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Research article

European coastal monitoring programmes may fail to identify impacts on benthic macrofauna caused by bottom trawling

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ABSTRACT

Bottom trawling (hereafter trawling) is the dominant human pressure impacting continental shelves globally. However, due to ongoing data deficiencies for smaller coastal vessels, the effects of trawling on nearshore seabed ecosystems are poorly understood. In Europe, the Water Framework Directive (WFD) provides a framework for the protection and improvement of coastal water bodies. It requires member states to track the status of 'biological quality elements' (including benthic macrofauna) using WFD-specific ecological indicators. While many of these metrics are sensitive to coastal pressures such as nutrient enrichment, little is known about their ability to detect trawling impacts. Here, we analysed a comprehensive data set of 5885 nearshore benthic samples – spatiotemporally matched to high-resolution trawling and environmental data – to examine how these pressures affect coastal benthos. In addition, we investigated the ability of 8 widely-used benthic monitoring metrics to detect impacts on benthic biological quality. We found that abundance (*N*) and species richness (*S*) were strongly impacted by bottom trawling. A clear response to trawling was also observed for the WFD-specific Benthic Quality Index (*BQI*). Relationships between *N* and *S*, and trawling were particularly consistent across the study area, indicating sensitivity across varying environmental conditions. In contrast, WFD indices such as AZTIs Marine Biotic Index (*AMBI*), multivariate AMBI (*M-AMBI*), and the Danish Quality Index (*DKI*), were unresponsive to trawling. In fact, some of the most heavily trawled areas examined were classified as being of 'high/good ecological status' by these indices. A likely explanation for this is that the indices are calculated using species sensitivity scores, based on expected species response to eutrophication and chemical pollution. While the *BQI* also uses species sensitivity scores, these are based on observed responses to disturbance gradients comprising a range of coastal pressures. Given the prominent use of *AMBI* and *DKI* throughout Europe, our results highlight the considerable risk that the metrics used to assess Good Ecological Status (GES) under the WFD may fail to identify trawling impacts. As trawling represents a widespread source of coastal disturbance, fishing impacts on benthic macrofauna may be underestimated, or go undetected, in many coastal monitoring programmes around Europe.

1. Introduction

Bottom trawling (hereafter trawling) is a fishing method where bottom-contacting fishing gears are towed over the seabed to capture demersal fish and invertebrate species (Eigaard et al., 2016; O'Neill and Ivanović, 2016). These activities can disturb or kill seabed biota (Hiddink et al., 2017; Lambert et al., 2011), affect important benthic processes (Bradshaw et al., 2021), and reduce the functionality of seabed communities (Bremner et al., 2003; McLaverty et al., 2021; Tillin et al., 2006). European waters are some of the most heavily trawled areas globally (Amoroso et al., 2018), with the majority of trawling effort concentrated in coastal areas (Eigaard et al., 2017). However, these coastal regions are also important areas of high biological productivity (Watanabe et al., 2018), supporting several seabed ecosystem functions

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(Waldbusser et al., 2004), and mosaics of habitat essential for the shelter, feeding, and breeding of marine species (Kritzer et al., 2016).

The introduction of key marine environmental policy directives in Europe, such as the Water Framework Directive (WFD) (2000/60/EC), has placed a greater emphasis on the quantification of ecological status in marine waters. The WFD relates specifically to groundwater, inland surface waters, transitional waters, and coastal areas. It defines coastal waters as the nearshore area extending 1 nautical mile (nm) from shore, or up to 12 nm where chemical status is assessed. In these areas, member states are required to monitor and assess the state of biological quality elements, such as benthic macrofauna. This is carried out using ecological indicators, designed to measure and track progress towards Good Ecological Status (GES), as defined by reference or desired conditions (Rice et al., 2012; Van Hoey et al., 2010). Due to the differing marine conditions across European coasts, the WFD allows indicators to be developed on a country-by-country basis, albeit requiring rigorous inter-calibration between predefined ecoregions (Borja et al., 2007, 2009). This has resulted in the development of a range of benthic indicators used to assess and monitor the quality of coastal macrofauna (Borja et al., 2015; Van Hoey et al., 2010).

Metrics selected for use under the WFD are typically designed to address its specific terms and definitions (Vincent et al., 2002), resulting in a preference for multi-metric indices i.e. single value metrics that uses several indicators in their calculation. These indices typically include abundance (*N*), species richness (*S*), a diversity index, usually Shannon's diversity index (H') (Shannon, 1948), as well as species sensitivity scores, used to reflect the relative abundances of stress tolerant/sensitive taxa in a sample. Most nations (including Denmark), use indices that incorporate species sensitivity scores from AZTIs Marine Biotic Index (*AMBI*) (Borja et al., 2000), while others use scores from the Benthic Quality Index (*BQI*) (Rosenberg et al., 2004). In either case, most WFD indices have been shown to be adept at monitoring diffuse coastal pressures such as eutrophication (nutrient enrichment) and chemical pollution, a key focus of the WFD (Borja et al., 2015). Nevertheless, comparatively little attention has been given to the response of indices to more direct physical pressures, such as trawling. Given that coastal trawling is widespread throughout Europe (Eigaard et al., 2017), and has been shown to reduce benthic abundance (Gislason et al., 2017), biomass (McLaverty et al., 2020a), functionality (McLaverty et al., 2021), and community composition (Bromhall et al., 2022) in Danish waters, this aspect merits further consideration. This is particularly the case as evidence from offshore areas suggest that WFD indices may be poor indicators of trawling disturbance (Gislason et al., 2017).

While the ability of monitoring programmes to detect and track pressures can be restricted by e.g. poor or inadequate coverage (Nygård et al., 2020), the choice of metrics used to carry out the task is also of critical importance (Rossberg et al., 2017). This is particularly the case in WFD monitoring, where a single indicator is applied to each biological element. However, selecting ecological indicators for a specific or general purpose is known to be problematic (Dale and Beyeler, 2001; McLaverty, 2020). In order to better select indicators for fisheries management, several criteria have been recommended as key properties (Rice and Rochet, 2005). However, when these criteria have been applied to evaluate the effectiveness of common ecological indicators, many were deemed to be *"unsuitable for the monitoring and assessment of bottom trawl impacts"* (Hiddink et al., 2020). One of the main reasons for this is that trawling-induced disturbance is difficult to separate from natural environmental variation (Szostek et al., 2016; Van Denderen et al., 2015). This is exacerbated in the presence of other human pressures, such as eutrophication and associated oxygen depletion, both of which are common around Danish and European coasts (Bromhall et al., 2022; Ferreira et al., 2011; McLaverty et al., 2020a; Petersen et al., 2020). Given that shallow water benthos recover relatively rapidly from disturbance (Jones, 1992), and strong natural (Aldridge et al., 2015) and human pressures (Halpern et al., 2008) are found in coastal areas, it has been assumed that nearshore fishery impacts are minor, or at least difficult to detect. However, little empirical evidence exists to support this assumption, especially due to ongoing deficiencies in fisheries data for nearshore fishing vessels. This, coupled with the WFD's lack of focus on trawling, mean there is a risk that nearshore benthic quality assessments may fail to identify trawling impacts on seabed biota, leading to a situation where monitoring programmes overestimate benthic quality in coastal areas.

A continuing obstacle to nearshore trawling impact assessments has been poor data coverage for coastal vessels. Certainly, fishing vessel monitoring has improved greatly with the introduction of Vessel Monitoring Systems (VMS) and Automatic Identification Systems (AIS). These satellite-based tracking systems gather data on the position, speed, and course of fishing vessels, which can be combined with logbook information to estimate the distribution of commercial fishing activities (Hintzen et al., 2012; Amoroso et al., 2018; Shepperson et al., 2018; Natale et al., 2015). However, the standard methodologies used to estimate vessel activities are poorly suited to coastal fisheries. VMS have only been mandatory for fishing vessels ≥15 m since 2005 (EC 2244/2003), and ≥12 m vessels since 2012 (EC 1224/2009), while AIS has been a requirement on vessels ≥15 m since 2014 (EC 1224/2009). Coastal vessels tend to be smaller, and are often not covered by these systems. Additionally, VMS receivers log data infrequently (e.g. once per hour in Denmark), requiring methods to detect fishing activity (Poos et al., 2013), and interpolate fishing tracks (Hintzen et al., 2012). Given that VMS data are mainly associated with larger vessels, the available interpolation methods are better suited to estimate fishing tracks which follow longer, linear, hauls. On the other hand, AIS log data in the order of seconds, and yield better estimations of fishing activity and location (Thoya et al., 2021). AIS are, however, associated with much lower coverage, which remains a major drawback (Shepperson et al., 2018). Dredgers targeting shellfish in Denmark are obliged to carry a so-called Black Box System (BBS). These were introduced in 2012 in Limfjorden, and in 2013 along the east coast of Jutland. BBS records the vessel position, speed, and winch activity every 10 s, allowing for highly accurate estimations of fishing activity (Nielsen et al., 2021). It is generally accepted that a combination of different sources of fisheries data will provide the most accurate spatial estimates of fishing activity (Thoya et al., 2021). In this study we therefore make use of a novel method to combine these data sources, to estimate nearshore fishing effort at a high accuracy.

This study examines the effects of bottom trawling and environmental pressures on nearshore benthic macrofauna, and evaluates the ability of 4 widely used benthic indicators and 4 WFD indices to detect these impacts. We matched 5885 macrofauna samples to bottom trawling intensity estimates and key physical variables across the Danish coastal zone. Bottom trawling intensity was estimated by combining VMS, AIS, and BBS data to provide high-resolution pressure estimates (~100 m resolution) covering a national coastal area over a 12-year period. Given the transitional location of Denmark between the North and Baltic Seas, the study area provides an appropriate setting to test the performance of indicators under wide ranges of salinity, temperature, bottom currents, and commercial trawling intensity. The results of this study offer new evidence of the ecological effects of bottom trawling in nearshore waters, and provide an evaluation of how well these metrics can monitor ecological status in coastal areas. We expect the results to be directly relevant to northern European and Baltic areas, but also to areas experiencing variable coastal environmental conditions and high bottom trawling.

2. Materials & methods

2.1. Study area

Denmark has one of the longest coastlines in Europe (~7310 km), stretching from the North Sea coast to the west, to the islands of Bornholm and Ertholmene in the Baltic Sea (Fig. 1). Coastal seabed

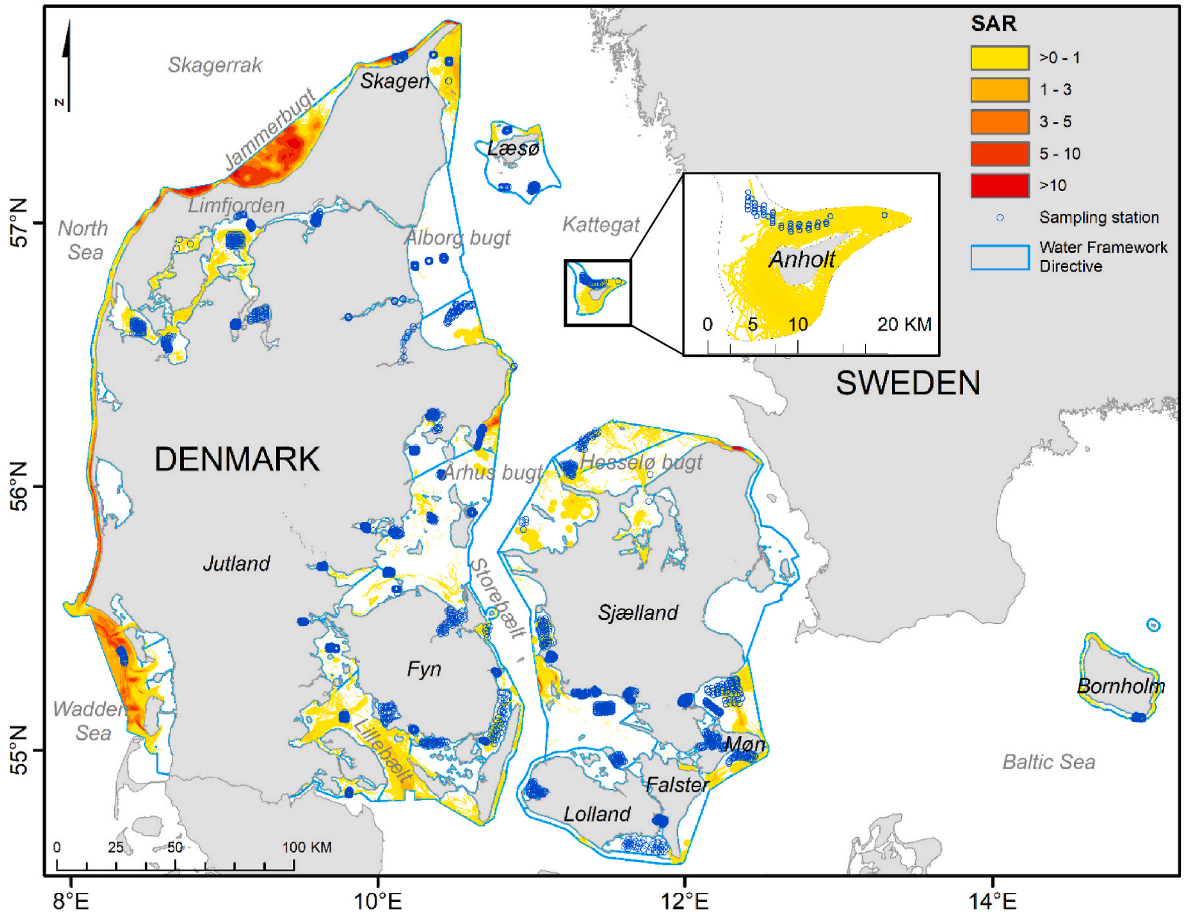

Fig. 1. Location of benthic sampling stations (blue circles) in Danish Water Framework Directive (WFD) areas. For presentation purposes, bottom trawling effort is shown as mean annual effort (based on grid cells of 100 × 100 m for all gears pooled) over the study period (2005–2017). Place names are provided in black and sea areas in grey text. Inset shows a grid of 41 individual sampling stations at the Anholt monitoring site, together with high-resolution trawling pressure. SAR: swept area ratio. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

conditions are characterised by large differences in salinity, temperature, current velocity, and nutrient loads. The western North Sea coast is marine influenced, and runs latitudinally from the macrotidal Wadden Sea in the south to the microtidal Skagerrak in the north. On the east coast, conditions also differ substantially from the Kattegat in the north, to the Baltic Sea in the southeast. A deep inflow of high saline (30–34%) marine water channels down from the North Sea and Skagerrak into the Kattegat, through the Belt Seas and Øresund, and into the Baltic Sea. This inflow is counterbalanced by a surface outflow of brackish water originating in the Baltic (Møller, 1996). Salinities in the western Baltic Sea are comparatively low (<10%). Tidal ranges on the west coast are relatively high, reaching ~2 m around the Wadden Sea. By contrast, east coast tides are is usually <20 cm, although large changes in water level (~1 m) are caused by winds and high/low air pressures (Conley et al., 2000). Habitat types sampled across the study area were chiefly composed of sand or mud (78% of samples), or a combination of the two i.e. sandy mud or muddy sand (96%) (Table S1). Muddy sediments are widespread along the southeast coastline and e.g. in Limfjorden, while sandy sediments are found predominantly along the west and northeast coasts, and around the islands of Læsø and Anholt (Figure S1 f).

Bottom trawling accounts for the highest effort of all fisheries in Danish waters (Skov et al., 2020), and is prevalent in many nearshore areas (Fig. 1). Otter trawls, beam trawls, dredgers, and Danish seines account for the majority of nearshore trawling activity. Beam trawling for brown shrimp *Crangon crangon* is dominant along the west coast (Wadden Sea and North Sea coast). In the northwest and Jammerbugt area, trawling is predominantly undertaken using Danish seines, targeting European plaice *Pleuronectes platessa,* and to a lesser extent Atlantic cod *Gadus morhua* and haddock *Melanogrammus aeglefinus*. The east and southeast areas (southern Fyn, Møn, Hesselø Bugt, Bornholm) are mainly trawled by otter trawls targeting a mix of Atlantic cod and European plaice. Otter trawls targeting Norway lobster *Nephrops norvegicus* are concentrated in muddy grounds around Læsø, Anholt and east of Skagen. Internal waters, such as the Limfjorden, and several areas along the east coast and in Isfjord, are trawled by dredgers for blue mussel *Mytilus edulis*, cockles *Cerastoderma* spp., and flat oyster *Ostrea edulis* (Gislason et al., 2021).

2.2. Fauna & sediment data

Benthic macrofauna and sediment data were gathered from the ODA database (Surface Water Database, ODA - https://odaforalle.au.dk), provided by the Danish national environmental monitoring programme (NOVANA). The database holds information on benthic and sediment samples collected annually (in spring or autumn) from fixed sites around the Danish coast (Fig. 1) using a HAPS corer (0.0143 m^2). At each monitoring site, multiple replicate samples are collected in a grid pattern (e.g. Fig. 1 inset). The number of stations per grid ranges from 5 to 45, although the vast majority of grids contain >40. During sampling, sediment is rinsed through a 1 mm mesh sieve, and the residuum preserved in 96% ethanol. In the laboratory, macrofaunal individuals are identified to the lowest possible taxon, enumerated, and weighed (detailed sampling methods are provided by Josefson and Hansen (2014)). To match benthic data with available fishing pressure and environmental data, macrofauna and sediment data from nearshore areas between the years 2005 and 2017 were extracted from the database. As biomass weights were provided as either dry weight or wet weight (differing between years), biomass values were standardised to ash-free dry weight (AFDW) using conversion factors provided by Ricciardi and Bourget (1998). Samples collected from enclosed or semi-enclosed estuaries and basins that were outside the envelope of the available hydrodynamic models (see section below) were not included in the analysis (e.g Nissum Fjord, Ringkøbing Fjord, Lillestrand, Kalundborg Fjord, Jammerland Bugt, west Mariager Fjord, and Sejerø Bugt). In addition, these areas are known to be unrepresentative of nearshore coastal areas. Additional information on data preparation is provided in Supplement Text S1. Semi-quantitative sediment descriptions associated with each macrofaunal sample were also provided in the database. These were standardised and converted to the Folk classification system (Folk, 1954), as described in Supplement Text S2. Subsequent to extraction and preparation, 5885 individual macrofaunal samples remained for analysis.

2.3. Environmental data

As a high degree of environmental variation was expected across the study area, we gathered high-resolution modelled data for 3 key physical water parameters; bottom current velocity (mean), bottom salinity (minimum), and bottom temperature (mean). Yearly parameter averages at each sampling station were extracted from several MIKE 3 HD Flexible Mesh (FM) models (Danish Hydraulic Institute, 2017) for the years 2005–2016. Further information on the models is provided in Supplement Text S3. Latitude was also included as a predictor variable to account for potential variation not described by the physical data. The environmental data used in analysis are summarised in Fig. S2 and Table S1.

2.4. Trawling intensity

Each of the 5885 macrofauna samples were matched with 1 year of trawling pressure data, estimated back in time from the day of sampling. To do this, all bottom trawling data (VMS, AIS, and BBS) in Danish nearshore areas for the years 2005–2017 was combined with logbook data. These sources of spatial fishing data comprise information on the vessels position, speed, and heading, and the daily logbooks contain information on the gears used, their configuration, and the species retained. The logbook and spatial data were merged according to Hintzen et al. (2012). Because of differences in the availability and recording frequency of the spatial data systems, we developed a hierarchical merging method which prioritises the most detailed data source (BBS, then AIS, then VMS). Gear-specific speed profiles were used to determine if a vessel was fishing, and only those recordings were kept in the analysed data set (Poos et al., 2013). We then reconstructed the fishing tracks by interpolating the fishing recordings of a vessel (Hintzen et al., 2012), combined with the modelled width of the fishing gear (Eigaard et al., 2017). For Danish seines, a deviating method was applied to compensate for their distinct fishing pattern (described further in Fig. S2). Trawling intensity was ascribed to each sampling station by creating circles with 100 m radius at each station, and aggregating the total surface of fishing tracks (swept area) within the circle over a 1 year period prior to sampling. The total swept area was then divided by the surface of the circle to determine the Swept Area Ratio (SAR year^{-1}), the metric used here to denote trawling intensity. As a different routine was used to estimate Danish seining intensity, and as this relatively light gear may result in different impacts to the seabed, we repeated the analysis excluding all records of Danish seines. Those results are provided in Fig. S3.

2.5. Benthic indicators

Macrofauna data were used to calculate 8 benthic coastal monitoring metrics. These comprised 4 univariate indicators; total abundance (*N*), species richness (*S*), Shannon diversity (*H′*), and biomass, and 4 WFD multi-metric indices; AZTIs Marine Biotic Index (*AMBI*), the multivariate *AMBI* (*M-AMBI)*, Benthic Quality Index *(BQI)*, and the Danish Quality Index *(DKI)*. *N*, *S*, and *biomass* represent relatively simple community metrics, while *H′* accounts for both *S* and *N*, and thus the evenness of a community. *AMBI* evaluates benthic quality by estimating the proportion of disturbance sensitive taxa in a sample relative to disturbance tolerant taxa (Borja et al., 2000). Species sensitivity/tolerance is categorised using five ecological groups, ranked by their sensitivity to eutrophication and pollution, based on literature and expert judgement. Unlike the other indicators in this study, *AMBI* describes high ecological quality by low values (0 = high quality), while poor quality is reflected by high values (7 = poor quality). To aid interpretation, we present the results for *AMBI* on an inverse scale. *M-AMBI* is the multivariate extension of *AMBI*, and is calculated by combining *H′*, *S* and *AMBI* in a multivariate factor analysis approach. The *BQI* index is the Swedish national benthic indicator used under the WFD, and combines *N* weighted species sensitivity scores with *S* into a single index. The *DKI* is the Danish national benthic indicator used for Danish WFD assessments, and combines several other indicators such as *N*, *S*, *AMBI*, and *H'* into a single index. Further descriptions of the metrics and their calculations are provided in Table S2. Metrics were calculated for each macrofauna sample, with no pooling of samples. This was done as monitoring sites typically varied in numbers of stations, and due to the large distances between stations (Fig. 1 inset). Samples containing no macrofauna (n = 327) were given a value of 0 (or 7 in the case of *AMBI*).

2.6. Data analysis

Generalised linear mixed effects models (GLMMs) were used to analyse the response of indicators to bottom trawling and environmental pressures. Each GLMM included trawling intensity (SAR year^{-1}), sediment type, water depth, bottom temperature, bottom salinity, bottom current speed, and latitude as fixed effects. We included the monitoring site and sampling year as random effects to account for potential factors other than those encompassed by the fixed effects, the non-independence of samples from within a site, as well as random inter-annual changes in benthic recruitment success. Note that the model for *S* also included *N* as predictor variable, to account for potentially sources of spatial and temporal variation (from e.g. recruitment success) caused by correlations between *N* and *S* (Gislason et al., 2017). Indicators were modelled using a negative binomial (*N*), Poisson (*S*), Tweedie (*H′, biomass*), Gaussian (*AMBI, BQI*), or beta (*M-AMBI, DKI*) distribution. The analysis of GLMMs were performed using the glmmTMB package (Brooks et al., 2017). We quantified the importance of each predictor variable using Relative Variable Importance (RVI) scores, which estimates and quantifies predictor importance using multi-model inference and provides a score irrespective of statistical significance. Scores were determined from the weighted AIC, calculated across all permutations of a global model (Burnham and Anderson, 2004). Variables can thus be interpreted as highly important (RVI >0.9), moderately important (RVI 0.9–0.6), or low to no importance (RVI <0.6). We interpreted a RVI score of >0.6 as a clear response to a pressure RVIs were calculated using the R package ‘MuMin’ (Barton, 2013). Positive or negative relationships between indicators and predictors were derived from the lowest AIC model (Burnham and Anderson, 2004). We present the results of models with delta AIC <5 in Table S3 of the Supplementary Material. Diagnostics for the best performing models were checked using the R package ‘DHARMa’ (Hartig, 2016). Collinearity was low between the fixed predictors (all <0.4). As we were interested in the ability of indicators to detect trawling, relationships between the variables were visualised using marginal effects plots using the ggeffects package (Lüdecke, 2018).

3. Results

3.1. Response of monitoring metrics to environmental pressures and bottom trawling

Depth, latitude, sediment type, and bottom trawling intensity were particularly important predictors, exhibiting RVI scores above 0.9 in the GLMMs (Fig. 2). Depth was an important predictor of *N*, *BQI*, and *DKI* (all RVI >0.9), and to a lesser extent *S* (RVI >0.6). Relationships between depth and these indicators were all negative, meaning that benthic macrofaunal quality declined with depth. With the exceptions of *N* (RVI 0.78) and *AMBI* (RVI 0.99), benthic macrofaunal quality increased with latitude. Latitude was also particularly important for *biomass* (RVI 0.84) and *BQI* (RVI 0.99). Sediment type had high importance (RVI >0.9) for several metrics, including *N*, *S*, *M-AMBI*, *BQI*, and *DKI*. By contrast, *biomass* and *AMBI* values were relatively unaffected by differences in sediment type. Temperature was of moderate importance for *N*, *biomass*, and *BQI* (all 0.69). However, the differing relationships between indicators would suggest a mixed effect of temperature i.e. negative relationship with *N*, and positive relationship with *biomass* and *BQI*. Salinity and current speed were generally of little to no importance in the models. The degree of explained variance in the models ranged considerably, from high values of 72% (*S*) and 63% (*N*), down to values of 7% (*DKI*), and 4% (*M-AMBI*).

Higher bottom trawling intensity was associated with lower values for all indicators except *AMBI* and *DKI* (Fig. 2). Trawling was particularly important for *N* (RVI 0.97), *S* (RVI 0.99), and *BQI* (RVI 0.77), but was of little to no importance for *H'* (RVI 0.45), *AMBI* (RVI 0.37), *M-AMBI* (RVI 0.3), and *DKI* (0.27). Fig. 3 reveals that *N* was markedly higher at untrawled sites (Fig. 3a), and reduced at heavily trawled sites (>20 SAR year^{-1}). Although the decline was not as sharp for *S* (Fig. 3b), richness declined steadily from the untrawled areas (max. 29 species) with increasing trawling intensity (max. ~18 species at sites >20 SAR year^{-1}). Values of *BQI* also showed a clear negative relationship with trawling (Fig. 3e), despite a subset of sites with *BQI* values of ~12 recorded at high trawling intensity. This group of sites were chiefly composed of samples from the Wadden Sea and Jammerbugt. Although most of the high *biomass* values were recorded in untrawled areas (Fig. 3c), the overall relationship with trawling was weak (RVI 0.28). This is likely a result of the relatively low number of high *biomass* values recorded in the dataset. The majority of untrawled *biomass* samples were <2 g, which was generally comparable to *biomass* in the trawled areas.

The indicators *H'* and *M-AMBI* exhibited no interpretable relationships with trawling (Fig. 3d/f). Conversely, *AMBI* (Fig. 3e) and *DKI* (Fig. 3g) exhibited positive trends (Fig. 3e/g), although these were not found to be related to trawling (RVI: 0.33 and 0.27, respectively). Excluding Danish seines from the analysis resulted in only minor differences in the overall performance of indicators (Fig. S3). The main difference being that the RVI for *BQI* (RVI 0.47) fell outside of the 0.6 threshold, meaning that it was no longer deemed important.

3.2. Spatial variation of monitoring metrics in relation to fishing pressure

Metrics and their associated trawling intensity were mapped to explore potential spatial patterns (Fig. 4). A relatively clear and consistent relationship between high *N* (darker red colour) and lower trawling intensity (smaller circles), and lower *N* (light red/white) and high trawling (large circles) was evident across the study area (Fig. 4a). The consistency of this relationship across sites would suggest that trawling effects on *N* were consistent across environmental conditions. A similar relationship was observed for *S* and trawling, albeit to a lesser degree (Fig. 4b). The association was less clear where high *S* occurred in moderately trawled areas (e.g. Anholt and Hesselø Bay). Although some lightly trawled areas had higher *BQI* values (Århus Bugt, Ålborg Bugt, Anholt), areas such as Skagen and Wadden Sea were also associated with high trawling (Fig. 4h), and potentially explaining the lower RVI score for *BQI*. In addition, low *BQI* values were often associated with low salinity areas, irrespective of trawling intensity (e.g. Sjælland, Fyn, Bornholm and in Limfjorden). This may suggest that *BQI* is better suited to detect trawling impacts in open marine coasts, as opposed to smaller enclosed inlets, broads and fjords which are freshwater influenced. The highest *biomass* values were recorded in relatively few untrawled or lightly trawled sites in the southern and western Kattegat (Fig. 4c). This pattern was inconsistent across locations, with many low *biomass* values observed in untrawled or lightly trawled areas (e.g. southern Sjælland, southern Fyn, and Limfjorden). *H'* was relatively high in areas such an Anholt, Skagen, Hesslø Bugt, and northern Lillebælt. Nevertheless, there was no apparent association between trawling and *H'* regardless of location. While *AMBI*, *M-AMBI*, and *DKI* values varied across the study area, there appeared little relationship between these indices and trawling intensity. In fact, high values were often recorded in heavily trawled areas along open coasts (e.g. Skagen and Wadden Sea) while low values were often associated with lightly trawled areas along the east coast (Århus Bugt, south Sjælland, south Fyn).

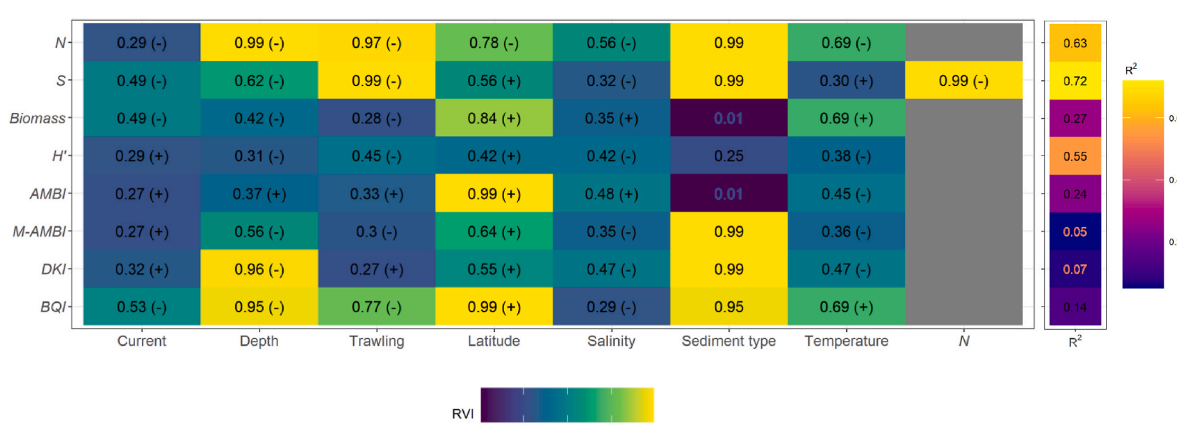

Fig. 2. Relative variable importance (RVI) of environmental and fishing pressures on macrofaunal indicators. RVI scores can be interpreted as >0.9 = highly important, 0.9–0.6 = moderately important, <0.6 = low to no importance. Symbols indicate the direction of relationships between variables: positive (+) or negative (−). Note: *AMBI* values are reversed to aid interpretation, while sediment type is a categorical variable. R^2 values indicate variance explained by best fitting model (Table S3).

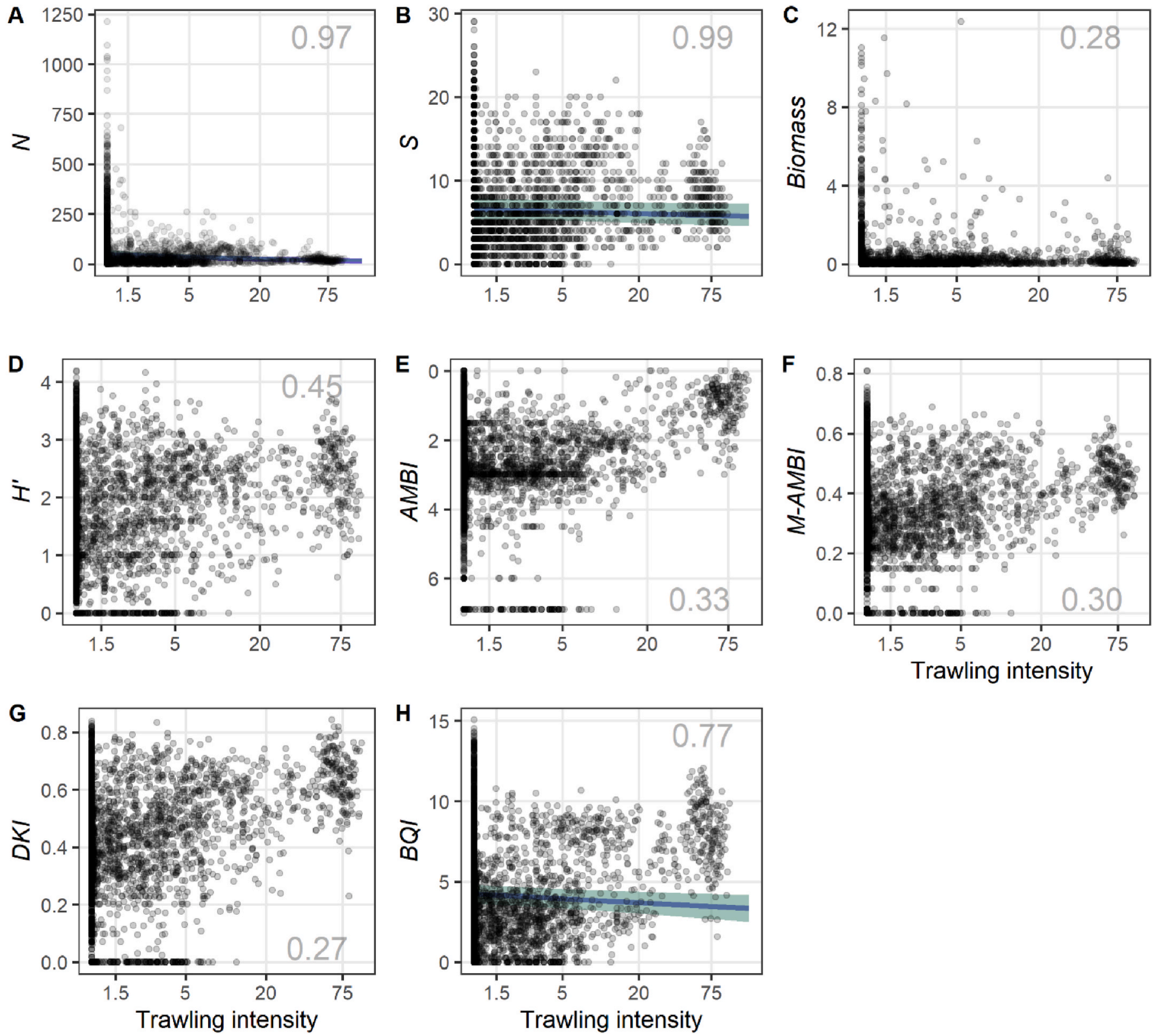

Fig. 3. Relationships between monitoring metrics and trawling intensity (SAR year^{-1}). Regression lines are shown for RVI scores >0.6. Greyed areas represent 95% confidence intervals. Inset values display the associated RVI score. Raw observations are overlaid as points. Trawling intensity is presented on a logarithmic scale for presentation purposes. Note: *AMBI* values are reversed to aid interpretation. Biomass values are in grams.

4. Discussion

Until now, studies investigating the impacts of bottom trawling in coastal areas have been hampered by poor data coverage for nearshore vessels. We used a hierarchical merging method in this study to combine complementary types of fisheries data (VMS, AIS, and BBS) and logbook information, resulting in high resolution fishing pressure estimates for the entire Danish coastal area. This allowed us to observed, for the first time, that coastal trawling has reduced benthic abundance (*N*) and species richness (*S*) across Danish nearshore water bodies. The *BQI* index, used to assess benthic quality in Swedish WFD areas, detected these changes to the community. However, the other WFD indices examined (*DKI*, *AMBI*, and *M-AMBI*) showed no response to trawling, nor did *H'* or *biomass*, corroborating observations from offshore waters (Gislason et al., 2017). The results of this study highlight a considerable risk that WFD metrics used monitor the health of coastal benthic ecosystems, and efforts towards Good Ecological Status (GES), may be unable to detect trawling impacts on the seabed. Although the *DKI* is primarily used in Denmark, *AMBI* is used as a key component in the majority of European benthic monitoring indices (Borja et al., 2015). These findings are therefore highly relevant to coastal monitoring programs and national assessments of GES based on these metrics. Given the widespread nature of trawling in European coastal areas (Eigaard et al., 2017), and the role of benthic macrofauna in seabed ecosystem function (Gammal et al., 2017; Kristensen et al., 2014), our results may also have implications for management of human pressures on coastal ecosystems.

The metrics *N* and *S* exhibited particularly clear negative relationships with trawling across varying environmental conditions. *N* was markedly reduced in the trawled areas, and showed a clear relationship with trawling in most locations, indicating a generally high sensitivity. The responsiveness of *N* to trawling is in line with previous studies in Danish waters (Bromhall et al., 2022; Gislason et al., 2017; McLaverty

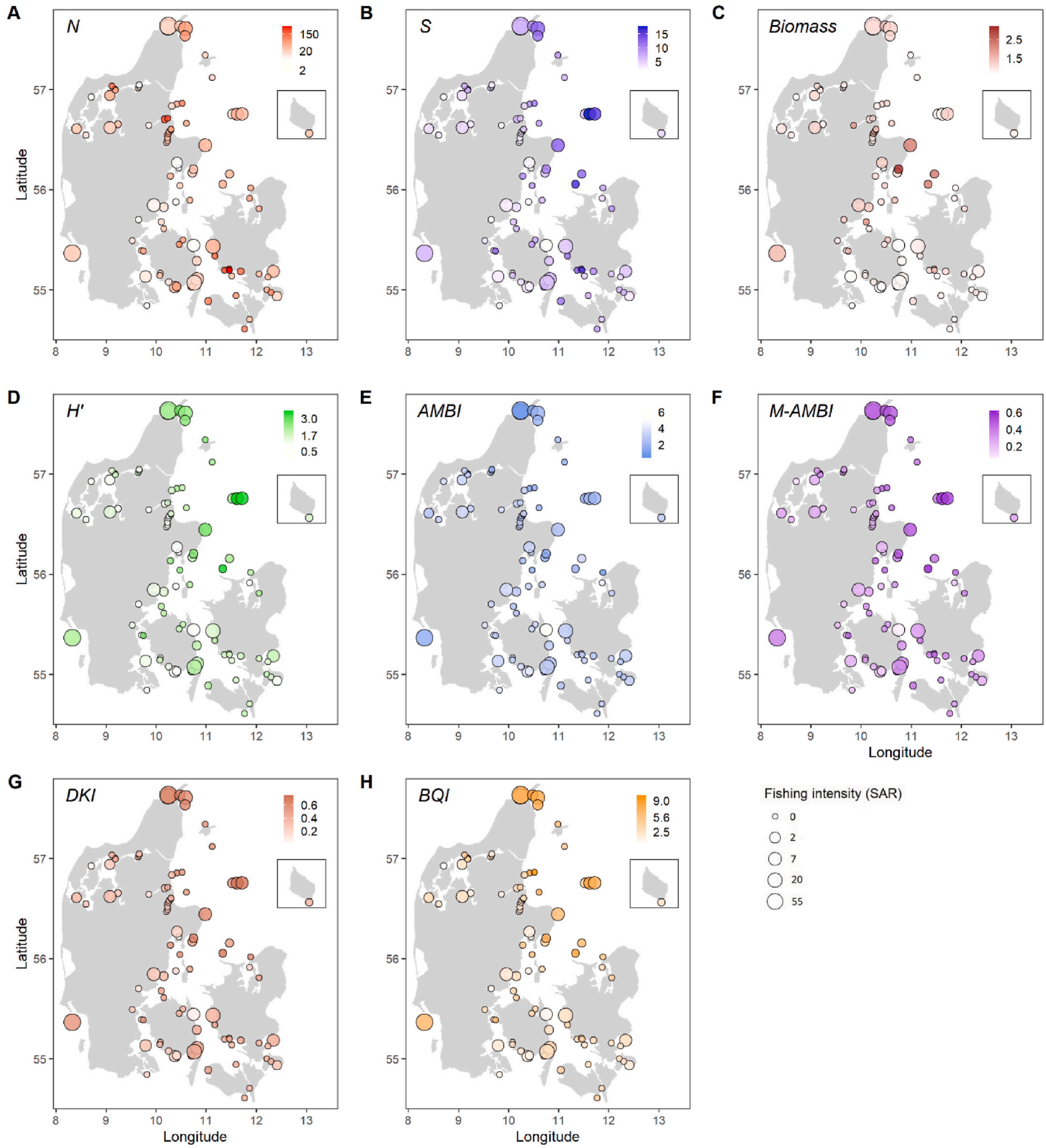

Fig. 4. Spatial distribution of monitoring metrics as a function of trawling intensity (SAR year^{-1}). The scale for *N*, *H′*, and *BQI* are log transformed to aid interpretation. The island of Bornholm is shown in the inset. Note: AMBI scale is reversed.

et al., 2020b) and other regions (Hiddink et al., 2020). In contrast, the literature regarding the sensitivity of *S* to trawling is less conclusive (Hiddink et al., 2020). For example, *S* has been shown to exhibit inconsistent responses to trawling between studies in Limfjorden (Bromhall et al., 2022; McLaverty et al., 2020a) and the Kattegat (McLaverty et al., 2020b; Sköld et al., 2018). However, these inconsistencies can often be due to the sensitivity of *S* to sampling design and effort (Chase and Knight, 2013; Hillebrand et al., 2018). In addition, the inherently strong correlation between *S* and *N* in a sample can introduce complicating sources of spatial and temporal variation (e.g. from inter-annual recruitment success), which has been shown to undermine the effectiveness of *S*-based multi-metric indices (Gislason et al., 2017). *Biomass* and *H′* did not exhibit changes attributable to trawling. Evidence regarding the inability of *H′* to detect trawling effects is mounting (Cyrielle et al., 2020; Hiddink et al., 2020). A possible reason for this is that *H'* is relatively sensitive to natural variation, as well as interspecific processes such as competition and predation within the community. These processes are potentially less affected by trawling than by differences in community dynamics between areas and over time (Svensson et al., 2012). Although *biomass* is generally considered a highly effective trawling indicator (Hiddink et al., 2020), its responsiveness to trawling can be reduced in the presence of high environmental variation, particularly in eutrophic areas (McLaverty et al., 2020a). Indeed, the strong latitudinal gradient in *biomass* observed in this study may indicate that the sharp transition from marine conditions in the north, to more brackish conditions in the south, masked any responses to trawling, particularly as *biomass* is often lower in brackish conditions (Edgar and Barrett, 2002).

In terms of the WFD-specific indices, our results found no relationship between trawling and *AMBI*, *M-AMBI*, or *DKI*. Given the prominent role of these indices in WFD monitoring, these findings are potentially significant. A likely explanation is that each of these indices were originally developed to monitor the effects of diffuse coastal pressures such as eutrophication and oxygen depletion (Borja et al., 2000; Josefson et al., 2009; Muxika et al., 2007). Accordingly, *AMBI* (and thus *M-AMBI* and *DKI*) are calculated using scores of species sensitivity to eutrophication, based on expert-judgement (Borja et al., 2000). Benthic quality is then assessed by estimating the proportion of pollution tolerant species, relative to the proportion of pollution sensitive species. Rigorous inter-calibration exercises have confirmed that these indices are indeed sensitive to various pollution sources (Borja et al., 2015), and to several human pressures (Borja et al., 2019). However, there is little evidence to suggest that pollution sensitive species are also sensitive to physical disturbance. On the contrary, *AMBI* has been shown to be a relatively poor indicator of physical impacts (Muxika et al., 2005), and in addition, physically disturbed communities can be dominated by pollution sensitive species, providing they do not receive high nutrient enrichment (Salas et al., 2004). Under such circumstances, physically disturbed communities would be ascribed 'high ecological quality', according to *AMBI* (Grémare et al., 2009; Labrune et al., 2006). This would explain the observed positive trend between trawling and *AMBI*, and why 'high ecological quality' *AMBI* values were associated with some of the most heavily trawled sites in our study. An alternative explanation is that the most heavily trawled samples in the analysis are located on the west coast i.e. in the Wadden Sea and Jammerbugt, where diversity and biomass is naturally higher. This aspect could potentially skew the results, and serve to mask trawling impacts when compared to east coast samples. However, these trends were not apparent for *N*, *S*, or *biomass*, which all declined clearly across the trawling gradient. Furthermore, excluding records of Danish seines from the analysis did not affect the results (Fig. S3). Given that the calculation of *DKI* is based on *AMBI* (Table S2), it is understandable why *DKI* and *AMBI* performed similarly, and why *DKI* has shown either mixed responses (Hansen and Blomqvist, 2018), or no response to trawling in the past (Eigaard et al., 2020; Gislason et al., 2017). In either case, there is strong evidence here to suggest that pollution/eutrophication monitoring indices are poorly suited to identify benthic impacts from physical pressures such as trawling.

The other WFD index examined in this study, the *BQI,* exhibited a clearer response to trawling. This may be due to the fact *BQI* chiefly uses *N* and *S* in its calculation (both sensitive to trawling), and a different set of species sensitivity scores than *AMBI*/*M-AMBI*/*DKI*. In contrast, the *BQI* sensitivity scores are based on observed species responses to an artificial disturbance gradient, composed of several pressures such as hypoxia, physical disturbance, and toxic substances (Leonardsson et al., 2015). The scorings may therefore better reflect the multiple pressures (anthropogenic and natural) typically occurring in coastal and nearshore areas. The scores were also determined from areas such as the Skagerrak and Kattegat, which may explain why relationships between trawling and *BQI* appeared stronger in the northern marine influenced areas. Furthermore, when data from Jammerbugt were excluded from the analysis (Fig. S3), the ability of *BQI* to detect trawling was reduced, also potentially indicating a higher sensitivity under fully marine conditions. The sensitivity of *BQI* to trawling is supported by studies in nearby deeper waters (Gislason et al., 2017; Sköld et al., 2018), and the index has been shown to be sensitive to other forms of physical disturbance (Trannum et al., 2021). Although we are not aware of any trawling studies using *BQI* in low saline areas, which may warrant further investigation, the *BQI* is successfully used as the primary indicator for soft sediment areas in the low salinity HELCOM (Baltic Sea) area (Nygård et al., 2020).

4.1. Considerations

The hierarchical merging method provided novel estimations of high resolution coastal trawling pressure, however, some challenges remain regarding this endeavour. Fishing vessels ≤12 m (≤15 m before 2012) are not obliged to carry VMS, nor do all vessels ≤15 m necessarily use AIS (EU 1224/2009). On the other hand, coastal shellfish dredgers have all carried BBS since 2013 (Nielsen et al., 2021). Although the vast majority of Danish trawlers and seiners fishing in coastal waters are above 12 m, a small proportion of trawling total effort is potentially not included in our estimates. Furthermore, as SAR estimates are based on a ratio, they can be strongly affected by scale. While global or multi-regional studies (e.g. Amoroso et al., 2018) typically use trawling pressure estimates at the scale of kilometres, our study is based on a scale of ~100 m. The high resolution trawling data used herein may therefore led to differences between the results of this study, and those that use a coarser cell resolution. A separate consideration is that the results of this study are exclusively based on monitoring data. We were thus restricted to data gathered from national monitoring sites (Fig. 1), which resulted in a poor representation of highly trawled areas in Jammerbugt, the Wadden Sea, and other hotspots in south Fyn and Lillebælt. Given the lack of coverage in these areas, the data may poorly capture the range of parameters and pressures which impact benthos around the coast. However, given that the gaps in coverage overlapped with areas heavily impacted by trawling, we would consider the results of this study to be conservative, rather than an overestimate. Furthermore, eutrophication impacts large swathes of the Danish coast. Unlike trawling tracks, which can be mapped with relatively accuracy, the effects of eutrophication are diffuse, and difficult to spatially estimate without direct sampling. It is likely that many samples, in both trawled and untrawled areas, were impacted by elevated nutrients. Although we excluded enclosed and semi-enclosed estuarine bays from the study, we do not directly account for the effects of eutrophication and oxygen depletion, as these data were not available. It is possible that some degree of trawling impacts is masked by these pressures. Similarly, background nutrient enrichment may explain why some indices were unaffected by trawling. However, if this were the case, it would indicate that the metrics were unsuitable for monitoring the range of pressures found in coastal waters. Finally, although the study area represents a wide range of human and environmental pressures, we cannot unequivocally say these results are applicable universally, and further validation and testing is likely needed across geographic locations.

5. Conclusions and perspectives

We demonstrate here that bottom trawling negatively impacts benthic macrofauna in nearshore areas, and that these impacts may to go undetected by monitoring metrics commonly used to assess Good Ecological Status (GES). The *DKI*, used to assess benthic quality in Denmark, and *AMBI*, used in the calculation of many European benthic quality indices (Borja et al., 2015), were found to be unresponsive to trawling. These indices are therefore potentially unsuitable to provide assessments of seabed quality in coastal areas, where multiple pressures are present. Moreover, we observed that some of the most heavily trawled sites were ascribed 'high ecological quality', by *AMBI* and the *DKI*. On the other hand, the *BQI*, used to monitor Swedish WFD areas, declined with increasing trawling. The *AMBI* and *DKI* indices are tailored to monitor the effects of eutrophication, and given that nutrient enrichment remains an enduring issue in all marine regions (Korpinen et al., 2019), these metrics are likely to remain important tools in benthic monitoring. We therefore conclude that no single indicator can provide a 'silver bullet' for seabed monitoring, particularly in highly dynamic areas experiencing multiple pressures. It is likely that several metrics with complimentary properties, which can integrate aspects such as community structure, function, and sensitivity, are required to effectively track and measure coastal benthic health. In this regard, our results show *N*, *S,* and *BQI* to be effective indicators of trawling impacts in coastal areas. Despite a growing toolbox of metrics used to assess trawling on benthic macrofauna (e.g. De Juan and Demestre, 2012; González-Irusta et al., 2018; Hiddink et al., 2019; Serrano et al., 2022), indices developed under the WFD remain the primary monitoring metrics used in coastal areas. Given their potential inability to identify trawling impacts, coastal monitoring of benthic quality and GES may result in overly optimistic estimates of ecological status, and thus undermine the objectives of the WFD. We hope that the results of this study highlight that the impacts of coastal trawling are significant, and that key benthic monitoring metrics may be incapable of detecting changes in benthic quality caused by bottom trawling. Similarly, we hope that these results can inform exiting coastal monitoring programmes, and contribute to the future development of monitoring metrics.

Credit author statement

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Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2023.117510.

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