WORKING GROUP ON BIODIVERSITY SCIENCE (WGBIODIV; outputs from 2024 meeting)

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International Council for the Exploration of the Sea
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Contents

i Executive summary

We set out to provide a multifaceted perspective of biodiversity change across the Northeast Atlantic, identify areas where biodiversity is particularly at risk given abrupt or gradual responses of marine biota, and examine the efficacy of spatial protection measures to conserve and protect nature. We used a suite of structural and functional metrics across plankton, benthos and fish assemblages, and considered various climate change, fisheries management and marine artificial structure scenarios. Our work either directly informed Indicator Assessments for the OSPAR Quality Status Report 2023 or could be applied to inform ecosystem-based management. This contribution is not ICES advice but the work of a science working group (WGBIODIV).

Across the various biodiversity metrics, the abundances, distribution and productivity of key groups representing different trophic levels were in flux in many of the areas assessed. We provide a contribution on how benthic effect trait composition determines seabed ecosystem functions, how these vary along environmental gradients but are similarly vulnerable to bottom trawling. Many of the observed changes in plankton and predicted changes in fish species distributions and foodweb properties were affected by climate change. For instance, most plankton lifeforms, which include groups of phytoplankton, zooplankton, and ichthyoplankton, have been declining in abundance throughout the Northeast Atlantic in the long term and these changes were associated either directly or indirectly with environmental change linked to climate change. The distribution of species richness across the fish component of the foodweb is also projected to be affected by climate change with spatially extensive increases in species richness largely driven by the expansion in range of smaller piscivores with relatively low predator–prey mass ratios by 2095 under RCP 4.5. Based on projections from ecosystem models, we provide how proactive, regionalized Nature-Based Solutions could ensure resilience and attain Good Environmental Status in future, contributing to the recovery of both the ecological integrity and the socio-economic benefits of marine ecosystems. Finally, using a global meta-analysis, we find that decommissioning options aimed at repurposing marine energy infrastructure into artificial reefs may not provide the intended benefits. Despite the wealth of evidence of biodiversity change presented here, we highlight the need to develop international consensus on how to operationalize effective biodiversity targets where thresholds and tipping points are insufficient to 'bend the curve' towards nature recovery.

Keywords: *plankton, benthos, fish, Good Environmental Status, ecosystem structure and functioning*

ii Expert group information

Expert group name	Working Group on Biodiversity Science (WGBIODIV)
Expert group cycle	Multiannual
Year cycle started	2021
Reporting year in cycle	3/3
Chairs	Ute Jacob (Germany)
	Murray Thompson
Meeting venues and dates	7–10 June 2022, Online meeting
	24–28 April 2023, Nantes, France
	15–19 April 2024, Oldenburg, Germany

1 Using complementary measures to provide a multidimensional perspective of biodiversity change (ToR A)

1.1 A summary of contributions from WGBIODIV to the OSPAR Quality Status Report 2023 for plankton

OSPAR's strategic objective with respect to biodiversity and ecosystems is to protect and conserve marine biodiversity, ecosystems and their services to achieve good status of species and habitats and thereby maintain and strengthen ecosystem resilience. WGBIODIV members have been involved in the development and assessment of multiple different indicators on marine biodiversity.

1.2 Plankton Community Change indicator assessment

The Plankton Community Change indicator (PH1/FW5) describes changes in plankton functional types, or lifeforms (D1C6, D4C1, D4C2; Figure 1). Once lifeforms are extracted from the PLET, the PH1/FW5 indicator uses a Kendall trend test to determine the direction of change in plankton abundance time-series (i.e. increasing, decreasing, or stable) and a Plankton Index (PI) to quantify the relative change among ecologically meaningful lifeform pairs.

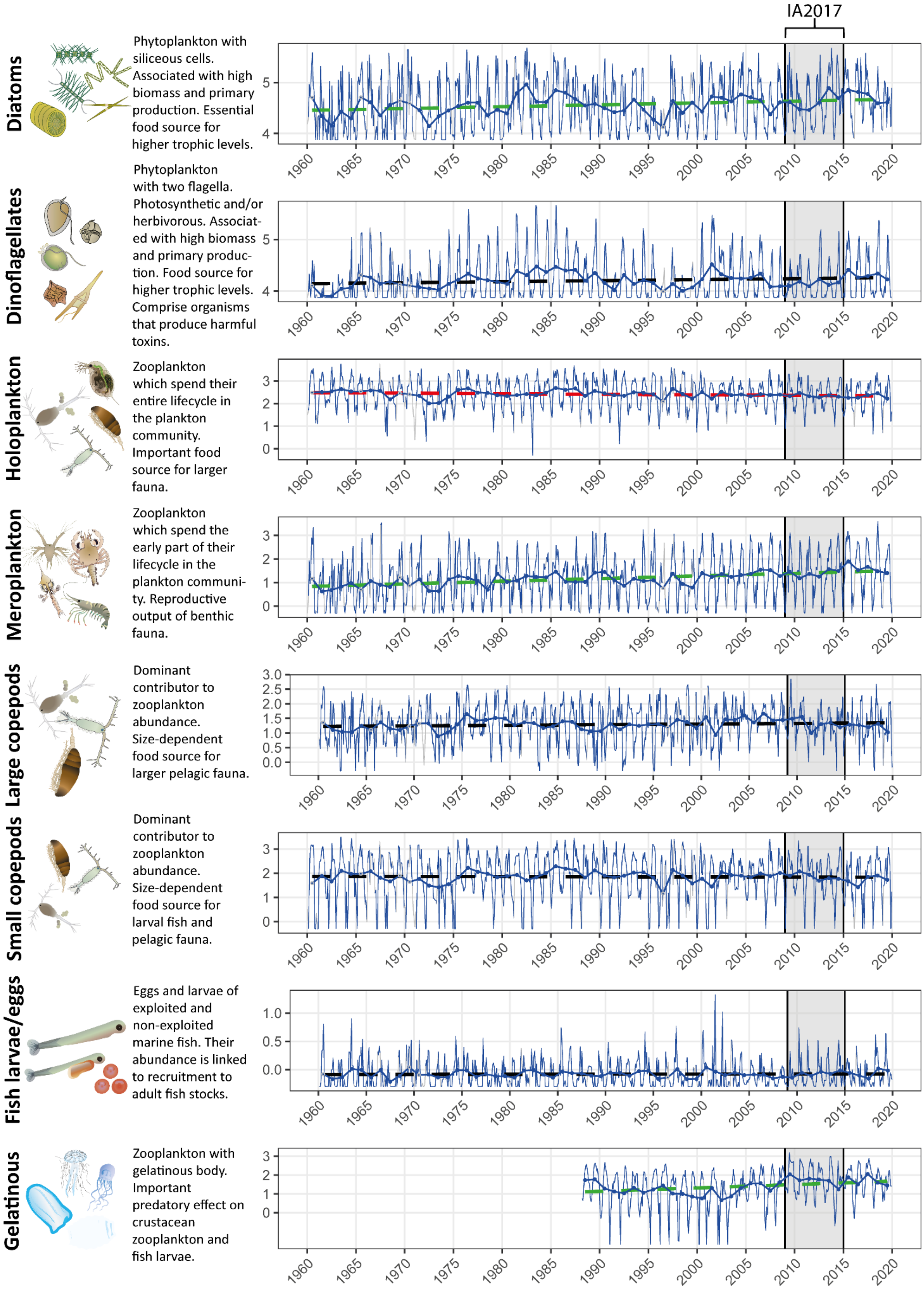


Figure 1.1 Long-term monthly and annual log10 transformed abundance time-series for eight plankton lifeforms in the Western Channel. Blue lines display monthly variability (thinner line), and annual mean abundance (thicker line) values. Dashed lines indicate linear trend lines in annual abundance without any inference on statistical significance. The Kendall trend test is used to infer significance of trends, with red: decreasing trend, green: increasing trend, and black: no trend. Data obtained from the Continuous Plankton Recorder (CPR) survey and Plymouth Marine Laboratory (for gelatinous zooplankton only due to non-quantitative capture of gelatinous taxa by CPR). The shaded region represents the period of IA2017. Plankton images courtesy of the Integration and Application Network, University of Maryland Centre for Environmental Science (ian.umces.edu/symbols/).

The PH1/FW5 indicator was tested with the COMP4 assessment units, a new set of spatial areas aligned with Contracting Parties national reporting regions and with those developed by OSPAR´s Intersessional Correspondence Group on Eutrophication (ICG-Eut). We first extracted the lifeforms based on these updated areas, generated a Kendall trend test and a PI for each lifeform, and then examined the results for consistency and robustness. Spatial modelling (Geographic Information System tools and statistical approaches) was employed to determine the most ecologically relevant scale in the relationships between activities, pressures and affects on pelagic habitat and state. The distribution of trends across a uniform square grid was closely aligned with the distribution of trends across the COMP4 assessment units (Figure 2), providing suitable justification for their use in OSPAR and MSFD assessments of the three Pelagic Habitats indicators. The spatial alignment of patterns in the PH1/FW5 indicator with the delineation of the COMP4 assessment units was further explored. The use of the COMP4 assessment units for Pelagic Habitats indicator assessments has now greatly improved the spatial comparability between PH1/FW5 and foodweb and Eutrophication assessments, as well as supporting the Pelagic Habitats Thematic Assessment.

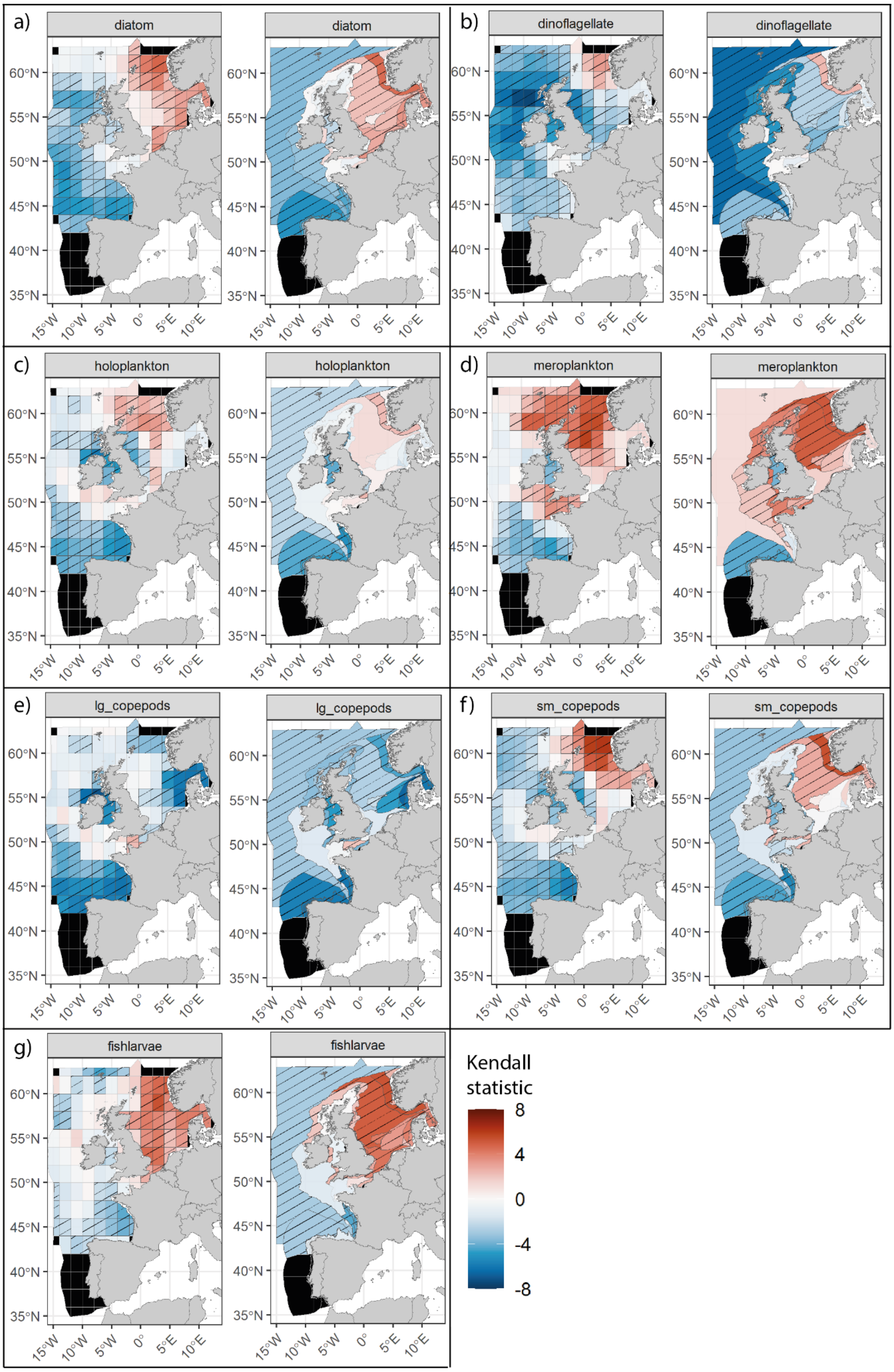


Figure 1.2 The distribution of Kendall trend test results (1960–2018) derived from CPR data and distributed across a 2° square grid (left panel for each lifeform) and across the COMP4 assessment units (right plot for each lifeform) for seven plankton lifeforms which were analysed for the PH1/FW5 indicator assessment, including: a) diatoms, b) dinoflagellates, c) holoplankton, d) meroplankton, e) large copepods, f) small copepods, and g) fish larvae/eggs. Assessment units are coloured according to the results of the Kendall trend test, which indicate the magnitude of long-term increase (> 0) and decrease (< 0) in lifeform abundance from 1960-2018. Patterned assessment units indicate statistically significant change (p ≤ 0,05). Assessment units filled in black indicate insufficient data to evaluate a trend.

PH1/FW5 currently uses monthly time-series of plankton data collected via light microscopy, which must be at least eight years in length (sampling must cover four years of the assessment period and at least four years of the comparison period). To further increase indicator robustness by including as many data as possible, the indicator was tested on different types of data, lengths of datasets, and frequencies of sampling. These data were gathered via an OSPAR data call. In addition, to improve the interpretability of the assessment, the list of lifeforms considered in the assessment was reduced to focus on a core group with easy identification to represent changes in phytoplankton (diatoms and dinoflagellates) and zooplankton (meroplankton, holoplankton, large copepods, small copepods, fish larvae, and gelatinous zooplankton).

OSPAR IA2017 identified changes in pelagic indicators but did not link these with drivers of change. For the current indicator assessment, we built on work of the EMFF-Funded ICEGRAPH project (Increasing Confidence in Evaluating GES for Regional Assessments of Pelagic Habitats) to identify the causes of change PH1/FW5 (Bedford *et al.*, 2020a). Indicator responses to key anthropogenic pressures and climate drivers were quantified using tree-based models (random forest) to determine magnitude and direction of indicator change relative to these drivers. We developed procedures for integrating results from these tree-based models to distinguish responses from anthropogenic pressures from those due to prevailing conditions (including climate change). This analysis was required to assess indicator progress in light of the lack of suitable assessment thresholds and across subregions. We examined these state-pressure links at multiple spatial scales, including the assessment scales used by ICG-Eut (i.e. COMP4 assessment units), as well as MSFD pelagic habitat types (i.e. variable salinity, coastal, shelf, and oceanic / beyond shelf habitats), and OSPAR regions.

The PH1/FW5 indicator assessment produced for the OSPAR QSR 2023 concluded that long-term trends (1960–2019; Figure 1.3) indicate that most plankton lifeforms, including diatoms, dinoflagellates, holoplankton, fish larvae/eggs, and large (adult ≥ 2 mm) and small (adult < 2 mm) copepods are declining in abundance throughout the Northeast Atlantic. Spatial patterns in the direction of change for both small and large copepods closely match those of the diatoms. Conversely, meroplankton demonstrate a pattern of increasing abundance or no change throughout all assessment units and all but one fixed-point station. Patterns apparent from the widely distributed CPR data are reflected by the fixed-point stations from adjacent transitional waters, except in the case of dinoflagellates, which demonstrate an increasing trend in transitional waters of Scotland (MSS, UK) and Germany (NLWKN, Germany).

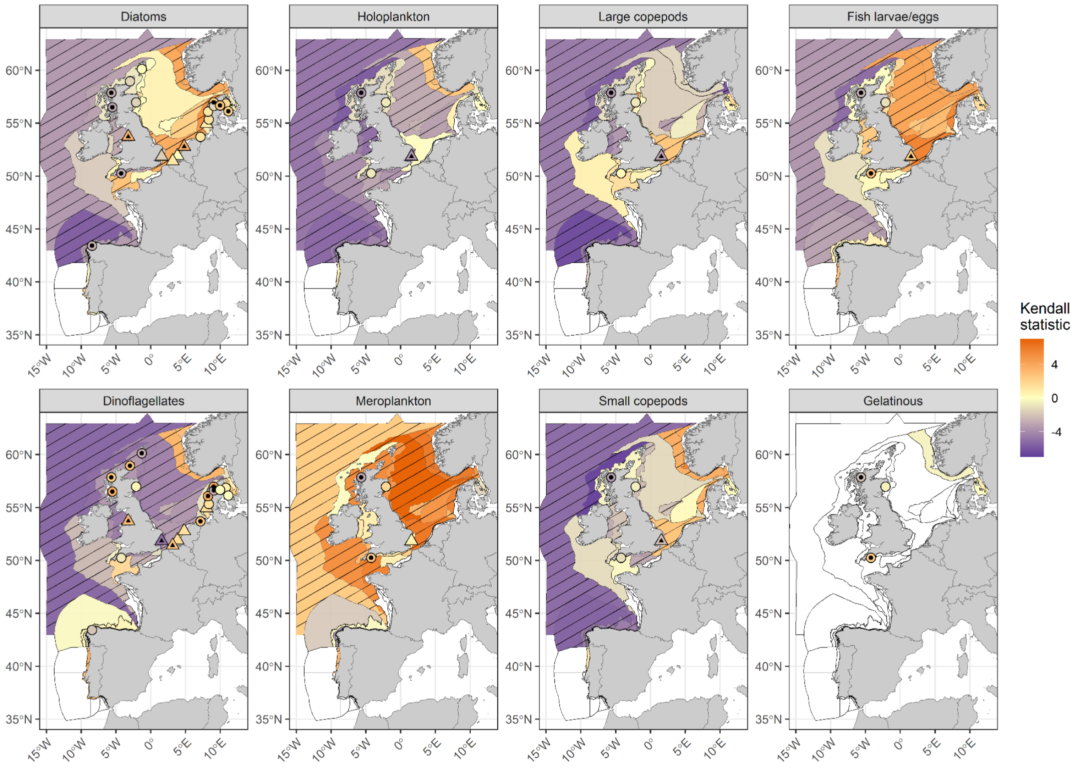


Figure 1.3 Kendall statistics for eight plankton lifeforms displayed over COMP4 assessment units and fixed-point stations in the Greater North Sea, Celtic Seas, Bay of Biscay and Iberian Coast. River plumes are represented as triangles. Assessment units are coloured according to the results of the Kendall trend test, which indicate the magnitude of long-term increase (> 0) and decrease (< 0) in lifeform abundance from 1960–2019 (or a shorter period for fixed-point stations). Patterned assessment units, fixed-point stations and river plumes with an internal black symbol indicate statistically significant change (p ≤ 0,05). Assessment units filled in white indicate insufficient data to evaluate a trend.

Changes in lifeform abundance were linked to variation in environmental pressures, acting both externally (e.g. precipitation, windspeed) and internally (e.g. pH, water temperature) on the marine environment. Many of these pressures are influenced indirectly by climate change (Figure 1.4). Modelling results have shown that increased sea surface temperatures were linked to declining abundances of plankton lifeforms, particularly small and large copepods in the Atlantic. Change in water temperature was also strongly linked to increasing meroplankton abundance within the Eastern North Sea. Impacts of increasing temperature were most frequently detected in shelf and oceanic pelagic habitats. While increasing temperatures can have direct impacts, they can also be linked to greater stratification and resulting nutrient limitation. Links to nutrient concentration were less clear since it is difficult to assess the impact of nutrient concentrations on lifeforms due to the lag period between occurrence and phytoplankton uptake and assimilation. However, model results suggest that they were more apparent in coastal regions. Changes in diatom and dinoflagellate abundance in these assessment units were mainly linked to increases in the N:P ratio, driven by reductions in phosphorus concentrations.

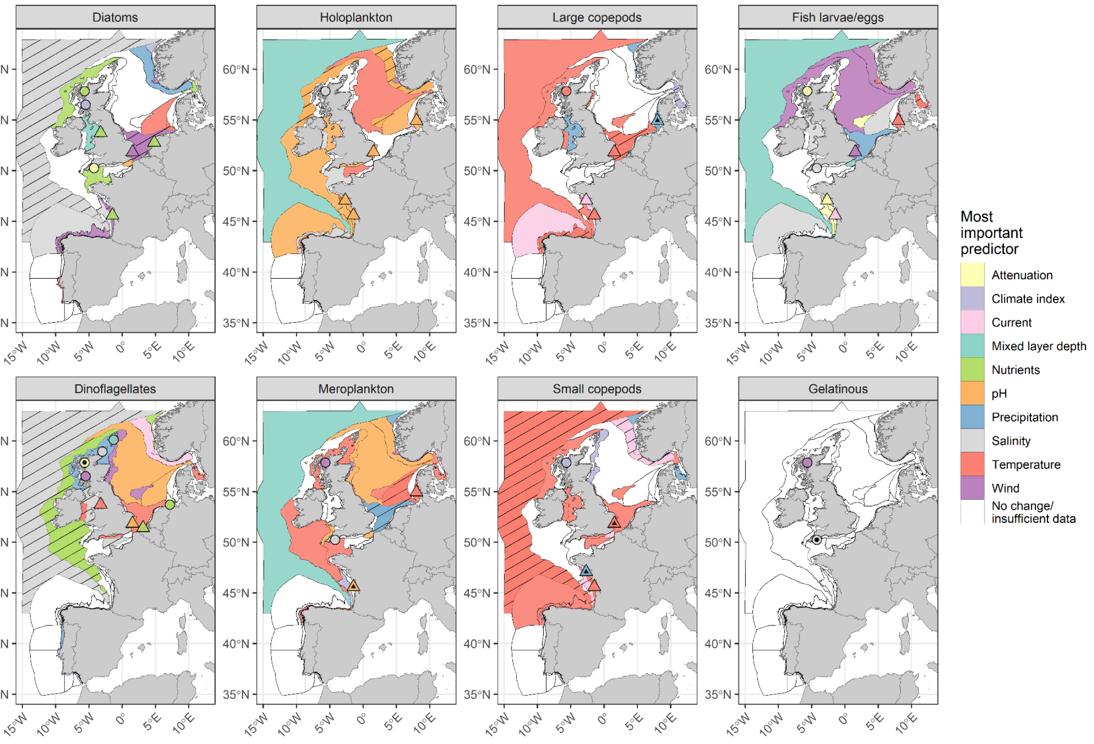


Figure 1.4 COMP4 assessment units for eight plankton lifeforms, coloured by a categorization of the most important environmental variable to predict lifeform abundance. Patterned assessment units, fixed-point stations and river plumes with an internal black symbol indicate statistically significant correlation between predicted and observed testing data, indicating greater reliability of reported results. Assessment units filled in white indicate no change in lifeform abundances or insufficient data to evaluate a trend. Fixed-point stations and river plumes are only displayed where a trend in lifeform abundance is present.

Based on the current criteria used to determine GES (McQuatters-Gollop *et al.*, 2022) and outlined in greater detail under Task 1.4, the relationships between PH1/FW5 indicator results and environmental pressures, the quality status of shelf habitats within the Greater North Sea (Region II), Celtic Seas (Region III), and Bay of Biscay and Iberian Coast (Region IV) was "Not good". Coastal habitats in the Celtic Seas and oceanic / beyond shelf habitats in the Bay of Biscay and Iberian Coast were also “Not good”. All remaining assessed pelagic habitats had an “Unknown" quality status.

1.3 Changes in phytoplankton biomass and zooplankton abundance indicator assessment

The Changes in phytoplankton biomass and zooplankton abundance (PH2) indicator provides a means of identifying changes (anomalies) in the quantities of two fundamental groups within a plankton community, phytoplankton biomass and zooplankton abundance as represented by the abundance of copepods since they are the most numerous zooplankton group. Such changes represent deviations from the assumed natural variability of a plankton time-series. Changes in phytoplankton biomass and zooplankton abundance are measured between a historic comparison period (prior to 2015) and a contemporary assessment period (2015–2019). The direction of change is statistically identified as either increasing, stable, or decreasing. This indicator has been assessed at the subregional scale, using COMP4 assessment units (Enserink *et al.*, 2019) to subdivide data for samples collected within OSPAR Regions II, III and IV.

Anomalies in phytoplankton biomass and zooplankton abundance exhibited decreasing trends across the majority of COMP4 units assessed (Figure 1.5). Change over the assessment period (2015–2019) was marked by strong and significant decreases in phytoplankton biomass in 82% of the assessment units studied. For zooplankton abundance, strong decreases occurred in 59% of the assessment units studied. In agreement with the results for this indicator presented in the IA2017, phytoplankton biomass has continued to increase in the North Sea. Although IA2017 reported a negative trend in zooplankton abundance for the Southern North Sea, the current assessment period (2015–2019) detected no significant change in zooplankton abundance for this area.

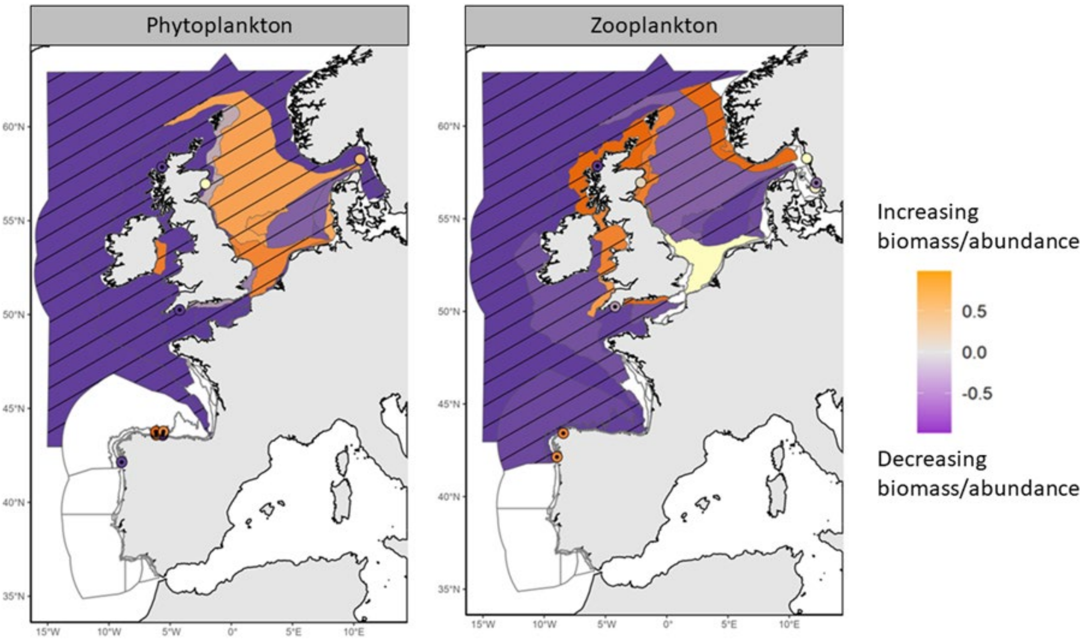


Figure 1.5 Trend in phytoplankton biomass and zooplankton abundance anomalies between the assessment period (2015–2019) and the comparison period (station data: 1992–2014; non-station data: 1997–2014 for phytoplankton and 1960–2014 for zooplankton). Hatched areas were characterized by significant changes ($p \leq 0.05$) in phytoplankton biomass or zooplankton abundance between the comparison and the assessment periods. White areas indicate no data or insufficient data to assess the area.

Environmental pressures shape phytoplankton biomass/zooplankton abundance across temporal and spatial scale (Figure 1.6). Changes in phytoplankton biomass/zooplankton abundance and climate change over the assessment period were evident in the Celtic Seas and the Bay of Biscay and Iberian Coast. Several environmental variables are indirectly linked to climate change, including increasing sea surface temperature (SST), decreasing windspeed, decreasing light attenuation, and increasing mixed layer depth. These pressures were linked to decreases in phytoplankton biomass and zooplankton abundance. Similarly, nutrient imbalance (only directly affecting phytoplankton) and decreases in pH were observed to co-occur with decreases in phytoplankton biomass in the variable salinity and coastal habitats of OSPAR Celtic Seas and OSPAR Greater North Sea region.

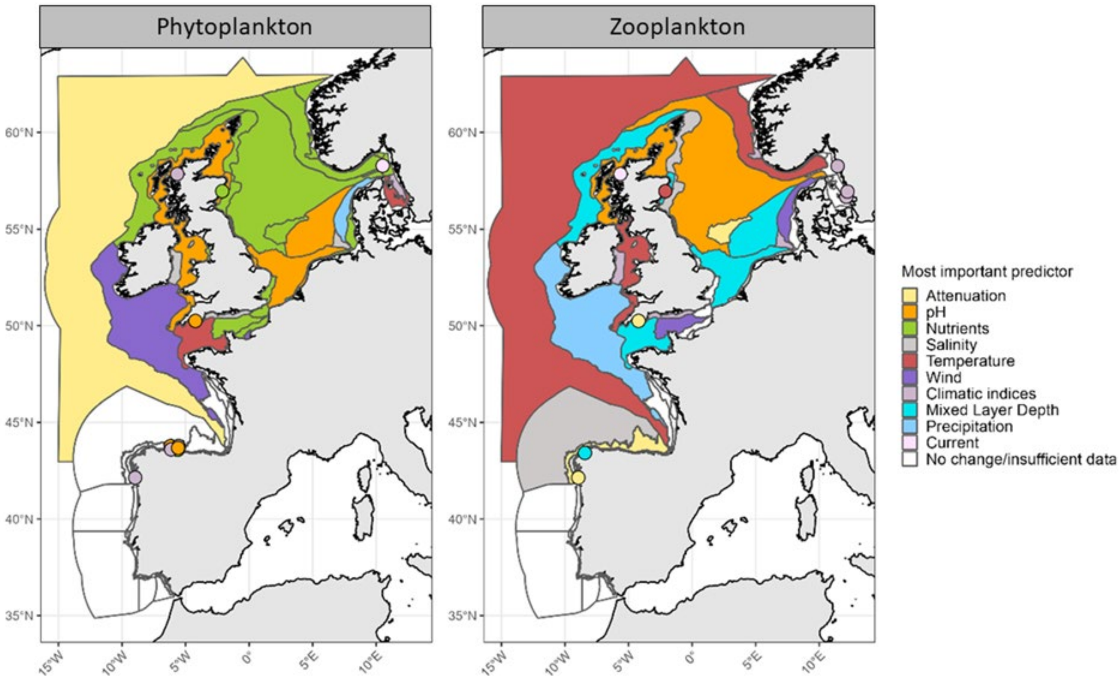


Figure 1.6 PH2 indicator results displaying the most important environmental variable linked to trends in phytoplankton biomass and zooplankton abundance. White areas indicate no data or insufficient data to assess the area.

Based on the criteria used to determine GES (McQuatters-Gollop *et al.*, 2022), the relationships between the indicators and environmental pressures, the quality status of most habitats within the OSPAR Regions was "Not good". Only variable salinity habitats in the Greater North Sea and the Celtic Seas had an "Unknown" quality status.

1.4 Changes in plankton diversity indicator assessment

The changes in plankton diversity (PH3) indicator quantifies changes in plankton diversity through the consecutive use of β- and α-diversity indices (through Local Contribution to Beta Diversity (LCBD) and Menhinick, Hulburt, Gini, and Patten indices respectively). In the PH3 assessment, the β-diversity focuses on the rate of change, or turnover, in species composition in a time-series (Rombouts *et al.*, 2019). Statistical significance is calculated separately for each year. Years with significant β-diversity corresponded to a deviation from the usual community composition. For assessment units with significant years, α-diversity indices were calculated to highlight whether species richness or dominance was responsible for driving the detected changes. The biodiversity indices were computed separately for each dataset, and separately for zooplankton and phytoplankton. In addition, the computation of the indices was considered independently of the taxonomic level. The Menhinick, Hulburt, Gini, and Patten indices were calculated for every month. Only years with more than eight months of sample data were used. This procedure is spatially consistent with the PH1/FW5, PH2 and FW2 indicator assessments since the same assessment units were used for all. Missing data were not interpolated in this assessment, as interpolation at the level of individual taxa can introduce large biases in species abundance. To examine spatial differences in diversity indices, we computed an ecological quality ratio (EQR). This EQR consisted of comparing the β-diversity of each year of the assessment period to the mean β-diversity of the comparison period. Finally, a Kendall trend test was run on the annual EQR of the β-diversity to account for interannual variation in diversity and to remove cyclical seasonal variation. While the annual β-diversity identified the short-term change of plankton diversity, the Kendal trend test allowed us to identify permanent or long-term change of plankton diversity during the assessment period.

Changes in diversity were addressed at the regional scale by assessing long-term changes in CPR data (1960–2019), and at local scale from fixed monitoring stations (1989–2019). To compare community composition across the assessment units, β-diversity was integrated through a yearly Ecological Quality Ratio. While in assessment the PH3 has been adopted as a common indicator in the Celtic Seas, the PH3 indicator remained a pilot assessment in the Greater North Sea and the Bay of Biscay and Iberian Coast.

From 2015 to 2019 on an annual basis, 70% of the assessment units had an atypical phytoplankton community composition compared to the period prior to 2015, while only 7% of the assessment units displayed a phytoplankton community composition similar to that of the comparison period (prior to 2015). 23% of the assessment units fell somewhere in between. However, only one assessment unit had significant atypical composition (Channel Well Mixed Tidal Influenced in 2019; Western English Channel; β-diversity p-value < 0,05; Figure 1.7). Kendall trend test results were not statistically significant, revealing no long-term change in the phytoplankton community composition.

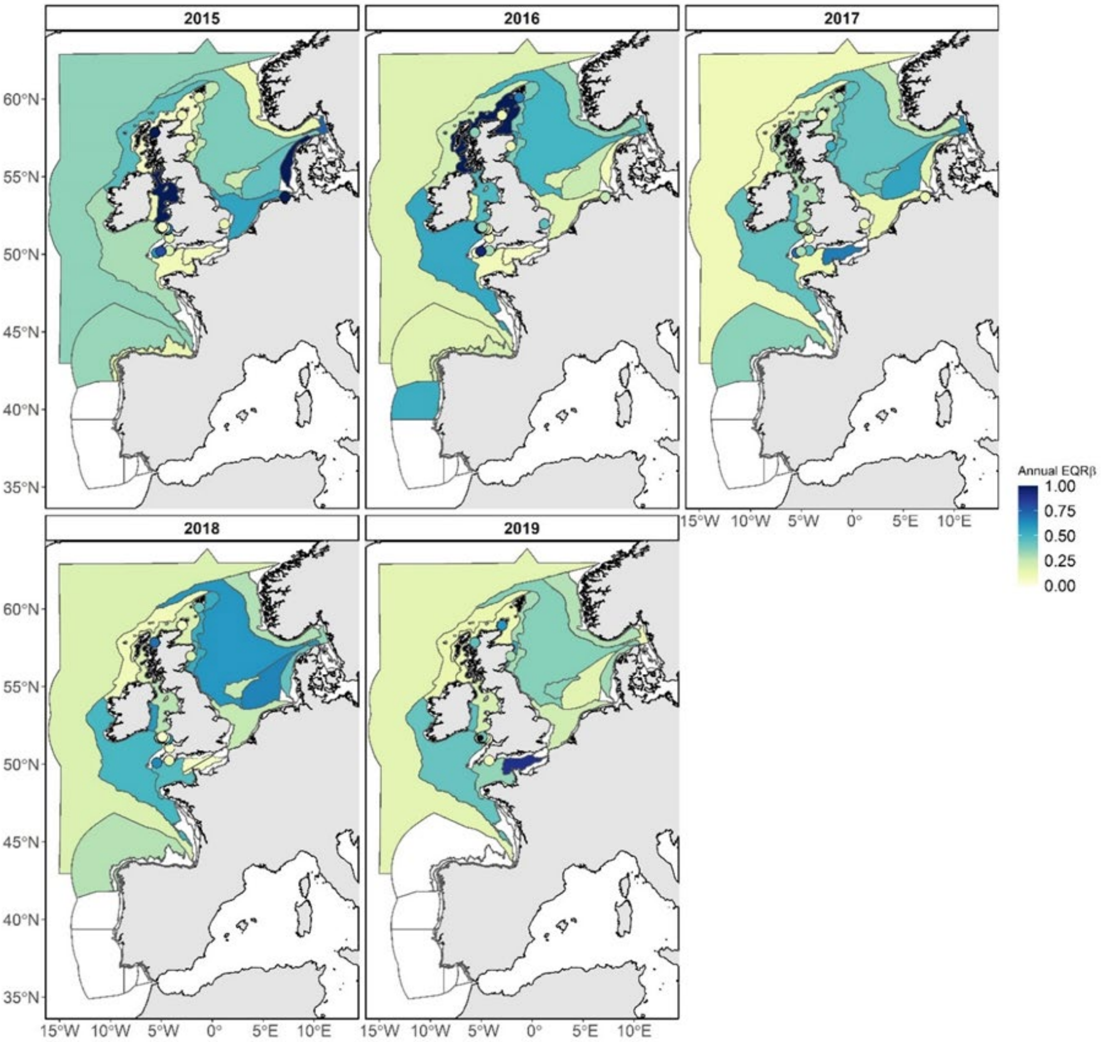


Figure 1.7 Evolution of the annual EQR_β of phytoplankton diversity indices during the assessment period (2015–2019). Low EQR_β indicating a large difference between the comparison value of EQR_β and the annual EQR_β value are displayed in yellow; High EQR_β indicating a slight difference between the comparison value of EQR_β and the annual EQR_β value are displayed in dark blue. White areas indicate no data or insufficient data to assess the area. COMP4 units with significant atypical composition (LCBD p-value < 0,05) are displayed as dashed areas. Monitoring fixed stations with significant atypical composition (LCBD p-value < 0,05) are displayed as dark dots. This is a hybrid figure showing results of the common indicator assessment for the Celtic Seas and for the pilot assessment for the Greater North Sea and the Bay of Biscay and Iberian Coast.

From 2015 to 2019 on an annual basis, 83% of the assessment units had an atypical zooplankton community composition compared to the period before 2015, while only 4% of the assessment units displayed a zooplankton community composition close to the period before 2015. 13% of the assessment units were in between. However, eight sites (assessment units and monitoring stations) had significant atypical composition (Channel Well Mixed: 2015; Norwegian Trench: 2015; East Coast Permanently Mixed 1: 2016; Eastern North Sea: 2016; Northern North Sea: 2016; Southern North Sea: 2016; Southern North Sea and Anholt station in 2019; β-diversity p-value < 0,05; Figure 1.8). Kendall trend test results were not statistically significant, revealing no long-term change in zooplankton community composition.

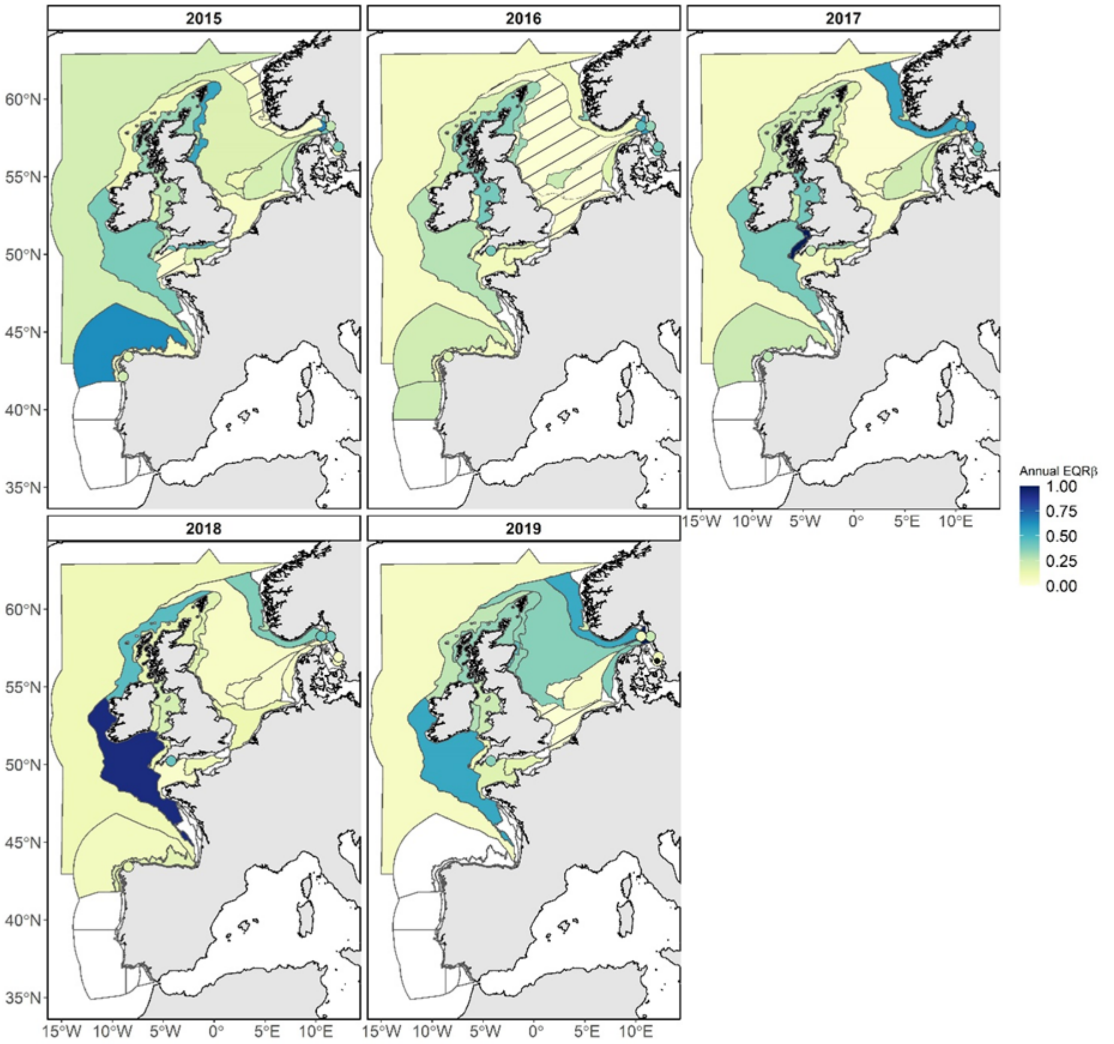


Figure 1.8 Evolution of the annual EQR_β of zooplankton diversity during the assessment period (2015–2019). Low EQR_β indicating a large difference between the comparison value of EQR_β and the annual EQR_β value are displayed in yellow; High EQR_β indicating slight difference between the comparison value of EQR_β and the annual EQR_β value are displayed in dark blue. White areas indicate no data or insufficient data to assess the area. COMP4 units with significant atypical composition (LCBD p-value < 0,05) are displayed as dashed areas. Monitoring fixed stations with significant atypical composition (LCBD p-value < 0,05) are displayed as dark dots. This is a hybrid figure showing results of the common indicator assessment for the Celtic Seas and for the pilot assessment for the Greater North Sea and the Bay of Biscay and Iberian Coast.

Environmental variables were selected according to their relevance to determine the most important pressure in plankton diversity. The set of environmental variables used originated from different models targeting the Northeast Atlantic area. The link between PH3 and pressures was conducted using the β-diversity results, as previous studies have demonstrated the ability to link environmental parameters to the LCBD (Vilmi *et al.*, 2017). The EQR was used to maintain consistency and harmonization among the COMP4 assessment units. Variations in plankton diversity were closely related to decreasing light attenuation and imbalance between nutrients for phytoplankton and primarily by natural climatic indices (e.g. NAO, AMO) for zooplankton (Figure 1.9).

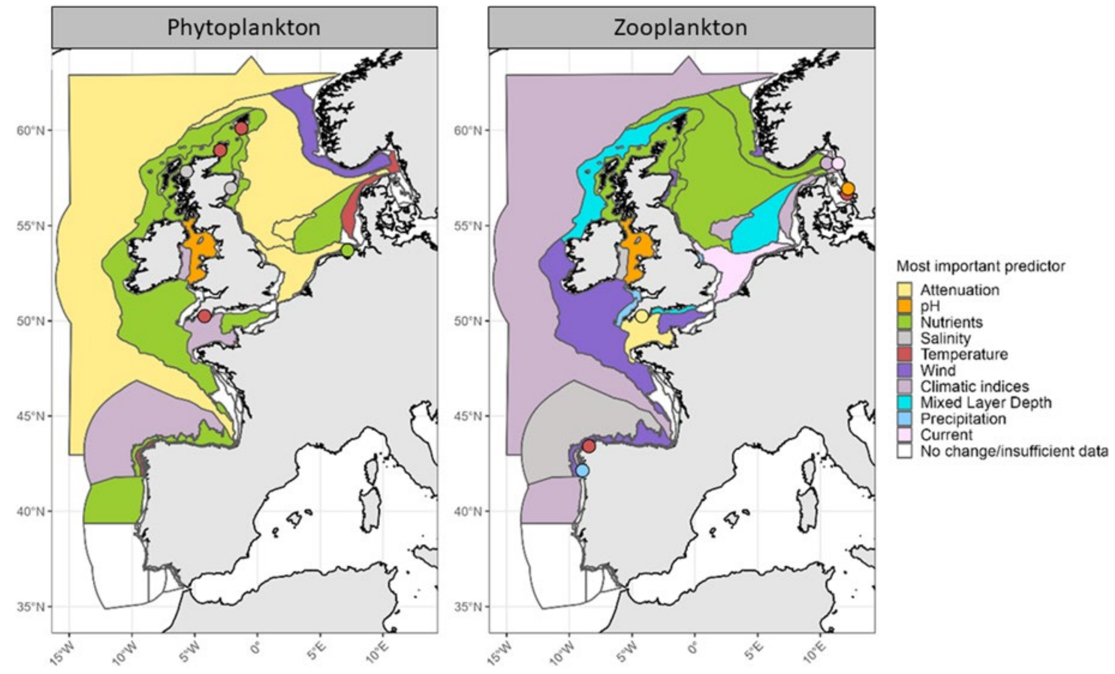


Figure 1.9 PH3 indicator results displaying the most important environmental variable linked to trends in phytoplankton and zooplankton diversity. This is a hybrid figure showing results of the common indicator assessment for the Celtic Seas and for the pilot assessment for the Greater North Sea and the Bay of Biscay and Iberian Coast.

Based on the current criteria used to determine GES (McQuatters-Gollop *et al.*, 2022), the relationships between PH3 indicator results and environmental pressures, the quality status of coastal habitats within the OSPAR Regions was "Not good". Variable salinity habitats in the Greater North Sea and shelf habitat in the Celtic Seas were also “Not good” (pilot assessment of the PH3 indicator for these two regions). Shelf habitats in the Greater North Sea and the Celtic Seas and Oceanic habitats of the Bay of Biscay and Iberian Coast had an “Unknown" quality status.

1.5 CEMP Guideline for Pelagic Habitats Thematic Assessment Integration Method

While it can be useful for more technical audiences to interpret indicator results at the level of the four distinct pelagic habitat types that make up each OSPAR region, additional integration may be necessary if a single regional determination of GES is desired. Integration of indicator results is also an important requirement of the MSFD. To integrate results of multiple Pelagic Habitats indicators, we determined that the simplest approach was to extend the rules currently used for integrating results within the PH1/FW5 and PH2 indicators, by applying majority rules to GES results at the indicator level. We selected the One Out-All Out approach to automatically downgrade GES results to be more negative in the case of a tie.

An intermediate step is required to transition from multiple indicators results for each habitat type to a single regional determination of GES for the region; however, there were two logical approaches for this. The first approach would be to integrate the results of the common pelagic habitats indicators for each habitat type to determine an overall quality status result for each habitat (Figure 1.10a). The second approach would be to integrate the indicator results across habitat types to determine an overall quality status result for each indicator (Figure 1.10b). We concluded that integrating across indicators (i.e. PH1/FW5, PH2, and PH3 where it is accepted as a common indicator) to generate a single GES determination for each MSFD pelagic habitat type. This approach provides granular information on how anthropogenically linked changes vary with geography. Results generated using this approach can inform whether changes are mainly occurring close to the coast, and therefore possibly linked to direct anthropogenic pressures such as eutrophication, or whether they are occurring further offshore and more likely associated with broad scale processes like climate change. For the Pelagic Habitats Thematic Assessment (Deliverable D1.4b) we integrated across the three Pelagic Habitats indicators and developed a Coordinated Environmental Monitoring Programme (CEMP) guideline document to describe the integration methodology, which was approved by the OSPAR Biodiversity Committee (BDC) during their meeting in Berlin, Germany (12–15 December 2022).

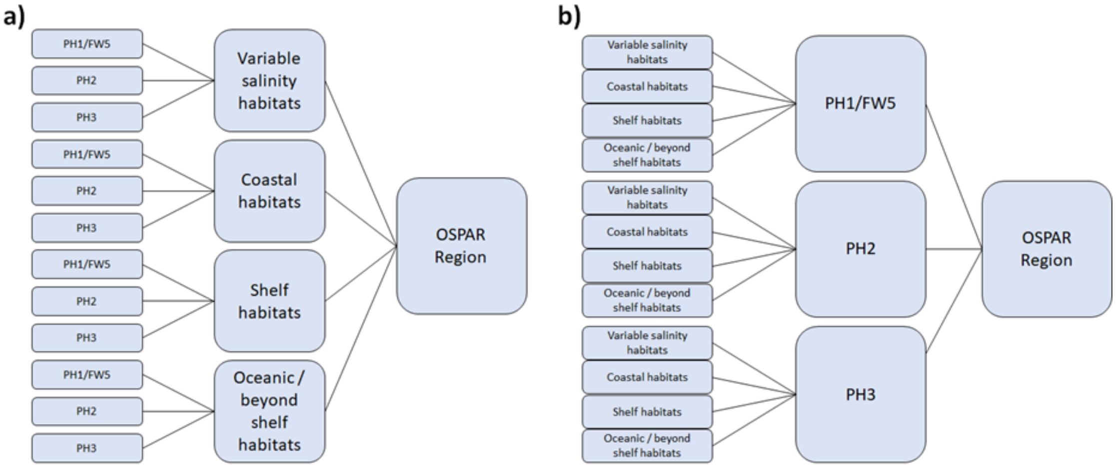


Figure 1.10 Conceptual flowchart diagram displaying two ways indicator results can be integrated to determine GES. Results can be integrated across the three indicators separately for each of the four pelagic habitat types within an OSPAR region (a), or they can be integrated across the four pelagic habitat types separately for each of the three pelagic habitat indicators within an OSPAR region (b).

Finally, regarding the setting of thresholds to determine GES, although pelagic habitats can in some cases be assessed for GES by evaluating and establishing threshold values based on primary productivity and chlorophyll-a (Heyden and Leujak, 2023; Tilstone *et al.*, 2023), there is currently no scientific consensus on what represents GES when it comes to the abundance of lifeforms, copepods, or biodiversity. Further, determination of GES is further complicated by the fact that plankton data collection began after the Northeast Atlantic was already heavily impacted by anthropogenic activities, thus there are no suitable data available to represent pristine conditions that would not be already adversely impacted by human activities.

The current OSPAR methodology used to assess GES for pelagic habitats (and developed through the NEA PANACEA Project; Task 1.2) avoids the use of threshold values. Rather than testing whether a particular threshold value has been attained, the methodology for pelagic habitats indicators developed for the QSR 2023 evaluates three criteria to establish whether there is a suitable burden of evidence, including:

1. A sufficient level of spatial and temporal confidence among assessed time-series,
2. A sufficient level of spatial representation to assess each habitat type, and
3. The most important pressure being one that is linked to anthropogenic activity.

This methodology also evaluates whether the available evidence shows a suitable level of internal agreement to support determination of GES, including:

4. Most assessment units showing the same direction of change, and
5. A sufficient mean rank for the most important pressure linked to changes in lifeform abundance).

It could be argued that the minimum levels applied to evidence and agreement criteria are themselves thresholds, however, these values only assess the burden of evidence of whether an important change may have occurred, rather than an indicator value above or below which GES is not achieved.

1.6 Pelagic habitats thematic assessment

The Thematic Assessment of Pelagic Habitats for the OSPAR QSR 2023 was created to determine the current state of Pelagic Habitats across the Northeast Atlantic, the influence of humans on this component of the marine environment, and to determine if there are any management measures available within OSPAR to mitigate negative changes driven by human-linked pressures.

The document itself was written in consultation with members of the COBAM and UK Pelagic Habitats Expert Groups and underwent several periods of peer review and revision over the course of 2022. NEA PANACEA grant recipients were responsible for writing the State and Climate Change chapters, as well as curating all other sections written by other OSPAR expert groups and acting as primary editors of the overall document. The State chapter of the assessment was written as a summary of integrated indicator results at the level of MSFD pelagic habitats (i.e. variable salinity, coastal, shelf, and oceanic / beyond shelf habitats) and more generally at the level of the assessed OSPAR regions (Regions II, III, and IV; Figure 1.11). For all five OSPAR Regions (including unassessed Regions I and V) current literature on pelagic habitats was reviewed to create a regional summary narrative. The structure of the contribution followed the DAPSIR approach (Drivers, Activities, Pressures, State, Impacts, Response), which related changes in natural ecosystems with anthropogenic influences that drive them.

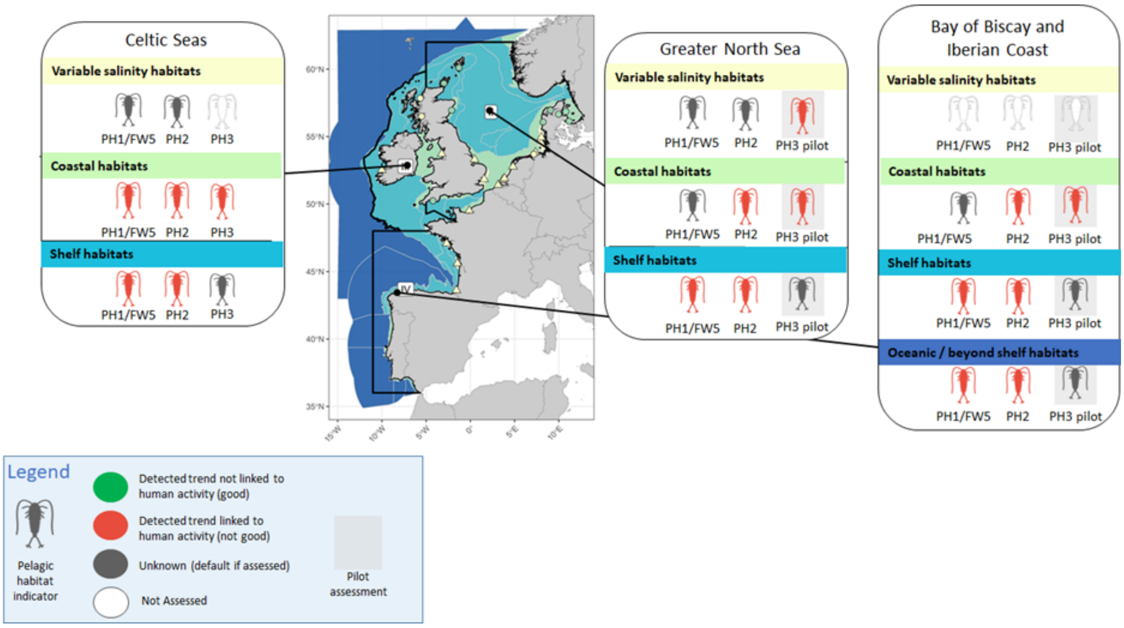


Figure 1.11 Indicator results schematic for pelagic habitat types (variable salinity, coastal, shelf, and oceanic / beyond shelf habitats) within the OSPAR Regions assessed for the Pelagic Habitats indicator assessments. This graphic has been designed following the format used and methodology described in McQuatters-Gollop et al. (2022). For the OSPAR Pelagic Habitats biodiversity indicators there was variability among indicators, pelagic habitat types, and OSPAR Regions. Icons have been coloured according to indicator status. Current integration methods for pelagic habitats do not allow for the determination of "Good" Environmental Status. Some indicators were not assessed in some Regions (grey) due to lack of data. Pelagic habitats without results displayed indicate that the particular habitat type is not present within the OSPAR region. Grey background and "pilot" labelling display that an indicator has candidate status in the respective region and a pilot assessment has been prepared.

For this contribution, indicator results were summarized following methodology described in McQuatters-Gollop *et al.* (2022). Results were summarized in the form of graphical maps (Figure 1.11) which spatially link indicator results to geographic locations, and tables (Table 1.1), which clearly display the integration steps from indicator results at the level of pelagic habitat types, to a determination of GES for each pelagic habitat type, to determination of GES for each assessed OSPAR region.

Table 1.1 The status for each pelagic habitat type within each OSPAR region, derived from integrating the status of common indicators for pelagic habitats. Uncoloured and diagonally hatched cells indicate that an indicator has candidate status in a particular region and a pilot assessment has been produced. As the PH3 indicator remains a candidate indicator for OSPAR Regions II and IV, the status of PH3 for these regions is given for information purposes only and was not considered in the integration of overall habitat or region status.

Region	Habitat	PH1/FW5	PH2	PH3	Habitat status	Region status
Greater North Sea (Region II)	Variable salinity	Unknown	Unknown	Not good	Unknown	Not good
	Coastal	Unknown	Not good	Not good	Not good	
	Shelf	Not good	Not good	Unknown	Not good	
	Oceanic	Not assessed	Not assessed	Not assessed	Not assessed	
Celtic Seas (Region III)	Variable salinity	Unknown	Unknown	Not assessed	Unknown	Not good
	Coastal	Not good	Not good	Not good	Not good	
	Shelf	Not good	Not good	Unknown	Not good	
	Oceanic	Not assessed	Not assessed	Not assessed	Not assessed	
Bay of Biscay and Iberian Coast (Region IV)	Variable salinity	Not assessed	Not assessed	Not assessed	Not assessed	Not good
	Coastal	Unknown	Not good	Not good	Not good	
	Shelf	Not good	Not good	Unknown	Not good	
	Oceanic	Not good	Not good	Unknown	Not good	

A confidence scoring methodology, based on an approach developed by ICG-Eut to validate the output from their COMPEAT Tool, was applied to evaluate the robustness of reported trends for each plankton dataset for each assessment unit it intersected. For each assessment unit or fixed-point station temporal confidence was evaluated by assessing the consistency of sampling throughout each time-series, integrating the consistency of sampling throughout each year (seasonal representation) and consistency of sampling across the entire time-series.

Spatial confidence was also evaluated for distributed datasets such as the CPR. For each COMP4 assessment unit spatial confidence was calculated based on the spatial distribution of samples relative to the boundaries of each assessment unit. This confidence scoring approach was incorporated into the assessments for all three Pelagic Habitats indicators (i.e. PH1/FW5, PH2, and PH3) and was further integrated in the Thematic Assessment of Pelagic Habitats, using the same integration rules which were applied to GES results (Table 1.2).

Table 1.2 Confidence assessment of the type, amount, quality, and consistency of evidence (i.e. Robust, Medium, or Limited), as well as the degree of agreement in the results (i.e. High, Medium, or Low) for the three pelagic habitats indicators across the five OSPAR regions. Colours are used for interpretation. Uncoloured and diagonally hatched cells indicate that an indicator has candidate status in the region and while a pilot assessment has been conducted, a confidence assessment has not been produced.

Region	Criteria	PH1/FW5.	PH2	PH3	Criteria status	Region status
Greater North Sea (Region II)	Agreement	Medium	Medium	Not assessed	Medium	Medium
	Evidence	Medium	Medium	Not assessed	Medium	
Celtic Seas (Region III)	Agreement	High	High	High	High	High
	Evidence	Medium	Medium	Medium	Medium	
Bay of Biscay and Iberian Coast (Region IV)	Agreement	High	High	Not assessed	High	Medium
	Evidence	Limited	Limited	Not assessed	Limited	

The general conclusions of the Thematic Assessment of Pelagic Habitats were that the growing global population has generated increasing demand for food production, waste disposal, coastal development, and energy systems, all of which contribute to human-induced climate change. Climate change is probably the greatest pressure currently impacting plankton communities across the OSPAR maritime area as a whole. These activities also influence the supply of nutrients entering coastal environments, which can generate eutrophication and impact the productivity of pelagic habitats.

Pelagic habitats in the OSPAR Maritime Area have experienced widespread changes over the past 60 years, with indicator assessments revealing a general pattern of decreasing phytoplankton and zooplankton abundance and/or biomass across the Greater North Sea, Celtic Seas, and Bay of Biscay and Iberian Coast. Long-term trends have largely continued into the current assessment period, and are expected to continue into the future, eventually impacting higher food-web levels. Due to widespread changes linked to pressures generated by human activities, the Greater North Sea, Celtic Seas, and Bay of Biscay and Iberian Coast had “Not good” status, given the current definition and categorization of quality status.

Global efforts to slow climate change are probably the best mechanism to counter widespread changes in plankton communities, although effective measures for reducing or preventing climate change mostly lie outside the remit of OSPAR. Regionally targeted management measures (e.g. controlling inputs of nutrients and organic matter) in coastal areas may affect pelagic habitats at the shelf and coastal scale. While these mitigation efforts are likely to only generate noticeable impact in coastal areas, they may also have some effect in areas where plankton communities are affected by the cumulative impacts of multiple pressures (i.e. both warming and eutrophication).

2 A summary of contributions from WGBIODIV to the OSPAR Quality Status Report 2023 for fish and foodwebs

2.1 Recovery of sensitive fish species

This species-level indicator addresses the extent of recovery among populations of fish deemed sensitive to additional mortality from fishing. Fish species with life-history traits such as large ultimate body size, slow growth rate, large length and late-age-at-maturity, are particularly sensitive to additional sources of mortality including through bycatch (Greenstreet *et al.*, 2012; Rindorf *et al.*, 2020). Populations of such species are known to have declined markedly in abundance through the 20th century (e.g. Bluemel *et al.*, 2021; Sguotti *et al.*, 2016), a period of marked expansion in fishing activity across the area assessed. Recovery in population status among a significant fraction of surveys that assess each species is therefore needed.

Taking into account all populations of sensitive species, including those without enough data to support assessment (i.e. no data and unknown outcomes), 28% of regional population assessments achieved the recovering threshold and 46% achieved the no further decline threshold (Figure 2.1). For sensitive fish species able to support assessment, this indicator assessment suggests an improving situation with 49% (of 114 populations) achieving the primary threshold of recovering in the long term (based on regionally integrated outcomes). A further 32% of assessable populations had a stable long-term integrated outcome, so strong evidence exists that further decline in species has been halted with 82% of assessable populations achieving the secondary assessment threshold (either recovering or stable integrated outcome), increasing to 89% when examining the recent short-term assessment period.

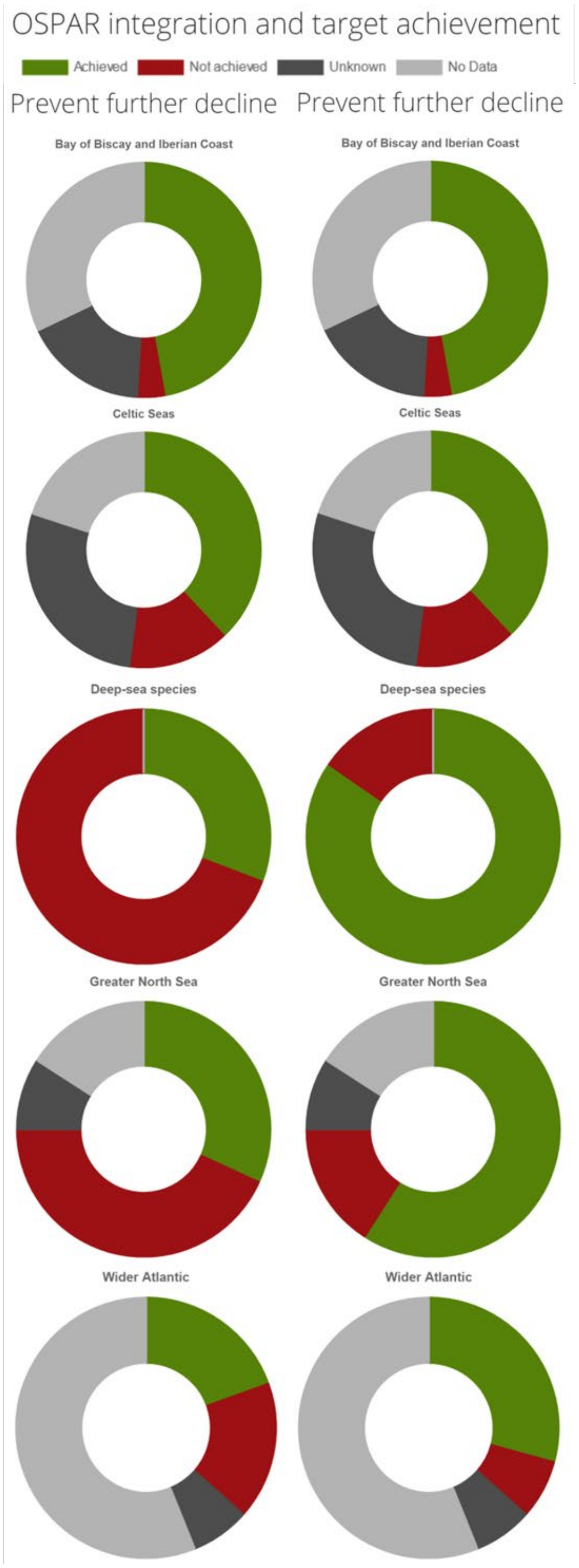


Figure 2.1 Summary of integrated assessment outcomes (achieved or not achieved) for two assessment objectives: (left) species populations are recovering in the long term and (right) species populations are no longer declining (either stable or recovering in the long term), by OSPAR Region. Results for deep-sea species are based on the single deep-sea survey on the Porcupine Bank in the Wider Atlantic only (WASpaOT3). Unknown result = species with insufficient data to assess against thresholds. No data = species present within a Region, but no data are available.

Nevertheless, 18% of regionally integrated outcomes for populations failed to achieve either threshold suggesting long-term population declines. Although the evidence of recent population declines falls to 11% in the short term, one species (*Cyclopterus lumpus*) is declining in both assessment periods and across multiple Regions (Bay of Biscay Iberian Coast and Celtic Sea) and should thus be prioritized for further examination and/or protection. *Molva macrophthalma* also failed both long-term objectives in Region III and IV but has stabilized at low levels in recent years in Region IV. Additional management action may be warranted for the following populations that failed to achieve both long-term assessment objectives, along with evidence of recent population declines in a single region: *Amblyraja radiata* (Region II), *Anarhichas lupus* (Region V), *Brama brama* (Region V), *Conger conger* (Region V), *Molva molva* (Region IV), and *Pollachius pollachius* (Region II) and the deep-sea species, *Mora moro*, assessed for the Porcupine Bank (Region V).

2.2 Size composition in fish communities

The Typical Length indicator is one of multiple foodweb indicators currently used by OSPAR to assess fish communities (Lynam *et al.*, 2022). It represents the average length of fish (demersal bony fish and elasmobranchs) and provides information on the size structure within communities. The indicator is calculated using catch data from species sampled by scientific surveys. Fishing mortality constrains the age structure of fish populations, reducing the proportion of larger individuals. A gradual, steady decline in Typical Length is expected in response to high fishing pressure and fishing at Maximum Sustainable Yield is expected to lead to a recovery in the indicator. This is because the size structure of the fish assemblage integrates the impacts of fishing pressure over long periods of time.

There was no consistent pattern across the whole OSPAR Maritime Area (Figure 2.2). Increases were found in the Irish Sea overall, Bristol Channel, part of Porcupine Bank, a small subdivision to the north of Scotland and the northern Isles (Orkney and Shetland) area within the North Sea, parts of the northern Bay of Biscay and the northern Celtic Sea. Decreases to minimum values were found in the central and southern North Sea and Kattegat and parts of the western edge of the shelf, the Clyde area and to the south of Portugal and part of the northern French coast.

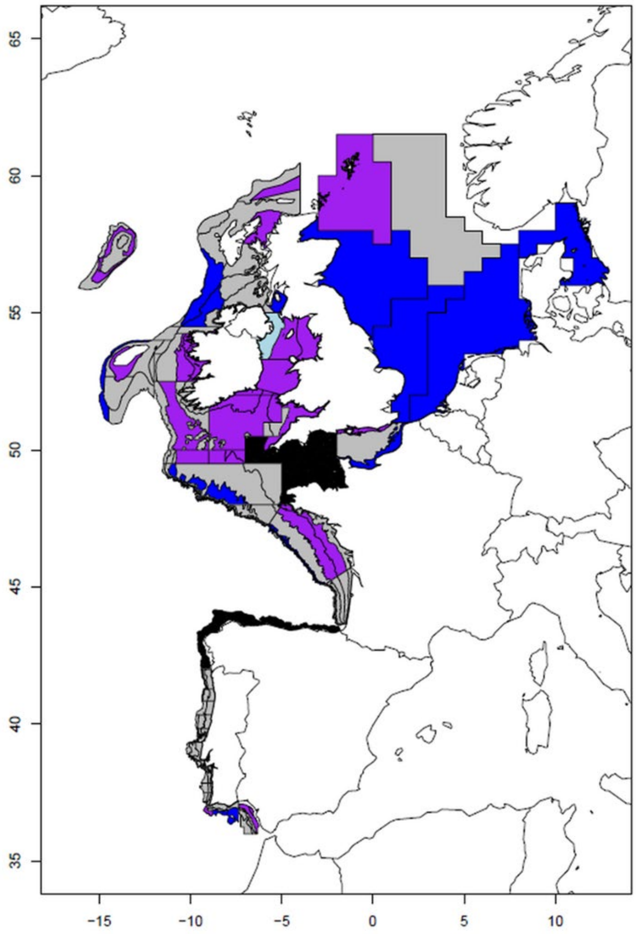


Figure 2.2 Spatial pattern in outcome of Typical Length indicator by subdivision for preferred surveys by Region. Purple colouring means long-term increase evident; dark blue shows decrease to minimum level; light blue shows decrease to low but not minimum level. Grey areas show areas with no long-term change evident and black area show surveys that are too short to detect long-term change. See also the assessment on Proportion of Large Fish for more information on change in the size structure of fish assemblages across OSPAR Regions (Lynam and Piet, 2023).

2.3 Pilot Assessment of Feeding Guilds

Foodweb indicators can reveal how ecosystems are responding to environmental change and anthropogenic pressure in a way that cannot be inferred from studying habitat, species or assemblages alone. Systematic differences in response of typically smaller bodied organisms, feeding lower in the foodweb (planktivores) relative to those with bigger bodies that utilize different resources (benthivores, pisco-crustivores) and feed higher up the foodweb (piscivores), could profoundly alter the uptake of nutrients and the efficiency of communities in converting resources into biomass (i.e. ecosystem functioning which supports the provision of ecosystem services).

Using change in functionally distinct feeding guilds to assess environmental status has been widely advocated to fulfil OSPAR and the Marine Strategy Framework Directive requirements (Boschetti *et al.*, 2021; ICES, 2018; Rombouts *et al.*, 2013; Tam *et al.*, 2017; Walmsley *et al.*, 2016), but international consensus on how to do this has been lacking. Thompson *et al.* (2023; https://oap.ospar.org/en/ospar-assessments/quality-status-reports/qsr-2023/indicator-assessments/feeding-guild-pilot-assessment/) drew together data and expertise from across the North Atlantic and Arctic Oceans to help achieve this. Their study aimed to further develop a feeding guild indicator, as proposed by Garrison and Link (2000) and Thompson *et al.* (2020), which uses stomach contents information to group predators into common functional roles within the foodweb. Stomach contents data were collated with support from EuroMarine. The newly collated dataset contains > 23 000 unique predator–prey interactions from > 1 250 000 fish stomachs from across the North Atlantic and Arctic Oceans. The data are used to demonstrate how feeding guilds (i.e. predator groupings based on their taxonomy, body size and diet) can be defined systematically and in a way that is conducive to their application internationally across ecosystems. These guilds are then applied to otter trawl survey data collected from across the OSPAR Maritime Area to demonstrate observed changes in the relative dominance of key energy pathways that are critical to maintaining ecosystem structure and function. The Pilot Assessment of Feeding Guilds revealed spatially extensive decreases in planktivore biomass lower in the foodweb in the North Sea and Bay of Biscay subdivisions. In contrast, biomass of the pisco-crustivore and piscivore feeding guilds higher in the foodweb increased in the Celtic Seas region and Bay of Biscay subdivision.

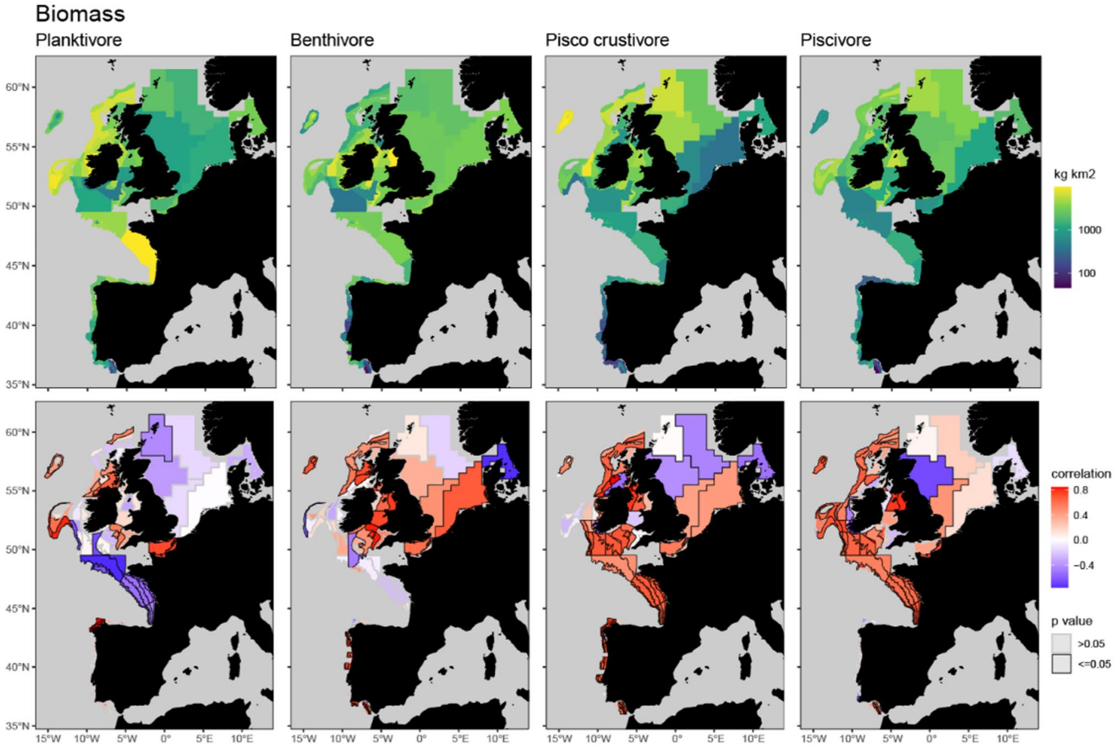


Figure 2.3 Mean spatial distribution (top row) and temporal change (bottom row) in feeding guild species richness by assessment strata based on otter trawl data. Temporal increases are shown by red cells (Pearson's correlation values between 0 and +1) and declines in blue cells (correlation values between 0 and -1). Assessment strata where the temporal change (correlation) is significant are highlighted with black borders.

2.4 Can we make predictions for feeding guilds against a shifting baseline? Climate change affects the distribution of diversity across marine foodwebs

Many studies predict shifts in species distributions and community size composition in response to climate change, yet few have demonstrated how these changes will be distributed across marine foodwebs. Thompson *et al.* (2023) use Bayesian Additive Regression Trees to model how climate change will affect the habitat suitability of marine fish species across a range of body sizes and belonging to different feeding guilds, each with different habitat and feeding requirements in the Northeast Atlantic shelf seas. Contrasting effects of climate change are predicted for feeding guilds (Figure 2.4-Figure 2.5), with spatially extensive increases in species richness largely driven by the expansion in range of smaller species with relatively low predator–prey mass ratios and who tend to be piscivorous by 2095 under RCP 4.5.

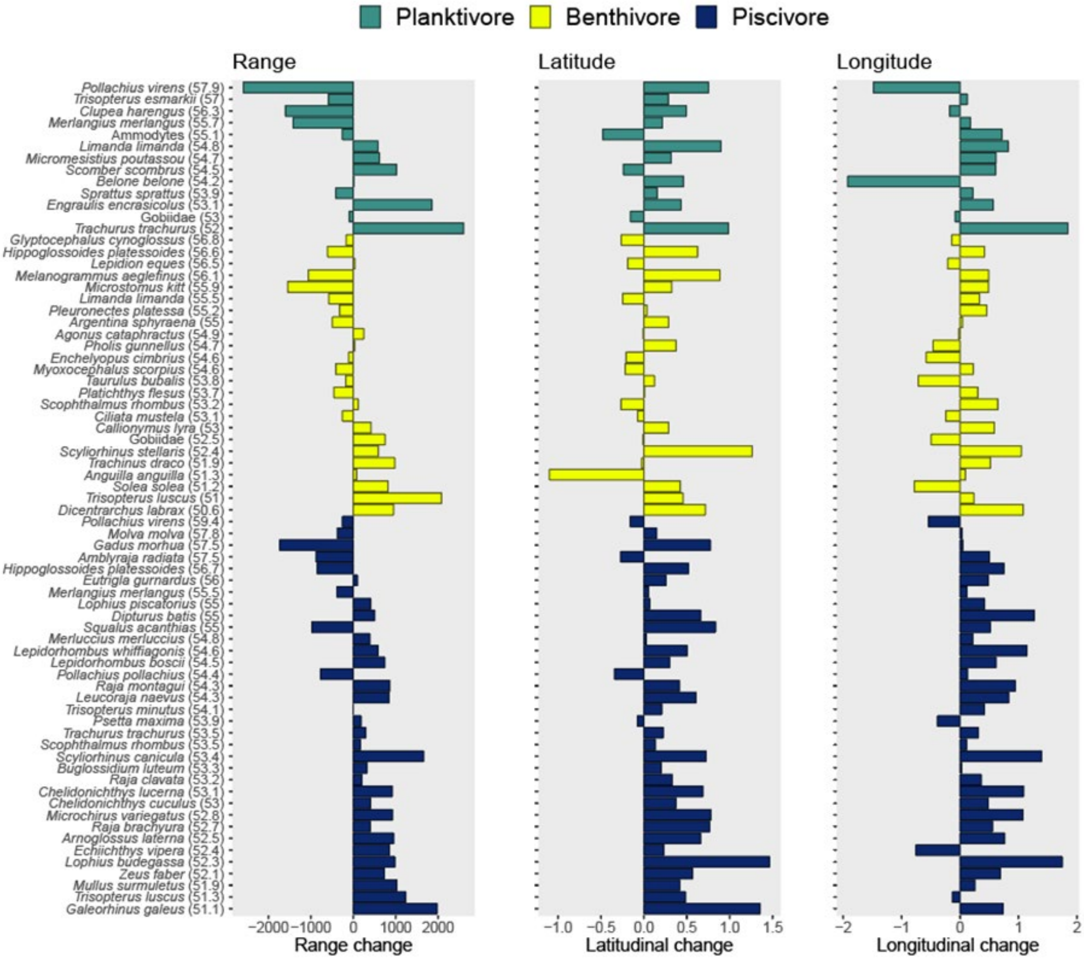


Figure 2.4 Predicted change in the distribution of species assigned to feeding guilds, feeding guild species richness, predator–prey mass ratios (PPMR) and mean maximum length (MML) from 2020 to 2095 based on RCP 4.5. Top row: species are ordered along the y-axis by feeding guild and then their mean latitudinal values (in parentheses). Change in range represents change in the number of cells occupied across the study region, each cell corresponding to an area of 100 km^2. Latitudinal and longitudinal change represent shifts in the mean latitudinal and longitudinal values of cells occupied by species, respectively. Species which appear multiple times on the y-axis switch guilds through ontogeny, such as juvenile planktivorous saithe (*Pollachius virens*) which switch from planktivore to piscivore at larger size classes and can have differing habitat requirements (note contrasting latitudinal changes).

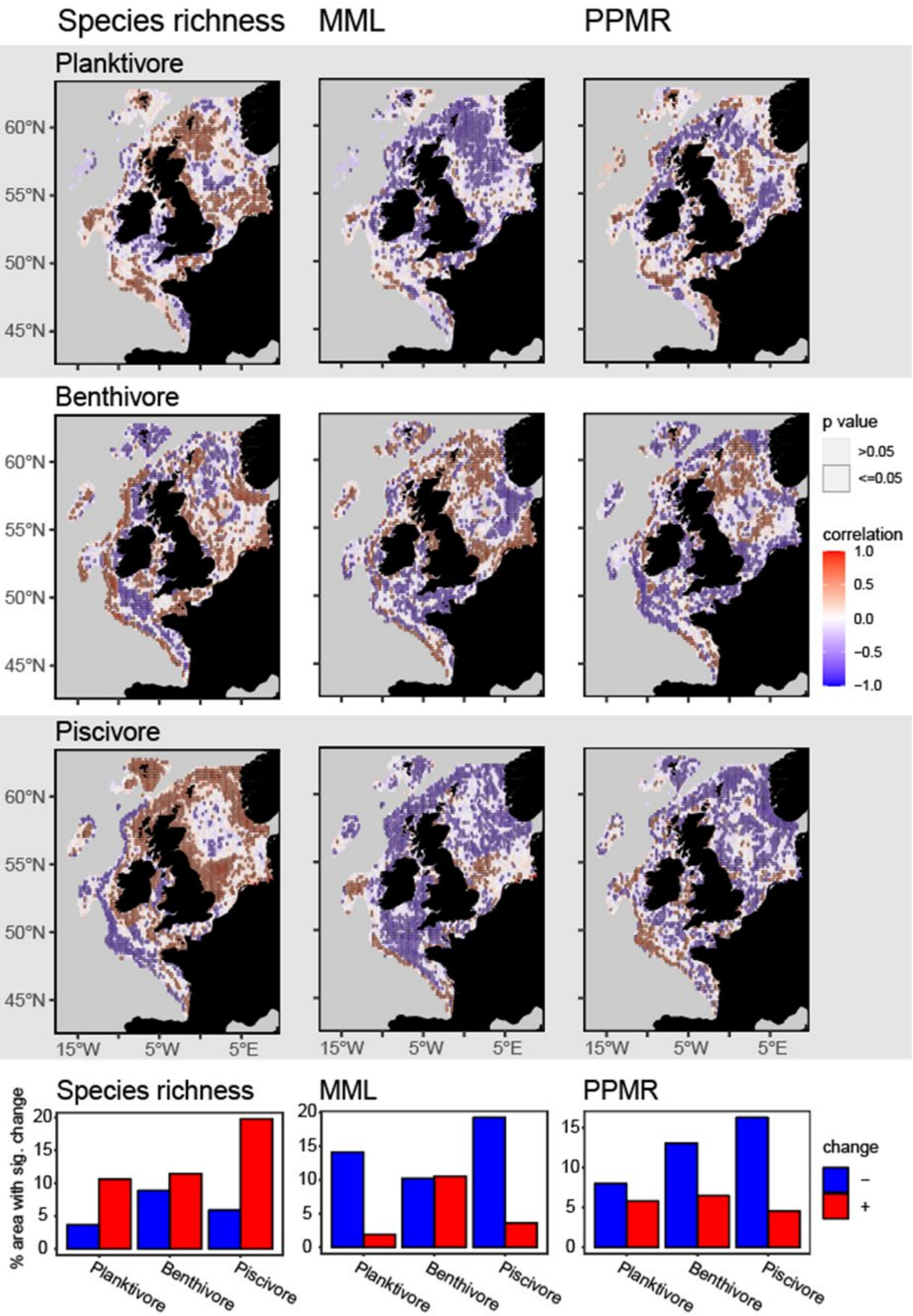


Figure 2.5 Temporal correlations in feeding guild species richness (left column), mean maximum length (MML; middle column), and predator prey mass ratio (PPMR; right column) over five-year intervals from 2020 to 2095 under RCP 8.5. Temporal increases are shown by red cells (Kendall's tau correlation values between 0 and +1), declines by blue cells (correlation values between 0 and -1), and cells with significant correlations have a black border. The bottom row shows the % of cells with a significant increasing (red) or significant decreasing (blue) correlation.

3 Using complementary measures to provide a multidimensional perspective of benthic ecosystem functions

3.1 Introduction

Efforts to manage and conserve Biodiversity and Ecosystem Functioning (BEF) depend on our ability to understand the mechanisms that affect ecosystem stability at a range of spatial scales, including sufficiently large-scales that meet international strategies and directives (e.g. OSPAR, MSFD). Both variability of species diversity and spatial heterogeneity play important roles in maintaining resource availability via so called "Portfolio Effects": "As in financial portfolios, where diversification across assets can stabilize returns (i.e. reduce risk), a wide assortment of diversifying features in biological systems stabilizes their performance" (Shindler *et al.,* 2015). Therefore, understanding BEF requires complementary indices which afford a multidimensional perspective of biodiversity change in taxonomy, functions and diversity in their alpha, beta and gamma forms (Wang and Loreau, 2014; Thompson *et al.*, 2020). In the marine environment, the benthos is a charismatic component that has raised tremendous research interests as part of BEF. Currently, knowledge of benthic BEF such as spatial distribution and vulnerability substantially dominate knowledge of the role of species in ecosystem functioning, especially regarding functions other than trophic interactions. To fill this gap, ToR A provides a theoretical framework to identify benthic ecosystem functions and predictions of ecosystem structure and function in space relevant to management and service provision.

3.2 Benthic ecosystem functions

The seabed hosts the majority of marine species, occupies the largest single ecosystem on Earth by area, and has been extensively disturbed following centuries of fishing, among many other anthropogenic pressures (Snelgrove 1999, Snelgrove *et al.,* 2014). However, most biological conservation studies have focused on species vulnerability (e.g. lifespan; Rijnsdorp *et al.,* 2018), without distinguishing between the functions species ensure in the marine ecosystem (e.g. bioturbation; Kristensen *et al.,* 2012). It is therefore unclear whether protecting vulnerable species conserves seabed functions, since species survival and ecosystem function are not necessarily expressed by the same traits (Díaz and Cabido, 2001; Lavorel and Garnier, 2002). Among the numerous studies using a large variety of traits, there are still ambiguities regarding the type of traits to use for specific research objectives. Here, we provide an analysis of a set of traits mostly linked to ecosystem functions.

In the first place, let's define the concept of ecosystem function: there is a large consensus on "fluxes of materials and energy" (Díaz and Cabido, 2001). Figure 3.1 provides a summary of ecosystem functions ensured by the seabed. Other than the passive effects of abiotic components, two main types of functions ensured by the biota can be distinguished. The first type, "non-engineering functions", includes species interactions and biomass dispersal. The second one, "ecosystem engineering", can be more straightforwardly defined: positive or negative effects of an engineering species on the fitness of co-occurring species indirectly by alterations of the physical environment (Jones *et al.,* 1997). While non-engineering functions are difficult to handle with traits since it involves species network relations that can strongly change from one area to another, engineering functions in the marine benthos are restricted to bioturbation (sediment mixing and bioirrigation; Kristensen *et al.*, 2012) and habitat creation (Thrush and Dayton, 2002).

In this contribution, we put an emphasis on the second type of traits, effect traits that express ecosystem engineering as ecosystem functions. We documented fifteen traits expressing these functions for 812 species of macro- and megabenthos found in European waters representing 11 invertebrate phyla and found from the Arctic to the Mediterranean (2473 references). The traits take into account sediment mixing types (biodiffusive, advective and regenerative sediment particle movements), bioerosion, pumping/ventilation into the sediment matrix, burrow type, width and depth, and type and size of biogenically created structures; details and data are provided in Beauchard *et al.* (2023b).

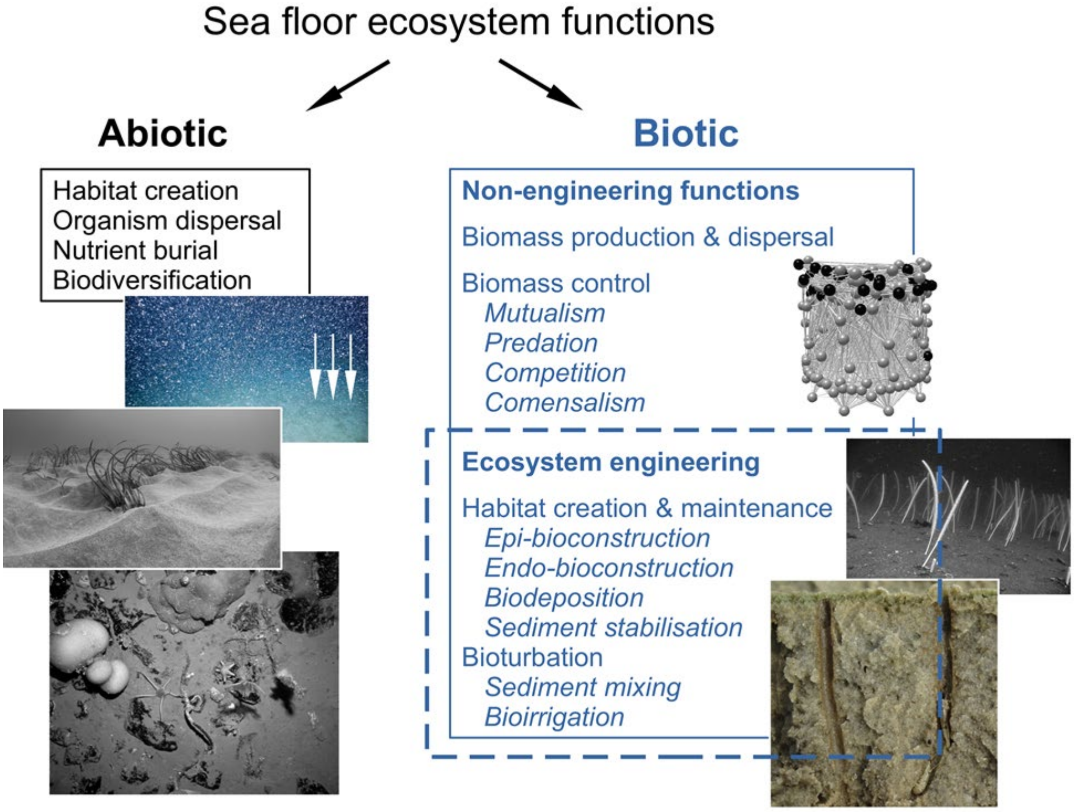


Figure 3.1 Ecosystem functions ensured by the seabed. Left, passive processes induced by physical conditions such as hydrology (e.g. dispersal), geomorphology (e.g. habitat creation), sedimentation (e.g. nutrient burial in abyssal plains) or biodiversification through mixed substrata. Right, biotic functions of which many are network-dependent and not considered in this contribution. Dashed frame: functions considered.

Using a specific multivariate ordination technique (MFA; Escofier and Pages, 1983), we derived a typology of 15 species functional groups (Figure 3.2). MFA is designed to handle datasets with multiple blocks of variables, such as different traits, by analysing the structure within and between blocks of variables. Prior to ordination, each block of variables is weighted so that small and large blocks equally contribute to the total variance. Then, the procedure generates a series of orthogonal axes, each representing a combination of traits that maximize the variance explained across the dataset. Here, the first four MFA axes capture the main structural variations in the data, separating functional groups according to engineering position (epi/endobenthic) and engineering types (shallow/deep, simple/complex biogenic structures). Table 3.1 provides a breakdown of the traits contributing the most to the axes and highlights the key traits that drive differentiation among the functional groups. Detailed group characteristics are provided in Table 3.2.

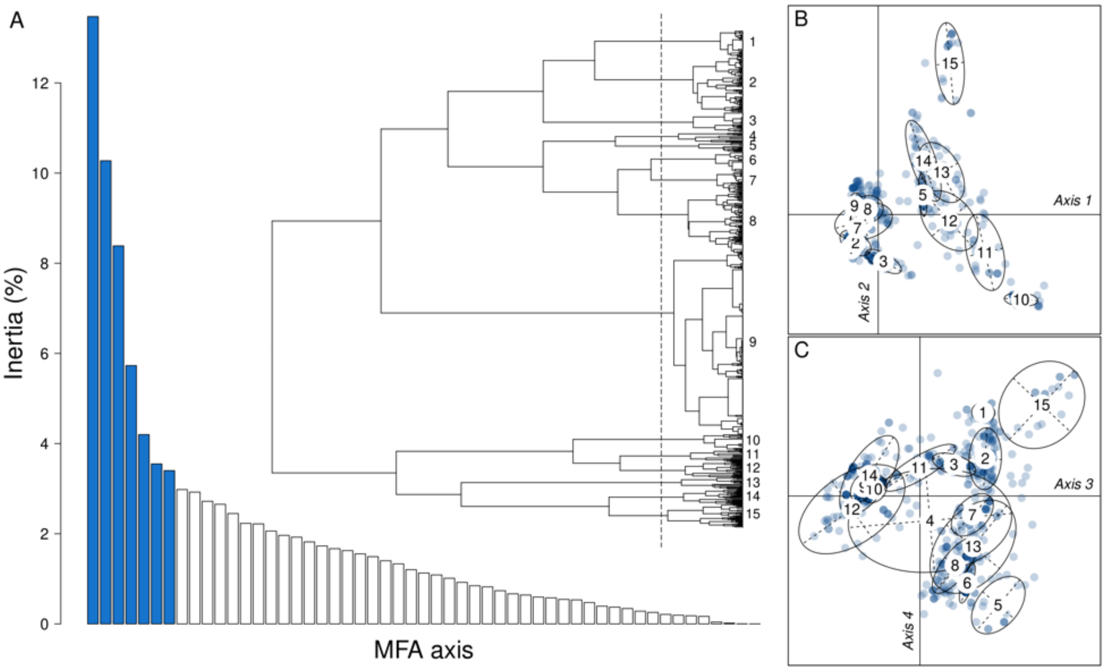


Figure 3.2 Multiple Factor Analysis (MFA) applied to the species × traits matrix. A) Eigenvalue diagram: each bar represents the multivariate variance of each axis; 7 axes (blue) can be considered for species clustering and interpretation. B) Axes 1-2; species (blue dots) functional groups; for details, see Table 3; axis 1 separates epibenthic species (left side) from burrowing ones (right); axis 2 indicates burrowing depth from 0-5 cm (bottom side) to > 30 cm (top). C) Axes 3-4; axis 3 expresses biodeposition from left to right and tubular protrusions (left) to other epibenthic bioconstructions (right); axis 4 opposed simple bioconstructions (shell, bottom) to more complex ones (corals, top). Axes 5, 6 and 7 are not represented as they brought only limited variation regarding a few species groups.

Table 3.1 Trait contributions to MFA axis variance (correlation ratios varying between 0 and 1) as complementary to Figure 18A-B interpretation.

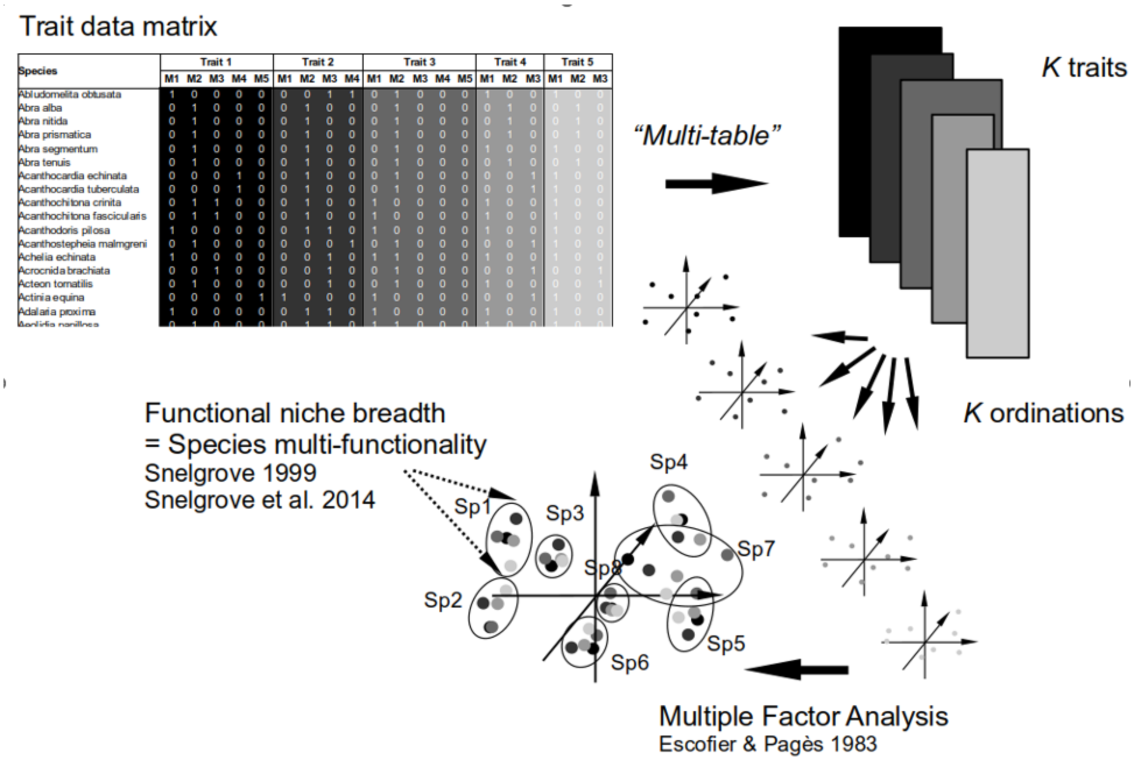
Trait	Axis 1	Axis 2	Axis 3	Axis 4
Substratum depth distribution	0.38	0.29	0.05	0.12
Biodiffusion	0.00	0.32	0.16	0.09
Downward conveying	0.52	0.13	0.04	0.00
Upward conveying	0.23	0.38	0.10	0.07
Regeneration	0.01	0.22	0.01	0.06
Bioerosion	0.07	0.05	0.08	0.19
Biodeposition	0.08	0.18	0.46	0.02
Biostabilization	0.42	0.19	0.01	0.01
Ventilation/Pumping	0.67	0.34	0.09	0.02
Endo-bioconstruction type	0.84	0.33	0.02	0.08
Endo-bioconstruction depth	0.81	0.45	0.07	0.09
Endo-bioconstruction width	0.81	0.35	0.17	0.00
Epi-bioconstruction type	0.54	0.62	0.86	0.70
Epi-bioconstruction extension	0.39	0.37	0.84	0.53
Epi-bioconstruction size	0.16	0.33	0.81	0.57
Inertia (%)	13	10	8	6

Table 3.2 Detailed descriptions of functional groups identified in Figure 18A-B.

Group	Description
1	Epibenthic species. Provide biogenic structures that offer habitats.
2	
3	
4	Fouling species. Create complex mats that provide habitats for small faunal and autotrophic organisms.
5	Boring and abrasive species. Typical of hard substrata. Some of them contribute to habitat complexity by boring. Rare in soft sediments.
6	
7	
8	Shallow-buried shelly species. Mainly suspension-feeders contribution to biodeposition. Provide shells as sediment stabilizers and habitats to fouling organisms.
9	Surficial sediment diffusive species. Epibenthic and mobile species, surficial sediment diffusors.
10	Tubicolous species. Stabilize sediments, contribute to biodeposition, and tube mats as habitats for many other benthic species.
11	
12	Bioturbative species (sediment mixing and bioirrigation), minor (12) or major (14) effect; 13, sessile, substantial sediment irrigation.
13	
14	
15	Deep burrower, bulldozing effect on the sediment. Powerful bioirrigators.

The analysis shows that ecosystem functions can exhibit a pattern fundamentally different from life-history traits ("response traits") where trade-offs of energetic allocation limit the diversification of life-history strategies. Rather than three or four functional groups as generally encountered in life strategies (Southwood, 1988; Kindsvater *et al.*, 2016), we highlighted 15 different groups consisting of species encompassing quite contrasting seabed functions (at least 6–9 major extremities appear; Figure 18A). Despite its rather fundamental nature, this aspect may have important implications from an applied perspective. While the limited number of life-history strategies in the marine benthos such as A, r and K-strategists is convenient to identify disturbance-resistant, resilient and vulnerable, respectively (Beauchard *et al.*, 2022), the greater multiplicity of functional types based on effect traits may not easily fit the former trilogy (see ToR b, vulnerability analysis).

Next to species clustering into functional groups, MFA was used to compute species functional niche breadth (Figure 3.3). Each of the 15 traits is a 812 species × m modalities matrix, and the 15 matrices enable the representation of each species through 15 positions within the same system of multivariate axes. Then, the functional niche breadth consists of the variance of the 15 axis scores (sum of variances for several axes). As a result, the more dissimilar the traits, the broader the niche and ultimately the more multi-functional the species.



Trait data matrix

Species	Trait 1					Trait 2				Trait 3					Trait 4			Trait 5		
	M1	M2	M3	M4	M5	M1	M2	M3	M4	M1	M2	M3	M4	M5	M1	M2	M3	M1	M2	M3
Abludomelita obtusata	1	0	0	0	0	0	0	1	1	0	1	0	0	0	1	0	0	1	0	0
Abra alba	0	1	0	0	0	0	1	0	0	0	1	0	0	0	0	1	0	0	1	0
Abra nitida	0	1	0	0	0	0	1	0	0	0	1	0	0	0	0	1	0	0	1	0
Abra prismatica	0	1	0	0	0	0	1	0	0	0	1	0	0	0	0	1	0	0	1	0
Abra segmentum	0	1	0	0	0	0	1	0	0	0	1	0	0	0	1	0	0	1	0	0
Abra tenuis	0	1	0	0	0	0	1	0	0	0	1	0	0	0	0	1	0	0	1	0
Acanthocardia echinata	0	0	0	1	0	0	1	0	0	0	1	0	0	0	0	0	1	1	0	0
Acanthocardia tuberculata	0	0	0	1	0	0	1	0	0	0	1	0	0	0	0	0	1	1	0	0
Acanthochitona crinita	0	1	1	0	0	0	1	0	0	1	0	0	0	0	1	0	0	1	0	0
Acanthochitona fascicularis	0	1	1	0	0	0	1	0	0	1	0	0	0	0	1	0	0	1	0	0
Acanthodoris pilosa	1	0	0	0	0	0	1	1	0	1	0	0	0	0	1	0	0	1	0	0
Acanthostepheia malmgreni	0	1	0	0	0	0	0	0	1	0	1	0	0	0	0	0	1	1	0	0
Achelia echinata	1	0	0	0	0	0	0	1	0	1	1	0	0	0	1	0	0	1	0	0
Acrocnida brachiata	0	0	1	0	0	0	0	1	0	0	1	0	0	0	0	0	1	0	0	1
Acteon tornatilis	0	1	0	0	0	0	0	1	0	0	1	0	0	0	1	0	0	0	0	1
Actinia equina	0	0	0	0	1	1	0	0	0	1	0	0	0	0	0	0	1	1	0	0
Adalaria proxima	1	0	0	0	0	0	1	1	0	1	0	0	0	0	1	0	0	1	0	0

Figure 3.3 Species functional niche breadth computation. For each species, each trait returns a profile of modality scores for which more than one modality can be attributed a score > 0. Therefore, each trait is a separate matrix processed by a separate multivariate ordination (here a Fuzzy Correspondence Analysis; FCA), and the K separate ordinations, weighted by the inverse of their first respective eigenvalue, are simultaneously coordinated in the same system of the MFA axes. Finally, each species has K scores along each MFA axis. Niche breadth simply consists in the variance of these scores for each species, and the K variances being summed across the K axes.

The comparative analysis of niche breadth between functional groups indicates that burrowing groups are generally more multi-functional. Although both epi- and endobenthic groups can remarkably engineer their environment, engineering from epibenthic groups was systematically limited to the substratum surface, while endobenthic groups can engineer both sediment matrix and surface. Indeed, prominent structures such as faecal pellets, mounds or tube protrusions are commonly encountered on soft sediments with important biogeochemical roles (Ziebis *et al.*, 1996; Murray *et al.*, 2002; Wild *et al.*, 2005).

3.3 Predictions of ecosystem structure and function in space

In order to predict the distributions of seabed ecosystem functions in space, we defined 24 theoretical habitats as combinations of water depth (intertidal, 0–20 m, 20–200 m and > 200 m) and substratum (hard, mixed hard-soft, gravel, sand, muddy sand and sandy mud-mud) categories, two major descriptors of the marine benthic environment (Snelgrove and Buttman, 1994; Etter and Mullineaux, 2001). Then, through Double Principal Coordinate Analysis (DPCoA, Pavoine *et al.*, 2004), we combined the species × habitats matrix with the species × species distance matrix derived from the 7 MFA axes (Figure 3.1); DPCoA axes maximize between-habitat variance (i.e. β-diversity) while averaging species functional distances per habitat (i.e. α-diversity) in accordance with Rao's quadratic entropy (Rao, 1982). This ordination resulted in a main axis, with α-functional diversity increasing from hard to soft sediments (Figure 3.4). Functional groups 10, 11, 13 and 15 (see Table 3.2), contributed the most to the trend as they are the most multi-functional whereby, they engineer both sediment matrix and surface. Finally, and partly noticeable, functions seem to be more vulnerable in deep habitats that are less exposed to natural disturbance, which induces longer lifespan in deep-sea communities (Montero-Serra *et al.*, 2018), while vulnerability in shallower habitats appears undetermined (Figure 20C).

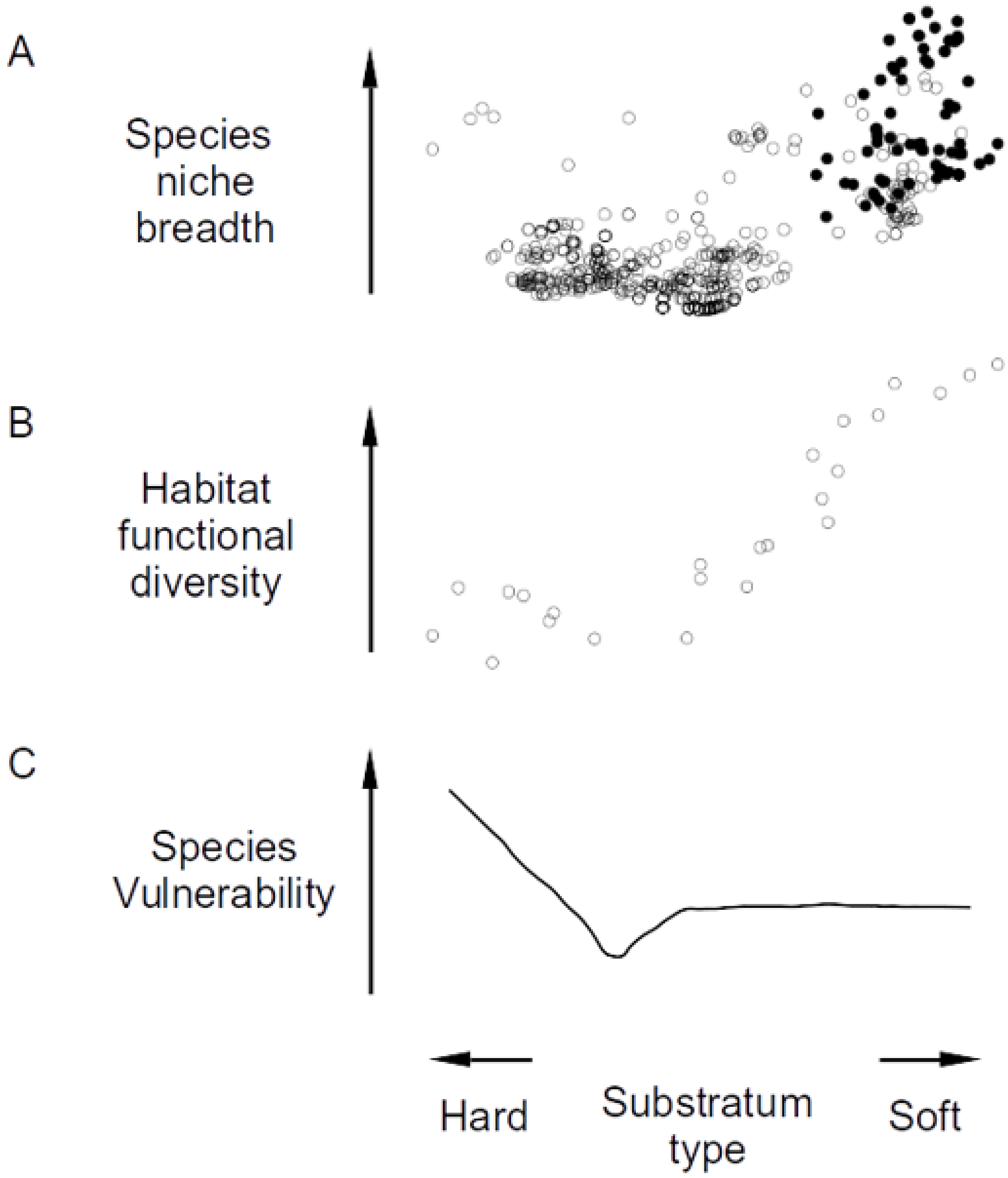


Figure 3.4 Trends in community functional aspects along the main environmental gradient (substratum type) of the European macro- and megabenthic fauna highlighted by the Double Principal Coordinate Analysis (DPCoA). A) Species niche breadth calculated as explained in Figure 19 (species as dots); black dots, species from groups 10, 11, 13 and 15, most contributing to the trend. B) Community functional diversity (Rao's quadratic entropy index) computed for each of the 24 theoretical habitats (4 depth × 6 substratum categories), substantially higher in soft sediments. C) Species vulnerability according to Beauchard et al. (2021, 2023b), indicating higher vulnerability in hard substrata, but limited to the deepest part (continental slope, far left side of the axis), especially due to tall coral forms from vulnerable marine ecosystems (VME; ICES 2023a), weakly variable everywhere else.

3.4 Summary

These analyses were based on trait data compiled from the literature and enabled to combined macro- and megabenthos in a same analytical space. The scientific value of the derive patterns remains more predictive than effective in the absence of empirical field data. The theoretical predictions can be summarized as follows.

- Based on the existing literature, it is possible to document a part of the benthic European fauna that represents a substantial fraction of the main seabed ecosystem engineering that contribute to ecosystem functions, and directly or indirectly to ecosystem services.
- The multivariate pattern in effect trait composition is more complex that the one derived from life-history traits, which suggests much more possible combinations of functional groups and greater β-functional diversity (i.e. complementary ecosystem functions at the regional scale).
- Among the identified functional groups, species multi-functionality is higher in groups specific of soft sediments, especially tubicolous and deep burrowing species; such species are clearly those that enhance seabed functional diversity.
- Along shallow environmental gradients (< 200 m), it is difficult to isolate habitats that are more functionally vulnerable. Consequently, bottom trawling can systematically impact benthic ecosystem functions irrespectively of the habitat.

4 Thresholds responses of marine biota and ecosystem structure: identify areas where marine biodiversity is particularly at risk (ToR B)

4.1 Identifying environments where diversity of seabed ecosystem functions is particularly at risk

4.1.1 Introduction

In the quest for methods to assess ecosystem health, marine benthic ecology has experienced a surge in ecological indicator development through the use of multiple biological traits (Beauchard *et al.,* 2017; Miatta *et al.*, 2021). Until now, most studies have focused on species vulnerability in the context of ecosystem degradation solely based on organism loss. For instance, it is now well accepted that lifespan is a proxy to growth rate and to some degree reproductive success (Kindsvater *et al.,* 2016). This trait can be used to express population recoverability following disturbance, as long-lived species can be more vulnerable than short-lived ones due to their slower population growth rates (Auster and Langton, 1999; Hiddink *et al.*, 2017; Rijnsdorp *et al.,* 2018; ICES, 2019). Biological traits that determine how species respond to disturbance and adapt to the environment in which they occur are often referred to as “response traits”. However, following disturbance, population reduction or species extirpation is even more critical if these species provide crucial ecosystem functions, something that response traits do not directly express. Rather, “effect traits”, that depict effects of organisms on the ecosystem, do have a direct expression on ecosystem functions and thus are more appropriate to relate disturbance to ecosystem functioning (see ToR a). The consideration for this response-effect trait dichotomy, raised some time ago (Lavorel and Garnier, 2002), was recently bolstered in marine benthic ecology (Beauchard *et al.,* 2023; Beauchard, 2023; Figure 4.1).

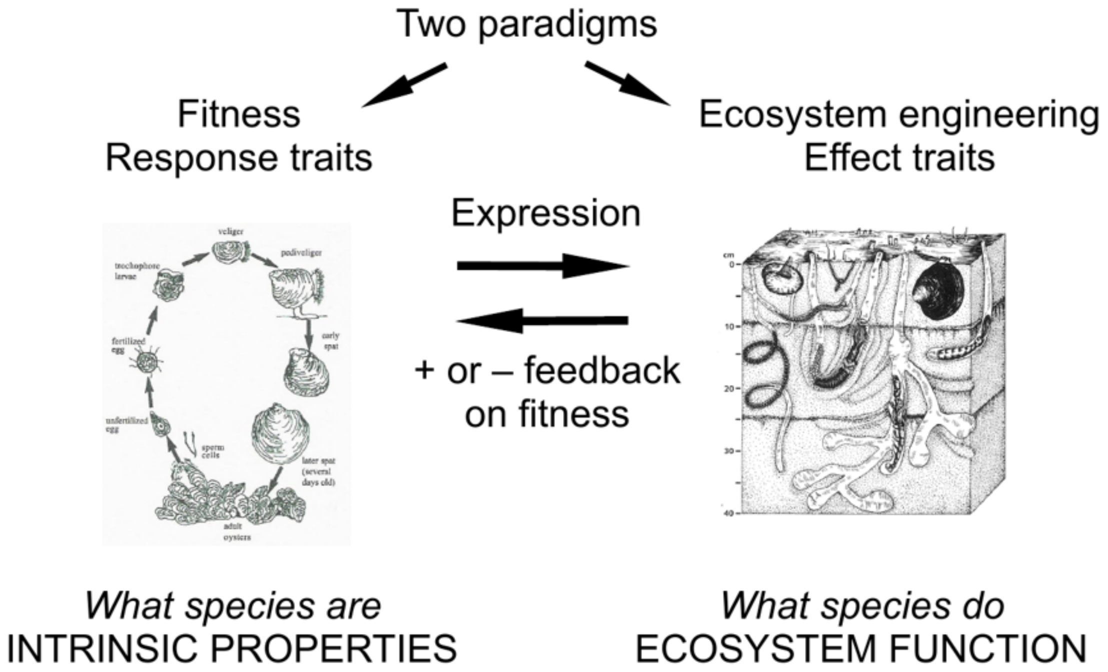


Figure 4.1 Response and effect trait dichotomy for theoretically consistent trait data applications. Left, response traits, expressing the three fitness components (growth, survival and reproduction), specific to characterize species vulnerability to natural and human-mediated disturbance. Right, effect trait characterizing ecosystem engineering as fitness expression; arrow, feedback of the engineering species on the fitness of co-occurring species, indiscriminately positive or negative. Figure from ICES (2024).

Bottom-trawl fisheries have long been regarded as the major threat to benthic ecosystem functions since there is no equivalent human physical disturbance on the seabed for spatial and temporal extents (Jennings and Kaiser, 1998; Thrush and Dayton, 2002; Crowder *et al.*, 2008). To explore the effects of bottom trawling on seabed ecosystem functions, the effect trait dataset analysed in ToR a can be combined with benthic survey data and corresponding trawling intensity estimations. Ultimately, functional diversity derived from effect traits is a meaningful concept from a management perspective since the greater it is the more ecosystem services can be directly or indirectly expected (Beauchard, 2023), at least “regulating services” when considering ecosystem engineering.

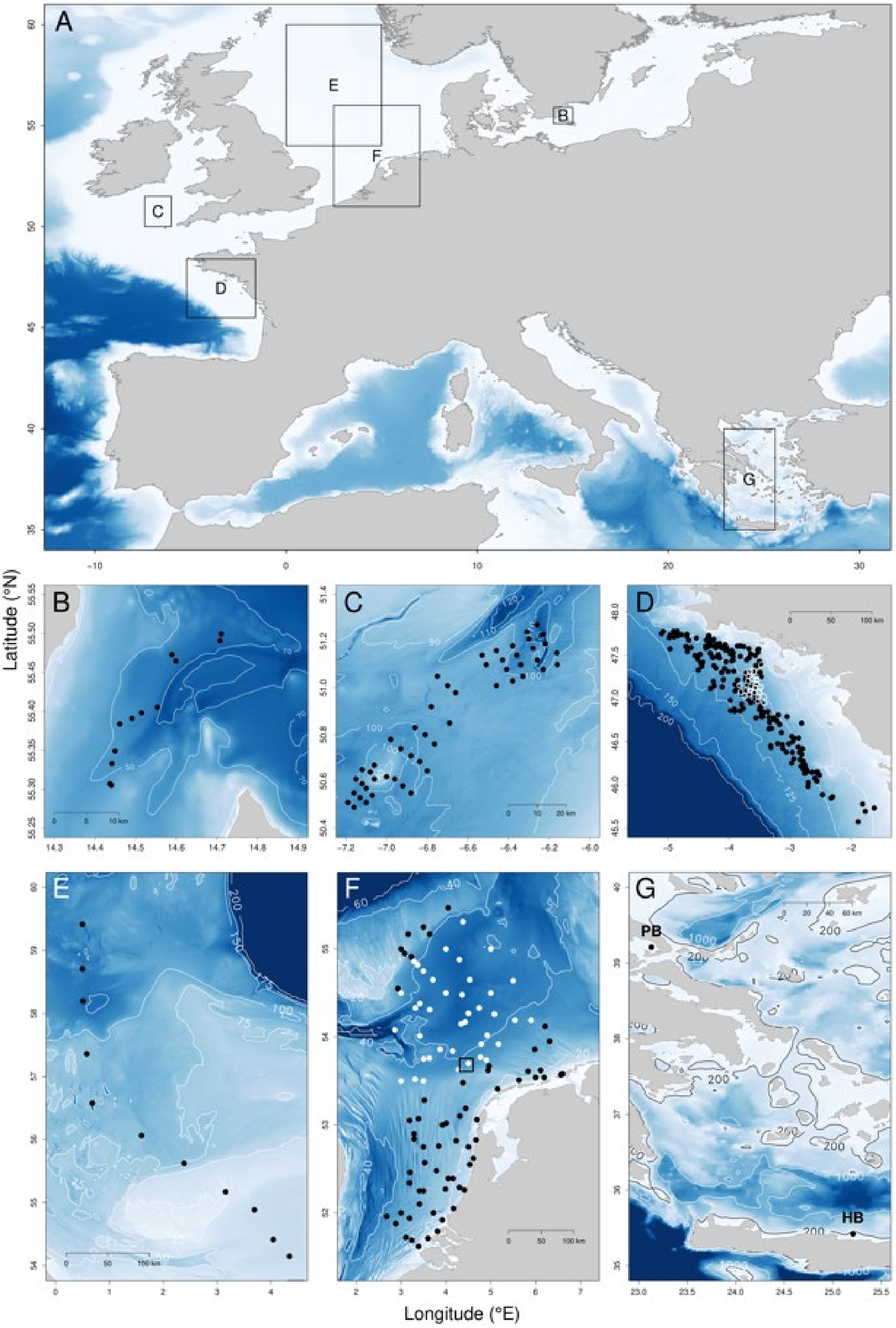


Figure 4.2 Study areas. A) Locations of the case studies in European waters; black frames delineate panels from B to G where sample positions (dots) are mapped. B) Baltic Sea (BS). C) Celtic Sea case study (CS). D) Bay of Biscay; black dots, BBL case; opened white circles, BBF case. E) North Sea transect (NST). F) Dutch sector of the North Sea (all dots, DSNS–WA); white dots, low dynamics case (DSNS – LD); black dots, high dynamics case (DSNS – HD); square in the middle, Frisian Front case (FF). G) Eastern Mediterranean Sea (EMS); PB, Pagasitikos Bay case; HB, Heraklion Bay case; samples are aggregated at very small scale (45 and 50 for PB and HB respectively). Contour lines indicate depth in meters, with contrasts emphasized by blue background adapted to each area.

4.2 Vulnerability of benthic ecosystem functions in European waters

We considered thirteen case studies for which similar information was available, allowing the application of the same analytical methodology (Figure 22); see Beauchard *et al.* (2023a) for details. For each case study, we considered: bottom-trawling intensity (mostly as swept-area ratio (SAR, yr-1); Eigaard *et al.*, 2017); habitat descriptors traditionally used in benthic ecology to explain species community composition such as depth, and descriptors that account for hydrology and sediment type. A brief summary of the main features of each case study is provided hereafter and in Table 4.1.

Table 4.1 Detailed descriptions of the case studies. Swept-area ratio: BACI, before-after control-impact experiment; pres./abs., presence-absence of commercial trawling, estimated close to 1 when present; CV, coefficient of variation. Commercial SAR estimation: ICES data.

Case study	Sampling year	Spatial extent (km)	Number of samples	Depth range (m)	Sedimentary nature	Benthic sampling technique	Total number of taxa	Number of taxa documented for traits	Trawling context	Swept-area ratio (mean ± SD)	Swept-area ratio (CV)
BS	2019–2020	30	33	47–66	From sandy mud to muddy sand	van Veen grab	23	23	Commercial SAR estimation	2.5 ± 2.4	96
DSNS – WA	2009–2018	300	103	6–54	From muddy sand to mixed gravel	Boxcorer	291	205	Commercial SAR estimation	3.5 ± 3.7	106
DSNS – LD	2009–2018	150	40	29–54	From muddy sand to sand	Boxcorer	213	153	Commercial SAR estimation	2.3 ± 1.9	83
DSNS – HD	2009–2018	150	63	6–41	From sand to mixed gravel	Boxcorer	215	166	Commercial SAR estimation	4.2 ± 4.3	102
NST	2018	500	34	26–225	From mud to coarse sand	Boxcorer	180	125	Commercial SAR estimation	2.3 ± 1.3	57
FF	2017	1	16	34	Mainly muddy sand	Boxcorer	57	34	BACI	_	_
CS	2014–2015	50	51	96–137	Sandy mud to coarse sand	Boxcorer	117	94	Commercial SAR estimation	4.7 ± 5.4	115
BBL	2011–2013	100	211	42–133	Mainly muddy sand	Otter trawl	121	85	Commercial SAR estimation	6.9 ± 4.6	67
BBF	2013	20	206	78–112	Mainly muddy sand	Day grab	295	156	Commercial SAR estimation	5.3 ± 2.6	49
EMS – WA	1994–1996	10	95	70–220	From mud to muddy sand	Smith-McIntyre grab	293	141	Pres./Abs.	_	_

Case study	Sampling year	Spatial extent (km)	Number of samples	Depth range (m)	Sedimentary nature	Benthic sampling technique	Total number of taxa	Number of taxa documented for traits	Trawling context	Swept-area ratio (mean ± SD)	Swept-area ratio (CV)
EMS – PB	1994	1	45	90	Mud	Smith-McIntyre grab	106	65	BACI	_	_
EMS – HBB	1995	1	30	70	From mud to muddy sand	Smith-McIntyre grab	239	115	BACI	_	_
EMS – HBS	1996	1	20	196–220	Mud	Smith-McIntyre grab	66	42	Pres./Abs.	_	_

The relationships between trawling intensity and trait composition were explored by combining the RLQ ordination technique and the Fourth-corner testing procedure (Dray *et al.*, 2014). RLQ ordination enables sample, habitat descriptor (matrix R; samples × descriptors), species (matrix L; samples × species) and trait (matrix Q; species × traits) positioning within the same multivariate space (Dolédec *et al.*, 1996), while the Fourth-corner testing provides a conservative assessment of the habitat-traits statistical relationships (Dray and Legendre, 2008, Peres-Neto *et al.*, 2017). This analytical methodology is now widely used in community ecology, but we applied here a modified version given the possible confounding effects between habitat descriptors and trawling on faunal composition. Indeed, correlations between habitat descriptors and trawling intensity could indicate that fishing effort was not equally exerted on all habitats, with impacts possibly biased toward benthic communities of specific functional attributes. To control for such constraints, we applied the "Partial-RLQ" procedure (Wesuls *et al.*, 2011) that consists in regressing trawling intensity and each taxon on habitat descriptors and in extracting residuals as new R and L matrices. In this way, we built a disturbance gradient perfectly correlated to trawling intensity while controlling for habitat variations. In all case studies, matrices L and Q were preliminarily processed by Correspondence Analysis and Fuzzy Correspondence Analysis (Chevénet *et al.*, 1994) respectively, whereas matrix R was processed either by standardized Principal Component Analysis or Hill and Smith Analysis (Hill and Smith, 1976) depending on the presence of qualitative habitat descriptors next to quantitative ones. Following detrending, residual values in matrix L were rescaled by subtraction to the minimum value of the matrix in order to get null or positive values as negative ones cannot be processed by Correspondence Analysis; this does not affect data structure and relationships between taxa (columns) which are attributed the same importance (same column sum) as in Non-Symmetric Correspondence Analysis (Grimaret-Carpentier *et al.*, 1998). Figure 4.3 illustrates the procedure.

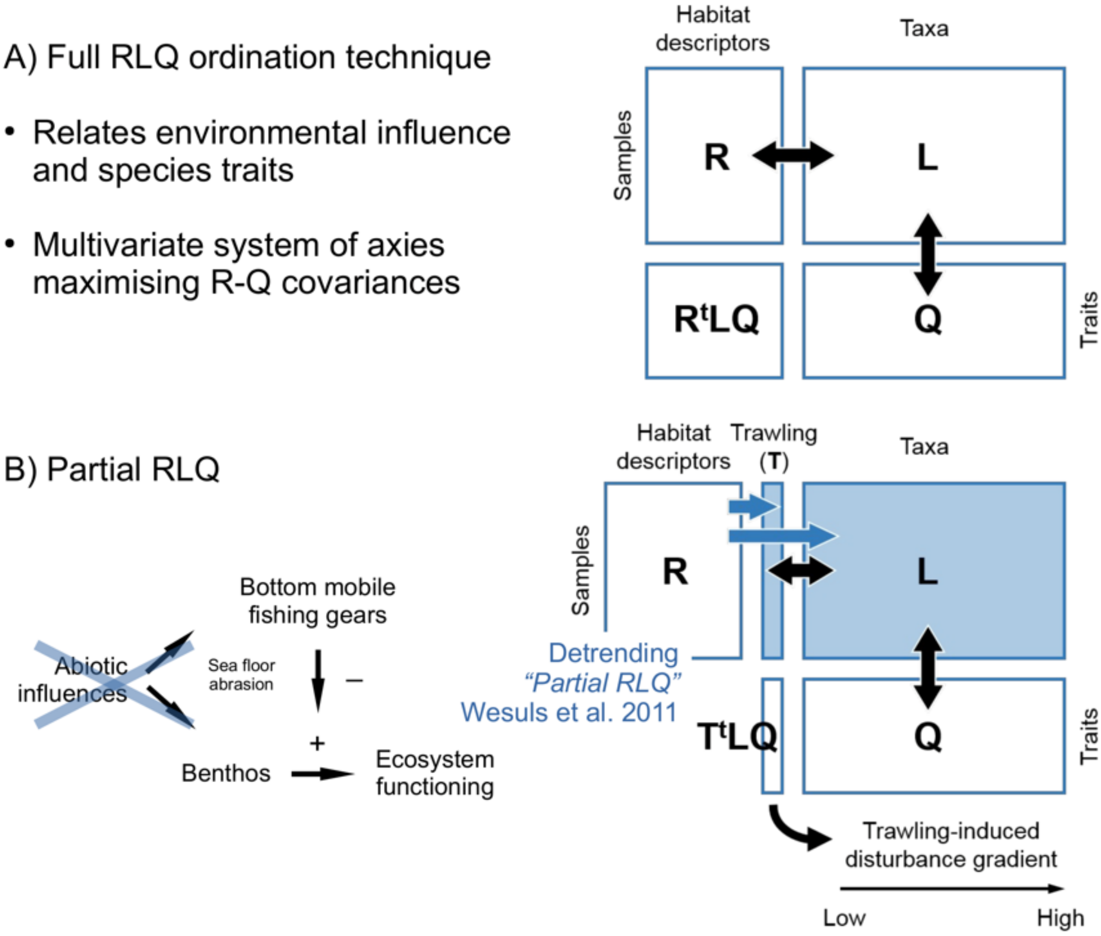


Figure 4.3 Summary of the data analytical procedure. A) Typical full RLQ matrix (rectangles) disposition; black arrows indicate row or column matching; multivariate axes that simultaneously enable sample, habitat descriptor, taxa and trait positioning are obtained through the product AtA with A = RtLQ. B) Partial RLQ, removing environmental influences; blue arrows indicate detrending of the trawling and L variables from the R variables through multiple regression (residual extraction, filled rectangles). As the matrix AtA with A = TtLQ contains only one value (sum of square covariances between the unique variable of T and Q variables through L abundances, i.e. presence-absence, individual or biomass density), the multivariate ordination results in a single axis, with increasing T values from left to right; detrending ensures that variations in trait compositions are solely due to variations in trawling intensity.

Trawling effect was significant in seven case studies. Tubicolous species groups were found at low trawling frequencies in case studies where the sampling technique was grab or boxcorer (endobenthos); in place of these groups, large biogenic structures were found in the BBL case study (otter trawl, megabenthos). At the opposite end of the trawling-induced disturbance gradient (high trawling frequency), deep burrowing species were systematically positioned. These results were clearly indicative of the importance of burrowing depth. As a result, and consistently with ToR a, in no case study lifespan was found significantly covariant along the trawling-induced disturbance gradient.

Also, the functional groups found at the extremities of the trawling intensity gradient were those of largest functional niche breadths, i.e. those that have many contributions to ecosystem functions and contribute the most to functional diversity (Figure 3.4A-B), including large biogenic structures in rocky and mixed substratum habitats (Beauchard *et al.* 2023b, group 1). As a consequence, and according to findings reported in ToR a, functional niche breadth and diversity could exhibit various patterns along the trawling gradient, as corroborated in Figure 4.4 and Figure 4.5, respectively.

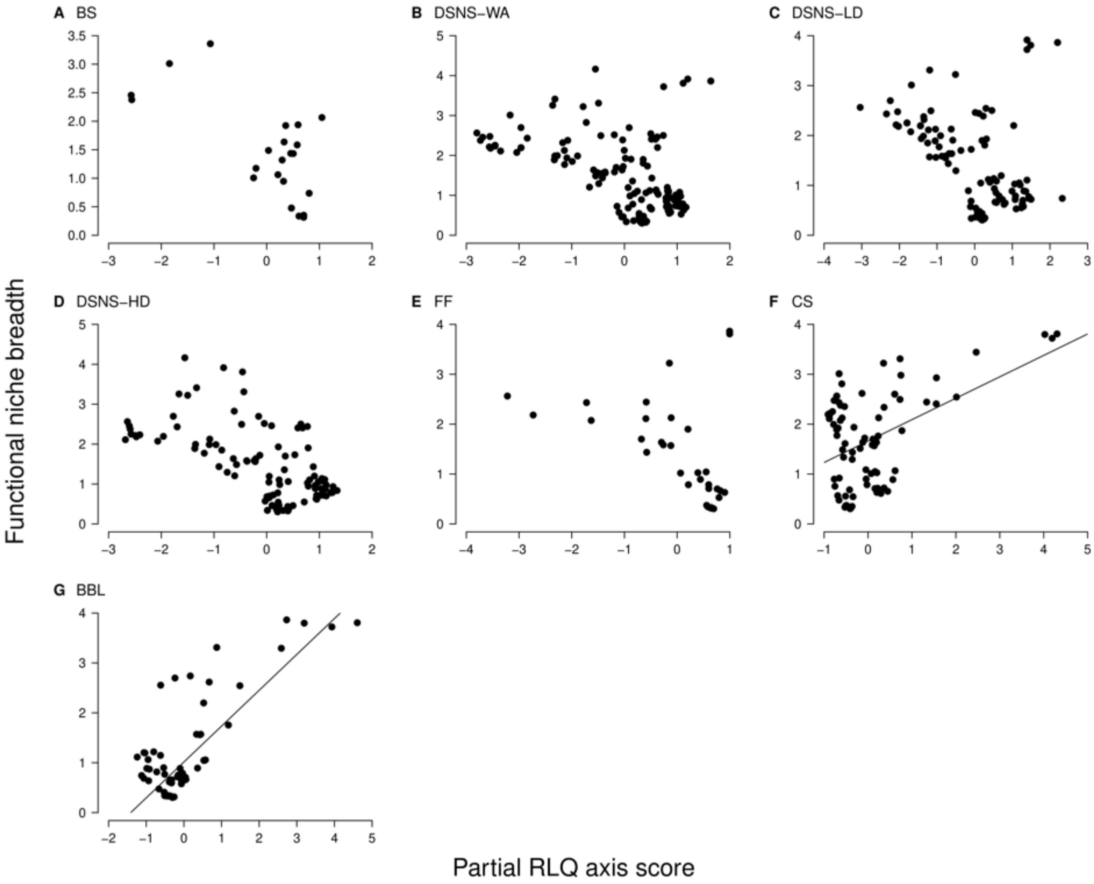


Figure 4.4 Trend in species functional niche breadth as species multi-functionality along the partial RLQ axis (black dots, species). Values can be compared between functional groups in Figure 22 that displays the same species positions along the x-axis: for instance, among species of broad functional niches, tubicolous species (e.g. SmallTub group) on the left side in BS, DSNS and FF case studies, and deep burrowing species (e.g. Deep3D group) on the right side in DSNS-WA, DSNS-LD, FF, and CS case studies. In every case study (see Material and methods for abbreviations), the RLQ multivariate space consists of a single axis (horizontal) expressing an increase in trawling intensity from left to right, but associated with various effect trait compositions (Table 4). Trends can be indistinctly positive or negative and were found significant only for CS and BBL case studies (Table 3). In several cases (e.g. DSNS, FF and CS), the range of variation is a function of trawling intensity. Only median-high values are found to be impacted under low trawling in DSNS-WA and FF case studies (tubicolous worms) while a wider range subsists at higher intensity where deep burrowing and surficial-resistant species occur. The opposite appears in CS and BBS cases: only deep burrowers subsist at high trawling intensity.

Similarly to functional niche breadth, functional diversity could be positively related to trawling intensity (Figure 4.5 F-G). Such relationships are wrongly suggestive of positive trawling effect on functional diversity as they simply result from the removal of highly multi-functional species (that substantially contribute to functional diversity) in some places while it has less effect in some others. Figure 4.6 summarizes the complex process of bottom-trawling impact on ecosystem function.

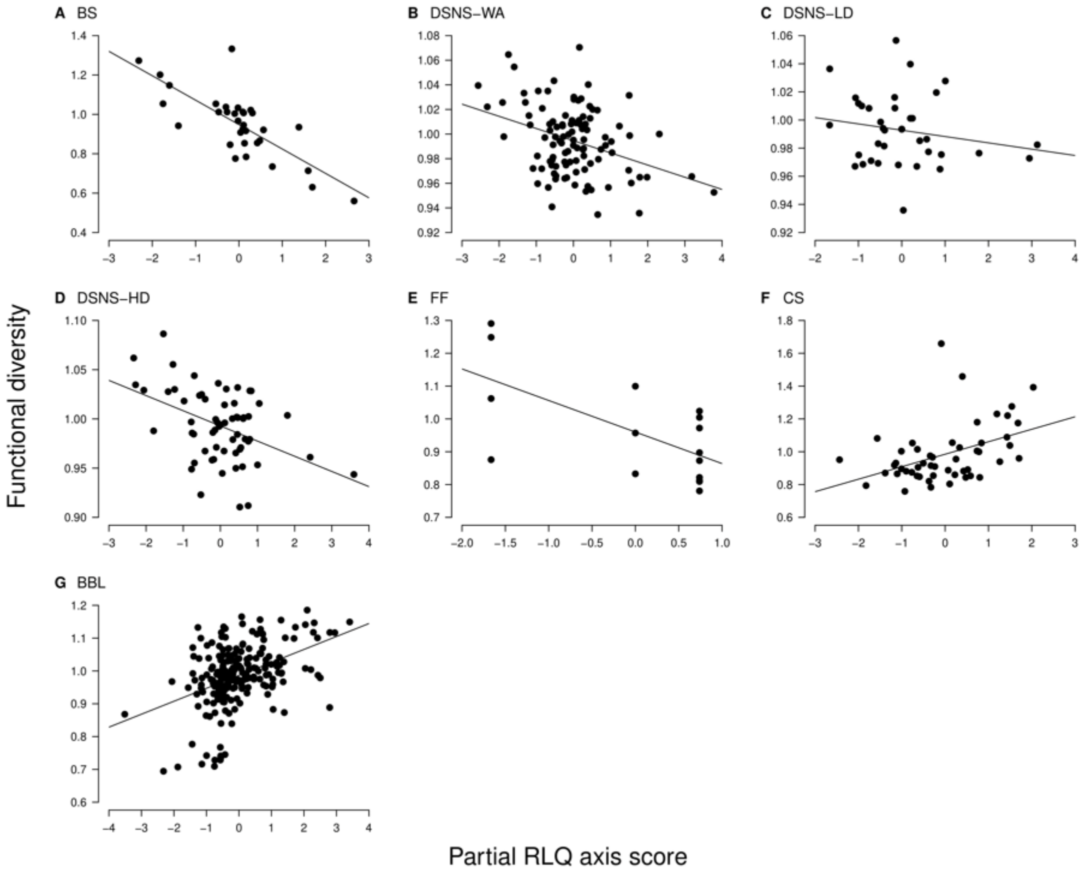


Figure 4.5 Trend in functional diversity (Rao's index calculated from species positions) along the partial RLQ axis (black dots, species communities). Trends, dominantly negative (loss of ecosystem functions), can be positive because of persisting highly multi-functional and deeply buried species at high trawling intensity.

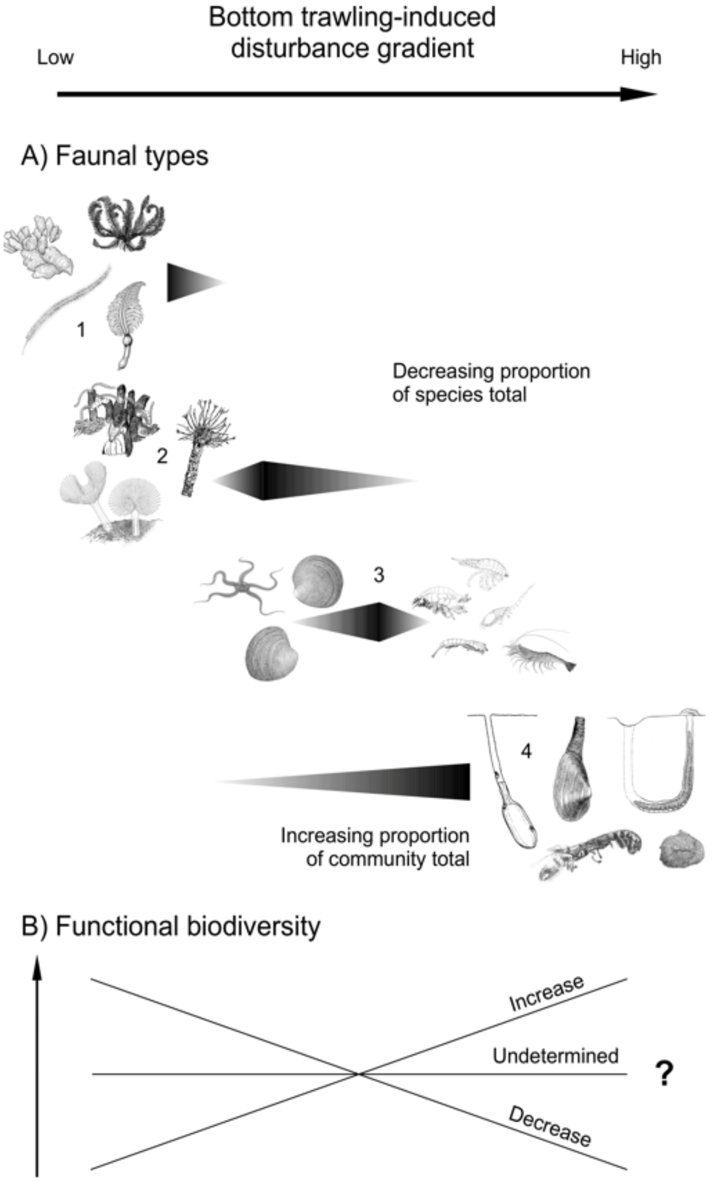


Figure 4.6 Bottom-trawling consequences on ecosystem functions performed by benthic organisms along the trawling gradient. A) Compositional changes along the gradient: (1) extremely vulnerable biogenic structures (e.g. reef builders as habitat providers); (2) tubicolous species (sediment stabilization, biodeposition and advective transfers of materials); (3) intermediately vulnerable (shell builders and mobile surficial sediment mixers); (4) deep burrowing engineers (bull-dozing effect on the sediment, gallery building and bioirrigation). B) Functional biodiversity trends possibly observed along the gradient; the trend can vary as vulnerability is relatively independent from ecosystem function. In no case does bottom trawling increase biodiversity; when observed, a positive trend simply results from the removal of vulnerable and multi-functional species on the left side of the gradient.

The impact of bottom trawling on the ecosystem functions performed by benthic organisms varies according to their functional characteristics. From low to high trawling intensity, organisms living on the sediment surface or within the surficial layer (Figure 4.4 A, type 1 and 2) are more impacted than deep burrowing organisms (Figure 4.4 A, type 4). While the frequencies of the former groups decrease, the latter groups, less exposed to trawling gears, represent a growing proportion of total community abundance. In this context, the related functional biodiversity (FD) increases with species trait dissimilarity within the community (Figure 4.4 B). As there is no general rule to predict functional composition along gradients of soft sediment habitats, FD trend can be context dependent. FD is generated by species that ensure multiple ecosystem functions and that can occur anywhere along the trawling gradient. The indeterminacy in FD trend is explained by the large independence between species multi-functionality and vulnerability (Beauchard *et al.*, 2023b). A decrease in FD indicates that trawling intensity increases in areas of higher abundances in species that are both multifunctional and vulnerable. An FD increase along the trawling gradient does not indicate that trawling promotes FD, such a trend simply results from the removal of vulnerable and multi-functional species on the left side of the gradient while deep burrowers promoting FD persist on the right side (high trawling intensity). Hence, even impact on lowly vulnerable species might have critical consequences on ecosystem function beyond a certain trawling frequency.

4.3 Summary

Complementing ToR a, ToR b provides analyses based on field empirical data that enable to partly state on the accuracy of ToR a predictions. Though the considered case studies do not include deep-sea habitats, one Bay of Biscay case study (BBL), related to megafauna, may analogically testify to the vulnerability of deep-sea community dominantly composed of large epibenthic structures (ICES, 2023a).

- Bottom trawling can clearly impact benthic effect trait composition that determine seabed ecosystem functions.
- There is no lifespan determinant of functional group vulnerability, due to at least three reasons. (1) Following Beauchard et al. (2023a, 2023b), there are only weak correspondences between functional groups and vulnerability. (2) Considering organism vulnerability solely based on lifespan may be too limiting as other functional attributes such as burrowing depth can impart resistance to trawling effect. (3) In most case studies, minimum trawling frequencies (swept-area ratio) range from 1 to 2 times per year, which is enough to prevent some short-lived species to achieve their cycle as evidenced by the systematic decline of tube worms in all case studies.
- Hence, benthic ecosystem engineering, although crucial to marine ecosystem functioning, is not indicative of ecosystem function vulnerability, deep seas excepted. This does not mean that bottom trawling could be exerted on any habitat without impacts as some species of important ecosystem multi-functionality already disappear at low trawling frequency.
- Finally, a relevant benthic ecosystem management should simply and solely focus on species vulnerability, in condition to provide robust biological expertise on traits beyond lifespan, currently insufficient to predict trawling impact on benthic communities.

4.4 Thresholds and tipping points are tempting but not necessarily suitable concepts to address anthropogenic biodiversity change

4.4.1 Overview

There is no number of species that we are safe to lose up to a limit, and setting a limit discourages cautious management. Thresholds and tipping points are commonly used concepts in environmental management; however, they are not suitable to be used to manage biodiversity change for conceptual, ethical, and empirical reasons:

Empirically, most biodiversity change appears to happen gradually, rather than suddenly and disproportionately, as would be expected if a threshold had been crossed. Ethically, thresholds and tipping points deem some species dispensable, when they could be keystone species.

Conceptually, defining a threshold for biodiversity change (a maximum value of acceptable loss) neglects the fact that ecosystems are complex and rely on a complete, entangled webs of species interactions.

Biodiversity change is not linear and there is often a time-lag between a 'cause' and the 'effect'. Few commonalities exist between species in their responses to environmental changes, and some species have stress memories that weaken their resilience to change. Alongside large-scale global policy targets, targets must be formulated at the local and regional scales - this is where management decisions are effectively being implemented.

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Alongside large scale global policy targets, targets must be formulated at the local and regional scales - this is where management decisions are effectively being implemented.

Biodiversity is more than just counting species. Essential Biodiversity Variables are divided into six distinct classes, half of which focus primarily on species, and the other half primarily on ecosystems (Figure 4.7).

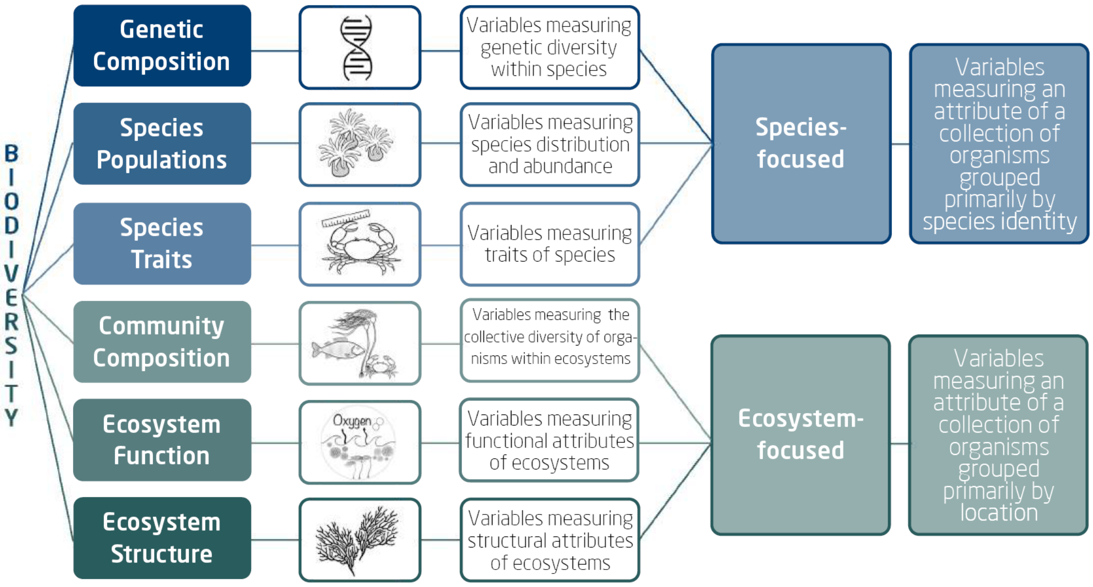

Figure 4.7 Essential Biodiversity Variables (adapted from Pereira *et al.*, 2013).

4.5 Recommendations for Actions

- High precision monitoring is needed so that scientists can use dependable and complete data to evaluate changes in biodiversity.

- Integrated management across scales is required to ensure that both local and global management decisions are aligned.
- Coproduction across scales including the availability of funding to compensate stakeholders for their involvement.

4.6 Thresholds and tipping points are tempting but not necessarily suitable concepts to address anthropogenic biodiversity change—an intervention

Hillebrand *et al.* (2023) consider whether thresholds and tipping points are effective biodiversity targets and communication tools. Thresholds and tipping points are frequently used concepts to address the risks of global change pressures and their mitigation. It is tempting to also consider them to understand biodiversity change and design measures to ensure biotic integrity. Here, we argue that thresholds and tipping points do not work well in the context of biodiversity change for conceptual, ethical, and empirical reasons. Defining a threshold for biodiversity change (a maximum tolerable degree of turnover or loss) neglects that ecosystem multifunctionality often relies on the complete entangled web of species interactions and invokes the ethical issue of declaring some biodiversity dispensable. Alternatively defining a threshold for pressures on biodiversity might seem more straightforward as it addresses the causes of biodiversity change. However, most biodiversity change appears to be gradual and accumulating over time rather than reflecting a disproportionate change when transgressing a pressure threshold. Moreover, biodiversity change is not in synchrony with environmental change but massively delayed through inertia inflicted by population dynamics and demography. In consequence, formulating environmental management targets as preventing the transgression of thresholds is less useful in the context of biodiversity change, as such thresholds neither capture how biodiversity responds to anthropogenic pressures nor how it links to ecosystem functioning. Instead, addressing biodiversity change requires reflecting the spatio-temporal complexity of altered local community dynamics and temporal turnover in composition leading to shifts in distributional ranges and species interactions.

4.7 Two approaches to defining thresholds

Abstractly, thresholds are defined on the x-axis of “drivers” and tipping behaviour on the y-axis of “response,” such that transgressing a critical pressure level (threshold) leads to disproportionately large changes (tipping) of the variable of interest. In a biodiversity context, two ways of defining driver and response emerged (Figure 4.8). The first considers biodiversity loss as the driver and ecosystem processes or properties as the response, asking at what threshold of biodiversity change do we observe a tipping in ecosystem functioning (Figure 4.8a). The second addresses biodiversity change as the response to environmental drivers, aiming to define a threshold level of environmental pressure at which a disproportional change in biodiversity occurs (Figure 4.8b). These two views are obviously closely interlinked, as the biodiversity response in the latter case is the driver of the ecosystem change in the former. But their differentiation is useful beyond subtle semantics, as it reflects two rather distinct fields of the ecological literature on biodiversity change.

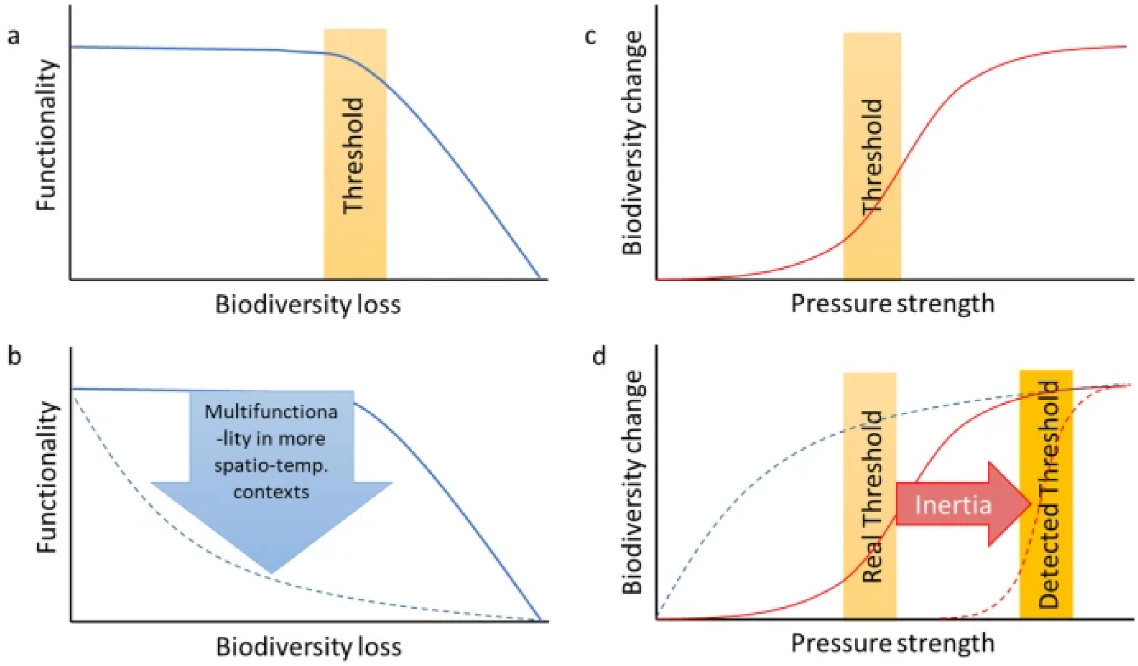

Figure 4.8 Conceptual approaches to defining thresholds of biodiversity loss (a, b) or thresholds of pressures on biodiversity (c, d); a Functionality of an ecosystem in relation to a proportional biodiversity loss, which may not exceed a certain threshold to maintain ecosystem integrity; b Considering this relationship across abiotic contexts and functions leads to estimates of much earlier loss of multifunctionality (dotted line); c Biodiversity change in relation to pressure strength, with a threshold pressure leading to disproportional acceleration of compositional shifts; d Including more gradual change in biodiversity (blue line) or delayed responses through demographic inertia (red dotted line) leads to misplaced thresholds or their absence.

4.8 Global and regional biodiversity targets

Despite the lack of empirical support and the unresolved ethical issues, thresholds of biodiversity change have been formulated in the context of planetary boundaries, initially as a safe operating space for biodiversity (Rockström *et al.*, 2009), later as biosphere integrity including functional and genetic diversity (Steffen *et al.*, 2015). Avoidance of global thresholds is a logical target emerging from these planetary boundaries, such as aiming to reduce global extinction rates to rates of speciation (Rounsevell *et al.*, 2020). However, such global targets incur a scaling issue as global extinction is only the final step of accumulated local and regional extinctions. For a given habitat or region, it is the local or regional extinction that matters, as the species' role in ecosystem functioning depends on its local presence, which is not alleviated by rescuing the species somewhere else.

Regionalizing the biodiversity target formulations such as "no net loss of biodiversity" will not be more useful. Superficially, it seems valid to assume that as long as extinction is less than species gains, a potential critical threshold for biodiversity change is avoided. However, the extinction side of the equation is hampered by the above-described inertia, such that "winners" and "losers" show shifted temporal dynamics poising towards transient increases in diversity rather than losses. Additionally, the local gain of species depends on the availability of further species in the regional species pool (Hodapp *et al.*, 2018), i.e. temporal compositional shifts are inseparably connected to changes in spatial biodiversity (Eriksson and Hillebrand 2019). Additionally, there are implementation issues as well, for example when offsets for biodiversity loss unintentionally risk further biodiversity declines (Maron, 2017).

The alternative is to formulate targets that address a certain amount of biodiversity change. Doing so at local to regional scales reflects the scale on which local assemblages and regional species pools respond to such pressures (Suding and Hobbs, 2009) but would require a case-by-case analysis of thresholds for all relevant pressures. As this will rarely be possible, data synthesis efforts were used to estimate at which level biodiversity disproportionately responds to pressures such as hypoxia (Vaquer-Sunyer and Duarte, 2008), aridity (Berdugo *et al.*, 2020), fragmentation (Andren, 1999), or nitrogen fertilization (Bai *et al.*, 2010). However, these studies often stress the high variability of the threshold estimates such that it seems highly uncertain to derive a target for one system by extrapolating local results from other regions without a proof that this generalization is possible.

Norberg *et al.* (2022) conceptually addressed the question whether basing a management decision on the assumed presence of a threshold does good or harm when the existence or position of this threshold is uncertain. Their simple model shows that the answer depends on (i) how "wrong" the threshold estimate is and (ii) how negative the effects of sub-thresholds pressures are on the management target. If we do not know the answer to either of these items, we must be aware that we should not set targets for pressures in the intellectual darkness of having limited knowledge of the magnitude and timing of the response of the entangled biodiversity web to these pressures. Precautionary principles would require setting targets at very low levels of pressure to prevent any threshold transgression, but this strategy often fails when negotiating the use and protection of natural resources and ecosystems. Taking deep-sea mining as an example, a true precautionary principle would require stopping all mining pursuits for decades, as this is the time estimated for filling gaps in our current knowledge (Amon *et al.,* 2022).

In summary, the threshold and tipping point narratives have little validity in the context of biodiversity change and the formulation of targets to mitigate this change. There is little empirical evidence of biodiversity showing a tipping response to environmental drivers and our ability to predict such threshold pressure levels seems highly limited. Likewise, assuming a tolerable amount of biodiversity change is conceptually flawed and ethically disputable. Thus, formulating targets as if such thresholds existed seems dangerous as they further suggest a "manageability" that in fact is not feasible. We further fear that suggesting actions based on a threshold that does not hold up to scrutiny in discussions of conflicting use and conservation undermines the biodiversity position in such target negotiations.

There is no easy alternative to propose, but that does not invalidate our argumentation. On the contrary, it might be even more important as the current lack of convincing alternatives increases the temptation to accept biodiversity targets based on an uncertain tipping point narrative. However, we can propose a number of important steps that might inform the formulation of future operational biodiversity targets. First, we need to understand the scale sensitivity and inertia in biodiversity responses better that lead to the intertwined temporal changes in composition and spatial changes in species distribution. Second, we must upscale our current knowledge of critical transitions from simple, species-poor systems to communities with a multitude of species interactions (Kéfi *et al.,* 2022). Biodiversity itself might be the reason why ecological systems are able to absorb environmental change without tipping into different states. Third, we need to accept that our ability to manage biodiversity is limited in the first place. Targets are often formulated as if biodiversity responses to further anthropogenic transformation were deterministic, but even in the case of simple pulse disturbances, a recent meta-analysis showed that functional recovery was the norm when the pressure ended, but not to compositional recovery (Hillebrand and Kunze, 2020). All three aspects together indicate that further actions potentially affecting biodiversity need to be discussed under the premise that their impact on biodiversity cannot be easily predicted or made undone.

4.9 Modelling drivers of biodiversity change emphasizes the need for multivariate assessments and rescaled targeting for management

The current policy and goals aimed to conserve biodiversity and manage biodiversity change are often formulated at the global scale. At smaller scales however, biodiversity change is more nuanced leading to a plethora of trends in different metrics of alpha diversity and temporal turnover. Therefore, large-scale policy targets do not translate easily into local to regional management decisions for biodiversity. Using long-term monitoring data from the Wadden Sea (Southern North Sea), Dajka *et al.* (2022) joined structural equation models and general dissimilarity models to enable a better overview of the drivers of biodiversity change (Figure 4.9). Few commonalities emerged as birds, fish, macroinvertebrates, and phytoplankton differed in their response to certain drivers of change. These differences were additionally dependent upon the biodiversity aspect in question and which environmental data were recorded in each monitoring program. No single biodiversity metric or model sufficed to capture all ongoing change, which requires an explicitly multivariate approach to biodiversity assessment in local ecosystem management (Figure 4.10, Figure 4.11, Figure 4.12, Figure 4.13).

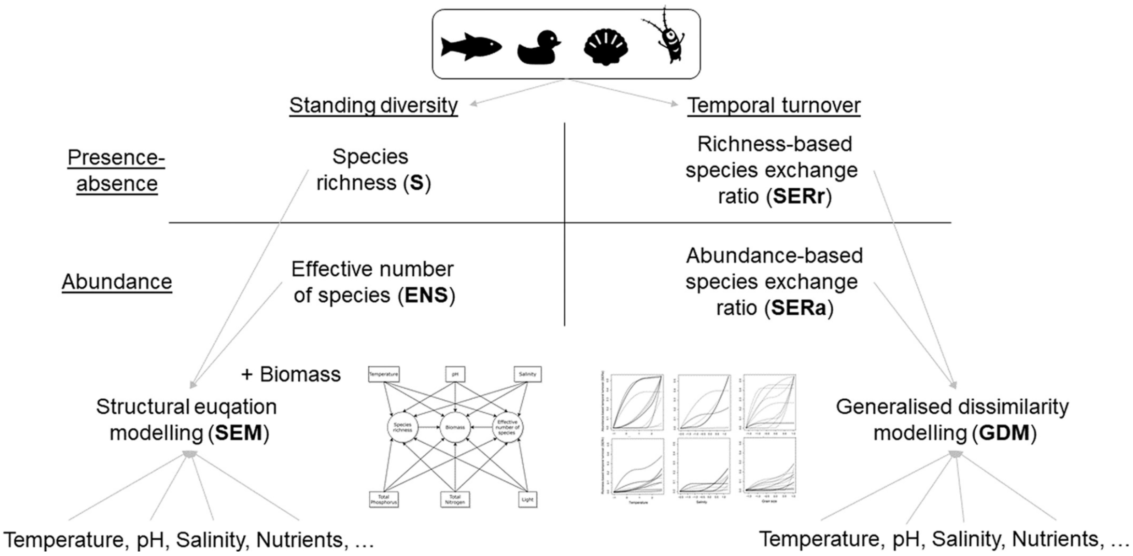

Figure 4.9 Flow diagram of our general approach to biodiversity models of four Wadden Sea organism groups: Fish, birds, macrozoobenthos, and phytoplankton.

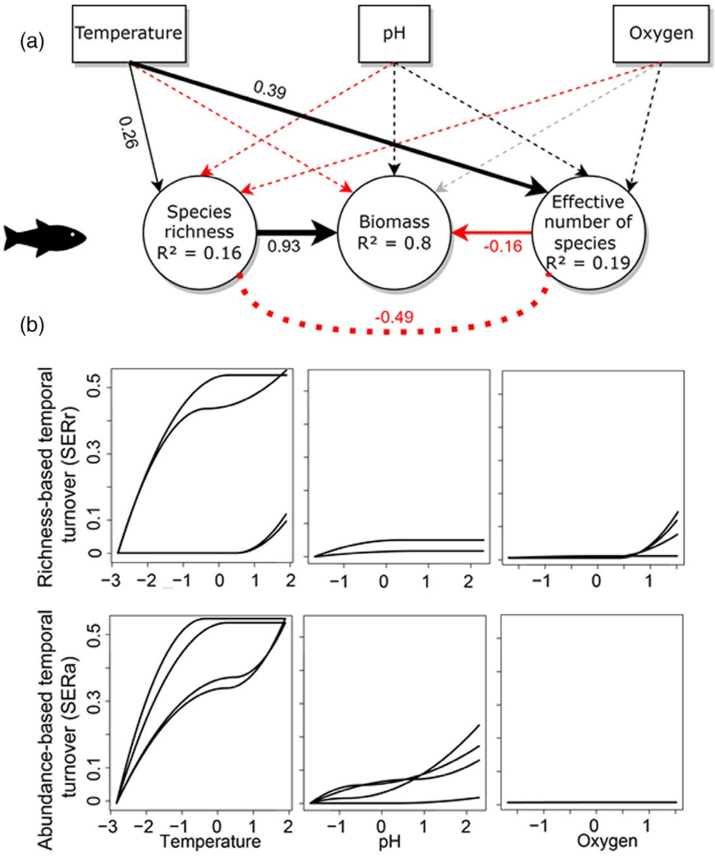

Figure 4.10 Effects of environmental factors on diversity metrics for fish of the Wadden Sea; estimates for standing diversity and biomass are modelled by SEM (a): Endogenous variables with arrows going into them; exogenous variables with arrows coming out of them; dashed arrows: insignificant relationship (p > .1); solid arrows: significant relationship; thicker arrows: higher significance; red: negative relationship; black: positive relationship; grey: indiscriminate relationship (coefficient value below 0.1 or − 0.1); numbers: coefficient strength (from −1 to 1, strengths of individual arrows can be compared with each other); dotted connectors without arrows: correlations (with correlation coefficient numbers) between variables; environmental effects on temporal turnover estimated with GDMs (b) richness-based SERr (top) and abundance-based SERa (bottom): Each line is a significant site, sites where certain links to variables were insignificant, had a coefficient of < 0.1 or the deviance explained by the model was < 15%, were not plotted; environmental variables are scaled to a mean of 0 and standard deviation of 1 to make their impacts directly comparable, the height of the lines signifies a higher impact on temporal turnover of the respective organism community, levelling-off of the curve shows where along the environmental gradient most (or least) of change in turnover occurs.

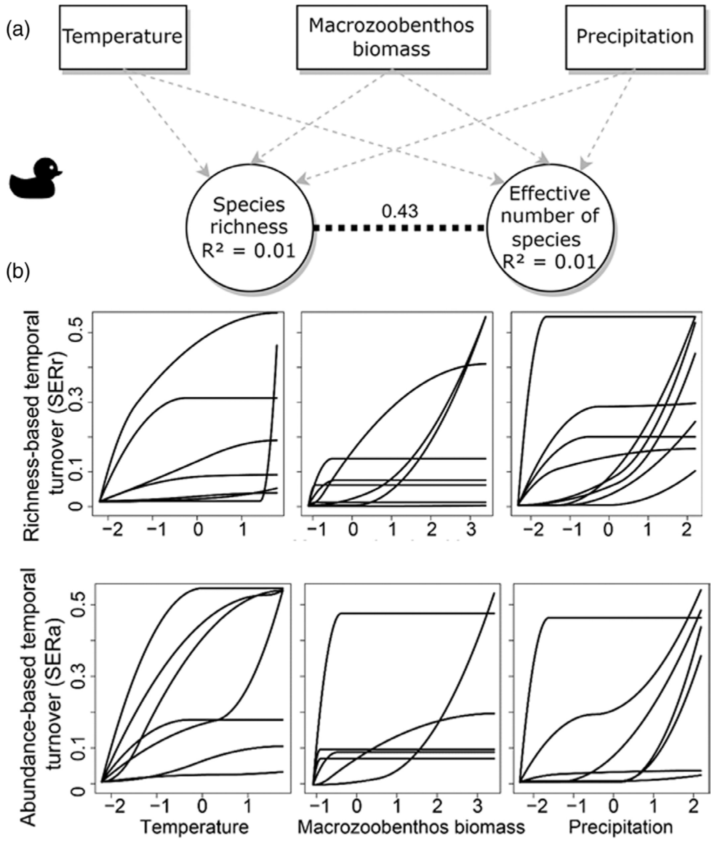

Figure 4.11 Effects of environmental factors on diversity metrics for birds of the Wadden Sea; estimates for standing diversity and biomass are modelled by a SEM (a): Endogenous variables with arrows going into them; exogenous variables with arrows coming out of them; dashed arrows: insignificant relationship ($p > .1$); solid arrows: significant relationship; thicker arrows: higher significance; red: negative relationship; black: positive relationship; grey: indiscriminate relationship (coefficient value below 0.1 or −0.1); numbers: coefficient strength (from −1 to 1, strengths of individual arrows can be compared with each other); dotted connectors without arrows, correlations (with correlation coefficient numbers) between variables; environmental effects on temporal turnover estimated with GDM (b) richness-based SERr (top) and abundance-based SERa (bottom): each line is a significant site, sites where certain links to variables were insignificant, had a coefficient of <0.1 or the deviance explained by the model was <15%, were not plotted; environmental variables are scaled to a mean of 0 and standard deviation of 1 to make their impacts directly comparable, the height of the lines signifies a higher impact on temporal turnover of the respective organism community, levelling-off of the curve shows where along the environmental gradient most (or least) of change in turnover occurs.

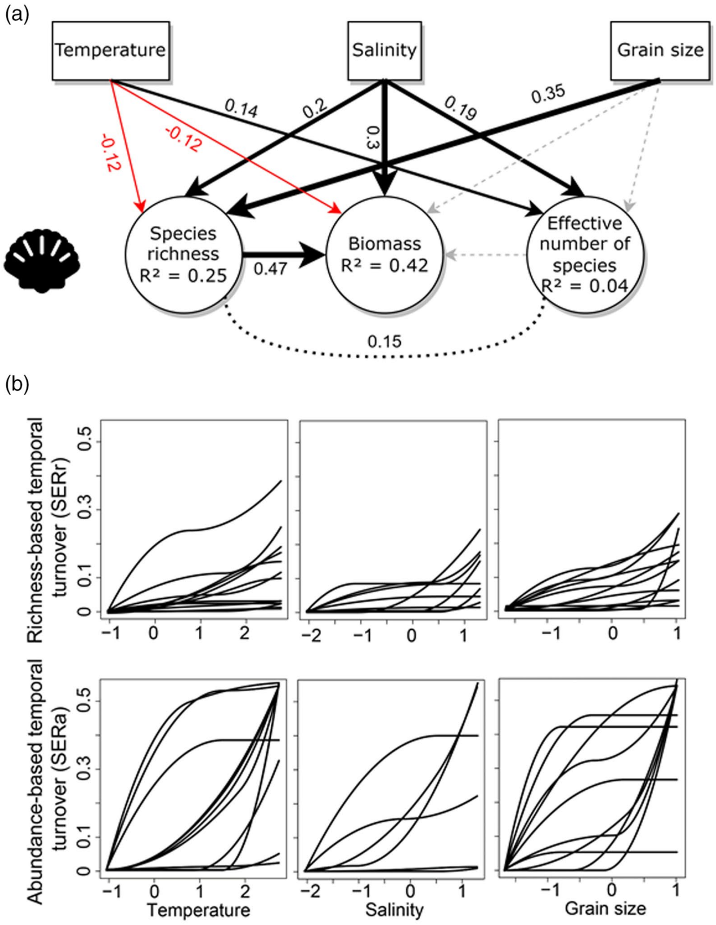

Figure 4.12 Effects of environmental factors on diversity metrics for macrozoobenthos of the Wadden Sea; estimates for standing diversity and biomass are modelled by SEM (a): Endogenous variables with arrows going into them; exogenous variables with arrows coming out of them; dashed arrows: insignificant relationship (p > .1); solid arrows: significant relationship; thicker arrows: higher significance; red: negative relationship; black: positive relationship; grey: indiscriminate relationship (coefficient value below 0.1 or –0.1); numbers: coefficient strength (from –1 to 1, strengths of individual arrows can be compared with each other); dotted connectors without arrows: correlations (with correlation coefficient numbers) between variables; environmental effects on temporal turnover estimated with GDM (b) richness-based SERr (top) and abundance-based SERa (bottom): Each line is a significant site, sites where certain links to variables were insignificant, had a coefficient of <0.1 or the deviance explained by the model was <15%, were not plotted; environmental variables are scaled to a mean of 0 and standard deviation of 1 to make their impacts directly comparable, the height of the lines signifies a higher impact on temporal turnover of the respective organism community, levelling-off of the curve shows where along the environmental gradient most (or least) of change in turnover occurs.

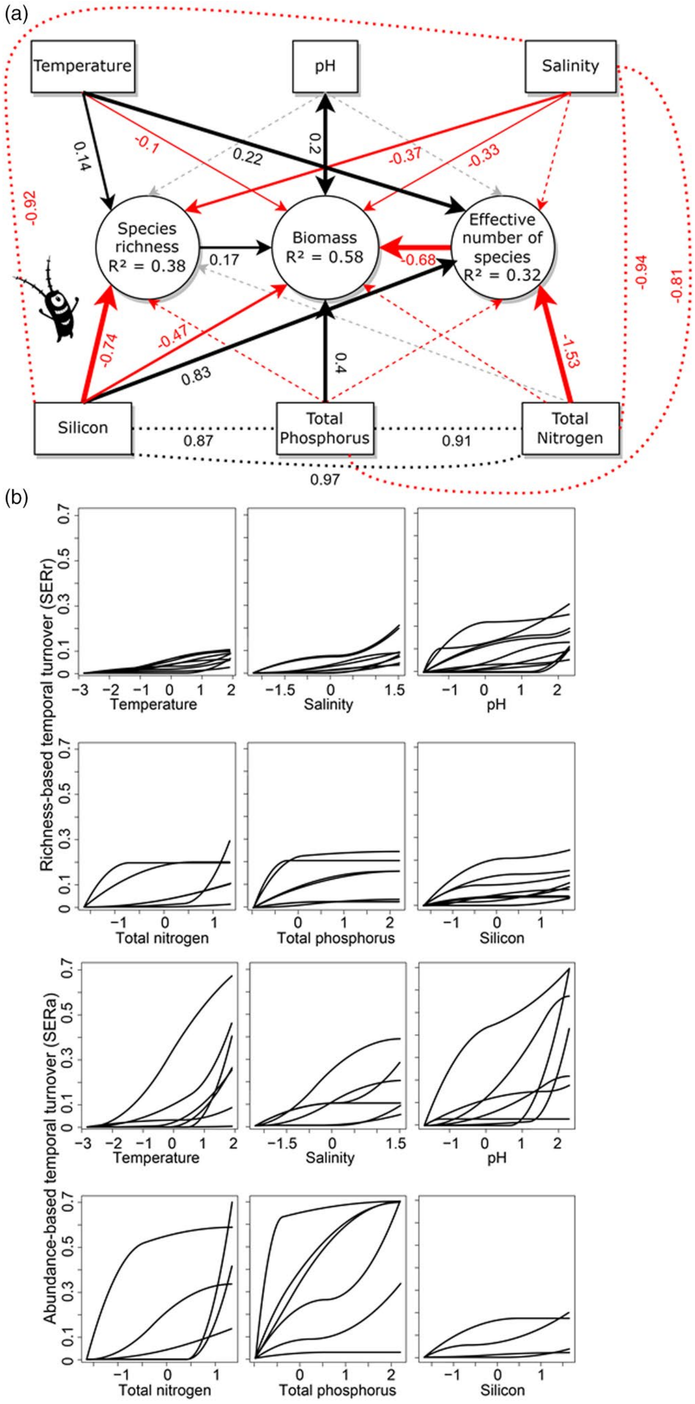

Figure 4.13 Effects of environmental factors on diversity metrics for phytoplankton of the Wadden Sea; estimates for standing diversity and biomass are modelled by SEM (a): Endogenous variables with arrows going into them; exogenous variables with arrows coming out of them; dashed arrows: insignificant relationship (p > .1), solid arrows: significant relationship; thicker arrows: higher significance; red: negative relationship; black: positive relationship; grey: indiscriminate relationship (coefficient value below 0.1 or − 0.1); numbers: coefficient strength (from −1 to 1, strengths of individual arrows can be compared with each other); dotted connectors without arrows: correlations (with correlation coefficient numbers) between variables; environmental effects on temporal turnover estimated with GDM (b) richness-based SERr (top) and abundance-based SERa (bottom): Each line is a significant site, sites where certain links to variables were insignificant, had a coefficient of <0.1 or the deviance explained by the model was <15%, were not plotted; environmental variables are scaled to a mean of 0 and standard deviation of 1 to make their impacts directly comparable, the height of the lines signifies a higher impact on temporal turnover of the respective organism community, levelling-off of the curve shows where along the environmental gradient most (or least) of change in turnover occurs.

Biodiversity change is nuanced, especially at local scales (Chase *et al.*, 2019). Our study confirms that this also applies to the environmental variables driving the changes in biodiversity. First, our study emphasizes the importance of using multivariate assessments of biodiversity. Simple univariate assessments would miss a large part of both biodiversity change itself and the impact of different environmental variables. This also has important ramifications for the ways in which the targeting efforts of management function. Effective management relies on effective targets—from which to measure change and meet goals. These in turn need to be established from data that enable understanding of biodiversity change and be reflective of their drivers as well. As our findings show, multivariate assessments at local scales give a lot more insight into the complex nature of biodiversity change (Rishworth *et al.*, 2020). Additionally, our study shows how multivariate measures of biodiversity change (Hillebrand *et al.*, 2018) can be linked to environmental variables using SEM and GDM. The primary strength of both modelling approaches is that the impacts of all modelled variables are directly comparable and conservation priorities as well as suitable conservation measures can be formulated along the links highlighted in our study. Through these links, we move closer to allowing management of biodiversity change at local scales, where tailored approaches will be more effective. At these smaller scales, targets may be more realistic and meaningful to set and reach.

Our assessment approach requires recording species identities, their abundance, their biomass, and respective environmental variables. We want to urge management to improve their efforts in environmental monitoring to not underestimate the extent of biodiversity change. A clear limitation in our study, for instance in the SEM for birds, arose from the lack of available data, especially those of environmental variables. Currently, the level of detail and sampling consistency of most Wadden Sea species data (especially from Lower Saxony) far outmatch those of corresponding environmental variables. Our assessment here provides a method on how to model the question of biodiversity change and its drivers but does not provide a final answer to it. To continually observe change, monitoring concepts need to consistently match more extensive environmental data to species data and, ideally, collect them together.

Better protection and management concepts can indeed be achieved through improved biodiversity monitoring, data analyses and modelling approaches. Approaches like ours can be part of the blueprint for these concepts. With increasing sample size from longer time-series and from fitting species data with more extensive environmental data, modelling approaches such as SEM and GDM become more accurate, can detect more links, and thus provide a better picture of the modelled system. The phytoplankton dataset in our analysis was paired with more extensive environmental data and we were thus able to create a more detailed model compared to the other organism groups. In contrast, macrozoobenthos species data were measured with higher temporal and spatial resolution but was only complemented with three environmental variables that were congruent between Germany and the Netherlands. This was also the case for the bird data, except that a congruent species-environmental-data combination was only available for Germany. The fish data presented themselves with the lowest sample size from Germany only, which is likely the reason for the few significant relationships in the models. The Wadden Sea stretches across three countries and more extensive datasets are especially hard to come across due to its transnational geography and trilateral governance. Cross-national agreements on environmentally matched, continuously collected time-series data will enhance the complexity and accuracy of models in future, and in turn effective management regimes.

The implications of biodiversity change are being increasingly well understood, but, due to their complexity, cannot be easily addressed. Recently proposed efforts to pursue univariate, global, targets for biodiversity comparable to the 2.0°C-climate target (Rounsevell *et al.*, 2020) would lead to ineffective governance of this complex problem. It is evident from our study and the wider literature (Hillebrand *et al.*, 2018; Rishworth *et al.*, 2020) that multivariate biodiversity assessments at local scales—the scales that are vital for management decisions—should be pursued instead. Focusing on only singular biodiversity metrics in one-dimensional targets will miss major developments of biodiversity change, multidimensional targets must be used instead. These multidimensional targets can be based on multivariate assessments such as ours. The four biodiversity metrics S, ENS, SERa, and SERr are based on species identity and abundance—data that are often already being collected in many monitoring programs. We added biomass to our assessments, but this is not a necessity. For practitioners, this means that multivariate assessments are then only an additional calculating effort. Instead of picking out a single biodiversity metric to focus on, biodiversity monitoring needs to consider at least these four and ideally monitor environmental variables alongside it. Next steps could include connecting model results such as ours to nature's contributions to people (Kadykalo *et al.*, 2019) to reveal the direct effect of biodiversity change on human communities. This provides a chain of links that policy-makers can follow to effectively align their local scale management priorities along meaningful multivariate assessments of biodiversity.

5 Spatial protection measures to conserve and protect biodiversity and meet wider aims (ecosystem function and services; ToR C)

5.1 Testing the Role of Nature-Based Solutions in Climate Change Mitigation and Sustainable Fisheries

5.1.1 Overview

FutureMARES used state-of-the-art digital laboratories for virtual experiments to investigate the effects of climate change and management interventions on human activities for three socio-political scenarios (Global Sustainability GS, National Enterprise NE and World Markets WM) for European regional seas. These experiments used an ecosystem-based perspective and combined Nature-based Solutions (NBS) with Nature-inclusive Harvesting (NIH). In a context of future climate change, with expected further increases in temperature and changes in primary production, these digital laboratories provide new tools to help management interventions to maintain and restore biodiversity and support productive, sustainable fisheries.

This text outlines the results of seven digital representations of European seas that take either a regional perspective (North Sea, Baltic Sea, Bay of Biscay and Western Mediterranean Sea), or a subregional perspective (Finnish Archipelago Sea, NW Mediterranean Sea, and the Portuguese Shelf). Specifically, spatial-temporal marine ecosystem models, using the Ecopath with Ecosim and Ecospace framework, were refined and developed to explore impacts of contrasting climate projections with and without additional management interventions. Follow https://www.futuremares.eu/policy-papers to see more information for this work (Policy Brief 4; Coll *et al.* 2024) and others.

Multiple management interventions were included simultaneously to capture their cumulative effects on the ecosystem, and management goals for actions were specific to each region or subregion according to ecological and policy contexts and stakeholders' preferences. For example, the restoration of habitat-forming species was a priority in the Mediterranean Sea (Posidonia meadows and in the North Sea (native flat oyster reefs), while reductions in nutrient loading were considered a priority for the Baltic Sea. Furthermore, in each area, the protection of areas was chosen specifically to achieve international targets (such as 10% full protection and 20% high protection by 2030) alongside the achievement of sustainable fishing practices (such as reducing discard and bycatch rates and the attainment of maximum sustainable yields). The digital marine laboratories highlight that taking management actions now can make a difference to the status of ecosystems in a future climate.

5.2 Key statements

- NBS – the restoration of habitat-forming species and protection of key marine areas, alongside reductions in excessive nutrient loadings and the attainment of conservation targets for MPAs -together with NIH actions - the reduction of fishing effort and associated rates of discards and bycatch-are predicted to play a vital role in mitigating the impacts of climate change in future. Using digital laboratories, we showed divergent trajectories between the FutureMARES scenarios and their climate change analogues for the biomass of many species and ecological indicators demonstrating likely impacts of management interventions.

- Significant spatial variability of the biomass of species and fishery catch projections has been projected, driven by diverse environmental and ecological conditions, as well as by fisheries activities and management actions. These spatially explicit results highlighted that tailored management interventions will be crucial to ensure favourable ecological and socio-economic outcomes for European marine ecosystems in future.

5.3 Context and Background

FutureMARES tested the ecological and socio-economic effects of climate change and management measures in three scenarios (with their associated climate change only scenarios, RCP2.6 and RCP8.5). First, general scenarios were translated into downscaled narratives and mechaniztically implemented into Marine Ecosystem Models (MEMs) representing both regional and subregional areas. These models were used to run contrasting scenarios that considered different management interventions (including a range of options for protection, restoration and ecosystem-based management of fisheries) across a range of differing regional contexts, current legislations and future developments of relevant legal frameworks (Figure 5.1).

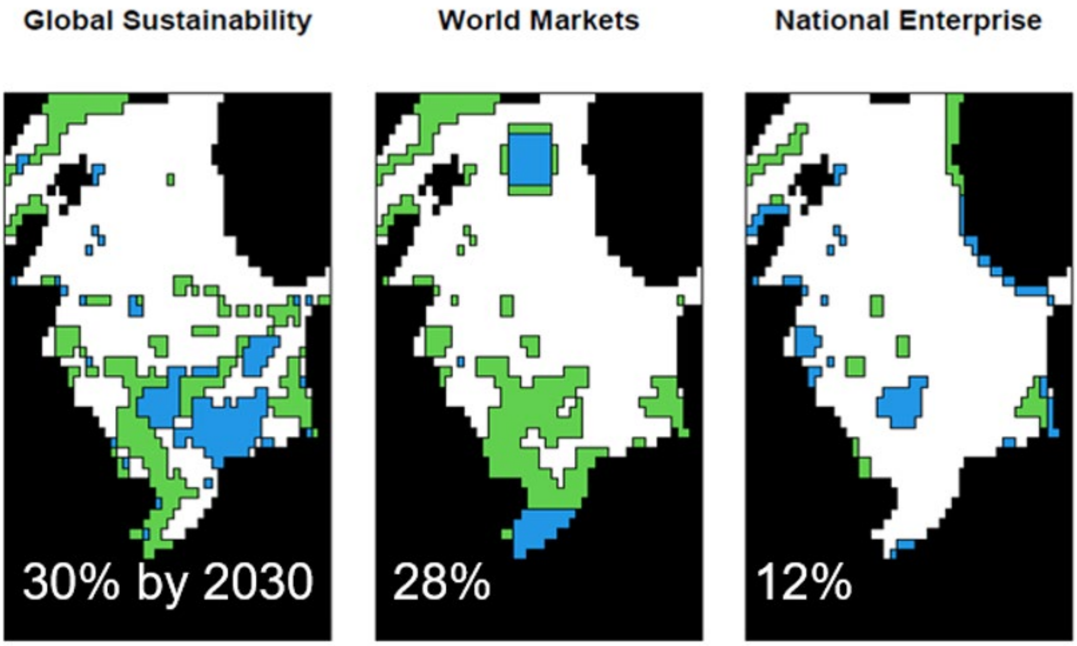

Figure 5.1 Placement of protected areas (areas where artisanal fisheries only are allowed – green - and no-take zones – blue) used in the digital laboratory scenarios for the North Sea.

Spatial-temporal impacts of climate change and human activities on the biomass of key commercial and key conservation species, spanning different trophic levels of the marine foodweb, were assessed for each scenario. Ecological indicators that integrated changes in multiple species and multiple fisheries were also included. Trade-offs between management strategies were investigated, contrasting changes in whole-system diversity, ecosystem structure and service delivery, and its resilience to climate change.

- Global Sustainability: digital labs tested the effects of reaching EU and international legal regulations and targets for restoration of habitat-forming species (flat oysters, blue mussels, seagrass, corals), for protection (Marine Strategy Framework Directive - MSFD, Habitats Directive, Biodiversity strategy, Green Deal) with priority for connectivity and climate-ready solutions, and for a full implementation of EU fisheries directives (Common Fisheries Policy, MSFD), Regional Sea Conventions, and Ecosystem Based Fishery Management principles. These included establishment of fisheries restricted areas, reductions in discarding and bycatch rates, and reduction of fishing effort to achieve values of fishing mortality below F MSY (Fishing mortality at the maximum sustainable yield).

- National Enterprise: experiments tested the effects of prioritizing restoration of high-value species according to food security, job security or coastal protection within EU Exclusive Economic Zone (EEZ), according to national targets. Regarding protection actions, the scenario included small MPAs focusing on national interests with little to no connectivity and high levels of fishing operating in national EEZ following economic subsidies to ensure food security and maximum landed volumes.
- World Markets: priority was given to restore high-value species (key commercial species) with limited-scale interventions, to establish small MPAs with economic value and no connectivity and to prioritize large-scale fisheries over small-scale fisheries, with the aim to achieve a maximum landed value. NE and WM scenarios considered lower targets of discard reduction and bycatch, while the three contrasting scenarios also differed in terms of fish price and fuel costs that alter the distribution of vessels. Finally, status quo scenarios simulated baseline conditions for management interventions with climate change only.

5.4 Key Results

Results showed distinct ecological and fisheries socio-economic outcomes by the mid-term (2050) and long term (2100) according to the three simulated scenarios. The final results were shaped by contrasting environmental conditions (RCP2.6 and RCP8.5) as well as the pivotal influence of the various management strategies (encompassing protection, restoration and ecosystem-based fisheries management) in each FutureMARES scenario (Figure 5.2). This study illustrates that NBS can yield clear benefits when the greenhouse gas emissions is reduced.

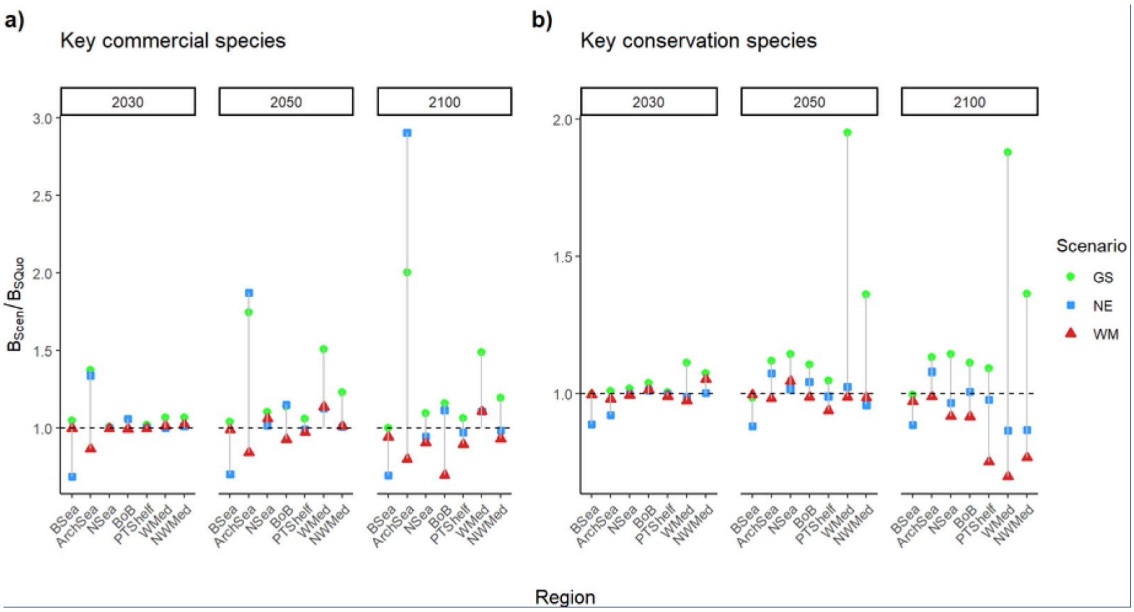

Figure 5.2 Biomass change due to additional management action of key commercial species and key conservation species in 2030, 2050 and 2100 in multiple regional seas. Outcomes are given for the three FutureMARES scenarios (Global Sustainability, GS, National Enterprise, NE, and World Markets, WM) once standardized to the status-quo simulation incorporating climate change but no new management (BScen / BSQuo). Regions reference: BSea: Baltic Sea, ArchSea: Finnish Archipelago Sea, NSea: North Sea, BoB: Bay of Biscay, PTShelf: Portuguese Shelf, WMed: Western Mediterranean Sea, NWMed: Northwest Mediterranean Sea.

5.4.1 Global Sustainability (GS) scenario

Several positive outcomes were projected, including the rebuilding of cod stocks as well as other commercially important fish species in the Baltic and the North Sea, the recovery of habitat-forming species such as corals, mussels and kelp in the Bay of Biscay, the Baltic Sea, the Portuguese shelf and the Western Mediterranean, or the rebuilding of Mediterranean seagrass, small pelagic fish and predators in the Mediterranean Sea and flat oysters and predatory biomass in the North Sea. Interestingly, indicators of ecosystem degradation, such as the increase of gelatinous zooplankton and of cyanobacteria, were predicted to stabilize or decline under GS conditions.

5.4.2 National Enterprise (NE) and World Markets (WM) scenarios

Declines of several important commercial and conservation species were projected. These declines were associated with important changes in ecosystem structure and functioning, and the occurrence of ecological changes in the foodweb, with trophic cascades and predation release effects in a diversity of species. In this context, several trade-offs and synergies were identified. Fisheries outcomes depended strongly on the status of the ecosystem, on the fisheries activities projected to occur in the areas, and on future changes in environmental conditions. In some cases, higher catches were projected under NE or WM scenarios, despite depletions of important commercial and conservation species. However, in other cases, higher catches were projected under the GS scenario. Overall, the status quo scenarios tended to project lower catches than historical levels due to climate effects and, in some cases, than the FutureMARES scenarios.

5.5 Policy Recommendations

In a context of climate change, with expected further increases in temperature and potential changes in primary production, management interventions should be implemented now to maintain biodiversity and support productive, sustainable fisheries in future. Proactive ecosystem-based management interventions that combine actions to restore, protect and conserve marine ecosystems, in combination with sustainable fishing practices, are crucial to shape the future ecological and socio-economic status of European Seas, under all climate change scenarios. Overall, our results show that proactive, regionalized interventions can ensure resilience and attain Good Environmental Status in future, contributing to the recovery of both the ecological integrity and the socio-economic benefits of marine ecosystems for generations to come.

5.6 Could decommissioned energy infrastructure become artificial reefs? A global meta-analysis of ecological effects from offshore marine artificial structures

Marine artificial structures (MAS), including oil and gas installations (O and G) and offshore wind farms (OWFs), have a finite operational period. Selecting the most suitable decommissioning options when reaching end-of-life remains a challenge, in part because their effects are still largely undetermined. Whether decommissioned structures could act (sensu 'function') as artificial reefs (ARs) and provide desired ecological benefits is of particular interest. Lemasson *et al.* (2024) use a meta-analysis approach of 531 effect sizes from 109 articles to assess the ecological effects of MAS, comparing O and G and OWFs to shipwrecks and ARs, with a view to inform their decommissioning. Their aim was to answer the overarching question: (1) What are the ecological effects of MAS? They address several related questions that guide current understanding of the effects of decommissioning: (2) What are the ecological effects of ARs? (3) Do O and G infrastructure and OWF installations act as ARs (otherwise formulated: are all MAS in the sea de facto ARs?)? To finally address (4) can specific decommissioning options be selected to benefit biodiversity and promote positive ecological outcomes?

This synthesis demonstrates that while MAS can bring ecological benefits, important idiosyncrasies exist, with differences emerging between MAS types, habitat types, taxa and ecological metrics (Figure 5.3). Notably, we find limited conclusive evidence that O and G and OWFs would provide significant ecological benefits if decommissioned as ARs. We conclude that decommissioning options aimed at repurposing MAS into ARs may not provide the intended benefits.

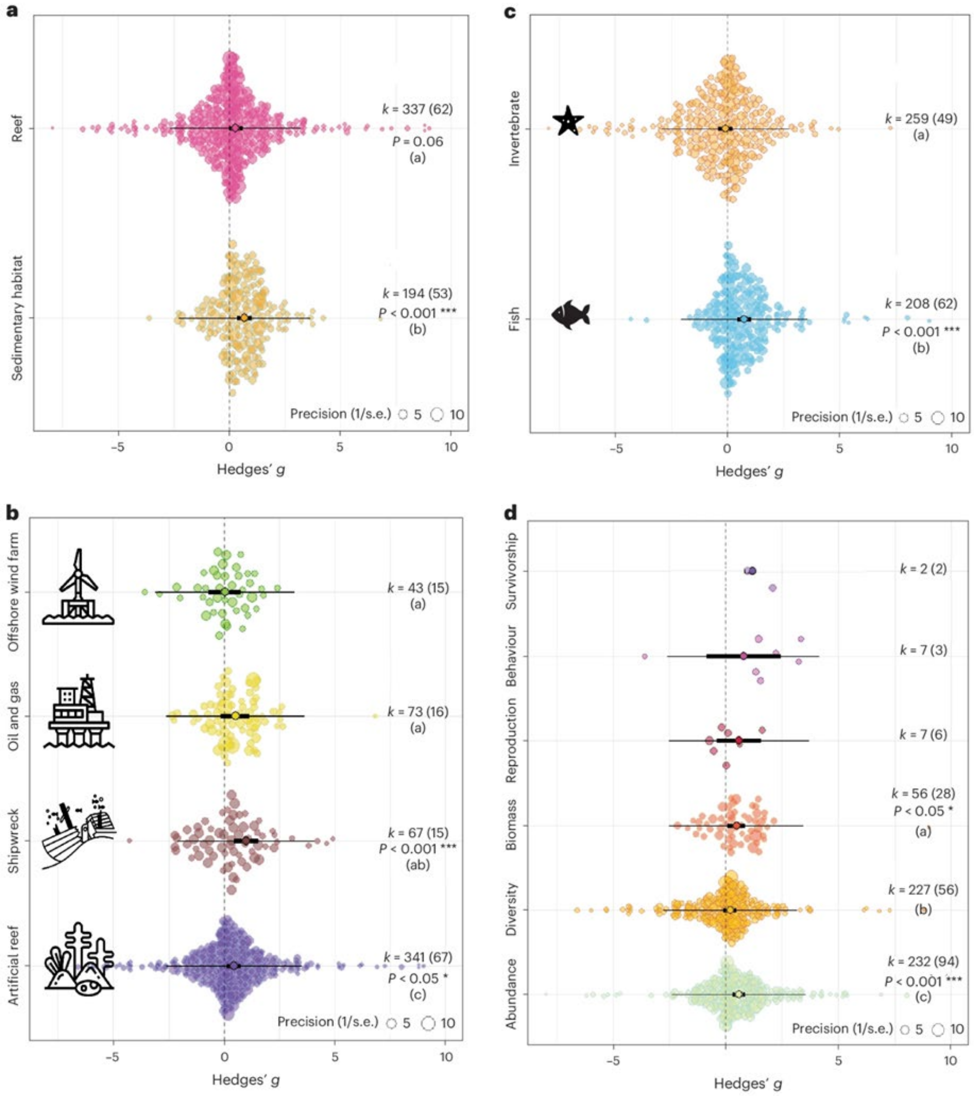

Figure 5.3 Effects of MAS across habitats and ecological metrics – results of subgroup analyses. Orchard plots for each single-moderator meta-analytic model. a, Seabed type as moderator. The model shows significant positive effects of MAS when compared with natural sedimentary habitats without structures (t_{529} = 4.0798, $P < 0.0001$), but not when compared with natural reefs without structures (t_{529} = 1.8431, P = 0.0659). Natural sedimentary habitat: g = 0.69 (95% CI 0.36, 1.03). Natural reef: g = 0.29 (95% CI −0.02, 0.60). b, MAS as moderator. The model shows significant very large and large positive effects of shipwrecks (g = 1.0; 95% CI 0.45, 1.56; t_{520} = 3.5549, P = 0.0004) and of artificial reefs (g = 0.45; 95% CI 0.13, 0.77; t_{520} = 2.7903, P = 0.0055) on ecological metrics compared with sites without structures present, but no significant effects of oil and gas infrastructure (g = 0.52; 95% CI −0.12, 1.16; t_{520} = 1.5914, P = 0.1121) or offshore wind farms g = 0.04; (95% CI −0.68, 0.76; t_{520} = 0.1113, P = 0.9114). c, Taxon type as moderator. The model shows significant large positive effects of MAS on fish (g = 0.77; 95% CI 0.47, 1.06; t_{465} = 5.1397, $P < 0.0001$), but no significant effect on invertebrates (g = −0.08; 95% CI −0.39, 0.24; t_{465} = −0.4713, P = 0.6377). d, Outcome (ecological response type) as moderator. The model shows significant large positive effects of MAS on abundance (g = 0.60; 95% CI 0.33, 0.87; t_{525} = 4.3376, $P < 0.0001$), significant moderate positive effects on biomass (g = 0.48; 95% CI 0.09, 0.87; t_{525} = 2.4034, P = 0.0166), but no significant effect on diversity (g = 0.21; 95% CI −0.09, 0.50; t_{525} = 1.3754, P = 0.1696). Note that survivorship, reproduction and behaviour had low to extremely low sample sizes; hence estimates may not reflect true effect sizes. For each plot, the coloured bubbles represent individual effect sizes from studies, the circled dots represent the estimated mean Hedges' g values, the bold error bars represent the 95% CIs, and the thin error bars represent the 95% prediction interval. k represents the number of effect sizes included for each group; in brackets is the number of studies they originated from. Asterisk denotes groups for which significant effects were detected. For each plot, groups that do not share a letter (for example, (a)) are significantly different from each other. The starfish icon was obtained from pngwing.com. The fish, oil rig, offshore wind, shipwreck and artificial reef icons were made by DinosoftLabs, Freepik, Ultimatearm, Amethyst prime and Eucalyp, respectively, all from www.flaticon.com.

6 Outlook: The people we need for the ocean we want

Due to the strong interconnectedness between the ocean and our societies worldwide, improved ocean governance is essential to sustainable development However, a multitude of different perspectives—ecological, societal, political, economic—and relations between these have to be understood and taken into consideration to foster transformative pathways towards marine sustainability. A core challenge that we are facing is that the 'right' response to complex societal issues cannot be known beforehand as abilities to predict complex marine systems are limited. Consequently, societal transformation is necessarily a journey to- wards the unknown and therefore requires approaches that must enable the involvement of everyone with stakes in future of our marine environment and its resources.

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Annex 1: List of participants

Name	Institute	Country (of institute)
Andrea Belgrano	SLU Department of Aquatic Resources	Sweden
Anik Brind’Amour	Ifremer	France
Chris Lynam	Cefas	UK
Elena Balestri	Scottish Fishermen's Federation	UK
Elena Couce	Cefas	UK
Felipe Artigas	University of the Littoral Opal Coast	France
Gerjan Piet	Wageningen University & Research	Netherlands
Jan-Claas Dajka	Helmholtz Institute for Functional Marine Biodiversity	Germany
Kari Ellingsen	Norwegian Institute for Nature Research	Norway
Matt Holland	University of Plymouth	UK
Michaela Schratzberger	Cefas	UK
Murray Thompson	Cefas	UK
Olivier Beauchard	Royal Netherlands Institute for Sea Research	Netherlands
Pascal Laffargue	Ifremer	France
Ute Jacob	Helmholtz Institute for Functional Marine Biodiversity	Germany

Annex 2: Resolutions

021/FT/EPDSG03 The **Working Group on Biodiversity Science** (WGBIODIV), chaired by Murray Thompson, UK, and Ute Jacob, Germany, will work on ToRs and generate deliverables as listed in the Table below.

	Meeting dates	Venue	Reporting details	Comments
Year 2022	7–10 June	Oldenburg, Germany	Interim e-evaluation	
Year 2023	24–28 April	Ifremer, France	Interim e-evaluation	
Year 2024	15-19 April	Oldenburg, Germany	Final report	

ToR descriptors

ToR	Description	Background	Science Plan Codes	Duration	Expected Deliverables
a	Using complementary measures to provide a multidimensional perspective of biodiversity change.	Efforts to manage and conserve marine biodiversity depend on our ability to understand the mechanisms that affect ecosystem stability at a range of spatial scales, including sufficiently large-scales that meet international strategies and directives (e.g. OSPAR, MSFD). The approach to understanding ecosystems using complementary indices which afford a multidimensional perspective of biodiversity change (e.g. trait and taxonomic diversity in their alpha, beta and gamma forms) provides a way to examine stability and variability at multiple scales (Wang and Loreau 2014; Thompson *et al* 2020). For instance, both variability of species diversity and spatial heterogeneity play important roles in maintaining resource availability via so called 'Portfolio Effects' (Thorson *et al.*, 2018). To our knowledge, however, such patterns in biodiversity across assemblages have not been considered in this context. This ToR (a) will: assess multidimensional biodiversity patterns across assemblages (1.2); make predictions of ecosystem structure and function in space and time relevant to management and service provision, e.g. in designated areas where there is currently limited information (i.e. spatial ToR) and in future under varying climate scenarios (1.3).	1.1 1.2; 1.3	3 years 3 years	Provide a framework and data on how to link ecosystem services directly to marine biodiversity. Production of peer reviewed paper and/ or conference paper. Evidence of ecosystem management and policy, e.g. maps showing biodiversity patterns in space and change over time delivered through WG report.

b	Thresholds responses of marine biota and ecosystem structure: identify areas where marine biodiversity is particularly at risk.	Robust information on the spatial distribution of biota and anthropogenic drivers of change are critical to direct conservation efforts (2.1). Identifying where along anthropogenic pressure gradients notable ecosystem shifts take place will help to inform meaningful and cost-effective interventions. Such thresholds were recently demonstrated for the benthos (Couce *et al.*, 2020) illustrating levels of trawling, for example, at which faunal communities undergo significant change, with consequent effects on ecosystem functioning. ToR (b) will: identify where diversity, ecosystem function (e.g. production and consumption) and ecosystem services (e.g. carbon storage) are particularly at risk, e.g. where thresholds for several pressure gradients coincide and where policy intervention may thus be particularly effective (2.2, 2.3).	2.1 2.2 2.3	3 years 3 years 3 years	Literature review of threshold responses across realms, habitats, ecosystem components (do biodiversity thresholds exist?) Compile relevant datasets across ecosystem components (benthic invertebrates, plankton, fish, elasmobranchs, mammals) Production of peer reviewed paper and/ or conference paper. Evidence of ecosystem management and policy, e.g. maps showing areas where marine biodiversity is particularly at risk delivered through WG report.
c	Examine the efficacy of spatial protection measures to conserve and protect biodiversity and meet wider aims (ecosystem function and services).	The United Nations Sustainable Development Goal 'to 'Conserve and sustainably use the oceans, seas and marine resources' requires governments to assess and mitigate our impact on marine ecosystems. We can manage our pressures through limiting their 'footprint' (spatial protection) and also the level of pressure (e.g. limiting fisheries catches). Understanding which management measures will be most effective requires us to assess, quantify and predict the trajectory of ecological recovery resulting from spatial protection measures. ToR (c) will use data on faunal distribution, spatial protection measures and heterogeneous (e.g. fishing) and large-scale pressures (e.g. warming) identified in ToRs a, b. ToR (c) will establish status, trends and threats (i.e. pressures) for species and habitats of interest (3.1); establish the wider effects of spatial protection measures on biodiversity in light of primary and cumulative pressures (e.g. as identified in ToRs a, b; 3.2); establish the effects of man-made structures on biodiversity (EcoStar, FuEComMS, DREAMS; 3.3).	3.1; 3.2; 3.3	3 years	Production of peer reviewed paper and/ or conference paper. Evidence of ecosystem management and policy, e.g. biodiversity status, trends and threats (inc. cumulative pressures and evolving energy infrastructure) in areas of interest, delivered through WG report.

Summary of the Work Plan

Year 1	Work on all ToRs
Year 2	Work on all ToRs, Publications
Year 3	Work on all ToRs, Publications

Supporting information

Priority	The current activities of this group will lead ICES into issues related to the integrated ecosystem assessments and the implementation of the ecosystem approach to marine management. Consequently, these activities are considered to have a very high priority.
Resource requirements	The research programmes which provide the main input to this group are already underway, and resources are already committed. The additional resource required to undertake additional activities in the framework of this group is negligible.
Participants	The Group is normally attended by ca. 20 members and guests.
Secretariat facilities	Standard EG support
Financial	No financial implications.
Linkages to ACOM and groups under ACOM	There is a linkage to the Integrated Ecosystem assessments Steering Group (IEASG). The results of WGBIODIV are important to WGECO and may be of relevance for IEA groups.
Linkages to other committees or groups	The outcomes of WGBIODIV will be important to the ICES high priority work area 'Marine Strategy Framework Directive (MSFD)'. Linkages to JWGBIRD and WGMME exist.
Linkages to other organizations	OSPAR, HELCOM, European Commission