



# Trawling for evidence: An ecosystem-based multi-method trawling impact assessment

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## ABSTRACT

Robust methods for assessing fisheries impacts are required to support evidence-based management and contribute to the assessment of protected areas. Under marine policies, such as the Marine Strategy Framework Directive, effective management requires an understanding of impacts at the fisheries scale and needs to account for the responses of various benthic components. This study investigated the impacts of trawling for Nephrops and whitefish along the Northeast England coast, comparing, and assessing benthic communities and habitats at muddy sediment sites across a gradient of fishing pressure. An ecosystem-based sampling approach assessed macrofauna, meiofauna, and sediment characteristics by combining traditional and innovative methods, conducting a balanced assessment of the north-east England trawl fishery. Combining multiple methods effectively highlighted contrasting trends between infaunal size classes. Significant negative impacts of trawling pressure were observed on macrofauna and meiofauna richness, diversity, and community composition, and sediment compaction and oxygenation. Key metrics were comparatively assessed for their suitability for the detection of benthic disturbance, with recommendations discussed in relation to future monitoring.

## 1. Introduction

Bottom fishing is the dominant anthropogenic activity occurring on the seafloor globally (Halpern et al., 2008). It is by far the most widespread, with a much larger and dispersed footprint than other activities (e.g. mining, sand and gravel extraction, and spoil disposal) (Foden et al., 2011). Trawling is a type of mobile bottom fishing where large, weighted nets are dragged along the seabed, efficiently catching a wide variety of species including crustaceans, bivalves, and bottom-dwelling fish depending on the structure of the gear (Eigaard et al., 2017; Watson et al., 2006). The North Sea has been trawled for centuries (Kerby et al., 2012), and it is estimated that over 60% of the seabed under 200 m depth is actively trawled (Eigaard et al., 2017).

The assessment of trawl fisheries and its impacts has become increasingly important in recent years, with a diverse range of policy drivers from international to national and regional levels (e.g. United Nations Convention on biological diversity (United Nations, 1992), the OSPAR Convention (OSPAR, 1992), and EU Habitats Directive

(European Commission, 1992)). At a European level, the Marine Strategy Framework Directive (MSFD) requires member states to assess predominant pressures and impacts. Descriptor 6, Seabed Integrity, states that benthic ecosystems must be safeguarded, and not adversely affected by human pressures (Directive, 2008/56/EC, 2008). At the local level, the UK Marine Strategy Regulation 2010 requires the delivery of MSFD along with the Fisheries Act ecosystem objective. The development of ecosystem-based fisheries management requires knowledge on the distribution and impacts of trawling, and the status of effected benthic habitats and species (Pikitch et al., 2004). Assessments of ecological status allow managers to develop informed management plans, set priorities for habitat protection, and achieve a socio-economic-environmental balance (Hiddink et al., 2017; McConaughy et al., 2020; Rice, 2005). At the local level, assessments for individual Marine Protected Areas (MPAs) are important if designated features are to be protected to meet conservation objectives (Clark et al., 2017).

Direct effects of trawling are caused by physical contact with the

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seafloor, leading to surface abrasion by scraping and ploughing, sediment penetration, destruction of benthic features, reduced structural complexity, and sediment mobilisation (Grieve et al., 2014; Jones, 1992; Rijnsdorp et al., 2015). For benthic communities, these often result in: a reduction in the abundance, richness, and diversity, diminished biomass, altered community structure, function and productivity (Collie et al., 2000; Hiddink et al., 2017; Jennings et al., 2001; Kaiser et al., 2006; Kaiser et al., 2002; Kaiser and Spencer, 1996; Tillin et al., 2006). In addition, there is growing concern over the impact on blue carbon stores (Black et al., 2022; Sala et al., 2021). Effects vary significantly between different trawl fisheries, influenced by gear design (e.g., otter or beam), target species, trawl frequency and intensity, target habitat, natural disturbance regularity, and benthic community composition (susceptibility and life histories) (Jones, 1992; Kaiser et al., 2002; McConnaughey et al., 2020). It is generally believed that structurally complex habitats such as biogenic reefs, and those with low levels of natural disturbance such as mud and deep-water habitats are most vulnerable to benthic disturbance, taking the longest time to recover (Bolam et al., 2014; Hiddink et al., 2019; Kaiser et al., 2002). Complex relationships between the fishery particulars and the impacts on species and habitats observed, combined with a patchy and dynamic distribution of trawling activity (Amoroso et al., 2018; Hiddink et al., 2017) make it difficult to provide generalised impact predictions.

Assessments of trawling impacts are often carried out experimentally, with discreet areas of seafloor subjected to predetermined trawling intensities, sampled before and after disturbance, by tracking changes in benthic communities (e.g. Kaiser et al., 1996; Schratzberger et al., 2002; Tuck et al., 1998). Whilst experimental studies provide exact data on the disturbance regime and ensure that samples are collected directly in the trawl path, their acute and short-term nature fails to reflect real fisheries and the long-term chronic disturbance they cause (Løkkeberg, 2005). A recent shift to more comparative assessments (e.g. Gislason et al., 2017; Pommer et al., 2016), to investigate communities across a gradient of actual fishing pressures, better accounts for the spatial extent, temporal variability, and frequency of fishing activity, and results in more representative fishery level findings (Hiddink et al., 2017). However, it is important to acknowledge that fishing gradient studies can have additional obstacles to overcome, due to the possibility of underlying environmental gradients driving fisheries distribution.

Many comparative studies use spatial data derived from satellite-based vessel monitoring systems (VMS) to assign fishing pressure levels to sampling areas (e.g. Eigaard et al., 2017; Pommer et al., 2016). VMS provide high-resolution real-time data for fishing vessels over 12 m (European Commission, 2009; Lee et al., 2010), and similarly AIS (Automatic Identification System) is compulsory for vessels over 300 tonnes (Jiang et al., 2016; Shepperson et al., 2017). But the UK fishing fleet consists of a high proportion of smaller vessels (MMO, 2019), particularly prosecuting inshore waters. An alternative method to account for smaller vessels is to use records of trawlers sighted by fishery patrols (Breen et al., 2014; Jennings et al., 2001), which are undertaken by the regional Inshore Fisheries and Conservation Authorities (IFCAs) within UK inshore waters.

Most trawling impact studies, and wider marine monitoring of sediment, focus on macro-infaunal communities as the biological component (Bolam et al., 2014; Collie et al., 2000; Hiddink et al., 2017; JNCC, 2001; 2016). Multimetric macrobenthic indices are increasingly used to interpret macrofaunal community data (Borja et al., 2000; Kennedy et al., 2011; Simbora and Zenetos, 2002), but whilst significant evidence exists for associations with chemical pressures, relatively few studies have assessed responses from physical pressures such as trawl disturbance (Gislason et al., 2017). Additionally, despite the desire to move to an ecosystem-based approach to management and monitoring, the response from meiofauna to trawling disturbance is often not included in impact assessments. Meiofauna are ubiquitous and key components in benthic communities (Brannock et al., 2018; Michaela Schratzberger and Ingels, 2018), but their high density, small size, and

taxonomic intractability, makes traditional methods of identification difficult, time consuming, and expensive (Bhadury et al., 2006; Chariton et al., 2015; Cordier et al., 2017; Danovaro et al., 2016). Molecular tools offer an alternative approach to efficiently analyse meiofaunal communities (Bhadury et al., 2006; Carugati et al., 2015; Schratzberger and Ingels, 2018), environmental DNA (eDNA) metabarcoding showing particular promise (Bijleveld et al., 2018; Rees et al., 2014).

Rapid and less destructive sampling methods used to assess sediment communities include Sediment Profile Imagery (SPI). SPI offers an alternative to macrofauna and meiofauna sampling, where coarse ecological data is gained from images of the sediment surface and internal profile, with no fauna extracted from the system (Birchenough et al., 2013; Birchenough et al., 2010; Germano et al., 2011; O'Reilly et al., 2006). Parameters extracted from the images are combined into indices such as the Benthic Habitat Quality index or the Organism-Sediment Interaction index (H. Nilsson and Rosenberg, 2006; Rhoads and Germano, 1982), providing an alternative monitoring tool where time and cost are pressing issues.

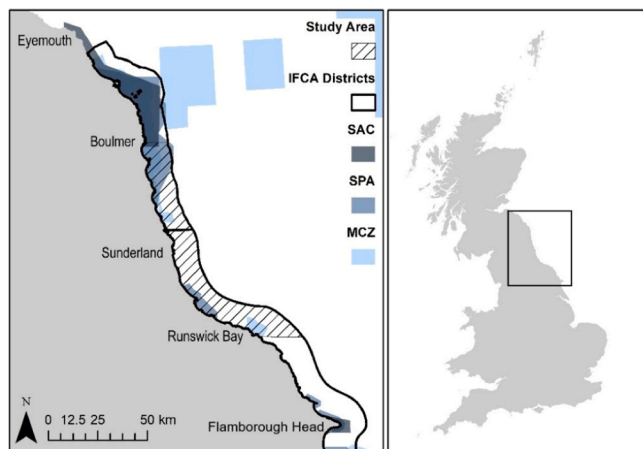
Combining multiple monitoring methods consistently demonstrates the most accurate and well-rounded results (Flannery and Przeslawski, 2015; Jørgensen et al., 2011), for example, combing grabs with trawls, monitoring both infauna and epifauna respectively, informing on the whole community (Jørgensen et al., 2011). Despite the advantages, monitoring studies reporting two or more gear types remain uncommon worldwide (Flannery and Przeslawski, 2015). The integration of innovative sampling methods alongside traditional methods is required for a balanced assessment of aspects of the marine ecosystem (Danovaro et al., 2016). It is also important to compare efficacy and confidence in detecting specific impacts in order to better inform monitoring programme plans.

The purpose of this study was therefore twofold: 1) to assess the impacts of trawling on benthic communities in inshore waters; and 2) to compare the responses of various traditional and innovative monitoring outputs to trawl disturbance. We compare biological communities across a gradient of trawling pressures along the north-east England coast, using maps modelled from fisheries patrol sightings. Traditional macrofaunal and SPI surveys are combined with more innovative eDNA meiofaunal sampling, to deliver a comprehensive ecosystem-based assessment. Multiple macrobenthic and SPI indices are then evaluated for suitability in trawling impact assessments, alongside community composition and common diversity indicators. Finally, we discuss the findings in relation to marine monitoring in the context of ecological quality status and MSFD requirements.

## 2. Methods

### 2.1. Study area & local trawl fishery

Demersal fish and shellfish make up around half of the UK fishery landings, and almost two thirds in England, using primarily bottom fishing gears (MMO, 2019). UK trawlers landed 160 thousand tonnes of demersal fish and 40 thousand tonnes of shellfish in 2018, worth over £ 430 million combined (MMO, 2019). This study focuses on the bottom trawl fishery within the Northumberland Inshore Fisheries & Conservation Authority (NIFCA) and North-East IFCA (NEIFCA) districts (Fig. 1). The fishery primarily uses otter trawls to target mainly prawn (*Nephrops norvegicus*), and to a lesser extent whitefish (e.g., Cod, *Gadus morhua*). Due to a large proportion of the fleet being less than 10 m (MMO, 2019), IFCA vessel sightings from routine sea patrols are the only formal data available to determine the fisheries spatial distribution. Within the study area, spatial restrictions exist to control mobile gears, including trawling, in some MPAs (NIFCA, 2020), but trawling is currently legitimate in many of the other protected areas (Fig. 1), such as the Coquet to St Mary's Marine Conservation Zone (MCZ), which is designated for subtidal mud and sand features, where trawling occurred at the time of the study.



**Fig. 1.** Nearshore waters (to 6 nm) off the coast of north east England within the Northumberland and North Eastern Inshore Fisheries and Conservation Authority districts, and the associated Marine Protected Areas including Special Areas of Conservation, Special Protection Areas, and Marine Conservation Zones.

**2.2. Mapping trawling pressure**

Fishing pressure was modelled for 2016–2018 following Turner et al. (2015), whereby NIFCA and NEIFCA trawling sightings were extracted

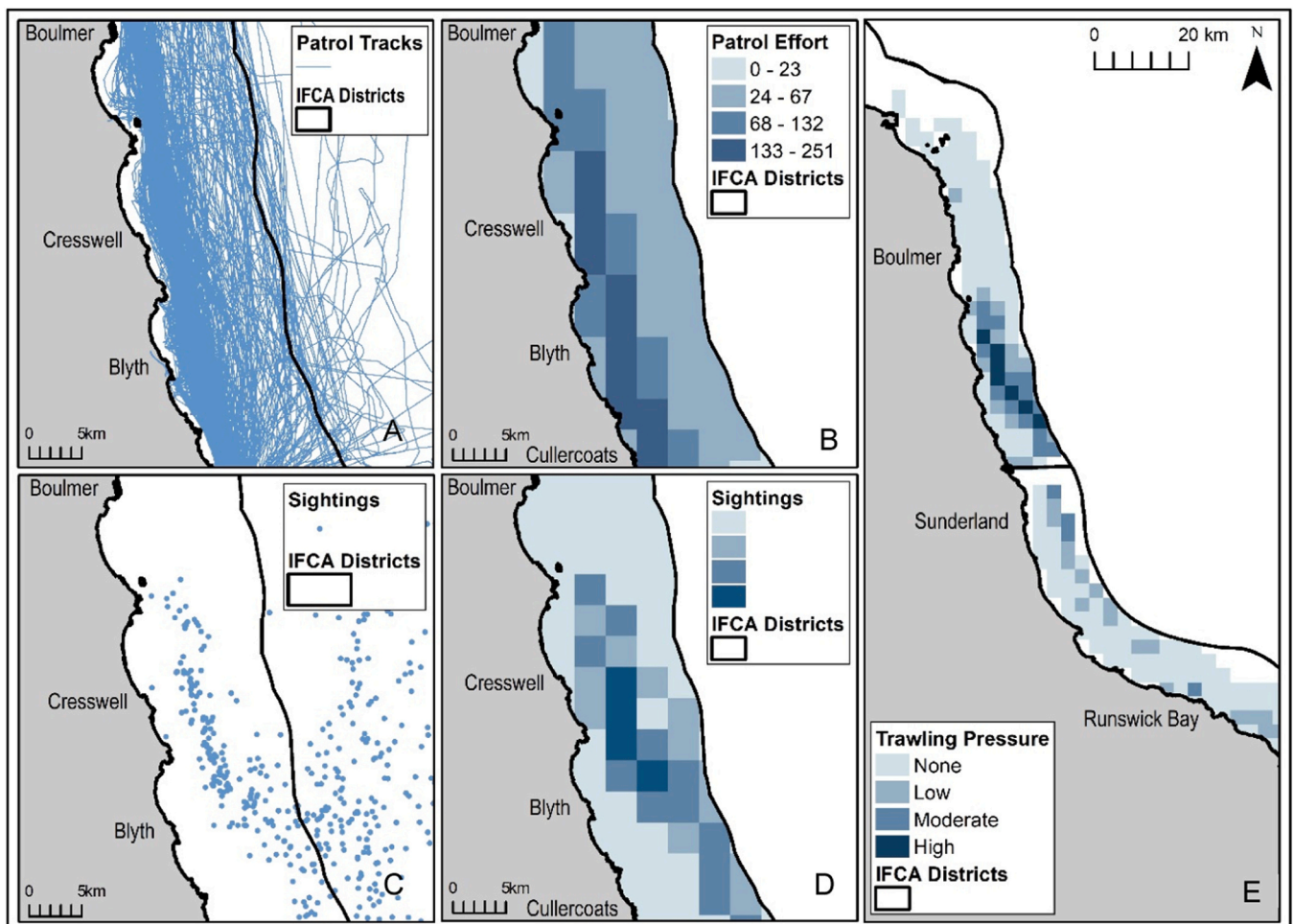
(n = 569) and standardised by patrol effort. Trawling pressure values (SPUE) were split by natural jenks function (ESRI ArcMap v.10.6.1) into 4 categories: none (0), low (0–2), moderate (2–8), and high (8–16) number of vessels per 3 km grid cell. Only grid cells with a moderate to high confidence were used (patrolled more than once in 2 months or 2 weeks respectively (Stephenson, 2016)), which removed much of the northern area furthest away from the NIFCA patrol vessel mooring and around the boundary line between the two districts (Fig. 2).

**2.3. Sample site selection**

Trawling pressure, broad scale habitat (EUSeaMap), substrate hardness (Olex echosounder data from IFCA patrol vessel), and bathymetry data were combined in GIS (ArcMap 10.6.1) to randomly select 12 sample sites satisfying the selection criteria: falling on muddy substrate (standardising the substrate to limit natural variability), covering a gradient of fishing pressures, and over 20 m depth (Fig. 3 & Table 1). Sample sites were inside and outside protected areas.

**2.4. Field sampling**

Grab surveys were carried out from NIFCA patrol vessel ‘St Aidan’ in the summers of 2018 and 2019 (15th and 16th August in 2018, 30th July and 3rd August in 2019). At each sample site (Table 1), five replicate samples of benthos were collected with a Van Veen grab (0.1 m<sup>2</sup>) for biological data, and a further five in the same locations for sediment characteristics (JNCC, 2001; Mason, 2016), resulting in a total of n = 60



**Fig. 2.** a) Patrol tracks, b) Patrol effort (passes per 3 km grid cell), c) Trawling sightings, d) Trawling sightings recorded per 3 km grid cell, e) Trawling pressure (SPUE) with high and moderate confidence, from sightings weighted by patrol effort. All data 2016–2018.

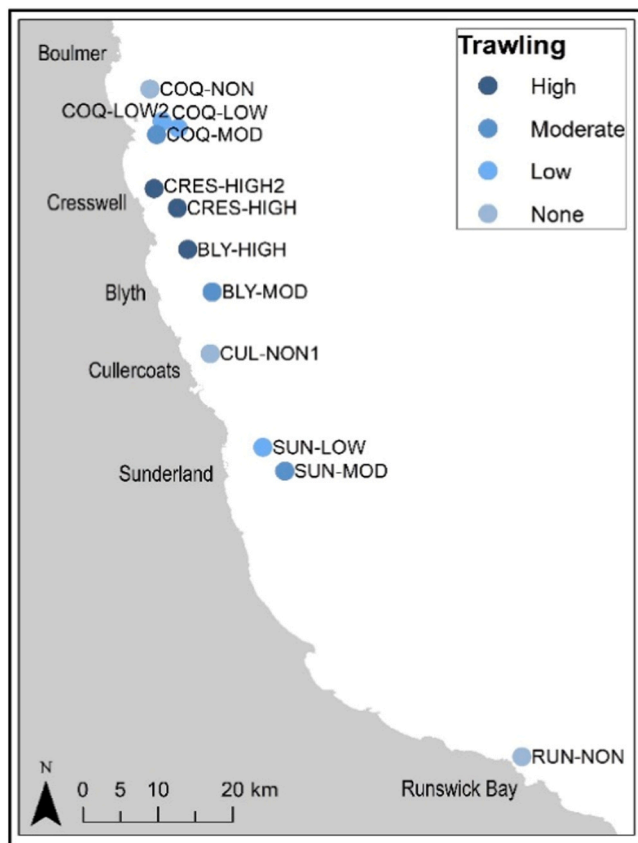


Fig. 3. Sites selected for sampling over a gradient of trawling pressure categories (none, low, moderate, high).

**Table 1**  
Sample sites with associated trawling pressures, depths, and samples.

| Site Code  | Trawling Category | Trawling SPUE | Depth (m) | Sample Year |
|------------|-------------------|---------------|-----------|-------------|
| COQ-LOW    | Low               | 0.80          | 32        | 2018        |
| CRES-HIGH  | High              | 10.92         | 36        | 2018        |
| BLY-MOD    | Moderate          | 5.15          | 39        | 2018        |
| CUL-NON1   | None              | 0.00          | 41        | 2018        |
| SUN-MOD    | Moderate          | 5.80          | 43        | 2018        |
| RUN-NON    | None              | 0.00          | 31        | 2018        |
| COQ-NON    | None              | 0.00          | 32        | 2019        |
| COQ-LOW2   | Low               | 0.91          | 46        | 2019        |
| COQ-MOD    | Moderate          | 4.38          | 38        | 2019        |
| CRES-HIGH2 | High              | 8.37          | 27        | 2019        |
| BLY-HIGH   | High              | 14.90         | 41        | 2019        |
| SUN-LOW    | Low               | 1.91          | 38        | 2019        |

of each type. Descriptive observations of each grab were noted upon retrieval (sample depth, surface colour and texture, colour change with depth) and digital photographs of the surface obtained. Grabs were rejected and re-deployed if sample depth was less than 7 cm (JNCC, 2001), grab jaws were ajar, signs of sediment washout present, and access doors open.

Biological grabs were first subsampled for environmental DNA analysis, using a modified 60 ml syringe corer to collect four replicate sediment samples with a surface area of 5 cm<sup>2</sup> and a depth of 5 cm. The remaining sample (0.098 m<sup>2</sup>) was then sieved on-board using a Wilson Auto-siever fitted with a 0.5 mm mesh sieve. Residual sediment containing macrofauna was immediately fixed in 10% phosphate buffered Formalin. Sediment characteristic grabs were subsampled for Particle Size Analysis (PSA) and Total Organic Carbon (TOC), using a modified

200 ml syringe corer to collect samples with a surface area of 15 cm<sup>2</sup> and a depth of 5 cm for TOC and full sample depth for PSA. All core contents were placed on ice immediately and frozen upon docking.

Adjacent to each grab pair, a Sediment Profile Imagery (SPI) camera system was deployed from St Aidan in August 2018 and the research vessel 'Princess Royal' in August 2019. The SPI operates as an upturned periscope, penetrating the sediment to obtain a cross-sectional image (Germano et al., 2011). Multiple replicate stills were captured of the sediment profile per deployment, acting as 'optical cores', whilst plan-view video was recorded of the substrate surface gathering information on landscape ecology and sediment typography (Wilson et al., 2009).

## 2.5. Sample processing and data extraction

### 2.5.1. Benthic grabs

Macrofauna samples were rinsed in freshwater and sorted into major groups (Annelida, Crustacea, Echinodermata, Mollusca (split into Gastropoda and Bivalvia), and others), before being identified to the lowest practical taxonomic level, and enumerated. Colonial organisms were recorded as present only. Blotted wet weight biomass was measured for each major group following NMBAQC Guidelines (Worsfold et al., 2010). Organisms were blotted to remove excess alcohol prior to weighing, and fauna weighing less than 0.0001 g were assigned a nominal mass of 0.0001 g. Conversion ratios were used to convert biomass values to ash free dry weight (Ricciardi and Bourget, 1998). Laboratory processing of grab subsamples was completed by Benthic Solutions Ltd.

Meiofauna samples underwent DNA metabarcoding. Samples were thawed, buffered, and homogenised, prior to being centrifuged at 10,150 x G for 15 min. The supernatant was discarded and up to 10 g of pelleted material was collected (Taberlet et al., 2012). DNA isolation was performed using QIAGEN DNeasy® PowerMax® Soil kits, following the manufacturer's instructions (Guardiola et al., 2016). DNA yield and purity for each sample was quantified using a QUBIT 3.0 fluorimeter. The nuclear 18 S ribosomal RNA subunit (18 S) (Guardiola et al., 2016) and mitochondrial cytochrome c oxidase subunit 1 (CO1) (Leray et al., 2016) barcode gene regions were selected for amplification using PCR and universal primers. Library preparation used QIAGEN QIAseq® 1-step Amplicon Library kit following the manufacturer's instructions and sequencing was performed on an Illumina MiSeq® sequencer. Processing of raw sequencing data to amplicon sequence variants was performed using an in-house pipeline of bioinformatics tools in R (R Core team, 2016) and Geneious (v11.0.3, Biomatters Ltd.). Taxonomic assignment employed a curated list of barcode gene sequences drawn from public databases, such as NCBI GenBank (Benson et al., 2005) and the international barcode of life database (Ratnasingham and Hebert, 2007).

Sediment samples for PSA were homogenised and small sub-samples removed for laser diffraction. The remaining material was passed through stainless steel sieves with mesh apertures of 8000 µm, 4000 µm, 2000 µm and 1000 µm. Any material retained on the sieve, such as small shells, shell fragments and stones were removed, and the weight was recorded. The laser sub-samples were wet screened through a 1000 µm sieve and determined using a Malvern Mastersizer 2000 particle sizer. The separate assessments of the fractions from sieving and laser were combined for each sample, and the percent of fines (% <63 µm), sand (% 63 µm – 2 mm) and gravel (% >2 mm) calculated, and classified using the Wentworth and Folk classification systems (Folk, 1954; Wentworth, 1922).

TOC was measured using an Eltra combustion method. The samples from each station were treated with 10% HCl to remove inorganic carbon, washed to remove residual acids, and dried. A Carbon Analyser heated the sample in a flow of oxygen, where carbon is converted to carbon dioxide, and measured by infra-red absorption. The percentage of carbon was calculated with respect to the original sample weight (%)

w/w C).

### 2.5.2. Sediment profile imagery

The best quality image was selected for each sample point, i.e., based on least smearing and resuspension, and best illumination and sediment structure. Images collected in 2018 were deemed low quality and were removed from further analysis. Selected still images from 2019 were cropped and scaled prior to data extraction. Illumination defects in the remaining images precluded the use of image analyses to accurately delineate the apparent redox potential discontinuity (aRPD – line marking the transition between oxidising to reducing sediment conditions (Fenchel and Riedl, 1970)), so this was traced manually using expert judgement in ImageJ software (Schneider et al., 2012). The resulting polygon area was measured and divided by the image width to determine mean depth of the aRPD. The presence, abundance, and size of biotic features were recorded for each image, including visible tubes, faecal pellets, feeding pits or mounds, epifauna, infauna, burrows, and oxic voids (Nilsson and Rosenberg, 2006).

## 2.6. Data analysis

### 2.6.1. Macrofauna

Differences in average macrofaunal abundance, biomass, taxonomic richness, and diversity (Shannon-Wiener and Margalef) between each trawling pressure category were tested using Kruskal-Wallis, as parametric assumptions were not met (Dytham, 2011; Underwood, 1997). To account for possible effects of the pressure category scoring bands, correlations were also explored, with abundance, biomass, richness, and diversity plotted against trawling pressure scores (SPUE) and tested with Spearman Rank Order Correlations. Linear mixed effect models were generated for each response variable combining environmental and anthropogenic factors to account for the complex environment and natural variation. The model used a combination of sample location as a random effect, and sample year, sediment type, sample depth, TOC, and trawling score (continuous) as fixed effects. A forward selection technique was used to select the most explanatory variables for model inclusion.

Community composition was analysed using multivariate statistics with PRIMER software (Plymouth Marine Laboratory, UK). Bray Curtis similarity was calculated on square root transformed averaged data to reduce the dominance of common taxa. The similarity matrices were visualised in non-metric multidimensional scaling plots (nMDS), and differences between trawling categories explored with ANOSIM tests. SIMPER analysis was used to determine the species responsible for any significant differences observed.

Macrofauna data were also scored using four biotic indices: AZTI Marine Biotic Index (AMBI, Borja et al., 2000 – benthic community health measure based on ecological groups relating to their sensitivity to disturbance), Multivariate-AMBI (M-AMBI, Muxika et al., 2007 – AMBI scores combined with habitat measures of species richness and diversity), BENTIX (based on relative percentages of 3 ecological groups relating to their sensitivity to disturbance) (Simboura and Zenetos, 2002), and Infaunal Quality Index (IQI, Phillips et al., 2014 – combination of AMBI, diversity, and richness, scored against reference habitat physio-chemical conditions and sampling techniques). AMBI and M-AMBI were calculated using version 5 of a software tool available from ATZI (ambi.azti.es). BENTIX was calculated manually in Microsoft Excel using the taxa scores available at HCMR (www.hcmr.gr/en/the-bentix-index) and the formulas in Simboura et al. (2005). IQI version 4 was calculated using a Microsoft Excel tool developed by the UK Environment Agency. The scoring parameters used for each index are shown in Table 2. Differences, correlations, and models were explored as above for abundance and diversity measures.

### 2.6.2. Meiofauna

The taxa matrices were first sorted to remove all non infaunal

**Table 2**

Classifications (ecological quality or level of disturbance) with scores for mud habitats for each biotic index (AMBI, M-AMBI, BENTIX, and IQI).

| Classification (Ecological Quality or Level of Disturbance) | AMBI        | M-AMBI      | BENTIX      | IQI         |
|---|-------------|-------------|-------------|-------------|
| High / Undisturbed  | < 1.20      | > 0.82      | > 4.00      | > 0.75      |
| Good / Slightly Disturbed                                   | 1.20 – 3.30 | 0.62 – 0.82 | 3.00 – 4.00 | 0.64–75     |
| Moderate / Moderately Disturbed                             | 3.30 – 5.00 | 0.41 – 0.62 | 2.50 – 3.00 | 0.44 – 0.64 |
| Poor / Heavily Disturbed                                    | 5.00 – 6.00 | 0.20 – 0.41 | 2.00 – 2.50 | 0.24 – 0.44 |
| Bad / Extremely Disturbed                                   | > 6.00      | < 0.20      | < 2.00      | < 0.24      |

metazoan taxa, such as insects. DNA concentrations were converted to presence / absence (Chariton et al., 2015), due to the relationship between abundance and the number of sequence reads being weak (Egge et al., 2013; Elbrecht and Leese, 2015). Taxa were merged into taxonomic families to allow for diversity calculations on presence/ absence data using the number of taxa per taxonomic family to estimate Shannon-Wiener diversity (Klunder et al., 2018). Differences in meiofaunal taxonomic richness, family richness, and family diversity between trawling pressure categories were tested with ANOVA, trends with trawling scores with Pearson's Correlations, and more complex relationships with mixed effect models as above for macrofauna. Meiofaunal community composition was analysed with Bray Curtis similarity calculated on presence / absence data, nMDS, ANOSIM, and SIMPER.

### 2.6.3. Sediment profile imagery

Image outputs were scored using two different multi-parameter SPI indices: Benthic Habitat Quality (BHQ, (HC Nilsson and Rosenberg, 1997; Rosenberg et al., 2003) – measure based on sediment surface faunal structures (fecal pellets, tubes, pits and mounds), subsurface structures (infaunal structures, burrows and oxic voids) and the measurement of the aRPD)) and Organism Sediment Index (OSI, (Rhoads and Germano, 1982); Rhoads and Germano (1987) – measure based on aRPD, the macrofaunal successional stage, and chemical parameters). For the BHQ, successional stage and Ecological Quality Status was determined using the updated classification derived for the Water Framework Directive in water depths greater than 20 m (Rosenberg et al., 2004), which was subsequently used for the calculation of OSI. Patterns in OSI and BHQ values along the trawling pressure gradient were explored as above for meiofauna (ANOVA Pearson's Correlations, and mixed effect models). Further examination of the individual parameters used in the calculation of BHQ was also carried out with Spearman's Rank Order Correlations.

## 3. Results

### 3.1. Macrofauna

#### 3.1.1. Abundance, richness, diversity, and biomass

Abundance, taxonomic richness, Shannon's diversity, and Margalef diversity of macrofauna did not appear to be altered by trawling pressure when examined alone, with no significant differences between averages for trawling categories (Kruskal-Wallis,  $p > 0.05$ ) or correlations with trawling scores (Spearman's Rank Order Correlation,  $p > 0.05$ ). In contrast, there were strong significant negative correlations between macrofauna abundance, taxonomic richness, and Margalef diversity with TOC levels (Spearman's Rank Order Correlation,  $r_{2abundance} = -0.359$ ,  $r_{2richness} = -0.439$ ,  $r_{2diversity} = -0.402$ ,  $p < 0.005$ ). Mixed effects models controlling for additional factors (location, sample year, sediment type, sample depth, and TOC) revealed that trawling pressure was in fact a significant influencing factor in the

observed taxonomic richness, Shannon's diversity, and Margalef diversity, but not abundance (Table 3). Trawling pressure and location were the two most influential factors across the board.

Macrofaunal biomass was robust to changes in trawling pressure at the levels examined here. Total biomass and major taxonomic groups' biomasses were not significantly different between trawling categories (Kruskal-Wallis,  $p > 0.05$ ), or significantly correlated with trawling scores (Spearman's Rank Order Correlation,  $p > 0.05$ ). Mixed effects models further confirmed this, with trawling score not significantly influencing biomass for any group (Table 3).

### 3.1.2. Community composition

Differences in community composition of macrofauna between sample years (ANOSIM,  $R = 0.467$ ,  $p < 0.001$ ) were stronger than trawling pressure (ANOSIM,  $R = 0.220$ ,  $p < 0.001$ ), so samples were stratified by year for further analysis of trawling effects. Trawling had a larger impact on community composition in the 2018 samples than 2019 (ANOSIM,  $R_{2018} = 0.425$ ,  $R_{2019} = 0.190$ ,  $p_{2018} < 0.001$ ,  $p_{2019} < 0.05$ ). Visualisation of these differences in nMDS plots clearly show separation patterns observed in order of increasing pressure in the 2018 data (i.e. none furthest from high), but not in 2019, or both years combined (Fig. 4). The samples from a single site, CRES-HIGH2, are responsible for the reduced patterns observed in the 2019 data. With CRES-HIGH removed from the dataset, there is a stronger difference between trawling pressures for 2019 alone and both years combined (ANOSIM,  $R_{2019} = 0.264$ ,  $R_{combined} = 0.457$ ,  $p < 0.001$ ). SIMPER analysis revealed that there was 69.42% dissimilarity between communities at the unfished and highly fished sites. The average abundances of the five taxa contributing most to these differences can be seen in Table 4, two of which significantly increased whilst three significantly decreased with increasing trawling pressure.

### 3.1.3. Indices

Macrofaunal index classifications did not differ greatly between samples, with the study area being generally good and high Ecological Quality Status (EcoQS) for all indices. AMBI classified all samples as slightly disturbed, and good EcoQS. BENTIX and IQI scores marked all samples as either good or high EcoQS, and M-AMBI scores for each trawling pressure category were all assigned as high.

Patterns in macrofaunal index scores along the trawling pressure gradient were varied and inconsistent (Fig. 5). AMBI scores displayed the strongest trend, having the only statistically significant difference between trawling categories (Kruskal-Wallis,  $H = 9.60$ ,  $p < 0.05$ ) and correlation with trawling scores (Spearman's Rank Order Correlation,  $r_2 = 0.461$ ,  $p < 0.001$ ). There was an increase in the AMBI score with increased trawling pressure, but not at a magnitude large enough to alter the ecological quality classification assigned. Mixed effects models controlling for additional factors (location, sample year, sediment type, sample depth, and TOC) revealed that trawling pressure, year, and TOC most influenced AMBI scores, whilst IQI was most influenced by

trawling pressure and location, and BENTIX by TOC (Table 5).

## 3.2. Meiofauna

### 3.2.1. Richness and diversity

Patterns in the meiofaunal data differed significantly between years. In 2018, taxonomic richness, family richness, and diversity were all significantly different between trawling pressure categories (ANOVA,  $F_{\text{taxa richness}} = 5.48$ ,  $F_{\text{family richness}} = 4.47$ ,  $F_{\text{diversity}} = 4.05$ ,  $p < 0.05$ ), with a general increase with higher fishing pressure. In 2019, only taxonomic richness was significantly different between pressure categories (ANOVA,  $F_{\text{taxa richness}} = 4.70$ ,  $p < 0.05$ ), with a decreasing pattern with increased pressure. When both years were combined, there was no significant differences observed in the three responses between pressure categories. Fig. 6 displays these contrasts for taxonomic richness. Correlations with trawling scores showed a similar pattern, with significant positive correlations observed in the 2018 data for all three responses (Pearson's Correlation,  $r_{\text{taxa richness}} = 0.473$ ,  $r_{\text{family richness}} = 0.409$ ,  $r_{\text{diversity}} = 0.373$ ,  $p < 0.05$ ), and a significantly negative correlation for taxonomic richness in 2019 (Pearson's Correlation,  $r_{\text{taxa richness}} = -0.389$ ,  $p < 0.05$ ), and no significant correlation when both years were combined.

Mixed effects models controlling for additional factors (location, sample year, sediment type, sample depth, and TOC) further emphasised the contrasts between sample years, with 'year' being the dominant influencing factor for richness and diversity measures. Trawling was only a significant factor for the 2019 and combined datasets, along with depth (Table 6).

### 3.2.2. Community composition

There were strong significant differences in meiofaunal community composition between years (ANOSIM,  $R = 0.803$ ,  $p < 0.005$ ), and less so between sediment type, location, and trawling pressure (ANOSIM,  $R_{\text{sediment}} = 0.262$ ,  $R_{\text{location}} = 0.139$ ,  $R_{\text{trawling}} = 0.084$ ,  $p < 0.05$ ). Data were therefore stratified by year prior to further analysis, which increased the differences observed between trawling pressure categories (ANOSIM,  $R_{2018} = 0.187$ ,  $R_{2019} = 0.107$ ,  $p < 0.05$ ). Visualisation of these differences in nMDS plots showed that the unfished sites had the most unique communities in 2018, with visual separation of the no pressure samples from the three fished pressure categories, whereas no such pattern was obvious in 2019 or when all samples were combined (Fig. 7). SIMPER analysis of the 2018 data revealed that there was 45.84% dissimilarity between communities at the unfished and highly fished sites. Of the five taxa contributing most to these differences, four increased in prevalence (*Paranehalia belizensis*, *Temora longicornis*, *Hubrechtella dubia*, *Isias clavipes*), whilst one decreased (*Aricidea minuta*) with increasing trawling pressure.

**Table 3**

Mixed effects model outputs for abundance, taxonomic richness, Shannon's diversity, Margalef diversity, and ash-free dry weight biomass (g) for all taxa and each major group. R-sq value for each model (goodness-of-fit), F-values for each model factor combination shown if significant only (with associated significance level depicted by asterisk: \* =  $p < 0.05$ , \*\* =  $p < 0.005$ , \*\*\* =  $p < 0.0005$ ).

|                       | R-sq  | Trawling  | Location  | Year      | Sediment | Depth  | TOC      |
|-----------------------|-------|-----------|-----------|-----------|----------|--------|----------|
| Abundance             | 75.7% | -         | 6.06 ***  | 34.55 *** | -        | 4.19 * | -        |
| Taxonomic Richness    | 82.4% | 13.71 **  | 17.18 *** | 5.49 *    | 5.32 **  | 4.37 * | 11.17 ** |
| Shannon Diversity     | 77.3% | 37.79 *** | 17.37 *** | 6.95 *    | -        | -      | -        |
| Margalef Diversity    | 80.4% | 19.63 *** | 19.98 *** | -         | 4.20 *   | -      | 9.88 **  |
| Total Biomass         | 24.0% | -         | -         | -         | -        | -      | -        |
| Polychaete Biomass    | 21.3% | -         | -         | -         | -        | -      | -        |
| Crustacea Biomass     | 33.3% | -         | -         | 7.33 *    | -        | -      | -        |
| Gastropoda Biomass    | 40.7% | -         | 2.82 *    | 11.05 **  | -        | -      | -        |
| Bivalvia Biomass      | 25.1% | -         | -         | -         | -        | -      | -        |
| Echinodermata Biomass | 35.1% | -         | 2.37 *    | -         | -        | -      | -        |
| Other Biomass         | 22.8% | -         | -         | -         | -        | -      | -        |

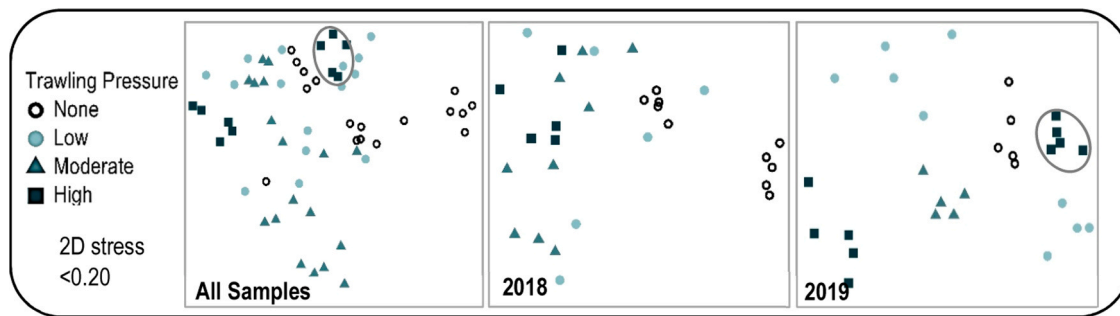


Fig. 4. nMDS plots of square-root transformed abundance data showing the similarity of community composition between trawling pressure categories for all samples combined, 2018 samples only, and 2019 samples only. Circled samples are from CRES-HIGH site which appear to be outliers.

Table 4

Median (+/- range) abundance for each trawling pressure category (none, low, moderate, high) of the five species most responsible for differences between unfished and highly fished sites from SIMPER analysis. Trend direction along the increasing fishing pressure gradient (↑ = increasing, ↓ = decreasing), and degree of difference between categories (H value, Kruskal-Wallis) with associated significance level (blank = not significant, \* =  $p < 0.05$ , \*\* =  $p < 0.005$ , \*\*\* =  $p < 0.0005$ ).

| Taxa                       | Average Abundance per Trawling Pressure |                 |                |                  | Trend | Difference |
|----------------------------|---|-----------------|----------------|------------------|-------|------------|
|                            | None                                    | Low             | Moderate       | High             |       |            |
| <i>Abra sp.</i>            | 4<br>(+/- 77)                           | 16<br>(+/- 139) | 3<br>(+/- 168) | 186<br>(+/- 168) | ↑     | 13.93 **   |
| <i>Amphiura filiformis</i> | 38<br>(+/- 73)                          | 11<br>(+/- 73)  | 0<br>(+/- 52)  | 0<br>(+/- 0)     | ↓     | 33.47 ***  |
| <i>Kurtiella bidentata</i> | 21<br>(+/- 87)                          | 3<br>(+/- 26)   | 0<br>(+/- 2)   | 0<br>(+/- 0)     | ↓     | 34.38 ***  |
| <i>Ophiuroidea</i>         | 16<br>(+/- 83)                          | 17<br>(+/- 137) | 2<br>(+/- 12)  | 1<br>(+/- 2)     | ↓     | 26.81 ***  |
| <i>Praxillella affinis</i> | 0<br>(+/- 0)                            | 0<br>(+/- 1)    | 0<br>(+/- 2)   | 6<br>(+/- 7)     | ↑     | 32.12 ***  |

### 3.3. Sediment profile imagery

Both BHQ and OSI scores varied widely over the study area, ranging between two and eleven, and covering bad to good ecological quality status categories. The BHQ and OSI index scores were both significantly different between trawling pressure categories and negatively correlated with trawling scores (Table 7). Mixed effects models controlling for additional factors (location, sediment type, sample depth, and TOC) revealed that BHQ scores were significantly influenced by trawling score (R-sq = 63.00%,  $F = 4.51$ ,  $p < 0.05$ ), whilst OSI scores were not (R-sq = 56.95%,  $F = 1.84$ ,  $p > 0.1$ ). Further investigation into the individual BHQ parameters revealed that only camera penetration depth and aRPD depth significantly correlated with trawling pressure scores, with a deeper penetration and shallower aRPD with increasing fishing pressure (Spearman's Rank Order Correlation,  $r_{2penetration} = 0.376$ ,  $r_{2aRPD} = -0.473$ ,  $p < 0.05$ ).

## 4. Discussion

Most trawling impact studies, or monitoring programmes, concentrate on a single target community, usually macrofauna (Bolam et al., 2014; Collie et al., 2000; Hiddink et al., 2017; JNCC, 2001; JNCC, 2016). This study took an ecosystem-based approach, assessing macrofauna, meiofauna, and sediment characteristics by combining traditional and innovative methods, conducting a truly balanced assessment of the

north-east England trawl fishery. Contrasting responses were observed, highlighting the complexity of the system, further stressing the importance of wider monitoring and the advantages of combining multiple methods for well-rounded assessments (Danovaro et al., 2016; Flannery and Przeslawski, 2015).

### 4.1. Macrofauna

Macrofaunal overall abundance and biomass were not affected by trawling pressure in this case, despite this being a common observation in previous studies (e.g. Ball et al., 2000; Hiddink et al., 2017). Many of these were experimental studies, where samples were collected shortly after a trawling incident when initial mortality effects are most evident. In this study, it was impossible to know when the most recent trawl pass had occurred, so it is possible that sites had begun to recover in terms of abundance, whether from migration in the short term or recruitment in the longer term (Kaiser et al., 2006). In contrast, there were signs of indirect longer-term impacts such as reduced richness and diversity, and altered community composition, suggesting communities remain in a changed state (Jones, 1992). Species richness and diversity can be reduced as communities lose the most sensitive species, whilst more resilient species excel (Hiddink et al., 2019; Hiddink et al., 2017; Kaiser and Spencer, 1996; Lambert et al., 2014). Of the taxa most responsible for the differences in community composition, those showing a positive response to trawling (*Abra sp.* and *Praxillella affinis*) are both small short-lived taxa that are classified as tolerant to disturbance (Horton et al., 2020; Marlin, 2006). Two of the taxa showing negative trends (*Amphiura filiformis* and *Ophiuroidea*) are larger, longer living brittlestars (Marlin, 2006), which are generally fragile (Marlin, 2020), live close to the sediment surface and are vulnerable where trawling impacts are the most intense (Pommer et al., 2016). The observed patterns fit well with previous observations of links with body size and longevity on impact and recovery (Hiddink et al., 2019; Jennings et al., 2001).

The index scores of macrofaunal communities were good or high at all sites, irrespective of trawling intensity. There were significant patterns in scores within those categories, with AMBI showing the strongest trend with trawling pressure. Indices have been tested significantly less on fishing impact detection than chemical pollution and organic enrichment, however AMBI has received mixed reviews to date. While it proved useful for detecting trawling impacts in the Arabian Gulf (Rabaoui et al., 2019), it was outperformed in the Kattegat, Gulf of Lion, English Channel, and southern North Sea by BQI (using rarefied species densities), invertebrate density, species richness, or Margalef index (Gislason et al., 2017; Jac et al., 2020). Here, AMBI performed better than IQI, BENTIX, and M-AMBI. IQI is used in the UK for WFD monitoring, so would perhaps be the obvious choice for wider impact detection in the study area. IQI has been widely tested for detecting impacts from hazardous substances, organic enrichment, and smothering, whilst the response to physical fisheries related disturbance is un-quantified (WFD - UKTAG, 2014). When benthic abrasion is a key

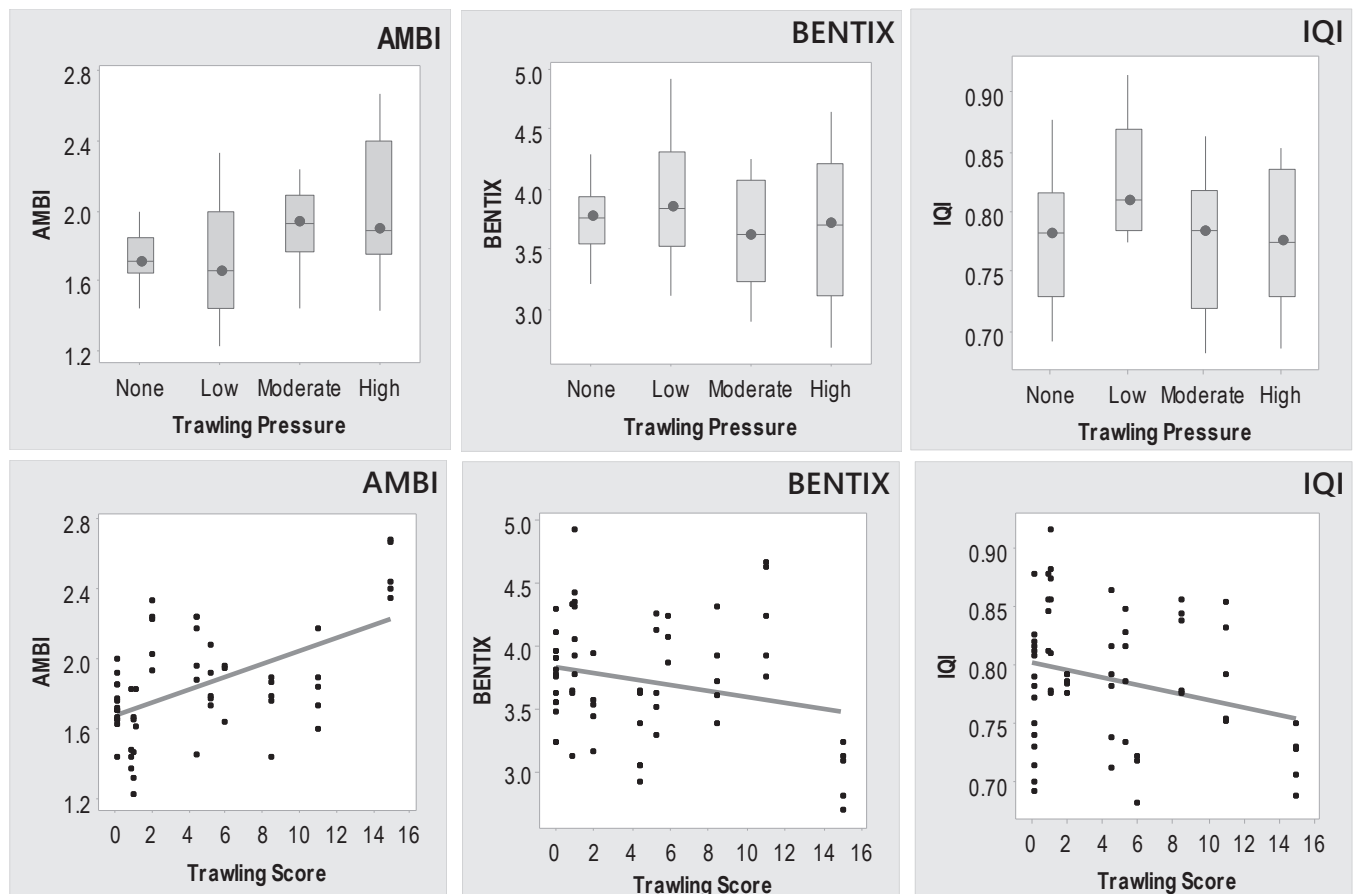


Fig. 5. AMBI, BENTIX, and IQI scores along the potting pressure gradient. Boxplots display the median, interquartile range, and range for each trawling category. Scatter plots display all sample values for each trawling score, with linear regression line.

Table 5

Mixed effects model outputs for AMBI, BENTIX, and IQI. R-sq value for each model (goodness-of-fit), F-values for each model factor combination shown if significant only (with associated significance level depicted by asterisk: \* =  $p < 0.05$ , \*\* =  $p < 0.005$ , \*\*\* =  $p < 0.0005$ ).

|        | R-sq  | Trawling | Location  | Year     | Sediment | Depth  | TOC      |
|--------|-------|----------|-----------|----------|----------|--------|----------|
| AMBI   | 77.5% | 11.28 ** | 4.38 **   | 11.85 ** | -        | -      | 12.22 ** |
| BENTIX | 62.9% | 7.41 **  | 6.11 ***  | -        | -        | 5.00 * | 9.55 **  |
| IQI    | 73.4% | 11.03 ** | 12.85 *** | 5.19 *   | 4.54 **  | -      | -        |

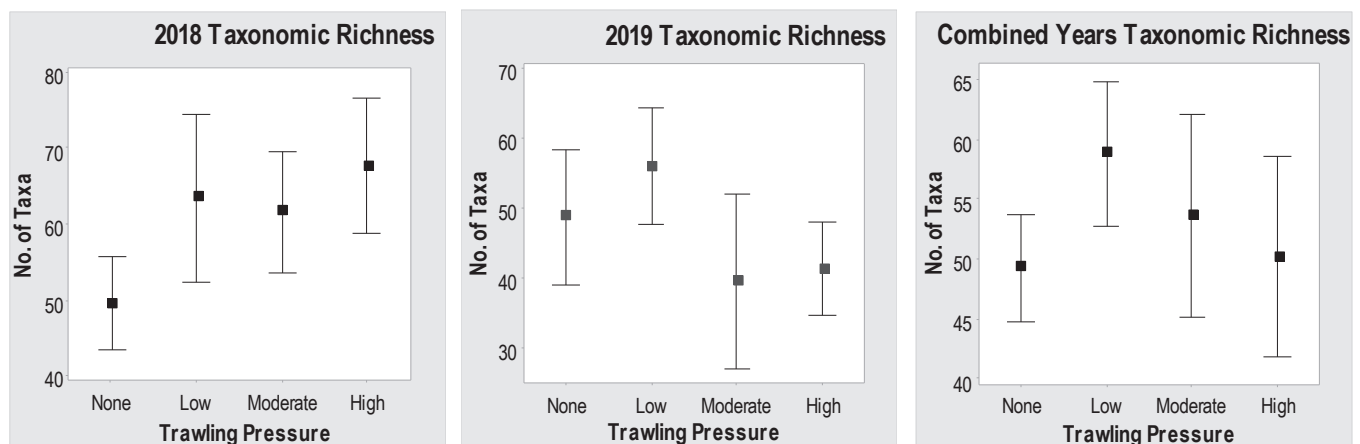
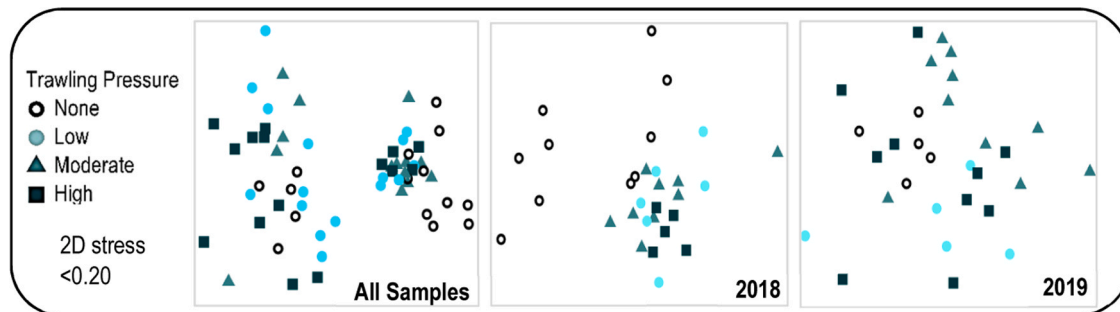


Fig. 6. Mean (+/- 95% CI calculated from pooled standard deviation) taxonomic richness for 2018, 2019, and combined samples, along the fishing pressure gradient categories (none, low, moderate, high).

**Table 6**

Mixed effects model outputs for meiofaunal taxonomic richness, family richness, and Shannon diversity for each dataset (2018, 2019, both years combined). R-sq value for each model (goodness-of-fit), F-values for each model factor combination shown if significant only (with associated significance level depicted by asterisk: \* =  $p < 0.05$ , \*\* =  $p < 0.005$ , \*\*\* =  $p < 0.0005$ ).

| Dataset        | Parameter          | R-sq  | Trawling | Location | Year      | Sediment | Depth     | TOC |
|----------------|--------------------|-------|----------|----------|-----------|----------|-----------|-----|
| Combined Years | Taxonomic Richness | 59.5% | 9.65 **  | 4.70 **  | 15.25 *** | -        | 17.09 *** | -   |
|                | Family Richness    | 52.7% | 6.19 *   | 2.46 *   | 15.87 *** | -        | 11.30 **  | -   |
|                | Shannon Diversity  | 47.0% | 5.79 *   | -        | 12.99 **  | -        | 8.64 **   | -   |
| 2019           | Taxonomic Richness | 57.3% | 5.35 *   | -        | N/A       | -        | 9.24 *    | -   |
|                | Family Richness    | 56.2% | 6.26 *   | -        | N/A       | -        | 10.01 **  | -   |
|                | Shannon Diversity  | 48.0% | 5.51 *   | -        | N/A       | -        | 6.69 *    | -   |
| 2018           | Taxonomic Richness | 57.5% | -        | -        | N/A       | -        | -         | -   |
|                | Family Richness    | 47.8% | -        | -        | N/A       | -        | -         | -   |
|                | Shannon Diversity  | 50.4% | -        | -        | N/A       | -        | -         | -   |



**Fig. 7.** nMDS plots of meiofauna presence/absence data showing the similarity of community composition between trawling pressure categories for all samples combined, 2018 samples only, and 2019 samples only.

**Table 7**

Mean (+/- SD) BHQ and OSI scores for each trawling pressure category (none, low, moderate, high). Degree of difference between trawling categories (F value, ANOVA) and correlation with trawling scores (r value, Pearson's' Correlation), with associated significance level (\* =  $p < 0.05$ , \*\* =  $p < 0.005$ ).

| Index | Average Score per Trawling Pressure |                  |                  |                  | Trend | Difference | Correlation |
|-------|-------------------------------------|------------------|------------------|------------------|-------|------------|-------------|
|       | None                                | Low              | Moderate         | High             |       |            |             |
| BHQ   | 7.8<br>(+/- 1.8)                    | 6.3<br>(+/- 1.5) | 8.8<br>(+/- 1.6) | 4.5<br>(+/- 2.0) | ↓     | 8.18 **    | -0.537 **   |
| OSI   | 9.0<br>(+/- 0.9)                    | 7.4<br>(+/- 1.0) | 9.0<br>(+/- 0.9) | 5.1<br>(+/- 1.4) | ↓     | 5.21 *     | -0.566 **   |

consideration, it appears that using IQI in combination with AMBI could show the best results. Although AMBI worked well along the north-east England coast to detect trawling impacts, index performance is strongly linked to geography, and so it is important to systematically screen the options locally to select the most appropriate (Jac et al., 2020). This makes selecting or recommending a single macrofaunal index for the MSFD impossible at the European scale, with indicators which perform inconsistently of little value at this scale. Perhaps a reduced list with recommendations for different regions and target impacts would be better suited, but still limitations need to be clear.

#### 4.2. Meiofauna

Meiofauna are considered fairly robust to the impacts of trawl disturbance due to their small size, likelihood of being resuspended rather than killed, and having short regeneration times to recover rapidly when necessary (M. Schratzberger et al., 2002). Theory suggests that benthic disturbance may in fact lead to their proliferation (Jennings et al., 2001). Responses in previous trawling studies are varied, with positive (Lampadariou et al., 2005), negative (Costa and Netto, 2014; Lampadariou et al., 2005; Pusceddu et al., 2014), and indifferent (M. Schratzberger et al., 2002) impacts observed. This study found that meiofauna responded positively to trawling pressure in samples collected in 2018, but mostly indifferently in 2019 and when years were

combined. The fishing pressure model used was most up to date for the 2018 samples, having been produced with 2016–2018 patrol records. Data for 2019 sightings were not digitised and available to incorporate at the time of this study. It is possible that a in shift fishing pressure in 2019 not accounted for by the model could explain the differences seen. However, significant shifts in fishing patterns are unlikely year to year locally, as the preferred habitat is limited in the area, and fishers are known to have strong ties to known productive grounds and tend to revisit them.

In 2018, the differences were greatest between the unfished and fished sites, with strong differences even at the lowest trawling pressure, suggesting that the threshold for change was low, and the presence of trawling was more important than the intensity in structuring meiofaunal communities. The increased richness and diversity recorded in 2018 could be due to factors such as: reduced competition or predation from macrofauna (Giere, 2008; Ingels et al., 2014), and boosted oxygen, organic matter, and nutrients created by sediment turnover (Lampadariou et al., 2005). Changes in community composition come about from the differences in sensitivity to changing conditions, and it is thought that nematodes are generally stress tolerant compared to more sensitive groups like copepods and ostracods (Pusceddu et al., 2014; Zeppilli et al., 2015). Of the taxa most responsible for the differences in community composition here, the positive response of *H. dubia* could be predicted by its classification as tolerant in the AMBI species list and a

scavenger, whilst the others are small with robust body structures (Borja et al., 2000; Horton et al., 2020). Similarly, the decrease in *A. minuta* could be predicted by an AMBI classification of 'very sensitive to disturbance', larger body size, and softer structure (Borja et al., 2000; Rouse and Pleijel, 2001).

Meiofauna are vital contributors to ecosystem functioning, undertaking nutrient cycling, playing an important role in food webs as both consumers and producers, and modifying the interactions between macrofaunal species (Gièrè, 2008; Piot et al., 2014; Woodward, 2010). Positive effects of altered communities include the enhancement of ecosystem functions from higher meiofaunal richness, (Covich et al., 2004), and 'farming the sea' from boosted productivity related to the dominance of smaller species (Jennings et al., 2001). Conversely, a skew towards meiofauna could have negative impacts, as they are highly metabolically active, consuming up to ten times more oxygen than macrofauna at the equivalent biomass (Gerlach, 1971). Abundance and biomass were not measured here, but if these increased in line with richness and diversity, negative impacts could result from reduced oxygen availability in benthic sediments.

#### 4.3. Sediment profile imagery

BHQ scores were most influenced by trawling, and so this index is recommended, over OSI, for use in future fishing impact studies to detect change. This may be due to the fact that the BHQ metric gives more weight to the sediment physical structures, such as surface mounds and pits created by fauna (Rosenberg et al., 2003), which can be easily removed or altered by trawling. BHQ scores significantly reduced with increased trawling pressure, indicating a reduction in habitat quality, in line with previous observations (Badino et al., 2004; Nilsson and Rosenberg, 2003; Rosenberg et al., 2003). In contrast to the meiofauna observations, here the threshold of change was much higher, with the largest drop in scores occurring only when trawling intensity entered the high-pressure category. Scores were highly variable between sites, much more so than the macrofauna indexes, suggesting that sediment dynamics were driving much of the variation rather than biological parameters. This was further supported by models revealing that sediment penetrability and aRPD depth were mostly responsible for the trends in BHQ scores with trawling. Trawling has previously been linked to sediment alterations, such as the resurfacing of toxic substances, change in sediment texture, reduction in the proportion of fine sediment, reduction in surface roughness, creation of plough furrows and spoil heaps, alteration of redox profiles, and increased sediment mixing (Bradshaw et al., 2012; Costa and Netto, 2014; De Biasi, 2004; Depestele et al., 2019; Lindeboom and Groot, 1998; Rosenberg et al., 2003). Sedimentary impacts appear to vary widely between studies, depending on the original sediment characteristics. Here, penetration was greater in trawled areas, presumably from the gear breaking up the surface layers or the presence of a finer top layer from settlement post resuspension, in contrast to studies describing shallower penetration due to sediment compaction from the weight of the gear or the removal of the soft surface sediment layer (Lindeboom and Groot, 1998; Smith et al., 2003). In agreement with previous studies, redox depth was shallower in trawled areas (Badino et al., 2004; H. C. Nilsson and Rosenberg, 2003). This depletion of the oxidised sediment layer could prove an obstacle to the reburrowing of infauna displaced by the fishing event (Badino et al., 2004). In addition to the effect on infaunal communities from altered sediment structure and dynamics, changes to sediment compactness and oxygenation have the potential to alter the rate of biogeochemical cycling (Badino et al., 2004; Trimmer et al., 2005). Therefore, significant knock-on effects beyond those directly measured by the SPI are probable. Additional chemical analysis of benthic sediment in the north-east trawl fishery would be useful to examine these effects further.

#### 4.4. Recommendations

A major limitation of studying inshore fisheries comprised of smaller vessels is the lack of reliable locational data, such as VMS. Although spatial models can be successfully produced from sightings data, they are generally lower spatial resolution and confidence. Whilst individual sightings are accurate records of fishing activity, a lack of sightings, especially at locations rarely patrolled, does not mean that no fishing is occurring, which can introduce bias (Breen et al., 2014). One solution is increased provision of resources to IFCAs to enable greater collection of sightings data, with the aim of standardising patrol routes, maximising spatial coverage, and reducing uncertainty (Breen et al., 2014). The maximum distance of patrol view to date necessitated the 3 km resolution used in this study (Turner et al., 2015), but can result in detail on the heterogeneity of fishing pressure within a 3 km area being overlooked, which is an important consideration due to the patchiness of fishing activities (Kaiser et al., 2002). Individual sample sites may not be fully representative of the assigned pressure score. Introducing VMS into smaller vessels would increase the accuracy of trawling pressure maps for inshore waters, but a major challenge is the resistance of fishers to share fishing locations (McCluskey and Lewison, 2008). The Department for Environment and Rural Affairs plans to roll out inshore vessel monitoring (I-VMS) in the near future for vessels under 12 m (DEFRA, 2018), at which time higher resolution pressure maps can be produced. Until then, the maps produced here are the best possible snapshot of trawling pressure with the limited data available, and contribute to filling a major knowledge gap in the spatial distribution and impacts of inshore fisheries (Breen et al., 2014), which could not be addressed with experimental studies at the fisheries scale (Hiddink et al., 2017).

Whilst the use of the most recent sightings data is essential to measure current impacts, it is also important to acknowledge that sites currently classed as unfished may have been fished in the past; they are unlikely to represent truly un-impacted scenarios for comparison. Bottom fisheries have been active for hundreds of years (Bennema and Rijnsdorp, 2015), with historical distribution largely unknown before the introduction of fisheries patrols and sightings in recent decades. It is possible that the entire study area has been heavily trawled and dredged in the past, resulting in a shifted baseline, where historical fishing has already removed the most sensitive species (Brown and Trebilco, 2014; Hiddink et al., 2017). Shifted baselines have the potential to mask the severity of impacts in current studies. Closing designated areas to fishing is one approach to reversing degradation and restoring ecosystems to their pre-exploited baselines (Pitcher, 2001), with the added advantage of allowing for enhanced comparative impact studies (Kaiser et al., 2002). Long-term closures have allowed the effects of scallop dredging on benthic communities to be better evaluated in the Isle of Man, and have led to significant recovery of benthic species and habitats (Beukers-Stewart et al., 2005; Howarth et al., 2015). Similar areas closed to trawling or modifying gear to reduce impacts in areas of the north-east coast could help to establish similar baselines as well as protecting areas under a precautionary approach (FAO, 1996). McConnaughey et al. (2020) reviewed trawl fishery management methods, concluding that there is no universal best practice, and that combining a variety of approaches is best to meet sustainability goals whilst balancing food production needs with environmental protection. Whichever measures are selected for a certain fishery, adaptive processes to monitor performance and allow for refinements are advised (McConnaughey et al., 2020). As such, it is important that the north-east England trawl fishery should continue to be studied, especially considering management amendments inside protected areas.

#### 5. Conclusions

Examining benthic communities and habitats subject to a gradient of trawling pressure revealed significant impacts associated with benthic disturbance within the study area. Macrofauna richness, diversity,

community composition, and index scores suffered, whilst in contrast, meiofauna richness and diversity were boosted in the presence of trawling in the first sample year. A higher-pressure threshold was observed for sediment effects, with sediment compactness and oxygenation influenced under the highest trawling pressures.

The lack of accurate positional data for fishing activities coupled with potential lasting effects of historic fishing pressure, have the potential to be masking more severe impacts. Although impacts are clear at current levels, the most intensely trawled areas continue to sustain abundant infaunal communities and may not be altered beyond their natural limits. However, negative trends in abundance were shown in fragile species sensitive to direct trawling impacts along the pressure gradient. It is important to acknowledge that lower confidence pressure data in data poor fishery areas makes assessing the true value of monitoring indicators more challenging, and therefore complimentary studies in areas with detailed trawling data would be beneficial to confirm this studies finding of which indicators perform best.

The various indicators used likely differ in their sensitivity based on the timing of the trawling events. For example, SPI indices might better capture effects of recent trawling events where the sediment has yet to resettle and reoxygenate, whilst community shifts in longer lived faunal species will take longer to appear after trawling and may require repeated events. This further supports the use of multiple methods and metrics together, as in this study, especially when the timing of disturbance events is unclear in the pressure data available.

The quantitative data collected on the habitats and communities present is valuable to local marine managers and provides an understanding of current conditions for future work, especially if pressures increase or management is modified. As of 2020, a NIFCA byelaw was put in place where light otter trawl gear only was permitted within the Coquet to St. Mary's MCZ, thus no scallop dredging, or heavy otter trawl activity is allowed. The data gathered in this study will be invaluable going forwards to investigate change in benthic communities and habitats following this change in management.

This study highlights the importance of combining multiple monitoring approaches to deliver a balanced view, rather than focussing on single factors which do not well represent wider ecosystem impacts. Combining more traditional morphological methods with metabarcoding techniques and sediment assessments helps to illuminate a bigger picture and is recommended in monitoring regimes and for consideration in further MSFD indicator testing and development.

#### CRediT authorship contribution statement

**ATM:** Conceptualization, Methodology, Formal analysis, Writing – original draft, Writing – review & editing, Visualization, Project administration, Funding acquisition. **CF:** Conceptualization, Methodology, Resources, Writing – review & editing, Supervision, Project administration, Funding acquisition. **HS:** Conceptualization, Methodology, Resources, Writing – review & editing, Supervision, Project administration, Funding acquisition. **CS:** Conceptualization, Writing – review & editing. **RK:** Methodology, Formal analysis, Resources, Writing – review & editing.

#### Declaration of Competing Interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Clare Fitzsimmons reports financial support was provided by European Maritime and Fisheries Fund.

#### Data Availability

Data will be made available on request.

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