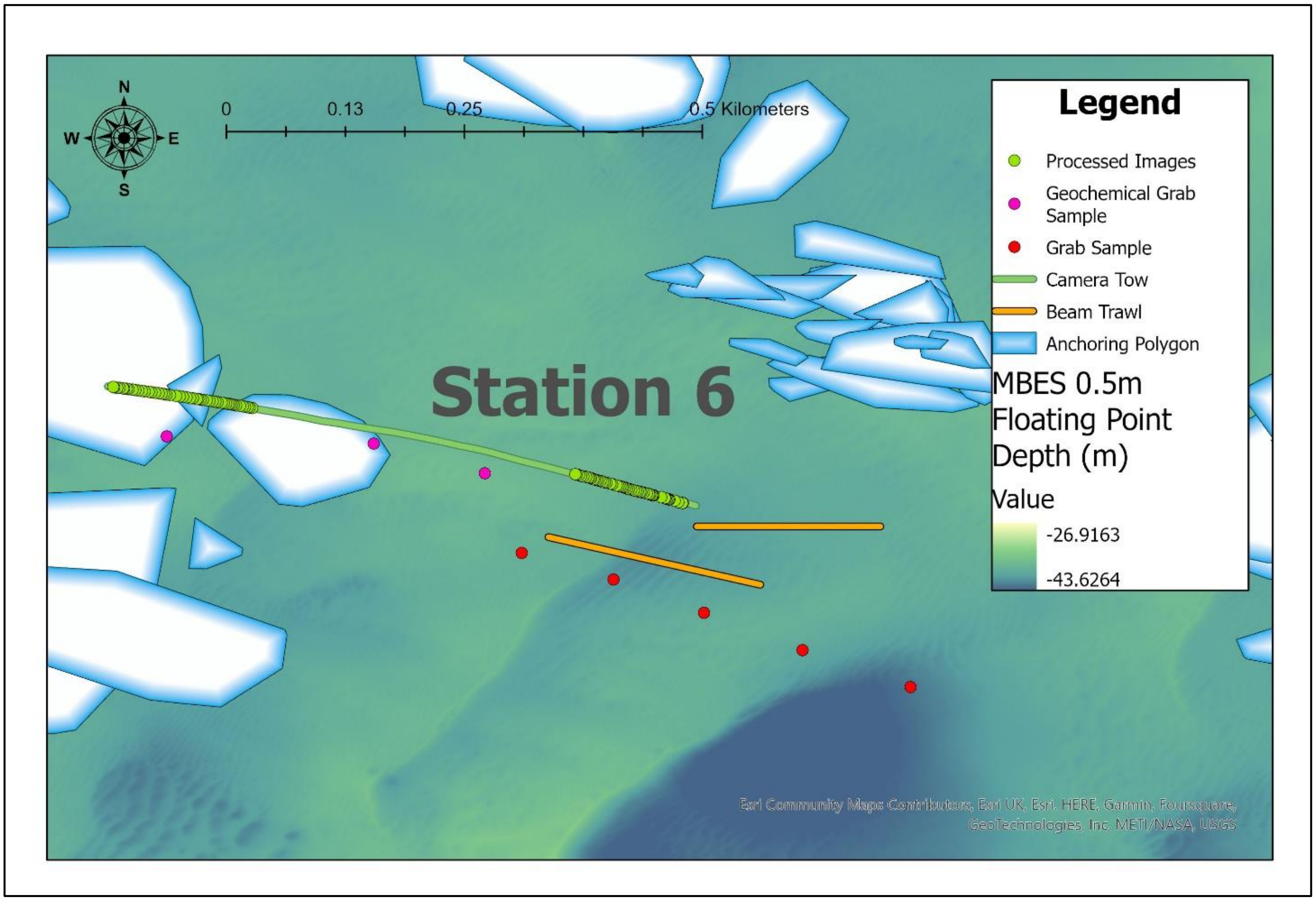


Figure 4. Samples taken at station 6 at Point Lynas anchorage off the east coast of Sir Ynys Môn (Isle of Anglesey), UK. Red circles show grab samples, pink circles show geochemical samples, green lines show camera tows with green circles showing processed seabed images, and yellow lines showing beam trawls. Areas of seabed affected by anchors and chains are defined by blue polygons with white centres and document areas impacted by anchoring between '2017-01-01 00:00' and '2021-07-01 23:59' UTC. Multibeam (MBES) data shows broad bathymetry of the study area.

Data received by terrestrial AIS bases concerning all vessels that entered the study area between '2017-01-01 00:00' and '2021-07-01 23:59' UTC were purchased from Marine Traffic. From these data, it was necessary to identify anchored vessels. Anchored vessels are not completely stationary and drift around the anchor with, waves, and tides in an arc or circular shape, as shown in figure 2. Therefore, an inclusion criterion based on published methods by Deter et al. (2017) was used for the AIS data from the 3 years prior to sampling to exclude vessels that were not anchored and to separate anchoring events from each other. Anchoring events were defined by a cluster of AIS pings with speed < 0.25 knot, less than 1 hour between AIS pings, and less than 200m between AIS pings. The aim was to create polygons of the seabed areas affected by anchoring disturbance by both the anchor and chain, and to quantify the intensity of disturbance in each polygon, as shown in figure 5.

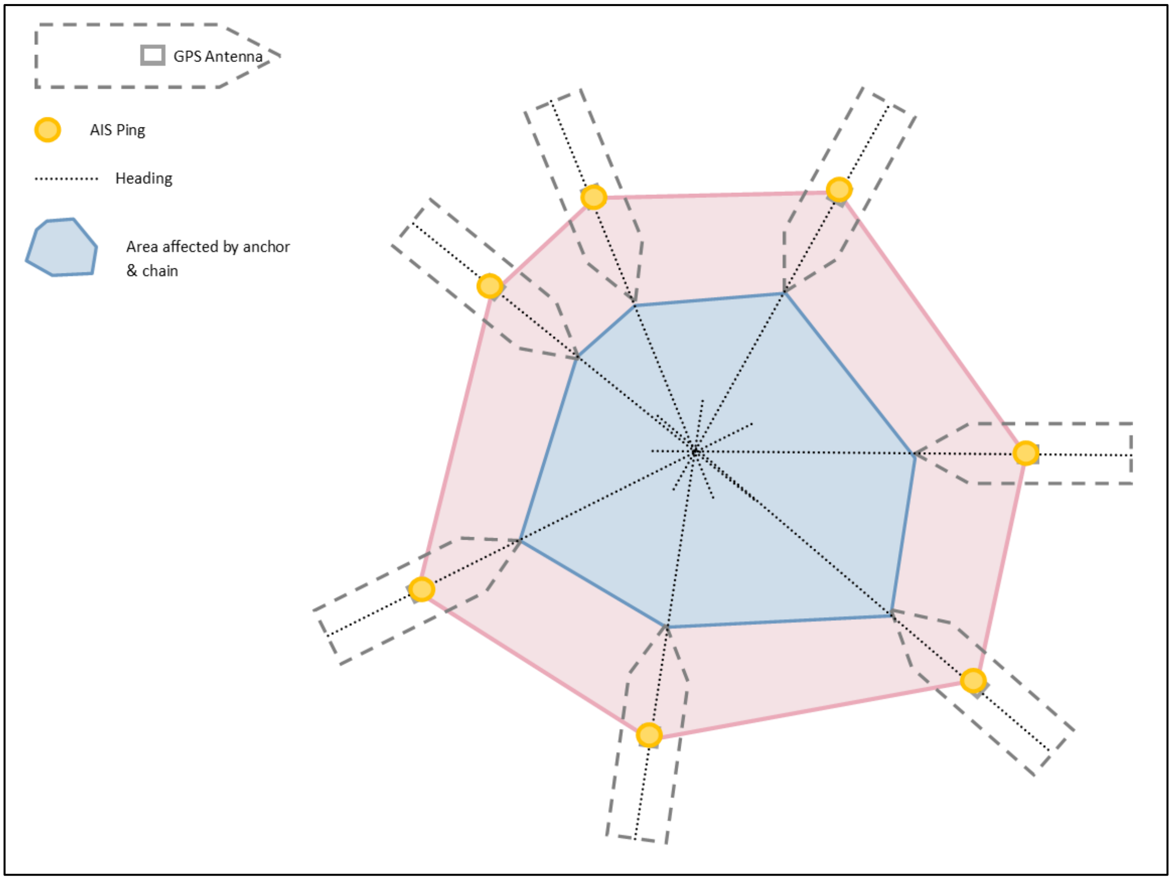


Figure 5. Schematic illustrating how the anchoring polygons were derived from AIS vessel position data and vessel length. The bow position was projected from the GPS antenna based on the length of the vessel and the blue anchoring polygon was derived by mapping the bow-to-bow position of each hourly vessel location.

As shown in figure 1 c, the optimum ship position in good weather has the chain going straight down from the bow of the ship to the seabed, to allow for the chain to absorb force in inclement weather. To ascertain the impact of anchoring on the seabed, it is necessary to plot the position of the bow of the ship with respect to the GPS antenna, because the chain goes down from the bow while the GPS antenna is usually positioned somewhere on the rear of the vessel. This was projected by estimating the distance from the ship antenna to the anchor/bow of the ship, based on the length of the vessel. For most vessels the heading of the vessel is shown for each AIS ping and the intersection is the centre of the anchoring polygon. The area of each anchoring polygon is mapped by linking all anchor/bow positions for each separate anchoring event, as shown in figure 5.

The cumulative anchoring pressure inside each anchoring polygon is defined as follows:

$$\text{Anchoring Pressure} = \frac{\text{Anchoring duration (hours) x DWT}^{2/3}\text{ (tons)}}{\text{Area of Anchoring Polygon}}$$

The deadweight tonnage (DWT) of the vessel was to the power of 2/3 to account for the relationship between typical anchor and chain size and vessel DWT. The size of the anchor and length of the chain used by a vessel is determined by the ships equipment number. The equipment number is a function of the vessel displacement$^{2/3}$, frontal (wind exposed) area of the ship, and lateral (wind exposed) area of the ship (Luger and Harkes 2013; DNV 2017), as shown in the equations published by Det Norske Veritas, (2017).

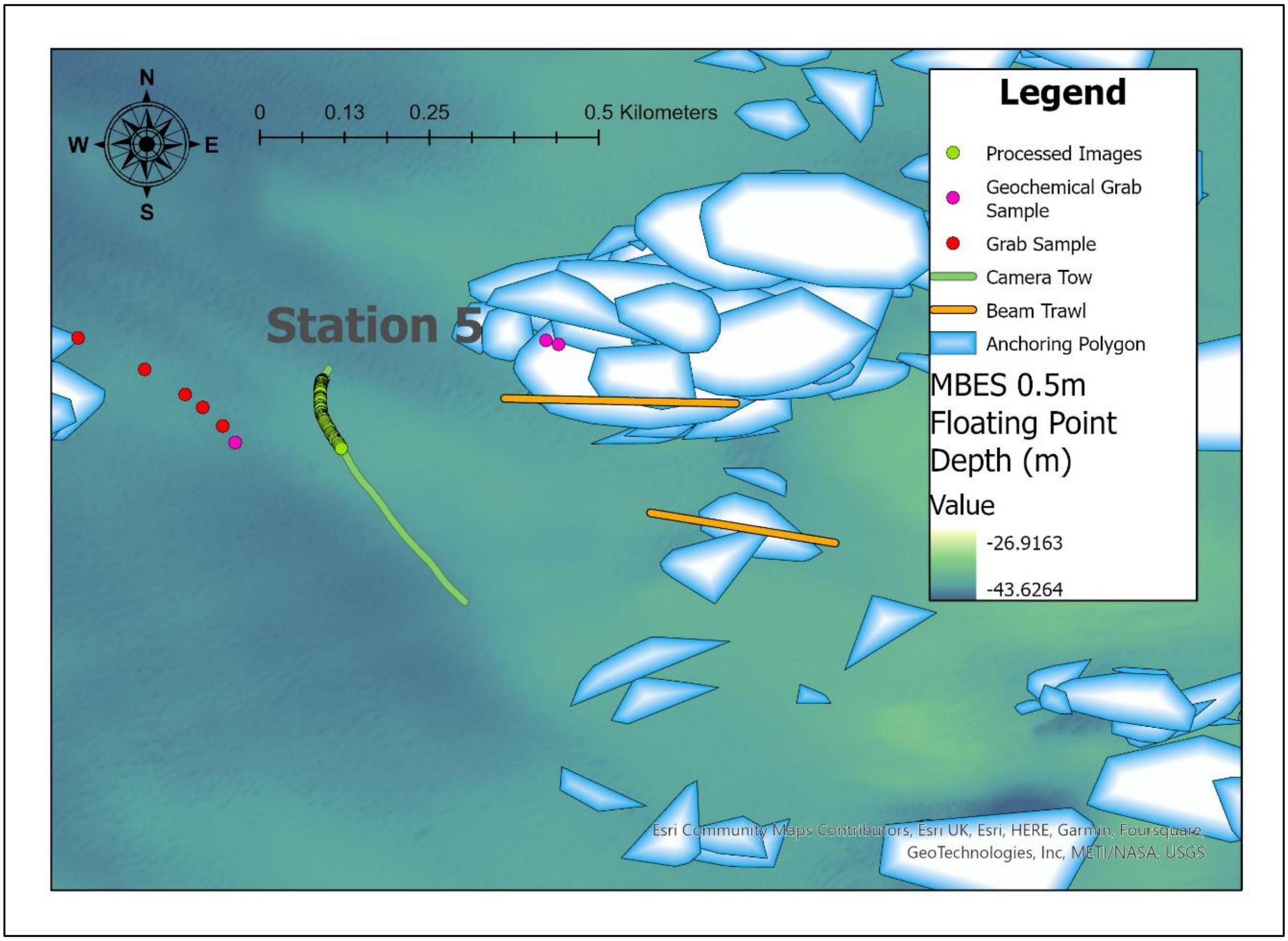


Figure 6. Samples taken at station 5 at Point Lynas anchorage off the east coast of Sir Ynys Môn (Isle of Anglesey), UK. Red circles show grab samples, pink circles show geochemical samples, green lines show camera tows with green circles showing processed seabed images., areas of seabed affected by anchors and chains are defined by blue polygons with white centres and document areas impacted by anchoring between '2017-01-01 00:00' and '2021-07-01 23:59' UTC, and yellow lines show beam trawls. Where beam trawls cross multiple anchoring polygons a mean anchoring pressure for the trawl was established by extracting the anchoring pressure at 1001 points along the trawl and then calculating a mean anchoring pressure per trawl. Multibeam (MBES) data shows broad bathymetry of the study area.

As shown in figure 6, beam trawls frequently crossed polygons of different anchoring pressures and across intersecting anchoring polygons; they also varied in length. Therefore, it was necessary to establish a mean anchoring pressure value for the entire trawl. To achieve this, the anchoring pressure at 1001 points along the trawl was extracted, and then the mean anchoring pressure per trawl was calculated.

5.5 DATA ANALYSIS

5.5.1 Grab sample Data Analysis

Each grab sample had a unique cumulative anchoring pressure, but we did not consider a single grab to be an acceptable level of replication because there was too much variation between grabs. Therefore, it was necessary to group grab samples from the same station that had similar anchoring pressures to create replicates. To allow for the inclusion of an appropriate number of samples in the analysis, inclusion and grouping criteria were established. The anchoring value was rounded to the nearest whole number, grabs within a station were assigned to these anchoring value levels, and when two or more grabs were present at the same anchoring level within a station, they were grouped together. Single grabs were not included. The mean of the grouped grabs was included as a replicate in the analysis. This allowed the inclusion of grab samples from higher anchoring pressures and the greatest number of replicates that came from anchoring values greater than zero. 34 grab samples of a possible 50 were included in the analysis, from 12 replicate groups, 5 of which had an anchoring pressure of 0, the replicate with the highest anchoring pressure was 4 tons$^{2/3}$ hours / m^2. The grouping of grabs into replicates resulted in 9 grabs from the highest anchoring pressures 4.1 - 28 tons$^{2/3}$ hours / m^2 that could not be included in analysis as there was not any replication of these.

The mean abundance and biomass per m2 were calculated for each replicate. To account for the variability in the number of grabs per replicate, it was not appropriate to calculate the mean species richness per replicate because varying numbers of grabs included in the replicates would lead to variation in the number of species present. In this instance a ‘FOR loop’ was coded to automatically randomly sample 2 grabs from each replicate 100 times by using a looping variable to repeatedly execute the section of code that samples how many species were present in the 2 samples. The mean species richness for each of replicates was then calculated from the species richness of each of these samples of 2 grabs. A univariate linear regression model was then used to investigate the relationship between cumulative anchoring pressure and each of the dependent variables (univariate community descriptors, abundance, richness, and biomass).

5.5.2 Geochemistry & Mineralogy Analysis

At each station three 0.1 m^2 day grab samples were collected for geochemical analysis by CGG at their laboratories. Geochemical samples were collected using a plastic spoon from the centre of each collected day grab geochemical sample. Samples were then immediately fixed in a 4% formalin solution diluted with distilled water. Thirty samples were analysed for bulk organic geochemistry and concentrations of major and trace elements. Total carbon (TC) was determined using an Eltra CS800 Carbon Sulphur Determinator calibrated to certified reference materials (Whitton et al. 2022).

5.5.3 Trawl Analysis & Missing values

Owing to the coronavirus restrictions on board the research vessel, there was limited crew and time available to sort the trawl catch and minimal contact between swapping shifts, which unfortunately led to a significant amount of missing data. Missing values were highlighted in the analysis of the beam trawl data, which are related to the weight and abundance of what would probably be the most abundant species that may be strongly affected by anchoring.

Table 1. Outline of data that were missing from the beam trawl dataset due to the Covid-19 restrictions on board the research vessel in June 2021.

Species	Affected trawls	Data available	Data missing
Brittle Star ***Ophiuroidea***	18, 8, 2, 1, 11, 10, 13, 9, 4, 5, 3, 7, 6	Total weight	Number of individuals
Gastropod mollusc ***Colus sp.***	8	Number of individuals	Total weight
Hermit crabs ***Paguridae***	All trawls	Number of individuals	Total weight
Dead Man's Fingers ***Alcyonium digitatum***	All trawls	Absent or present	Total weight per trawl Number of colonies per trawl
Green sea urchin ***Strongylocentrotus droebachiensis***	All trawls	Absent or present	Total weight per trawl Number of Individuals
Sand stars ***Astropecten irregularis***	2	Total weight	Number of individuals
Painted top shell ***Calliostoma zizyphinum***	2	Number of individuals	Total weight
Octopus ***Octopodia***	9	Number of individuals	Total weight
Weever Fish ***Trachinidae***	9	Number of individuals	Total weight

Making estimates of these missing values using the other available data might have been possible by extrapolation; however, the potential for the introduction of bias here is great. It is likely that organisms were weighed and accurately counted when they were larger and easier to separate or when there were not many in a sample; therefore, it was decided to only calculate and analyse total species richness, *Ophiuroidea* biomass, *Paguridae* abundance and number of *Alcyonium digitatum* colonies for the trawl data as this is the most reliable data available from this set.

Species richness was established per trawl, and a univariate linear regression model was used to investigate the relationship between the cumulative anchoring pressure and species richness.

5.5.4 Seabed Image Data Analysis

As each image had a distinct cumulative anchoring pressure, images were grouped into 'replicates' that came from the same station and had the same anchoring pressure, and replicates with fewer than 5 images were excluded. This means that 749 images from a possible 800 were included in the statistical analysis.

The mean abundance and coverage were calculated per replicate. To account for the variability in the number of images per replicate, it was not appropriate to calculate the mean species richness per replicate because varying numbers of images included in the replicates would lead to variation in the number of species present. In this instance, a 'for-loop' was used to randomly sample five images for each replicate 100 times before calculating the mean species richness for each of these samples. A univariate linear regression model was then used to investigate the relationship between cumulative anchoring pressure and each of the dependant variables (univariate community descriptors: abundance, richness, and coverage).

5.5.5 Statistical Analysis

Data were collated and organised using Microsoft Excel, and subsequent analysis was performed in the GUI R studio. The residuals were not normally distributed; therefore, all data were log-transformed (log10(x+1)) to achieve homoscedasticity and satisfy the assumptions for parametric analyses by linear regression. To ascertain whether merchant vessel anchoring impacted community indicators (abundance, richness, and biomass), univariate linear regression analyses were performed. These linear regression analyses identified whether anchoring significantly contributed to differences in these community indicators. The resulting degrees of freedom, residual spread, histograms, and QQ plots of the different linear models were examined to determine their fit. For all statistical analyses, the significance (α) was set at <0.05. No exploration on the interactions between depth and anchoring pressure was undertaken as part of this analysis.

6 RESULTS

6.1 ENVIRONMENTAL CONDITIONS

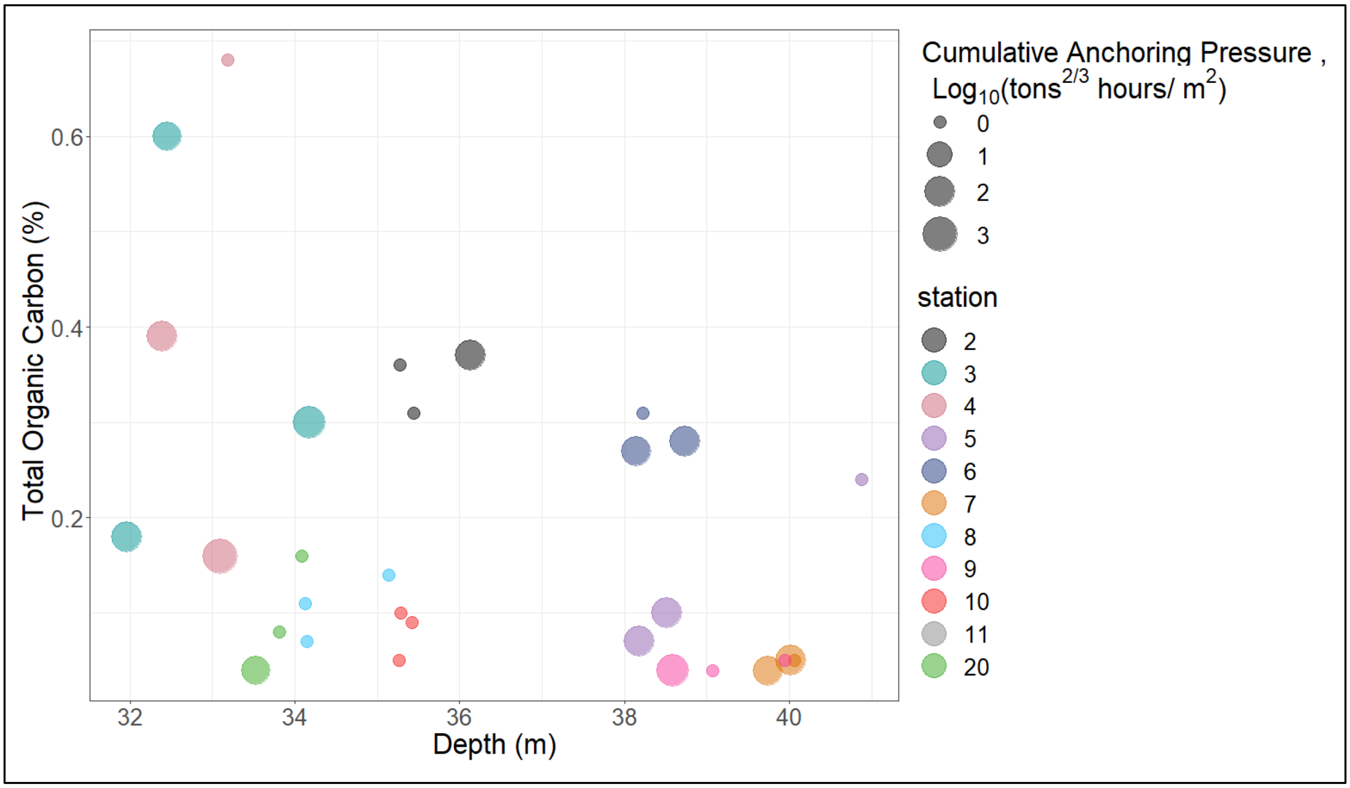
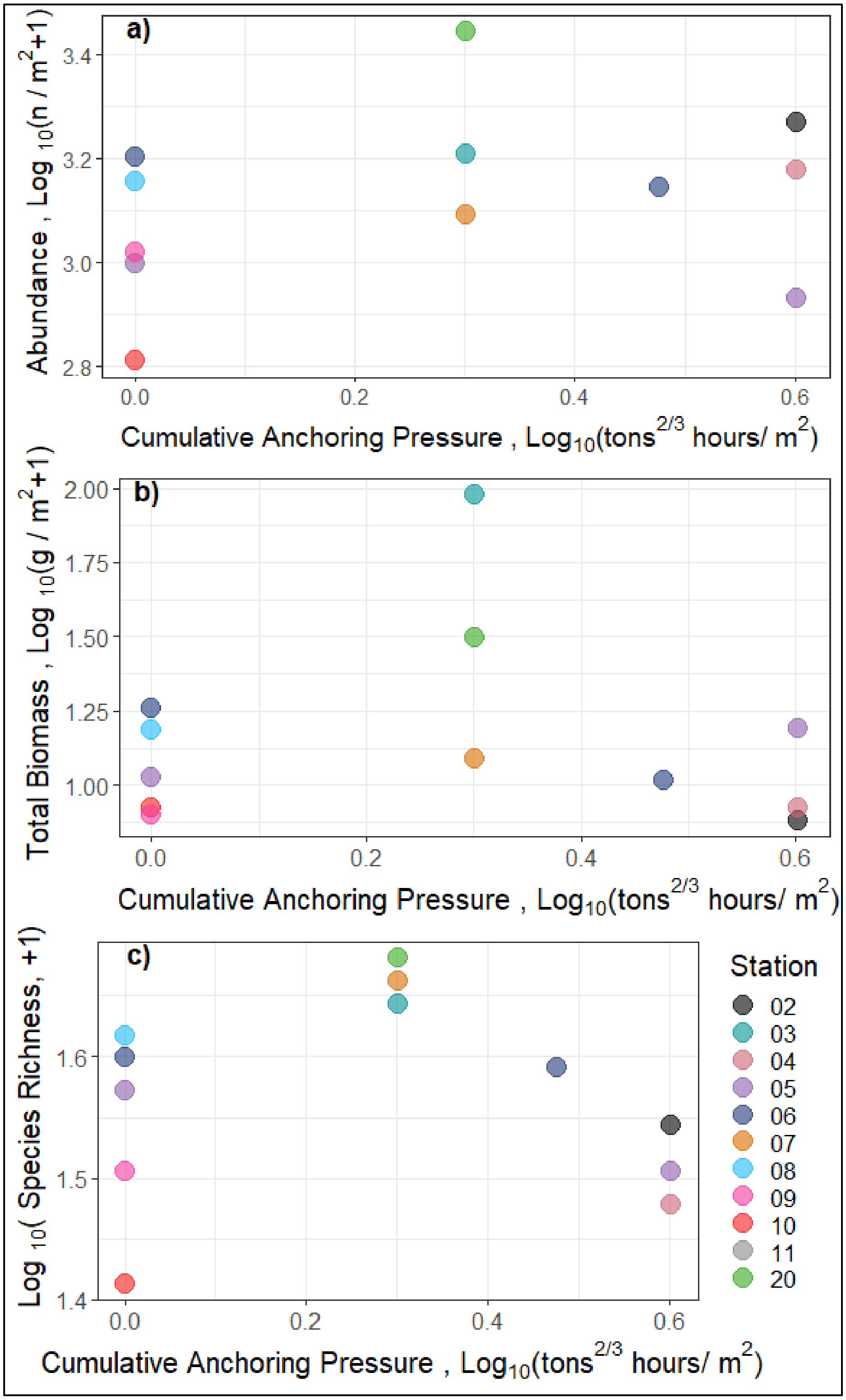


Figure 7. Total organic carbon percentage by depth of the sediment at the Point Lynas anchorage sampled in June 2021. The bubble size is proportional to the cumulative anchoring pressure. Colours denote the sampling station that replicates originated from and the size of the point relates to the anchoring intensity experienced by the sampled area in the three years leading up to the sampling.

Geochemical grab samples showed no clear relationship between total organic carbon, depth, and anchoring pressure. The ranges for depth and total organic carbon for the geochemical samples were small. It is desirable that different anchoring values are mixed throughout the parameter space, as this indicates that there is less risk of confounding effects of anchoring with other environment gradients.

6.2 GRAB SAMPLES - INFAUNA



In all 50 fauna grab samples 127 taxa were identified with the highest total number of individuals from the annelid families of *Orbiniidae* (1,267), *Spionidae* (931), and amphipod genus *Urothoe* (653). These three groups were analysed individually, as well as part of the community data.

No relationship was found between the cumulative anchoring pressure and the total abundance of infauna per replicate (figure 8. a) ($P = 0.3717$, $t_{10} = 0.935$, $R^2 = 0.08043$, Adjusted $R^2 = -0.01153$). No relationship was found between cumulative anchoring pressure and total biomass of infauna per replicate (figure 8. b) ($P = 0.9455$, $t_{10} = -0.070$, $R^2 = 0.0004917$, Adjusted $R^2 = -0.09946$). The species richness of replicates was estimated by randomly sampling two grabs for each replicate 100 times to account for variability in the number of available grabs per replicate. No relationship was found between cumulative anchoring pressure and species richness of infauna per replicate (figure 8.c) ($P = 0.6933$, $t_{10} = -0.407$, $R^2 = 0.00311$, Adjusted $R^2 = -0.09658$).

6.2.1 Grab Samples – Focus Species

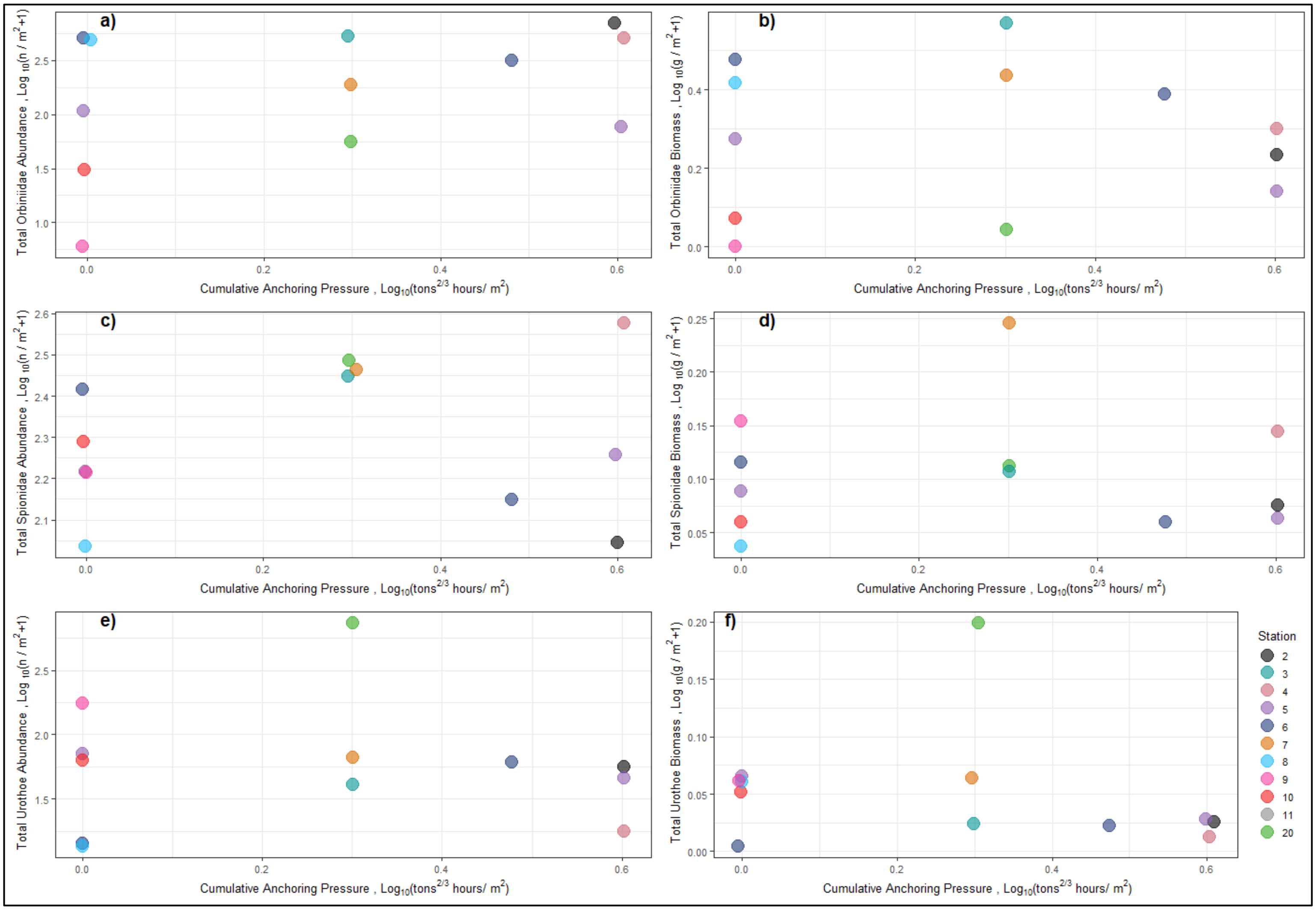


Figure 9. a) Total *Orbiniidae* abundance (*values jittered to enhance visualisation by h = 0.0000, w = 0. 006), b) total biomass of *Orbiniidae*, c) Total *Spionidae* abundance (*values jittered to enhance visualisation by w = 0. 006), d) total *Spionidae* biomass, e) Total *Urothoe* abundance, and d) total *Urothoe* biomass (*values jittered to enhance visualisation by w = 0. 008) in day grab replicates by AIS-derived cumulative anchoring pressure in the Point Lynas anchorage sampled in June 2021. Colours denote the sampling station from which the replicates originated.

No relationship was found between cumulative anchoring pressure and the total abundance of *Orbiniidae* per replicate (figure 9. a) ($P = 0.2164$, $t_{10} = 1.320$, $R^2 = 0.1483$, Adjusted $R^2 = -0.06316$). No relationship was found between cumulative anchoring pressure and total biomass of *Orbiniidae* per replicate (figure 9. b) ($P = 0.9216$, $t10 = 0.101$, $R^2 = 0.001018$, Adjusted $R^2 = -0.09888$). No relationship between cumulative anchoring pressure and total abundance of *Spionidae* per replicate was found (figure 9.c) ($P = 0.6856$, $t_{10} = 0.417$, $R^2 = 0.01708$, adjusted $R^2 = -0.08121$). No relationship between cumulative anchoring pressure and total biomass of *Spionidae* per replicate was found (figure 9.d) ($P = 0.9411$, t_{10} =

0.076, R^2 =0.0005747, Adjusted R^2 = -0.09937). No relationship was found between the cumulative anchoring pressure and the total abundance of *Urothoe* per replicate (figure 9. e) (P = 0.9933, t_{10} = -0.009, R^2 =0.00000751, Adjusted R^2 = -0.09999). No relationship was found between cumulative anchoring pressure and total biomass of Urothoe per replicate (figure 9. f) (P = 0.5629, t_{10} = -0.598, R^2 = 0.03457, Adjusted R^2 = -0.06197).

6.3 BEAM TRAWLS – EPIFAUNA

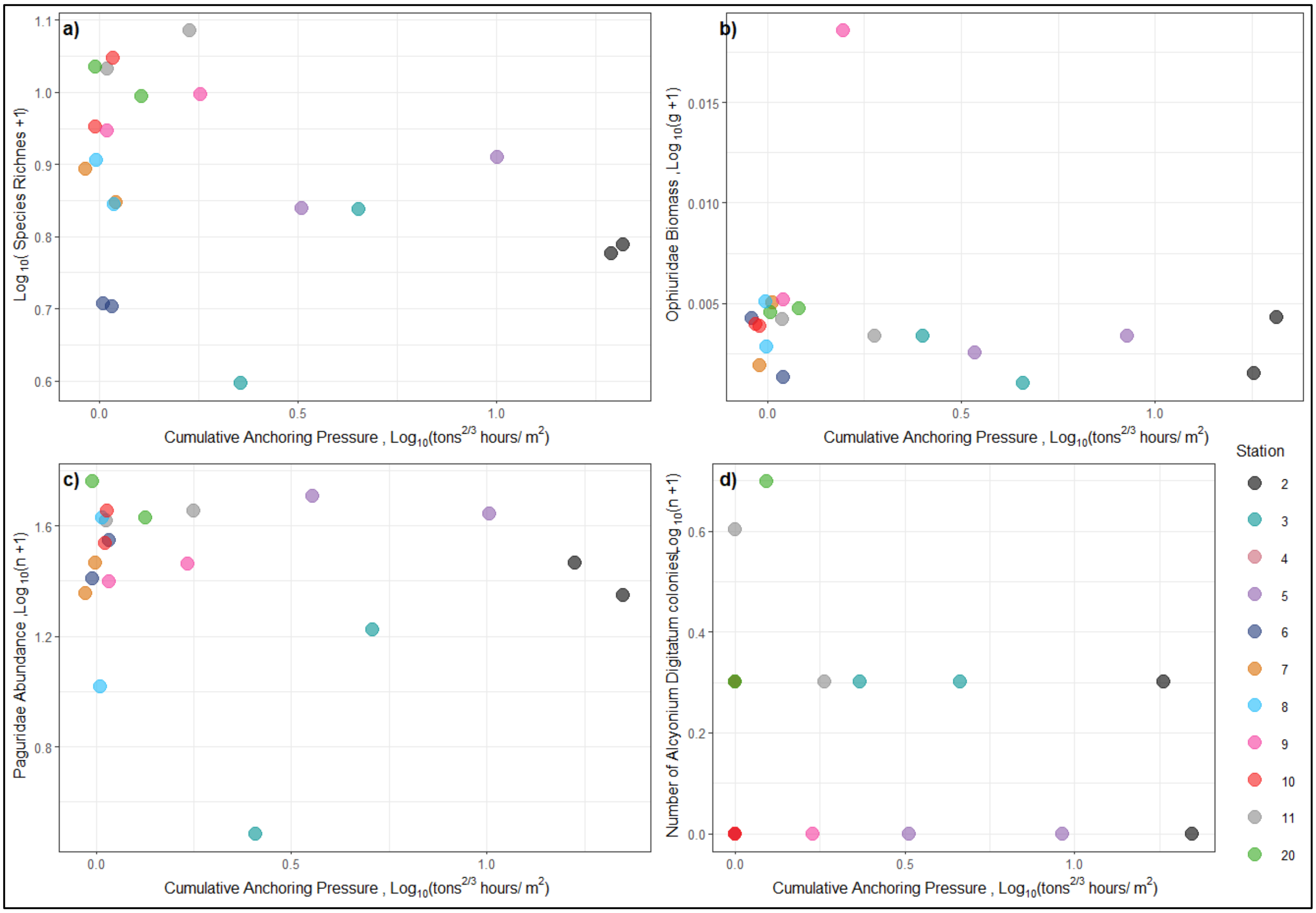


Figure 10. a) Species richness of epifauna excluding fishes and other more mobile epifauna. (*values jittered to enhance visualisation by h = 0.01, w = 0. 045), b) *Ophiuridae* biomass. (*values jittered to enhance visualisation by h = 0.001, w = 0. 045), c) *Paguridae* abundance (*values jittered to enhance visualisation by h = 0.02, w = 0. 045) and d) number of *Alcyonium digitatum* colonies in beam trawl samples by AIS derived cumulative anchoring pressure in the Lynas anchorage sampled in June 2021. Colours denote the sampling station that replicates originated from. Colours denote the sampling station that replicates originated from. *All values jittered to enhance visualisation by h = 0.03, w = 0.03

No relationship was found between species richness and cumulative anchoring pressure (figure 10a) (P = 0.1825, t_{18} = -1.387, R^2 =0.09261, Adjusted R^2 = 0.04633). No relationship was found between the

Ophiuridae biomass and cumulative anchoring pressure (figure 10b) ($P = 0.6689$, $t_{18} = -0.435$, $R^2 = 0.01039$, adjusted $R^2 = -0.04459$). No relationship was found between *Paguridae* abundance and cumulative anchoring pressure (figure 10c) ($P = 0.7387$, $t_{18} = -0.339$, $R^2 = 0.006337$, adjusted $R^2 = -0.04887$). No relationship was found between the number of *Alcyonium digitatum* colonies and the cumulative anchoring pressure (figure 10d) ($P = 0.754$, $t_{18} = -0.318$, $R^2 = 0.005594$, Adjusted $R^2 = -0.04965$).

6.4 SEABED IMAGES - EPIFAUNA

The following groups were identified from the image data and were included in the analysis. Fish and more mobile species were excluded from the analysis as they are unlikely to remain within specific areas of different anchoring pressures.

Table 2. Outline of species identified in the seabed images and included in the analysis.

Included in Count analysis of Seabed images		Included in Coverage analysis of Seabed images		
Ophiuridae	Brittle star	Unidentified brown fibrous hydroids		
Adamsia palliata	Cloak anemone			
Colus sp.	gastropod mollusc			
Macropodia	Spider crab	***Vesicularia spinosa***		Indistinguishable Hydroids
		Hydrallmania falcata	Helter skelter hydroid	
		Sertularia	Squirrel’s tail hydroid	
Asterias rubens	Common starfish	***Tubulariidae***		Oaten pipes hydroid
Solasteridae	Sun star	***Alcyonidium diaphanum***		Sea chervil
Buccinum undatum	Common whelk	***Nemertesia ramosa***		Branched antenna sea fir
Aeolidia papillosa	Common grey sea slug	***Nemertesisa antennina***		Sea beard
Paguridae sp.	Hermit crab	***Hydractinia echinata***		Hermit crab fur
Processa sp.	Shrimp	***Flustra folicaea***		Hornwrack
Ebalia sp.	Nut crab	***Alcyonium digitatum***		Dead man’s fingers

Sabella pavonina	Peacock worm	***Crustose corilinales sp.***	Crustose corraline algae
Lanice conchilega	Sand mason worm	***Carbasea carbasea***	
Astropecten irregularis	Sand star		
Actiniaria sp.	Sea anemones		
Aphrodita-aculeata	Sea mouse		
Phlebobranchia sp.	Sea squirt		

6.4.1 Seabed Images - Count Data

The two groups with the highest total abundance across seabed images were *Ophiuroidea* (672) and *Paguridae* (63); therefore, they were analysed separately, as well as part of the community data.

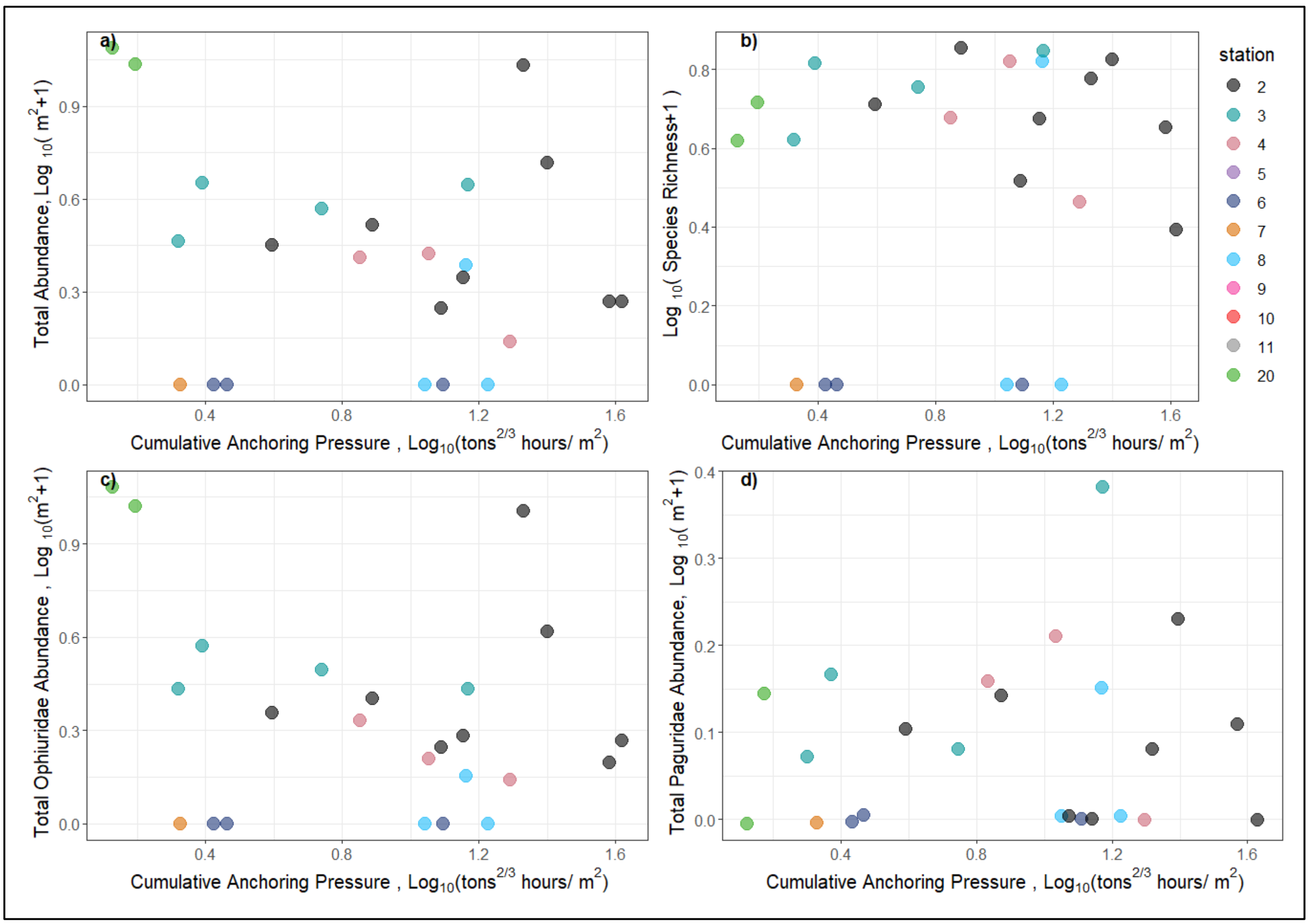


Figure 11. a) Total abundance of epifauna b) Species richness c) Total *Ophiuridae* abundance and d) Total *Paguridae* abundance (**Paguridae* values jittered to enhance visualisation by h = 0.005, w = 0. 02), from seabed image replicates by AIS derived cumulative anchoring pressure in the Point Lynas anchorage sampled in June 2021. Excluding fishes and other

more mobile epifauna. Colours denote the sampling station from which the replicates originated. *All values jittered to enhance visualisation by h = 0.03, w = 0.03

No relationship was found between the total abundance of epifauna and cumulative anchoring pressure (figure 11a) ($P = 0.346$, $t_{22} = -0.963$, $R^2 = 0.04045$, adjusted $R^2 = -0.003169$). No relationship was found between species richness and cumulative anchoring pressure (figure 11b) ($P = 0.6021$, $t_{22} = 0.529$, $R^2 = 0.01256$, Adjusted $R^2 = -0.03232$). No relationship was found between the total abundance of *Ophiuridae* and the cumulative anchoring pressure (figure 11c) ($P = 0.2263$, $t_{22} = -1.245$, $R^2 = 0.0658$, Adjusted $R^2 = 0.02333$). No relationship was found between the total abundance of *Paguridae* and the cumulative anchoring pressure (figure 11d) ($P = 0.5618$, $t_{22} = 0.589$, $R^2 = 0.01553$, adjusted $R^2 = -0.02922$).

6.4.2 Seabed Images - Coverage Data

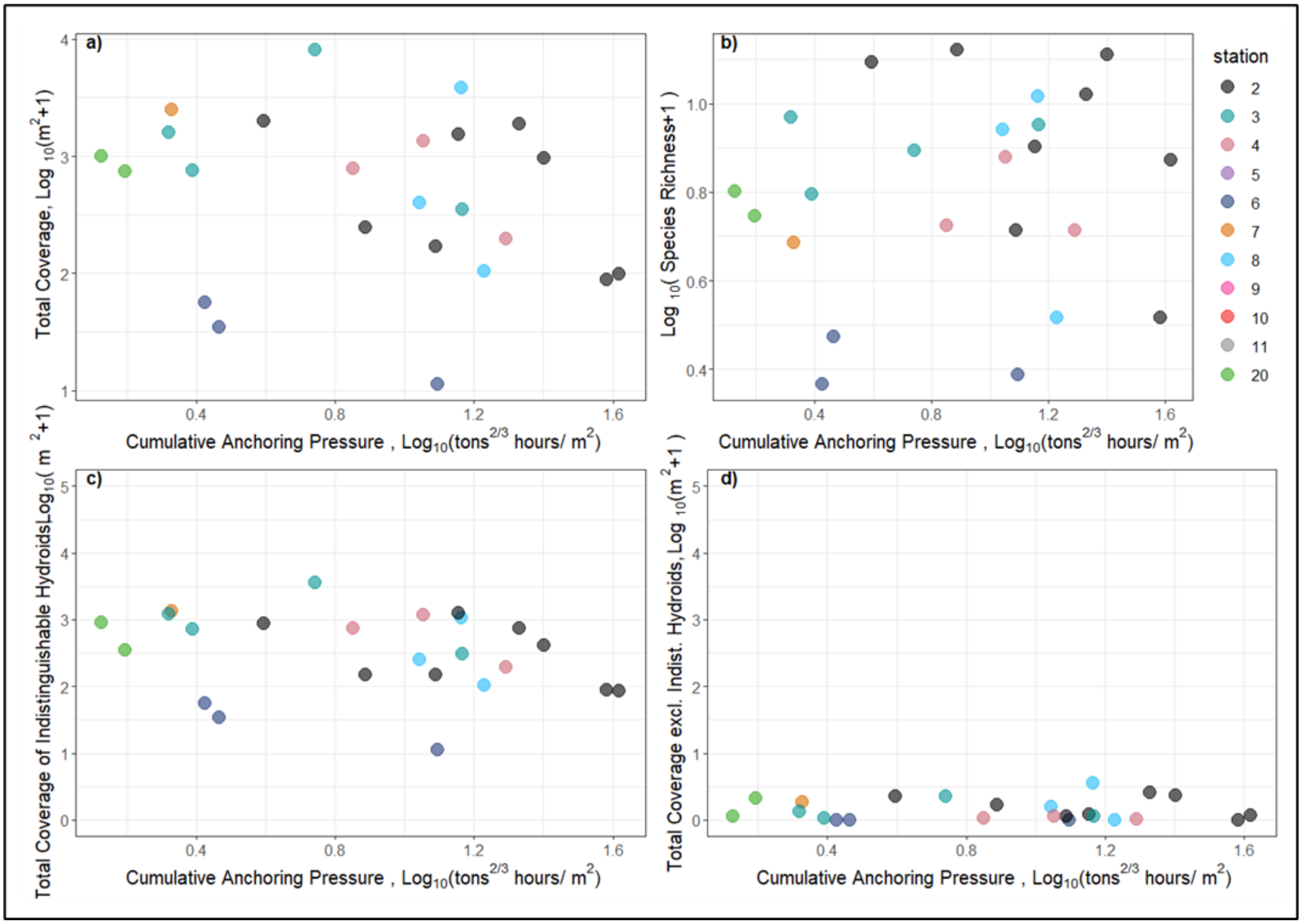


Figure 12. a) Mean coverage of epifauna b) Species richness c) Total coverage of indistinguishable hydroids and d) Total coverage excluding indistinguishable hydroids from seabed image replicates by AIS derived cumulative anchoring pressure in the Lynas anchorage sampled in June 2021. Colours denote the sampling station that replicates originated from. *All values jittered to enhance visualisation by h = 0.03, w = 0.03

No relationship was found between total epifauna coverage and cumulative anchoring pressure (figure 12a) ($P = 0.2888$, $t_{22} = -1.087$, $R^2 = 0.05097$, Adjusted $R^2 = 0.007827$). No relationship was found between species richness and cumulative anchoring pressure (figure 12b) ($P = 0.5663$, $t_{22} = 0.582$, $R^2 = 0.01518$, Adjusted $R^2 = -0.02959$). No relationship was found between the total coverage of the group of indistinguishable fibrous brown hydroids and the cumulative anchoring pressure (figure 12c) ($P = 0.2199$, $t_{22} = -1.263$, $R^2 = 0.06759$, Adjusted $R^2 = 0.02521$). No relationship was found between the coverage of all epifauna, excluding the indistinguishable fibrous brown hydroids, and cumulative anchoring pressure (figure 12d) ($P = 0.9553$, $t_{22} = -0.057$, $R^2 = 0.0001464$, Adjusted $R^2 = -0.0453$).

7 DISCUSSION

Previous studies that investigate abrasive activity on the seabed by different mechanisms have been shown to result in a variety of negative community effects (Riesen and Reise 1982; George and Warwick 1985; Churchill 1989; Beukema 1995; Dayton et al. 1995; Collie et al. 1997; Duplisea et al. 2002; Hiddink et al. 2006; Rossi 2013; Palanques et al. 2014; van Denderen et al. 2015; O'Neill and Ivanović 2016; Sciberras et al. 2016). Merchant vessel anchoring in sediment habitats has been the focus of limited study and may perpetuate significant disruption to benthic communities due to the size of anchor gear used for these vessels and the frequency at which they anchor during routine operation. Identifying the effects of merchant vessel anchoring will allow for the development of effective strategies for the mitigation and management of anchorages and accurate assessment of benthic habitats subject to anchoring pressure (Davis et al. 2016; Broad et al. 2020; Watson et al. 2022).

In this study, seabed community indicators were measured in areas of varying anchoring activity in the Point Lynas anchorage to identify if a significant difference in community indicators existed between these areas. Community indicators were measured from data collected using day grabs, beam trawls, and seabed images, and statistical analyses were performed using linear regression models. The fundamental aim of this research was to gain a comprehensive understanding of the cumulative ecological effects of repeated disturbance by anchoring activity of merchant vessels on the soft-sediment communities at the Point Lynas anchorage. This study aimed to establish whether anchoring pressure affected the biomass, species richness, and abundance of benthic infauna and epifauna. It attempted to test the hypothesis that repeated mechanical disturbance by anchor scour of merchant vessels is causing disruption to benthic communities, resulting in significant changes in community structure at high anchoring intensity compared to communities that are not impacted by anchoring pressure.

The results do not support the rejection of the null hypothesis, with no significant relationships shown in any analyses. The abundance, biomass, and species richness at the sampled sites showed no correlation with cumulative anchoring pressure. Whilst literature that assesses the impacts of merchant vessel anchoring in soft sediment habitats is sparse, there is evidence to suggest that our results are unexpected as recreational anchoring and other kinds of disturbances such as otter trawling have been shown to have significant negative effects on benthic macrofaunal communities in soft-sediment habitats (Hinz et al. 2009), and merchant vessel anchors are significantly larger and more abrasive than these other mechanisms of disturbance in that they penetrate more deeply into the sediment (Broad et al. 2020). A 2021 report by the Dorset Wildlife Trust, in which Multibeam Echosounder (MBES) surveys were used, showed significant anchor and chain scars on the seabed associated with the increased anchoring of cruise ships resulting from the Covid-19 pandemic (Tinsley 2021). This report took place in an area of seabed characterised by muddy sand and stiff clay which generally experience lower natural disturbance and can bear the marks from disturbance for a significantly longer amount of time than habitats that experience a natural regime of greater disturbance. It is reasonable to expect that this large-scale physical disturbance would be accompanied by ecological consequences. Previous studies on the effects of bottom fishing show that gear that penetrates further into the seabed has greater impacts and recovery tends to be slower in finer sediments such as mud where natural disturbance is typically lower than in coarse sediments such as sand (Collie et al. 2000a; Kaiser et al. 2006; Hiddink et al. 2017).

No relationship was detected between cumulative anchoring pressure and abundance, biomass, or species richness of epifauna from beam trawls or seabed images. An effect on epifaunal organisms would have been expected because contact with large merchant vessel anchors and chains is likely to cause damage to the seafloor, altering sediment composition and biogeochemistry and reducing topographic complexity, subsequently affecting benthic communities, and directly damaging biota, leading to mortality (Collie et al., 2005; Mayer et al., 1991; O'Neill and Ivanović, 2016; Sciberras et al., 2016). The effects of this, however, may be short-lived, and recovery after the disturbance may be swift as organisms may be adapted to high levels of disturbance. In addition, no relationship was detected between cumulative anchoring pressure and abundance, biomass, or species richness of infauna suggesting that cumulative anchoring pressure across the Point Lynas anchorage does not produce negative community effects in benthic communities. An effect on infaunal organisms may have been expected because of the large volume of sediment that is resuspended by high-tonnage vessels at anchor, which has been estimated to be as much as 2,733 m^3 (Watson et al. 2022). The presence of suspended sediment alters the physical environment triggering a variety of responses from aquatic organisms which can include reduced bivalve pumping rates, visibility, and even direct mortality (Wilber and Clarke 2001). The results from our study do not appear to suggest that this is taking place, as there was no significant difference in abundance, biomass, or species richness across sites from different anchoring pressures. In contrast, bottom trawling has been shown to have significant impacts on the sediment dynamics of an area through sediment resuspension (Churchill 1989), remineralisation of nutrients and contaminants (Kaiser et al. 2010) and leading to changes in particle size distribution and suspended sediment settling in areas adjacent to trawling sites (O'Neill and Ivanović 2016). However, in this study there was a lot of variability in the mean abundances per site, which indicates that these benthic communities are highly dynamic, and suggests on a site level there are likely to be other variables that influence the community structure such as bed shear stress. Infaunal communities naturally display high levels of temporal and spatial variation even over small scales and consequently the impacts of disturbance such as that by anchor gear can be difficult to detect and quantify (Thrush 1991; Drabsch et al. 2001). Infaunal biota have rapid growth rates (Haig et al. 2012) and typically show short recovery times (~months) after disturbance (Backhurst and Cole 2000). Thus, attempting to assess the long-term effects of anchoring without weighting the timing of a specific anchoring activity may produce confounding results. In addition, it can be difficult to accurately assess the impacts of anthropogenic disturbances and recovery as there are very few sites where disturbance has not already been going on for a considerable amount of time and therefore a shift in the ecological baseline could already have taken place at these sites (Pauly 1995). Therefore, it could be suggested that the sustained popularity of the anchorage for vessels waiting to access the Mersey or dock at Liverpool over recent decades has resulted in such widespread effects that the scale of this experiment cannot detect the shift in ecological baseline of the wider area that has already taken place.

Conversely, anchoring may not be as damaging at this study site, as has been speculated in the available literature; therefore, it may be correct that these results show no effect of anchoring on infaunal communities. It is reasonable to assert that the effects of relatively small-scale merchant anchoring disturbances may be masked by the regime of natural disturbances that occur at the study site. The infaunal communities that inhabit the study site may be resistant to disturbances caused by anchoring activity because of the resilience required to withstand the environmental conditions to which they are exposed. This is supported by literature where benthic communities that experienced high shear stress are less affected by bottom trawling (Rijnsdorp et al. 2018).

There was no relationship between cumulative anchoring pressure and organic carbon content, which is in contrast with the literature, as trawled areas have shown both increased and decreased organic carbon content in addition to increased coarsening of sediment at sites with high trawling activity, indicating that winnowing takes place (Palanques et al. 2014; Epstein et al. 2022). This study may have been improved by analysing sediment grain size at different sites as this has been shown to play a significant role in determining the composition of a soft-sediment community (Gray 1981; Etter and Grassle 1992) because deposit feeders selectively ingest sediments of specific particle size (Taghon 1982; Self and Jumars 1988) and there are interspecific differences in particle size preference (Fenchel et al. 1975).

While the detrimental effects of anchoring activity in habitats such as seagrass and coral reefs are clear, and recovery times are generally well documented, anchoring activity in unvegetated sediments can be more difficult to assess because the effects are more difficult to observe. Anchor scour directly damages seagrass fronds (Hastings et al. 1995), uprooting shoots (Milazzo et al. 2004), and creating anchor pits(Collins et al. 2010), with full recovery of the ephemeral seagrass *Halodule wrightii* (Williams 1988) reportedly taking nine months (Creed and Amado Filho 1999). In 1998, a cruise ship dropped its anchor onto a coral reef in Virgin Island National Park, creating an anchor scar ~128m long and 3m wide, where coral cover did not significantly increase over the subsequent decade (Rogers and Garrison 2001b). For soft sediments it is generally much more difficult to assess the level of community disturbance and recovery because of the nature of the sediment habitat. In addition to a scarcity of research into the effects of merchant vessel anchoring in unvegetated soft sediments making predictions based on the existing research into the recovery of benthic habitats can be problematic. Defining ecosystem recovery is complex and rarely is complete restoration of an ecosystem to pre-disturbance state, including identical states of abundance, diversity, structure, and function (Hiscock and Tyler-Walters 2006) used as a metric for recovery, with most studies focusing on the recovery of key species (Foden et al. 2011). Recovery from bottom fishing is dependent on the frequency and the gear used (Kaiser et al. 2006) and biomass and productivity can be inhibited if fishing occurs too frequently for recovery to take place (Hiddink et al. 2006).

Presumably, there exists a critical threshold of frequency and duration of anchoring activity that, once surpassed, will lead to detectable and critical and sustained ecological effects despite the natural regime of disturbance. However, these results provide limited insight into this as it remains difficult to establish whether this critical threshold has been reached in the Point Lynas anchorage. It may be that at the Point Lynas anchorage the anchoring activity is fragmented and sporadic such that the frequency at which a particular area of the seabed is anchored allows recolonisation and recovery to pre-disturbance levels to occur rapidly and uninterrupted by further anchoring disturbance; therefore, the effects of the disturbance remain largely undetectable. It is important to understand how anchoring intensity across unvegetated sediment habitats affects communities, so that management regimes that balance anchoring requirements with habitat conservation can be designed and implemented. It is also possible that this study failed to detect an effect that exists which would constitute a type II error and would most likely have occurred due to a low power of the study to detect effects of anchoring due to the limitation of having few replicates particularly few replicates from higher anchoring pressures.

Higher concentrations of epifaunal invertebrate scavengers (*P. bernhardus, A. rubens, O. ophiura and B. undatum*) would likely be found at areas very recently anchored (2-3 days) as these species feed on damaged and exposed crustaceans, whelks, polychaetes, echinoderms, and bivalves (Ramsay et al. 1998). A long-term increase in scavenger species with trawl disturbance has also been shown which is likely due to increased food availability and more favourable conditions for growth and reproduction (Rumohr and Kujawski 2000). For this reason, it is unfortunate that because of Covid-19 restrictions and time limitations, abundance and biomass data were not available for analysis of all organisms from the beam trawl. It is important to quantify the most abundant species well because they are most likely to show a strong response to disturbance. However, it is a disadvantage to not know their total abundance and weight. Therefore, future studies that investigate pre- and post-disturbance species abundance to investigate the short-term effects of anchoring on these soft sediment habitats may help draw stronger comparisons between the effects of trawling and anchoring. However, epifauna data derived from seabed images also did not show an increased number of scavenger species in replicates at higher anchoring pressures. When assessing why there is no significant difference in the number of scavenger species at higher anchoring pressures, it is important to consider how the natural disturbance regime affects the resilience of the community to anthropogenic disturbance. Many authors have previously suggested that the effects of anthropogenic disturbances are inextricably linked to the regime of natural disturbance at a site (Kaiser 1998; Auster and Langton 1999). It seems logical to assert that organisms that inhabit unconsolidated mobile sediments with high levels of natural disturbance would be adapted to elevated turbidity, more intense physical disturbance, and periodic smothering by the settlement of suspended sediment (Kaiser et al. 2003), Therefore would be less impacted by anthropogenic disturbance. For example, Van Denderen et al., (2015) showed that communities exposed to high levels of natural disturbance (0.3 -1.4 N m^2) show limited or no trawling effect whereas, responses to trawling were found in areas of low bed shear stress and resulted in communities of small-sized, deposit-feeding animals, mobile scavengers, or predators. The mean bed shear stress, as established by SEACAMS2, Bangor University, of the study area ranged from 0.52 N $m\text{-}^2$ to 0.76 N $m\text{-}^2$. it may therefore reasonably be asserted that this study area with its soft sediment composition and high level of natural disturbance results in a benthic community that is resilient to the levels of anchoring disturbance observed at the site.

7.1 Strengths & Limitations

A strength of this study relates to the wide variety of datasets that are collated here. The inclusion of grabs, trawls, and seabed images allows greater confidence in the conclusion that there is no detectable impact of merchant vessel anchoring at the Point Lynas anchorage in this study. This study displays the effective use of AIS data to quantify and assess the nature of anchoring pressure across an area. Similarly, the inclusion of three years of anchoring data and the development of this method for assigning precise anchoring values to specific samples taken using AIS data have produced a quality anchoring dataset to allow for these conclusions to be drawn. However, it is prudent to consider that anchoring pressure may not be well estimated using these methods. It may be possible that the area impacted by anchoring, as shown in figure 5, may be an overestimate or underestimate of the actual area impacted by the anchor and chain. If this were the case, then the anchoring pressure value assigned to samples near the borders of these impact areas may be over or under inflated. This may further reduce the power of the study to detect effects by confounding the results. In addition, this method of assessing anchoring damage may not be suitable across all sites and anchoring studies as the use of AIS data excludes many smaller recreational vessels which may represent more anchoring vessels at some sites.

It is also important to note that this study suffers from the limitation of lack of replication at higher anchoring values. This means that the samples with the highest anchoring pressures were often excluded from the analysis because there were no suitable replicates, limiting the conclusions that could be drawn. Owing to the fragmented and sporadic nature of anchoring activity and the development of the AIS technique for assessing anchoring activity after the sampling design and research cruise, there are many replicates that come from zero anchoring pressure, and much fewer from higher anchoring pressures. This limits the conclusions that can be made, as the range of anchoring pressures that have been tested is relatively small and the number of samples included in these replicates is low, with a minimum of two grab samples per replicate and a minimum of five seabed images per replicate. This results in lowered power to detect the effects of anchoring on communities due to the relatively small sample sizes. The study may have benefitted from uncoupling the samples taken from their original sampling station and grouping them into replicates based solely on anchoring pressure, as environmental conditions were broadly consistent across the site. Separating samples from the initial sampling station may have revealed effects which may have been masked by the exclusion of samples with matching high anchoring pressure but that originated from different initial sampling stations. If the study were repeated, it would be prudent to attempt to increase the number of replicates across the board but with particular emphasis on collecting more samples from higher anchoring pressures with more targeted sampling at specific sites of high cumulative anchoring pressure, as this greater range of anchoring may produce greater insight into the effects on benthic community abundance and biomass of anchoring at higher frequencies. In addition, a greater number of samples taken would allow for the minimum number of samples required for replicates to be increased. This would hopefully eliminate the occurrence of replicates that contain no identifiable benthic fauna as occurred in some of the seabed image replicates.

The present study may have been improved by categorising anchoring activity with respect to how recently anchoring occurred. Currently, equal weighting was given to anchoring, regardless of when anchoring took place in the last three years. In this method, a sample taken from an area anchored twice 2.5 years ago would likely have a higher cumulative anchoring pressure than a sample taken from an area anchored once just 1 week before sampling, and which may have skewed the results or have unduly influenced the conclusions that could have been made about the effects of anchoring on communities as those areas with high anchoring pressures from a long time ago are likely to have undergone significant recovery since the disturbance. Future studies should, therefore, consider how long ago the anchoring activity occurred to give more weight to more recent anchoring events while still attempting to consider the long-term effects of anchoring activity.

8 CONCLUSION

The AIS data analysis in this study has revealed that anchoring activity across the study area is highly spatially localised thus grab samples, seabed images and beam trawls have not revealed clear changes in community indicators in response to anchoring. I conclude that high levels of natural disturbance in the soft-sediment habitat at the Point Lynas anchorage likely results in a benthic community that is resilient to anchoring disturbance or one in which the effects of anthropogenic disturbance are masked by the natural regime of disturbance. In addition, the insight gained from this study are likely limited by the total number of replicates and in particular the relatively small number of replicates from higher anchoring pressures which has potentially resulted in low power to detect the effects of anchoring pressure.

These results highlight the inaccuracies that may result from attempting to extrapolate the effects from studies on other abrasive activities such as trawling to predict ecosystem effects of merchant vessel anchoring. It is evident that subtle cumulative effects of anchoring in soft sediment communities may emerge only when anchoring disturbances are investigated over greater spatial and temporal scales and across a gradient of natural disturbance and this would be my recommendation for further study. Identifying and comprehensively understanding the effects of merchant vessel anchoring will allow for the development of effective strategies for the mitigation and management of anchorages and the accurate assessment of benthic habitats subject to anchoring pressure.

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