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Developing appropriate, and ecologically significant, sampling units for broad scale monitoring of fisheries activity using up to date Vessel Monitoring System data.

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Developing ecologically significant sampling units for fishing pressure using Vessel Monitoring System data for the purpose of planning broadscale benthic monitoring surveys

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Executive Summary

The UK Marine Biodiversity Monitoring R&D Programme aims to deliver status and trend information capable of detecting change in the condition of marine habitats and species across the whole UK marine environment, both within protected sites and outside. The Programme is led by JNCC on behalf of the Statutory Nature Conservation Bodies and delivered through the UK Marine Monitoring and Assessment Strategy (UKMMAS).

This study was carried out jointly by Cefas and JNCC in partnership to investigate and test how currently available data on fishing activity obtained from Vessel Monitoring Systems (VMS) can be used within the UK Marine Biodiversity Monitoring R&D Programme to help understand and monitor the effects of fishing activity on seabed habitats.

The UK Continental Shelf covers an area greater than 870,000km^2 and therefore efficient sampling strategies will be required to collect monitoring data from a representative sample of that area. Sampling design is expected to use a risk-based approach to sample allocation in order to make the design cost-effective. Within the Programme, risk is defined as 'the risk to habitats and species of being adversely affected by pressures caused by human activities'. Collecting data along a gradient of pressure will not only result in an efficient sample design, but also provide new data concerning the response of habitats and species to various levels and combinations of pressures.

The primary aim of this study was to test the suitability of currently available data layers displaying offshore fishing activity for planning surveys to monitor seabed habitats. The results would also inform the development of physical abrasion pressure layers. At present, JNCC use aggregated VMS data gridded at 0.05dd (decimal degrees) for offshore fishing activity data layers. Data are aggregated in order to anonymise individual fishing vessels and at this spatial scale because vessel positions are recorded at least every two hours.

A survey was carried out at two sites in December 2011: Eastern English Channel (EEC) and Outer Thames Estuary (Th). These sites were selected because, for the time period of interest, they represented two of the most fished areas of the UK seabed in terms of hours fished per annum with demersal gear and had similar sandy sediment types. A third site was intended to be sampled, but because of time lost due to bad weather during the survey, it was not possible. Changes in infauna and epifauna across a gradient of fishing pressure were measured, and analysed using univariate and multivariate statistical techniques. Alternative ways of spatially expressing VMS data were then tested against benthic community data to find which was relatively best at explaining any observed biological variability.

Regional maps of fishing pressure abrasion at the two sites for the 18 months prior to the experiment were produced using a point summation method for grid sizes of 0.05^{o}, 0.025^{o}, and 0.0125^{o} and two variations of nested grid with an estimation of swept area as proposed by Gerritsen *et al* (2013). Regional maps of fishing pressure abrasion scores were visually different, depending on the method chosen to present the processed VMS data, despite the maps being based on the same underlying raw data. Areas of "no fishing" were only apparent when smaller grid sizes were used, indicating that larger grids overestimate the spatial footprint of fishing. This overestimation of fishing activity was more apparent when fishing activity had a patchy distribution, such as in the Outer Thames, whilst those areas of homogeneous fishing activity, such as at Eastern Channel, appeared to be more robust to changes in the method of presentation of VMS data. As the cell size decreased, fishing effort could be seen to be reapportioned unevenly within the larger cells, indicating that an assumption that fishing effort is homogeneously spread within each cell was not always valid for larger grid sizes.

Sediment characteristics, including sediment group, silt (%), and organic carbon content were found to be the predominant factors driving variability in biodiversity indices across the fishing pressure gradients at the two sites. These results suggest that, in this predominately sandy habitat, sediment variability is having a larger influence than any observed effect of demersal

fishing. In a simple linear regression, fishing pressure had a statistically significant, positive, but small effect on all biodiversity indices measured (abundance, species richness and biomass) in this sandy habitat. The high proportion of scavengers at both sites and the *W*-statistics suggest that the benthic communities may already have been adapted to a perturbed environment, either because of natural disturbance due to strong tidal currents or as a result of historical fishing pressure. This might explain why the fishing pressure gradient had a relatively small effect. Similar studies have found that recently trawled sand habitats can have elevated species richness due to an increase in scavengers that benefit from feeding from dead and damaged species as a result of trawling. However, in this study there was no clear pattern in species or traits to explain this increase. Equally, it has been shown in some studies that fishing is more common in areas with higher benthic biodiversity, leading to a positive trend. The overall small observed effect size of abrasion from fishing pressure on biodiversity indices could be due to a combination of other interrelated factors, including the heterogeneity of fishing pressure within a cell, natural disturbance of the biological communities, historic fishing disturbance, and/or the resilience of the current benthic community to pressures associated with demersal fishing. Regardless of the direction of the relationship between fishing abrasion pressure and biodiversity indices, it appeared largely independent of the method of processing VMS data, as no particular method of deriving pressure gave a consistently better fit for the combined dataset from both sites.

At Outer Thames, there were more significant relationships between fishing abrasion pressure scores and biodiversity indices and the models were a better fit when using a larger grid size (0.05dd). At Eastern Channel, there were more significant relationships between biodiversity indices and fishing scores derived from the smallest grid size (0.0125dd). Across the entire dataset, the lack of consistent trend could be due to the different patterns of fishing activity at the two sites discussed above. This result suggests that the size of grid used to aggregate fishing data in different regions could be critical in terms of explaining variability in benthic communities, as well as visual presentation of the distribution of fishing activity.

This study deliberately focused on a single, standardised, broadscale sedimentary habitat. The habitat was chosen because it coincided with some of the highest levels of demersal fishing on the UK Continental Shelf. Multivariate analysis of epifauna from both sites demonstrated that there was a difference in epifaunal community composition and structure between the two sites, which were both identified in EUSeamap as the same broadscale seabed habitat. This may have been partly due to the relatively low sample size, which was a result of bad weather compromising the original survey plan, but also illustrates the challenge of using predictive broadscale habitat maps developed from modelled data, to plan such surveys, where finer scale seabed habitat maps based on more detailed remote sensed data and ground validation would be preferable.

Of the environmental parameters measured, sediment gravel content (%) and fishing pressure (0.025dd) best described multivariate variability at EEC. Many factors were found to describe the community patterns observed at Th, suggesting we had not identified which factors were actually structuring the benthic community.

Infauna communities were predominately composed of free-living predators and scavengers, suggesting a high level of disturbance. There was an observed relationship between fishing activity and a gradual increase in the 11-20mm infauna size class within EEC and a slight decrease in the 21-100mm infauna size class at Th, demonstrating an overall trend towards a smaller species size as fishing activity increased. Smaller organisms are sometimes able to survive physical abrasion caused by demersal trawling, due to displacement from the seabed by a pressure wave that forms in front of the towed gear. In the future, these findings could be used to select biological traits that respond consistently to fishing pressure abrasion to develop into future monitoring indicators.

It is recommended that in the future a similar study be carried out on different seabed habitat types with a higher sensitivity to benthic fishing pressure and/or where historical fishing has not taken place in order that patterns can be observed across different sedimentary habitat types in relation to fishing pressure gradients.

In the two regions where the study was carried out, fishing activity was predominately beam trawling, but different fleets fishing on other ground types could behave differently and hence require an alternative approach to aggregating VMS data. In future work, either a compromise could be chosen to suit most fishing behaviours or the most appropriate grid size could be chosen to reflect the heterogeneity of fishing activity in a given region.

In terms of developing more effective methods for monitoring, it can be seen that, in instances where habitats are relatively resilient, a large survey effort is required to pick up significant changes caused by abrasive fishing pressure. This could be considered intuitive, as more experimental power is required to detect a smaller effect size. It is suggested that a prioritisation exercise is required, based on cost-benefit analysis, to see whether the effect size (i.e. benthic response to pressure) which can be detected is proportionate to the increased survey cost.

The findings from this study highlight the importance of considering the most appropriate scale for the end user when aggregating VMS point data to grids. As shown, when aggregating VMS data to a regular grid, there is a risk of over- or underestimating fishing pressure spatially, depending on the scale of the grid selected.

In carrying out this study, we have demonstrated that monitoring surveys can be planned along fishing pressure abrasion gradients using our current aggregated fishing activity data layers when the seabed habitats are relatively homogeneous, and fishing pressure is also relatively homogeneous and well represented at the grid size used. Our primary finding is that, of the new methods for processing fishing activity data tested here, none are consistently significantly better or worse than the others in terms of analysing benthic variability in offshore sandy habitats. In this study, differences in environmental characteristics, particularly sediment characteristics, were the main factors driving differences in biological composition of benthic communities. Fishing pressure had a small statistically significant effect on the benthic community, and this further highlights the need for a robust survey design in order to elucidate subtle relationships between biology and interrelated drivers, both natural and anthropogenic. When considering pressure-state relationships, background variability must be taken into account and this appears to vary with region, as different factors were demonstrated to be important within similar, but geographically distinct, habitats.

This review has been peer-reviewed by Gwladys Lambert of Bangor University who is a leading expert in the field of VMS processing and analysis. It was not possible to address all of her comments in this report, but all of her suggestions and resulting actions have been annexed to the report and will be addressed in the development of this work and tested in future offshore Marine Protected Area (MPA) surveys. This study forms part of the ongoing development of the UK Marine Biodiversity Monitoring R&D Programme led by JNCC.

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1 Introduction

1.1 JNCC-Cefas partnership

The Joint Nature Conservation Committee (JNCC) is a Committee of non-departmental public bodies working at the interface of science and policy that advises the UK Government and devolved administrations on UK-wide and international nature conservation. JNCC is responsible for developing a biodiversity surveillance and monitoring strategy for the marine environment, and undertaking the development of some components of this strategy. It is also responsible for a contribution to the establishment of offshore marine protected areas. These will form the basis of independent advice to governments to meet statutory duties on both JNCC and governments. Additionally, JNCC lead on the development of advice to governments on options for monitoring seabirds, cetaceans and benthic habitats, both within and outside marine protected areas.

The Centre for Environment, Fisheries and Aquaculture Science (Cefas) is a multidisciplinary scientific research and consultancy centre providing research and advisory services to government and the private sector. It has a comprehensive portfolio of research, consultancy and training services in fisheries science and management, environmental monitoring and assessment, and fish and shellfish health, hygiene and cultivation. It also operates a sophisticated offshore field sampling capability provided through a network of research vessels and commercial platforms.

JNCC and Cefas have formed a formal partnership, in order 1) to develop relationships within JNCC and Cefas, 2) to enhance the provision of monitoring, surveillance and assessments in UK waters, and 3) to ensure delivery of high quality products that meet the aspirations of both partners. This collaborative partnership is intended to make a significant contribution, and offer a cost-effective approach, to the development and delivery of a regional sea surveillance and monitoring programme, especially in the offshore area. It also aims to make more efficient use of the expertise and survey resources of the two organisations.

JNCC considers that collaboration with Cefas could make a significant contribution, and offer a cost-effective approach, to the development and delivery of any regional sea surveillance and monitoring programme, especially in the offshore area. It could also establish principles for the future application of such partnerships across government.

Within this integrated project, one of the overarching aims is to develop ways of working effectively as partners in relation to evidence-gathering, the use of data for multiple cross-organisational purposes, and in carrying out integrated offshore surveys to satisfy the requirements of both partners.

1.2 Legislation protecting benthic marine biodiversity

Concern over the possible impacts of anthropogenic pressures on the marine and coastal environments has led to development of national, regional, and global legislation that aims to preserve and, where possible, to mitigate impacts on marine environments. At the European level, the Marine Strategy Framework Directive (MSFD) and the OSPAR Biological Diversity and Ecosystems Strategy both require assessment of human activities within the marine environment (Benn *et al* 2010).

The MSFD requires Member States to take measures to achieve or maintain Good Environmental Status (GES) for their marine waters by 2020. In the UK, the Directive was transposed into law through the Marine Strategy Regulations 2010. Member States are required to develop Marine Strategies that include: an initial assessment of their marine waters; characteristics, targets and indicators of GES; monitoring programmes for measuring progress towards GES, and; programmes of measures to achieve or maintain GES. There are 11 qualitative descriptors within the Directive for determining GES:

- D1: Biological diversity,
- D2: Non-indigenous species
- D3: Population of commercial fish/shellfish
- D4: Elements of marine food webs
- D5: Eutrophication
- D6: Sea floor integrity
- D7: Alteration of hydrographic conditions
- D8: Contaminants
- D9: Contaminants in fish and other seafood for human consumption
- D10: Marine litter
- D11: Introduction of energy, including underwater noise.

As statutory advisor to the UK Government, JNCC has a role in advising on aspects of the MSFD which relate to biodiversity and ecosystem protection. This includes advice in relation to both monitoring and assessment of the state of the marine environment, and the programmes of measures needed to achieve or maintain GES. JNCC is facilitating, via the Healthy and Biologically Diverse Seas Evidence Group (HBDSEG), the development of options for GES targets and indicators for several Descriptors. Part of this work aims to increase the understanding of the contribution of anthropogenic activities to pressures that affect the marine environment.

Marine Protected Areas (MPAs) are one of the main tools available to conservation scientists and policy makers in the UK to protect marine benthic habitats. JNCC provide advice to governments on the identification, management, monitoring and assessment of MPAs. Currently, UK MPAs are a combination of Special Areas of Conservation (SACs) from the EC Habitats Directive (92/43/EEC) and other MPAs that have stemmed from the Marine and Coastal Access Act 2009 and the Marine (Scotland) Act, Marine Conservation Zones (MCZs) and Scottish MPAs respectively. As a signatory to the OSPAR Convention, the UK is committed to establishing an ecologically coherent network of well-managed MPAs to help conserve marine ecosystems and biodiversity (Tillin *et al* 2010). This network will also help support the achievement of MSFD GES descriptors that would benefit spatial protection measures. These sites must be monitored by UK agencies to ensure they are in an acceptable condition, sufficiently managed, and are not at risk from human activities in accordance with the above legislation.

1.3 Monitoring marine biodiversity in the UK

The UK Marine Biodiversity Monitoring R&D Programme is led by JNCC on behalf of the Statutory Nature Conservation Bodies and delivered through UKMMAS. The Programme aims to deliver status and trend information capable of detecting change in the condition of marine habitats and species across the whole UK marine environment, both within protected sites and outside. The UK continental shelf covers an area greater than 87 million hectares (much greater than the land area) and therefore a sampling strategy is required to collect data from a representative sample of this area.

The final sampling design is expected to use a risk-based approach to sample allocation in order to make the design cost-effective. Within the Programme, risk is defined as ‘the risk to habitats and species of being adversely affected by pressures caused by human activities’. The implementation of a risk-based approach involves the identification of habitats and species which are at different levels of risk (with an associated confidence level), depending on the pressures that they are exposed to. It will then be possible to stratify sampling of habitats and species along a gradient of risk using these data, e.g. taking samples from a single habitat type at low risk, medium risk and high risk. Collecting data along such a gradient will not only result in an efficient sample design, but also provide new data concerning the response of habitats and species to various levels and combinations of pressures.

1.4 Priority pressures on the UK seabed

A list of defined pressures has already been formally agreed by the OSPAR Intersessional Correspondence Group on Cumulative Effects (ICG-C) and been accepted by HBDSEG and PSEG (the Productive Seas Evidence Group). From this list, a prioritisation exercise has identified different priority pressures, based on the findings of Charting Progress 2, Scotland's Marine Atlas, and the 2010 OSPAR Quality Status Report (UKMMAS 2010; Baxter *et al* 2011; OSPAR 2010). In prioritising anthropogenic pressures on the marine environment, consideration was given to spatial extent of pressures, the coincidence of pressures and receptors, and the intensity/significance of the effect of the pressures on the receptors. From this exercise, the highest priority pressures on seabed habitats were considered to be removal of target and non-target species by fishing and physical habitat damage. Abrasion and other physical damage was ranked as one of the highest priority pressures, predominately due to the contribution of demersal fishing activity, such as bottom trawling (e.g. Baxter *et al.* 2011; OSPAR 2010). A description of the pressure, as defined by ICG-C, is given below:

<u>Penetration and/or disturbance of the substrate below the surface of the seabed, including abrasion</u>
The disturbance of sediments where there is limited or no loss of substrate from the system. Abrasion relates to the damage of the sea bed surface layers (typically up to 50cm depth)... Activities associated with abrasion can cover relatively large spatial areas and include: fishing with towed demersal trawls (fish & shellfish); bio-prospecting such as harvesting of biogenic features such as maerl beds where, after extraction, conditions for recolonisation remain suitable or relatively localised activities including: seaweed harvesting, recreation, potting, aquaculture. Change from gravel to silt substrate would adversely affect herring spawning grounds.

Physical damage to the seabed as the result of abrasion caused, for example, by commercial demersal fishing, is also identified on the indicative lists of characteristics, pressures and impacts in Annex III of the MSFD (2008/56/EC).

Physical abrasion can directly damage seabed habitats and their associated species. Physical abrasion is likely to reduce the structural complexity of seabed features, damage erect epifaunal species and reduce biodiversity, through the selective removal of large, sessile, long-lived species from the community (Sewell & Hiscock 2005). Such species may be replaced by more mobile species, rapid colonisers and juvenile stages (Gubbay & Knapman 1999; Sewell & Hiscock 2005). Increases in suspended sediment levels as a result of abrasive activities such as dredging may adversely affect some filter feeding organisms by clogging filter and respiration mechanisms (Hartnoll 1998). Additionally, a shift in soft sediment habitats from sand and gravel to finer sediments may result in a decrease in species richness, abundance and biomass (Desprez 2000).

In order to be able to apply our understanding of this pressure in relation to possible prioritisation of monitoring effort, it is important to understand how best to spatially represent its constituent contributing anthropogenic activities, so that sampling designs can be planned accordingly. In particular, the Charting Progress 2 report (UK Monitoring and Assessment Strategy 2010) highlighted bottom towed fishing gear to be one of the most widespread, yet manageable, anthropogenic impacts on the seabed of the UK continental shelf. Knowing this demonstrates a requirement to understand the spatial distribution and intensity of fishing activity within UK waters before attempting to investigate the effects on benthic ecosystems.

1.5 VMS data as a proxy of fishing activity

Bottom trawling is one of the most spatially extensive anthropogenic contributors to both priority pressures: removal of target and non-target species and physical habitat damage (Jennings & Kaiser 1998, Eastwood *et al* 2007, Benn *et al* 2010, Tillin *et al* 2010). It is therefore considered vital that the effects of this activity are investigated and understood against a wider context of natural community variability, so as to distinguish natural and anthropogenic effects (Lambert *et al* 2012). In order to assess the scale and context of effects of bottom trawling, it is necessary to define the level of this activity occurring within the marine environment.

In UK waters, fishing vessels greater or equal to 15m in length have been required to be fitted with a Vessel Monitoring System (VMS) since 2005. VMS data provide information on vessel activity at regular intervals and may be used as an indication of fishing intensity (e.g. Lambert *et al* 2012). Although there are many vessels under 15m fishing in UK waters (98.4% of the UK fleet in 2010), the majority of activity in offshore waters (waters beyond 12 nautical miles, within British Fishery Limits and the seabed within the UK Continental Shelf Designated Area) is from vessels equal to or exceeding this overall length. As JNCC has responsibility for the provision of nature conservation in offshore waters, where this research was carried out, it was considered acceptable, within the parameters of this study, to consider only activity from vessels greater or equal to 15m in length, hence to use VMS data as a proxy of activity. Smaller vessels were therefore not represented.

Vessel Monitoring System (VMS) data were initially collected for the purpose of control and enforcement of fisheries under legislation introduced by the European Commission (European Commission 1997). Interpretations of fishing fleet activity have long been attempted prior to the introduction of VMS. Techniques such as estimates from vessel sightings by fishery protection aircraft and anecdotal evidence from local fishermen are employed in attempts to understand fishing fleet behaviour (Jennings *et al* 2002; Duplisea *et al* 2002):

- Jennings *et al* (2002) used fisheries sightings data to estimate fishing effort and reported as sightings of beam trawlers per unit of searching effort (SPUE);
- Rijnsdorp *et al* (1998) used an automated recording system with an accuracy of approx. 0.1 nautical miles to track the movements of 25 Dutch commercial beam trawlers. Positions were collected by connection to an onboard GPS navigator and data recorded onto a removable memory card. Data collected in this way were prone to system failures as well as hardware malfunction and were only able to provide a small snap shot in time and space of a single fishery, with the 25 vessels being scaled up to account for the total fleet activity; and
- Hinz *et al* (2009) estimated trawling intensity and spatial distribution by combining over-flight data and log book records of hours spent fishing per ICES rectangle, collected by the then UK Marine Fisheries Agency (now the Marine Management Organisation).

The introduction of VMS allowed delivery of high volume empirical data on fishery vessel movements without incurring additional costs from direct observation and without relying on anecdotal evidence from industry insiders. It also provided data at a good spatial resolution when viewed at a regional or national level, allowed time-series data to be gathered, and provided a standard whereby different studies could be compared. Prior to its introduction, measures of fishing intensity were relative and specific to each study (Hinz *et al* 2009). At its introduction in 2000, only fishing vessels of >24m overall length were included, but, more recently, the value of the data has been realised, and vessels of smaller size are now legally obliged to comply with the VMS requirement (Eastwood *et al* 2007). At time of writing, all fishing vessels ≥12m are included, though historic data are missing for vessel size classes up until their inclusion within the system (European Commission 2009).

VMS transmits, approximately once every 2 hours, information on vessel identity, speed and heading for all areas and for all UK and non-UK vessels fishing within UK waters to the MMO (e.g. Lee *et al* 2010a). Even though more frequent pings generate higher spatial resolution on fishing vessel movement (Lambert *et al* 2012), the costs associated with the increased amount of data are often prohibitively expensive, given that the primary purpose of collection is enforcement of fisheries regulations. Furthermore, it is unlikely that higher frequency VMS would be available within the timescales required to comply with advice to UK Governments. With that in mind, efforts are being made to investigate VMS data at its current collection frequency. Care is taken, when discussing processed VMS outputs, to discuss fishing activity rather than fishing effort, as the methods applied are interpreting vessel activities rather than establishing a link to the performance of any of the deployed gear (Jennings & Lee 2012).

1.6 Mapping of pressures associated with fishing

The introduction of VMS in northern European fisheries has allowed scientists to investigate, for the first time, the response of benthic fauna to a gradient of fishing activity in time and space (Jennings *et al* 2001a, b; Hiddink *et al* 2006b; Hinz *et al* 2009). For this reason, there has been a huge amount of interest in how to best visualise VMS data at different ecological scales and the most appropriate units to use (e.g. Mills *et al* 2007; Witt & Godley 2007; Lee *et al* 2010a,b). This is significant from a conservation and policy perspective as a better understanding of how pressures interact with habitats could lead to more informed and cost-effective management of marine activities.

Effort has already been invested in examining how best to visualise VMS data, heavily dictated by the aspirations of the end user in each case (Eastwood *et al* 2007; Mills *et al* 2007; Bastardie *et al* 2010; Hintzen *et al* 2010; Lee *et al* 2010b; Jennings and Lee; 2012; Hintzen *et al* 2012; Lambert *et al* 2012; Gerritsen *et al* 2013).

1.6.1 Data holdings and confidentiality

There are certain restrictions governing how VMS data can currently be used that constrain what is possible in terms of precision and data manipulation (Hinz *et al* 2013). Data are collected and held by MMO for enforcement purposes and it is not readily accessible to all for external use, though competent authorities are allowed to use it in discharging their statutory duties. Understandably, some fishers do not want information on their fishing grounds publicly known. However, there are various ways of aggregating the data in order to preserve sensitive information:

- vessel identification tags can be removed to ensure the fisher's anonymity;
- data can be grouped into gear types (e.g. beam trawls, otter trawls and scallop dredges can be grouped to form demersal gears); and
- data can be aggregated spatially to a coarser level or over longer time periods.

Cefas have agreed with MMO that, when a raster grid is used to display data, data are removed from cells where less than five separate fishing vessels have been identified to protect vessel anonymity, thus leading to underestimations of fishing effort in some regions (Janette Lee *pers comm)*.

1.6.2 VMS pings

In its raw format, VMS data can be graphically represented as a series of "pings" that appear as geographically distinct points on a map. Each ping can be represented spatially as a single point generated approximately every two hours. Each point has information associated with it, described earlier, which can subsequently be used during processing and in presentation. For instance, pings can be classified based on gear type or the nationality of the vessel. However, in this format, VMS data are hard to interpret, as it is not possible to see where vessels are between pings, whether there is a consistent direction of activity, or to assess whether vessels are actively fishing.

1.6.3 VMS speed filters

In order to use VMS data as a proxy of fishing activity, the data have to be processed in order to select those that represent active fishing rather than simply indicating vessel position. A typical approach is to filter the data based on vessel speed, which assumes that the vessel actively fishes only at a particular range of speeds for specific gear types. Mills *et al* (2007) identified speed bands for fishing activity:

- Beam Trawls (TBB): 2-8 knots
- Boat Dredges (DRB): 2-8 knots
- Mechanised Dredges (HMD): 2-8 knots
- Bottom Otter Trawls (OTB): 1-6 knots
- Midwater Otter Trawls (OTM): 1-6 knots
- Twin Otter Trawls (OTT):1-6 knots

A more basic approach is to apply a broad range to the whole dataset (i.e. 2-8 knots). Though, it should be recognised that there are no speed records for 17% of VMS points generated (Mills *et al* 2007).

1.6.4 VMS vectors

Alternatively, track reconstruction methods estimate the route a vessel has taken between two pings. This can only be achieved with raw VMS data that has not had the identification information from individual pings removed, as otherwise pings cannot be associated with one another. Track reconstruction itself can fall into two categories:

- Straight line (SL) interpolation draws a straight line between two points as an estimate of the vessel's track; and
- Cubic Hermite spline (cHS) interpolation attempts to take into account the likelihood that a vessel has actually not travelled in a straight line between two points, but instead uses both heading and speed recorded at the start and end points of each particular interpolation to construct a predictive path (Hintzen *et al* 2010).

Eastwood *et al* (2007) mapped the footprint of human activities (specifically direct physical pressure) on the seabed in English and Welsh sectors of UK waters, though they made no attempt to assess intensity, longevity or impacts arising from such activities. As part of their analysis, Eastwood *et al* (2007) found that selective extraction caused by demersal trawling affected a greater area than all other activities combined. They used a speed rule of 1-6 knots for otter trawlers and scallop dredgers and 2-8 knots for beam trawlers. Trawl lines and the swept areas were then created by linking consecutive points into a track and using gear width information to determine the area fished (ICES 2000):

- Beam trawlers: 24m (2 x 12m beams)
- Otter trawlers: 4m (2 x 2m scour from trawl doors)
- Shellfish dredgers: 20.4m (24 x 0.85m wide dredges)

It was acknowledged that this method was likely to generate an underestimate of trawling effort, as it assumes that vessels do not deviate from straight-line tracks between pings.

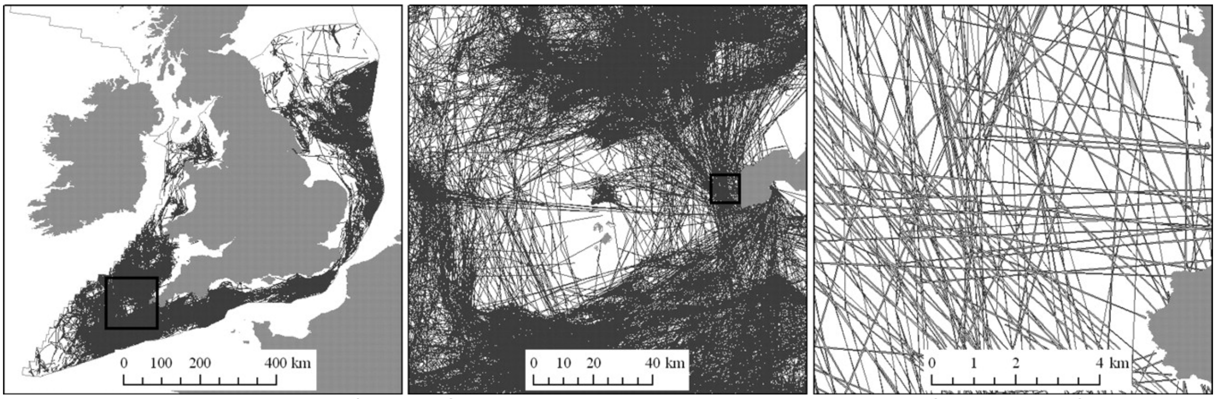

Figure 1. Trawl tracks estimated from VMS records in 2004 shown at three map scales from small (left panel) to large (right panel). The boxes drawn on the left and central panels show the spatial extent of the maps immediately to their right. (from Eastwood *et al* 2007).

1.6.5 Interpolated path VMS

There is an inherent error in drawing a simple straight line directly from point to point to infer a vessel's path (Mills *et al* 2007; Fock 2008). Mills *et al* (2007) attempted to compensate for the underestimate of track length, highlighted by Deng (2005), by expressing the degree of deviation from the straight line in three ways: minimum deviation, maximum deviation and a 'best estimate' of deviation. Fock (2008) similarly approached the generation of trawling estimates by reallocating fishing effort based on a straight line interpolation, whilst still considering the uncertainty of the trawl track. Following a two step process, Fock first defined a ping as fishing and then calculated a corresponding effort value. Each fishing location (or ping) was substituted for four new points, with each differing by one quartile from the next and based on the statistically averaged behaviour of the vessel. cHS takes considerably longer than SL interpolation. cHS constructions are likely to lack confidence in a vessels 'true' path, but do add value to estimates where tracks are used to infer footprint at the seabed that can then be subsequently aggregated to give a value of swept area per unit area. Hintzen *et al* (2010) found this method to provide a better estimate of the true track length than those estimates calculated from a SL interpolation and underestimated the true track length by less than 3% on average. Aggregation of VMS data in this way, and its subsequent visualisation, has been used to effectively identify fishing grounds in order to use the data layer outputs in marine spatial planning (Jennings & Lee 2012; Mills *et al* 2007; Eastwood *et al* 2007).

1.6.6 Raster gridded VMS

In order to help inform selection of Marine Conservation Zones, Cefas (supported by ABPmer) led on development of key socio-economic data layers in UK waters, including production of fishing activity layers per gear type and year, in order to contribute to preparation of an abrasion layer (Lee *et al* 2010b).

The fishing activity layers used VMS data from 2006 and 2007 and were generated by Cefas from VMS, log-book and EU vessel register data. Data included both UK and non-UK vessels and used a speed rule of 1-6 knots for all types of fishing.

Cefas used a point density approach that allowed the user to visualise aggregated VMS pings as cells/blocks that could be created at varying scales from the point data (Fock 2008; Piet & Quirijns 2009; Gerritsen *et al* 2013). This generated a unit of 'hours fished per unit area', where each ping was estimated to contribute approximately two hours of fishing activity to the total for each block.

Lee *et al* (2010b) recognised that "for trawled gears, the values within the raster grids [indicated] estimated hours fished. For non-trawled gears this [could not] be taken as being fully representative of the intensity of fishing as no indication [was] given of the time that set nets/pots etc. [remained] *in situ*. Rather it [gave] an indication of the spatial extent affected by these gears and the likely intensity, based on time spent setting and retrieving gear”.

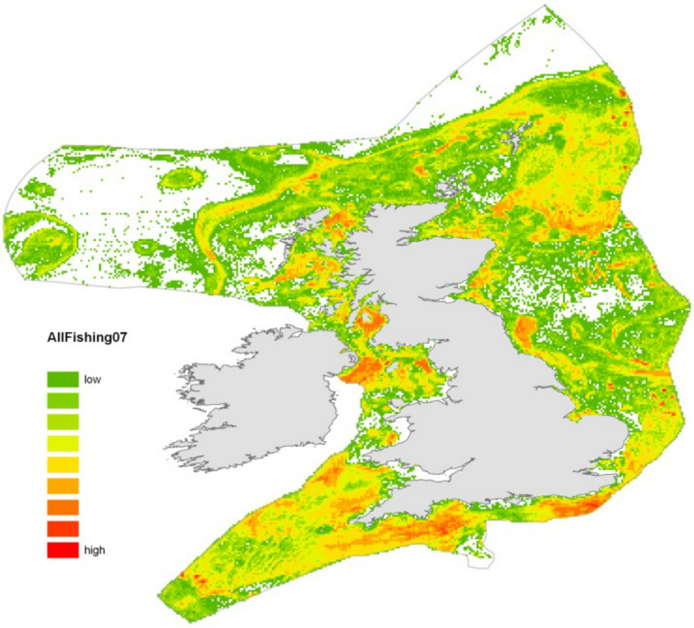

Figure 2. MB0106-derived fishing activity data layer for 2006, all gear types. Taken from Lee *et al* 2010b.

1.6.7 Nested block VMS

Gerritsen *et al* (2013) recently proposed an alternative nested grid approach that attempted to estimate the swept area of the seabed for each VMS ping. The authors used a swept area approach to relate the area of a cell to the proportion of the seabed within impacted by the gear. As VMS points tend to be highly clustered, the number of pings and their distribution within any regular grid will vary greatly. It can be surmised, therefore, that any cell with high numbers of observations will have higher precision than those with a lower number. Gerritsen *et al* (2013) used this rationale to demonstrate the importance of a coarser grid size at regions of low observation density, whilst applying a greater resolution to those areas with a higher number of observations. Hence, a nested grid approach became appropriate in order to best visualise and interpret VMS data holdings.

Starting with an arbitrary coarse grid cell size (0.16° latitude x 0.18° longitude in this case), the technique divided cells containing >20 observations into two and continued this process until no more divisions could be made, or until a minimum grid cell size had been reached (after 11 divisions in this case) (Figure 3). It is the changing cell sizes and the nesting of the cells that makes the Gerritsen method novel. Estimates of swept area per cell made using this methodology are not calculated based on a track interpolation, but rather by summing the effort (time interval since last record in hours) multiplied by the vessels instantaneous speed (km/h) and then again multiplied by the observed fishing gears estimated width that is in contact with the seabed (km).

Dividing the swept area calculation by the grid cell area provides a swept-area ratio that can be used to interpret the mean number of times the seabed in a cell has been impacted by fishing gear. It is, however, important to note that, although a swept area of one demonstrates that the swept area equals the grid cell area, it does not suppose that 100% of the cell has been impacted by fishing, as some areas may have been repeatedly trawled i.e. a swept area of one might suggest the entire cell has been fished once, but equally that, for example, half the cell may have been trawled twice, or that a quarter of the cell may have been trawled four times. Gerritsen *et al* (2013) make the assumption that tracks are distributed randomly and, therefore, derive an estimate of the proportion of a cell that is impacted at least once. Validation of this estimate demonstrated a slight bias, with the swept area ratio slightly overestimating the area that was impacted once and slightly underestimating the area that was impacted more than once. The advantage of this method is its robustness to the time interval between VMS observations, along with there being no requirement to interpolate vessel tracks so that the method is wholly based on real, rather than inferred, data. Currently there has been no investigation of how this method might be used to calculate biologically significant effects at the seabed.

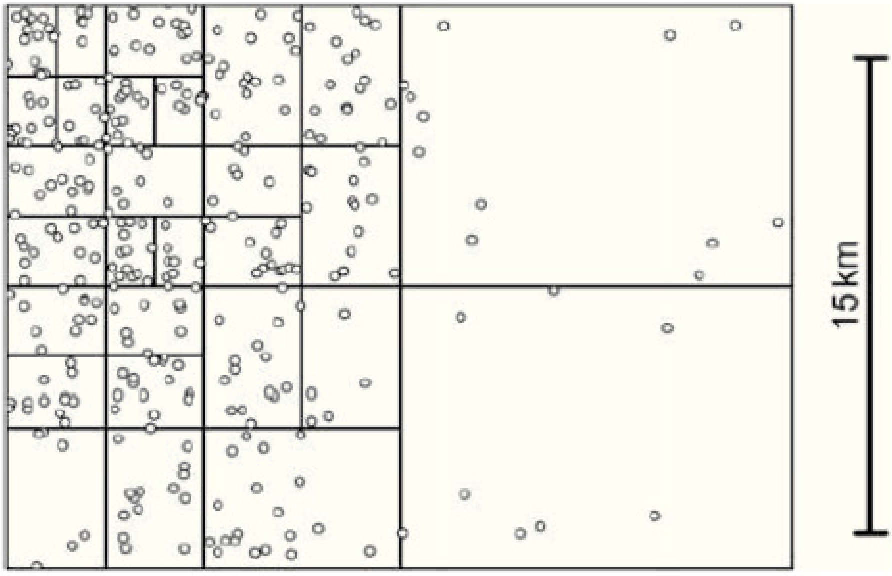

Figure 3. Example of a nested grid from Gerritsen *et al* (2013). Points correspond to VMS records. Cells with ≥20 VMS records are recursively divided by two.

1.7 Limitations in use of VMS

The vessel monitoring system is potentially a powerful tool for a wide array of end users and offers a number of advantages over old methods of estimating the level of fishing activity. There are, however, several issues associated with obtaining and using the data.

1.7.1 Linking VMS to logbooks

VMS data are collected on all UK fishing vessels as well as for all non-UK vessels fishing in UK waters. The associated data for non-UK vessels are, however, not as comprehensive as that collected for UK vessels. In many cases, non-UK vessel gear types are listed as those which are registered to the vessel and are not necessarily the gear type that they most commonly fish with. Also, some gear information is provided at a coarser level and some gear is unspecified. This can lead to estimation errors when attempting to assign relative levels of effort to fishing activity, as well as for swept area calculations. Efforts have been made to link data from vessel log books to VMS activities data to make better estimations of gear type based on landings information (Bastardie *et al* 2010; Hintzen *et al* 2012). Vessel logbooks collect data on quantities and values of the dominant species caught, locations, dates and the gear types used. The logbook location is, however, only reported at the scale of an ICES rectangle, approximately 30 nautical miles x 30 nautical miles (Piet & Quirijns 2009).

1.7.2 Speed filters

Using speed as an identifier of activity has been used to provide valuable information on fleet fishing grounds (Jennings & Lee 2012; Eastwood *et al* 2007). Though it is a commonly used proxy, the actual link between speed and fishing activity can vary greatly, depending on the gear type and the fishery of interest (Lee *et al* 2010a; and references therein). Using speed thresholds in isolation to distinguish between drifting, fishing and steaming is, therefore, not ideal, although improvements have been demonstrated by including vessel heading information (Mills *et al* 2007). As alluded to above, issues can be compounded when speed thresholds are based on vessel gear types that have not been explicitly identified. However, the use of a speed filter cannot at present be completely removed from consideration until measures are put in place to both consistently identify gear types of non-UK vessels and to establish a way of distinguishing those vessels that are actively engaged in fishing.

Lee *et al* (2010a) attempted to lay out a standardised procedure for the processing of raw VMS data that aimed to help facilitate wider exchange and use of fishing activities information, after having made a review of the methods that had already been employed recently. The method they describe is a way of estimating fishing activity from the raw VMS data by discriminating between vessels that are actively fishing from those which may be engaged in other activities:

(1) remove any duplicate records;
(2) remove those pings in the locality of any given port;
(3) calculate time intervals between given records;
(4) establish fishing gear and exclude data points within VMS for which a gear type cannot be assigned;
(5) Use reported speed as a proxy for fishing vs. non-fishing vessels (different gear types may have a different associated 'fishing speed'); and
(6) Make an estimation of the spatial distribution of fishing activity.

Hintzen *et al* (2012) have provided a resource for standardisation of VMS processing, analysis and visualisation. *VMStools* was built using the freeware environment R and is a package of open-source software. The software package itself is capable of combining VMS and logbook data formats and automates the process of distinguishing fishing from other vessel activities. High resolution maps can be produced in order to investigate both effort and landings data and the software is able to interpolate vessel tracks, using either a straight line interpolation or a cubic Hermite spline method that can, in turn, be used in conjunction with an uncertainty estimator of trawl activity or with a method that can represent tracks as gear widths.

1.7.3 Size of vessels carrying VMS

It has been previously mentioned that for the period of this study, VMS data were only collected for those vessels ≥15m. This is likely to have less of an impact for regions further offshore, but is liable to lead to underestimates in more nearshore areas as the majority of vessels fishing inshore are <15m in length, as mentioned above. Although some data are collected on smaller vessels from sightings and boardings, the data are not comprehensive, not comparable between English, Welsh, and Scottish inshore waters, and are not in a format that is easily linked to the data which are gathered under the VMS system. In order to consider fishing activity across all UK waters, it will be necessary, in the future, to try and combine inshore and offshore fishing activities on a meaningful, standardised scale. Landings information collected at a scale of an ICES rectangle represents a way of coarsely estimating the contribution of different size vessels to landings (by weight or value), and hence activity, in a region.

1.7.4 Frequency of VMS pings

One of the core considerations of the data is its limitation to a two hour ping in the UK. The track interpolation methods discussed previously, for example, would be markedly improved by an increase in ping resolution, as this should reduce the possible track variability between two pings (Lambert *et al* 2012). Lambert *et al* (2012) proposed a 30 minute interval between pings, a more suitable polling interval that would not only support a finer grid resolution but also mitigate the effect of increasing costs.

1.7.5 Raster grid scale

Tied in with considerations of ping resolution are other issues associated with subsequent aggregation to gridded formats. Aggregating the data into a grid format allows for a wider appreciation of fishing activity over broader scales, particularly at regional and national level. If the aggregated resolution is too coarse, then broad areas of the seabed appear to at least have some level of anthropogenic disturbance, when in fact fishing effort is not homogenous across a grid cell, so there are large areas which are not fished. However, if the data are gridded at too fine a resolution, then fishing intensity begins to appear overly patchy (Dinmore *et al* 2003; Piet & Quirijns 2009).

At smaller cell sizes, VMS ping frequency introduces cells into the grid where there appears to be no fishing activity. These cells are, however, artefacts of the 2 hour ping rate, as cell size may be reduced below the distance over which a vessel can travel within a 2 hour window. Essentially, a vessel can ping in one block and fish across a subsequent block before pinging in a further block. In this scenario, we have two blocks of the three with an associated fishing score, though all three will have been fished. This kind of artefact would have dramatic implications for study designs that require a prior estimation of impact, the most dramatic example being where the presumption of non-impact is used to assign that grid cell as a control and/or a baseline region. Hence, larger cell sizes tend to over-represent fishing activity, whilst smaller cell sizes tend to underestimate the activity. In most work to date, grid resolution has been directly attributable to study aims at an appropriate scale (Lee *et al* 2010b). Broadscale assessment of fisheries distribution requires different considerations to a localised study attempting to understand fisheries impact at a community level and both require a detailed knowledge of the limitations of VMS data.

1.8 Linking fishing activity to benthic disturbance

1.8.1 Spatial scale

Several studies have attempted to assess the most appropriate way to link VMS data at appropriate scales to possible benthic disturbance. Piet & Quirijns (2009) showed that, for a fished region, the proportion that is thought to be impacted varies depending on both the spatial and the temporal scales used. Their implication was that, for those studies looking to identify fishing-induced mortality of benthic species, a resolution that takes into account the spatial and temporal traits of the impacted communities should be considered. Lambert *et al* (2012) demonstrated that the scale at which VMS data are gridded, to a large degree, dictates the subsequent interpretation of fishing distribution and hence its inferred impact. Dinmore *et al* (2003) suggested analysing fishing intensities at as fine a resolution as is possible, or at a scale where the activity is presumed random. High resolution analysis is, however, hindered by the varying data limitations discussed earlier, with track reconstruction estimates failing to capture the true path of vessels and leading to a poor estimation of fishing intensity and effort. Opinion on the precise scale at which this occurs is a matter of dispute. For instance, though Mills *et al* (2007) suggest an ideal cell resolution of 3x3km, Lambert *et al* (2012) consider this resolution inappropriate for determining the effects of localised disturbance on seabed habitats with specific relevance to the resolution of biological sampling.

An estimation of fishing impact on a specific community will be affected by the time since the last trawling event[1] and the community impact expected is likely to occur at a much finer resolution than that suggested by Mills *et al* (2007). Lambert *et al* (2012) also looked directly into the implications of using alternative methods of VMS analysis to describe fishing activities and impacts. Coupling their results with those of Hintzen *et al* (2010), they showed that, not only did SL and cHS interpolations lead to differing interpretations of the same data, but also that they differed between fleets. The authors attributed this variability to the behaviour of vessels in different fisheries, where estimating long individual tows as found in the Dutch beam trawl fishery, described by Hintzen *et al (*2010), did not equate to the behaviour of vessels within a scallop fishery off the Isle of Man, where fishing was restricted to smaller pockets of habitat and vessels were likely to make multiple turns to cover the same patch of seabed. Fishing, in the case of the scallop dredgers, created a complex track pattern that could not be picked up within a two hour ping frequency, nor reconstructed in a consistent manner using current interpolation methods. The complexity here could be mitigated through increased polling, but would come at a consequently increased cost. In summary, the behaviours of different fishing fleets could be

[1] During the reporting of this study a paper was published by Lambert *et al* (2014) which looked at recoverability of benthic communities and related it to the last trawl event. It is planned that this approach is tested in relation to the UK Marine Biodiversity Monitoring R&D Programme later this year.

included as additional information that would increase confidence in some underlying assumptions (e.g. repeated fishing of same lines), but would also increase complexity.

1.8.2 Temporal scale

Piet & Quirijns (2009) demonstrated that, when assessing fishing effort year-to-year, error might arise due to fishing activity appearing to become more evenly distributed over increasing temporal scales. This is due to the nature of fisheries, which may gradually shift from one year to the next, alternating grounds or searching for more productive grounds. A further complexity lies in tasks looking to identify a temporal scale that will say something meaningful about the habitat condition and/or its recoverability. The key component here is a requirement to draw together habitat exposure to an activity, along with its sensitivity to the associated pressure. Gaining greater understanding in these areas may provide an evidence base for a reporting time frame with clearer relevance to specific habitats and communities. Piet & Quirijn's (2009) work clearly demonstrates the importance of scale for all investigations using such data as that derived from VMS. The interactions between spatial and temporal scaling can have implications for its appropriate application within survey designs and/or reporting assessments and must be carefully assessed at the planning phase.

1.9 Effects of demersal fishing on benthic communities

Many studies have tried to unravel the complex issue of how bottom trawling influences benthic species and habitats. Both Hiddink *et al* (2006) and Tillin *et al* (2006) looked at how long-term trawling affected benthic communities.

Hiddink *et al* (2006) showed a reduction in biomass, production and species richness with chronic trawling activity. Using an assessment of natural disturbance, the study found trends were strongest in areas of relatively lower than higher natural disturbance. Though their model was tested against real data, there was a strong dependence on size as a predictor of mortality and a bias was noted, whereby larger species may be more robust than predicted within the model. For instance, when two species of a similar biomass, but a varying resilience, occur in the same habitat, the effect of trawling on biomass may be masked when following removal of a more sensitive species (such as a seapen) it is quickly replaced by a more robust species (such as a starfish *Asterias rubens).* Though their model simplifies the structure of the benthic community and does not factor in recruitment to the local population, it does provide evidence that trawling affects the productivity of the benthos and, therefore, reduces the potential food availability for fish stocks that prey on them.

Large-scale and long-term shifts in the functional composition of benthic invertebrate communities caused by trawling were also investigated by Tillin *et al* (2006). The authors argued that understanding how the functioning of a species relates to their intolerance of trawling is the best way to understand impacts on benthic ecosystems, and that using a set of functional traits best demonstrates this interaction between benthic assemblage structure and activity. Working with a full set of traits, rather than a size-based model alone (see Hiddink *et al* 2006), might make it possible to identify whether life history traits influence the subsequent impact of trawling on population size. In their final results, they were able to demonstrate an increase in the relative proportion of small, short-lived species that reproduce sexually and produce pelagic larvae, in areas that were more heavily trawled.

In order to better understand what effects bottom fishing is having on benthic systems, we need to be able to separate the effects arising as a result of natural variability and disturbance. Diesing *et al* (2013) linked fisheries activities in the UK part of the English Channel with estimates of disturbance by natural processes, in order to try and identify faunal preconditioning to disturbance and how this may impact upon estimates of seabed integrity. This work is based on the assumption that benthic assemblages may already be naturally exposed to a highly disturbed environment that pre-conditions them to be resilient to direct physical anthropogenic impacts, such as trawling.

1.10 Pressure-based indicators

The use of biological indicators is potentially a cost-effective way of monitoring the marine environment (e.g. Hiddink *et al* 2006). State indicators are the most commonly used, with a number proposed in the literature, including parameters such as species diversity, the number of species and the proportion of different types of species in benthic samples (Borja *et al* 2011; Rice *et al* 2012). Piet & Hintzen (2012) argued that pressure-based indicators that can be directly linked to management actions are more cost-effective and can be measured more precisely than state-based indicators. However, the authors highlighted that there are large caveats associated with only focusing on the distribution of anthropogenic pressures on particular habitats, including the requirement to identify an 'intensity threshold' dependent on the recovery capacity of the benthic community. Effectively, there is a necessity to fully understand the pressure-state relationship for the pressure and habitat in question before using pressure levels as a proxy of habitat condition.

A trawl disturbance indicator has already been developed to quantify large-scale fishing impacts on benthic ecosystems based on five traits (Juan & Demestre 2012): mobility, fragility, position on substrata, average size and feeding mode. The authors scored each trait from zero to three based on its vulnerability to trawling: 0 – traits advantageous to support trawling; 1 – traits that determine low vulnerability to trawling; 2 – moderate vulnerability; 3 – high vulnerability. This produced a score for each species that ranges from 0-15, where the highest scores represented organisms that are highly vulnerable to fishing. These 15 scores were then grouped into 5 functional indicator groups which were converted to a Trawling Disturbance Index (TDI) by formula. The authors showed that there was a high correlation between the TDI and trawling effort, but with only six data points used.

1.11 Pilot Study

A pilot study was commissioned jointly by JNCC and Cefas in January and March 2011 to investigate the response of benthic assemblages to abrasion resulting from bottom trawling. The survey aimed to gather data to facilitate development of the risk-based approach to monitoring, and to understand how this relates to the requirements of the MSFD assessment of Good Environmental Status. It aimed to do this through collection of benthic samples to further our understanding of anthropogenic abrasion on benthic systems against a background of natural variation. The basic survey design examined benthic community structure at two sites approximately 10km apart with a similar habitat of sand and coarse sand, but with higher and lower fishing intensities (Whomersley *et al* 2012). An aggregated grid of VMS data at 0.05 decimal degrees was used as an indicator of fishing intensity, following the methodology outlined by Lee *et al* (2010a). Sidescan sonar and real time counts of trawl scars at both sites were used as a methodology for ground truthing the VMS data. Though there were no available data on longevity of trawl scars within that particular habitat, the data were used to establish the presence of activity where scars were recorded and so validate the VMS information. Quantitative data on benthic communities was determined in samples collected using a 0.1m^2 Day grab. Sample size was predicted using an *a priori* power analysis that looked to approximate the number of samples required to detect a statistically significant variance in species richness between the benthic communities at each of the two sites.

Data analysis demonstrated that, using the techniques employed, no significant differences between the treatments (relatively higher vs. lower fishing intensity) could be observed in relation to any of the biological metrics measured or to biological trait assessments. Consideration of the survey design demonstrated that there were likely to be several reasons why this pilot investigation was unable to demonstrate a difference between the two sites:

- The difference in fishing activity between the relatively higher and lower sites was not large when put into context with the range of fishing activity across the whole UK seabed. The maximum level of fishing intensity recorded at a national level was

2810 hrs.pa (within a 0.05dd cell), contrasting with the much lower maximum level of fishing intensity recorded within the defined survey area of 410 hrs.pa. As such, both sites within the study were likely to be exposed to a relatively similar level of demersal fishing activity;

- The dominant fishing gear used within the study area (otter trawl) has a small abrasive footprint when compared to other types of bottom fishing gear e.g. beam trawls or scallop dredges (Hinz *et al* 2012). When taken into consideration with the relatively low levels of fishing, this made it unlikely that the grab would sample a disturbed area of seabed; and
- The two factors above, combined with the aggregated VMS grid used for survey planning, made it even more unlikely that a disturbed area of seabed would be sampled.

Even though no direct difference between treatments was established, it is thought that this was largely down to problems in survey design and a lack of substantial difference in fishing levels. Nonetheless, the pilot work was fundamental in establishing some key criteria for future work and in highlighting the inherent complexity of demonstrating habitat impacts when exposed to varying levels of fishing activity. These considerations were a core component of the survey design outlined in this report, especially the understanding of data limitations and the potential mismatch between the spatial scale of biological sampling and fishing activity data.

The resolution at which VMS data were aggregated was suggested to warrant further investigation, as this information is likely to be used in order to prioritise monitoring effort in future. In this study, we test whether the current scale at which JNCC use VMS data, that proposed by Lee *et al* (2010a) (0.05dd), is the most appropriate for informing the design of monitoring surveys.

1.12 Aims and Objectives

The primary aim of this work was to test the suitability of current aggregated VMS layers for use in designing monitoring surveys. To this end, the survey was designed around currently available VMS layers delivered through the MB0106 (Lee *et al* 2010b) contract, updated using equivalent methodology as required. Alternative ways of spatially expressing VMS data were then tested, in order to find which was most suitable in terms of survey planning and assessing benthic response. Through investigation of the primary aims, information was gathered on the impacts of demersal trawling on sedimentary habitats, specifically the relationship between pressures associated with fishing activity and benthic response parameters and, in doing so, the identification of possible response variables. It was not the specific purpose of this work to thresholds at which demersal fishing begins to have an impact on benthic habitats, nor was it designed to identify benthic indicators of disturbance. However, it is hoped that the evidence gathered through this work may assist such research in the future.

2 Survey Design and Methods

2.1 Survey planning

2.1.1 Site selection

Maps of raw VMS data were examined in order to identify areas of highest relative demersal fishing activity in UK waters. Care was taken to ensure that areas of high fishing occurred on similar sedimentary benthic habitats. Areas selected were in the Eastern English Channel (EEC) and the Outer Thames Estuary (Th). The habitats present at the sites were broadly sandy based on the most up to date habitat maps and survey data available at the time of planning: Regional Environmental Characterisation (REC) survey data, high confidence[2] EUNIS maps included in the MESH (Mapping European Seabed Habitats) database and UKSeaMap (Figure 4).

A third area to the south of the Dogger Bank was also selected but, due to inclement weather during the survey cruise, it was not possible to sample at this site.

2.1.2 VMS processing and demersal fishing pressure for survey planning

Geographic Information System (GIS) layers were produced by gear type for both UK and EU vessels, and these were then weighted based on the relative spatial footprint of the fishing gear primarily used by the vessel. The weighted and summed VMS grid cells were then extracted for areas identified as sandy substrata, based on EUNIS level 3 descriptions. Consideration was also given to the scale of fishing intensity observed across the extracted cells and 12 blocks which represented the relative gradient of fishing intensity were selected.

Unprocessed Vessel Monitoring Systems (VMS) data were obtained by Cefas for the 18 months leading up to the survey (all of 2010 and first half of 2011) for UK vessels. An assumption was made that this would be a suitable period for capturing possible impacts of demersal fishing on the seabed, based on recovery periods of similar benthic habitats (e.g. Dernie *et al* 2003; Kaiser *et al* 2006). For non-UK vessels, VMS data were only obtained for 2010 as the data for 2011 were not available at the time of survey planning[3]. VMS data for UK vessels were linked to skipper logbook information in order to determine the type of fishing gear being used. The gear types used were categorised in accordance with the UN Food and Agriculture Organization (FAO) fishing gear classification (FAO 1990). For non-UK registered vessels, where logbook information was not available, information on fishing gear was obtained from the "primary gear" listed on the EU vessel registration database (European Commission 2005).

VMS data are collected irrespective of vessels' activity. As such, VMS data include records collected during both fishing and non-fishing activities, such as steaming. The maximum speed at which seabed fishing gear can be used is thought to be approximately 6 knots (Eastwood *et al* 2007). Therefore, the unprocessed VMS data were filtered for records with speeds between 1 and 6 knots to ensure that the dataset only contained information from vessels engaged in fishing activities. The date and time information attached to the unprocessed VMS data were used to determine the interval between consecutive VMS pings for each vessel, which was usually approximately two hours.

VMS data were imported into ArcGIS v.10.1® (Environmental Systems Research Institute) with ping interval included as an attribute to act as a proxy for fishing effort. The data were grouped by year and fishing gear and whether the data were from UK or non-UK registered vessels.

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[2] *Based on three criteria: (i) remote sensing data, (ii) ground-truthing data and (iii) data interpretation. See www.searchmesh.net/Default.aspx?page=1635 for original methodology*

[3] *Our inability to acquire VMS data for non-UK vessels for the purposes of planning in time for this research cruise again highlights the general difficulty in obtaining fishing data for the purposes of marine seabed monitoring. This is an important consideration when using the recommended methodology to plan future monitoring surveys.*

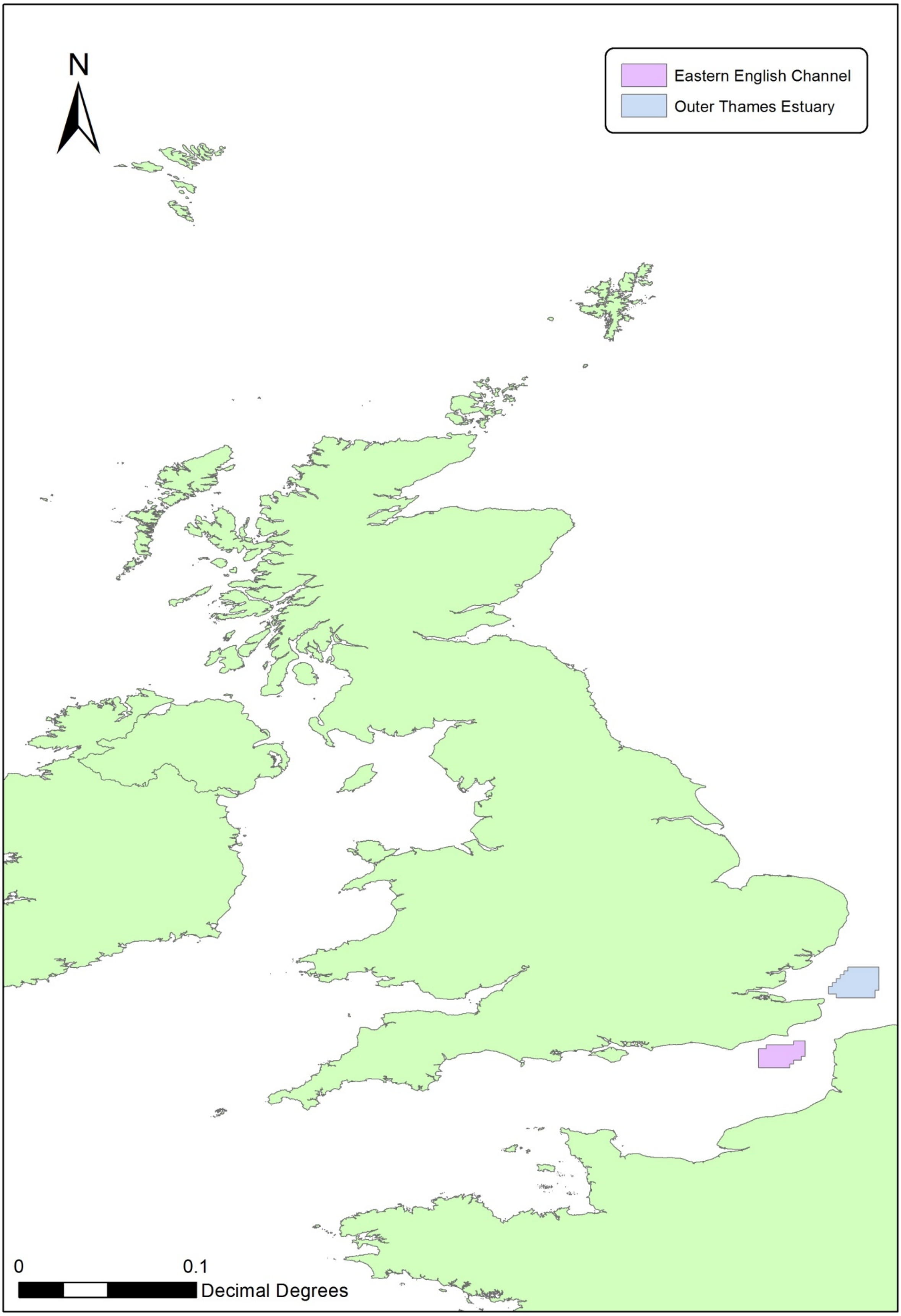


Figure 4. Two study areas were selected: Eastern English Channel and Outer Thames Estuary.

Subsets of the GIS layers were created to capture VMS data that fell within the three planned survey areas.

The ping intervals from the VMS points were aggregated to a Geographic Coordinate System 0.05dd (WGS84) raster grid using a spatial join. This grid was based on the MB0106 grid (Lee *et al* 2010b), which itself is a subdivision of the ICES rectangle system. This meant that the VMS-derived fishing activity data could potentially be compared with fisheries landings data in the future, as these are also reported using ICES rectangles/sub-rectangles.

Each gear type was assigned a relative spatial footprint weighting, based on the gear widths described by Eastwood *et al* (2007) (Table 1). The smallest gear width belonged to the bottom otter trawl (4m), which was therefore given a weighting of x1. Other gears' weightings were determined by the size of their footprint in relation to that of the bottom otter trawl. The gear type layers were multiplied by their weighting, providing each grid cell with a 'score' that acted as an indicator of relative fishing pressure on the seabed. It should be noted that these scores are dimensionless and were intended to give a relative measure of fishing intensity rather than a quantification of the seabed area affected by demersal trawling. Null values (for cells in which no VMS points were present) were converted to zeroes using the 'conditional evaluation' tool of the ArcGIS Spatial Analyst toolbox. This enabled rasters with null values in different cells to be combined. The 'raster calculator' tool in the ArcGIS Spatial Analyst toolbox was used in an iterative model to sum the gear/year layers based on their fishing intensity scores.

Table 1. Fishing gear types and their associated footprint widths and relative weightings.

Gear	Gear code	Gear width (m)[1]	Weighting
Otter trawls – bottom	OTB	4	x 1
Otter trawls – not specified	OT	Unknown (8)[4]	x 2
Otter twin trawls	OTT	8	x 2
Boat dredges	DRB	20.4	x 5.1
Mechanized dredges	HMD	20.4	x 5.1
Nephrops trawls	TBN	20.4	x 5.1
Beam trawls	TBB	12	x 3
Other trawls - not specified	TX	Unknown	-
Pair trawls - bottom	PTB	24	x 6

Based on Eastwood *et al* (2007)

2.1.3 Power analysis

A statistical power analysis was carried out prior to the survey to inform on the number of samples needed within each region based on existing abundance data from the selected sites. Based on the power analysis, it was decided that 12 blocks, each representing a cell within the 0.05dd grid and a level of fishing pressure along the full range, would each contain 10 randomly positioned sampling stations. Figure 4 and Figure 5 demonstrate how the sampling stations were distributed over Th and EEC.

The sampling was also not completed at Th due to bad weather, but sufficient samples had been collected to allow a comparison to be made with EEC. In total, 198 infauna samples were taken, the full 120 samples from EEC and 78 from Th. Due to the higher priority given to collection of infauna samples, fewer epifauna samples were taken; six samples from Th and twelve from EEC.

[4] *After discussion with JNCC and Cefas fisheries experts it was concluded that this was most likely to be a twin rig otter trawl.*

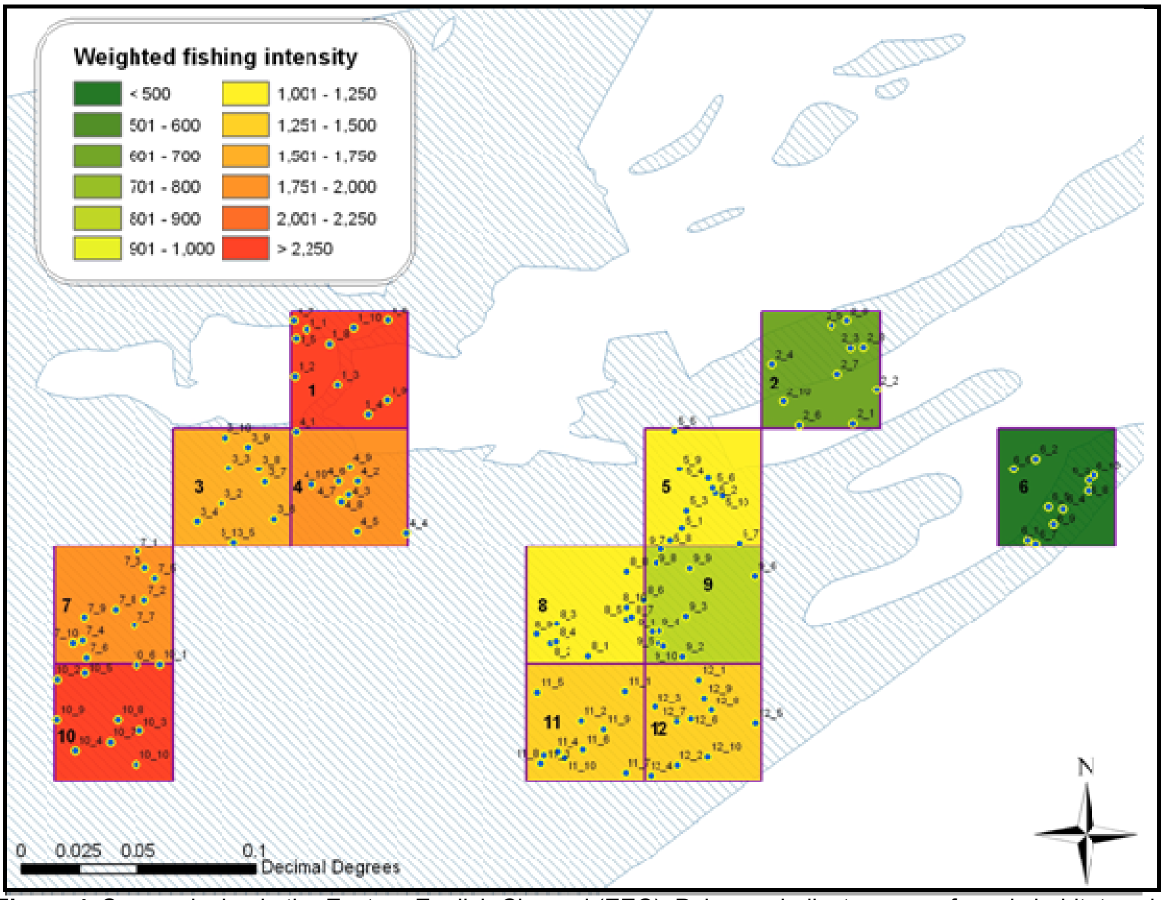


Figure 4. Survey design in the Eastern English Channel (EEC). Polygons indicate areas of sandy habitats selected for survey.

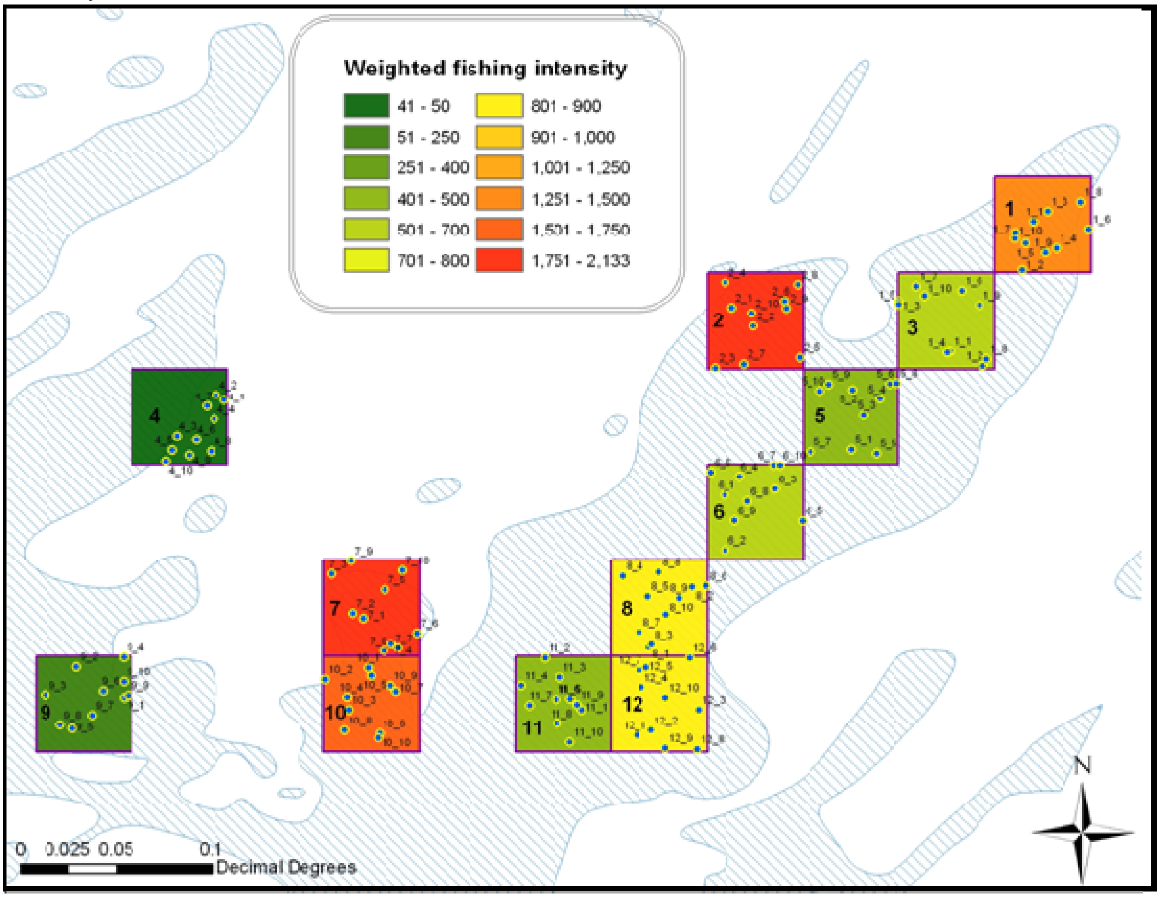


Figure 5. Survey design in the Outer Thames Estuary (Th). Polygons indicate areas of sandy habitats selected for survey.

2.2 Sample collection and processing

The survey took place from 10 December to 22 December 2011 on the *RV Cefas Endeavour* with a crew composed of JNCC and Cefas survey scientists.

2.2.1 Benthic sampling

At each station, a benthic grab sample was taken using a $0.1m^2$ Hamon grab for infaunal analysis and Particle Size Analysis (PSA), as well as a Shipek grab sample, for determination of organic carbon and nitrogen content of the sediment, two 2m beam trawls (for epifaunal analysis) and an extended camera tow (to assess sediment boundary changes). The camera tows unfortunately had to be dropped from the plan due to poor underwater visibility during survey operations.

After the removal of the sediment subsample required for PSA, the remaining sediments were sieved over a 1mm sieve. Sediments and organisms retained on the 1mm sieve were preserved in 4% formaldehyde and returned to the laboratory for later faunal analysis.

2.2.2 Biological samples

On return to the laboratory, biological samples were washed over a 1mm sieve and all retained organisms identified where possible to species level, or if not to the lowest possible taxonomic level. Each individual taxon was then weighed to give a blotted wet weight (g) /species. Ash Free Dried Weight (AFDW) per species was calculated using conversion factors (Eleftheriou & McIntyre 2005; Ricciardi & Bourget 1998; Rumohr *et al* 1987) embedded in the UNICORN database and output as Excel spreadsheets.

2.2.3 Sediment samples

A sub-sample of sediment was removed from the total sediment sample collected using the $0.1m^2$. Hamon grab. PSA methodology is based on recommendations made by the National Marine Biological Analytical Quality Control Scheme (Mason 2011). A subsample of the sediment, screened at 1mm, was analysed using the Malvern Mastersizer 2000 laser sizer. The remaining sediment was split at 1 mm by wet-sieving. Sediment >1mm was dry-sieved at 0.5 φ intervals, from 1 to 63mm. The dry-sieve and laser results were combined to give the full particle size distribution at half phi intervals, between 0.1 μm and 63 000 μm (63mm) (11.5 phi to –6 phi).

Sieve samples were weighed before sieving, during sieving and after sieving. Totals were checked and any samples with anomalies in the results were re-sieved. Two repeats were run for each sample on the laser-sizer and variability compared against set limits. A third repeat was completed for any samples outside expected limits. Outliers were removed and an average taken from the repeat runs completed. Glass-certified reference and in-house reference materials are used at regular intervals. Cefas participates in PSA ring tests as part of the NMBAQC scheme.

The full-resolution particle size distribution (PSD) data (at 0.5 φ intervals) for all sediments collected at the Outer Thames Estuary and the Eastern English Channel were grouped separately using Entropy, a non-hierarchical clustering method that groups sediments. EntropyMax is a Windows-based software that groups large matrices of PSD datasets into a finite number of groups. It has been described in detail by Stewart *et al* (2009).

Organic carbon and nitrogen content is determined in the <2mm fraction of sediment. The sediment is freeze-dried and ground. Inorganic carbon is removed using sulphurous acid, and then organic carbon and nitrogen contents are determined using an elemental analyser.

A certified reference material is run within each sample batch for quality control. Results are reported in % m/m dry weight.

2.2.4 Biological traits allocation

To determine the variability of biological traits within the EEC and Th survey areas, 10 traits were chosen as relevant to the resistance and resilience of benthic communities to seabed disturbance caused by fishing activity (Table 2). Each of these traits were subdivided into various modalities chosen to encompass the range of possible attributes of all taxa, resulting in the identification of 46 modalities in total. Some of the traits referred to measurable characteristics (e.g. size range, longevity) whose modalities presented a 'hierarchical' organisation (Paganelli *et al* 2012) while others (e.g. mobility) were wholly qualitative characteristics whose modalities represented discrete classes.

Table 2. Description of traits and modalities used in the biological traits analysis.

Trait	Modality	Description
Size Range (mm)	<10 (VS) 10-20 (S) 21-100 (SM) 101-200 (M) 201-501 (ML) >501 (L)	These reflect the maximum size the individual can reach in any dimension (either in height or width/breadth). For colonials such as bryozoans and hydroids, the size of colony is given, not the size of the individual cell.
Morphology	Fragile	Fragile or shell/structure
	No Protection	Body covered by a protective outer tissue made up of, for example, cellulose, e.g., tunicates
	Protected	Body covered or encased in either tough skin or exoskeleton
	Robust	Hard shell/ability to regenerate
Longevity	<1 year 1-3 years 3-10 years >10 years	The maximum lifespan of the adult stage
Larval Development Location	Pelagic – Planktotrophic	Larvae feed and grow in water column, generally spend a few weeks there enabling great dispersal potential
	Pelagic - Lecithotrophic	Larvae enter water column but are reliant on yolk reserves; typically pelagic for <1wk. Limits dispersal potential
	Benthic (direct)	Larval stage missing (eggs develop into juvenile forms) or larvae are limited to the bed
Egg Development Location	Asexual / budding	Species can reproduce asexually, either by fragmentation, budding, epitoky, etc. Often this is in addition to some form of sexual reproduction
	Sexual – shed eggs (pelagic)	Eggs are released into the water column
	Sexual – shed eggs (benthic)	Eggs are released onto/into the bed, either free or maintained on bed by mucous or other means
	Sexual – brood eggs	Eggs are maintained by adult for protection, either within parental tube or within body cavity
Living Habit	Tube-dwelling	Organism lives within a permanent structure within the sediment. Tube may be lined with sand, mucus or calcium carbonate and thus afford some kind of physical protection
	Burrow-dwelling	Lives within a permanent or temporary burrow, organism capable of fabricating new burrows quickly.
	Free-living	Species in which adult is not limited to any restrictive structure at any time. Able to move freely within sediments
	Crevice/hole/ under stones	Adults are typically cryptic, predominantly found inhabiting spaces made available by coarse/rock substrate and/or tubes made by biogenic species or algal holdfasts
	Epi/endo zoic/phytic	Organisms which are found directly attached to other organisms. May be found attached to external shells of animals or fronds of macroalgae. Includes those found within cavities of animals (e.g. mantle cavity of gastropods)
	In shell/tube of other animal	Organisms that primarily inhabit shell/tube of other animal
	Attached to substratum	Organisms actively attached to larger substrata or rock
Sediment Position	Surface	Species which are found on or just above the seabed. These do not cross the sediment/water interface whilst undertaking biological activities (feeding, locomotion).
	Shallow infauna (0-5cm)	Species whose bodies are found almost exclusively below sediment surface between 0 and 5cm sediment depth. Such species may have connection (either permanent or temporary) with overlying water column for feeding.
	Mid-depth infauna (5-10cm	Species whose bodies are partly or exclusively found below sediment surface at a depth generally between 5 and 10 cm

	depth)	sediment depth. The species may also be capable of occupying other sediment depth classes. Such species may have connection (either permanent or temporary) with overlying water column for feeding.
	Deep-infauna (>10cm)	Species whose bodies are partly or exclusively found below sediment surface at a depth greater than 10 cm sediment depth. The species may also be capable of occupying other sediment depth classes. Such species may have connection with overlying water column for feeding.
Feeding mode	Suspension	The removal of particulate food taken from the water column, generally via filter-feeding
	Surface deposit	Active removal of detrital material from the sediment surface, either via palps or ‘hoovering’, using an inhalant siphon. This class includes species which scrape and/or graze algal matter from surfaces.
	Sub-surface deposit	Removal of detrital material from within the sediment matrix. Generally involves non-selective ingestion of sediment and active egestion of sediment
	Scavenger / opportunist	Species which feed upon dead animals
	Predator	Species which actively predate upon animals (including the predation on smaller zooplankton)
	Parasite	Species which have a parasitic mode of life on other invertebrate species. An uncommon trait, found in eulimid gastropods and RHIZOCEPHALA crustaceans
Mobility	None	Species in which the adults have no, or very limited, mobility either because they are attached or are limited to a (semi-) permanent tube or burrow
	Low	Species in which adults are capable of some limited movement along the sediment surface or rocky substrata
	High	Species in which the adults are capable of movement along the sediment surface, rocky substrata and burrowing
Bioturbation	Diffusive mixing	Vertical and/or horizontal movement of sediment and/or particulates resulting from the activities of, for example, some free-living polychaetes, subsurface deposit feeders and carnivores, and burrow excavating species
	Surface deposition	Deposition of particles at the sediment surface resulting from e.g. defecation or egestion (pseudofaeces) by, for example, filter and surface deposit feeding organisms
	Upward conveyor	Translocation of sediment and/or particulates from depth within the sediment to the surface during subsurface deposit feeding or burrow excavation.
	Downward conveyor	The subduction of particles from the surface to some depth by feeding or defecation.
	None	Do not perform any of the above.

Many taxa display multi-faceted behaviour depending upon, for example, the specific conditions that prevail and resources available. Therefore, each of the taxa were coded using a “fuzzy coding” approach (Chevenet *et al* 1994) on the basis of the extent to which they displayed the modalities of each trait. Fuzzy coding allows taxa to exhibit differing modalities to different degrees. This approach, therefore, avoids the obligate assignment of a taxon to a single category/modality, which can lead to inaccurate characterisation of biological or ecological taxa profiles (Usseglio-Polatera *et al* 2000). In order to classify a taxon according to its affinity for more than one modality, each modality was given a score between 0 and 3 where 0 conveys that the taxon has no affinity for that modality, 1 or 2 express increasing partial affinity and 3 indicates total and exclusive affinity for that modality. In reality, certain traits, such as size range, longevity, larval and egg development, were predominantly expressed as partial modalities for most taxa. This reflected (a) variability of the attribute within a particular taxon, (b) variability in the attribute for a taxon from different published sources, and (c) variability displayed between different species within a genus. In contrast, entries for other traits, e.g. morphology and mobility, were often represented by a total affinity for one particular modality.

When all taxa had been coded for the species x trait matrix, the codes were converted to proportions for each taxon so that the total sum for each taxon x trait = 1. For example, for the trait ‘feeding mode’, *Ophiothrix* sp. was assigned a ‘3’ for suspension feeding and a ‘3’ for surface-deposit feeding; this was standardised to 0.5 and 0.5, respectively. The taxon x trait matrix was then combined with the station x taxon abundance matrix to create a final matrix of station x trait.

2.3 Data processing and analysis

2.3.1 Retrospective calculation of fishing pressure scores

In order to test which method of processing VMS data was best at explaining biological variability; alternative methods were applied to the same raw VMS data that was used in survey planning. Mapped demersal fishing pressure layers were produced, in order that five different fishing pressure scores could be attributed to each sampling station:

1. VMS gridded at 0.05dd (method described in Section 2.1.2)
2. VMS gridded as above but at a resolution of 0.025dd
3. VMS gridded as above but at a resolution of 0.0125dd
4. Nested VMS using 20 pings to trigger division
5. Nested VMS using 100 pings to trigger division

Each aggregated VMS grid was overlain with a shape file containing points at which samples had been collected. The fishing intensity scores of the underlying grids were appended to the points using the ‘extract values to points’ tool of the ArcGIS Spatial Analyst toolbox, so as to allow further analysis of the data.

For the nested layers, the method proposed by Gerritsen *et al* (2013) for VMS point data summation using a nested grid approach was broadly followed. This approach calculated ‘swept area’ (area covered by fishing gear) for each VMS ping and produced a ‘Swept area ratio’ showing the area of a grid cell that had been trawled relative to the area of the cell. For instance, a cell with a score of 100% or 1 could mean that the cell had been trawled completely once, or that half the cell had been trawled twice, or other permutations. The method, written in R code, sets a maximum cell size (0.18dd x 0.16dd). If a maximum given number of VMS pings (e.g. 20) within a grid cell is reached, the cell was halved. This process was repeated until the cell contained fewer than the maximum number of pings specified or until a given number of divisions had been completed. The result was a nested grid of varying cell sizes. Each cell contained the maximum specified number of pings or fewer, with the exception of cells that contained more but had already reached the maximum permitted number of divisions.

The R code provided by Gerritsen *et al* (2013) was adapted for use with these data, as per the other method. The swept area was calculated for each ping as per the Gerritsen method (Equation 1).

Equation 1– *Swept Area*

$$\mathbf{SA} = \sum_{i=1}^{n} e_i \, v_i w_i$$

Where i is the VMS record (1,…n), e is the hours fished per year (ping interval time), v is Average Speed Vessel (km/h), w is Total width of fishing gear (km) [5] *impacting the seabed and SA is the swept Area – adapted from Gerritsen et al (2013)*

[5] Gear widths taken from Eastwood *et al* (2007).

The nested grid sizes are then calculated. The maximum cell size was set to replicate the MB0106 VMS data at 0.05dd to ensure consistency. The maximum number of divisions for the 0.05dd cell size was set to 6 in order to aid computer processing time and leave the smallest cell at a manageable size (7.81 x10^{-3} dd). The number of VMS data points that triggered a divide was set first to 20 pings and subsequently to 100 to allow for comparison of the outputs.

A summation of the swept area for all VMS points contained within each grid cell was then calculated. The summed swept area was then divided by the area of the cell to calculate the swept area ratio (Equation 2)

Equation 2 – *Swept Area ratio*

$$\boldsymbol{SAr} = \frac{\boldsymbol{SA}}{\boldsymbol{CA}}$$

Where SAr is Swept Area ratio (Number of times the cell is fished), and CA is cell Area. – adapted from Gerritsen et al (2013).

The resulting grid was one of varying sizes nested in the maximum grid size of 0.05dd showing a more comparable set of impact scores. The grid of swept area calculations (20 and 100) were then appended to the survey sample locations using a spatial join for comparison.

2.3.2 Linear regression

All univariate analyses were carried out using sigmaPlot 12.0. All data were first assessed for normality and transformed where appropriate.

Simple linear regressions were carried out between univariate biology indices for both infauna and epifauna and environmental predictors. The following three measures of diversity were considered:

Number of species (*S*)
Margalef index (*d*): (*S* - 1) / ln (total number of individuals)
Biomass (*b*)

To create linearity and constant variance in the infauna data, square root transformations were made of abundance (number of individuals) and richness (number of species) and natural log for biomass (g). For the epifauna data, outliers present for abundance and biomass make assumptions behind the linear regression models invalid (even when taking the stronger natural log transformation for abundance). Thus, the non-parametric Mann-Kendall (Mann, 1945; Kendall, 1975) test was used to assess potential trends.

We defined the biological variables by $Y_1,...,Y_n$ such that the pressure variables are sorted from low to high; then calculated the Mann-Kendall statistic:

Equation 3

$$S = \sum_{j=1}^{n} \sum_{k>j} I(Y_j, Y_k)$$

where $I(.)$ is an indicator variable defined by the sign of $D = Y_k - Y_j$. If D is positive then $I(.) = 1$, if D is negative then $I(.) = -1$, if D = 0 then $I(.) = 0$.

In non-mathematical terms, each Y value is considered in turn and then the number of 'future' Y values that are greater (+1) or less (-1) are calculated. When this score is summed over all the Y values, the statistic gives an indication of trend (a positive value suggesting an upward trend and a negative value suggesting a downward trend).

We calculated p-values for the Mann-Kendall statistic by using a randomisation procedure in the spirit of Manly (1998). Under the null hypothesis of "no trend", the order of the pressure values is irrelevant. Thus, to see how S behaved under this null hypothesis, we randomised the order of the n pressure values and calculated S. When repeated many (e.g. 10,000) times then we generated the null distribution of S. Given that we did not know *a priori* which direction the trend would follow, we calculated the absolute value of S in our randomisation procedure. The p-value was the proportion of the randomised S values that were greater than the observed value.

2.3.3 Dominance plots and the W-statistic

Dominance plots, also referred to as species abundance plots, were created for both abundance and biomass data. Species from each station were then ranked in decreasing order of abundance/ biomass and used as the basis for the x-axis (logarithmic scale), before being plotted against cumulative relative abundance/ biomass. Dominance plots for species abundance and biomass were then combined at each station to create Abundance-Biomass Comparison (ABC) curves. The interaction between abundance and biomass curves is described as being indicative of a communities disposition to either *r* or *k* selection[6]. Clarke and Warwick (2001) highlight the problems associated with dealing with large datasets and the cumbersome nature of drawing ABC curves for them all. To overcome this they suggest the use of a single summary statistic, the *W* (for Warwick) Statistic, that will allow the comparison of stations according to their potential disturbance. The value calculated is standardised to a common scale between 1 and -1, with values approaching 1 undisturbed (even abundance across species but biomass dominated by a single species) and values of -1 heavily disturbed. The *W* statistic can be calculated using the Equation 4 below where B_i = Biomass, A_i = Abundance and S_i = Number of species at a particular station.

Equation 4

$$\boldsymbol{W = \sum_{i=1}^{S} (B_i - A_i)/[50(Si - 1)]}$$

2.3.4 Predictive ability of models

During the formulation of the model, $\sqrt{S}$ and ln(b) were used in order that the linear model assumptions were broadly met. For biomass, we used the standard back-transformation for a lognormal distribution: $\exp(P_j + v/2)$, where P_j is the prediction for observation j and v is the residual variance from the model fitted to the training data set (see below).

For infauna data, the explanatory variables used were:

1. VMS_05 (VMS gridded at 0.05dd)
2. VMS_025 (VMS gridded at 0.025dd)
3. VMS_0125 (VMS gridded at 0.0125dd)
4. VMS_20 (nested VMS using 20 pings)
5. VMS_100 (nested VMS using 100 pings)
6. % silt

[6] *In this context, r-selection makes a species prone to numerous reproduction at low cost per individual offspring, while K-selected species expend high cost in reproduction for a low number of more difficult to produce offspring. Organisms that live in stable environments tend to make few, "expensive" offspring. Organisms that live in unstable environments tend to make many, "cheap" offspring.*

7. % gravel
8. Depth
9. Distance (from shore)
10. Carbon content
11. Nitrogen content
12. Sediment group
13. Modal sediment group

Different fishing pressure scores were calculated based on the same raw VMS data and were each tested to see whether they were a better predictor of biological variability. Methods for processing are presented below. However, only one of the VMS variables was used in each model. Linear models were fitted with a biodiversity measure as the dependent variable and subsets of the 11 explanatory variables.

For the EEC data set, there were three missing values for carbon and nitrogen (stations 4, 37 and 58). In order to make fair comparisons between various subsets of the explanatory variables, we did not use these cases in our model comparisons.

Fitting all possible subsets is time-consuming and so expert judgement was used in selecting the models fitted. Collinearity was also checked for between explanatory variables. In particular, we started with the best performing variable when used on its own and then added further variables depending on how they each performed alone. Various subsets were checked in this way before coming up with a minimal model that performed optimally. As in Devlin *et al* (2008), the ability of the models to predict S was used as a measure of their performance. Whereas, Devlin *et al* (2008) used cross-validation to do this (effectively, a prediction data set of size 1), the procedure was modified by using a larger prediction data set. The procedure was as follows:

Randomly divide the cases into training and prediction data sets. The prediction data set constitutes one third (N_p) of the cases (the training data set is the other two thirds). The training data set was used to fit a linear model and then the model was used to predict the observations actually observed in the prediction data set.

A mean relative absolute error (MRAE) was used to measure the performance of the prediction over all observations in the prediction data set. The whole process was repeated 1,000 times and the mean MRAE taken.

The MRAE measure for prediction case j is defined as:

Equation 5

$$MRAE_j = \frac{100}{N_p} \sum_{j=1}^{N_p} \frac{|O_j - P_j|}{P_j}$$

where O_j is the observed value for prediction case j and P_j is the predicted value for prediction case j. One can interpret the MRAE as a measure of the mean relative percentage error of prediction (i.e. the error is scaled by the size of the predicted value). Models were built and interrogated using the free software R.

2.3.5 Multivariate analyses

Multivariate analysis of the macroinvertebrate data was carried out using Primer v6.1.15 (Clarke 1993; Clarke & Warwick 2001; Clarke & Gorley 2006). Nonparametric multi-dimensional scaling analysis (MDS) was performed on the square root transformed abundance and biomass data for infaunal and epifaunal communities in order to identify differences in the composition of the benthic ecosystems at the two sites. The similarity percentage (SIMPER) routine was also performed on the macroinvertebrate community data to identify any differences at the two survey sites in relation to the dominant species of the resident fauna. Analysis of similarity (ANOSIM) and the similarity profile routine (SIMPROF) were used to determine the statistical significance of any differences in benthic community composition.

The Primer v6.1.15 software offers the option to investigate differences in communities between pre-defined groups of samples (predefined and not calculated from 'cluster' routines). Analysis of similarity (ANOSIM) tests can be used as an approximate analogue of the standard univariate ANOVA test, so long as the pre-mentioned condition is met (Clarke & Gorley 2006). ANOSIM tests were carried out on infaunal communities from both EEC and Th sites with groups pre-defined using pressure scores derived from the VMS gridded resolutions at 0.05dd and 0.025dd. ANOSIM tests were not carried out for the 0.0125dd grid resolution as this was deemed to have too many independent pressure scores with too few stations per potential 'group'.

With so many potential one way tests it would be difficult to interpret data visualised in a tabular format. Primer v6.1.15 offers the potential to overcome this by out putting R values for each test to a resemblance worksheet. This can subsequently be re-visualised as an MDS plot that allows graphical representation of group separation, so that the higher the R value between two groups of samples the greater the separation of the two groups in high-dimensional space (Clarke & Gorley 2006). Though p-values are not represented in these plots it is important to balance the value of interpretations based on either p values or R values. Clarke & Gorley (2006) report that p values can be highly affected by sample sizes whereas R is not, but rather is an absolute measure of differences between two groups. Global test statistics must be observed i.e. observed non-significant global p-values will prevent the further running of subsequent pairwise tests.

The BEST routine was used in an attempt to discern which of the environmental parameters best fit the community variability. Multivariate analyses were then carried out on the corresponding square root transformed biological traits (BT) data to assess whether there were any differences in the life-history characteristics of the resident fauna.

The sediment Particle Size Analysis (PSA) data along with other environmental parameters (distance from shore, depth, organic carbon content and nitrogen content) were normalised and a similarity resemblance matrix based on Euclidean distance was used to carry out an MDS. A principal component analysis (PCA) was further employed to identify the causes of any differences in sediment particle size between survey sites.

3 Results

3.1 Pressure assessment

At the coarser resolution of 0.05dd, all stations within each respective cell were allocated the same pressure score. However, when the original cells were re-gridded to higher resolutions (0.025dd and 0.0125dd) it was apparent that, within several cells, fishing activity was not homogenous across the whole area (Figure 6 and Figure 7) resulting in a redistribution of pressure scores within each original survey block. This would have led to the relative abrasion scores changing for associated biological sampling stations and hence the different processed layers may explain or predict biological variability differently.

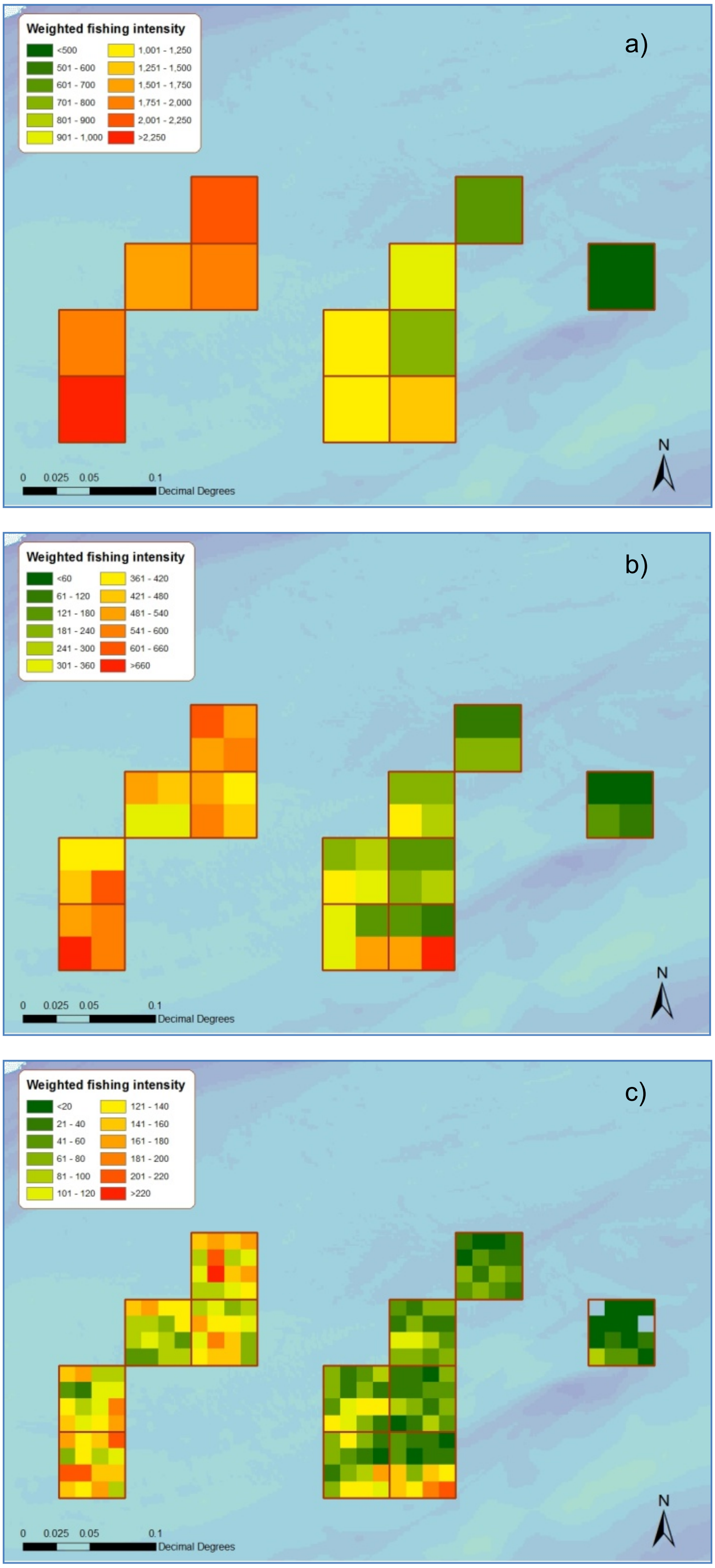


Figure 6. Re-aggregation of VMS point data from the Eastern English Channel to a) 0.05dd, b) 0.025dd and c) 0.0125dd grids.

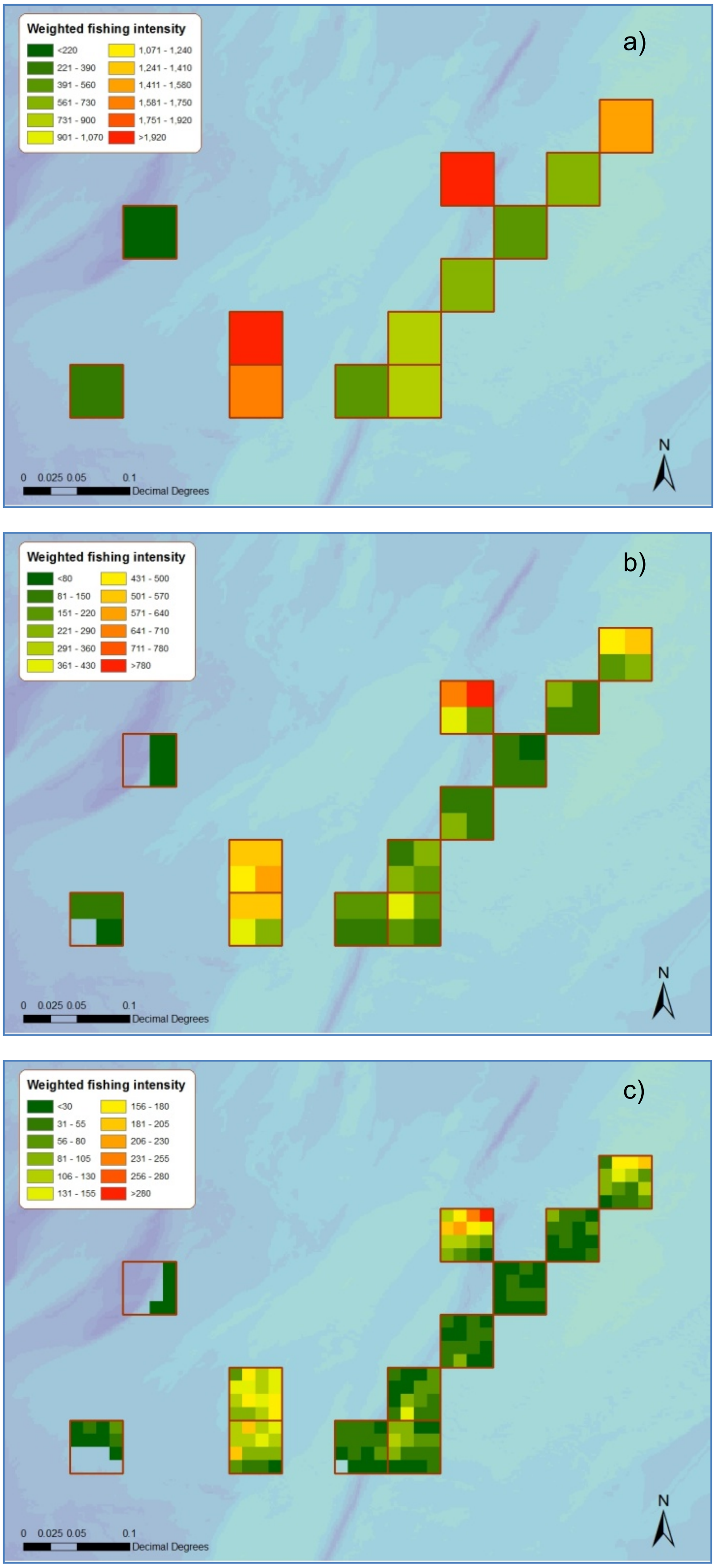


Figure 7. Re-aggregation of VMS point data from the Outer Thames Estuary to a) 0.05dd, b) 0.025dd and c) 0.0125dd grids.

As fishing pressure scores were altered, so the spread of sampling stations across the planned gradient of pressure would also change, as they were matched to the original 0.05dd gridding. However, distribution across the gradient changed only negligibly and even at the finest resolution the data still met all of the assumptions for robust analysis (Figure 8).

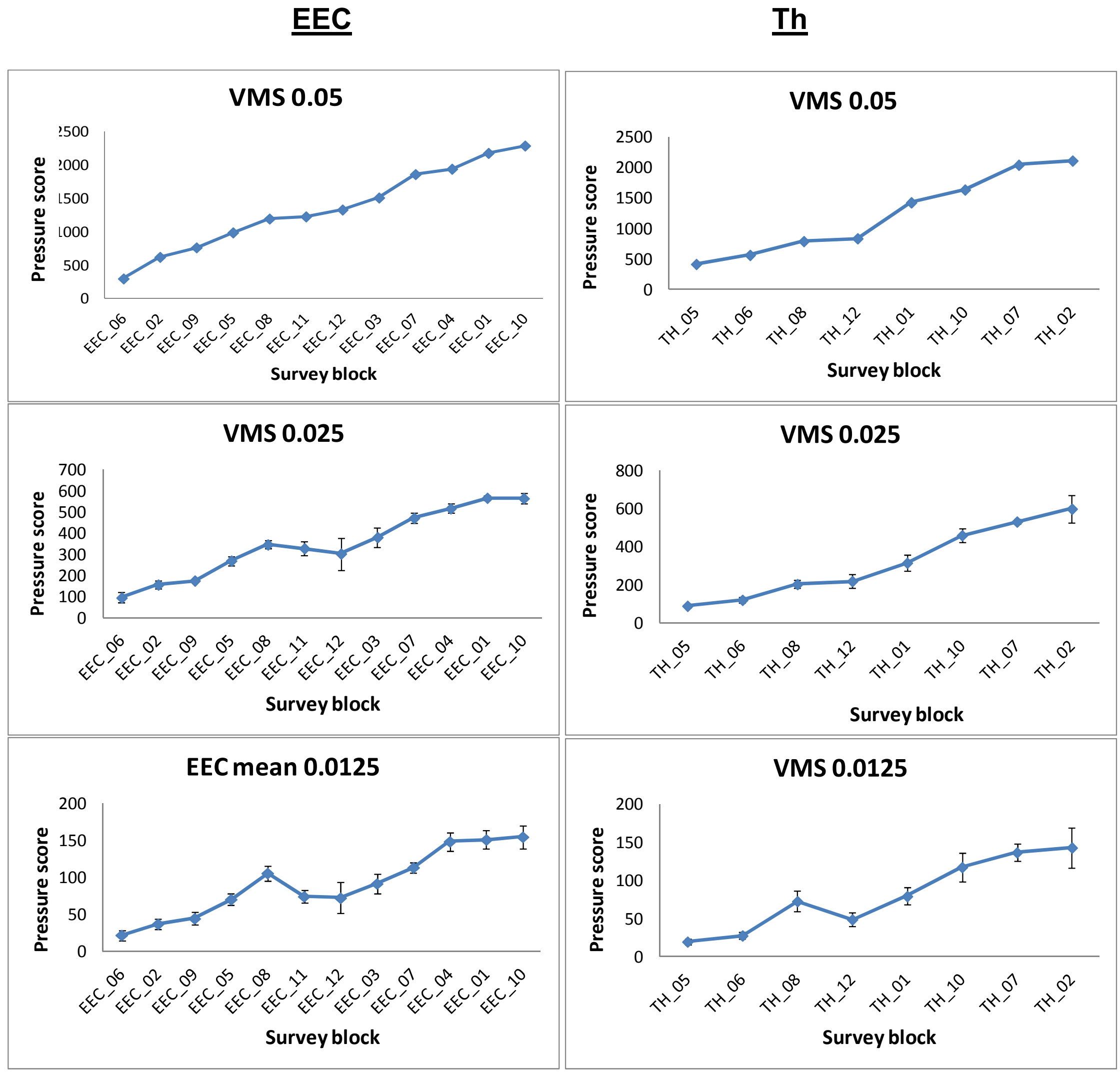
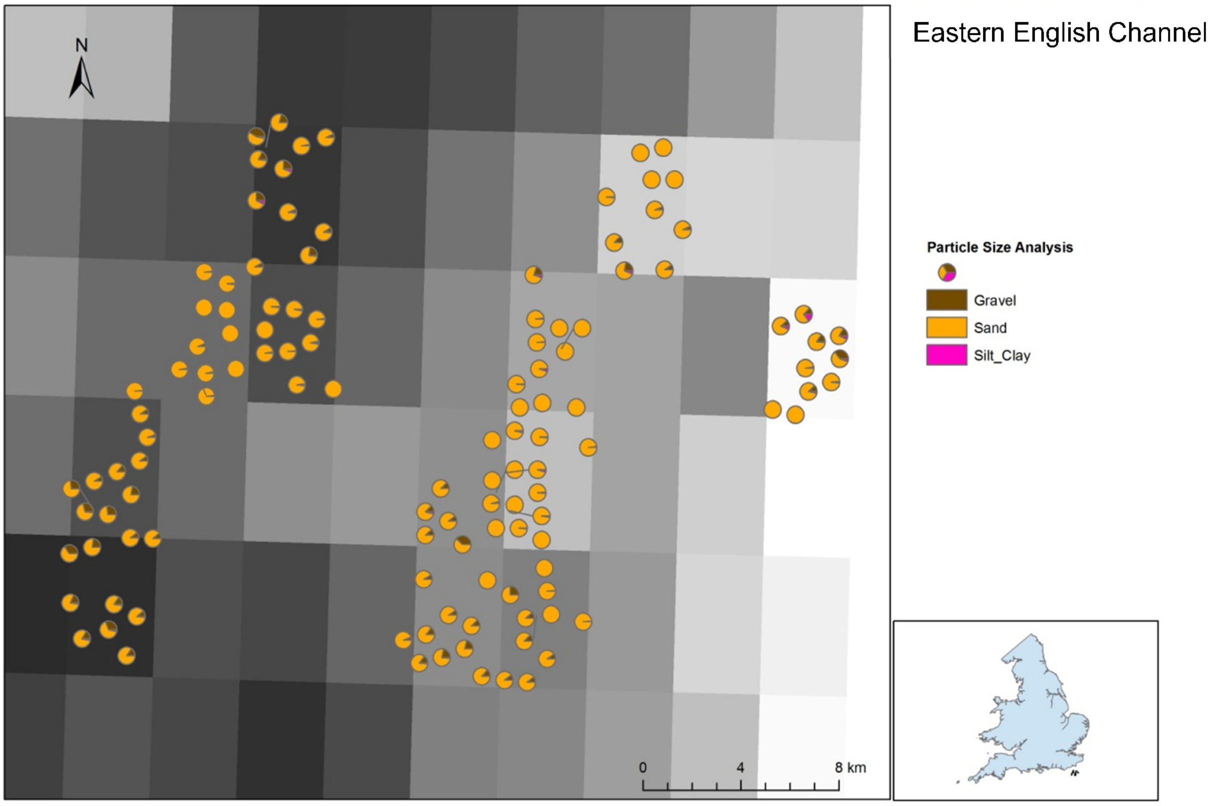


Figure 8. Plots for average abrasion pressure scores calculated from the regridding of the 0.05dd abrasion layer to a resolution of 0.025dd and 0.0125dd, with standard error bars.

3.2 Sediment analysis

Sedimentary habitats were found to be broadly similar between the survey regions, with sediments characteristic of sand and gravelly sand (Figure 9). However, a slightly higher degree of silt/clay fraction was observed in blocks 7 and 10 within the Th survey area. There also appears to be marginally more gravel across EEC.



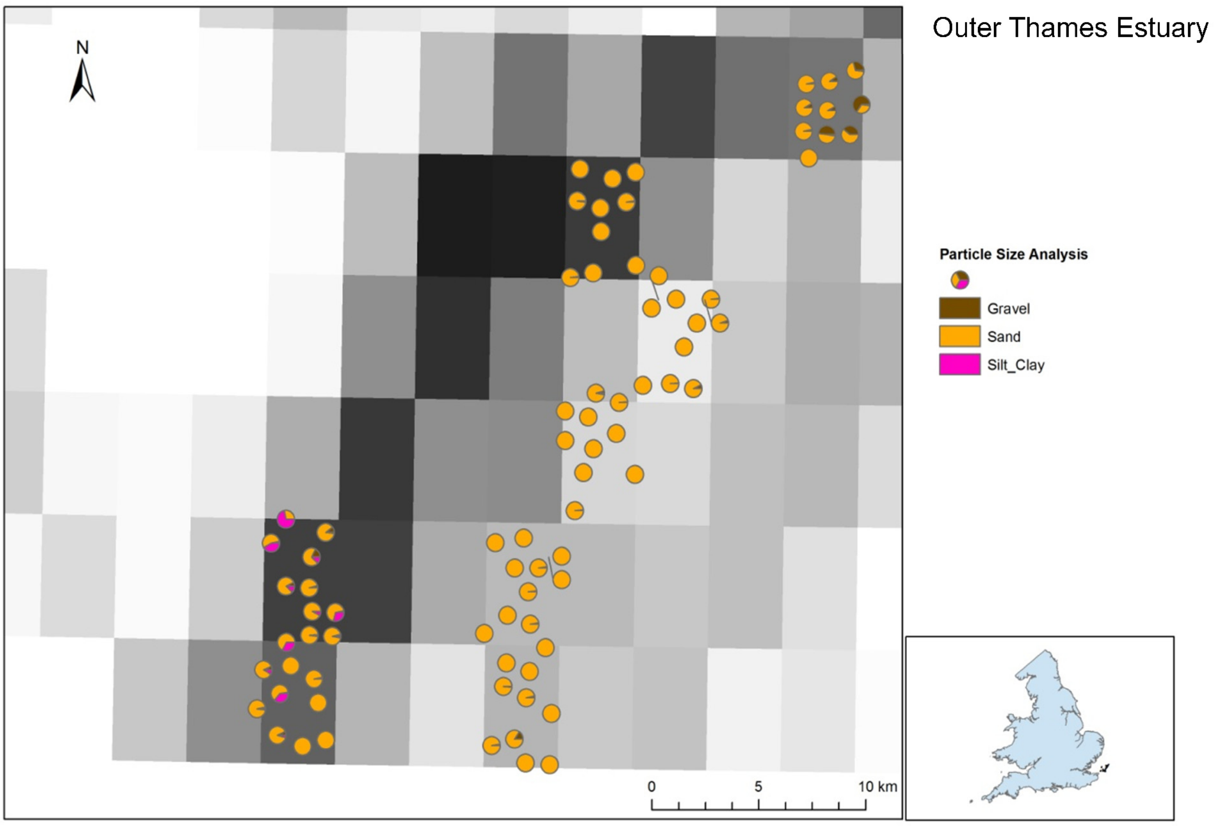
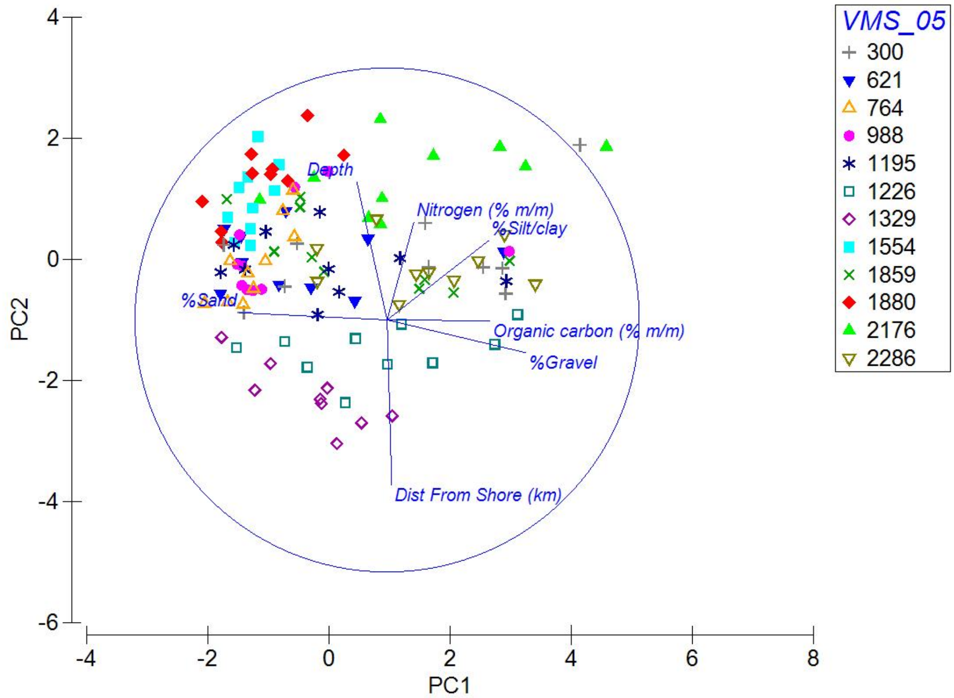


Figure 9. GIS plots representing spatial distribution of Particle Size Analysis (PSA) results as pie charts per station for Eastern English Channel and Outer Thames Estuary sites.

A Principal Components Analysis (PCA) was carried out to investigate whether environmental variables, including sediment type, water depth, organic carbon content and distance from shore based on station pressure scores could explain the observed distribution of stations (Figure 10 and Figure 11). The results demonstrated that water depth, % gravel, and % sand (within both survey areas) contributed the most to the observed distribution of stations. The difference in % sand and % gravel does not appear to be linked to either high or low pressure activity scores; therefore there is no evidence to link the observed sediment variability to the activity of fishing or to suggest that fishers are targeting a particular sediment type within our survey regions.



Eigenvalues

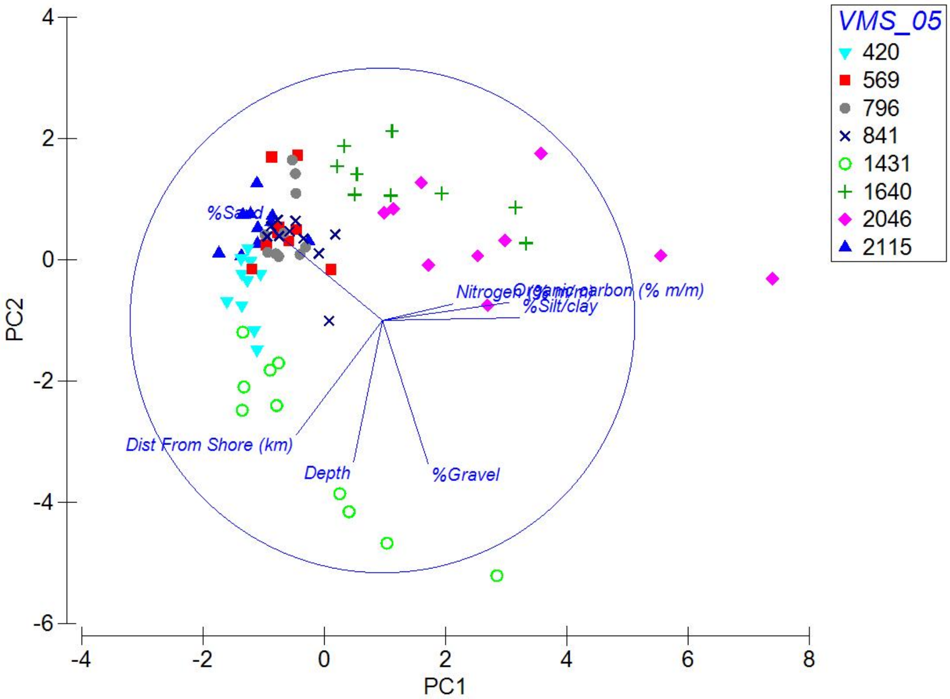
PC	Eigenvalues	%Variation	Cum.%Variation
1	2.55	36.4	36.4
2	1.31	18.8	55.2
3	0.953	13.6	68.8
4	0.839	12.0	80.8
5	0.751	10.7	91.5

Eigenvectors

(Coefficients in the linear combinations of variables making up PC's)

Variable	PC1	PC2	PC3	PC4	PC5
Depth	-0.121	0.550	-0.500	-0.164	-0.637
%Gravel	0.551	-0.130	0.041	0.332	-0.337
%Sand	-0.584	0.029	-0.002	-0.289	0.207
%Silt/clay	0.403	0.315	-0.200	-0.258	0.387
Dist From Shore (km)	0.018	-0.656	-0.042	-0.547	-0.410
Organic carbon (% m/m)	0.408	-0.004	-0.218	-0.537	0.259
Nitrogen (% m/m)	0.106	0.388	0.812	-0.354	-0.231

Figure 10. PCA produced in Primer v 6.1.15 for the Eastern English Channel, with Eigen values below. Environmental variables included depth, % gravel, %sand, %silt clay, distance from shore, organic carbon and % nitrogen. PCA is overlaid with pressures scores at the coarsest (0.05dd) pressures scores relating to original block selection.



Eigenvalues

PC	Eigenvalues	%Variation	Cum.%Variation
1	2.8	40.0	40.0
2	1.99	28.5	68.5
3	0.919	13.1	81.6
4	0.498	7.1	88.7
5	0.402	5.7	94.5

Eigenvectors

(Coefficients in the linear combinations of variables making up PC's)

Variable	PC1	PC2	PC3	PC4	PC5
Depth	-0.115	-0.562	0.313	-0.433	0.579
%Gravel	0.182	-0.570	-0.166	0.598	0.180
%Sand	-0.461	0.379	0.175	-0.058	0.306
%Silt/clay	0.542	0.010	-0.219	-0.271	-0.071
Dist From Shore (km)	-0.341	-0.453	0.106	-0.297	-0.709
Organic carbon (% m/m)	0.504	0.071	0.112	-0.445	0.040
Nitrogen (% m/m)	0.279	0.065	0.879	0.304	-0.170

Figure 11. PCA produced in Primer v 6.1.15 for the Outer Thames Estuary, with Eigen values below. Environmental variables included depth, % gravel, %sand, %silt clay, distance from shore, organic carbon and % nitrogen. PCA is overlaid with pressures scores at the coarsest (0.05dd) pressures scores relating to original block selection.

3.3 Infauna community analysis

Summary biodiversity indices per station are provided in Appendix 6.1.

Initial analyses were conducted on the entire dataset (both regions) and the observed trends were positive and all but one relationship were statistically significant at the 5% level (Table 3).

Table 3. P-values and direction of trend for the biological variables against different pressure scores.

Trend	p-value	Observed S
Biomass - VMS05	0.06	+ve
Biomass - VMS025	0.004	+ve
Biomass - VMS0125	0.001	+ve
Richness - VMS05	0.006	+ve
Richness - VMS025	<0.001	+ve
Richness - VMS0125	<0.001	+ve
Abundance - VMS05	0.019	+ve
Abundance - VMS025	<0.001	+ve
Abundance - VMS0125	<0.001	+ve

However, the magnitudes of the trends were small, indicated by small (<13%) associated R^2 values (coefficient of determination) (Figure 12). This suggests that there was a small, but significant, positive relationship between fishing pressure and biodiversity indices. As correlation does not imply causation, then this does not necessarily mean that one variable causes the other, as they may simply be occurring together or both be responding similarly to an additional covariate.

3.3.1 Eastern English Channel Infauna

A similar pattern was observed in the subset of EEC infaunal data: relationships between fishing pressure and biological indices tended to be statistically significant, small and positive (Figure 12). Outputs of simple linear regression analyses are summarised in Appendix 6.3.

For EEC, the number of biodiversity indices that have a significant relationship with fishing pressure scores increases with decreasing cell size and the coefficient of determination also increases accordingly. For the 0.05^{o} grid size, only the Shannon diversity index (H') correlated significantly with fishing pressure score, whereas for the 0.0125^{o} grid size, number of species ($\sqrt{S}$), Margalef species richness (d), and Shannon diversity index (H') all correlated significantly with fishing pressure score. Best fit was at the 0.0125^{o} grid size using the Shannon index (H') where p<0.001and R^2 was 12%. Neither of the fishing pressure scores derived from nested grids performed as well as the standard grid.

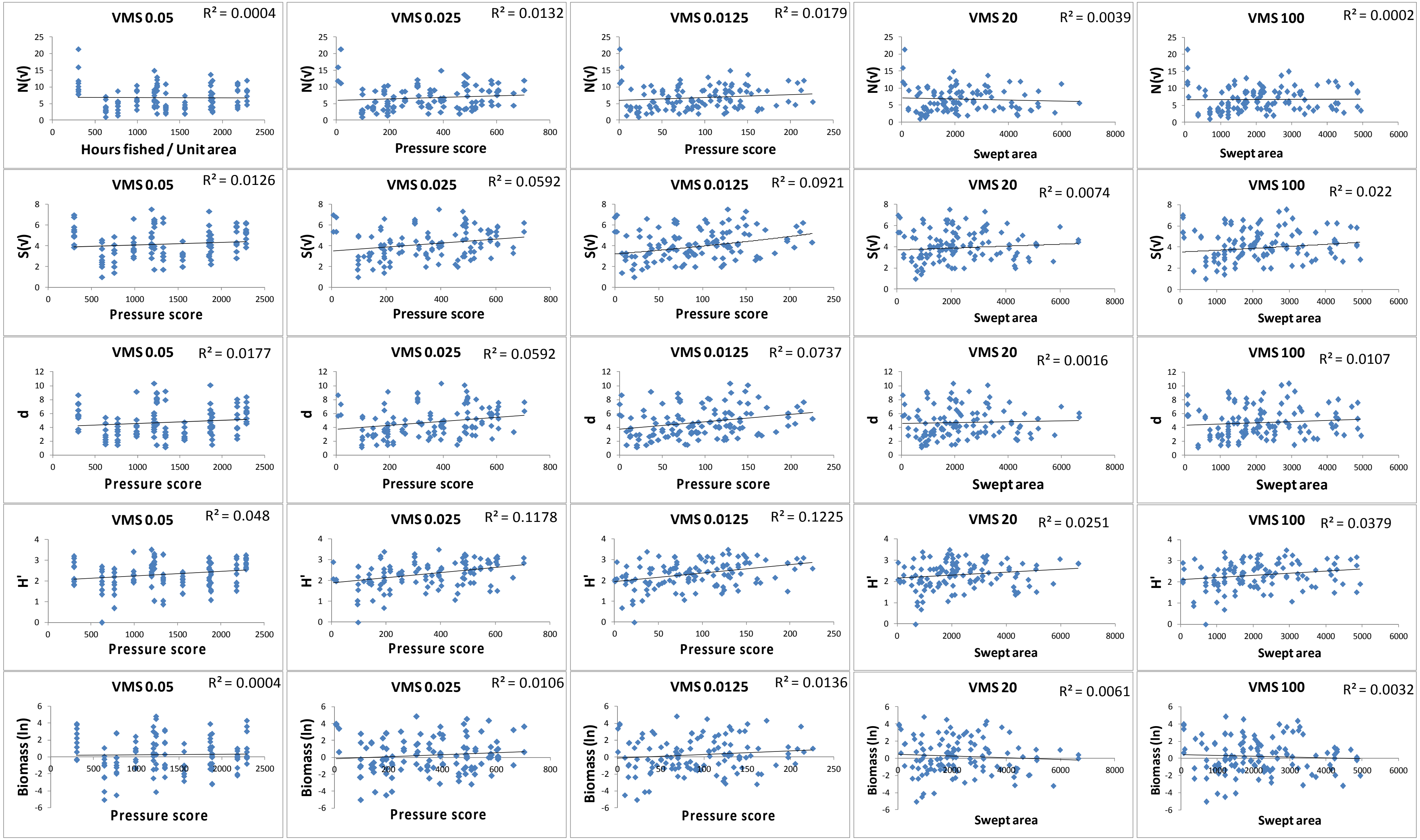


Figure 12. Linear regression plots based on univariate community metrics SQRT N (number of individuals),SQRT S (number of species), d (Margalef species richness), H' (Shannon Weiner species diversity) and ln Biomass calculated for the Eastern English Channel infaunal community. Linear regressions were carried out at VMS grid resolutions of 0.05dd, 0.025dd and 0.0125dd and for nested gridding based on the minimum ping numbers of 20 and 100.

Biodiversity indices correlate strongly with a suite of environmental factors, where most relationships are significant and the environmental parameters explain a large proportion of the biological variability (Table 4). Depth and sediment group were most consistently significant of the predictors across biological indices, but silt content, distance from shore and organic carbon content of the sediment were also present in most of the best performing models.

Table 4. Multi-linear regression analysis for community univariate indices √s, √N, d, H' and ln Biomass, at EEC were all run against the variables depth, %silt/clay, distance from shore, Organic Carbon, %Nitrogen, sediment group and sediment Mode 1, to identify which were having a significant effect on regression fit and how much each variable was proportionally adding to that fit.

	√S			√N			d		
	P	SSinc[7]	SSMarg	P	SSinc	SSMarg	P	SSinc	SSMarg
Depth	0.010	0.77	7.39	0.061	11.86	25.26	0.020	2.34	14.53
%Silt/clay	0.025	42.88	5.55	<0.001	344.99	85.67	0.040	86.33	11.32
Dist Shore	0.006	11.62	8.53	0.016	47.04	42.07	0.004	32.25	22.42
Organic Carbon	0.015	24.90	6.59	0.038	102.45	31.18	0.012	58.57	17.25
Nitrogen %	0.583	0.51	0.33	0.924	0.94	0.07	0.573	1.04	0.83
Group	<0.001	37.33	37.32	<0.001	154.30	161.81	<0.001	72.25	71.15
Mode 1	0.811	0.06	0.06	0.128	16.54	16.54	0.798	0.17	0.17
	$r^2 = 0.583$			$r^2 = 0.551$			$r^2 = 0.554$		

	H'			(ln) Biomass			VIF[8]
	P	SSinc	SSMarg	P	SSinc	SSMarg	Global
Depth	0.014	0.008	1.700	0.059	0.540	11.445	1.339
%Silt/clay	0.462	3.517	0.148	0.092	33.909	9.046	1.472
Dist Shore	0.077	1.507	0.872	0.002	38.447	32.275	1.363
Organic Carbon	0.308	2.622	0.285	0.013	34.152	20.121	1.390
Nitrogen %	0.782	0.014	0.021	0.792	0.512	0.218	1.077
Group	<0.001	6.973	6.642	0.061	10.501	11.292	1.564
Mode 1	0.222	0.411	0.411	0.380	2.429	2.429	1.093
	$r^2 = 0.413$			$r^2 = 0.329$			

Tables showing collinearity of explanatory variables are given in Appendix 6.2. Pearson's correlation coefficient was found to be high only between individual fishing scores[9] and between particular measures of sediment size[10].

[7] ***SSincr** is the incremental (Type) I sum of squares, a measure of the new predictive information contained in an independent variable, as it is added to the equation. One can gauge the additional contribution of each independent variable by comparing these values. **SSmarg** is the marginal (Type III) sum of squares, a measure of the unique predictive information contained in an independent variable, after taking into account all other independent variables. One can gauge the independent contribution of each independent variable by comparing these values.*

[8] *Variance Inflation Factor (VIF) indicates whether there is multiple collinearity in the model. A VIF of ≥5 typically indicates a multi-collinearity problem (O'Brien, 2007).*

[9] *Fishing pressure scores derived from 0.05° and 0.025° grids; fishing pressure scores derived from 0.05° and 0.0125° grids; and fishing pressure scores derived from 0.025° and 0.0125° grids*

[10] *For all data, between sand (%) and gravel (%); for EEC, between sand (%) and gravel (%); and for Th, between silt/clay (%) and sand (%)*

3.3.2 Outer Thames Infauna

The same overall pattern was observed in the Th infaunal data: relationships between fishing pressure and biological indices tended to be significant, small and positive (Figure 13). However, at this site, there were more significant relationships between biodiversity indices and fishing pressure scores at the largest cell size (0.05dd) and the values for the coefficient of variation were highest. (Appendix 6.3). Here, the best fit was with abundance (√) with $p=0.002$ and an associated R^2 of 12%. Again, the performance of the nested pressure scores was not as good as those derived from standard grids.

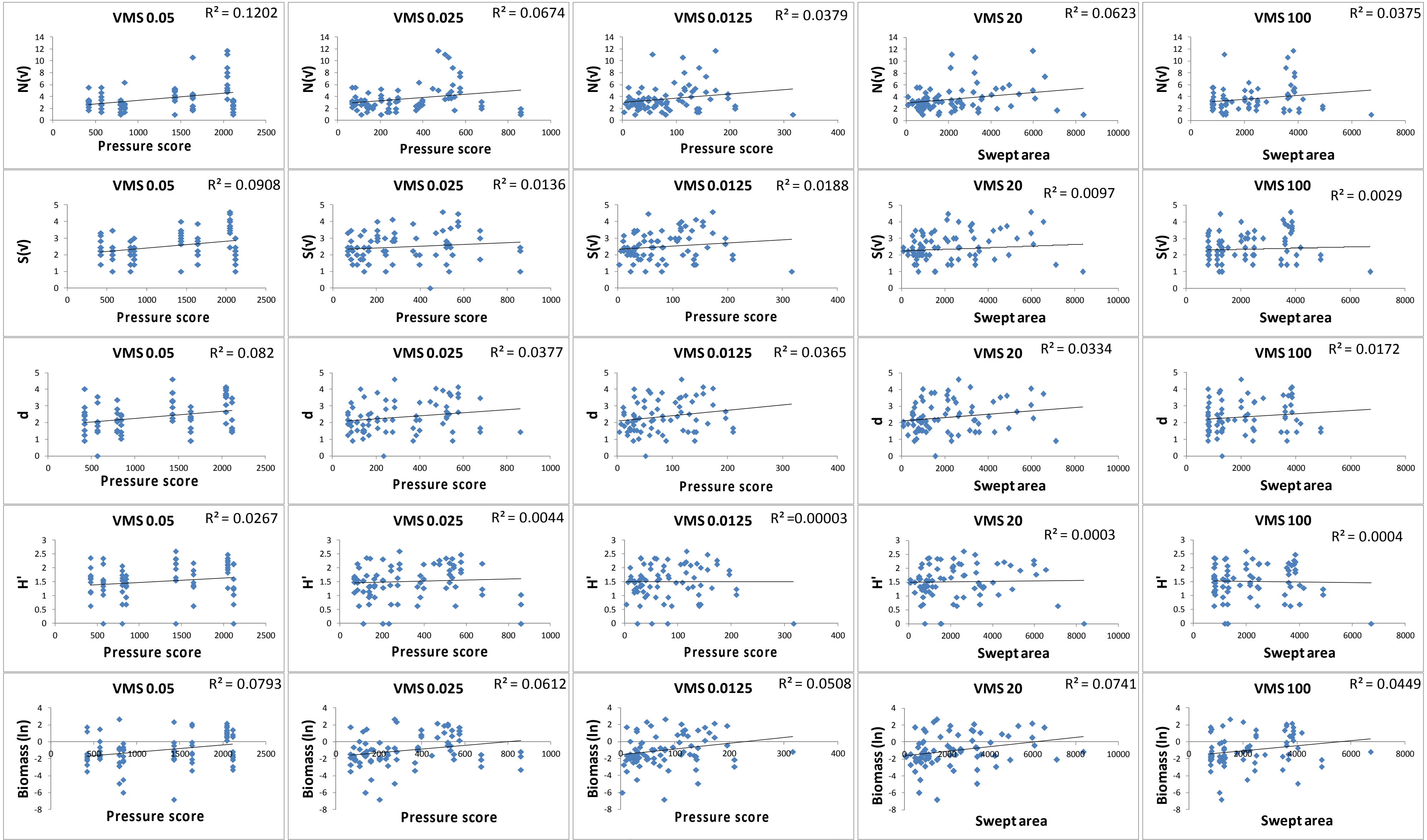


Figure 13. Linear regression plots based on univariate community metrics √N (number of individuals),√S (number of species), d (Margalef species richness), H' (Shannon Weiner species diversity) and ln Biomass calculated for the Outer Thames Estuary infaunal community. Linear regressions were carried out at VMS grid resolutions of 0.05dd, 0.025dd and 0.0125dd and for nested gridding based on the minimum ping numbers of 20 and 100.

Biodiversity indices at the sites again appeared to be largely driven by a suite of environmental factors, which are statistically significant and explain a large proportion of the biological variability (Table 5).

Table 5. Multi-linear regression analysis for community univariates √s, √N, d H' and ln Biomass, at Th were all run against the variables depth, %silt/clay, distance from shore, Organic Carbon, %Nitrogen, sediment group and sediment Mode 1, to identify which were having a significant effect on regression fit and how much each variable was proportionally adding to that fit.

	√S			√N			d		
	P	SSinc	SSMarg	P	SSinc	SSMarg	P	SSinc	SSMarg
Depth	0.019	1.874	3.328	0.158	0.002	6.533	0.011	5.784	5.335
%Silt/clay	0.828	5.412	0.028	0.122	38.773	7.854	0.828	3.979	0.037
Dist Shore	0.282	1.245	0.682	0.209	17.456	5.151	0.615	0.265	0.198
Organic Carbon	0.011	5.083	3.932	<0.001	81.404	61.372	0.122	2.781	1.899
Nitrogen %	0.782	0.028	0.045	0.799	0.230	0.208	0.981	0.012	0.000
Group	0.324	0.116	0.572	0.054	8.300	12.278	0.616	0.003	0.197
Mode 1	0.110	1.519	1.519	0.207	5.200	5.200	0.108	2.054	2.054
	r^2= 0.276			r^2= 0.407			r^2= 0.225		

	H'			(ln) Biomass			VIF
	P	SSinc	SSMarg	P	SSinc	SSMarg	Global
Depth	0.023	1.441	1.840	0.854	3.440	0.103	1.550
%Silt/clay	0.891	1.281	0.006	0.698	22.919	0.461	1.965
Dist Shore	0.291	0.272	0.282	0.305	3.896	3.236	1.947
Organic Carbon	0.079	0.860	1.073	0.145	12.955	6.567	2.144
Nitrogen %	0.332	0.263	0.323	0.512	1.816	1.317	1.326
Group	0.959	0.123	0.001	0.693	0.340	0.476	1.674
Mode 1	0.155	0.701	0.701	0.050	12.040	12.040	1.270
	r^2= 0.175			r^2= 0.216			

At the Th site there were fewer significant relationships between environmental parameters and biodiversity indices, relative to the EEC site. This was significant in itself, as one might have expected similar factors to relate to the community structure given the similar sedimentary habitats in the two areas. Both depth and sediment organic carbon content were the most consistently significant predictors, often on their own.

3.4 Epifauna community analysis

In general there were far fewer significant relationships between the epifauna biodiversity indices and fishing pressure scores. This was possibly an artefact of the smaller number of samples and hence reduced sampling power.

3.4.1 Eastern English Channel Epifauna

Significant p values were observed for $\sqrt{S}$ (number of species) and d (Margalef species richness) at a VMS grid resolution of 0.05dd (Figure 14). In this instance, the r value was much larger than for the infauna, suggesting that fishing pressure is a relatively good predictor of biodiversity at this site. As with the infauna relationships, the trend was positive.

3.4.2 Outer Thames Estuary Epifauna

The only significant p values observed was $\sqrt{S}$ (number of species) at a VMS grid size of 0.0125dd (Figure 15). In this instance, the r value was much larger than for the infauna, suggesting that fishing pressure is a relatively good predictor of biodiversity at this site. As with the infauna relationships, the trend was positive.

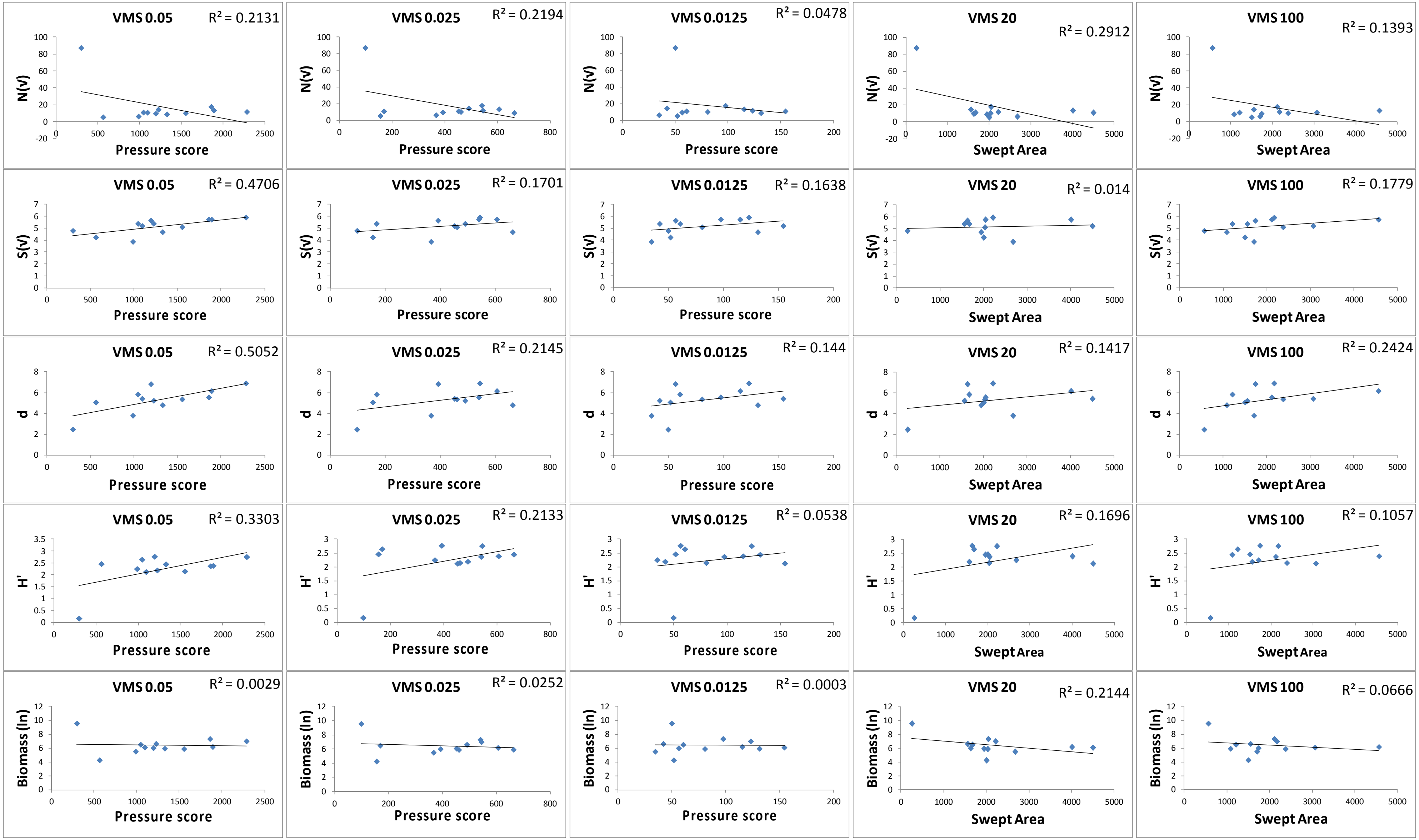


Figure 14. Linear regression plots based on univariate community metrics √N (number of individuals), √S (number of species), d (Margalef species richness), H' (Shannon Weiner species diversity) and ln Biomass calculated for the Eastern English Channel epifaunal community. Linear regressions were carried out at VMS grid resolutions of 0.05dd, 0.025dd and 0.0125dd and for nested gridding based on the minimum ping numbers of 20 and 100

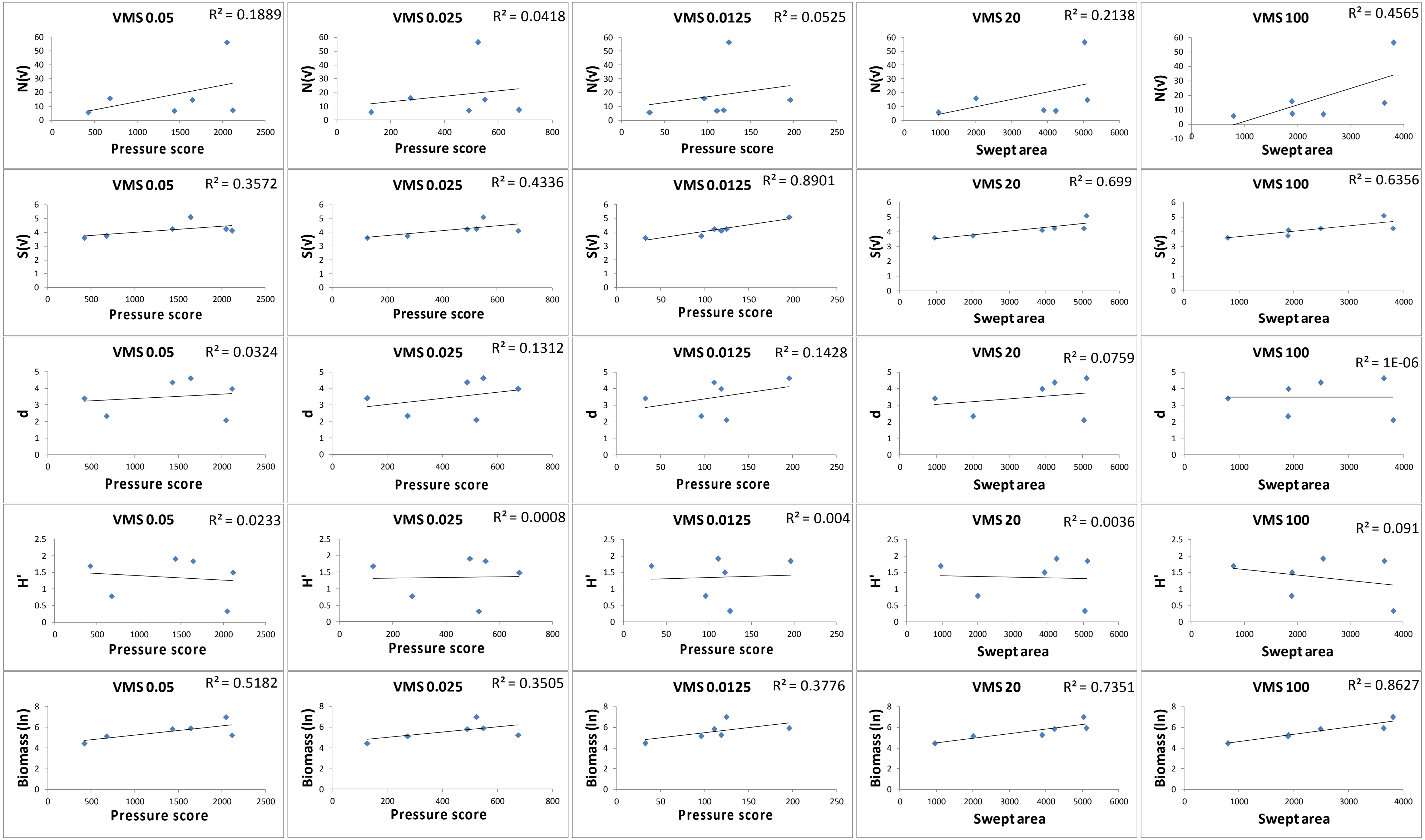


Figure 15. Linear regression plots based on univariate community metrics √N (number of individuals),√S (number of species), d (Margalef species richness), H' (Shannon Weiner species diversity) and ln Biomass calculated for the Outer Thames Estuary epifaunal community. Linear regressions were carried out at VMS grid resolutions of 0.05dd, 0.025dd and 0.0125dd and for nested gridding based on the minimum ping numbers of 20 and 100

3.5 Modelling infauna biodiversity indices

Number of species (S)
Table 5 shows MRAE values for different models. For EEC, the VMS variables do not significantly add to the best models and the VMS_20 and VMS_100 variables seem marginally worse than the others. Of the environmental predictors, sediment group (45.6), % gravel (46.9) and carbon content (49.2) perform better.

Differences are less obvious for Th but, again, carbon content performs well (52.9). The best minimal model for Thames includes carbon and VMS.05 (52.6). However, this is only marginally better than for carbon alone.

Table 6. MRAE for different univariate prediction models for infaunal data at EEC and Th.

	No. Of species (S)		Margalef (d)		Biomass	
	EEC	Thames	EEC	Thames	EEC	Thames
Constant	58.4	57.9	37.7	33.8	122.8	110.7
VMS_05	59.8	54.6	37.9	32.8	128.6	117.7
VMS_025	59.2	55.3	37.1	33.0	133.5	114.2
VMS_0125	58.5	56.8	37.1	33.3	131.2	115.1
VMS_20	59.7	56.0	38.3	33.2	124.0	118.8
VMS_100	60.4	57.9	38.4	34.0	124.9	116.8
% silt	53.8	56.4	34.8	34.0	132.3	121.8
% gravel	46.9	58.3	28.6	34.5	123.0	114.2
Depth	59.7	57.4	38.2	32.6	125.7	112.8
Distance	58.7	60.5	37.4	34.9	110.2	122.7
Carbon content	49.2	52.9	32.0	32.9	114.7	124.0
Nitrogen content	59.8	58.4	38.4	34.8	126.7	126.9
Sediment group	45.6	-	29.5	-	121.5	-
Mode	58.0	-	36.5	-	122.4	-
Best minimal model	41.7 Sediment group, Carbon, %silt	52.6 Carbon, VMS 0.05dd	26.4 VMS 0.0125dd, %silt, Carbon, Sediment group	31.3 VMS 20, Depth Carbon	103.8 Distance from shore, Carbon	110.7 No variables

Margalef (d)
For EEC, %gravel (28.6), sediment group (29.5) and organic carbon content (32.0) perform well. The best minimal model we could find uses VMS.0125, % silt and organic carbon content (26.4).

For Th, the single models perform little better than the one with no variables (33.8). As a general principle, prediction is worse for Th than for the EEC. Depth, organic carbon content and VMS.20 combined yield the best minimal model (31.3). This is marginally better than the alternative model with depth, carbon content and VMS.05.

Biomass (b)
For EEC, distance from shore (110.2) and organic carbon content (114.7) yielded the best predictions when used individually. The best minimal model was with distance from shore and organic carbon content considered together (103.8). The VMS_20 and VMS_100 pressure variables performed better relatively than the other pressure variables, but worse than no variables and much worse than for the variables making up the minimal model.

For Th, no model containing explanatory variables produced a better fit than simply using the mean level to predict.

In general at the EEC site, VMS_20 and VMS_100 were slightly worse predictors than the other pressure variables, whereas at the Th site VMS_20 was found to be equal to or better than other pressure variables, though prediction power was found to be less for Th than EEC.

Figure 16, Figure 17 and Figure 18 show plots of each explanatory variable and the three best performing pressure scores for the EEC data. Note that there are some outliers – however, these did not particularly affect the MRAE conclusions from Table 6, and so they were not removed from the EEC data.

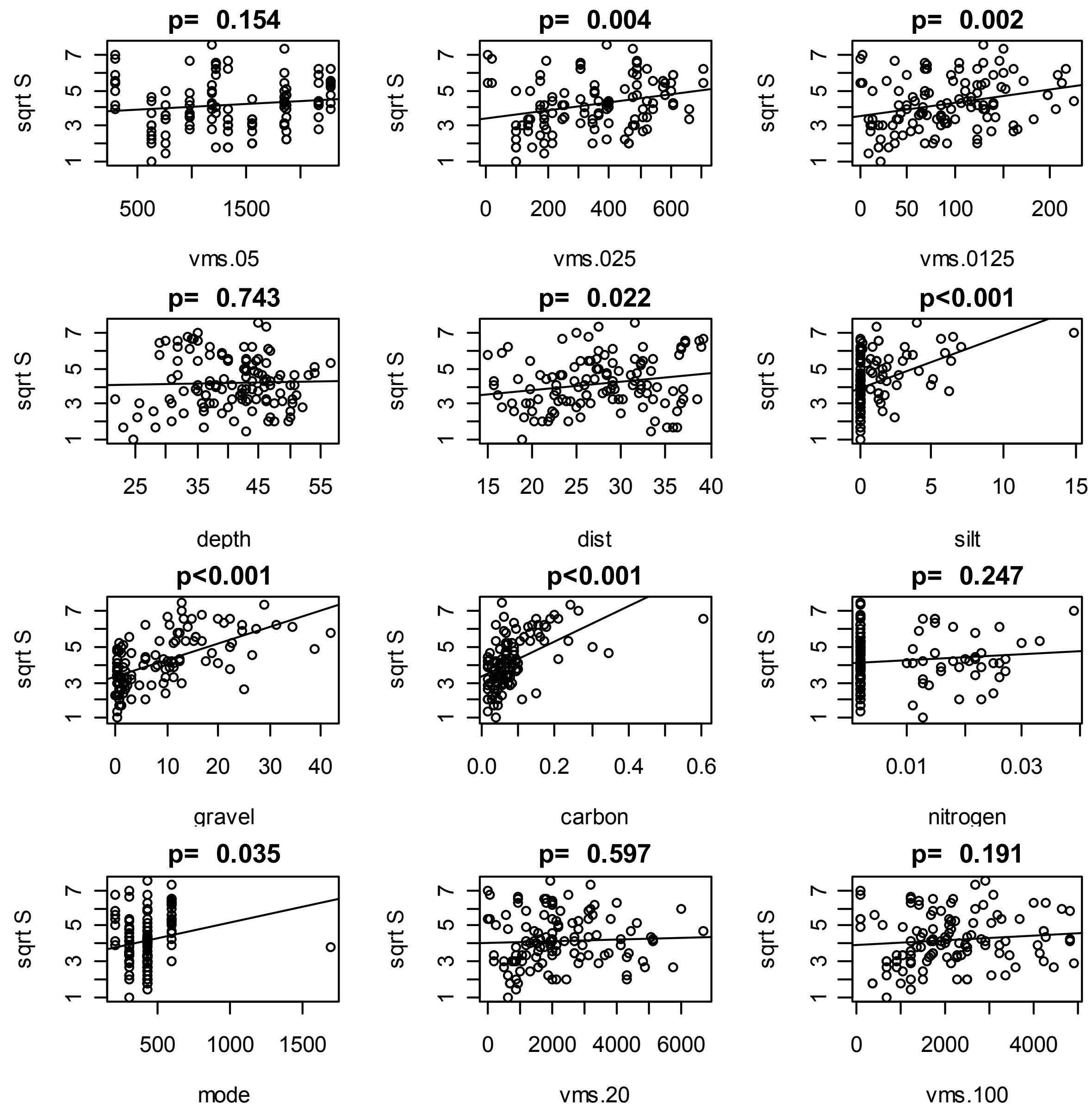


Figure 16. EEC infaunal data plots for square root of S against variables, created in R.

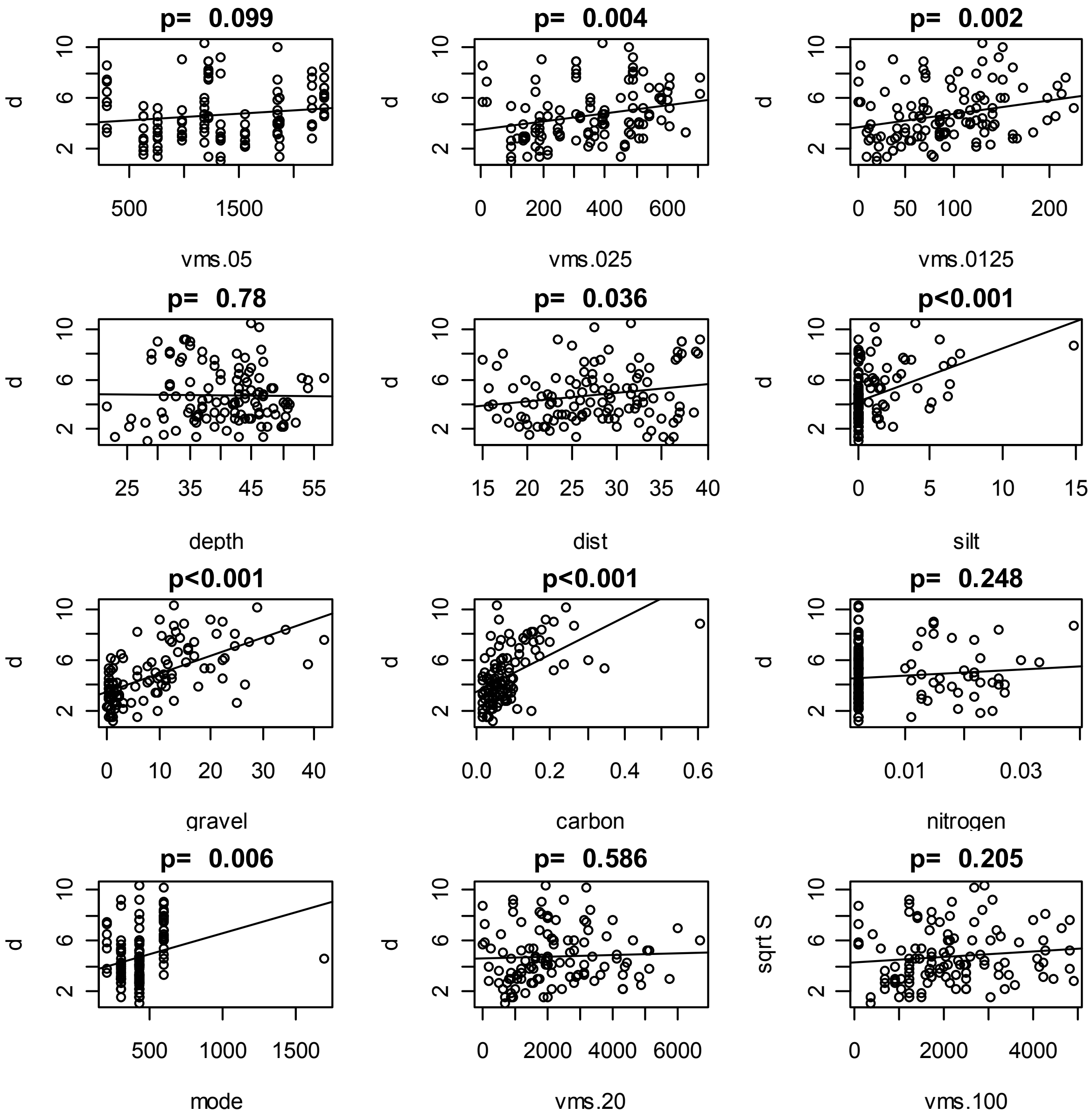


Figure 17. EEC infaunal data plots for d against variables, created in R.

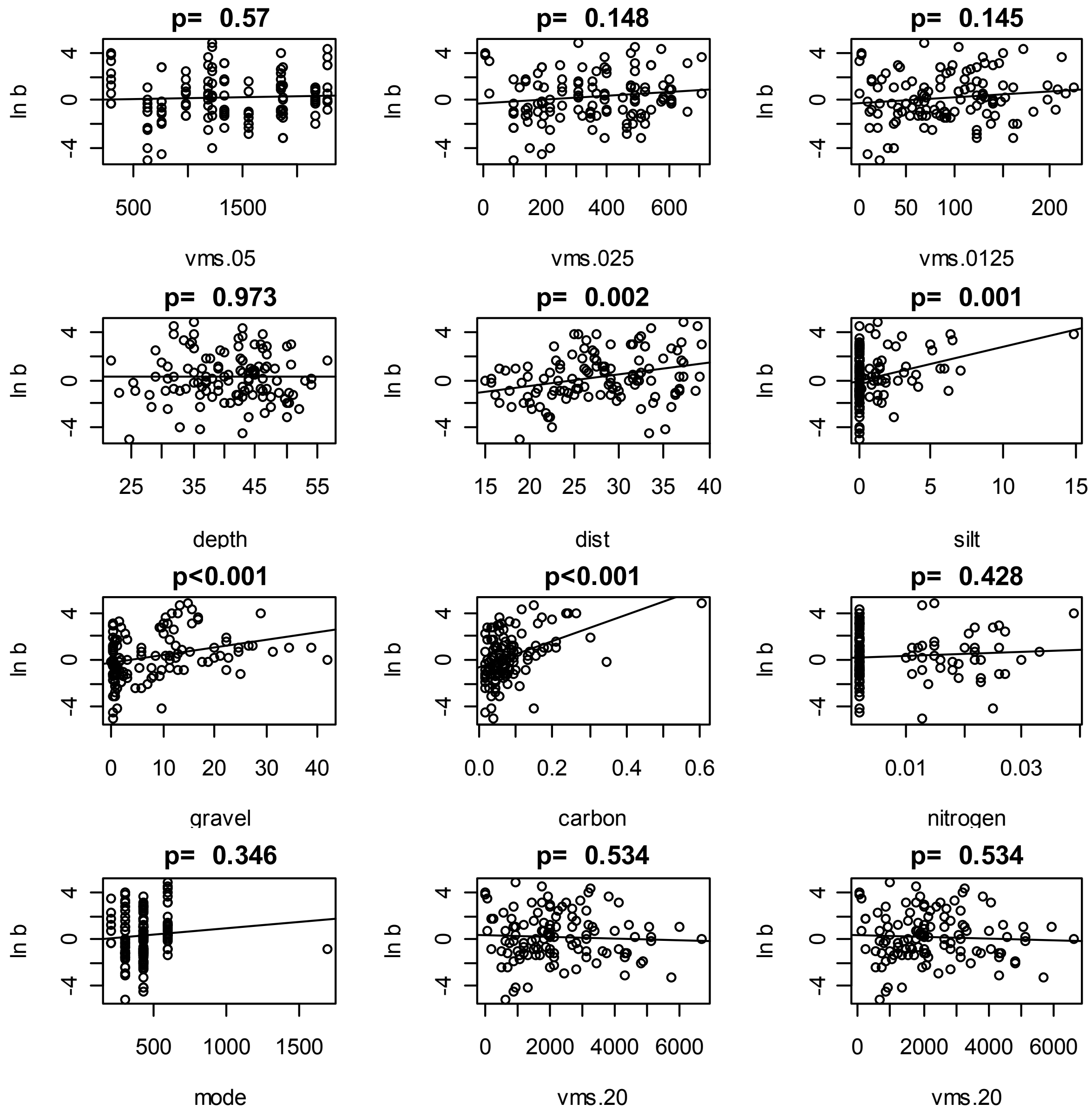


Figure 18. EEC infaunal data plots for natural log of biomass against variables, created in R.

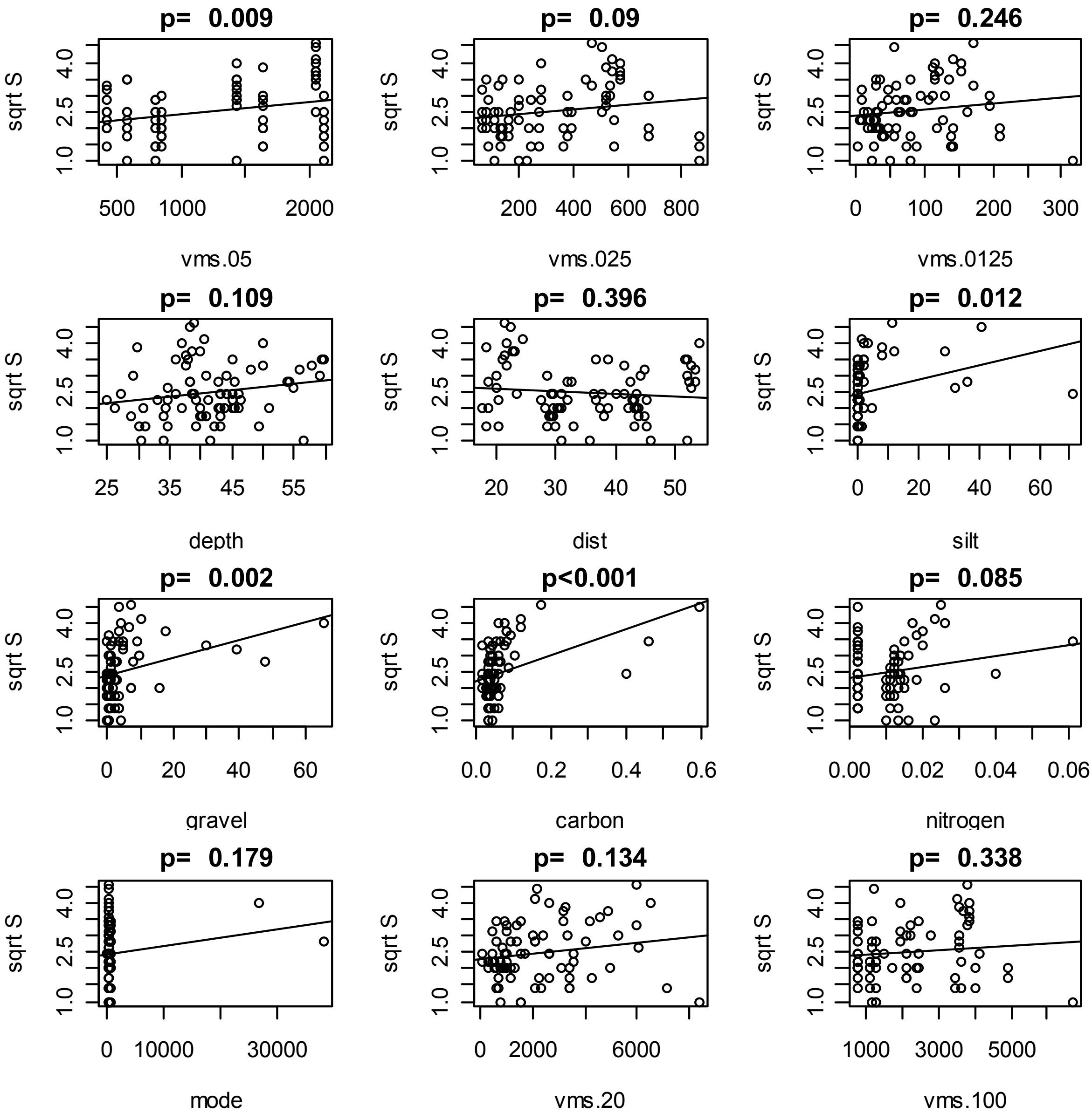


Figure 19. Th infaunal data plots for square root of S against variables, created in R.

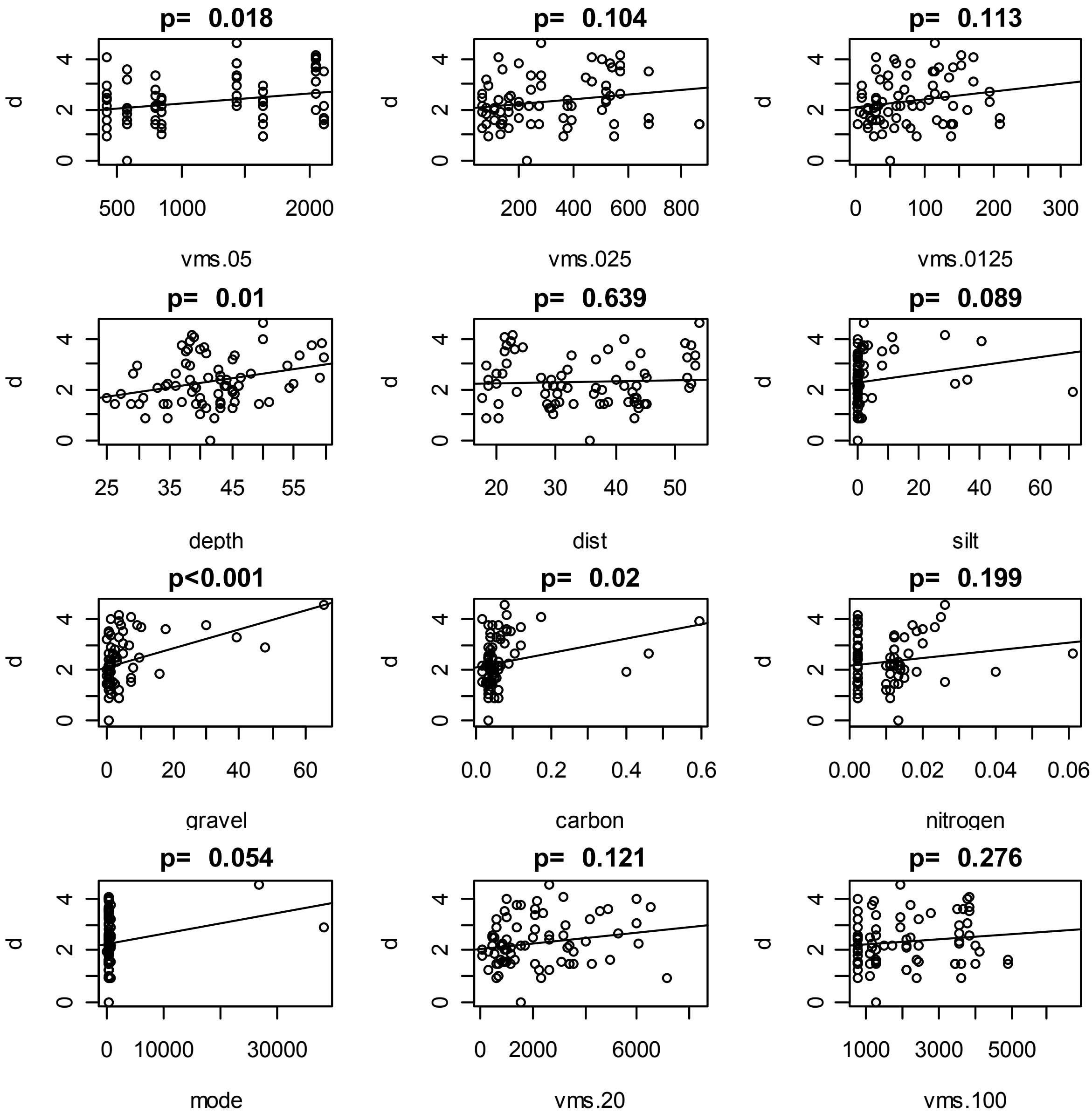


Figure 20. Th infaunal data plots for d against variables, created in R.

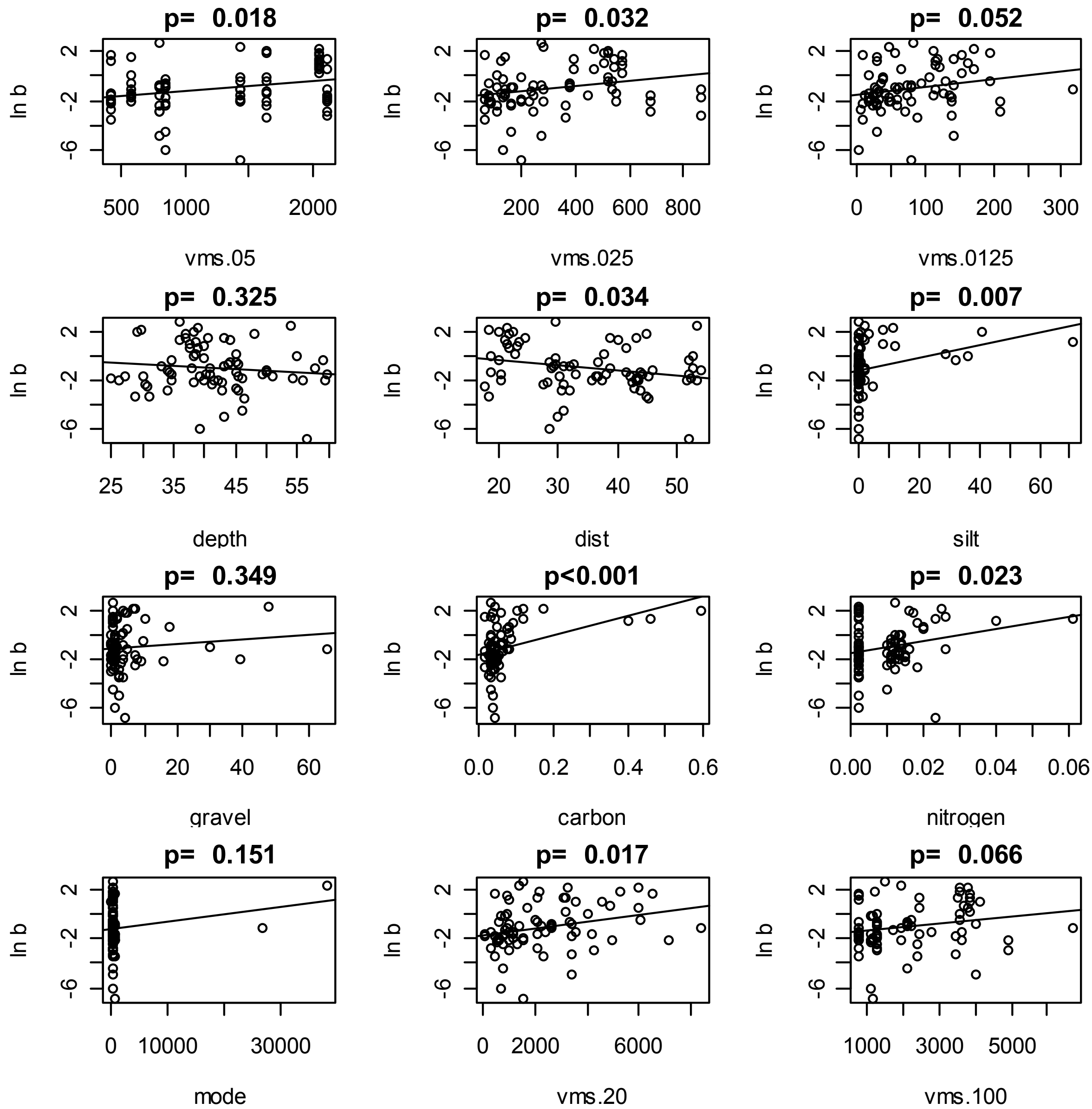


Figure 21. Th infaunal data plots for the natural log of biomass against variables, created in R.

Figure 19, Figure 20 and Figure 21 show plots of each explanatory variable against the three environmental variables for the Thames data. There are two very large outliers for Mode. We did not want to remove both of these cases and so this variable was not used for the MRAE calculations. In addition, the sediment group variable had 8 classes – some of which had only 2 or 3 cases. This meant that the MRAE prediction method could not be used for this variable because the training data set did not always contain cases with all 8 classes.

3.6 Modelling epifauna biodiversity indices

Analysis was more difficult for the trawl data because there were far fewer observations (12 for EEC, 6 for Thames). In addition, one of the EEC biomass values was very high (14,390 for station EEC_06_04). Given that the other biomass values were of the order of 500, this high value made prediction on the original scale problematic. Thus, this value was removed for the prediction results reported below.

As there are only four potential explanatory variables for the trawl stations (the three pressure variables plus distance to shore) we fitted all possible models (i.e. no variables, each variable on its own, distance to shore coupled with each pressure variable).

Table 7 summarises the MRAE values for the various models, though these should be treated with caution because models have been fitted with only a very few observations.

The best models are shown in red. Note that for the Margalef index for Thames, none of the explanatory variables help with the prediction (i.e. just using the mean value of the biodiversity index does better than adjusting this mean value for explanatory variables).

Table 7. MRAE for univariate prediction models for epifaunal data at EEC and Th.

	No. Of species (S)		Margalef (d)		Biomass	
	EEC	Thames	EEC	Thames	EEC	Thames
Constant	21.4	22.2	18.1	30.5	59.8	103.4
VMS_05	17.5	21.8	17.2	37.5	55.5	69.5
VMS_025	22.0	21.5	21.5	40.3	86.2	93.7
VMS_0125	23.5	14.8	20.4	46.4	67.1	130.8
Distance	25.1	35.2	20.8	44.8	72.1	163.7
VMS_05 + Distance	19.6	33.1	19.6	47.7	50.5	105.1
VMS_025 + Distance	25.7	34.1	24.6	49.7	89.4	117.7
VMS_0125 + Distance	26.4	19.4	23.1	57.9	67.6	346.9

NB: Lowest MRAE is shown in red.

Plots of each of the four variables against each biodiversity variable are shown in Figure 22(EEC) and Figure 23(Th). No very strong relationships were observed.

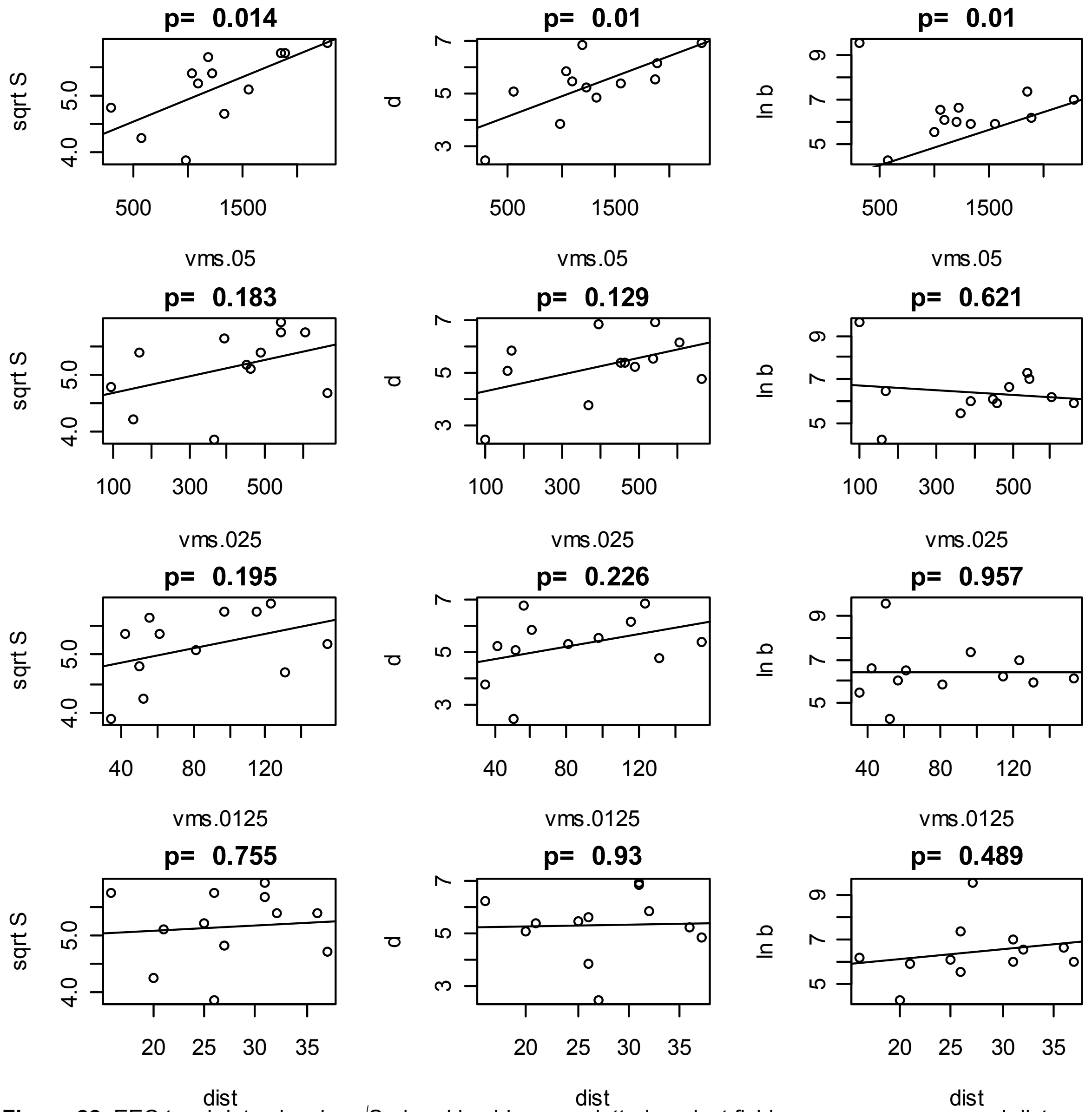


Figure 22. EEC trawl data showing √S, d and log.biomass plotted against fishing pressure scores and distance from shore.

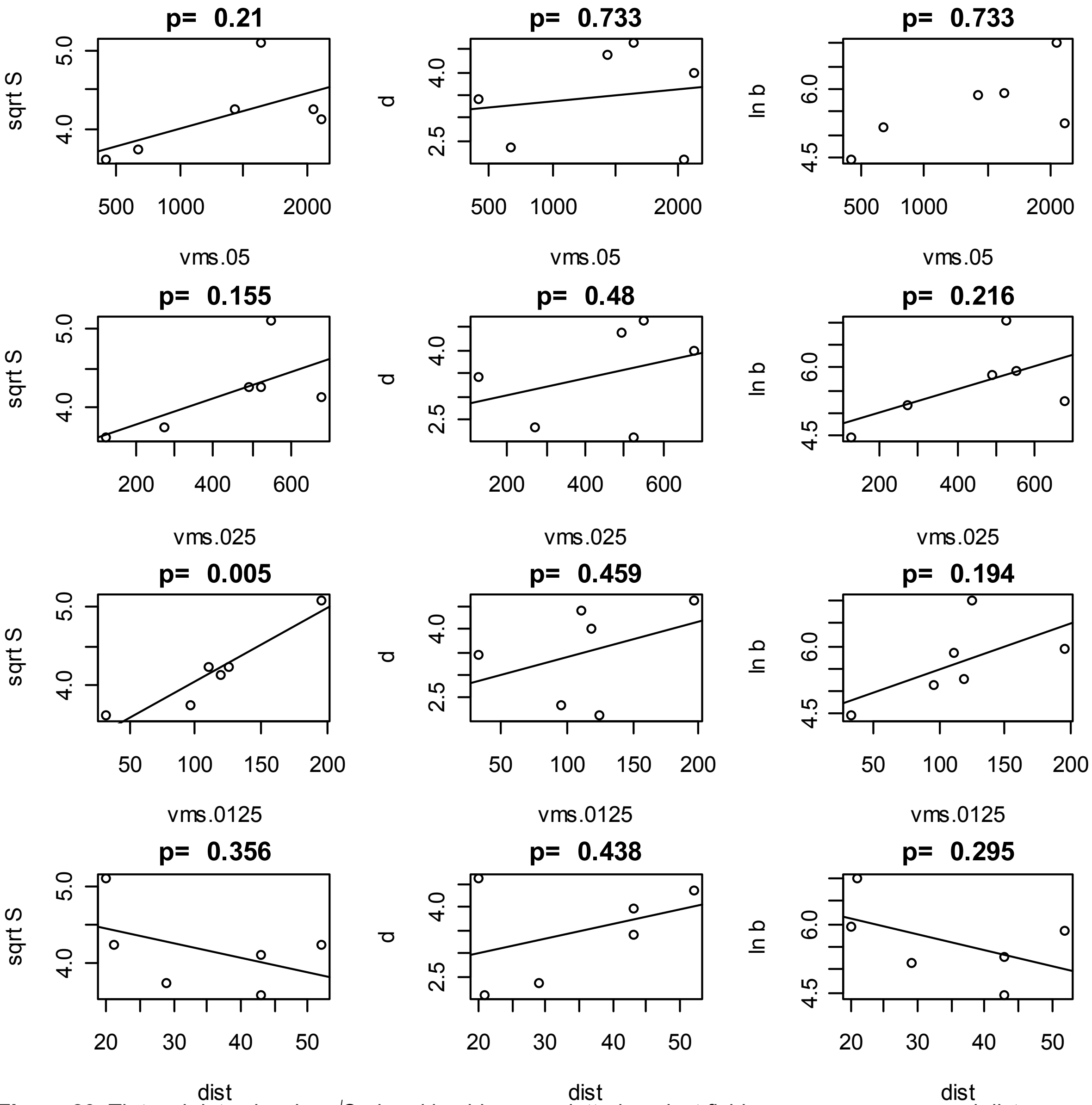


Figure 23. Th trawl data showing √S, d and log.biomass plotted against fishing pressure scores and distance from shore.

3.7 W-stat analysis

Infaunal W-statistics, at EEC, for all grid resolutions, showed a very poor increasing linear fit, with no p-values being statistically significant. Infaunal communities at Th, however, demonstrated a decreasing W-statistic with increasing fishing pressure, though again with a very poor linear fit and no statistical significance. To that end it would be assumed that infaunal communities' W-statistics were not responding to increases in exposure to abrasive pressure at these two sites.

When looking at the epifaunal community W-statistics at Th and EEC we again find them mirroring one another in terms of their trends, but this time it is EEC that demonstrated a decreasing W-statistic with increasing abrasive fishing pressure scores whilst for Th the W-statistic was increasing.

What is of note here is that both EEC and Th present stations below zero, which is quoted by Clarke and Warwick (2001) as the value at and below which a community is expected to be impacted. Though linear regression fits the data better for epifauna than it does for infauna, at both EEC and Th, neither are statistically significant. This is likely to be an artefact of reduced variability caused by the lower sample size. It does appear that Th is a more disturbed site overall than EEC for epifauna, with many of the communities with W-statistics less than zero regardless of attributed fishing pressure score.

3.7.1 Infauna

No changes in the infaunal communities at the EEC and Th sites in terms of changes in abundance/biomass dominance were observed across the perceived fishing activity gradient (Figure 24). The associated coefficient of determination (R^2) values were also extremely low, indicating that the linear regression was a poor fit to the data, with none of these returning statistically significant p-values at the 5% level (Appendix 1; Table 5).

3.7.2 Epifauna

A decrease in the W statistic was observed across the fishing activity gradient at the EEC site, indicating a shift towards a disturbed community. However, the low coefficient of determination (R^2) values indicate that the linear regression is a poor fit to the data, with no statistically significant p-values being observed (Appendix 1; Table 5), therefore no relationship could be inferred between the W statistic and fishing activity (Figure 25).

At the Th survey sites, values of the W statistic were primarily found to be negative, indicating a disturbed community. However, as the fishing activity increased, the W statistic became more positive, indicating a move towards an undisturbed community, the opposite of what might be expected. However the low coefficient of determination (R^2) values indicate, along with no statistically significant p-values (Appendix 1; Table 5), that the linear regression is a poor fit to the data. The negative values across the fishing activity gradient at the Th site may indicate that the whole site has been disturbed at some point and has not yet recovered (Figure 25).

The low number of epifaunal samples obtained meant that there was little statistical power, and therefore low confidence, in these particular results.

EEC **Th**

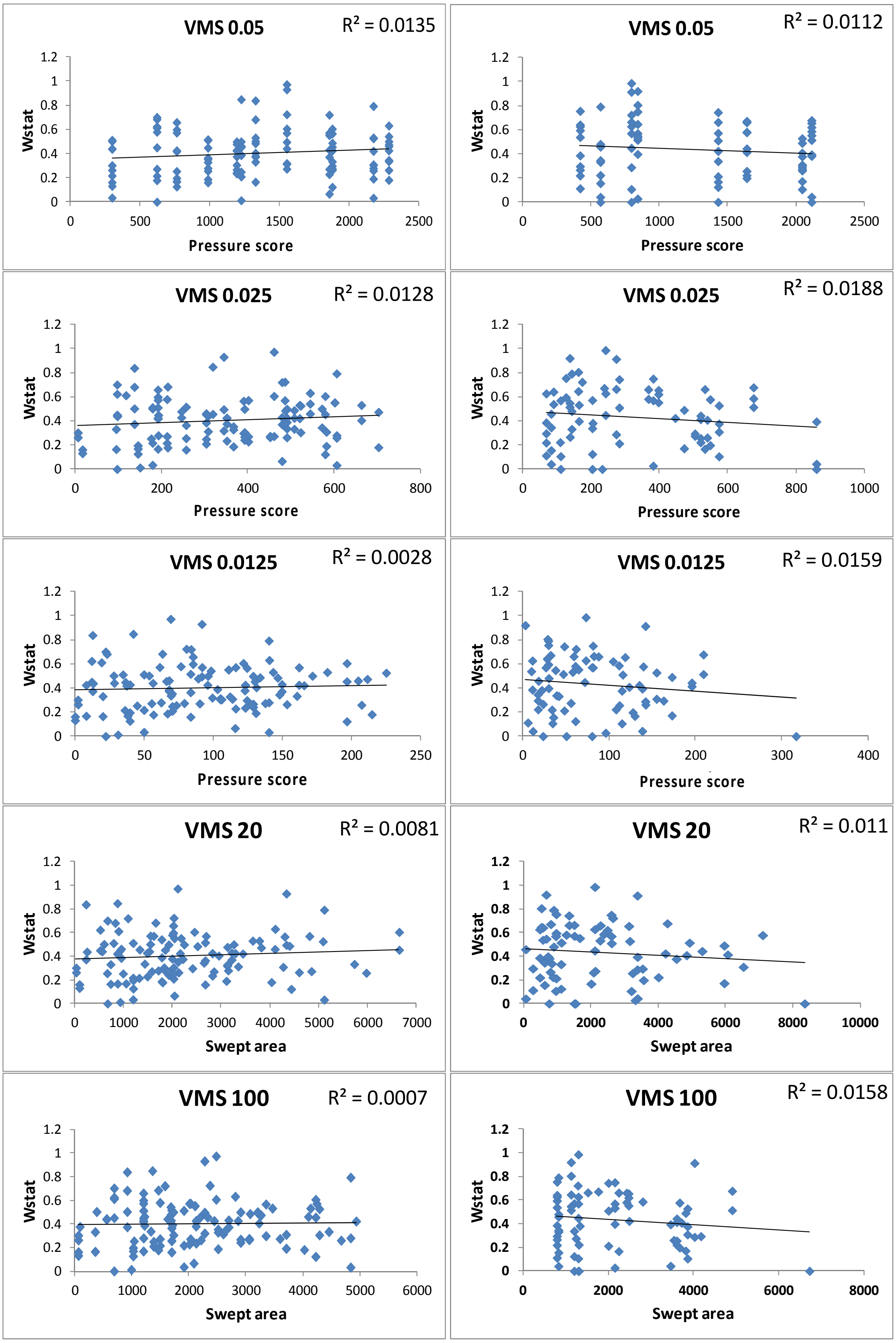


Figure 24. Linear regression plots based on values of the W statistic for the Eastern English Channel and Outer Thames Estuary infaunal communities. Linear regressions were performed at VMS grid resolutions of 0.05dd, 0.025dd and 0.0125dd.

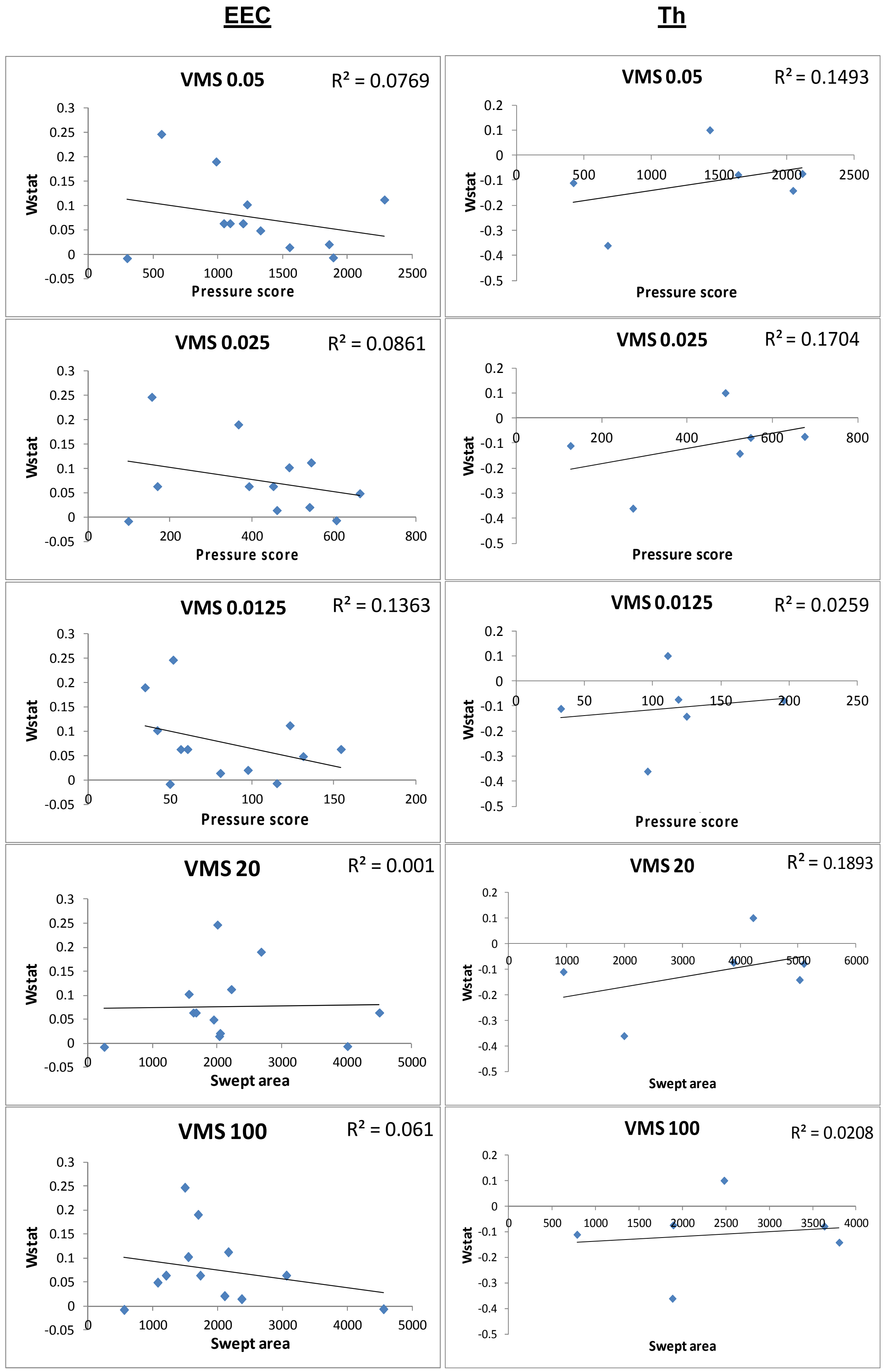


Figure 25. Linear regression plots based on values of the W statistic for the Eastern English Channel and Outer Thames Estuary epifaunal communities. Linear regressions were performed at VMS grid resolutions of 0.05dd, 0.025dd and 0.0125dd.

3.7.3 Average Wstat per block

One way ANOVA of each site in relation to differences between blocks at a grid resolution of 0.05dd, identified few blocks that were significantly different from one another at the 5% level (Figure 26; Appendix A Tables 6 and 7).

Block 3 at EEC (pressure score of 1554) appeared to be significantly different from all other blocks except from blocks 2, 7 10 and 12 (pressure scores of 621, 1859, 2286 and 1329 respectively).

This suggests that communities at both sites may already have been in a disturbed state, either from natural disturbance, historic fishing, or due to another factor.

EEC **Th**

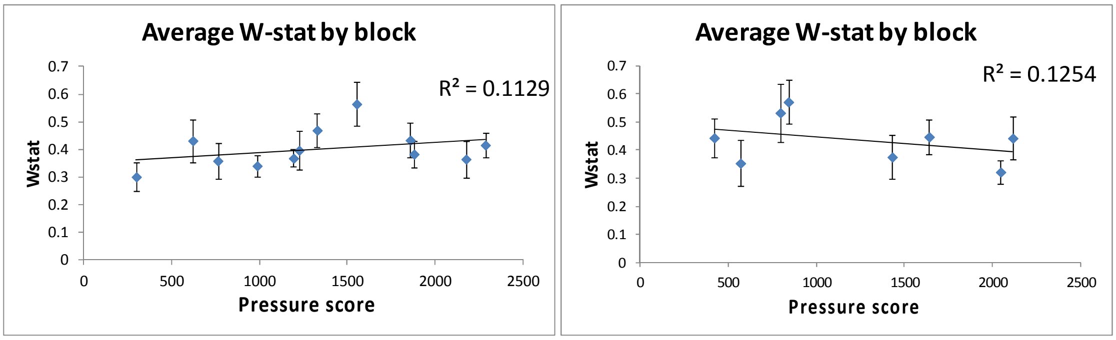


Figure 26. Linear regression plots based on values of the W statistic (block averages) for the Eastern English Channel (left) and Outer Thames Estuary (Right) communities.

3.8 Multivariate community analysis

MDS plots for all the infaunal abundance data were created and overlaid by site (Figure 27). There appears to be a moderate amount of clustering by site and the ordination has a high associated stress value. The high stress is likely a result of the large sample size.

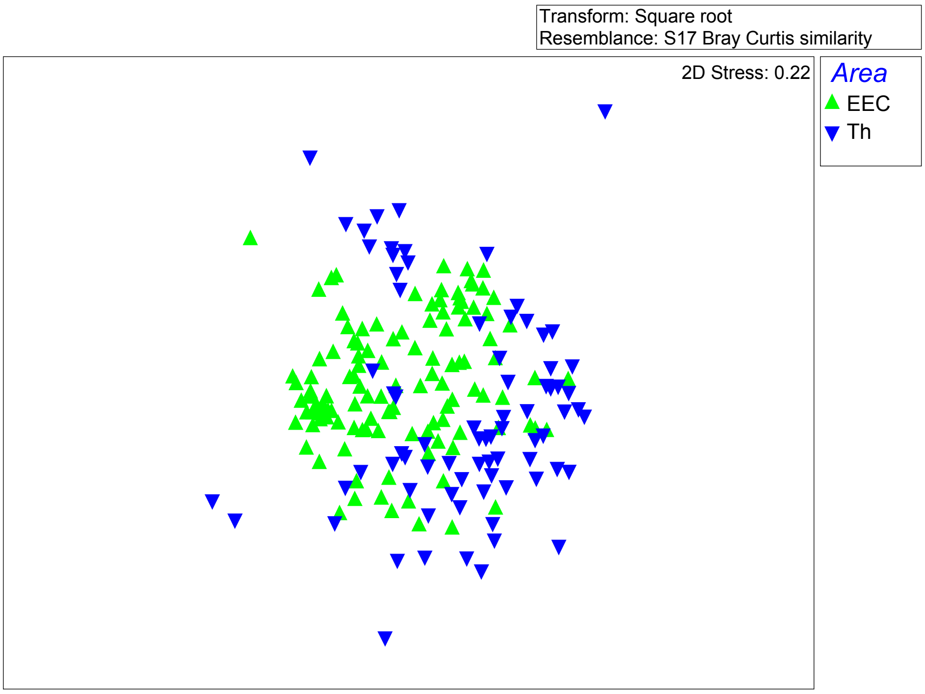


Figure 27. MDS plot for all infaunal community abundance stations and overlaid by site.

There is a much stronger differentiation between the epifaunal communities collected at Th and Eastern English Channel (Figure 28).

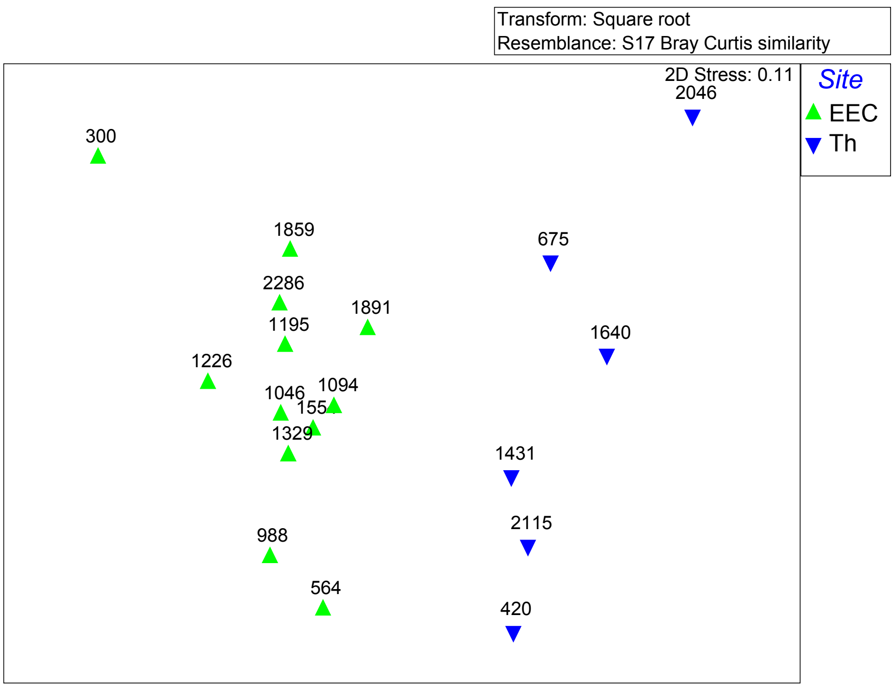


Figure 28. MDS plot for all epifaunal community biomass stations and overlaid by site and labelled by 0.05dd resolution pressure scores.

Figure 29 and Figure 30 show infaunal community abundance data at EEC and Th respectively and both are overlaid by Modified Folk classifications that are used in the EUNIS classification scheme. At EEC, stations group by sediment type, suggesting that this is driving community structure, compared to Th, where there is less clear distinction.

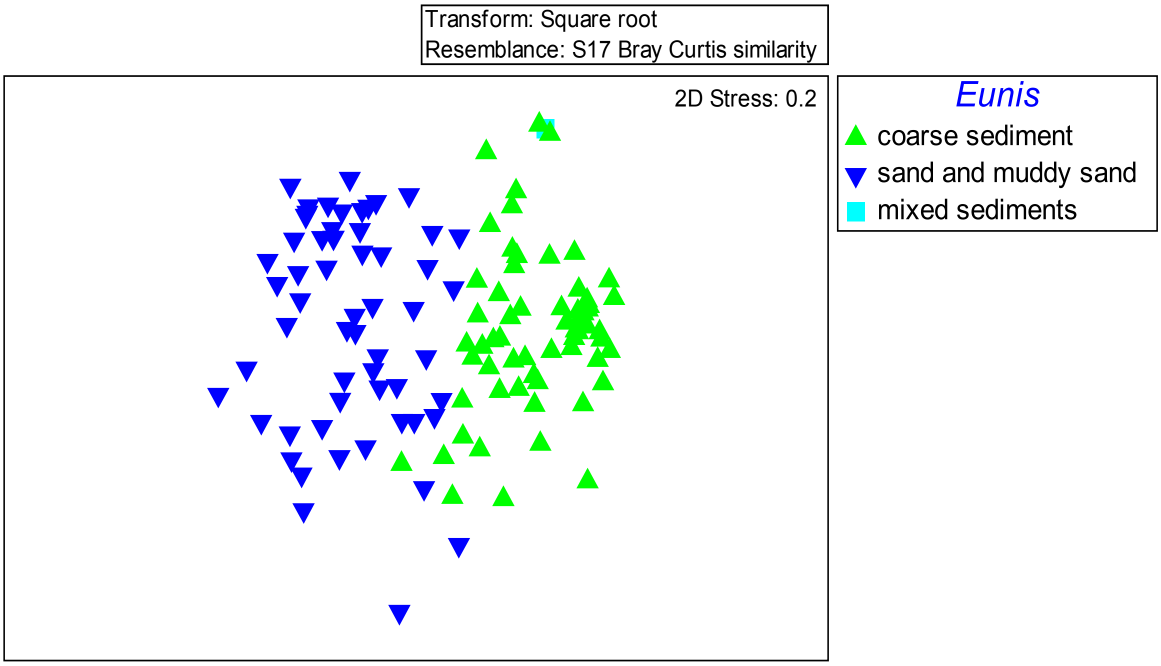


Figure 29. MDS plot at EEC for infaunal species abundance showing Modified Folk classification for each station.

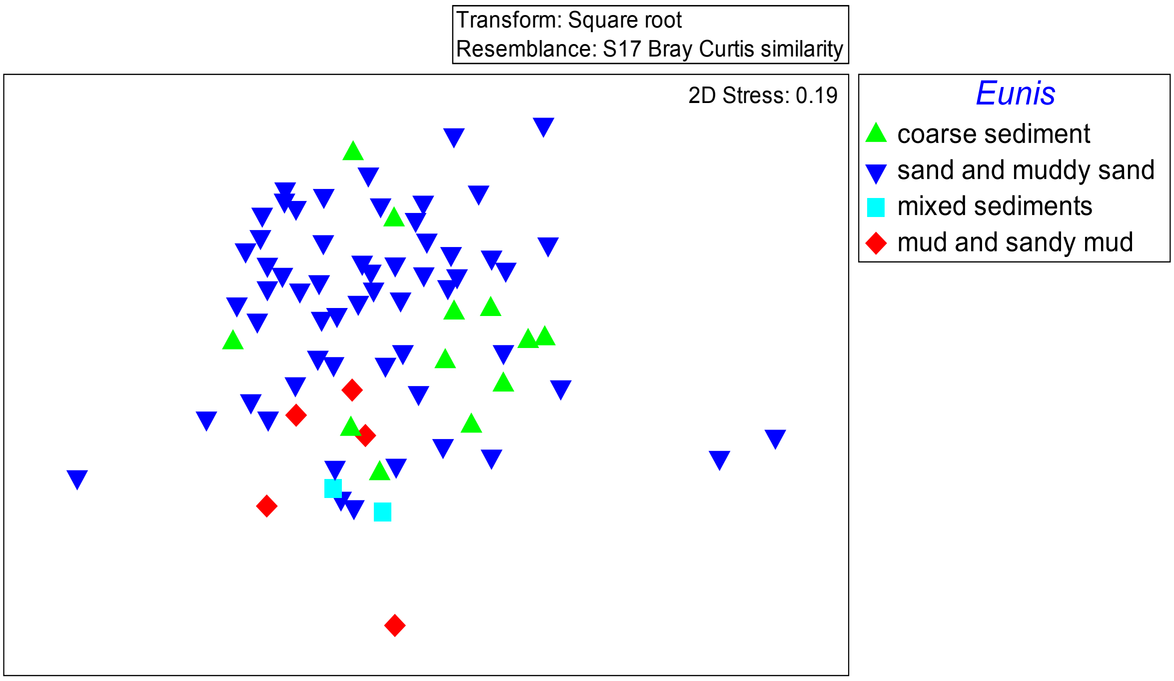


Figure 30. MDS plot at Th for infaunal species abundance showing Modified Folk classification for each station.

3.8.1 Fishing pressure scores and community structure

Eastern English Channel

There is no strong indication of a relationship between fishing pressure scores and community structure for the 0.05dd resolution grid (Figure 31). Even though there appears to be some grouping of stations based on the original abrasive fishing pressure scores, this not consistent across the identified gradient of pressure.

When VMS data are regridded, and new fishing pressure scores are overlaid on the community data, it can be seen that there is a tendency for communities associated with higher fishing pressure scores to be clustered more closely together. This could be indicative of a “Babushka effect” as described by Rabaut *et al* (2007). In this case, the disturbed community may be a subset of the undisturbed community rather than one composed of different species, which would create a separate discrete cluster instead.

Figure 32 shows abundance data overlaid with swept area scores calculated using 20 and 100 ping minima. In both cases, similar, but less strong, patterns were seen, as for the other fishing pressure scores.

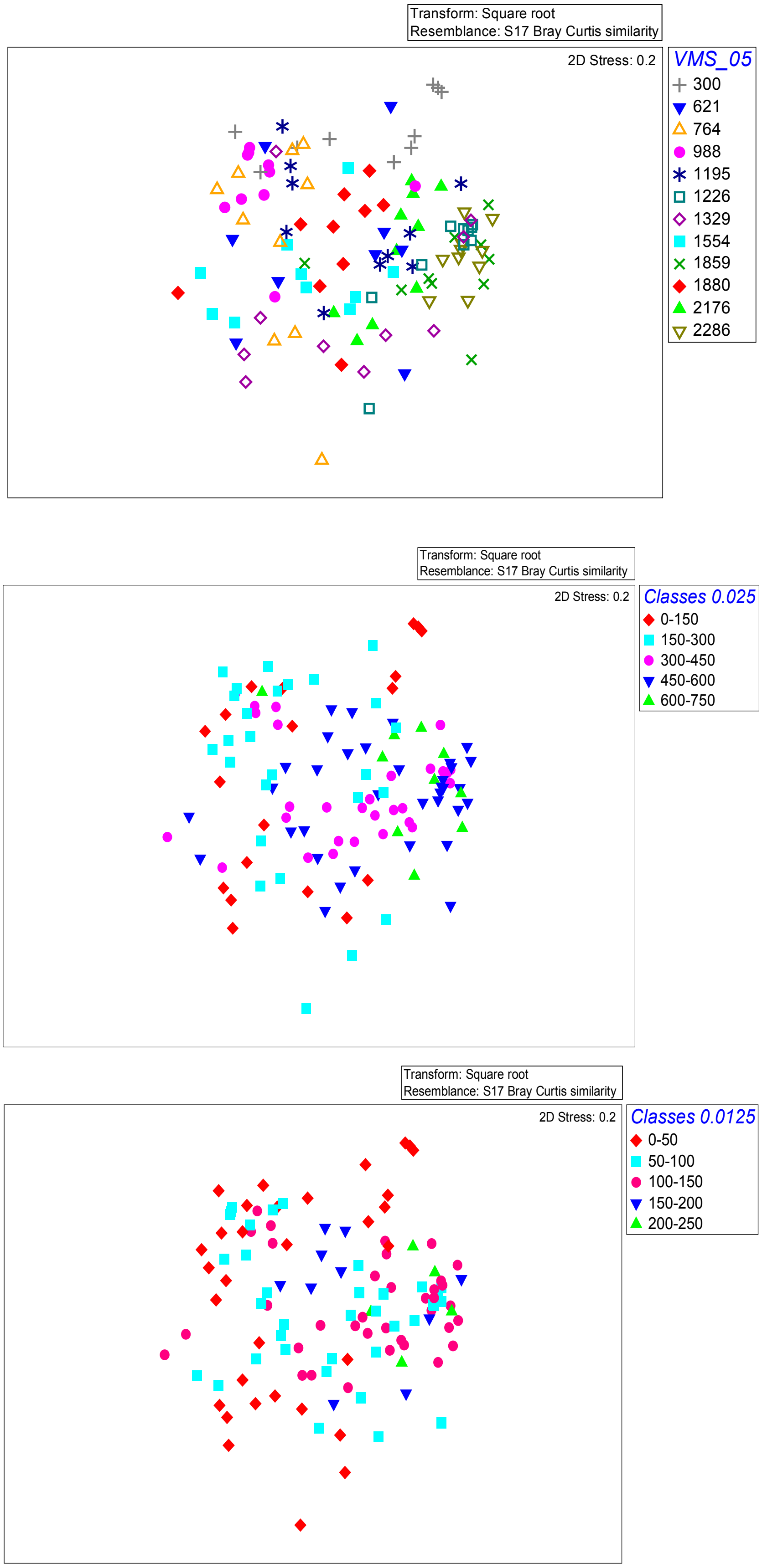
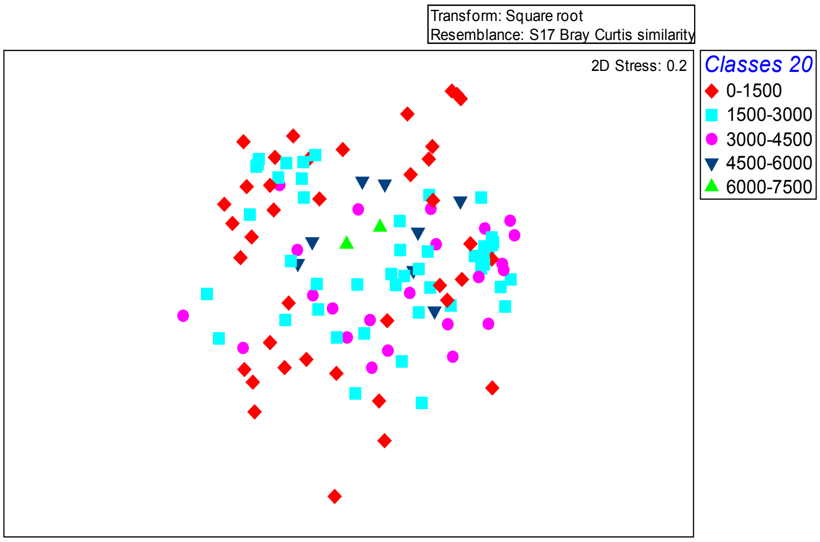


Figure 31. MDS plots for infaunal abundance data at EEC overlaid with classed pressure scores derived from a VMS resolution of 0.05dd, 0.025dd and 0.0125dd.



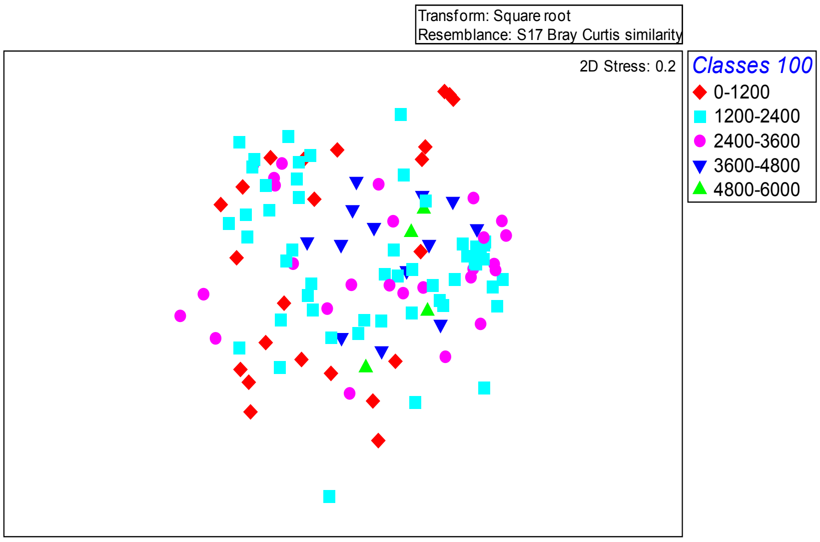


Figure 32. MDS plots for infaunal abundance data at EEC overlaid with classed pressure scores derived from nested regridding of VMS data based on ping minima of 20 (above) and 100 below.

Outer Thames

No grouping of stations with similar pressure scores, derived at a gridded resolution of 0.05dd, was observed at the Th site. However, when community data was overlaid with pressure scores derived from VMS gridded at 0.025dd, some loose clustering of stations with similar pressure scores was observed.

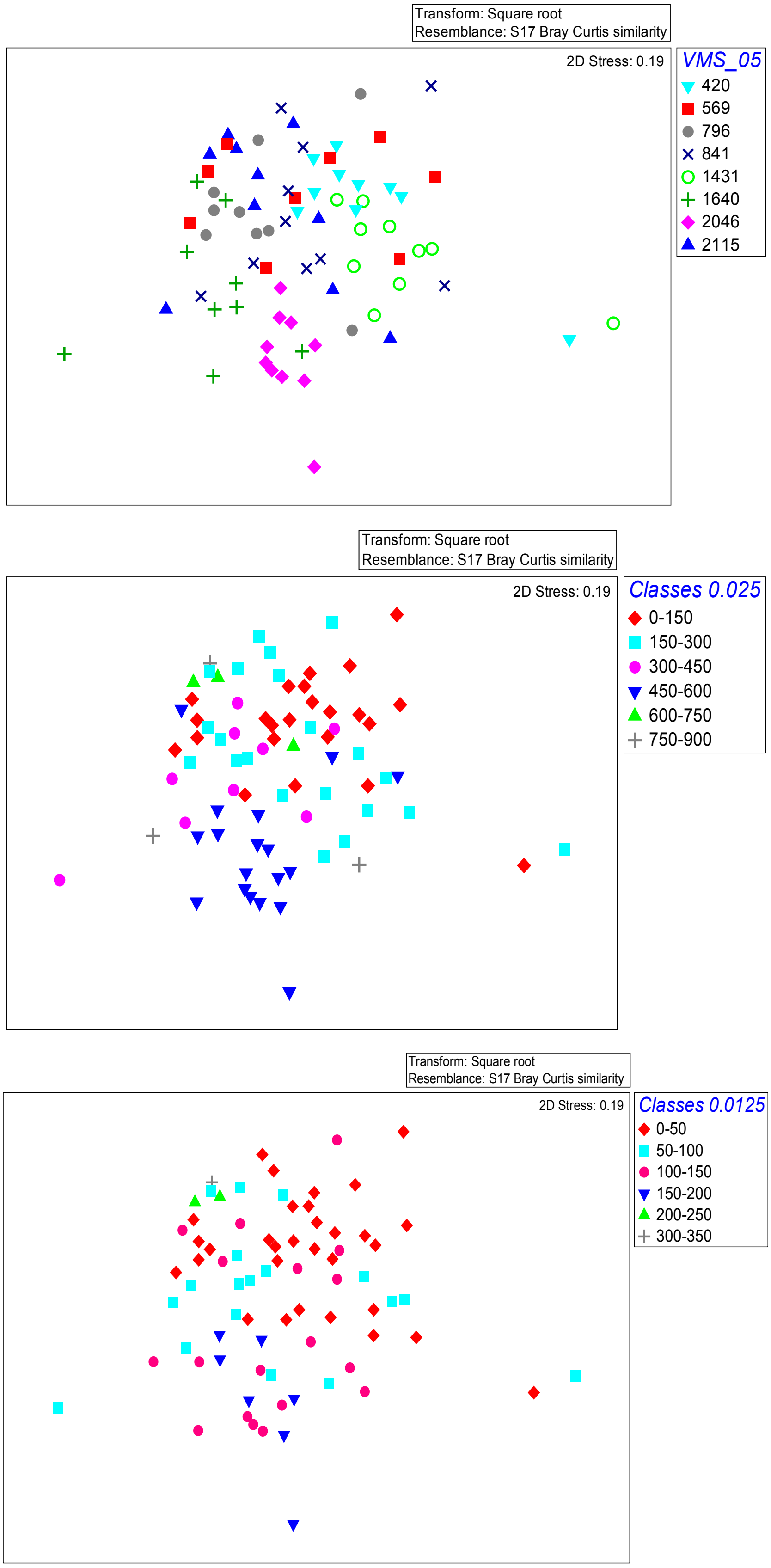


Figure 33. MDS plots for infaunal abundance data at Th overlaid with classed pressure scores derived from a VMS resolution of 0.05dd, 0.025dd and 0.0125dd.

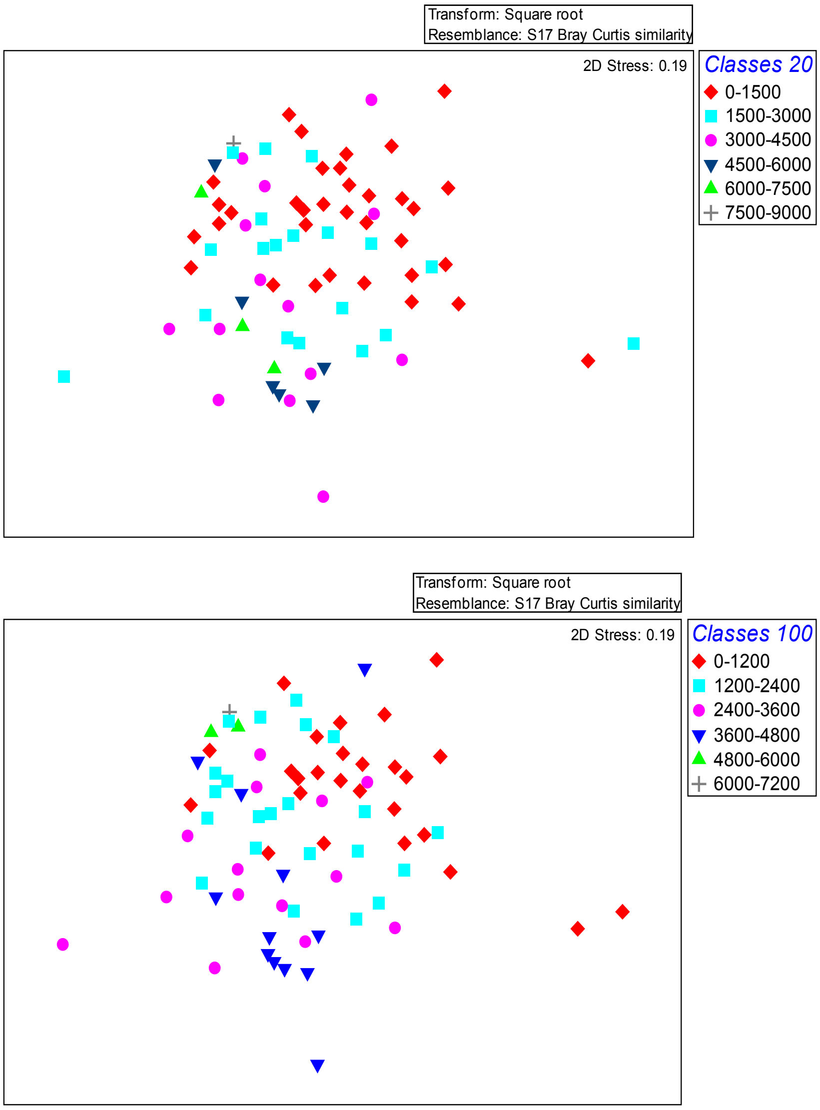


Figure 34. MDS plots for infaunal abundance data at Th overlaid with classed pressure scores derived from nested regridding of VMS data based on ping minima of 20 (above) and 100 below.

There were too few trawl stations at both EEC and Th sites to conclude anything of significance from the MDS ordinations performed.

3.8.2 Analysis of similarity (ANOSIM) between VMS groups

All global ANOSIM tests were found to be significant and therefore further interpretation was justified.

Figure 35 and Figure 36 present MDS plots for pairwise ANOSIM comparisons of abundance data at 0.05dd and 0.025dd grid resolutions for EEC. With neither 0.05dd nor 0.025dd gridding can any patterns in community structure be seen.

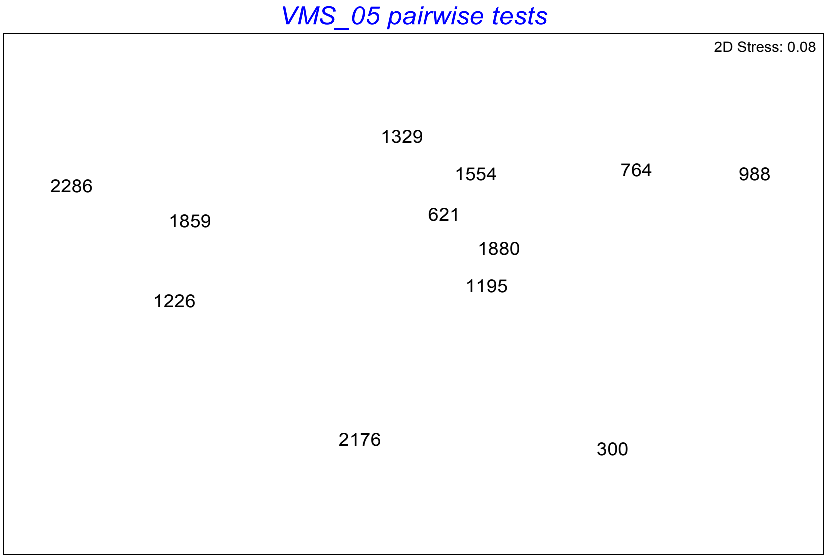


Figure 35. MDS plot for EEC R values for ANOSIM community abundance data between groups at a 0.05dd grid resolution.

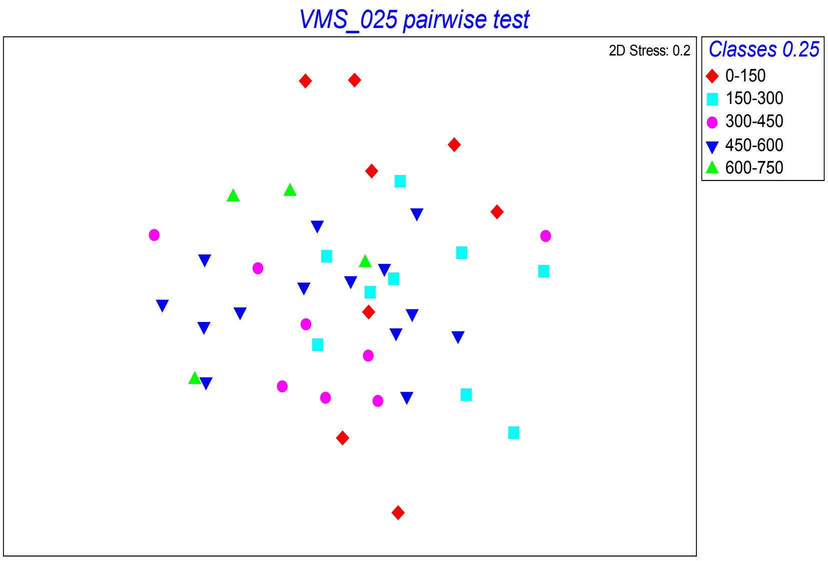


Figure 36. MDS plot for EEC R values for ANOSIM community abundance data between groups at a 0.025dd grid resolution, and overlaid with class designations.

Figure 37 and Figure 38 present the same data for the Th. When gridded at 0.05dd there are no clear patterns in the community data but, when regridded at 0.025dd, some clusters appear to form for both biomass and abundance.

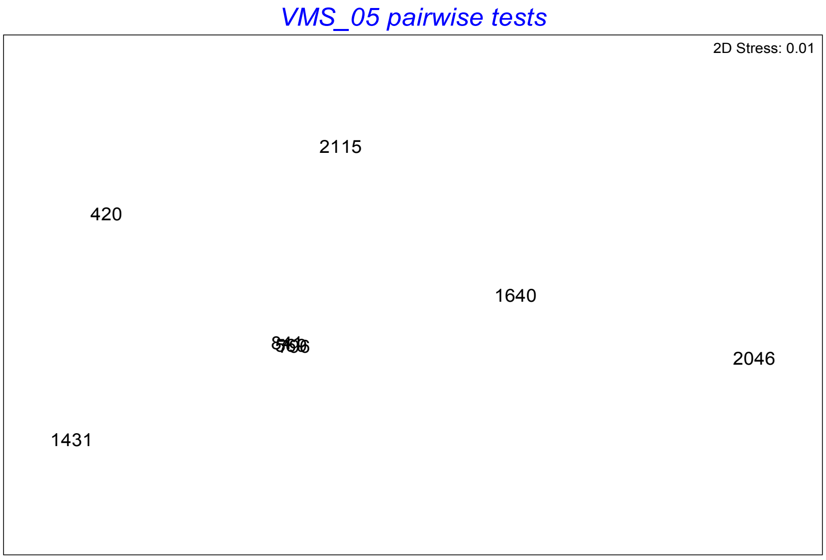


Figure 37. MDS plot for Th R values for ANOSIM community abundance data between groups at a 0.05dd grid resolution.

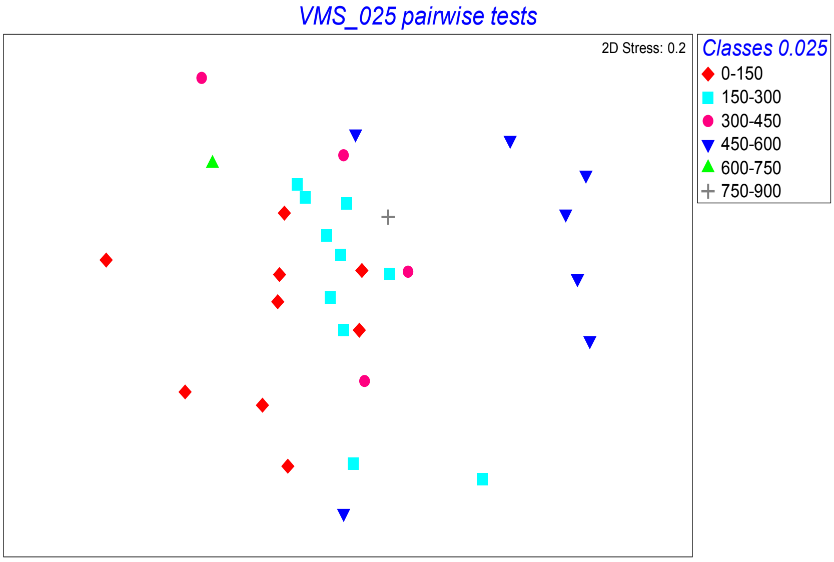


Figure 38. MDS plot for Th R values for ANOSIM community abundance data between groups at a 0.025dd grid resolution, and overlaid with class designations.

3.8.3 BEST analysis of environmental variables

Of the ten best single environmental factors explaining community variability, sand content (%) scored highly at both the EEC and the Th locations. Otherwise, none of the top five factors were the same for the two areas. For the EEC communities, gravel content (%) was highly significant and explained a good proportion of the observed variability in community structure. For the Th, none of our measured parameters performed as well, but distance from shore best explained the variability. In general, the patterns observed at the Eastern English Channel site were better explained by the environmental parameters measured.

Of the abrasive fishing pressure scores, VMS gridded at 0.0125dd and 0.05dd performed well at EEC, whilst VMS gridded at 0.025dd and nested at 100 pings scored better at Th.

BEST Analysis for EEC	***BEST Analysis for Th***
Variables	*Variables*
1 VMS_05	1 VMS_05
2 VMS_025	2 VMS_025
3 VMS_0125	3 VMS_0125
4 VMS_20	4 VMS_20
5 VMS_100	5 VMS_100
6 Depth	6 Depth
7 %Gravel	7 %Gravel
8 %Sand	8 %Sand
9 %Silt/clay	9 %Silt/clay
10 Dist From Shore (km)	10 Dist From Shore (km)
11 Organic carbon (% m/m)	11 Organic carbon (% m/m)
12 Nitrogen (% m/m)	12 Nitrogen (% m/m)

Best results (EEC)			*Best results* (Th)		
No.Vars	Corr.	Selections	No.Vars	Corr.	Selections
1	0.434	7	1	0.270	10
1	0.268	2	1	0.214	9
1	0.264	8	1	0.200	8
1	0.207	3	1	0.181	5
1	0.205	1	1	0.159	2
1	0.116	5	1	0.155	6
1	0.110	11	1	0.148	7
1	0.099	6	1	0.132	1
1	0.094	4	1	0.123	3
1	0.057	9	1	0.120	4

Figure 39. Primer output for BEST analysis for community abundance at EEC and Th with trial variable output limited to 1.

At EEC, when fishing pressure scores were included in the procedure, sediment gravel content (%) was still found to be an important factor but a more effective predictor when coupled with abrasion scores derived from VMS gridding of 0.025dd. In fact, of the top five models, four where found to include at least one fishing pressure score based on uniform gridding. The nested fishing pressure scores performed less well and were not present in any model. The degree of fit in general was better for species abundance data than for species biomass.

Best Analysis for EEC

Variables
1 VMS_05
2 VMS_025
3 VMS_0125
4 VMS_20
5 VMS_100
6 Depth
7 %Gravel
8 %Sand
9 %Silt/clay
10 Dist From Shore (km)
11 Organic carbon (% m/m)
12 Nitrogen (% m/m)

Best results

No.Vars	Corr.	Selections
2	0.472	2,7
2	0.451	3,7
1	0.434	7
3	0.434	2,3,7
3	0.429	1,2,7
3	0.429	2,6,7
3	0.422	1,3,7
2	0.417	1,7
3	0.415	3,6,7
4	0.413	2,3,6,7

Best Analysis for EEC excluding VMS

Variables
1 Depth
2 %Gravel
3 %Sand
4 %Silt/clay
5 Dist From Shore (km)
6 Organic carbon (% m/m)
7 Nitrogen (% m/m)

Best results

No.Vars	Corr.	Selections
1	0.434	2
2	0.394	1,2
2	0.365	2,3
3	0.365	1-3
3	0.357	1,2,4
2	0.352	2,4
2	0.350	2,6
3	0.348	1,2,6
4	0.342	1-4
2	0.333	2,5

Figure 40. BEST analysis for community abundance at EEC with (left) and without (right) fishing pressure scores.

BEST analysis for Thames abundance and biomass data, when run with pressure scores included, both demonstrated a best fit with five variables: nested 100 scores, depth, sediment silt content (%), distance from shore, and sediment nitrogen content (%). This shows that no single environmental factor is driving the community structure at the Thames site and that a large number of variables must be looked at to try and interpret the community variability. Selection of a large number of variables suggests that actually the best driver of community structure has not been discovered and included in the model.

Best Analysis for Th

Variables
1 VMS_05
2 VMS_025
3 VMS_0125
4 VMS_20
5 VMS_100
6 Depth
7 %Gravel
8 %Sand
9 %Silt/clay
10 Dist From Shore (km)
11 Organic carbon (% m/m)
12 Nitrogen (% m/m)

Best results

No.Vars	Corr.	Selections
5	0.311	5,6,9,10,12
5	0.307	2,6,9,10,12
4	0.304	5,6,9,10
5	0.304	3,6,9,10,12
5	0.303	1,6,9,10,12
5	0.303	5,6,9-11
4	0.303	5,6,10,11
5	0.302	4,6,9,10,12
4	0.299	4,6,10,11
5	0.299	5,6,10-12

Best Analysis for Th excluding VMS

Variables
1 Depth
2 %Gravel
3 %Sand
4 %Silt/clay
5 Dist From Shore (km)
6 Organic carbon (% m/m)
7 Nitrogen (% m/m)

Best results

No.Vars	Corr.	Selections
4	0.299	1,4,5,7
4	0.294	1,4-6
3	0.293	1,4,5
5	0.293	1,2,4,5,7
5	0.292	1,4-7

Figure 41. BEST analysis for community abundance at Th with (left) and without (right) fishing pressure scores.

3.9 Traits analysis

3.9.1 Infaunal relative trait abundance

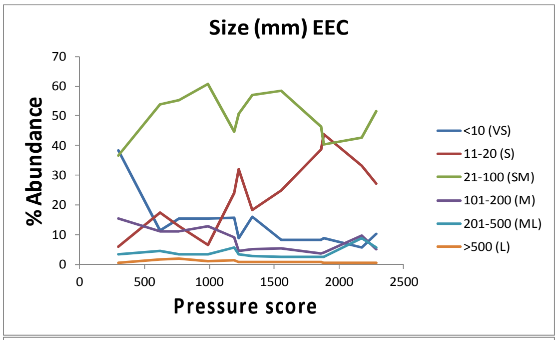
In general, the infaunal community at both sites was dominated by free-living species, which deposit feed and promote diffusive mixing within the sediment (Figure 42). However, the infaunal dominance is likely to be an artefact of the sampling method.

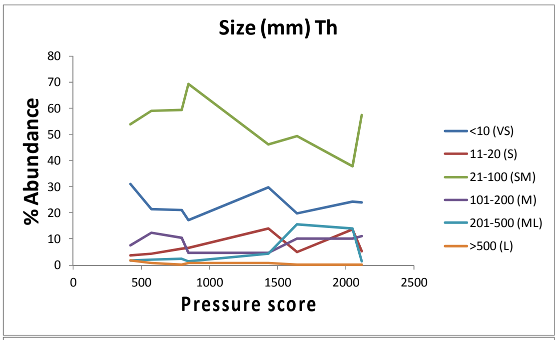
Traits of longevity (1-10 years) and larval development (planktotrophic) are comparable at both sites and correspond with what has previously been reported within the Greater North Sea subregion.

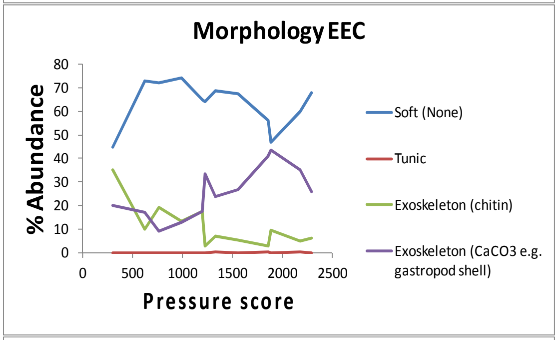
A gradual increase in the 11-20mm size group proportionally was observed at EEC and a slight decrease in the 21-100mm size class at Th, demonstrating a trend towards smaller size species as fishing activity increased. An increase in the proportion of species with exoskeletons (gastropod shells) was observed at EEC, unlike at the Th site which was dominated by soft bodied organisms. An increase in burrow dwellers was observed within the Th site in relation to an increase in fishing activity, compared to the EEC site, which was dominated by free-living species.

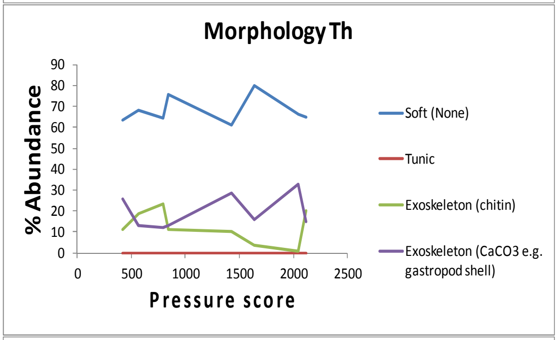
A general decline in species which brood their eggs and an increase in species that shed their eggs was observed at EEC in relation to increased fishing activity, while Th was consistently dominated by species that shed their eggs.

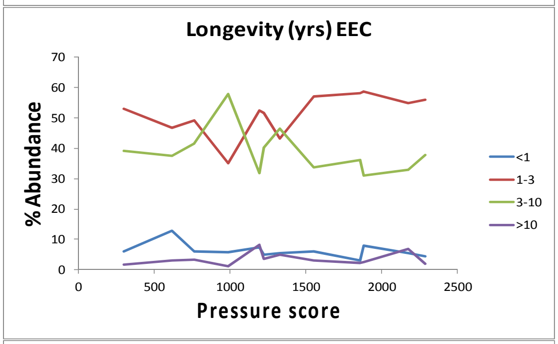
Sediment position was primarily found to be 0-5mm at both sites. Subsurface deposit feeders were observed to increase as fishing activity increased within Th. This correlates well with the observed increase in burrow-dwelling also observed at this site.

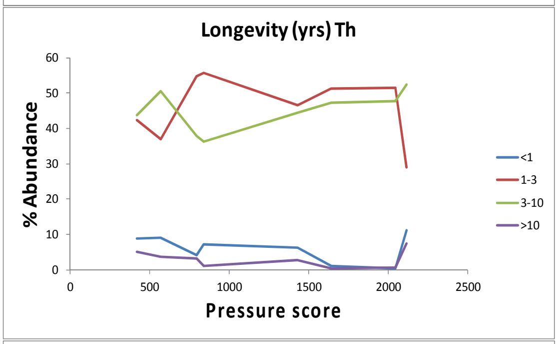

Size (mm) EEC
% Abundance
Pressure score
0
10
20
30
40
50
60
70
0
500
1000
1500
2000
2500
<10 (VS)
11-20 (S)
21-100 (SM)
101-200 (M)
201-500 (ML)
>500 (L)

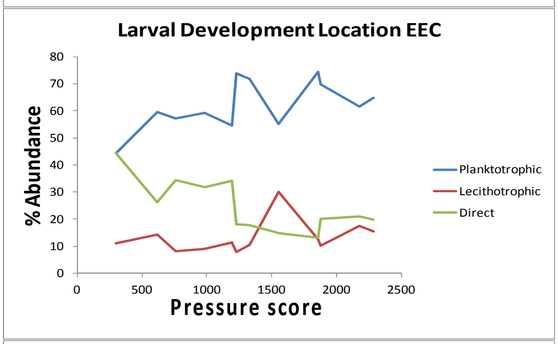

Size (mm) Th
% Abundance
Pressure score
0
10
20
30
40
50
60
70
80
0
500
1000
1500
2000
2500
<10 (VS)
11-20 (S)
21-100 (SM)
101-200 (M)
201-500 (ML)
>500 (L)

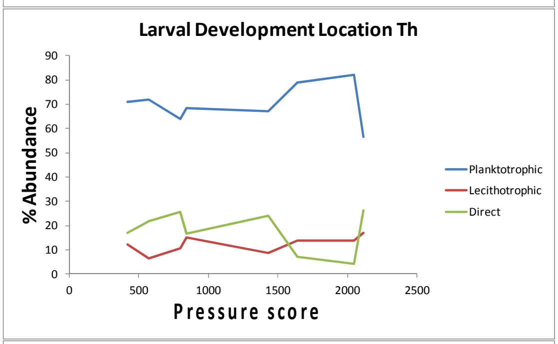

Morphology EEC
% Abundance
Pressure score
0
10
20
30
40
50
60
70
80
0
500
1000
1500
2000
2500
Soft (None)
Tunic
Exoskeleton (chitin)
Exoskeleton (CaCO3 e.g. gastropod shell)

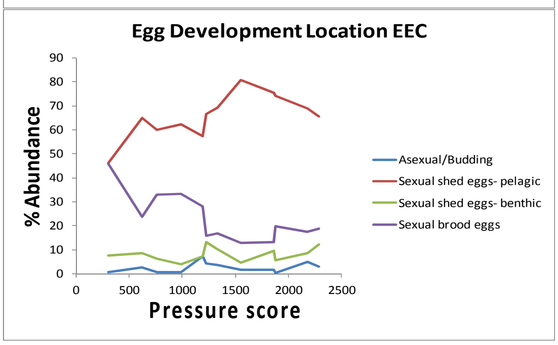

Morphology Th
% Abundance
Pressure score
0
10
20
30
40
50
60
70
80
90
0
500
1000
1500
2000
2500
Soft (None)
Tunic
Exoskeleton (chitin)
Exoskeleton (CaCO3 e.g. gastropod shell)

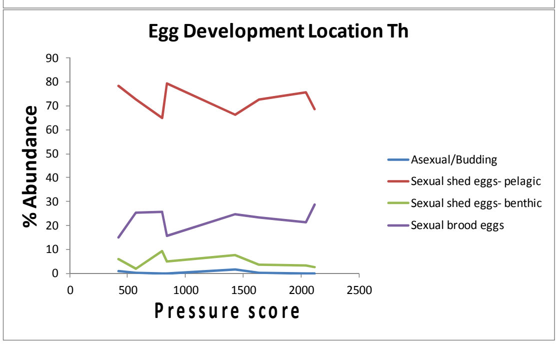

Longevity (yrs) EEC
% Abundance
Pressure score
0
10
20
30
40
50
60
70
0
500
1000
1500
2000
2500
<1
1-3
3-10
>10


Longevity (yrs) Th
% Abundance
Pressure score
0
10
20
30
40
50
60
0
500
1000
1500
2000
2500
<1
1-3
3-10
>10


Larval Development Location EEC
% Abundance
Pressure score
0
10
20
30
40
50
60
70
80
0
500
1000
1500
2000
2500
Planktotrophic
Lecithotrophic
Direct


Larval Development Location Th
% Abundance
Pressure score
0
10
20
30
40
50
60
70
80
90
0
500
1000
1500
2000
2500
Planktotrophic
Lecithotrophic
Direct


Egg Development Location EEC
% Abundance
Pressure score
0
10
20
30
40
50
60
70
80
90
0
500
1000
1500
2000
2500
Asexual/Budding
Sexual shed eggs- pelagic
Sexual shed eggs- benthic
Sexual brood eggs


Egg Development Location Th
% Abundance
Pressure score
0
10
20
30
40
50
60
70
80
90
0
500
1000
1500
2000
2500
Asexual/Budding
Sexual shed eggs- pelagic
Sexual shed eggs- benthic
Sexual brood eggs

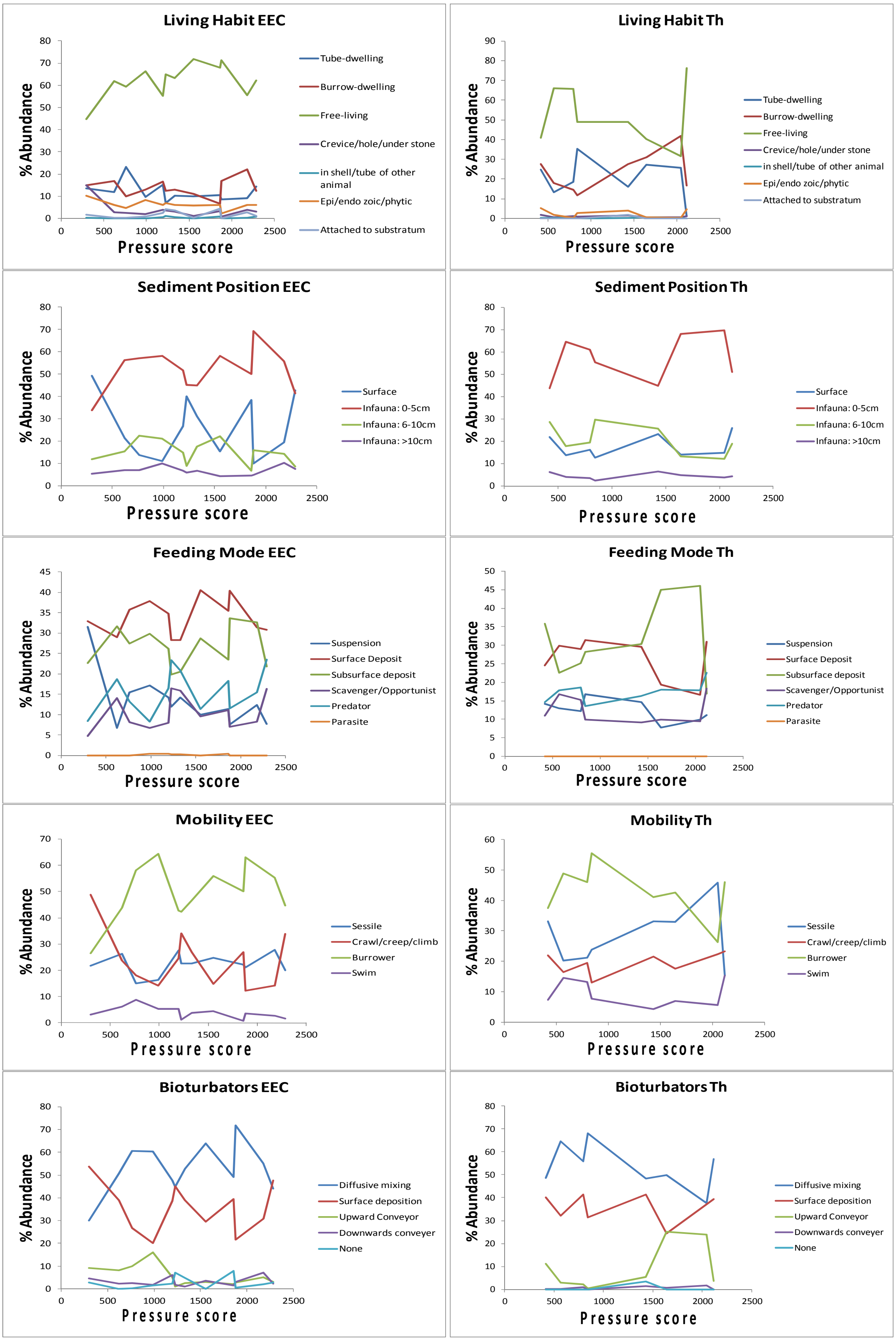


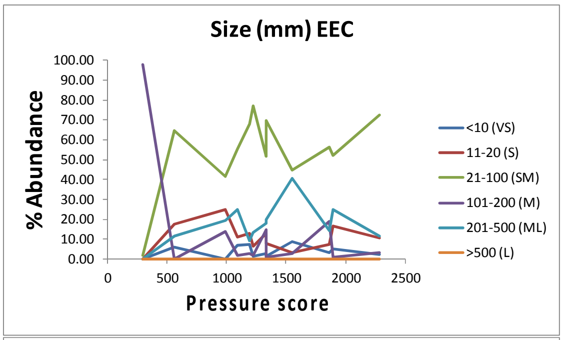
Figure 42. Proportional abundance line plots for traits: size, morphology, longevity, larval development location, egg development location, living habit, sediment position, feeding mode, mobility and bioturbation. Plots are for EEC (left) and Th (right) infaunal % abundance against increasing pressure scores derived from a VMS aggregated grid of 0.05dd.

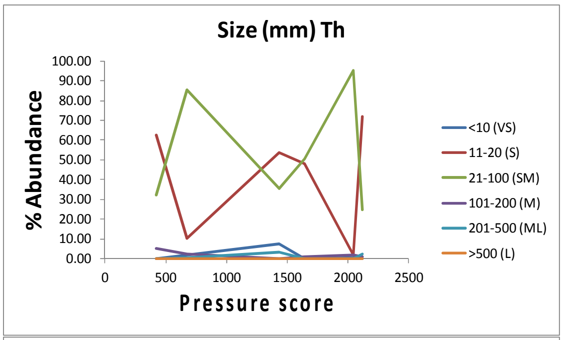
3.9.2 Epifaunal average trait abundance

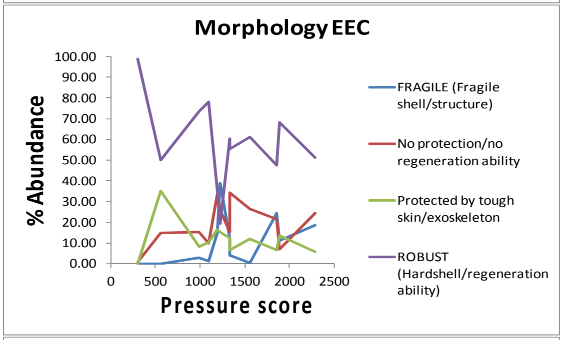
A greater range of size classes was observed at EEC when compared with Th, though both sites were dominated by species within the 21-100 mm size class. Both sites were dominated by robust species, as would be expected due to the high levels of natural disturbance documented at the two sites (Diesing *et al* 2013). Similarly, both sites were dominated by species within the 3-10 years longevity class and planktotrophic larval development class.

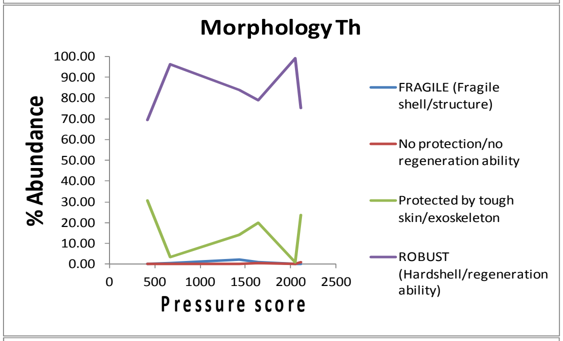
An increase in species which shed their eggs (pelagic) was observed at the EEC site as fishing activity increased. Th was dominated by species which both brood their eggs at mid levels of fishing activity and those that shed their eggs (pelagic) at higher levels of fishing activity.

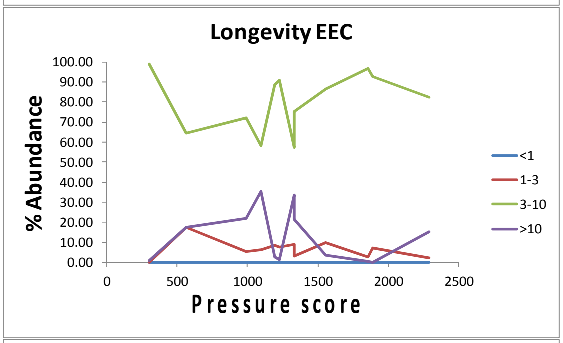
EEC and Th were both found to be dominated by free living species and species found within shells and tubes of other species. The dominant feeding type at both sites was predominantly predators and scavengers with no relationship observed with increased levels of fishing activity. Similarly both sites were dominated by species which exhibited low mobility (crawlers) and, again, no relationship with fishing activity was observed.

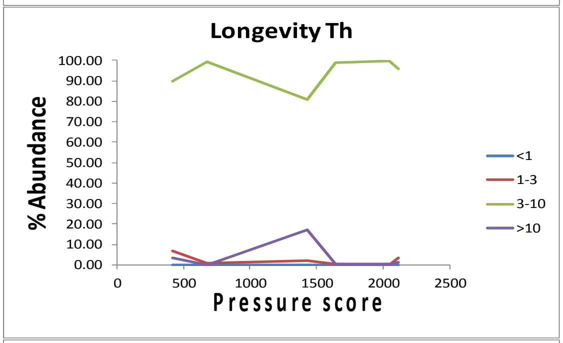

Size (mm) EEC
% Abundance
Pressure score
<10 (VS)
11-20 (S)
21-100 (SM)
101-200 (M)
201-500 (ML)
>500 (L)

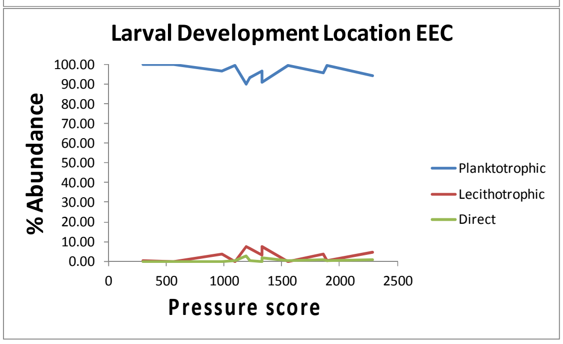

Size (mm) Th
% Abundance
Pressure score
<10 (VS)
11-20 (S)
21-100 (SM)
101-200 (M)
201-500 (ML)
>500 (L)

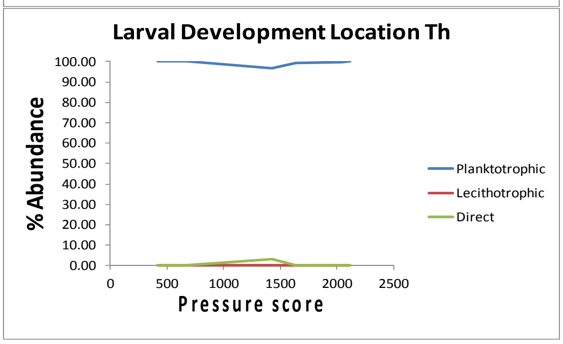

Morphology EEC
% Abundance
Pressure score
FRAGILE (Fragile shell/structure)
No protection/no regeneration ability
Protected by tough skin/exoskeleton
ROBUST (Hardshell/regeneration ability)


Morphology Th
% Abundance
Pressure score
FRAGILE (Fragile shell/structure)
No protection/no regeneration ability
Protected by tough skin/exoskeleton
ROBUST (Hardshell/regeneration ability)


Longevity EEC
% Abundance
Pressure score
<1
1-3
3-10
>10


Longevity Th
% Abundance
Pressure score
<1
1-3
3-10
>10


Larval Development Location EEC
% Abundance
Pressure score
Planktotrophic
Lecithotrophic
Direct


Larval Development Location Th
% Abundance
Pressure score
Planktotrophic
Lecithotrophic
Direct

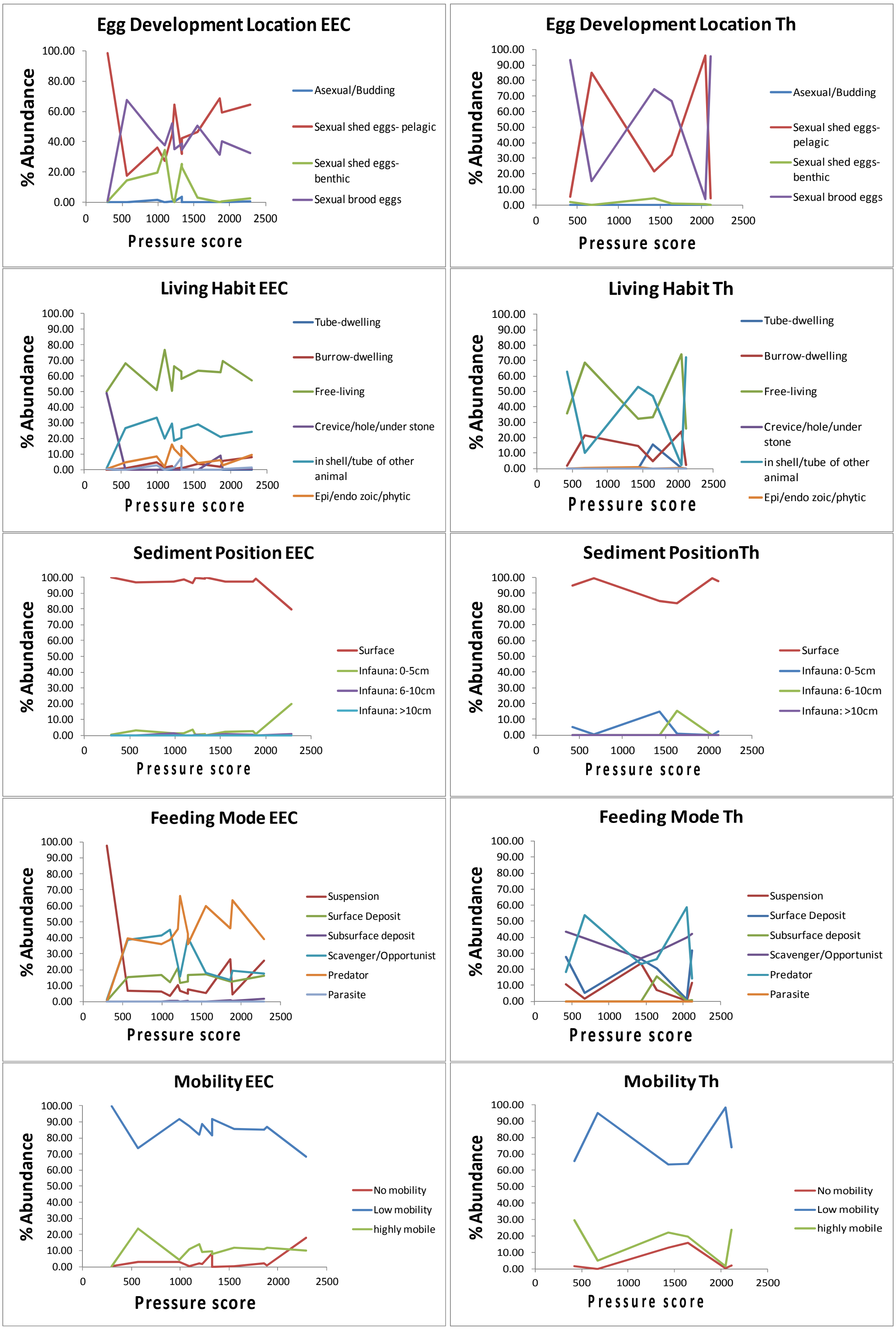


Figure 43. Proportional abundance line plots for traits; Size, Morphology, Longevity, Larval development location, Egg development location, Living habit, Sediment position, Feeding mode and Mobility. Plots are for EEC (left) and Th (right) epifaunal % abundance against increasing pressure scores derived from a VMS aggregated grid of 0.05dd.

3.9.3 Multivariate analysis of community traits

For the trait data, the most conspicuous patterns in MDS plots appear when data are grouped by EUNIS sediment class. At EEC, mixed sediment communities are separated, but coarse and sand and muddy sand largely overlap (Figure 44). At Th, mixed sediments, coarse sediment, and mud and sandy mud were found to cluster together, within the range of sand and muddy sand (Figure 45).

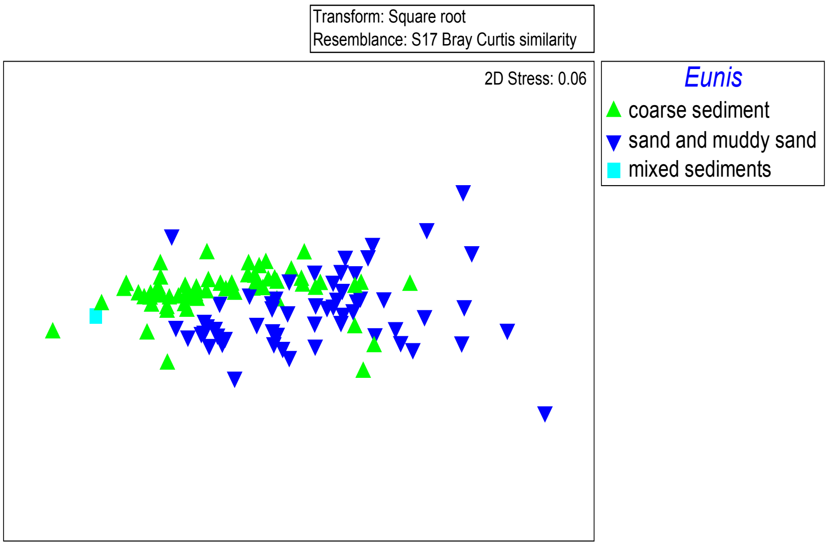


Figure 44. MDS plot for community trait abundance data at EEC overlaid with EUNIS sediment classes.

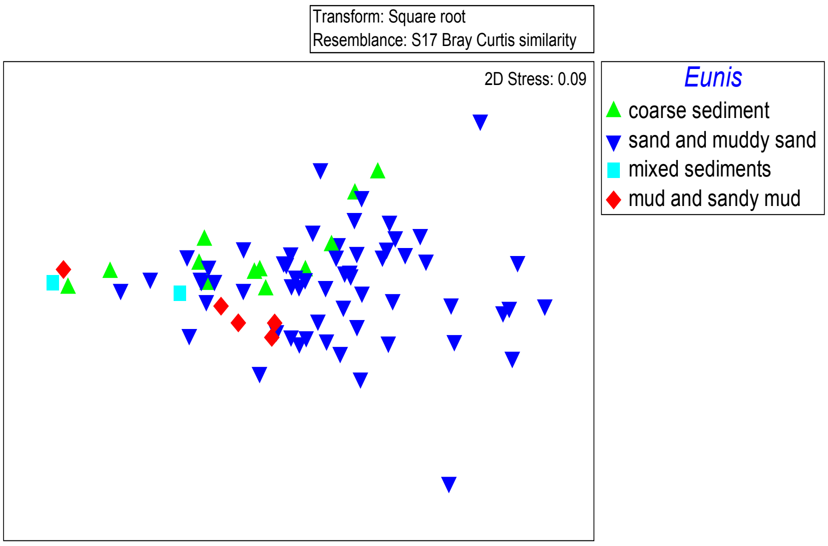


Figure 45. MDS plot for community trait abundance data at Th overlaid with EUNIS sediment classes.

There were no obvious patterns at EEC with fishing pressure scores overlain. However, at the Th site, stations clustered together. There was no clear trend with abrasion score, but it is thought that an additional factor not measured here may have been driving differences in community structure within the region.

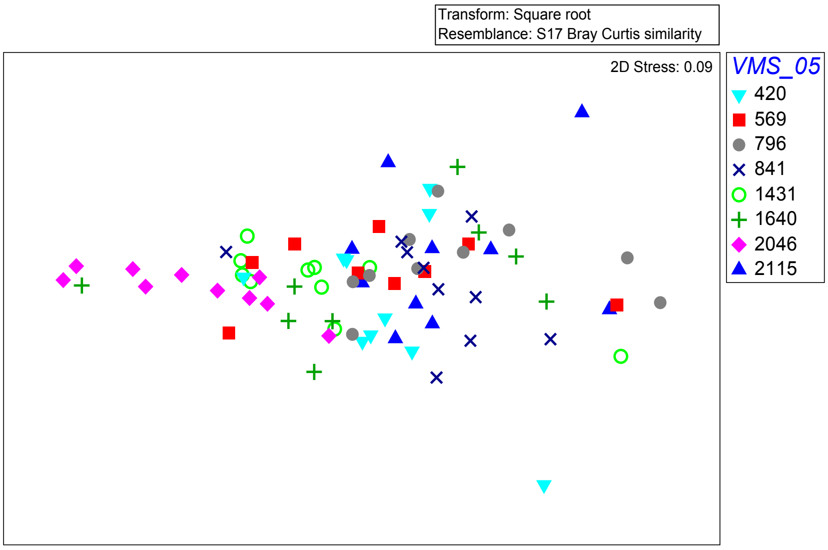


Figure 46. MDS for community trait abundance data at Th with a VMS grid resolution of 0.05dd overlaid.

ANOSIM testing of trait data, grouped by fishing abrasion pressure scores per station derived from a VMS grid resolution of 0.05dd, showed no clear R value similarities between groups with a similar degree of abrasive fishing pressure. R value similarities (derived from pairwise testing) at both EEC and Th were not explained by the derived fishing pressure scores used here.

4 Discussion

4.1 Distribution and presentation of fishing effort

An important consideration when assessing the different ways of processing VMS data is the presentational aspect. Often VMS data are presented on maps in order to demonstrate the spatial distribution and intensity of fishing effort on the seabed. As discussed below, the distribution of fishing activity can appear very different depending on how the data are processed and visualised. It is thus important to determine what resolution is most appropriate for representation of the VMS data.

4.1.1 Regular grids

There are some clear differences in the fishing layers when the VMS data were aggregated on 0.05dd, 0.025dd and 0.0125dd grids. Some of the trends mentioned below have already been well described in the literature, for example by Lambert *et al* (2012) and Lee *et al* (2010b), but are discussed here in terms of applied marine conservation.

1. *At the smallest cell size there is a risk that the spatial footprint of abrasive fishing pressure is underestimated.* However, this is most likely to be localised to areas of very low fishing effort where there are few vessels and correspondingly results in higher resolution information on the relative distribution of fishing activity. The slowest vessels within our speed filter can travel over 3.5km in the 2 hour period between pings.
2. *At the largest grid size there is a risk that the spatial footprint is overestimated.* This is obviously a trade-off with 1) and is particularly likely to be the case for very large cell sizes. At Th, as the fishing pressure cell size decreases, gaps appear in the pressure layer representing areas that are not fished (Figure 47). Looking at the patterns of pressure around the gaps suggests that this is likely to represent genuinely unfished areas rather than to be an artefact of the processing.
3. *Distribution of abrasive fishing pressure within a raster grid cell.* Due to the nature of aggregating the point data into a grid, an assumption has to be made that the fishing effort is distributed evenly over the cell. Pressure scores derived from these data are therefore attributed to all of the sample stations that fall within that cell. In order to try and account for this uneven distribution of activity within each original-sized cell, we used ten sampling stations per cell in order to try and sample the range of variability. When smaller grid cells are nested within the original size (0.05dd) it can be seen that the fishing effort in some of the cells is not distributed homogeneously. For instance, the central bottom cell in the EEC at the 0.05dd cell size is of intermediate intensity relative to the other cells, but, as the cell size decreases, it can be seen that the effort is reattributed into the bottom half of the larger cell (Figure 6).
4. *Differences between grid sizes are more apparent at smaller scales.* For regional maps presented below (Figure 47 and Figure 48), the same broad areas of high and low fishing effort are visible regardless of the processing method used. Maps presented at what could be considered a marine protected site level (Figure 6) show that the patchiness of fishing activity becomes more apparent as the cell size decreases.
5. *Differences in VMS presentation vary with region and potentially fishing type.* The different responses of the fishing pressure layers to changing grid size at EEC and Th indicate that the most appropriate processing method is context dependent. EEC appears more robust to changes in processing method, which might indicate that fishing effort is spread more homogeneously across the region. This in turn might be a reflection of the predominant gear type used in the region or behaviour of a given fleet. If a particular gear is more likely to be towed in a homogeneous “lawnmower”

fashion then the resulting VMS data will be more robust to the effect of the method of processing employed.

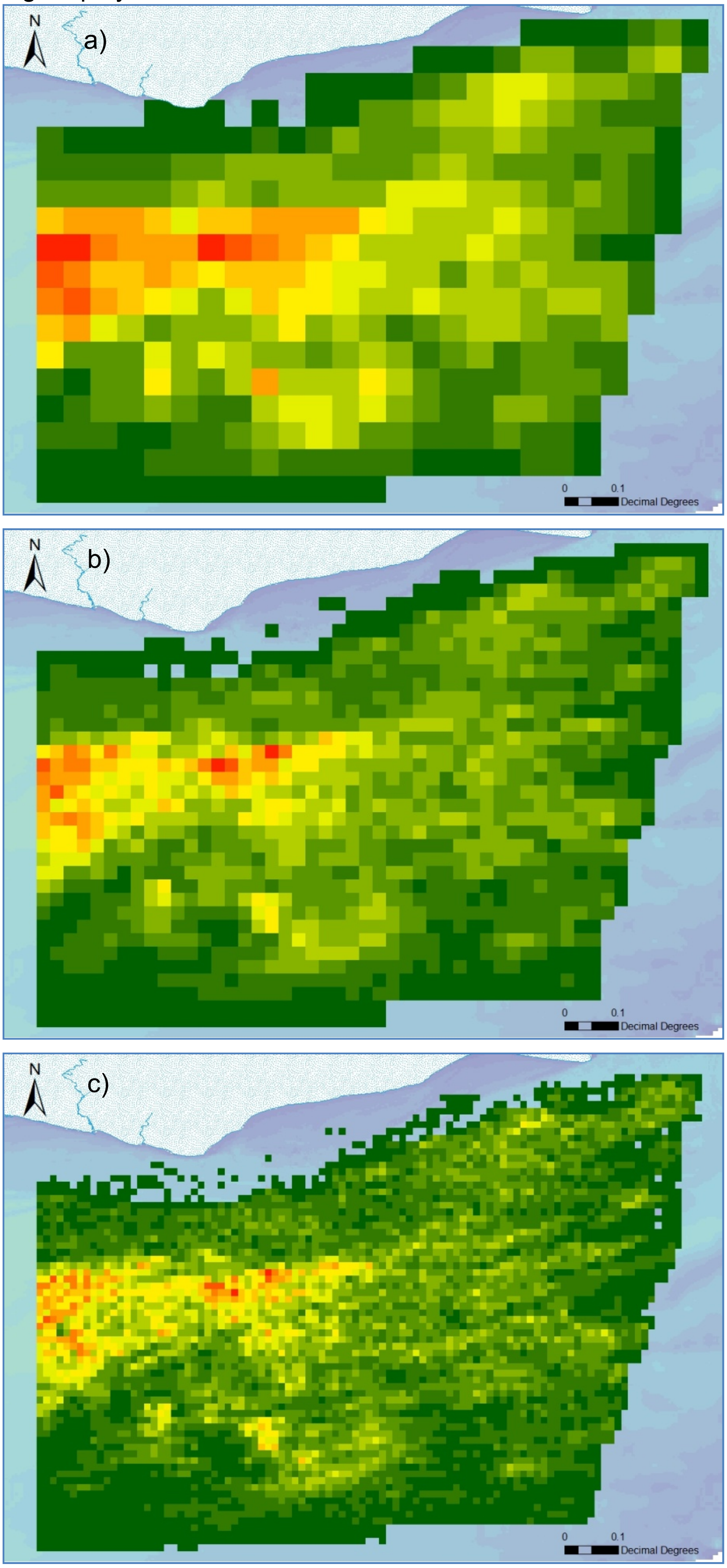


Figure 47. VMS data from the Eastern English Channel aggregated at a) 0.05dd, b) 0.025dd and c) 0.0125dd grids. Relative pressure scores are coloured red for high and green for low. Gaps indicate pressure scores of zero.

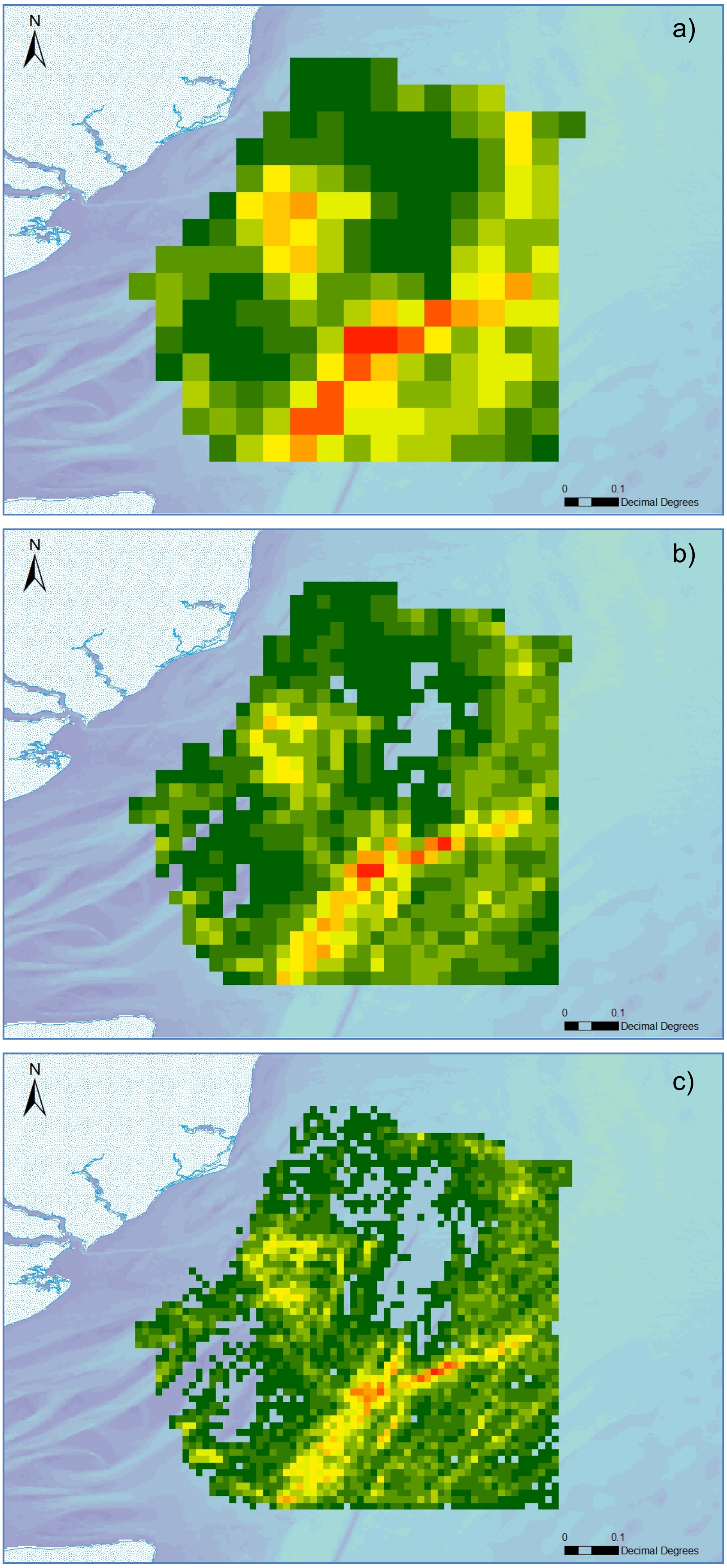


Figure 48. VMS data from the Outer Thames Estuary aggregated at a) 0.05dd, b) 0.025dd and c) 0.0125dd grids. Relative pressure scores are coloured red for high and green for low. Gaps indicate pressure scores of zero.

4.1.2 Nested grids

In terms of presentation, our modified Gerritsen (2013) method did not offer any significant advantages over the simple uniform grids.

When 20 pings were used to trigger cell division, the final cells were analogous to those of the 0.0125dd grid used in the study (Figure 49, Figure 50 and Figure 51). As such, the fidelity to the VMS point data in heavily fished areas is high. However, there does appear to be some evidence of under representation of fishing effort, although not as much as the 0.0125dd gridded data. Using 100 pings as the cell division trigger resulted in a larger minimum cell size and fewer inter-ping cells being assigned a fishing abrasion pressure score of 0. However, this approach may suffer from the same overestimation issue afflicting lower resolution standard grids.

The nested grids display characteristics of the three regular grid layers used in this study. In areas of sparse VMS data, there remains the risk of overestimation of fishing effort. The nested grids exhibit a low spatial resolution within these areas. However, in areas of high fishing intensity, the increasingly finer grid resolution captures a high degree of spatial information on the distribution of pings. These regions of the grid closely model the raw VMS data.

An advantage of using nested grids, rather than regular grids of an arbitrarily selected cell size, is that the nesting algorithm results in cell sizes appropriate for the density of VMS points in the underlying data. This effectively gives you the ‘best of both worlds’ of gridding to large and small grid sizes. The nested grid approach also provides absolute estimates of fishing activity in the form of swept areas. Conversely, the standard gridding approach only yields a relative fishing activity score at present.

One potential problem, which is likely to be an artefact of the method we modified, and which should be manageable in the future, is the absence of “no fishing” areas in the nested grid layers.

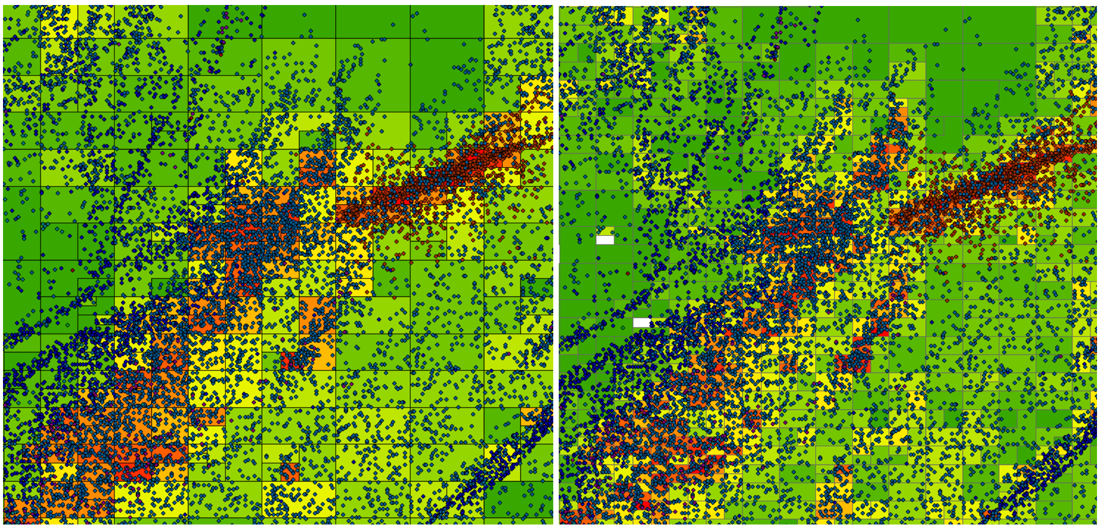
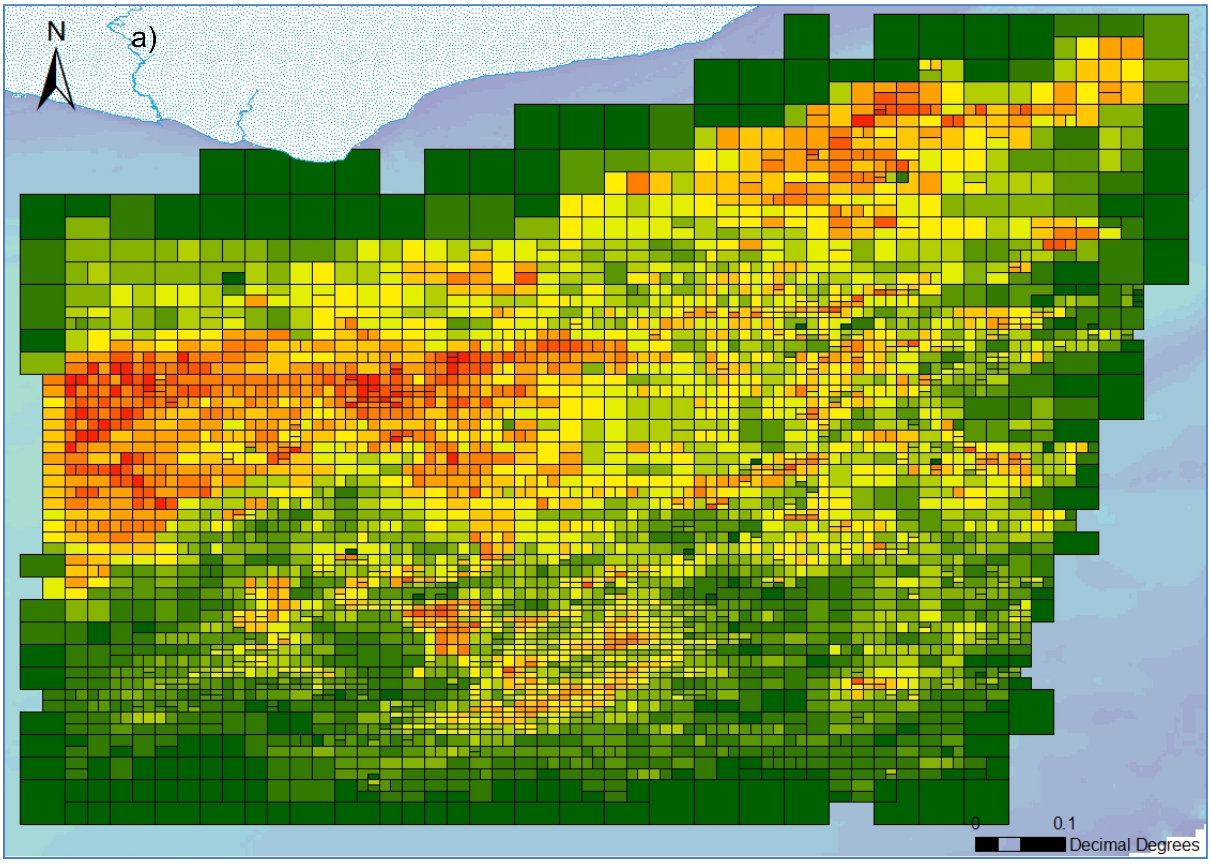


Figure 49. Placeholder figure showing nested grids (100 and 20) vs. VMS point data. Relative pressure scores are coloured red for high and green for low. Gaps indicate pressure scores of zero.



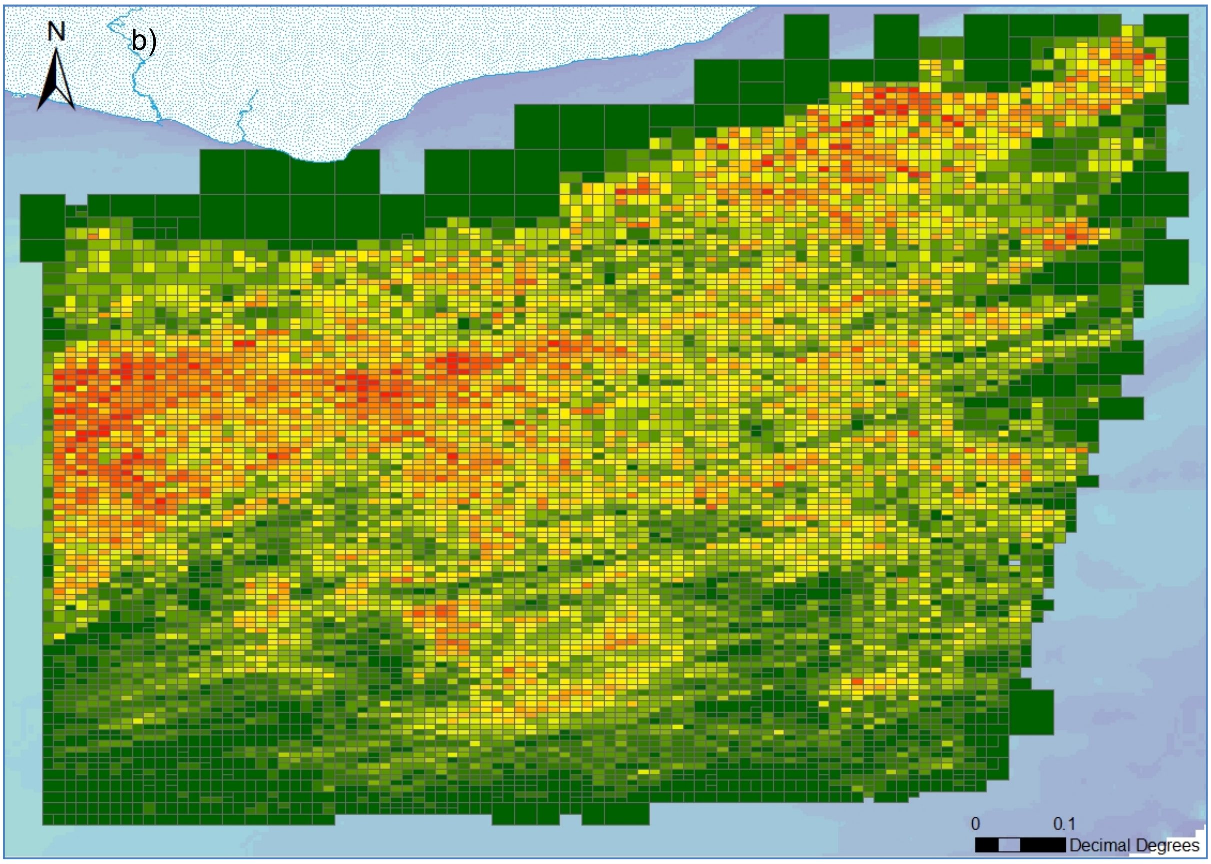


Figure 50. VMS data from the Eastern English Channel aggregated using the nested grid approach with cell division triggers of a) 100 pings and b) 20 pings. Relative pressure scores are coloured red for high and green for low. Gaps indicate pressure scores of zero.

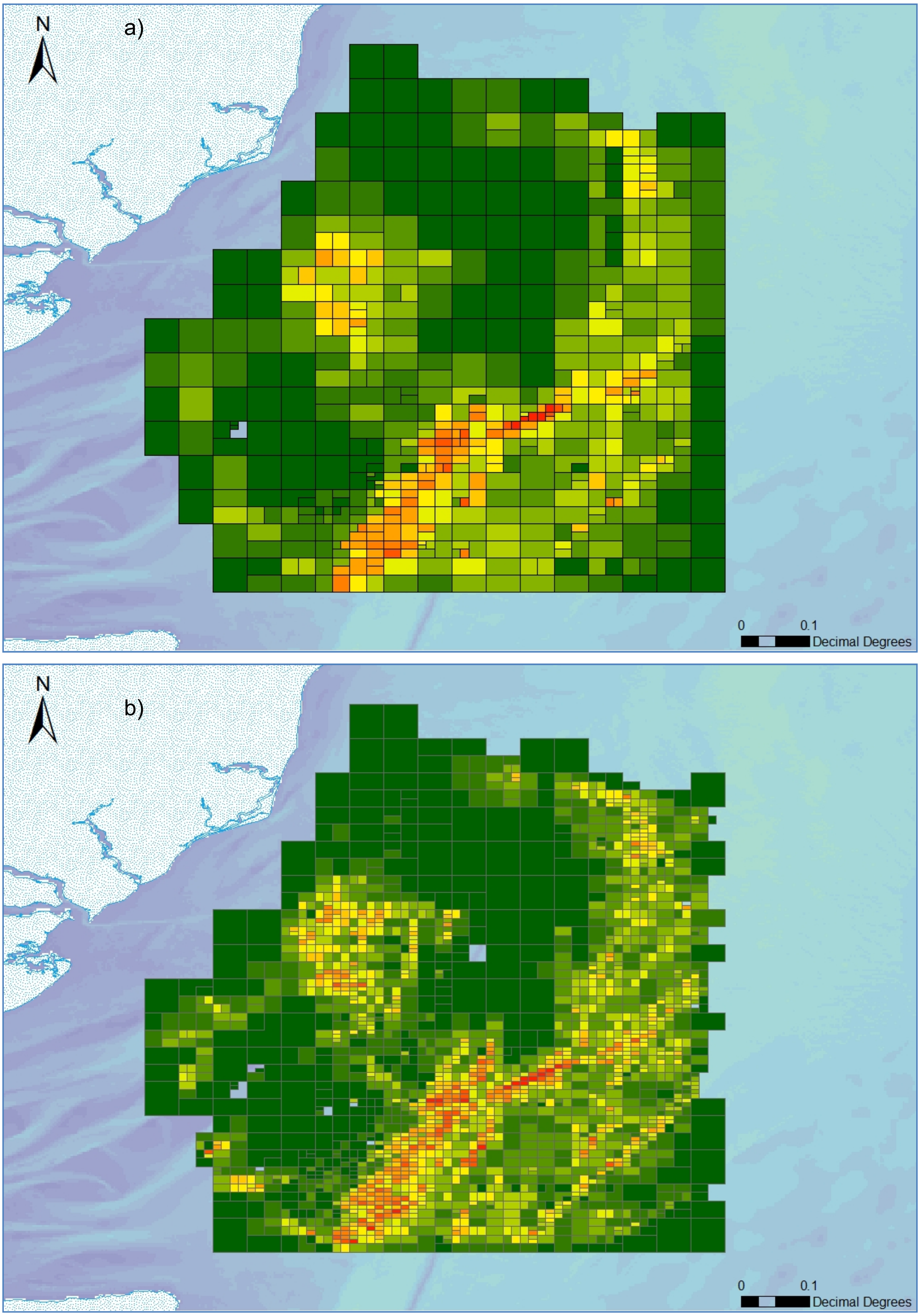


Figure 51. VMS data from the Outer Thames Estuary aggregated using the nested grid approach with cell division triggers of a) 100 pings and b) 20 pings. Relative pressure scores are coloured red for high and green for low. Gaps indicate pressure scores of zero.

The findings from this study highlight the importance of considering the most appropriate scale for the end user when aggregating VMS point data to grids. As shown, when aggregating VMS data to a regular grid, there is a risk of over or under estimating fishing pressure spatially, depending on the scale of the grid.

For mapping the distribution of fishing activity, 3km x 3km grid squares have been suggested previously (Mills *et al* 2007). A coarse resolution may therefore be acceptable for mapping purposes. However, it is expected that finer resolutions would be required to model the effect of fishing activities on seabed habitats, as the relative spatial footprint of benthic sampling gear is small.

4.2 Relating biological variability to fishing pressure

4.2.1 Univariate biodiversity indices

Univariate measures of abundance, richness and diversity calculated for both infaunal and epifaunal communities have regularly been used to describe and quantify the effects of fishing activity on benthic communities (Jennings & Kaiser 1998; Blanchard *et al* 2004). These studies have frequently linked reductions in these measures to fishing activity. However, in the current study, a small, but significant, increase in these indices was found consistently, regardless of fishing pressure method. Where statistically significant relationships were identified, the associated coefficient of determination (R^2) values were relatively low, indicating that the linear regression was a poor fit to the data. It is likely that this is due to confounding factors, such as natural variability and historical disturbance, which will be discussed further under limitations. Another possibility is that, for this habitat, impacts are minimal as associated species are tolerant of the disturbance. Collie *et al* (2001) conducted a meta-analysis of 39 published fishing impact studies. The analysis showed that, in general, communities in less consolidated sediments, such as sand, are less adversely affected by trawling.

What was significant about the univariate models was that, despite the similar habitats at both sites, there was relatively little agreement in the best predictors of biodiversity indices. In fact, there was no consistent best predictor, neither across sites, nor across infaunal univariate responder. Sediment organic carbon content (%) was the only consistent factor in models for both species richness and Margalef's richness at both sites. Though the best minimal models for predicting Margalef's richness at EEC and Th both utilised VMS derived fishing abrasion pressure scores for prediction (though different resolution derivatives for pressure scoring for each site) the value of their addition was minimal.

It is apparent that community composition and diversity across EEC and Th are best described by the environmental conditions within which they exist rather than by the anthropogenic impact to which they are exposed. Fishing pressure has a significant, but small, influence in relation to natural drivers within this habitat.

In general, it appears that data from EEC had a greater power for prediction than that found at Th. This suggests that there was an additional influencing factor at Th that we had not included within our model.

4.2.2 Biological traits analysis

The benthic infaunal communities of the North Sea, and to a lesser extent the English Channel, have been well-studied by benthic ecologists for over one hundred years (e.g. Ford 1923; Rees *et al* 1999; Reis & Krönke 2004). These studies have focussed on the main anthropogenic pressures affecting marine benthic communities including fishing, pollution, construction, extraction and disposal activities (Eggleton *et al* 2012). The impacts of these activities on the receiving benthic communities have been traditionally assessed using standard univariate metrics of abundance, richness and diversity and multivariate techniques using species identity (Hawkins *et al* 2003). However, more recently, there has been a greater focus on assessing the affect on benthic community function based on species life history traits.

An assessment of the changes in community traits along the fishing activity gradient defined during this study identified few distinct changes within the community traits of the infaunal and epifaunal communities that could be attributed to a decrease/increase in fishing activity.

Traits patterns of the infaunal communities were found to be similar to those previously reported from the Greater North Sea subregion (Eggleton *et al* 2012).The key relationships observed between changes in trait proportionality and fishing activity included a gradual increase in the 11-20mm size group within the EEC site and a slight decrease in the 21-100 mm size class at the Th site, demonstrating a trend towards a smaller species size as fishing activity increased. Such a relationship, i.e. a decrease in species size within areas impacted by fishing, has previously been described as a recognised community response to fishing activity (Blanchard *et al* 2004). Other observed changes in community traits included an increase in species with exoskeletons (gastropod shells) at the EEC site as fishing activity increased and an increase in subsurface deposit feeders which correlated well with an increase in burrowing species. Although the above relationships could be attributed to an increase in fishing activity, further consideration must be given to whether infaunal communities are a good indicator of fishing activity i.e. to what extent will fishing gear impact communities already dominated by small, subsurface dwelling fauna ?

As with the infaunal community traits, many of the observed dominant epifaunal community traits were similar to what has previously been described for the Th and EEC regions (Eggleton *et al* 2012). An increase in the feeding trait of predators and scavengers within the epifaunal community, as observed in this study, has been previously described as an effect of fishing, and attributed to the fact that larger species are often those at higher trophic levels (Jennings *et al* 2001b). The higher relative proportion of robust species which have shells and tubes, as observed in this study, may also be expected in areas affected by physical disturbances, such as fishing. The increase in species protected by tough shells and exoskeletons is thought to be due to the increased survivability afforded to these species when they are directly impacted by fishing gear when compared with more fragile species (Hall 1999, Hiddink *et al* 2006).

A multivariate assessment of all stations at each site using the abundance weighted traits matrix implied that there was no particular separation of stations based on their traits identity i.e. none were found to be functionally-distinct. This might be due to variability within survey blocks in terms of fishing activity. Likewise, the fact that samples were collected from areas that had very similar environmental characteristics may explain why little difference in species traits makeup was identified.

4.2.3 Natural disturbance

In this study, the W-statistic did not significantly relate to any scores of fishing pressure. However, for infauna, the magnitude of the W-statistic at Th may indicate that the site was already disturbed and the trend at EEC was for increasing disturbance with fishing pressure score. Another reason for the lack of significant relationships might be that the W-statistic itself is not suited to the broad scale nature of this study, where impact related effects are too dispersed, both spatially and temporally, to be identified by the community k-dominance.

Species/community traits makeup from areas of high levels of natural disturbance are thought to be similar to those species/community traits attributed to benthic communities that are exposed to high levels of fishing activity. The relative impact of such activities on benthic communities is also thought to be partly due to whether the anthropogenic disturbance exceeds background levels of natural disturbance (Jennings & Kaiser 1998). Although levels of natural disturbance were not factored into the design of this study, previous studies (Eggleton *et al* 2011) and (Diesing *et al* 2013) have demonstrated that the Th and EEC sites are both characterised by high levels of natural disturbance.

4.3 Wider policy context

4.3.1 Detecting change

As mentioned previously, the UK Marine Biodiversity Monitoring R&D Programme is led by JNCC on behalf of the Statutory Nature Conservation Bodies and delivered through UKMMAS. The Programme aims to deliver status and trend information capable of detecting change in the condition of marine habitats and species across the whole UK marine environment, both within protected sites and outside.

What this study has shown is that, for this sandy sedimentary habitat, effects of fishing are significant, but small. Any seabed monitoring intended to detect possible impacts of fishing pressure on benthic communities will need to be sufficiently robust and powerful to pick up these subtle changes against a background of natural variability. Here, statistical models were used to distinguish natural and anthropogenic effects on the benthic community.

The fact that benthic communities exposed to high levels of natural disturbance and high levels of fishing activity tend to express the same functional traits makes it very difficult to define and quantify which of these is having the greatest affect on the benthic communities and thus driving community structure. In addition, benthic communities exposed to high levels of natural disturbance will be less sensitive to physical disturbance when compared with communities accustomed to low natural disturbance, therefore making it more difficult to detect changes in community traits across a fishing activity gradient (Collie *et al* 2000; Eggleton *et al* 2011). In order to carry out any power analysis it is first necessary to determine the magnitude of change that is of interest. In the case of this study, the effect size of fishing was very small. In the future, it may be worth carrying out a cost-benefit analysis in order to help decide whether it is worth carrying out expensive offshore monitoring programmes in order to detect marginal impacts with a background of natural disturbance, especially as, typically, survey costs are likely to be inversely related to measurable effect size.

It is important to remember that the majority of the seabed around the UK coast may have been impacted (at some time) by fishing activity (or other anthropogenic pressures) and that species and communities present may have adapted to this pressure and have thus been influenced by both past and continued disturbance. Thus, historical context should be considered in future studies in order that the sum of influencing factors on the environment can be considered.

In this study, 18 months of VMS data were used to produce fishing pressure layers based on recovery times of this habitat from previous studies (e.g. Dernie *et al* 2003). Future work could examine the best temporal scale for such studies, as well as the best spatial scale on which to produce fishing pressure layers.

4.3.2 VMS processing methods

The ability to detect the impact of fishing on the seabed relies on being able to accurately attribute a known level of fishing intensity to biological samples. Whilst raw VMS “ping” data and vectorised tracks are not currently available due to issues of vessel anonymity and privacy, there are also issues with positional accuracy. VMS systems are positioned on the vessel, not the gear, so it is unlikely that if an individual ping were targeted by a grab it would actually land on the trawl scar. It is expected that, for the immediate future, aggregated VMS data are the best spatial information available on fishing pressure.

What this study has shown is that it is possible to use the current standard of a 0.05dd grid in order to plan surveys along fishing pressure gradients and investigate pressure-state relationships in different benthic habitats. However, the performance of the gridded fishing pressure scores does appear to vary slightly with site, possibly based on the distribution of fishing effort. Future studies could look at the performance of different gridding methods for different fishing fleets to see how robust they are to different gear and fleet behaviours.

Over estimation of fishing intensity may result in the selection of sample locations that are less likely to represent areas where trawling has taken place. In a low resolution grid, the chances of taking a sample from an area of seabed that has actually been trawled from a cell with few VMS points are small. This is one reason why we may have seen such high variability in the response variables for each of the fishing pressure scores.

To detect change in benthic ecosystems in relation to fishing activity, it is desirable to sample across a gradient of fishing pressure. The fishing intensity scores that result from gridding provide a means of identifying cells that represent a gradient of fishing pressure, which can be used to inform sampling strategies. As mentioned previously, scale is an important factor when aggregating VMS point data. The scale, along with associated over or under estimation of fishing intensity, will have a direct effect on which cells best represent this gradient. Cells identified as having a high pressure score on a low resolution grid, for example, may be comprised of a combination of high and low scored cells on a smaller scale grid (Figure 6).

The plots of average intensity scores per block for each site (Figure 8) show that the pressure gradient across the selected blocks remained intact irrespective of the scale at which the VMS data are aggregated. However, the amount of variability in the scores for each block increases as the scale becomes finer. This clearly demonstrates the heterogeneity of fishing effort within each original 0.05dd block when the data are examined at a finer spatial scale. Blocks with scores in the middle of the gradient seem to be most affected by the change in aggregation scale, as mean scores at both sites decrease with the increase in resolution. However, what the statistical analysis demonstrated was that there was little difference in terms of explanatory power between the different VMS processing methods. Perhaps surprisingly, the most sophisticated method, that proposed by Gerritsen *et al* (2013) had the lowest explanatory power.

4.3.3 Other ways to present VMS data

As mentioned previously, track interpolation based on VMS points could provide an alternative to gridding. However, these techniques rely on vessel identity data for each point being available and have inherent issues with accuracy. Skaar *et al* (2011) suggest that interpolated VMS data using current techniques may be suitable for mapping the large scale distribution of fishing effort but may be unsuitable for linking fishing activity to benthic impacts, particularly with a two hour ping interval. The present study broadly agrees with this. Shortening the ping interval to 1 hour has been shown to improve the accuracy of track interpolation (Skaar *et al* 2011). However, Gerritsen *et al* (2013) suggest that the swept area ratio approach, as used in the nested grid element of this study, is not appreciably affected by changes in the VMS reporting interval. Gerritsen *et al* (2013) suggest that the precision of the effort estimate in each cell is not directly determined by the size of the cell, but rather by the number of observations in the cell, as the VMS points are effectively random samples in the absence of supplementary information that link related points sequentially. As such it is unclear if a reduction in ping interval would be of benefit to the approaches used in this study.

4.3.4 Monitoring pressures

A risk-based approach to monitoring broadly recommends the identification of habitats and species which are at different levels of risk (with an associated confidence level), depending on the pressures that they are exposed to. It will then be possible to stratify sampling of habitats and species along a gradient of risk using this data, e.g. taking samples from a single habitat type at low risk, medium risk and high risk. Collecting data along such a gradient will not only result in an efficient sample design, but also provide new data on the response of habitats and species to various levels and combinations of pressures.

This study suggests that, particularly for this habitat, due to variability around response variables, no natural groupings of this pressure, and the subtle effects fishing pressure has on the benthic communities, a regression approach may be preferable to a categorical one (e.g. ANOVA type). Furthermore, there was no obvious tipping point where the fishing pressure began to have a greater effect on biodiversity indices or community structure.

The Charting Progress 2 report (UK Monitoring and Assessment Strategy, 2010) highlighted bottom towed fishing gear to be one of the most widespread, yet manageable, anthropogenic impacts on the seabed. It should be noted that the conclusions of this study are only relevant to the habitat on which the study was undertaken and in order to inform future monitoring surveys, studies of this kind are required on other sedimentary habitat types.

4.3.5 Limitations of study

- It was assumed that fishing abrasion pressure within each cell was homogeneous. Regridding pressure layers to finer resolutions showed that this was not always the case. This in turn could have led to the wrong abrasion scores being attributed to biological samples and may have been why variability was so high in the linear regressions undertaken.
- Environmental factors had a stronger effect than fishing on this habitat. At EEC, these factors were included in the model and a large proportion of the biological variability was explained, but the low explanatory power often associated with Th indicates that an additional, unmeasured, factor was also having a significant effect.
- The experimental design did not account for historical fishing disturbance or natural disturbance at the two sites. Natural and anthropogenic factors may have already modified the biological communities present at the sites in ways that have not been fully investigated;

    - Historical fishing disturbance at the sites may have meant that the communities were being maintained in an already modified state. As such, short-term changes in fishing pressure would have had no further effect on community organisation. This scenario requires further work to be undertaken not only to investigate historical data at both sites but to also identify a meaningful time scale (community/habitat significant) for fisheries data to be aggregated over before subsequent abrasive fishing pressure calculations for future studies.
    - Environmental conditions at both sites may have pre-disposed the benthic communities to exist in a dynamic environment which masks any community change which might be expected to occur with increasing abrasive fishing pressure exposure.

4.3.6 Future work

The results of this study identify several areas that may warrant future work, mostly to address the limitations outlined above.

The present study does not take into account historical fishing activities, as only VMS data for the 18 month period prior to the survey were used. As a result, we do not know how heavily impacted the two sites were by fishing activities prior to 2010. Not considering historic fishing activity, even qualitatively, could hamper the ability of the statistical analyses to detect fishing-induced changes in the communities, as there is no way to determine the state of disturbance of the sampling locations at the time of sampling.

Similarly, neither the resilience of the benthic community nor the amount of natural disturbance that a site is subjected to could be accounted for. The benthic assemblages in the Th and EEC, for example, may already be exposed to a high amount of natural disturbance. An estimate of this disturbance is required in order to disentangle the effects of natural and anthropogenic disturbance. It may be possible to use a proxy of natural physical disturbance, such as energy or grain size, to try and distinguish these effects.

Other studies have recently looked at the frequency of trawling events and the recovery capacity of the community, in order to predict whether communities are being maintained in a modified state. This may help to address issues of historic impacts and natural disturbance.

Finally, this study was standardised to a single sediment type. In reality, monitoring surveys are likely to be across multiple habitat types. It is recommended that a similar study to this one be carried out on a different sediment type, preferably finer, less naturally disturbed and therefore more sensitive to physical abrasion, in order to combine the results and draw more general conclusions about the use of VMS data to produce useful fishing pressure layers.

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6 Annexes

6.1 Biodiversity indices

Table 8. Average community univariates per 0.05dd block from infaunal data.

Block	VMS 0.05	S	N	d	J'	H'	Lambda	Biomass
EEC_06	300	22	88	4.814	0.790	2.334	0.171	7.681
EEC_02	621	10	17	3.265	0.928	1.820	0.252	0.645
EEC_09	764	19	57	4.561	0.890	2.400	0.135	6.496
EEC_05	988	17	50	4.063	0.851	2.245	0.157	2.941
EEC_08	1195	22	61	5.128	0.840	2.454	0.145	12.467
EEC_11	1226	25	67	5.779	0.888	2.665	0.117	14.168
EEC_12	1329	32	95	6.767	0.865	2.816	0.101	29.584
EEC_03	1509	12	24	3.501	0.894	2.030	0.191	2.022
EEC_07	1859	25	75	5.623	0.817	2.528	0.160	8.203
EEC_04	1939	17	51	4.151	0.848	2.213	0.177	3.005
EEC_01	2176	10	17	3.265	0.928	1.820	0.252	0.645
EEC_10	2286	24	65	5.600	0.886	2.658	0.116	18.015
TH_05	420	6	11	2.225	0.907	1.548	0.273	1.033
TH_06	569	6	12	1.896	0.908	1.412	0.336	0.810
TH_08	796	5	6	2.175	0.956	1.285	0.357	1.680
TH_12	841	4	9	1.789	0.916	1.242	0.334	0.300
TH_01	1431	9	18	3.178	0.880	1.835	0.263	1.342
TH_10	1640	6	22	1.932	0.881	1.408	0.315	1.911
TH_07	2046	14	59	3.436	0.790	2.067	0.197	3.931
TH_02	2115	5	7	2.009	0.930	1.250	0.373	0.933

6.2 Collinearity in explanatory variables

Table 9. Pearson correlation coefficients for explanatory variables showing collinearity between them and which to exclude from further analysis

	VMS 0.05	VMS 0.025	VMS 0.0125	Depth (m)	%Gravel	%Sand	%Silt/clay	Dist From Shore (km)	Organic carbon (% m/m)
All data									
VMS_025	0.841								
VMS_0125	0.754	0.877							
Depth	0.428	0.441	0.470						
%Gravel	0.363	0.349	0.287	-0.135					
%Sand	-0.310	-0.298	-0.250	0.131	-0.980				
%Silt/clay	-0.128	-0.122	-0.073	-0.026	0.258	-0.447			
Dist From Shore (km)	-0.160	-0.117	-0.133	-0.211	0.071	-0.032	-0.160		
Org carbon (% m/m)	0.063	0.087	0.027	-0.070	0.422	-0.469	0.372	0.092	
Nitrogen (% m/m)	0.033	0.021	0.083	0.036	0.086	-0.119	0.190	-0.137	0.068
EEC									
VMS_025	0.842								
VMS_0125	0.737	0.862							
Depth	0.121	0.099	0.161						
%Gravel	0.277	0.213	0.172	0.088					
%Sand	-0.329	-0.271	-0.228	-0.022	-0.811				
%Silt/clay	0.178	0.168	0.151	-0.085	0.000	-0.586			
Dist From Shore (km)	-0.191	-0.172	-0.187	0.227	0.124	0.027	-0.217		
Org carbon (% m/m)	0.201	0.180	0.101	-0.109	0.242	-0.466	0.460	-0.185	
Nitrogen (% m/m)	0.144	0.114	0.191	0.007	0.055	-0.167	0.210	-0.123	0.236
Th									
VMS_025	0.847								
VMS_0125	0.712	0.845							
Depth	-0.269	-0.277	-0.204						
%Gravel	0.137	0.044	0.009	0.387					
%Sand	-0.351	-0.246	-0.207	-0.168	-0.685				
%Silt/clay	0.351	0.296	0.276	-0.140	-0.017	-0.717			
Dist From Shore (km)	-0.182	-0.190	-0.204	0.579	0.336	0.033	-0.367		
Org carbon (% m/m)	0.358	0.270	0.172	-0.146	-0.001	-0.467	0.643	-0.352	
Nitrogen (% m/m)	0.323	0.243	0.351	-0.042	0.061	-0.229	0.257	-0.221	0.477

6.3 Linear regression

Table 10. p-values and r^2 values for linear regression between infaunal univariate responders and the abrasive pressure scores derived from VMS grids of 0.05, 0.025 and 0.0125decimal degrees along with swept area scores derived from the 20 and 100 VMS ping rules, at EEC.

EEC	0.05		0.025		0.125		20		100	
	p	r^2	p	r^2	p	r^2	p	r^2	p	r^2
Sqrt N	0.827	0.000	0.211	0.013	0.145	0.018	0.496	0.004	0.881	0.000
Sqrt S	0.223	0.013	0.007	0.059	0.003	0.072	0.694	0.001	0.252	0.022
d	0.150	0.018	0.008	0.059	0.003	0.074	0.662	0.002	0.263	0.011
H'	0.016	0.048	<0.001	0.118	<0.001	0.123	0.084	0.025	0.031	0.038
Ln Biomass	0.833	0.000	0.264	0.011	0.205	0.014	0.397	0.006	0.540	0.003

Table 11. p-values and r^2 values for linear regression between infaunal univariate responders and the abrasive pressure scores derived from VMS grids of 0.05, 0.025 and 0.0125decimal degrees along with swept area scores derived from the 20 and 100 VMS ping rules, at Th.

Th	0.05		0.025		0.125		20		100	
	p	r^2	p	r^2	p	r^2	p	r^2	p	r^2
Sqrt N	0.002	0.120	0.022	0.067	0.088	0.038	0.028	0.062	0.089	0.038
Sqrt S	0.007	0.091	0.084	0.039	0.232	0.019	0.130	0.030	0.328	0.013
d	0.013	0.082	0.095	0.038	0.100	0.037	0.117	0.033	0.262	0.017
H'	0.153	0.027	0.564	0.004	0.988	0.000	0.877	0.000	0.855	0.000
Ln Biomass	0.012	0.079	0.029	0.061	0.047	0.051	0.016	0.074	0.062	0.045

Table 12. p-values and r^2 values for linear regression between epifaunal univariate responders and the abrasive pressure scores derived from VMS grids of 0.05, 0.025 and 0.0125decimal degrees along with swept area scores derived from the 20 and 100 VMS ping rules, at EEC.

EEC	0.05		0.025		0.125		20		100	
	p	r^2	p	r^2	p	r^2	p	r^2	p	r^2
Sqrt N	0.131	0.213	0.125	0.219	0.495	0.0478	0.0702	0.291	0.2322	0.139
Sqrt S	0.014	0.471	0.183	0.17	0.192	0.164	0.7144	0.014	0.172	0.178
d	0.01	0.505	0.129	0.214	0.224	0.144	0.2279	0.142	0.104	0.242
H'	0.051	0.33	0.131	0.213	0.468	0.0538	0.1834	0.17	0.3025	0.106
Ln Biomass	0.868	0.00289	0.622	0.0252	0.96	0.000261	0.1296	0.214	0.4179	0.067

Table 13. p-values and r^2 values for linear regression between epifaunal univariate responders and the abrasive pressure scores derived from VMS grids of 0.05, 0.025 and 0.0125decimal degrees along with swept area scores derived from the 20 and 100 VMS ping rules, at Th.

Th	0.05		0.025		0.125		20		100	
	p	r^2	p	r^2	p	r^2	p	r^2	p	r^2
Sqrt N	0.389	0.189	0.697	0.0418	0.662	0.0525	0.3559	0.214	0.1407	0.457
Sqrt S	0.21	0.357	0.155	0.434	0.005	0.89	0.0381	0.699	0.0575	0.636
d	0.733	0.0324	0.481	0.131	0.46	0.143	0.5971	0.076	0.9984	0.000001
H'	0.773	0.0233	0.958	0.00078	0.906	0.00398	0.91	0.004	0.5612	0.091
Ln Biomass	0.107	0.518	0.216	0.35	0.194	0.378	0.0291	0.735	0.0074	0.863

Table 14. p-values and r^2 values for linear regression between epifaunal and infaunal Wstat scores and the abrasive pressure scores derived from VMS grids of 0.05, 0.025 and 0.0125decimal degrees along with swept area scores derived from the 20 and 100 VMS ping rules, at EEC and Th.

Infauna	EEC Wstat		Th Wstat		Epifauna	EEC Wstat		Th Wstat	
	p	r^2	p	r^2		p	r^2	p	r^2
0.05	0.2065	0.013	0.3568	0.011		0.3823	0.077	0.4494	0.149
0.025	0.2189	0.013	0.2318	0.019		0.3545	0.086	0.4152	0.171
0.0125	0.5694	0.003	0.2715	0.016		0.2389	0.136	0.7577	0.027
20	0.3293	0.008	0.3612	0.011		0.9237	0.001	0.3886	0.189
100	0.7681	0.001	0.2731	0.016		0.4388	0.061	0.7852	0.021

6.4 W-Stat ANOVA

6.4.1 Eastern English Channel

Table 15. One way ANOVA comparison of W-stat scores between abrasion pressure blocks derived from a VMS grid resolution of 0.05decimal degrees at EEC.

	DF	Sum of Squares	Mean Square	F-Value	P-Value	Lambda	Power
Block	11	0.519	0.047	1.332	0.2168	14.652	0.679
Residual	108	3.826	0.035				

Means Table for WStat
Effect: Block

	Count	Mean	Std. Dev.	Std. Err.
EEC_01: 2176	10	0.365	0.212	0.067
EEC_02: 621	10	0.431	0.247	0.078
EEC_03: 1554	10	0.564	0.248	0.079
EEC_04: 1880	10	0.383	0.151	0.048
EEC_05: 988	10	0.34	0.124	0.039
EEC_06: 300	10	0.3	0.167	0.053
EEC_07: 1859	10	0.433	0.199	0.063
EEC_08: 1195	10	0.368	0.098	0.031
EEC_09: 764	10	0.359	0.201	0.064
EEC_10: 2286	10	0.416	0.135	0.043
EEC_11: 1226	10	0.397	0.219	0.069
EEC_12: 1329	10	0.469	0.188	0.06

Interaction Bar Plot for WStat
Effect: Block

Fisher's PLSD for WStat
Effect: Block
Significance Level: 5 %

	Mean Diff.	Crit. Diff	P-Value	
EEC_01, EEC_02	-0.067	0.167	0.4292	
EEC_01, EEC_03	-0.2	0.167	0.0194	S
EEC_01, EEC_04	-0.018	0.167	0.8297	
EEC_01, EEC_05	0.024	0.167	0.7727	
EEC_01, EEC_06	0.064	0.167	0.4461	
EEC_01, EEC_07	-0.069	0.167	0.4161	
EEC_01, EEC_08	-0.003	0.167	0.9698	
EEC_01, EEC_09	0.006	0.167	0.9424	
EEC_01, EEC_10	-0.051	0.167	0.5459	
EEC_01, EEC_11	-0.032	0.167	0.7057	

EEC_01, EEC_12	-0.105	0.167	0.217	
EEC_02, EEC_03	-0.133	0.167	0.1173	
EEC_02, EEC_04	0.049	0.167	0.5645	
EEC_02, EEC_05	0.091	0.167	0.2812	
EEC_02, EEC_06	0.131	0.167	0.1221	
EEC_02, EEC_07	-0.002	0.167	0.9818	
EEC_02, EEC_08	0.064	0.167	0.4515	
EEC_02, EEC_09	0.073	0.167	0.3884	
EEC_02, EEC_10	0.016	0.167	0.8514	
EEC_02, EEC_11	0.035	0.167	0.679	
EEC_02, EEC_12	-0.038	0.167	0.6549	
EEC_03, EEC_04	0.182	0.167	0.0332	S
EEC_03, EEC_05	0.224	0.167	0.009	S
EEC_03, EEC_06	0.264	0.167	0.0022	S
EEC_03, EEC_07	0.131	0.167	0.1227	
EEC_03, EEC_08	0.196	0.167	0.0214	S
EEC_03, EEC_09	0.206	0.167	0.0161	S
EEC_03, EEC_10	0.149	0.167	0.0801	
EEC_03, EEC_11	0.168	0.167	0.0487	S
EEC_03, EEC_12	0.095	0.167	0.2608	
EEC_04, EEC_05	0.043	0.167	0.6145	
EEC_04, EEC_06	0.083	0.167	0.3291	
EEC_04, EEC_07	-0.051	0.167	0.5492	
EEC_04, EEC_08	0.015	0.167	0.8593	
EEC_04, EEC_09	0.024	0.167	0.7738	
EEC_04, EEC_10	-0.033	0.167	0.6972	
EEC_04, EEC_11	-0.014	0.167	0.8708	
EEC_04, EEC_12	-0.086	0.167	0.3071	
EEC_05, EEC_06	0.04	0.167	0.6357	
EEC_05, EEC_07	-0.093	0.167	0.2712	
EEC_05, EEC_08	-0.028	0.167	0.7439	
EEC_05, EEC_09	-0.018	0.167	0.8285	
EEC_05, EEC_10	-0.075	0.167	0.3726	
EEC_05, EEC_11	-0.056	0.167	0.5054	
EEC_05, EEC_12	-0.129	0.167	0.1286	
EEC_06, EEC_07	-0.133	0.167	0.1168	
EEC_06, EEC_08	-0.068	0.167	0.424	
EEC_06, EEC_09	-0.058	0.167	0.4903	
EEC_06, EEC_10	-0.115	0.167	0.1734	
EEC_06, EEC_11	-0.096	0.167	0.2554	
EEC_06, EEC_12	-0.169	0.167	0.0473	S
EEC_07, EEC_08	0.066	0.167	0.438	
EEC_07, EEC_09	0.075	0.167	0.3761	
EEC_07, EEC_10	0.018	0.167	0.8336	
EEC_07, EEC_11	0.037	0.167	0.6624	
EEC_07, EEC_12	-0.036	0.167	0.6713	
EEC_08, EEC_09	0.009	0.167	0.9123	
EEC_08, EEC_10	-0.048	0.167	0.5713	

EEC_08, EEC_11	-0.029	0.167	0.734
EEC_08, EEC_12	-0.101	0.167	0.2312
EEC_09, EEC_10	-0.057	0.167	0.4991
EEC_09, EEC_11	-0.038	0.167	0.6528
EEC_09, EEC_12	-0.111	0.167	0.1915
EEC_10, EEC_11	0.019	0.167	0.8207
EEC_10, EEC_12	-0.054	0.167	0.5261
EEC_11, EEC_12	-0.073	0.167	0.3899

6.4.2 Outer Thames Estuary

Table 16. One way ANOVA comparison of W-stat scores between abrasion pressure blocks derived from a VMS grid resolution of 0.05decimal degrees at Th.

ANOVA Table for WStat

	DF	Sum of Squares	Mean Square	F-Value	P-Value	Lambda	Power
Block	7	0.577	0.082	1.474	0.1908	10.317	0.575
Residual	70	3.917	0.056				

Means Table for WStat
Effect: Block

	Count	Mean	Std. Dev.	Std. Err.
TH_01: 1431	10	0.375	0.248	0.078
TH_02: 2115	10	0.442	0.243	0.077
TH_05: 420	10	0.444	0.216	0.068
TH_06: 569	9	0.314	0.246	0.082
TH_07: 2046	10	0.322	0.132	0.042
TH_08: 796	10	0.532	0.324	0.103
TH_10: 1640	9	0.447	0.188	0.063
TH_12: 841	10	0.571	0.246	0.078

Interaction Bar Plot for WStat
Effect: Block

Fisher's PLSD for WStat
Effect: Block
Significance Level: 5 %

	Mean Diff.	Crit. Diff	P-Value
TH_01, TH_02	-0.067	0.211	0.5276
TH_01, TH_05	-0.068	0.211	0.5208
TH_01, TH_06	0.061	0.217	0.5753
TH_01, TH_07	0.053	0.211	0.6148
TH_01, TH_08	-0.157	0.211	0.1418
TH_01, TH_10	-0.072	0.217	0.5114

TH_01, TH_12	-0.196	0.211	0.0685	
TH_02, TH_05	-0.001	0.211	0.9917	
TH_02, TH_06	0.128	0.217	0.2417	
TH_02, TH_07	0.121	0.211	0.258	
TH_02, TH_08	-0.09	0.211	0.3977	
TH_02, TH_10	-0.005	0.217	0.9665	
TH_02, TH_12	-0.129	0.211	0.2284	
TH_05, TH_06	0.129	0.217	0.2377	
TH_05, TH_07	0.122	0.211	0.2537	
TH_05, TH_08	-0.089	0.211	0.4035	
TH_05, TH_10	-0.003	0.217	0.9746	
TH_05, TH_12	-0.127	0.211	0.2323	
TH_06, TH_07	-0.008	0.217	0.9436	
TH_06, TH_08	-0.218	0.217	0.0484	S
TH_06, TH_10	-0.133	0.222	0.2373	
TH_06, TH_12	-0.257	0.217	0.0209	S
TH_07, TH_08	-0.211	0.211	0.0504	
TH_07, TH_10	-0.125	0.217	0.2532	
TH_07, TH_12	-0.249	0.211	0.0213	S
TH_08, TH_10	0.085	0.217	0.4344	
TH_08, TH_12	-0.039	0.211	0.7168	
TH_10, TH_12	-0.124	0.217	0.2579	

6.5 Biomass Multivariate Analysis

6.5.1 Eastern English Channel

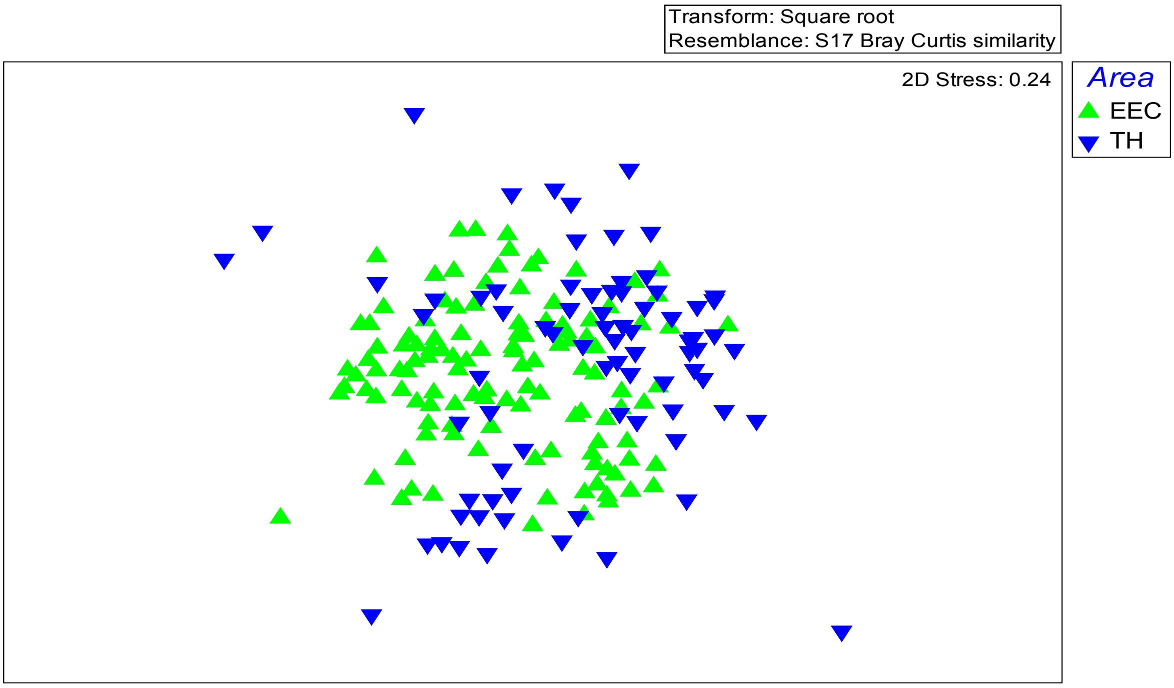


Figure 52. MDS plot for all infaunal community biomass stations and overlaid by site.

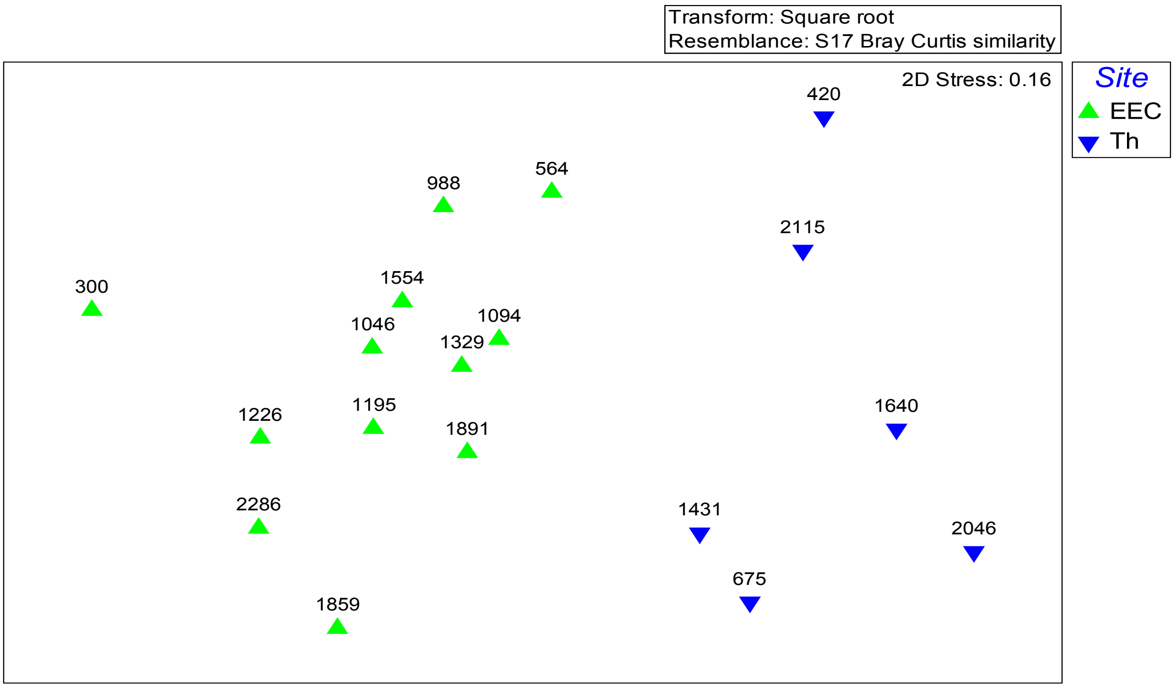


Figure 53. MDS plot for all epifaunal community biomass stations and overlaid by site and labelled by 0.05 resolution pressure scores.

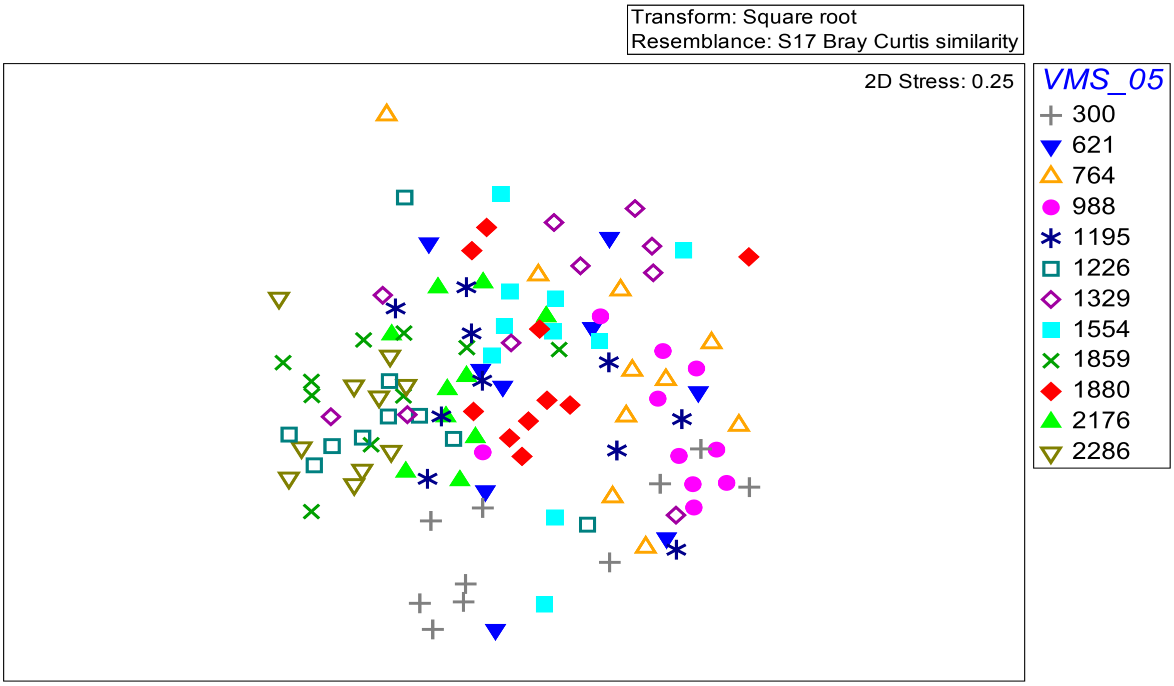


Figure 54. MDS plot for infaunal biomass data at EEC overlaid with pressures scores derived from a VMS resolution of 0.05 decimal degrees.

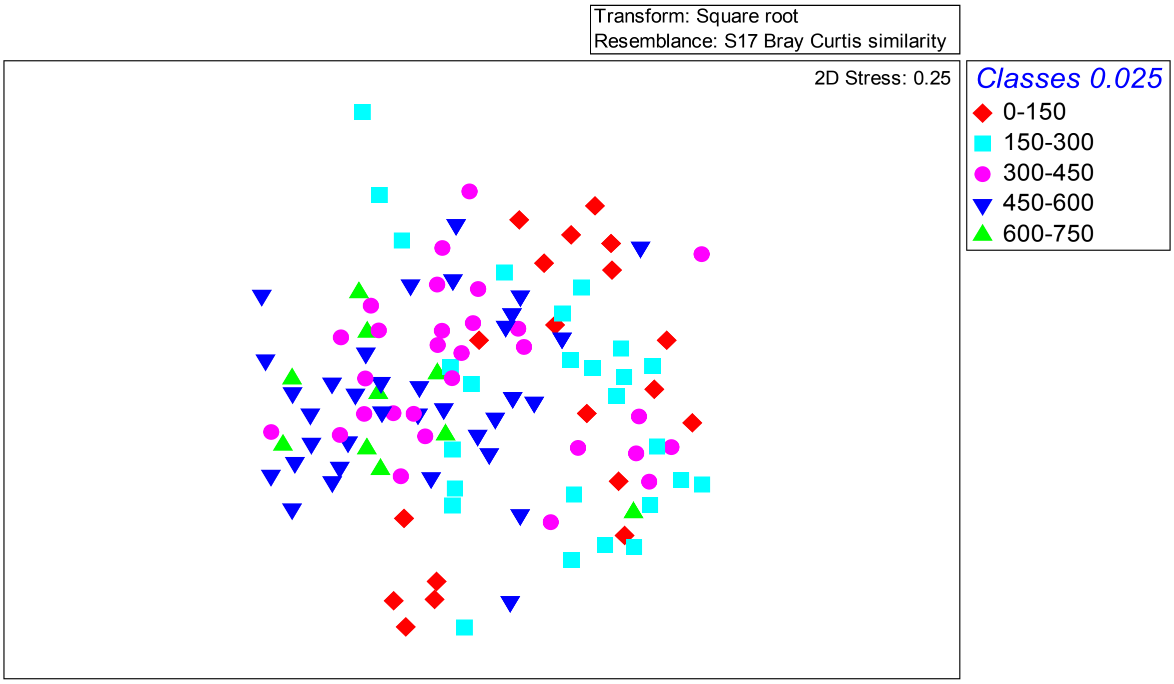


Figure 55. MDS plot for infaunal biomass data at EEC overlaid with classed pressures scores derived from a VMS resolution of 0.025 decimal degrees.

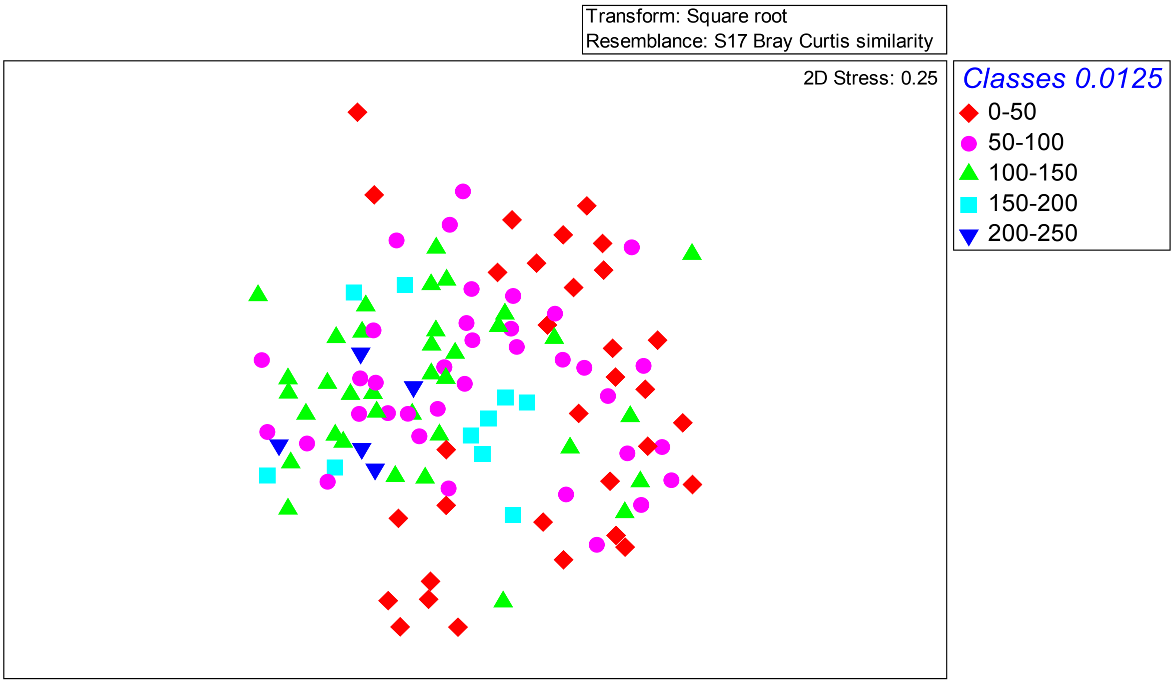


Figure 56. MDS plot for infaunal biomass data at EEC overlaid with classed pressures scores derived from a VMS resolution of 0.0125 decimal degrees.

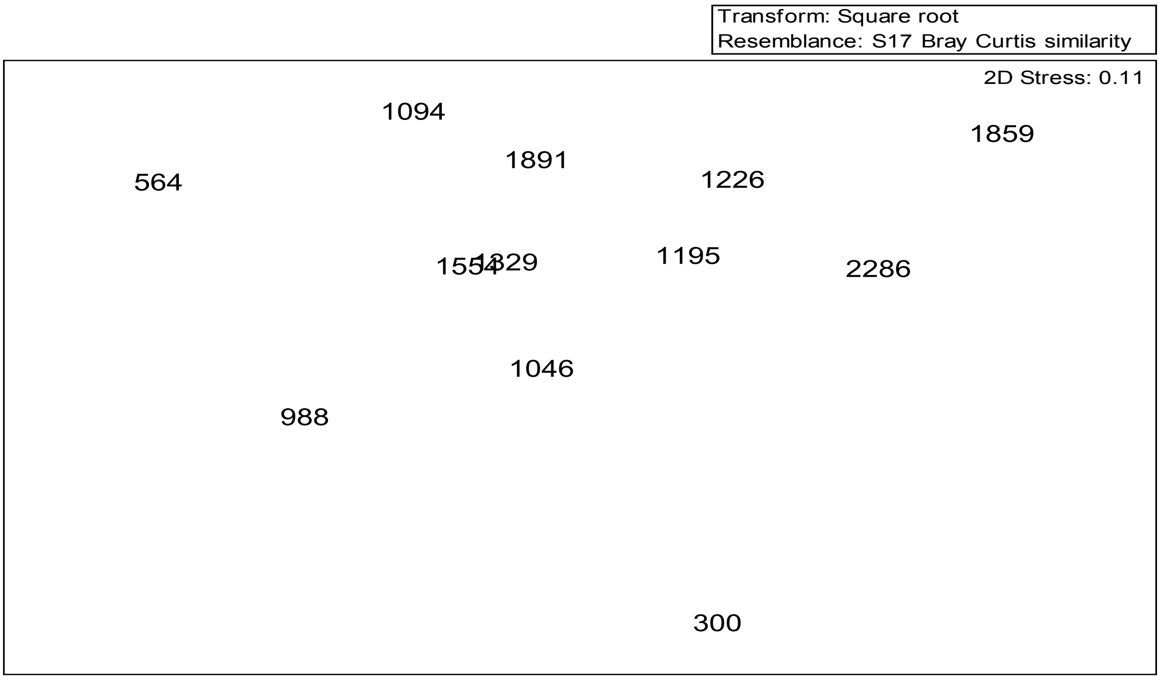


Figure 57. MDS plot for epifaunal biomass data at EEC overlaid with pressures scores derived from a VMS resolution of 0.05 decimal degrees.

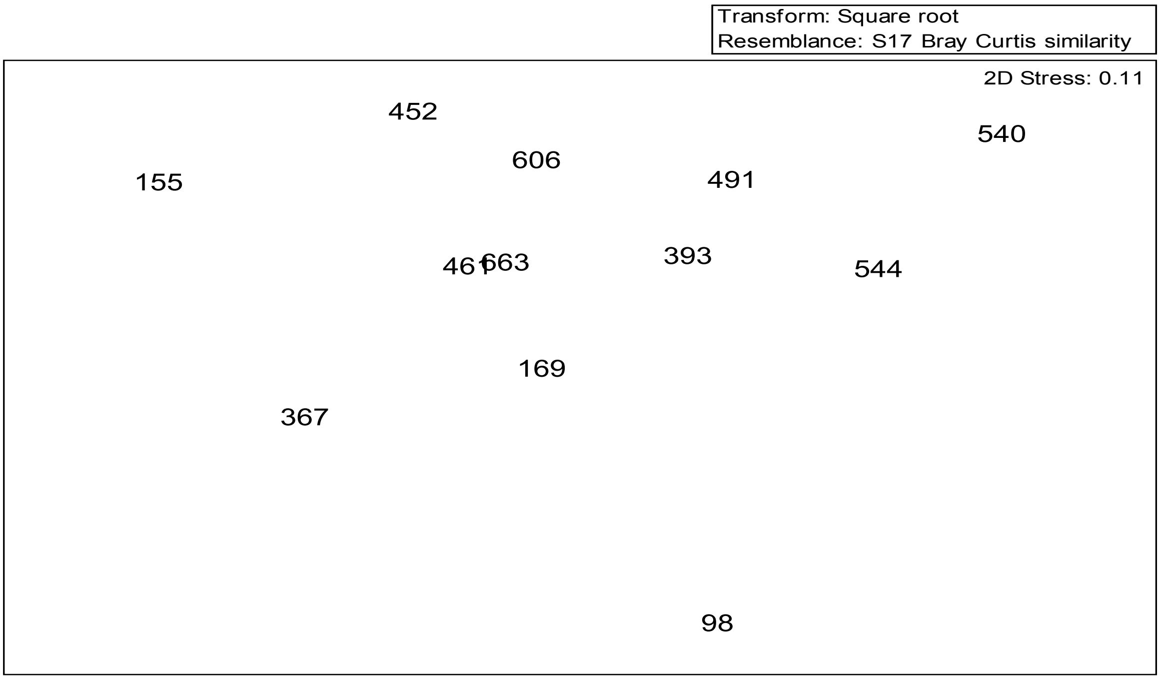


Figure 58. MDS plot for epifaunal biomass data at EEC overlaid with pressures scores derived from a VMS resolution of 0.025 decimal degrees.

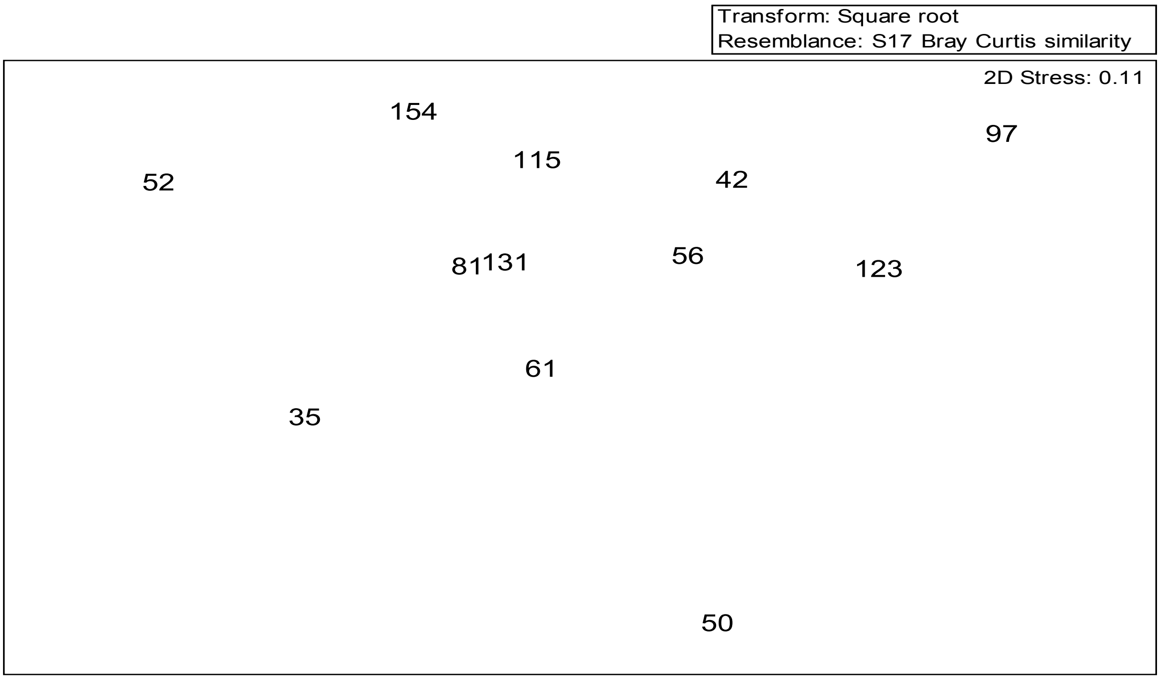


Figure 59. MDS plot for epifaunal biomass data at EEC overlaid with pressures scores derived from a VMS resolution of 0.0125 decimal degrees.

6.5.2 Outer Thames Estuary

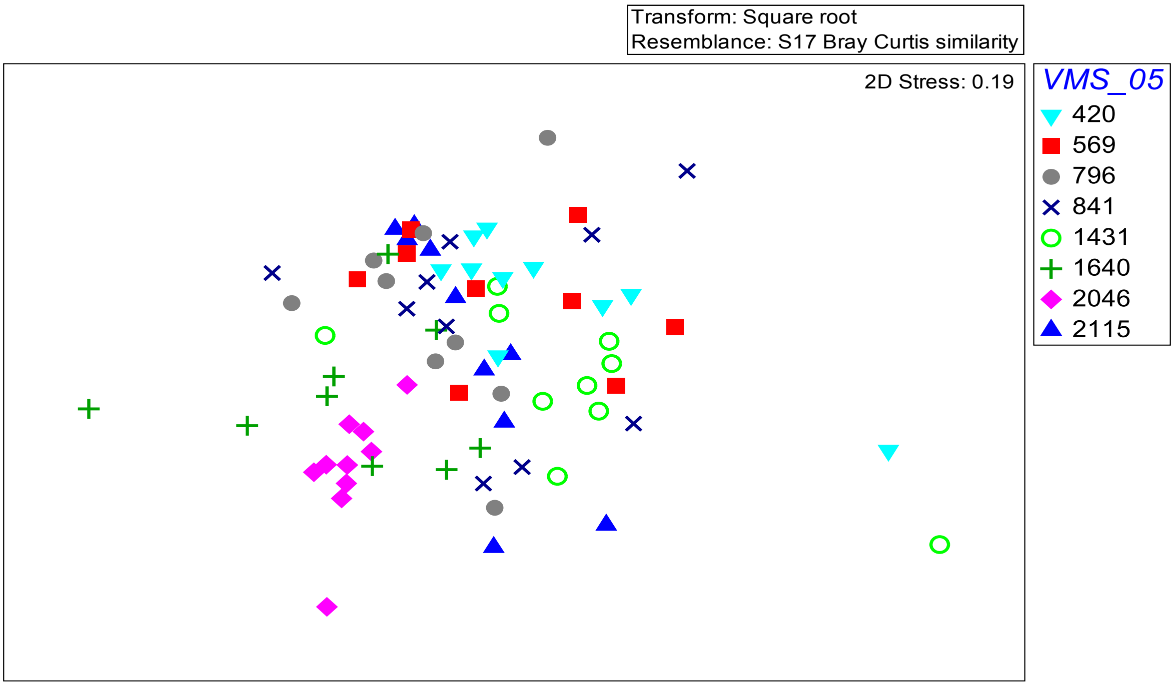


Figure 60. MDS plot for infaunal biomass data at Th overlaid with classed pressures scores derived from a VMS resolution of 0.05 decimal degrees.

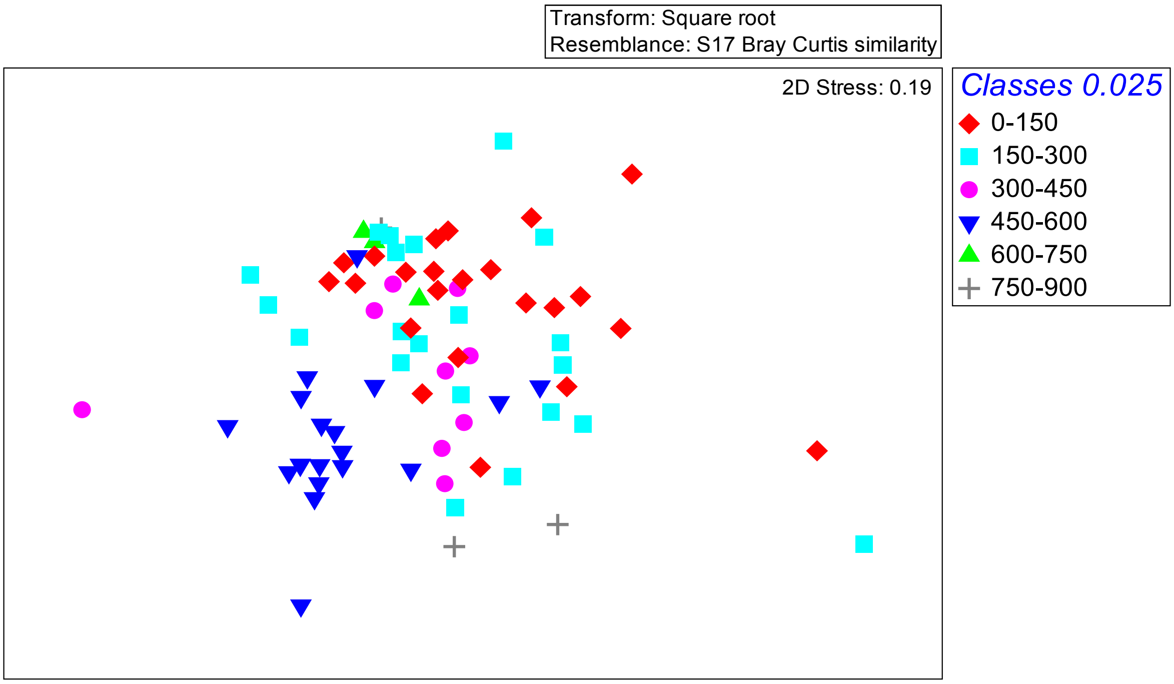


Figure 61. MDS plot for infaunal biomass data at Th overlaid with classed pressures scores derived from a VMS resolution of 0.025 decimal degrees.

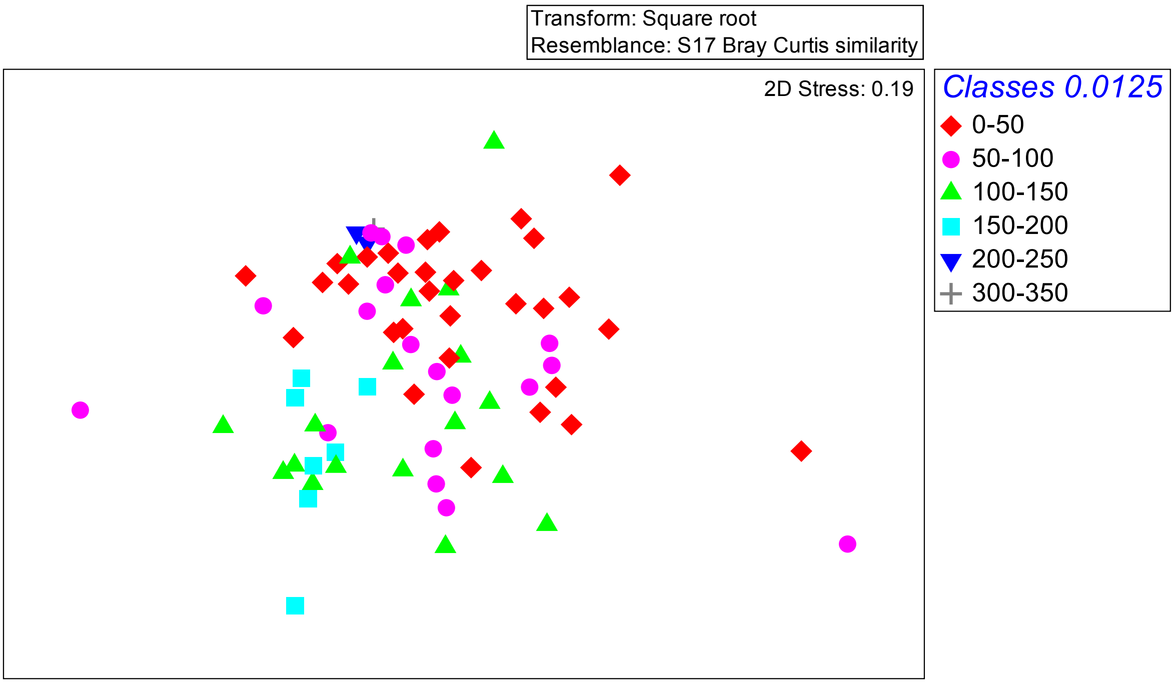


Figure 62. MDS plot for infaunal biomass data at Th overlaid with classed pressures scores derived from a VMS resolution of 0.0125 decimal degrees.

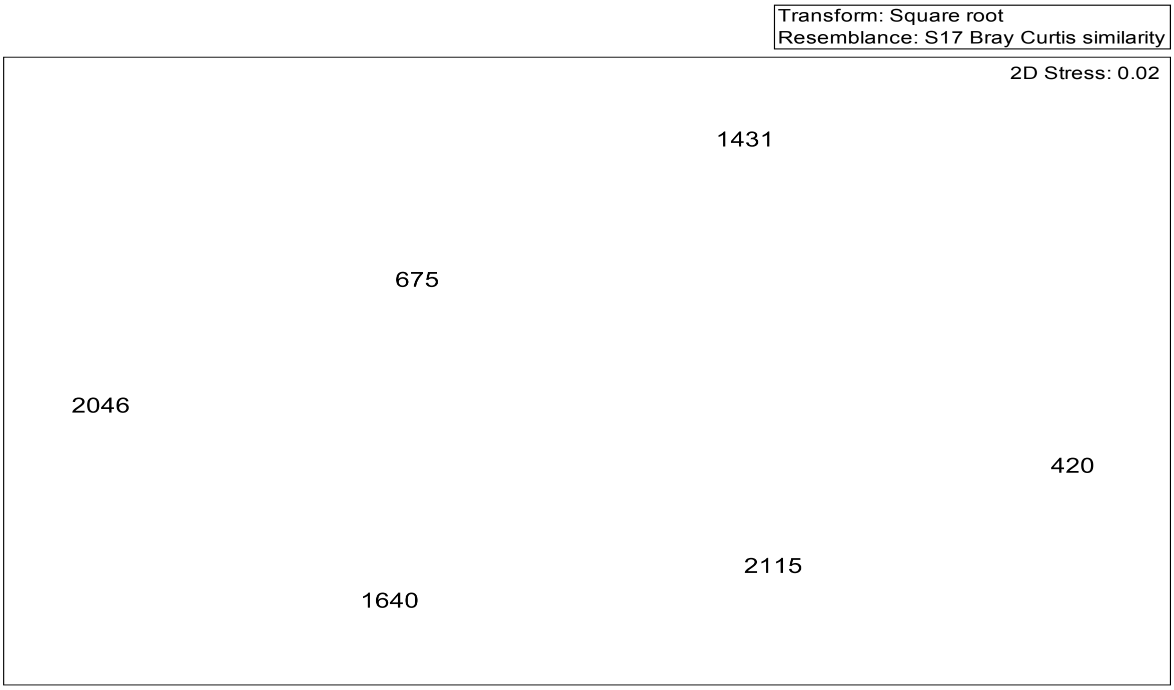


Figure 63. MDS plot for epifaunal biomass data at Th overlaid with pressures scores derived from a VMS resolution of 0.05 decimal degrees.

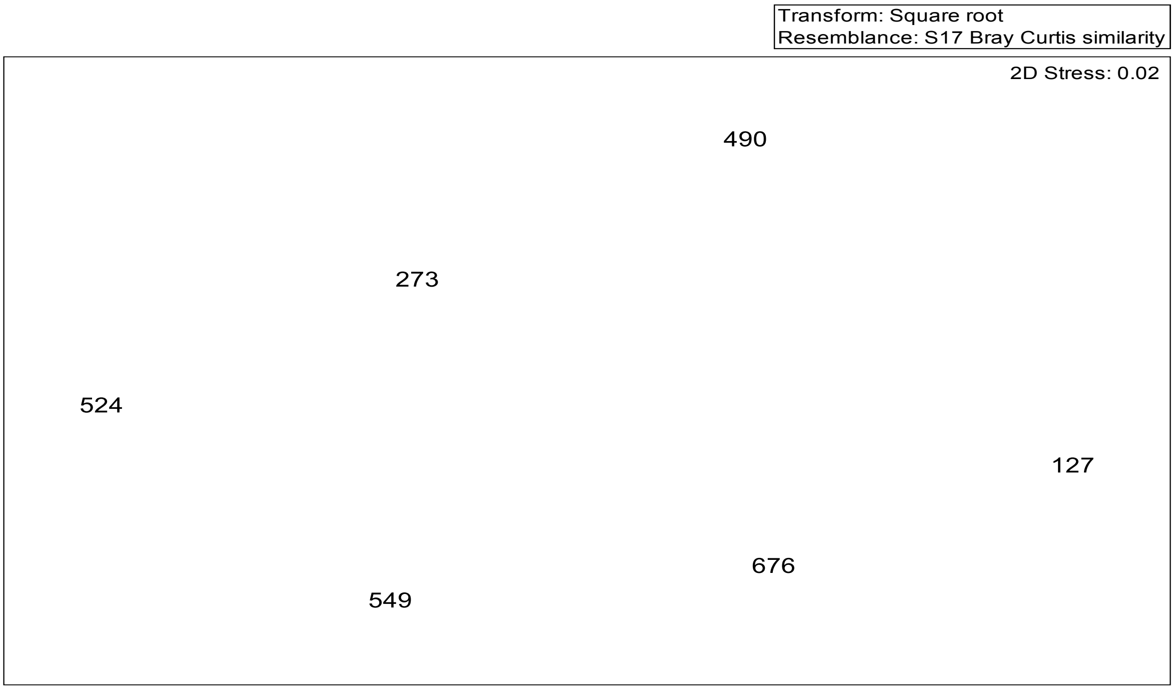


Figure 64. MDS plot for epifaunal biomass data at Th overlaid with pressures scores derived from a VMS resolution of 0.025 decimal degrees.

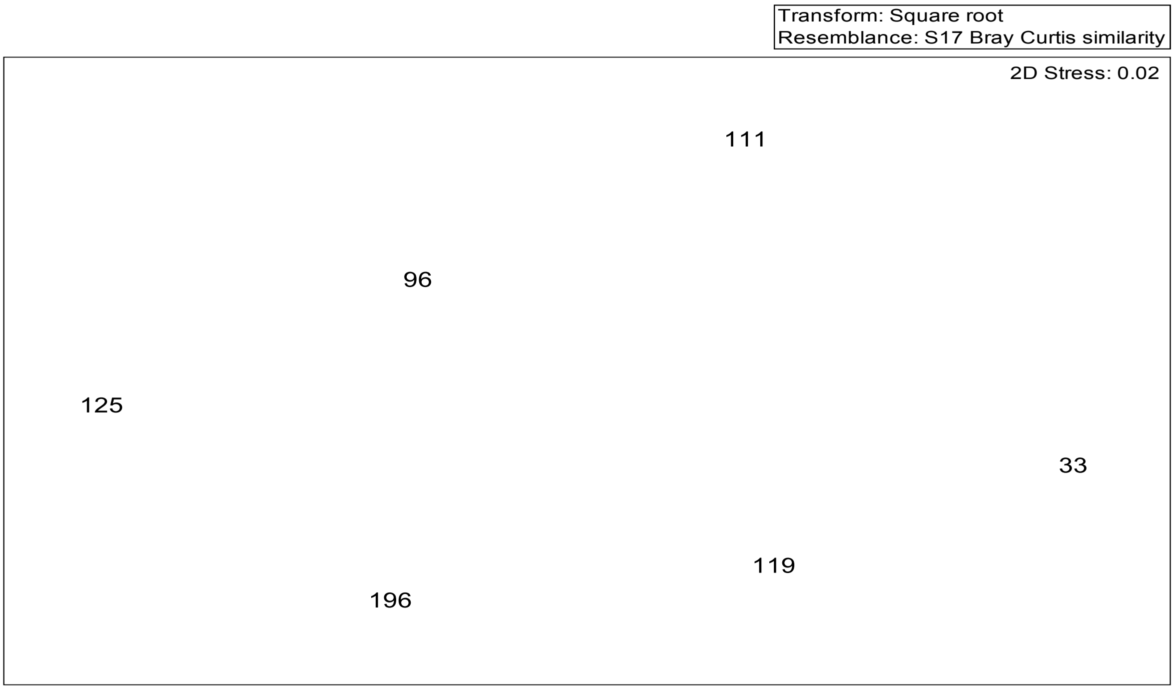


Figure 65. MDS plot for epifaunal biomass data at Th overlaid with pressures scores derived from a VMS resolution of 0.0125 decimal degrees.

6.6 ANOSIM test for Biomass data

6.6.1 Eastern English Channel

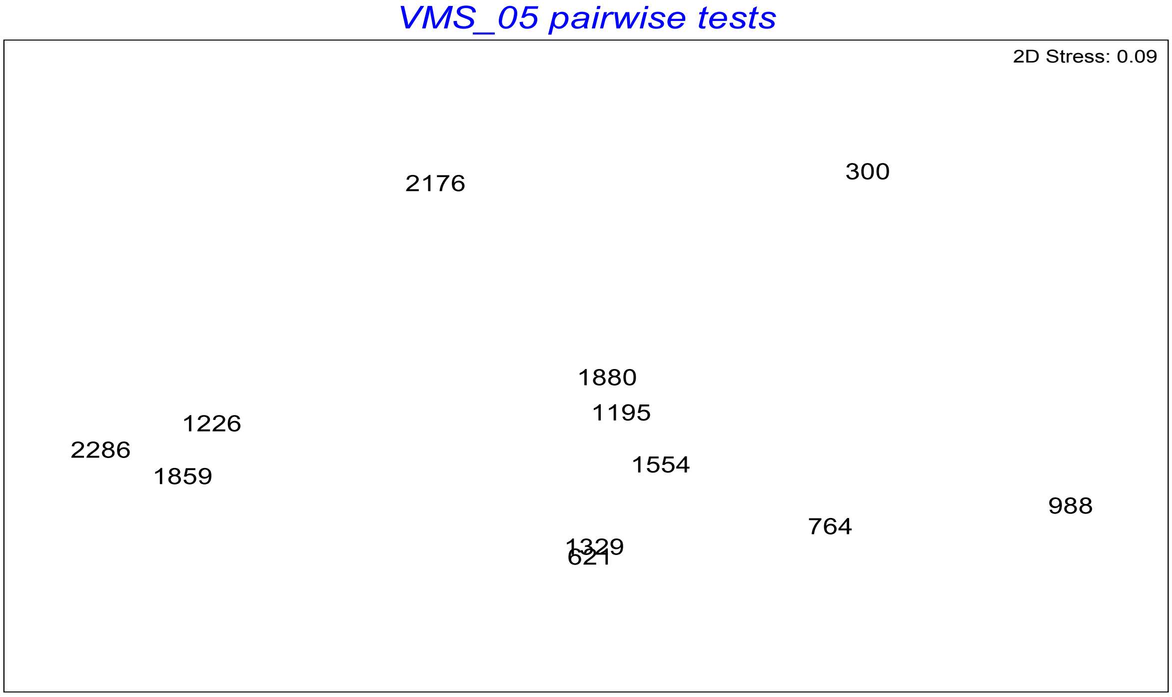


Figure 66. MDS plot for EEC R values for ANOSIM community biomass data between groups at a 0.05 grid resolution.

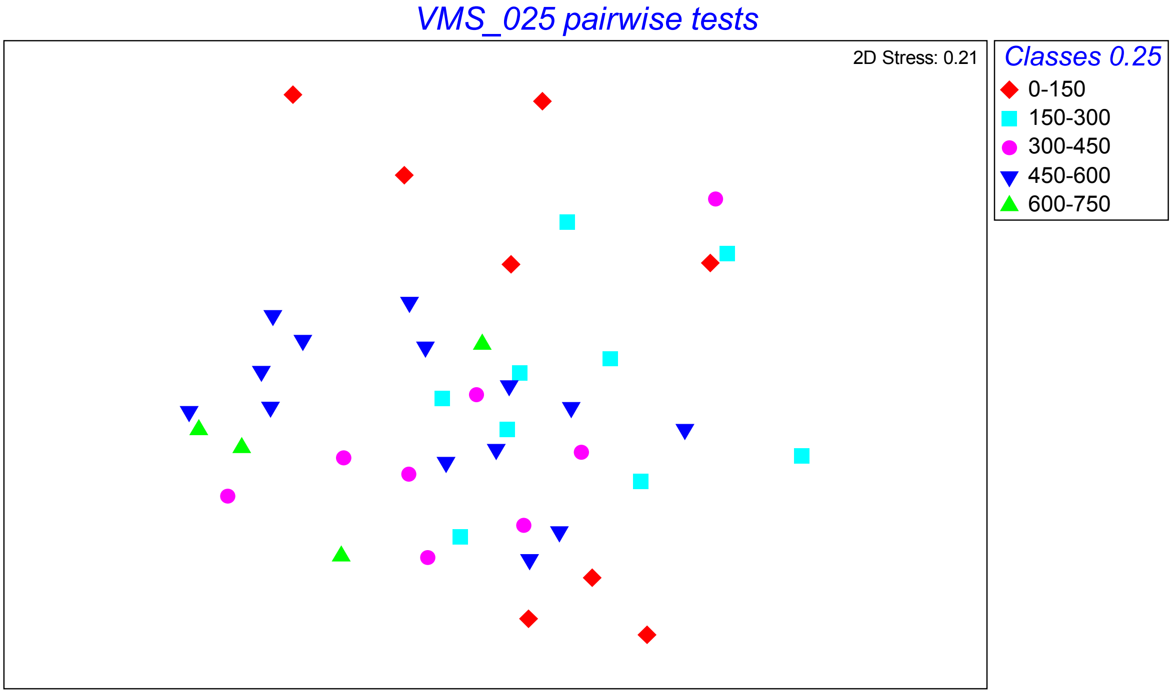


Figure 67. MDS plot for EEC R values for ANOSIM community biomass data between groups at a 0.025 grid resolution, and overlaid with class designations.

6.6.2 Outer Thames Estuary

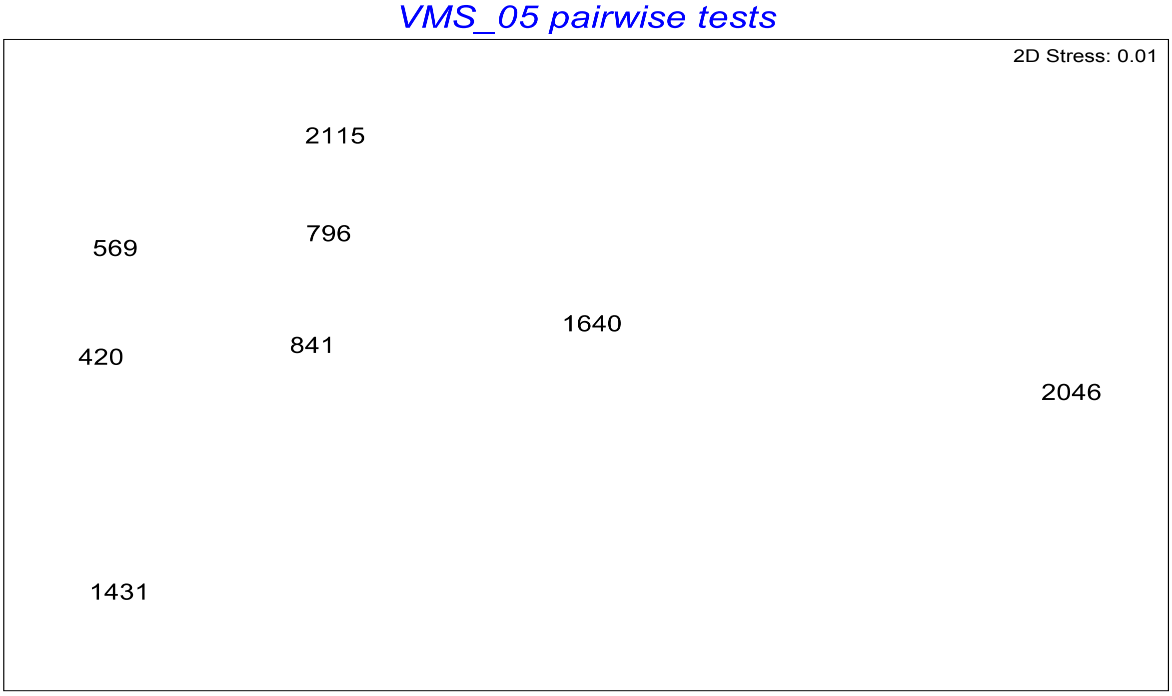


Figure 68. MDS plot for Th R values for ANOSIM community biomass data between groups at a 0.05 grid resolution.

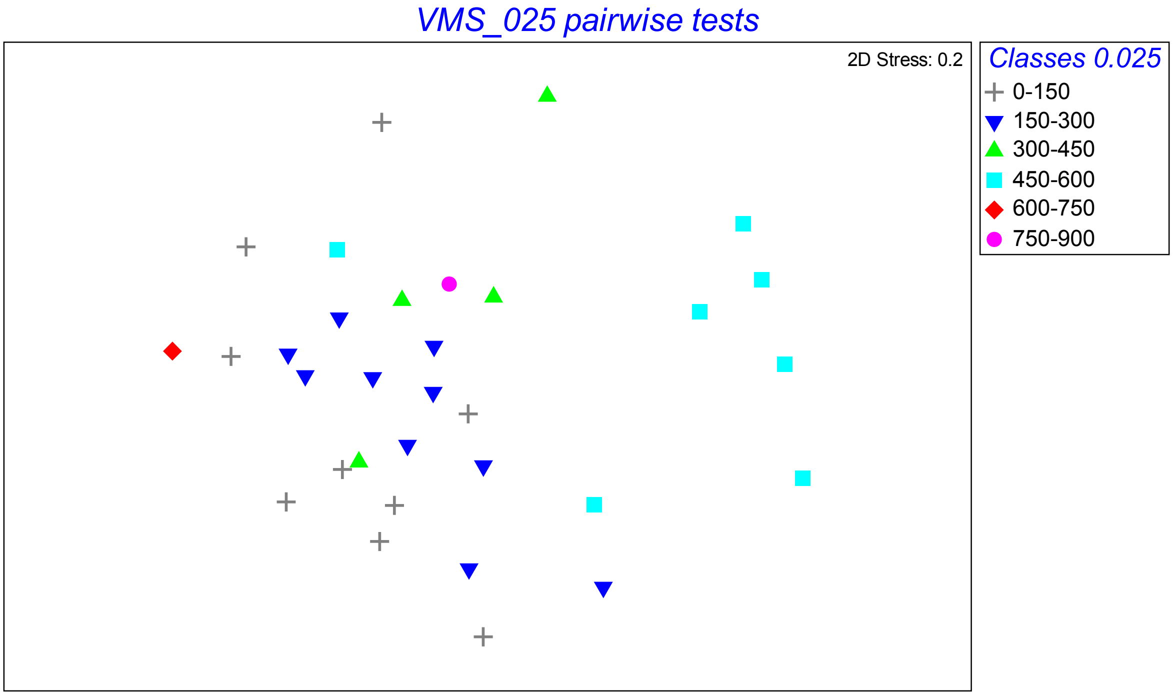


Figure 69. MDS plot for Th R values for ANOSIM community biomass data between groups at a 0.025 grid resolution, and overlaid with class designations.

6.7 BEST Analysis of Biomass Data

6.7.1 Eastern English Channel

Best Analysis for EEC

Variables
1 VMS_05
2 VMS_025
3 VMS_0125
4 Depth
5 %Gravel
6 %Sand
7 %Silt/clay
8 Dist From Shore (km)
9 Organic carbon (% m/m)
10 Nitrogen (% m/m)

Best results

No.Vars	Corr.	Selections
2	0.390	2,5
2	0.377	3,5
3	0.374	1,2,5
3	0.371	1,3,5
2	0.365	1,5
3	0.365	2,3,5
4	0.361	1-3,5
4	0.347	1,2,5,7
1	0.347	5
4	0.346	1,2,5,9

Best Analysis for EEC excluding VMS

Variables
1 Depth
2 %Gravel
3 %Sand
4 %Silt/clay
5 Dist From Shore (km)
6 Organic carbon (% m/m)
7 Nitrogen (% m/m)

Best results

No.Vars	Corr.	Selections
1	0.347	2
2	0.309	2,6
2	0.306	2,4
2	0.294	1,2
2	0.293	2,3
3	0.284	1,2,4
3	0.283	1,2,6
3	0.278	2-4
3	0.278	2,4,6
3	0.272	2,3,6

Figure 70. Primer output for BEST analysis for community biomass at EEC.

6.7.2 Outer Thames Estuary

Best Analysis for Th

Variables
1 VMS_05
2 VMS_025
3 VMS_0125
4 Depth
5 %Gravel
6 %Sand
7 %Silt/clay
8 Dist From Shore (km)
9 Organic carbon (% m/m)
10 Nitrogen (% m/m)

Best results

No.Vars	Corr.	Selections
5	0.284	2,5,7,8,10
5	0.278	1,5,7,8,10
5	0.276	2,5,8 -10
3	0.276	5,8,9
4	0.275	2,5,8,9
5	0.275	2,5,6,8,10
4	0.274	5,7,8,10
4	0.273	2,5,8,10
5	0.272	3,5,7,8,10
5	0.272	2,5,7 -9

Best Analysis for Th excluding VMS

Variables
1 Depth
2 %Gravel
3 %Sand
4 %Silt/clay
5 Dist From Shore (km)
6 Organic carbon (% m/m)
7 Nitrogen (% m/m)

Best results

No.Vars	Corr.	Selections
3	0.276	2,5,6
4	0.274	2,4,5,7
3	0.270	2,4,5
4	0.269	2,4 -6
5	0.266	2,4 -7

Figure 71. Primer output for BEST analysis for community biomass at Th.

6.8 SIMPER Analysis

Table 17. SIMPER analysis output for EEC and Th

Similarity Percentages - species contributions

One-Way Analysis

Data worksheet
Name: All Stns Transform (AB)
Data type: Abundance
Sample selection: All
Variable selection: All

Parameters
Resemblance: S17 Bray Curtis similarity
Cut off for low contributions: 80.00%

Group EEC
Average similarity: 16.25

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Echinocyamus pusillus	2.10	3.03	0.77	18.65	18.65
NEMERTEA	1.03	1.70	0.70	10.44	29.09
Nephtys cirrosa	0.46	0.90	0.33	5.57	34.65
Pseudonotomastus southerni	0.72	0.74	0.41	4.58	39.23
Aonides paucibranchiata	0.70	0.70	0.46	4.33	43.56
Ophelia borealis	0.43	0.67	0.25	4.12	47.68
Polycirrus	0.64	0.66	0.41	4.07	51.75
Glycera lapidum	0.68	0.63	0.41	3.87	55.62
Scoloplos armiger	0.50	0.53	0.25	3.26	58.88
Magelona filiformis	0.61	0.48	0.27	2.98	61.86
Poecilochaetus serpens	0.52	0.47	0.29	2.88	64.73
Spiophanes bombyx	0.29	0.33	0.25	2.05	66.79
Caulleriella alata	0.47	0.28	0.31	1.74	68.53
Bathyporeia elegans	0.31	0.27	0.18	1.69	70.22
Eulalia mustela	0.56	0.27	0.26	1.67	71.89
Glycera oxycephala	0.31	0.27	0.23	1.66	73.55
ACTINIARIA	0.50	0.27	0.24	1.64	75.19
Moerella pygmaea	0.32	0.25	0.21	1.51	76.70
Chaetozone zetlandica	0.28	0.24	0.19	1.48	78.18
Lumbrineris cingulata	0.38	0.22	0.23	1.38	79.56
Magelona johnstoni	0.40	0.17	0.16	1.06	80.62

Group TH
Average similarity: 11.67

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Nephtys cirrosa	0.61	3.87	0.47	33.18	33.18
Ophelia borealis	0.31	0.87	0.22	7.47	40.66
Moerella pygmaea	0.36	0.68	0.20	5.85	46.51
Spiophanes bombyx	0.43	0.68	0.26	5.79	52.30
Bathyporeia guilliamsoniana	0.26	0.65	0.20	5.60	57.90
Pseudonotomastus southerni	0.36	0.61	0.23	5.23	63.12
NEMERTEA	0.28	0.49	0.23	4.21	67.33
Urothoe brevicornis	0.26	0.45	0.14	3.87	71.20
Spisula elliptica	0.20	0.39	0.17	3.38	74.58
Lagis koreni	0.53	0.27	0.16	2.35	76.93
Polycirrus	0.18	0.24	0.13	2.07	79.01
Aonides paucibranchiata	0.17	0.24	0.10	2.06	81.07

Groups EEC & TH
Average dissimilarity = 90.96

Species	Group EEC Av.Abund	Group TH Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Echinocyamus pusillus	2.1	0.13	5.26	1.07	5.78	5.78
NEMERTEA	1.03	0.28	2.81	0.96	3.09	8.86
Nephtys cirrosa	0.46	0.61	2.72	0.71	2.99	11.85
Ophelia borealis	0.43	0.31	2.58	0.59	2.83	14.68
Pseudonotomastus southerni	0.72	0.36	2.45	0.75	2.69	17.37
Poecilochaetus serpens	0.52	0.22	2.01	0.6	2.21	19.58
Scoloplos armiger	0.5	0.04	2.01	0.53	2.21	21.79
Aonides paucibranchiata	0.7	0.17	2	0.69	2.2	23.99
Magelona filiformis	0.61	0.03	1.97	0.53	2.17	26.16
Moerella pygmaea	0.32	0.36	1.96	0.55	2.16	28.32
Polycirrus	0.64	0.18	1.95	0.68	2.14	30.46
Spiophanes bombyx	0.29	0.43	1.82	0.67	2	32.46
Magelona johnstoni	0.4	0.21	1.73	0.47	1.9	34.36
Glycera lapidum	0.68	0.08	1.69	0.72	1.86	36.22
Urothoe brevicornis	0.16	0.26	1.53	0.44	1.68	37.9
Bathyporeia elegans	0.31	0.13	1.48	0.51	1.62	39.52
Lagis koreni	0.11	0.53	1.38	0.46	1.51	41.04
Bathyporeia guilliamsoniana	0.17	0.26	1.36	0.5	1.5	42.54
Notomastus latericeus	0.33	0.41	1.36	0.53	1.5	44.04
Kurtiella bidentata	0.21	0.42	1.25	0.48	1.37	45.41
ACTINIARIA	0.5	0	1.23	0.47	1.35	46.76
Chaetozone zetlandica	0.28	0.11	1.22	0.46	1.34	48.1
Glycera oxycephala	0.31	0.13	1.21	0.53	1.33	49.44
Eulalia mustela	0.56	0.03	1.16	0.58	1.28	50.71
Lumbrineris cingulata	0.38	0.09	1.16	0.55	1.27	51.98
Caulleriella alata	0.47	0.09	1.12	0.61	1.23	53.21
Spisula elliptica	0.05	0.2	0.87	0.41	0.95	54.17
Spio filicornis	0.18	0.09	0.84	0.43	0.92	55.09
Scalibregma inflatum	0.34	0.09	0.83	0.51	0.91	56
Urothoe poseidonis	0.26	0.03	0.79	0.39	0.87	56.87
Pholoe baltica (sensu Petersen)	0.16	0.25	0.74	0.46	0.81	57.68
Bathyporeia tenuipes	0.22	0	0.67	0.31	0.74	58.41
Pista cristata	0.37	0	0.66	0.45	0.72	59.13
Syllis (Type H)	0.32	0	0.64	0.46	0.71	59.84
Syllis (Type E)	0.3	0	0.63	0.51	0.69	60.53
Abra prismatica	0.1	0.07	0.6	0.34	0.66	61.2
Moerella donacina	0.26	0	0.6	0.38	0.66	61.86
Eteone longa	0.14	0.12	0.58	0.45	0.63	62.49
Ophiura albida	0.08	0.2	0.56	0.37	0.62	63.11
Clymenura	0.23	0.01	0.56	0.41	0.61	63.73
Euspira pulchella	0.11	0.12	0.56	0.41	0.61	64.34
Bathyporeia gracilis	0.19	0	0.55	0.33	0.61	64.95
Mediomastus fragilis	0.22	0.13	0.55	0.4	0.61	65.56
Diplodonta rotundata	0.26	0	0.55	0.4	0.61	66.17
Abra alba	0.07	0.22	0.55	0.32	0.61	66.78
Eurydice pulchra	0.17	0	0.54	0.3	0.59	67.36
Spirobranchus triqueter	0.29	0.01	0.52	0.36	0.57	67.94
Nephtys hombergii	0	0.23	0.52	0.27	0.57	68.5
Lysilla nivea	0.18	0.01	0.51	0.32	0.56	69.07
Nucula hanleyi	0.23	0	0.5	0.41	0.55	69.62
Malmgrenia ljungmani	0.27	0.01	0.5	0.46	0.55	70.17
Lumbrineris	0.14	0.03	0.49	0.34	0.54	70.72
Ampelisca brevicornis	0.16	0	0.49	0.34	0.54	71.26
Podarkeopsis capensis	0.1	0.12	0.48	0.39	0.53	71.78
Grania	0.18	0.02	0.47	0.32	0.52	72.3
Phoronis	0.19	0.03	0.46	0.35	0.51	72.81
Polynoidae	0.2	0.01	0.45	0.4	0.49	73.3
Hesionura elongata	0.15	0	0.44	0.27	0.49	73.79
Schistomeringos neglecta	0.2	0	0.43	0.38	0.47	74.26
Apseudes latreillii	0.25	0.03	0.41	0.19	0.46	74.72
Abludomelita obtusata	0.01	0.09	0.38	0.21	0.41	75.13
Protodorvillea kefersteini	0.17	0.01	0.34	0.36	0.37	75.5
Leucothoe incisa	0.13	0	0.34	0.28	0.37	75.87
Echinocardium cordatum	0.08	0.03	0.33	0.28	0.36	76.23
Sphaerosyllis taylori	0.08	0.06	0.32	0.26	0.35	76.58
Ophiothrix fragilis	0.21	0	0.32	0.18	0.35	76.93
Chaetozone christiei	0.1	0.01	0.31	0.3	0.35	77.28
Polygordius	0.18	0	0.31	0.33	0.34	77.61
Syllis (Type D)	0.18	0	0.3	0.34	0.34	77.95
Goodallia triangularis	0.06	0.04	0.29	0.24	0.32	78.27
Glycymeris glycymeris	0.16	0	0.29	0.34	0.32	78.58
Thia scutellata	0.07	0.03	0.29	0.26	0.31	78.9
Pseudomystides limbata	0.15	0	0.28	0.3	0.31	79.21
Ensis arcuatus	0.04	0.03	0.28	0.2	0.31	79.52
Malmgrenia darbouxi	0.1	0.06	0.27	0.36	0.3	79.82
Marphysa bellii	0.11	0.03	0.27	0.32	0.3	80.12

6.9 Particle Size Analysis

Eastern English Channel

Six sediment groups were determined as the best group output from Entropy. The optimum number of clusters is achieved when the Calinski–Harabasz (C–H) statistic is at its maximum (Orpin & Kostylev, 2006). In addition to this statistic, expert judgement meant that in some cases where groups were sufficiently similar, they are considered to be the same group, and suffixed with an 'a' or a 'b' to show original grouping. Sediment characteristics and histograms for each of these final groups (5 groups, group 1 being split into a and b) are given in Figure 75 and Table 16.

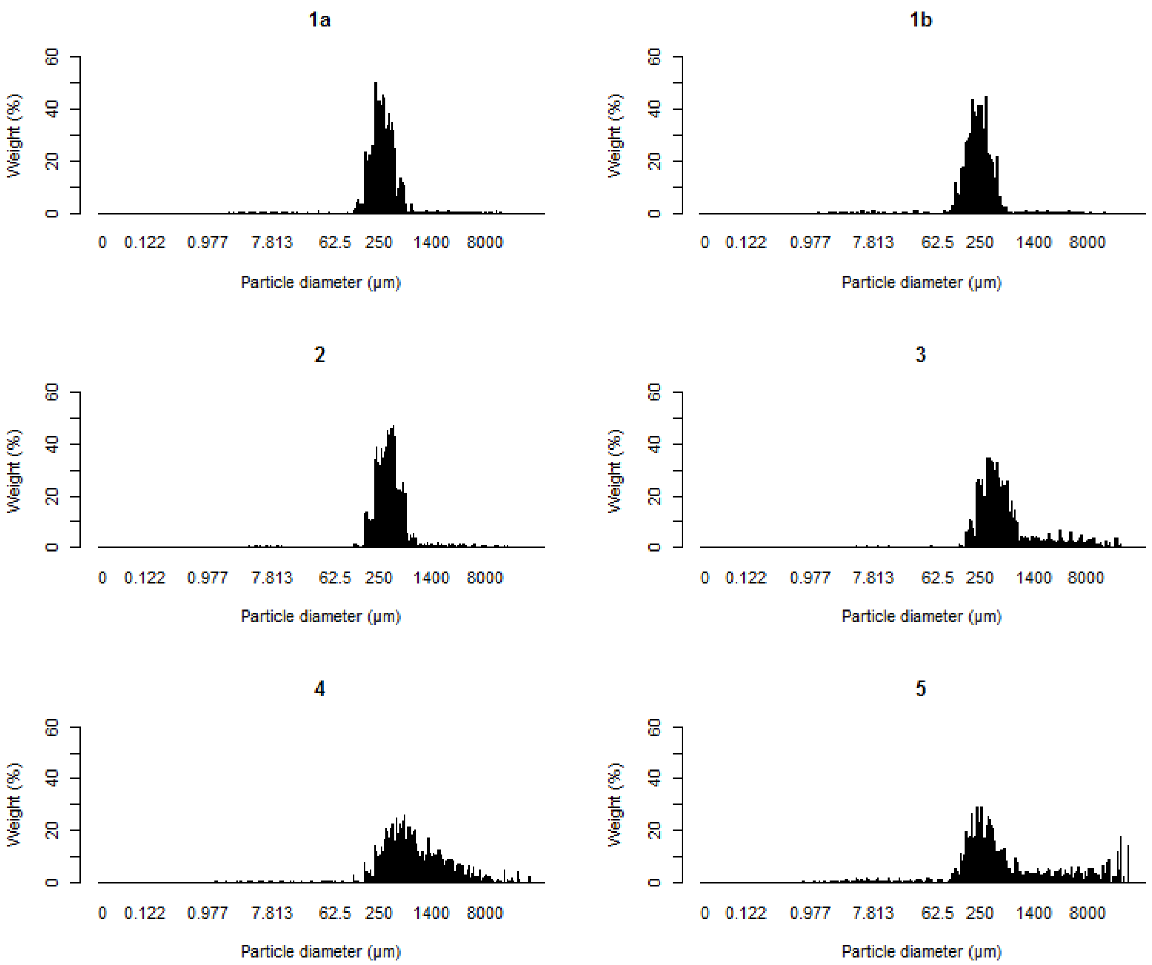


Figure 72. Particle size distribution histograms for each sediment group. All samples are represented within the histograms.

Table 18. Sediment characteristics of the six sediment groups determined for EEC PSA results, produced on the average particle size distribution for each sediment group, produced with Gradistat (Blott & Pye, 2001).

Sediment group	Number of samples	Sample type	Sediment description	Mode 1 (µm)	Mode 2 (µm)	Mode 3 (µm)
1a	18	Unimodal, Moderately Well Sorted	Slightly Gravelly Sand	301.8		
1b	12	Unimodal, Moderately Well Sorted	Slightly Gravelly Sand	301.8		
2	30	Unimodal, Moderately Well Sorted	Slightly Gravelly Sand	426.8		
3	20	Unimodal, Poorly Sorted	Gravelly Sand	426.8		
4	25	Unimodal, Poorly Sorted	Gravelly Sand	603.55		
5	15	Unimodal, Very Poorly Sorted	Gravelly Sand	301.8		

Sediment group	Gravel (%)	Very coarse sand (%)	Coarse sand (%)	Medium sand (%)	Fine sand (%)	Very fine sand (%)	Silt/clay (%)
1a	0.58	0.37	7.32	70.59	20.41	0.00	0.73
1b	0.59	0.27	2.94	53.25	41.25	0.63	1.08
2	1.71	0.87	18.69	69.69	8.66	0.00	0.38
3	11.09	4.61	29.53	48.80	5.56	0.01	0.40
4	18.16	17.34	35.15	25.50	2.86	0.09	0.90
5	20.13	5.21	12.70	37.57	19.17	1.04	4.17

Outer Thames Estuary

Eight sediment groups were determined as the best group output from Entropy. The optimum number of clusters is achieved when the Calinski–Harabasz (C–H) statistic is at its maximum (Orpin & Kostylev, 2006). In addition to this statistic, expert judgement meant that in some cases where groups were sufficiently similar, they are considered to be the same group, and suffixed with an ‘a’ or a ‘b’ to show original grouping. Sediment characteristics and histograms for each of these final groups (4 groups, group 2 being split into a and b, group 3 split into a, b and c and group 4 being split into a and b) are given in Figure 76 and Table 17.

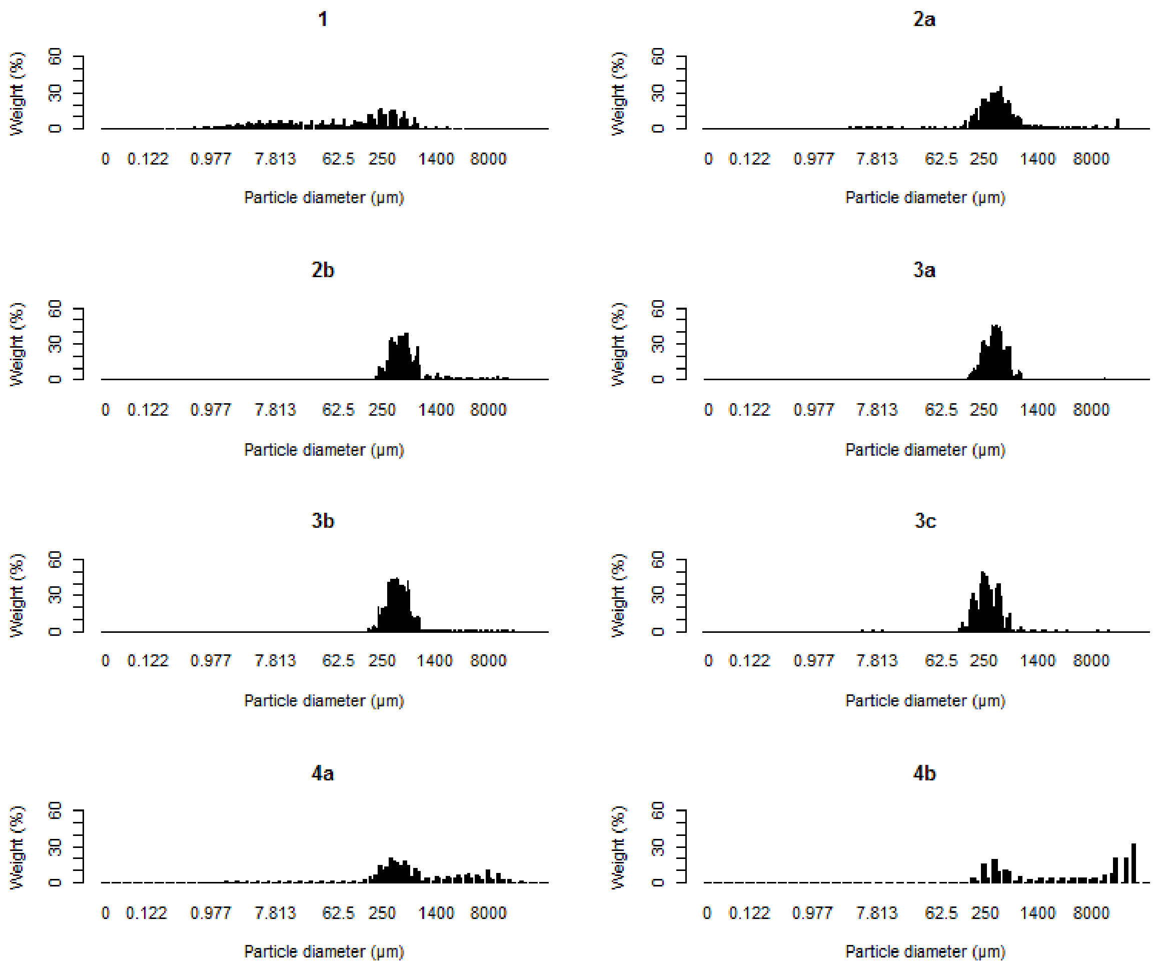


Figure 73. Particle size distribution histograms for each sediment group. All samples are represented within the histograms.

Table 19. Sediment characteristics of the six sediment groups determined for Th PSA results, produced on the average particle size distribution for each sediment group, produced with Gradistat (Blott & Pye, 2001).

Sediment group	Number of samples	Sample type	Sediment description	Mode 1 (µm)	Mode 2 (µm)	Mode 3 (µm)
1	5	Trimodal, Very Poorly Sorted	Slightly Gravelly Muddy Sand	426.8	6.6685	53.345
2a	12	Unimodal, Moderately Sorted	Gravelly Sand	426.8		
2b	8	Unimodal, Moderately Sorted	Gravelly Sand	603.55		
3a	15	Unimodal, Moderately Well Sorted	Slightly Gravelly Sand	426.8		
3b	24	Unimodal, Well Sorted	Slightly Gravelly Sand	426.8		
3c	11	Unimodal, Moderately Well Sorted	Slightly Gravelly Sand	301.8		
4a	3	Trimodal, Very Poorly Sorted	Gravelly Sand	426.8	4800	9600
4b	2	Polymodal, Very Poorly Sorted	Sandy Gravel	38250	426.8	19200

Sediment group	Gravel (%)	Very coarse sand (%)	Coarse sand (%)	Medium sand (%)	Fine sand (%)	Very fine sand (%)	Silt/clay (%)
1	2.09	1.28	11.82	23.42	12.41	6.96	42.02
2a	5.72	2.99	28.50	49.17	9.53	0.38	3.70
2b	5.74	3.89	51.33	38.37	0.67	0.00	0.00
3a	0.90	0.40	27.71	65.54	5.19	0.00	0.26
3b	0.87	0.69	43.32	53.64	1.48	0.00	0.00
3c	0.30	0.14	7.93	69.48	22.05	0.02	0.09
4a	28.79	6.28	24.33	29.90	5.00	0.85	4.86
4b	56.82	3.37	12.36	23.19	2.04	0.08	2.16

7 Actions from independent peer review by Dr Gwladys Lambert

7.1 Presentation of fishing effort distribution

GL: Study how fishing effort distribution changes from year to year. Without information on this, any attempt to monitor based on a fishing gradient will be difficult to understand.
In future surveys, VMS data from 2006, when the systems were widely taken up, to present will be used to examine historic intensity levels and changes in spatial distribution over time. This has already been done for both the Fladen Ground NC MPA and Dogger Bank SCI MPA monitoring R&D surveys in 2014 to put results into historical context.

Maps of fishing effort from 2006-11 have been added as an Appendix.

GL: How does this study relate to seabed monitoring? I would recommend areas where there is always a lot of fishing vs. areas where there is hardly ever any fishing so you can see how both areas change over time
This approach forms part of the Monitoring Strategy developed for the UK Marine Biodiversity Monitoring R&D Programme. The Strategy is divided into three components: Type 1 which considers long-term change over time, Type 2 which considers change along gradients of pressure, and Type 3 which uses manipulative studies to test particular hypotheses. This study followed principles set out in Type 2 monitoring to investigate whether bottom-contact fishing had an effect on a particular habitat. Being that the analysis was correlative and could not prove causality, any identified relationships would need to be tested with a BACI-type experiment to see whether seabed communities begin to recover once fishing pressure is removed.

GL: It would be interesting to show how many 12 to 15m vessel fish outside 12nm compared to how many >15m since the data exist since 2012
We intend to analyse the additional information provided by including vessels from 12-15m in the VMS scheme in the future to identify how the effort is distributed. Unfortunately, the uptake of VMS by 12-15m vessels to date has not been comprehensive.

GL: What methods are available to map inshore fishing effort?
In the past overflight and patrol vessel sightings have been used to map inshore fishing. These were biased towards sampling effort and were not undertaken in a standard way between regions. More recently, Fishermap attempted to map inshore fishing effort in English waters for the MCZ Project using interviews with fishers and similar initiatives in Wales and Scotland have been piloted, but these are also patchy. At present, there are a number of initiatives to try and map inshore fishing activities using satellites, including iVMS, and this shows much promise for the future. To be most useful to a monitoring programme, effort data would need to be provided periodically in a standard format. Invariably, methods of assessing <12m effort are different between devolved administrations and, at present, there is no reliable way of standardising these across boundaries. Furthermore, most are one-off exercises that do not provide a process for easily obtaining data in the future.

GL: What is the best unit to express fishing pressure in?
In this paper fishing activity is considered to contribute to the pressure defined by OSPAR as “Physical Damage (Reversible Change) – penetration and/or disturbance of the substrate below the surface of the seabed, including abrasion”. Specifically in relation to this abrasion pressure, it was considered most apposite to use swept area to represent the extent of the pressure.

GL: Why not present grids in kms?
By using decimal degrees, grid cells are then easily nestable within ICES statistical rectangles and can subsequently be related to landings data. In order to address the issue of the slight change in

grid size across UK waters, the swept area will be presented weighted to the area of the grid cell in the future.

GL: How are natural fluctuations in benthic communities taken into consideration?
There is ongoing work within JNCC to look at the effects of effects on benthic habitats against a backdrop of natural variability. In particular, an MMO contract has been let to look at our ability to detect the effects of mobile fishing on mobile sediments using the data that has been collected and archived in Marine Recorder.

GL: Why is swept area not presented as area fished per yr or % area fished per year?
Here, the layers were still under development and, whilst it would make it easier to consider the magnitude of the effort to present the layers in this way, it was felt at the time that they could also be misconstrued. For instance, a cell with a score of 100% for 2009 means that an area equivalent to the area of the cell has been swept within the cell that year; which could mean half of the cell has been swept twice, or a quarter has been fished four times. Our more recent pressure layers have all been presented as ratios per year though, with clear accompanying text explaining any assumptions.

7.2 Experimental design

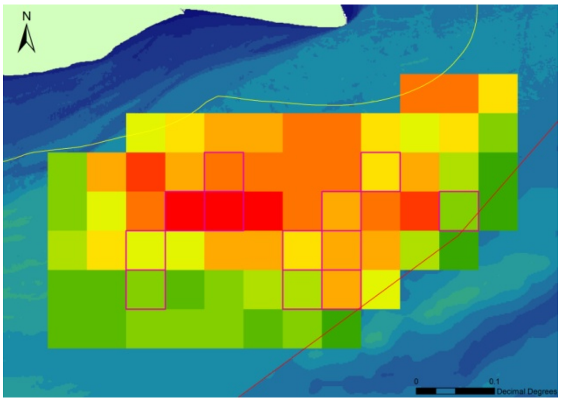
GL: Your samples within grid cells are not truly independent. Mixed models should be used throughout the study with block being the random factor. It could even be nested within site if both sites were integrated in the same model. This is very important, even when it comes to splitting the blocks into smaller (non-independent) units. Blocks should be a random factor, this will most likely modify the resulting p-values.
For this survey, under advice from Cefas statisticians, the samples were considered to be independent. However, this is something that is being looked at for future surveys as it will fundamentally change the way in which the surveys are planned and experiments designed.

GL: Why not use Akaike Information Criterion (AIC) for model selection?
The Mean Relative Absolute Error (MRAE) is an alternative method of model selection and a common method of comparing forecasts with outcomes. It was recommended to us over AIC in this instance by Cefas statisticians. In the future, however, we would consider using AIC if it makes the results clearer to the wider scientific community, particularly marine ecologists who more commonly use AIC or Hierarchical Partitioning for model selection.

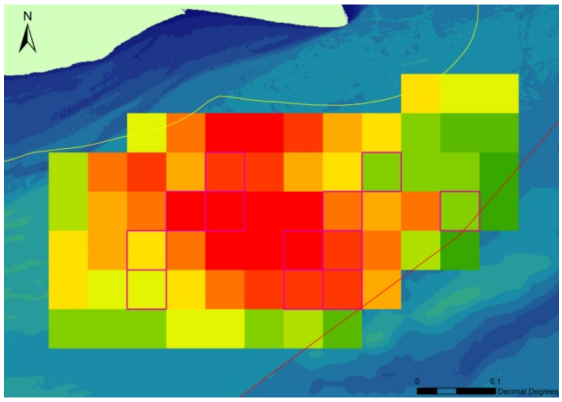
GL: I am not sure how to deal with the "pseudoreplicate" nature of the design in multivariate analysis - probably an analysis of spatial autocorrelation could give an answer - if the samples are not spatially auto-correlated maybe there is no major issue. For instance relative similarity between all pairs could be estimated then a geostatistical variogram could be used to study spatial autocorrelation.
We have passed this recommendation on to the Cefas statisticians who are advising on future survey designs for advice.

8 Maps of historic fishing effort in the survey areas

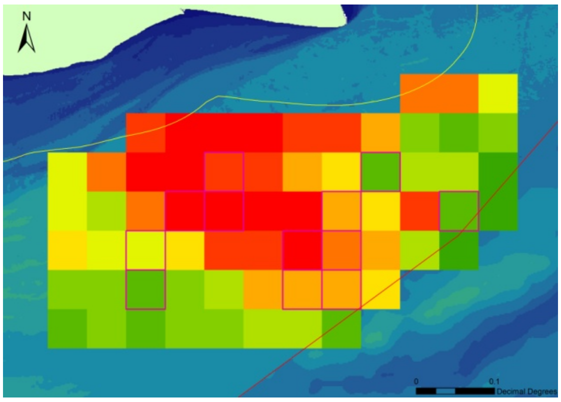
2006



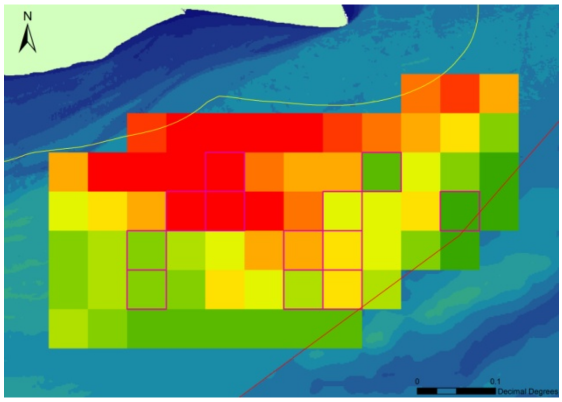
2007



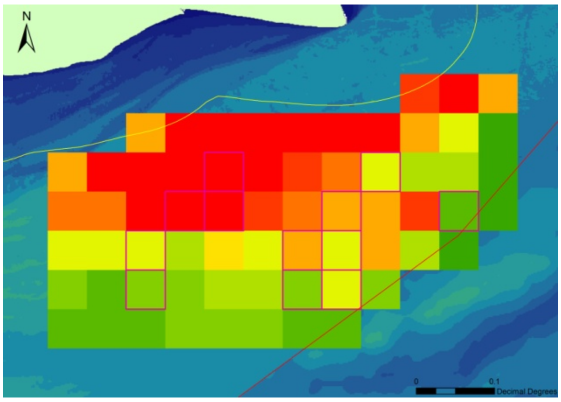
2008



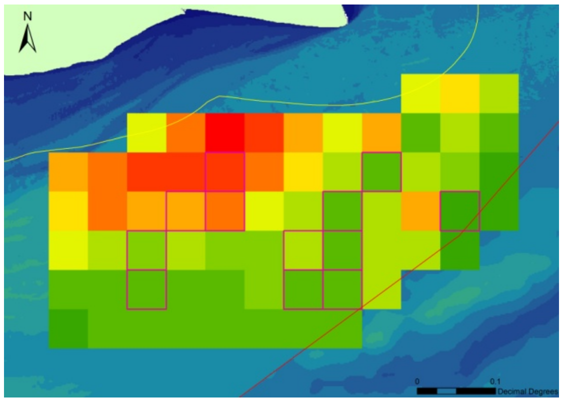
2009



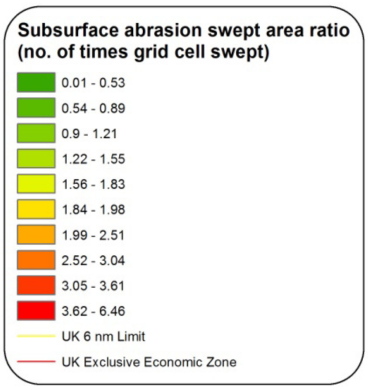
2010

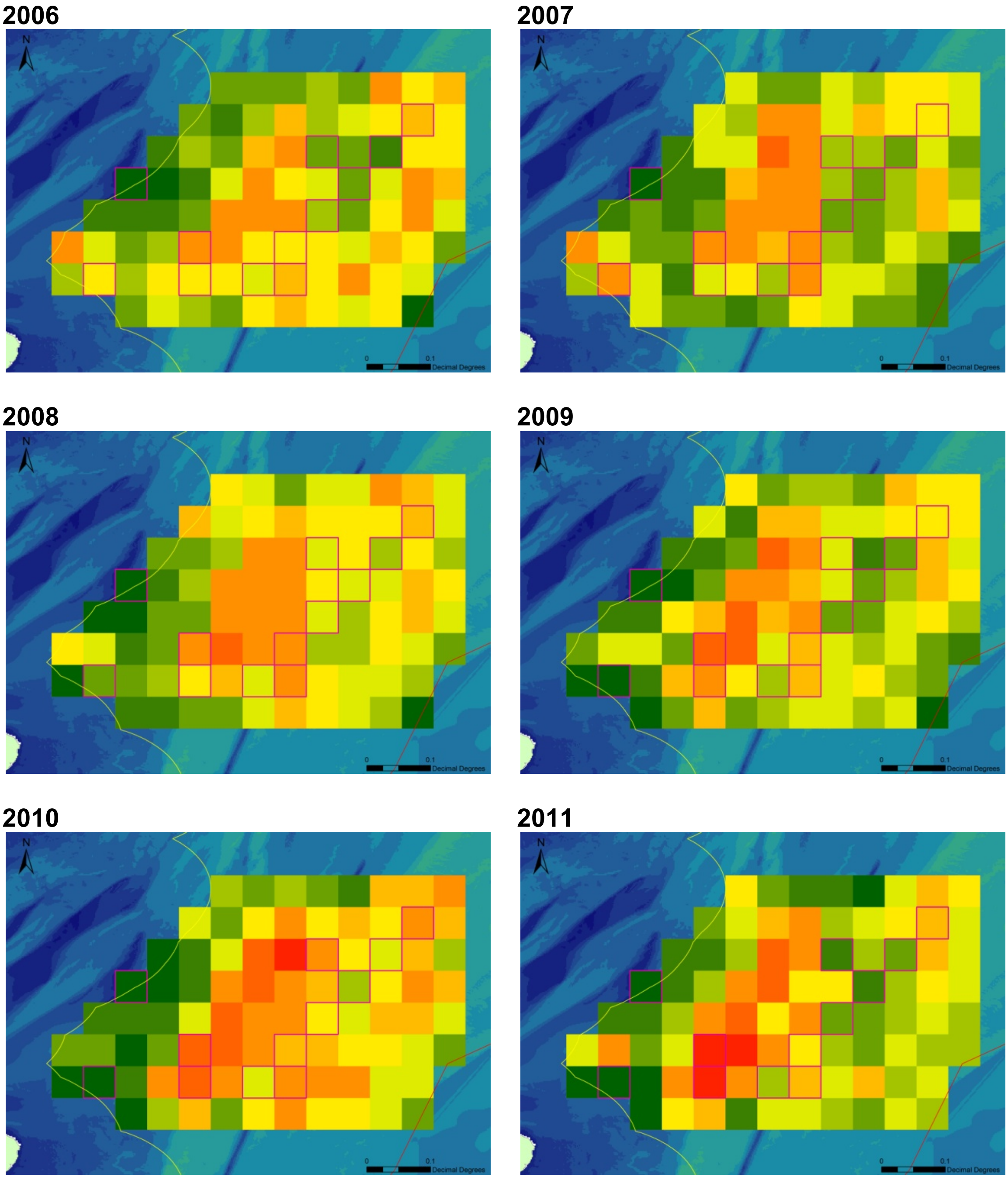


2011

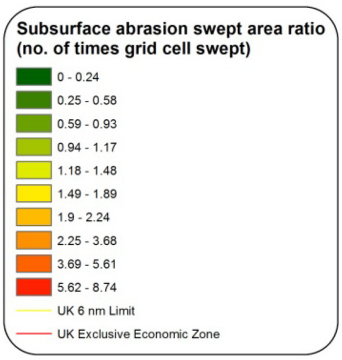


Historical subsurface abrasion pressure at the Eastern English Channel site estimated from VMS data and vessel logbooks that demonstrates change in fishing effort over time. Highlighted grid cells delineate cells that were sampled within this study.





Historical subsurface abrasion pressure at the Outer Thames Estuary site estimated from VMS data and vessel logbooks that demonstrates change in fishing effort over time. Highlighted grid cells delineate cells that were sampled within this study.



Cefas

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JNCC/Cefas Partnership Report Series. *Developing appropriate, and ecologically significant, sampling units for broad scale monitoring of fisheries activity using up to date Vessel Monitoring System data*, **No. 1**. Jenkins, C., Nelson, M., Whomersley, P., Johnson, G., Cameron, A., Barry, J., Eggleton, J., Church, N & Webb, K. 2015. ISSN 2051-6711