Ecological Network Analysis Indices

Pilot Assessment

OSPAR

QUALITY STATUS REPORT 2023

2023

Pilot Assessment of Ecological Network Analysis Indices

OSPAR Convention

The Convention for the Protection of the Marine Environment of the North-East Atlantic (the "OSPAR Convention") was opened for signature at the Ministerial Meeting of the former Oslo and Paris Commissions in Paris on 22 September 1992. The Convention entered into force on 25 March 1998. The Contracting Parties are Belgium, Denmark, the European Union, Finland, France, Germany, Iceland, Ireland, Luxembourg, the Netherlands, Norway, Portugal, Spain, Sweden, Switzerland and the United Kingdom.

Convention OSPAR

La Convention pour la protection du milieu marin de l´Atlantique du Nord-Est, dite Convention OSPAR, a été ouverte à la signature à la réunion ministérielle des anciennes Commissions d´Oslo et de Paris, à Paris le 22 septembre 1992. La Convention est entrée en vigueur le 25 mars 1998. Les Parties contractantes sont l´Allemagne, la Belgique, le Danemark, l´Espagne, la Finlande, la France, l´Irlande, l´Islande, le Luxembourg, la Norvège, les Pays-Bas, le Portugal, le Royaume- Uni de Grande Bretagne et d´Irlande du Nord, la Suède, la Suisse et l´Union européenne
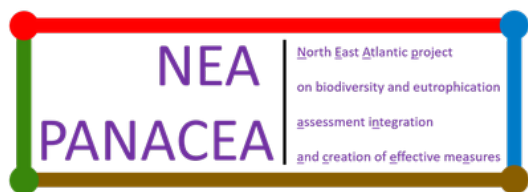
Contents

Key Message

The Ecological Network Analysis indices support the assessment of the structure and functioning of food webs by analysing interactions between all ecosystem components for four case studies. Different trends of indices were found over time and space, some linked to variability in primary production.

Message clé

Les indices d'analyse des réseaux écologiques soutiennent l'évaluation de la structure et du fonctionnement des réseaux trophiques en analysant les interactions entre toutes les composantes de l'écosystème pour quatre études de cas.

Différentes tendances des indices ont été observées dans le temps et dans l'espace, certaines étant liées à la variabilité de la production primaire.

Background

Food webs are complex networks of both feeding (trophic) and non-trophic interactions among species from different functional groups (primary producers, plankton, benthos, fish, birds and mammals) in ecosystems. Therefore, holistic and integrative approaches are needed to assess the ecological status of food webs and ecosystems. This model-derived indicator considers: (i) all ecosystem compartments starting from primary producers via plankton to top predators like fish, birds and mammals, (ii) all direct and indirect trophic interactions between ecosystem compartments within the food web and (iii) non-feeding pathways such as respiration, export out of the systems and pathways to detritus pools. The FW9 indicator allows (1) to assess the structure and functioning of food webs based on the analysis of the interactions among all compartments (e.g., species, functional groups, trophic guilds); (2) to identify the most important trophodynamic links between compartments, and (3) to analyse the effect of specific pressures on Ecological Network Analysis indices or biomass distribution of specific compartments.

In this pilot assessment, the indicator is applied in three COMP areas in OSPAR Region II (Bay of Seine, Elbe Plume and Kattegat) and one in OSPAR Region V (Azores). Several monitoring and survey data collected from different ecosystem components were used to build up the food web models. These pilot studies demonstrate the applicability of this indicator across different ecosystems (coastal, shelf, deep sea), which will be further elaborated under the discussion section.

Background (extended)

Food webs are complex networks of feeding (trophic) and non-trophic interactions among diverse species from different functional groups (primary producers, plankton, benthos, fish, birds, mammals) in ecosystems (**Figure a**). Therefore, holistic and integrative approaches are needed to assess the ecological status of food webs and ecosystems. The Ecological Network Analysis (ENA) is one methodology to holistically analyse food webs and its environmental interactions.

Ecological Network Analysis indices have been used worldwide to compare food web structure and functioning spatially (e.g., Baird and Ulanowicz 1993; Scharler and Baird, 2005; Niquil *et al.*, 2012; Chrystal and Scharler, 2014; Tecchio *et al.*, 2015), seasonally (e.g., de la Vega *et al.*, 2018a) and over time (e.g., Heymans *et al.*, 2004; 2007; Schückel *et al.*, 2015). Consequences of changes in food web structure and functioning due to different pressures were successfully evaluated with Ecological Network Analysis indices. Consequences of changes in food web structure and functioning due to different pressures were successfully assessed with Ecological Network Analysis indices, namely: fisheries (e.g., Heymans *et al.*, 2004, Preciado *et al.*, 2019), invasive species (Baird *et al.*, 2012, Jung *et al.*, 2020), harbour construction (Tecchio *et al.*, 2016), construction of offshore windfarms (Raoux *et al.*, 2017), dredging and dumping (de Jonge and Schückel, 2019) and cumulative impacts of effects of marine renewable energy and climate change (Nogues *et al.*, 2021). Ecological Network Analysis captures and assesses the complexity of ecological systems with the potential to bridge the ecological and socio-economic systems (de Jonge *et al.*, 2012; Borrett *et al.*, 2018). Ecological

Network Analysis allows the analysis of food webs at different hierarchical levels: the level of the whole ecosystem, the level of inter-compartment (bilateral) comparison, and the level of the individual compartment (Kay *et al.*, 1989).
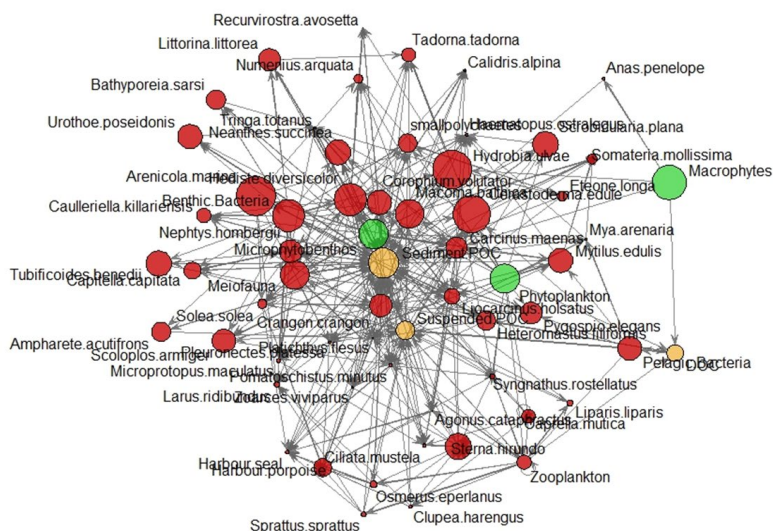
Figure a: The Ecological Network Analysis (ENA) is an integrated system ecology-oriented approach and analyses all trophic interactions (flows in carbon) among all ecosystem compartments in a food web. This example shows a coastal food web in the German Wadden Sea (Schückel *et al.*, 2015). Size of circles represents input biomass in carbon, green circles = primary producers, red circles = heterotrophs, yellow circles = detritus pools. Arrows: interactions (flows in carbon) among compartments.

Assessment Method

Ecosystems comprise a rich food web of many interactions and it would be remiss to exclude, *a priori*, most of them or to rely on analysis techniques that do so. Ecological Network Analysis is capable of analysing the structural and functional properties of this food web of interactions without reducing the model. Therefore, network models aim to include all ecological compartments and interactions and the analysis determines the overall relationships and significance of each (Fath *et al.*, 2007).

A detailed description of the Ecological Network Analysis (ENA) methodological framework and its ENA indices is provided in the FW9 CEMP Guidelines under EcApRHA Deliverable 3.4.2 and published by Fath *et al.*, (2007), Safi *et al.*, (2019) and Fath *et al.*, (2019). **Table 1** provides a step-by-step list how to construct an ecological network.

The construction of the food web model for the targeted ecosystem should be complete before calculating the ENA indices. The first step is to identify the ecosystem of interest and its spatial and/or temporal scale. Once the ecosystem has been chosen, it is necessary to compartmentalise the system into the major species, functional groups or trophic guilds.

Once the compartments have been chosen, biomass data of each compartment are needed to create the:

1) Community table: In general, time-series data of the plankton biomass (i.e., phytoplankton, zooplankton) is required along with biomass data of benthic organisms, fish and invertebrates, mammals, and birds. In addition, data of non-living compartments is required. Annual averages were used to build up the food web models over time. Biomass is converted to carbon. Monitoring programmes of Contracting Parties, scientific publications and data provided by OSPAR data calls of remaining biodiversity assessments that already cover

many specific compartments (primary producers, zooplankton, benthos, fish, birds, mammals) to fulfil demands of other biodiversity indicators were used to build up the pilot food web models.

2) Energetic ratios: In addition to biomass, physiological parameters, such as consumption (C), production (P), respiration (R) and egestion (E) must be quantified. To do so, several ratios of processes over biomass [such as production over biomass ratios (P/B), consumption over biomass ratios (C/B) or respiration over biomass (R/B)] were used from literature. To support this step, Germany already created a database including all relevant information (ratios, also diet) of >300 species that could be included into an OSPAR "living" food web database in future.

3) Diet matrix: Finally the dietary matrix is required. To gain a better understanding of the strength of the relationships occurring within a food web, a measure of the amount of each compartment representing a prey-item for another is needed (Who eats whom and at what rate?). In general, stomach content analysis and stable isotope analysis are used to account for these measures. In case that, for example, not all prey species are available for mainly omnivorous predators in a given year, it is assumed that more of the available prey species will be eaten (not the case for food web models using trophic guilds).
If this information is not locally available, literature or online databases (e.g., DAPSTOM, fishbase.org, MARLIN BIOTIC) can be used to estimate these measures. In addition, diet information was also collected to calculate OSPAR food web indicators (FW4, FW7) and thus could also be used to fulfil demands of the diet matrix needed for FW9.

By using the community table and the diet matrix a flow matrix is generated. This quadratic matrix is arranged in columns and rows to determine whether or not, and how much energy transfer (carbon transfer) flows from compartment j to compartment i. In addition to within system flow transfers (feeding transfers), it also contains flows to the detritus pools due to egestion of species/trophic guilds and outputs out of the system (due to respiration). Finally, the network will be balanced. For all compartments, inputs should balance outputs (Consumption = Production + Respiration + Egestion), in accordance with the conservation of matter and the laws of thermodynamics. Final output is a steady-state ecological flow network, which also includes non-feeding pathways such as export or respiration out of the system and pathways to detritus pools. This network represents the input matrix for the calculation of all relevant ENA indices. Besides Ecological Network Analysis, other approaches have been used to construct flow networks from ecological data. In the OSPAR region other existing operational Ecopath with Ecosim (EwE) models provide mass-balanced flow network from ecological data at the spatial resolution required by the MSFD Another approach for constructing ecological flow networks has been inverse modelling (Vezina and Platt, 1988).

Table 1: Steps for constructing ecological networks.

Step 1	**Identify ecosystem of interest and its spatial and/or temporal scale.**
Step 2	Make a list of important species, functional groups or trophic guilds in the ecosystem (primary producers to apex predators).
Step 3	Select a unit of currency for the network. Typically, the currency is biomass (in carbon).
Step 4	Construct the community table by including the biomass (carbon/area, annual averages) for each compartment in the network. If no measured biomass data for some compartments are available, literature data can be used.
Step 5	Include physiological parameters such as Consumption (C), Production (P), Respiration (R) and Egestion (E) for each compartment into the community table.
Step 6	Construct the diet matrix.
Step 7	Construct the flow matrix by using the community table and the diet matrix.
Step 8	Balancing, steady state of network
Step 9	Apply ecological network analysis to network. Calculate ENA indices.

Ecological Network Analysis indices are numerous and each ENA index describes different aspects of the food web. The FW9 indicator comprises a set of indices derived by Ecological Network Analysis. The set of indices including an explanation and its linkages to the MSFD criteria is given in **Table 2**. It is important to emphasise that the spatial scale that the ENA is run for in the present pilot assessment is based on COMP4 areas (see below) and does not reflect the requirement of MSFD for the time being. This proposed set of ENA indices reflect the outcome of the EU project EcApRHA (Safi *et al.*, 2019), and outcomes of common international workshops in 2017, 2018 and 2022 between experts of different ecosystem model approaches such as Ecological Network Analysis or Ecopath (Schückel *et al.*, 2018, Wolff *et al.*, 2019, Fath *et al.*, 2019). The selection of these indices is based on (i) published results and expert judgement of the high sensitivity of theses indices to capture changes in food webs, (ii) sensitive to specific pressures, (iii) analysing the structure, diversity and functioning of ecosystems, which is an important quest emerging from European Directives and (iv) the potential of theses indices to be easily communicated to stakeholders. The proposed indices focus on both specific parts of the food web (e.g., primary producers, secondary producers) or the whole food web. A detailed description of ENA indices and their related algorithms is provided in Kay *et al.*, (1989). Several of these indices follow Margalef´s (1968) and Odum´s (1969) theory of ecosystem development that occurs as a food web develops with succession.

Table 2: ENA indices and their explanation.

ENA Indices	Explanation	MSFD criteria
Detritivory over Herbivory ratio	The second trophic level of a food web can rely on two feeding modes, herbivory (consumption of autotrophic organisms) and/or detritivory (consumption of non-living organic matter). Detritivory is the sum of all predation flow to detritus, herbivory the sum of all predation flows to primary producers (Kay *et al.*, 1989). Odum (1969) relates changes in Detritivory to the maturity of ecosystems. The ratio indicates if the food web rely on freshly produced algal material (primary producers: pyhtoplankton, algae) or detritus. Thus, an ecosystem that shifts from high Detritivory to low Detritivory is less dependent on detritus and more dependent on primary production. The index is an intuitive ratio and is easily understandable to every end-user.	D4C4
Finn Cycling Index	The Finn Cycling Index (FCI) quantifies the importance of cycling and recycling in the system (Finn 1976, Finn 1980). It thus indicates the independency of the system of additional carbon input (input of primary production or dissolved organic carbon and particulate organic carbon). Going from high FCI towards low FCI values is represented by the reduction in the number of cyclic pathways within the food web or the reduction in the flow value within those cycling pathways (Finn, 1976, 1980). Finn (1976) states that if there is no cycling the index is zero.	D4C4
Relative Redundancy	This index focusing uniquely on prey-predator interactions. It measures the number of parallel pathways in the food web, but it is also a way for ecosystems to show a high resilience against	D4C4

	disturbances as one pathway can replace another one (Ulanowicz, 1997). It decreases as the food web´s specialisation increases, i.e., food web compartments become more dependent on one prey species or resource. It has been shown as sensitive to persistent ecosystem changes such as overfishing combined with impacts such as temperature fluctuations (Heymans and Tomczak, 2016).	
Relative Ascencendy / Relative Overhead	The Relative Ascendency describes both growth and organisation of flows in the food web and thus represents the carbon flow efficiency of any food web. Growth is measured as any increase of total system throughput, which is the sum of all flows within the food web and between the system and its outside (imports, exports, respiration). Total system throughput can increase either by increasing the spatial extent of the system or by including more species/trophic guilds, or by an increased activity of the system (e.g., during phytoplankton blooms). The Relative Overhead is the complementary component and represents the unorganised and inefficient part of the food web flow structure. If a system become overly organised, it will be prone to disturbances. The Relative Overhead can be used to express the systems' resilience because i) redundant pathways create alternative paths for carbon flows, ii) carbon import represents potential resource for the food web species to grow while iii) the respiration product as carbon dioxide represents a resource for primary producers (de Jonge and Schückel, 2021).	D4C2
Flow Diversity	Flow Diversity measures the diversity of flows and with which components of the system are used as food resources by other living components, in other words it measures the evenness of flows and reaches a maximum if all flows are equal in magnitude (Kay *et al.*, 1989).	D4C2, D4C1* (*food webs on species level)

Spatial scales:

The analysis was performed in different 'COMP4 assessment areas'. A detailed description of the assessment areas and related boundaries are provided in the FW2 pilot assessment. Areas within the COMP4 assessment areas assessed are the Elbe Plume, the Kattegat (coastal and deep), the Bay of Seine and the Azores. The Elbe Plume covers an area of 7 832 km², the Bay of Seine model covers an area of 5 000 km² and the Kattegat models cover an area of 4 957 km² (deep) and 9 629 km² (coastal), respectively. The Azores model represent a study area to the boundary of the EEZ, which covers an area of 954 563 km².

Data provided and used in this assessment

The datasets used for this pilot assessment have been collated from OSPAR data calls and different research projects carried out by experts from the OSPAR food web expert group and from peer-reviewed scientific publications. The datasets have different temporal and spatial scales (**Table 3**).

Table 3: Contracting Parties and institutes that provided the datasets for the FW9 assessment.

Contracting Party	Institute	Datasets	Date range
Portugal	Marine and Environmental Sciences Centre, Universidade dos Açores, Horta	local data, fisheries research cruises, mesopelagic trawling surveys	1997-2018
Denmark	Dept. of Bioscience, Aarhus University	Benthic data from monitoring program NOVANA, fish data DATRAS	2009-2018
Sweden	Hafok AB, Department of Aquatic Resources, Institute of Marine Research, Swedish University of Agricultural Science	benthic data (Sköld *et al.*, 2018), coastal monitoring programme	2009-2018
France	Normandie Université, UNICAEN	29 annual time series, including 21 time series of catches from the IFREMER database SACROIX and 8 time series of biomass from multiple stock assessment campaigns.	2000-2015
Germany	LKN-SH, Nationalpark authority	Trilateral Monitoring and Assessment Programme (TMAP, https://www.waddensea-worldheritage.org/trilateral-monitoring-and-assessment-programme-tmap), monitoring nationalpark authority and LLUR	2009-2019

OSPAR Region II

Bay of Seine (COMP area Coastal waters)
This food web model, the extended Bay of Seine model (or eBoS), focused on the area of the future offhore windfarms of Courseulles-sur-Mer (Normandy, France), that will be located in the lower middle part of the Bay of Seine. The bay opens onto the English Channel in its northern part, its depth is relatively shallow with a mean of 30 m. Habitats can be described as coarse sand and sandy gravels. The food web model was built up using Ecopath and is composed of 40 living groups including a wide range of 170 marine species – fish, invertebrates, birds and marine mammals, and 2 non-living groups, detritus and fishing discards. Multiple fishing techniques were modeled (trawling, nets, angling, traps, and other minor gears). A full description of the food web model is available in Halouani *et al.*, (2020). The eBoS Ecosim model was set to run from 2000

to 2015 and used 29 annual time-series, including 21 time-series of catches from the IFREMER database SACROIX (Système d'Information Halieutique, 2017) and 8 time series of biomass from multiple stock assessment campaigns. A primary production forcing was also used to force primary production in the model. This function was built using a satellite SeaWifs annual map of chlorophyll a between 2000 and 2010 (SeaWifs level3, 9 km resolution), and a satellite MODIS monthly map of chlorophyll a between 2011 and 2015 (MODIS aqua, 4 km resolution). Additional information on used time series can be found in (Halouani *et al.*, 2020).

Coastal North Sea (COMP area Elbe Plume)

This food web model focused on the coastal area of the German North Sea (COMP area Elbe Plume). Habitats represents the MSFD broad scale habitat "Circalittoral sand". Sediments can be described as fine sands. Depth is relatively shallow with a mean auf 14 m (ranges 9 to 19 m water depth).

The food web model was built up using an own R script (Schückel & Martinez, in prep.) and the R package (enaR: Ecological Network Analysis with R). The food web models composed of 235 marine species, and 3 non-living groups (detritus pools). Marine species include benthic invertebrates (207), pelagic and demersal fish species (19), birds (2) and marine mammals (3). Marine species were grouped into 11 trophic guilds in accordance with the MSFD Art. 8 Guidance (2022). Food web models were run from 2009 to 2019. Standing stock data were provided by the monitoring programmes of the national park authority in Schleswig-Holstein, the trilateral monitoring assessment programme (TMAP), the Landesamt für Landwirtschaft, Umwelt und ländliche Räume (LLUR), ICES DATRAS and published data.

Kattegat (COMP areas Kattegat coastal and deep)

The Swedish benthic data originates from a research project on effects of trawling on macrofauna in the Kattegat (see Sköld *et al.*, 2018 for a closer description). Some of the data used are collected within a coastal monitoring programme run by the County Administration of Halland. The Danish benthic data originates from the monitoring program NOVANA. Fish data were compiled from ICES DATRAS database. The plankton data is collected and processed according to the so-called HELCOM Monitoring Manual, a basin-wide (Baltic Sea) collaboration that standardises methods for monitoring various aspects of marine life (Chlorophyll https://www.helcom.fi/wp-content/uploads/2019/08/Guidelines-for-measuring-chlorophyll-a.pdf; Primary production https://helcom.fi/media/publications/Manual-for-Marine-Monitoring-in-the-COMBINE-Programme-of-HELCOM.pdf; Phytoplankton https://helcom.fi/wp-content/uploads/2020/01/HELCOM-Guidelines-for-monitoring-of-phytoplankton-species-composition-abundance-and-biomass.pdf; Zooplankton manual https://www.helcom.fi/wp-content/uploads/2019/08/Guidelines-for-monitoring-of-mesozooplankton.pdf). In total, 255 species/functional groups were grouped to 10 trophic guilds according to the EU Guidance 2022, and 1 non-living compartment (detritus pool). Primary producers were represented by phytoplankton, while zooplankton comprises one functional group. No data were available for top-predators (mammals, birds). Annual biomass data in Carbon per species or functional groups were used for the ENA models. In total, 126 models were calculated covering a period from 2009-2018. Since the available data provided occurred on trawled areas and areas closed for fisheries, different models were calculated for trawled and closed areas over time.

OSPAR Region V

The Azores ecosystem model describes the food web of the open-ocean and deep-sea ecosystem of the Azores in 1997, giving special highlight to the deep-sea environment. This baseline year was chosen for reference since most of the input data (diet and growth parameters) originated from that year. Temporal dynamics of the Azores ecosystem have been assessed between 1997 and 2018, using Ecosim - the time dynamic module of Ecopath.

Model domain is confined to the boundary of the Azores Exclusive Economic Zone (EEZ, 954 563 km^2), thus covering multiple and interconnected macrohabitats – deep-sea, open-ocean, some seamounts, parts of the Mid Atlantic Ridge and island slopes.

Pilot Assessment of Ecological Network Analysis Indices

The updated version of the Azores model (based on Morato *et al.*, 2016) was built upon previous models, developed for this region and associated seamounts (Guénette and Morato, 2001; Morato *et al.*, 2009), with the main goal of addressing deep-sea fisheries and ecosystem related research questions. This last version of the model emphasises intermediate and deep-water species present in the Azores ecosystem. Nevertheless, the model does also include pelagic, mesopelagic, bathypelagic, coastal, shallow and intermediate water species, and captures the full range of trophic guilds that characterize the food-web. The model is thus appropriate to provide ecological information about the structure, functioning and dynamics of the Azores and deep-sea food webs.

In total, 223 fish species were included in the model, representing 38% of the known fish biodiversity) (WoRMS, 2016). Further, they were allocated into 29 fish groups, based on diet composition, asymptotic length and average habitat depth for each species. Non-fish functional groups were defined based on a previous Ecopath model of the Azores (Guenette and Morato, 2001) and an Ecopath model for a hypothetical seamount in the North Atlantic (Morato *et al.*, 2009). The Azores model thus consisted of 45 functional groups: one detritus group, two primary producer groups, eight invertebrate groups, 29 fish groups, three marine mammal groups, one sea-turtle and one seabird group. The Azores fleet is dominated by a small-scale artisanal fishing fleet (Carvalho *et al.*, 2011) with only 10–20% of the fleet being a large-scale, semi-industrial fishing fleet. From an overview of the main fisheries in the Azores, their gear types, target species, fishing vessels and regulations (Gaspar 2011), eleven fishing fleets were considered in the model.

Total marine catch data of the year 1997 was obtained from Pham *et al.*, 2013 and Fauconnet *et al.*, 2019. The dataset included both official fishery statistics and estimated illegal, unreported, and unregulated catch (IUU) within the Azores EEZ. Discards (fish returned to the sea), were included in total catches. Species catch data was assigned to the different fishing fleets and the functional groups. Catch data that could not be assigned to a specific functional group or fleet (e.g., unidentified marine species) were redistributed into the groups exploited by the various fleets. Catch data was constructed for the reference year 1997 and then expressed in tonnes of wet weight per square kilometre of the model area.

Historical temporal dynamics of the Azores ecosystem have been assessed between 1997 and 2018, using Ecosim - the time dynamic module of Ecopath. The effects of fishing pressure on different ecosystem components, were simulated through time-series of fishing effort. A multi-model or ensemble approach to describe the temporal dynamics of the Azores ecosystem was used. We built a "biomass-model" that aims at describing the variability of biomass oscillations over time; a "catch-model" that was fitted to explain variability in the catch and a "catch-biomass model" that gives equal emphasis to the biomass and catch time-series. Since we relied on an ensemble of models to represent temporal dynamics, the information provided by model-derived ecological indicators, must be based on converge on common model predictions. The results presented in this report have been documented and described in the H2020 MISTIC SEAS III project (GA No. 110661/2018/794676/SUB/ENV.C2) deliverable 2.3 and 2.4.

Results

This pilot assessment identified temporal changes in food web structure and functioning derived by Ecological Network Analysis indices at regional and local scales in OSPAR Regions II and V. Consequences of changes in food web structure and functioning due to different pressures were successfully evaluated with Ecological Network Analysis Indices. Besides Ecological Network Analysis indices temporal trends in primary production and biomass of different trophic guilds were showed to better analyse the evolution of the ENA indices. To analyse the trend of the ENA indices, we used the SiZer R package, and piecewise linear and bent-cable models. Regarding OSPAR Region II, the coastal food webs (COMP areas Coastal waters (Bay of Seine) and Elbe Plume) several trophic guilds were very much linked to changes in primary production and phytoplankton biomass over time. However, different parts of the food web showed changes over time. Changes in the pelagic food web structure occurred in the Bay of Seine while the benthic trophic guilds showed changes in the Elbe Plume over time. Regarding ENA indices, the detritivory over herbivory ratio and the Finn Cycling index showed similar trends to the dynamics of primary production. In the Kattegat food web (COMP area Kattegat coastal and deep) most of the ENA indices showed no significant trend probably due to the short time-series. However, ENA indices indicate different trends in food web structure and

functioning among closed versus trawled areas. Structure and organization of the food web in the closed area, mirrored by the Relative Ascendency Index, increased from 2009 until 2014 but decreased afterwards. Food web structure and functioning of the Azores food web in OSPAR Region V is characterised by two main energy pathways (demersal pathway, pelagic pathway) reaching the top of the food-web. The dynamic of the Azores deep-sea ecosystem is highly influenced by bottom-up processes that act at the primary producer's level and control biological characteristics of the system. Fishing does not fully explain the variability observed in the indices of relative abundance of deep-sea fish.

Ecological Network Analysis indicator results suggest no significant correlations between the great majority of temporal ecological indicators and time. The only exception was found for the relative overhead indicator, which significantly decreasing over time suggesting the resilience of the ecosystem is decreasing.

Results (extended)

OSPAR Region II:
Bay of Seine (COMP area Coastal waters)
In the eBoS, the most dominant cluster of trophic groups include the benthic invertebrates (which include the meiofauna, the suprabenthos, invertebrate surface and subsurface deposit feeders, invertebrate filter feeders and invertebrate predators), followed by zooplankton (micro, meso and macrozooplankton) and phytoplankton. Other important clusters include zooplanktivorous fish (sprats, pilchards, horse mackerels and mackerels), demersal fish (poutings, seabass, poor cods, sharks and rays), flat fish (flounders, soles, sea breams and limands) and piscivorous fish (whitings and atlantic cods). Finally, the least abundant groups include cephalopods (benthic and benthopelagic cephalopods), sea birds (gulls, northern gannets and many others) and marine mammals (dolphins, porpoises and seals; **Figure b**).

Several groups are very much linked to primary production and to the biomass of phytoplankton. Primary production is indeed important in 2002 and in 2015 (**Figure c**), time periods which also saw a strong increase in the biomass of certain groups like zooplankton, bacteria and zooplanktivorous fish but also in other groups like the benthic invertebrates in 2015 and fish groups based on the important increase in discards indicative of increased fishing mortality on fish.

Regarding ENA indices, the Finn Cycling Index (FCI), that quantifies the amount of cycling in the food web, displayed a similar dynamic to primary production, with a decrease from 2000 to 2014, and an increase in 2015 (**Figure d**). Overall, the FCI seems to decrease between 2000 and 2015 as seen on the first SiZer derivative (**Figure d**), even though the last increase in 2015 seems to indicate that the shape of the curve would tend to increase (second SiZer derivative). On the contrary, the relative redundancy of the trophic pathways has an opposite response to primary production and thus to the FCI, with an increase between 2000 and 2014, and a decrease in 2015 (**Figure e**). The trend of the indices is, like the FCI, changed in the last year of the model (2015), resulting in a downward evolving curve, as seen on the second SiZer derivative (**Figure e**). The Detritivory on Herbivory on the other hand evolved in a concave shape between 2000 and 2015, with an initial increase in 2000, until it reaches a peak in 2006/2007, to then decrease until 2015. This is clearly visible through both piecewise linear model and the bent cable model (**Figure f**). The symmetry in both SiZer derivative, and especially in the second derivative further support this and could even support the use of a three-threshold points model as threshold points can also be observed in 2001/2002 and in 2013/2014 with either an abrupt decrease in 2001 or an abrupt increase in 2015 (**Figure f**).
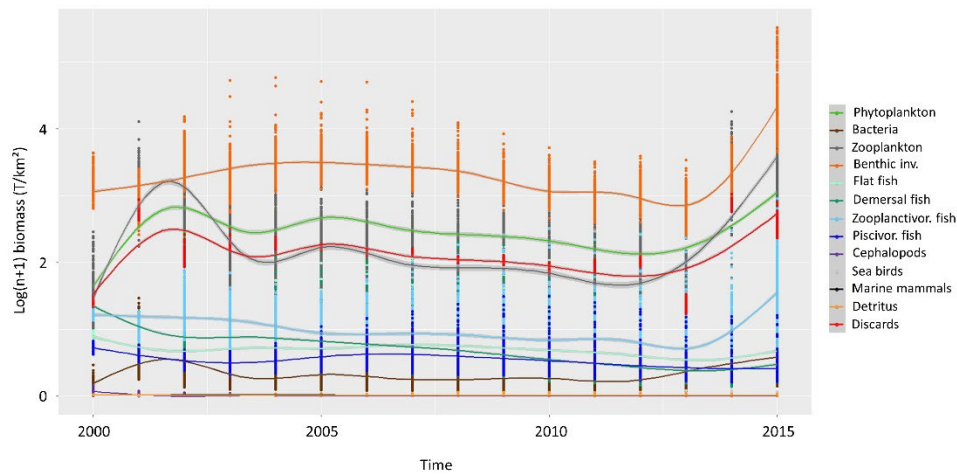
Figure b: Evolution of the biomass of 13 clusters of trophic groups between 2000 and 2015 (log(n+1) in T/km²). Clusters include phytoplankton, bacteria, zooplankton, benthic invertebrates, flat fish, demersal fish, zooplanktivorous fish, piscivorous fish, cephalopods, sea birds, marine mammals, detritus and discards.
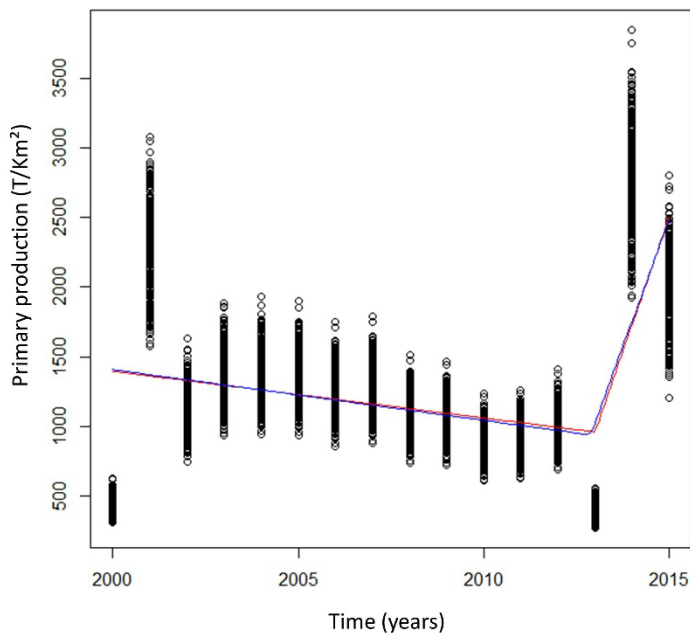
Figure c: Primary production in the eBoS models between 2000 and 2015. Primary production was fitted by two models: the piecewise linear (in red, threshold: 2013) and bent-cable (in blue, threshold: 2012.9).
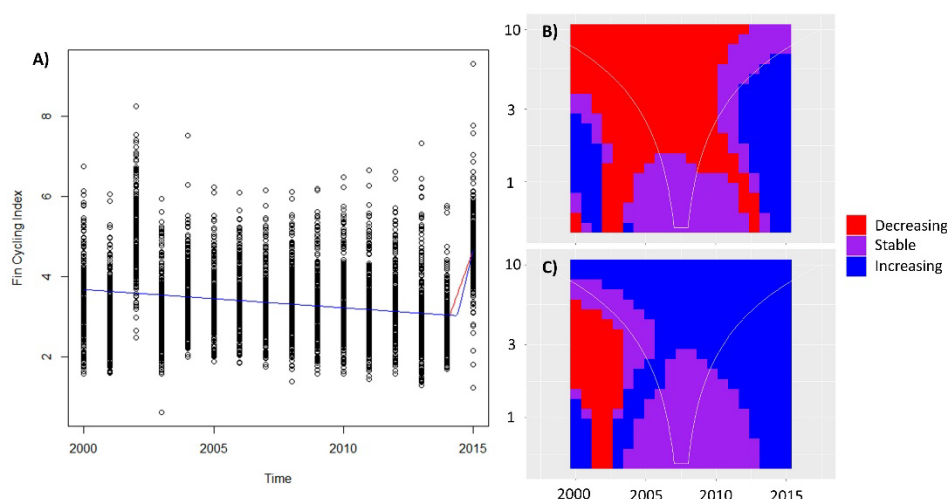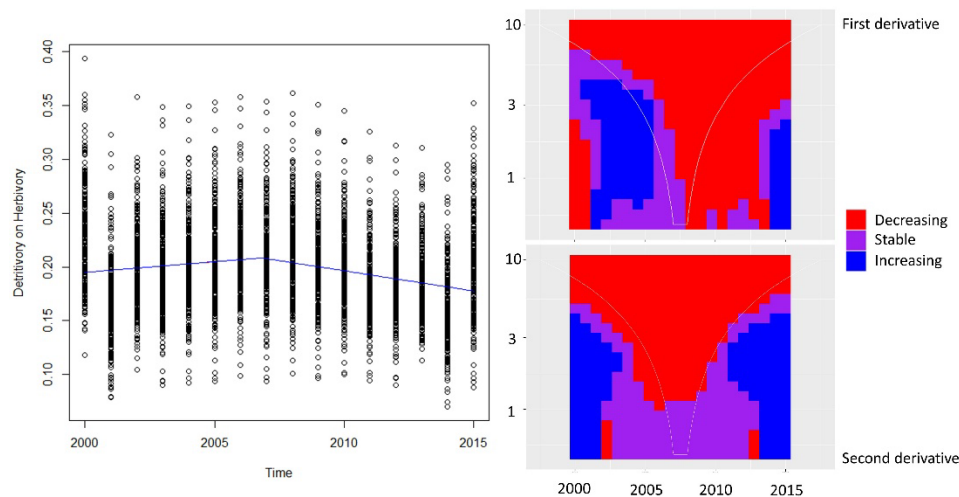
Figure d: Evolution of the Finn Cycling Index (FCI) between 2000 and 2015 in the eBoS models. A) Time variation of the FCI, it was fitted by two models: the piecewise linear (in red, threshold: 2014.1) and bent-cable (in blue, threshold: 2014.4). B) First derivative SiZer map. C) Second derivative SiZer map. SiZer maps categorize the derivative as positive (blue), negative (red), or possibly zero (purple).
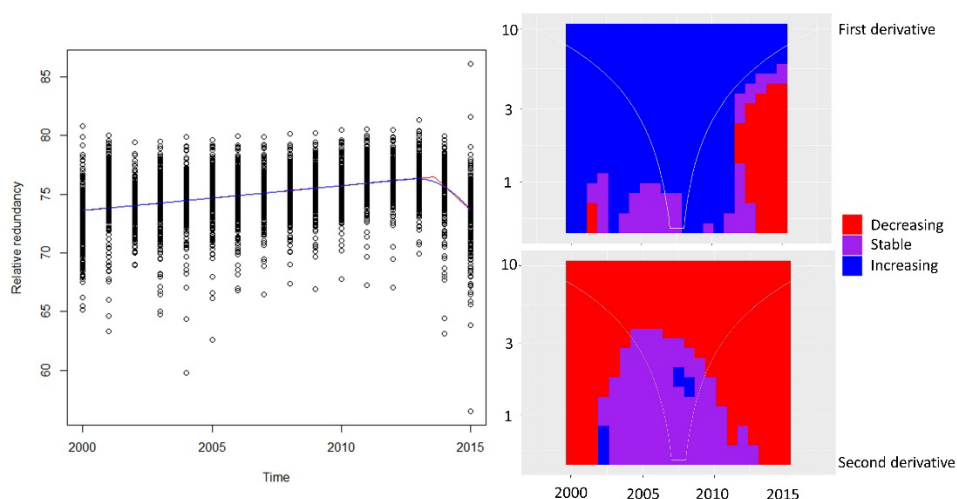
Figure e: Evolution of the relative redundancy between 2000 and 2015 in the eBoS models. A) Time variation of the redundancy, it was fitted by two models: the piecewise linear (in red, threshold: 2013.6) and bent-cable (in blue, threshold: 2015). B) First derivative SiZer map. C) Second derivative SiZer map. SiZer maps categorize the derivative as positive (blue), negative (red), or possibly zero (purple).

Figure f: Evolution of the Detritivory on Herbivory between 2000 and 2015 in the eBoS models. A) Time variation of the Detritivory on Herbivory, it was fitted by two models: the piecewise linear (in red, threshold: 2016.8) and bent-cable (in blue, threshold: 2006.8). B) First derivative SiZer map. C) Second derivative SiZer map. SiZer maps categorize the derivative as positive (blue), negative (red), or possibly zero (purple).

OSPAR Region II

**Coastal North Sea (COMP area Elbe Plume)**

In the coastal food web model, the most dominant trophic guilds (in terms of biomass in Carbon) include the benthic invertebrates (which include the meiofauna, invertebrate surface and subsurface deposit feeders, invertebrate filter feeders and invertebrate predators) (**Figure g**). Other important trophic guilds include bacteria and primary producers. Finally, the least abundant groups include marine birds (common scoter, eider duck) and marine mammals (porpoises, seals).
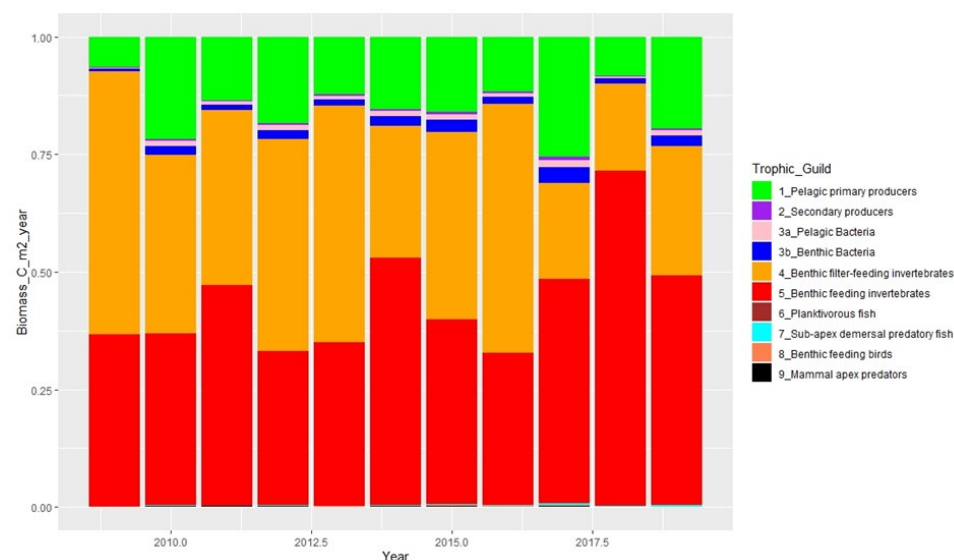
Figure g: Temporal variability of biomass of trophic guilds between 2009 and 2019.

Similar to the Bay of Seine food web, several groups are very much linked to primary production and to the biomass of phytoplankton (**Figure h**). Primary production decreased from 2009 until 2014 but increased since 2015 concomitant with an increase in biomass of benthic filter feeding invertebrates and benthic feeding birds (in 2016) that probably benefit from the increasing biomass of filter feeding invertebrates. However, an overall decrease in benthic filter feeding invertebrates was found over time.
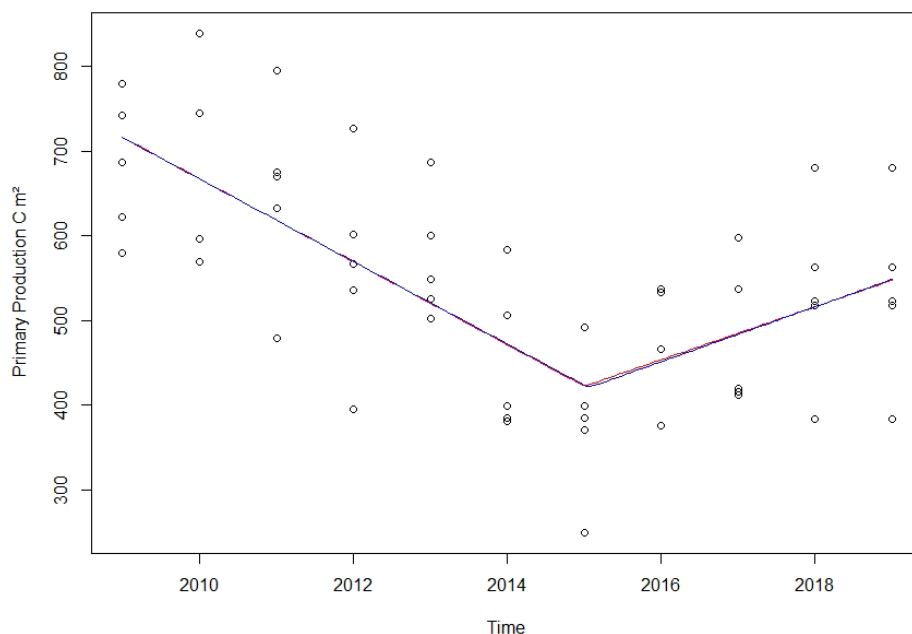
Figure h: Primary production between 2009 and 2019. Primary production was fitted by two models: the piecewise linear and bent-cable (both threshold: 2015).

Regarding ENA indices, the Detritivory to Herbivory ratio (**Figure i**) showed a sharp increase since 2015 similar to the dynamics of primary production. This is also visible through both piecewise linear model and the bent-cable model as well as the SiZer maps pointing to a threshold point in 2015.

The Finn Cycling Index (**Figure j**) showed a significant overall increase over time. High values indicate that the system is more independent from imports. The Flow Diversity (**Figure k**) is a measurement for the number of interactions and the evenness of energy flows. Comparable to the biodiversity index, a high value shows a highly diverse and well-developed system. The Flow Diversity increased from 2009 until 2015 but decreased sharply afterwards as supported by the threshold point in 2015. Since all models have the same number of compartments, energy flows seem to be more unevenly distributed. No significant changes were found for the Relative Redundancy in the North Sea coastal food web.
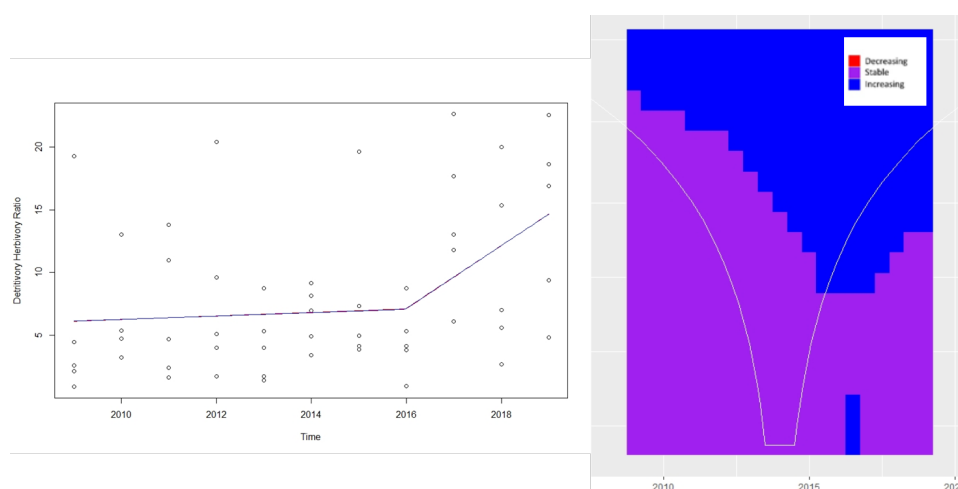
Figure i: Evolution of the Detritivory Herbivory ratio between 2009 and 2019 in the coastal North Sea food web model. Time variation of the FCI, it was fitted by two models: the piecewise linear (in red,

threshold: 2015) and bent-cable (in blue, threshold: 2015). SiZer maps were used for visualisation and to categorise the derivative as positive (blue), negative (red), or possibly zero (purple).
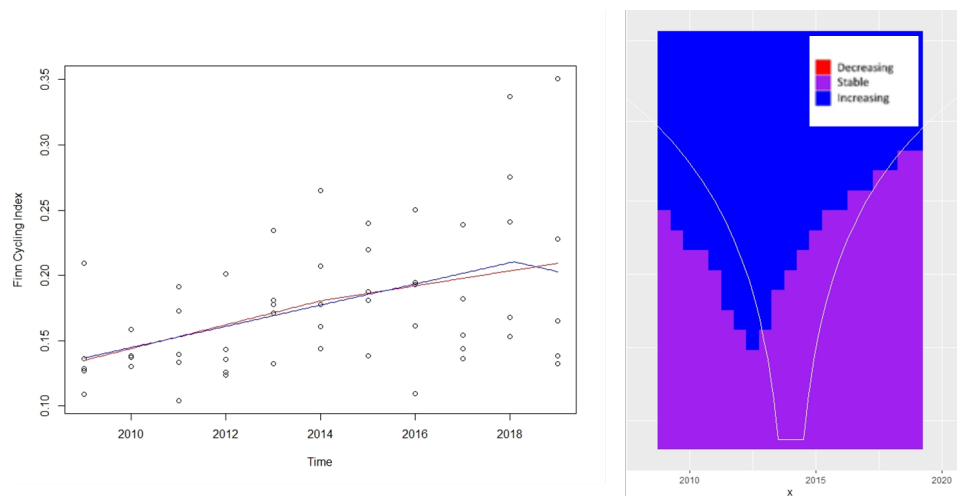

Figure j: Evolution of the Finn Cycling Index (FCI) between 2009 and 2019 in the coastal North Sea food web model. Time variation of the FCI, it was fitted by two models: the piecewise linear (in red) and bent-cable (in blue). SiZer maps were used for visualisation and to categorise the derivative as positive (blue), negative (red), or possibly zero (purple).
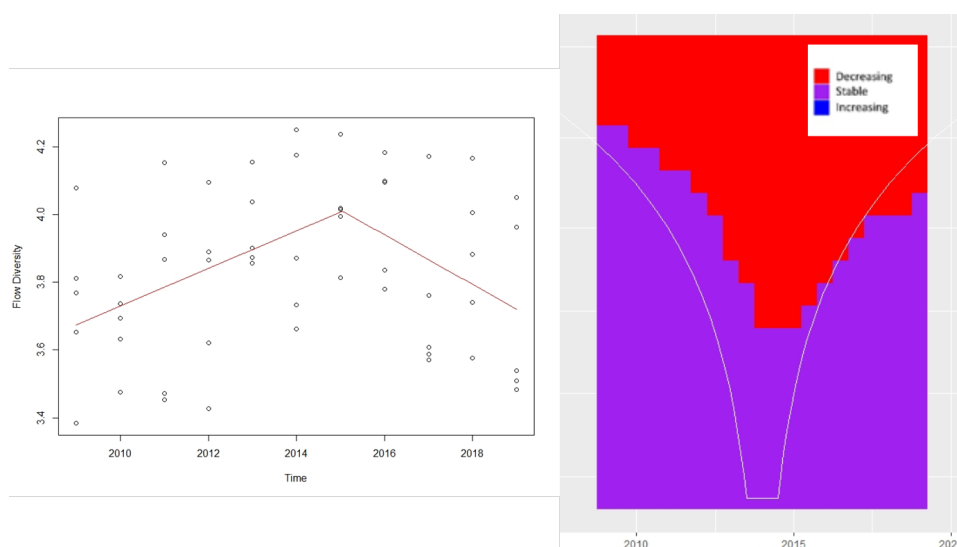

Figure k: Evolution of the Flow Diversity between 2009 and 2019 in the coastal North Sea food web model. Time variation of the FCI, it was fitted by two models: the piecewise linear (in red, threshold: 2015) and bent-cable (in blue, threshold: 2015). SiZer maps were used for visualisation and to categorise the derivative as positive (blue), negative (red), or possibly zero (purple).

Kattegat (COMP areas Kattegat coastal and deep)
In the Kattegat food web models, the most dominant trophic guilds (in terms of biomass in Carbon) include benthic invertebrates, in particular benthic filter and deposit feeding invertebrates (**Figure l**). Other important trophic guilds include bacteria, primary producers and planktivorous fish. A higher proportion of benthic filter feeding invertebrates was observed in the closed area compared to the trawled areas.
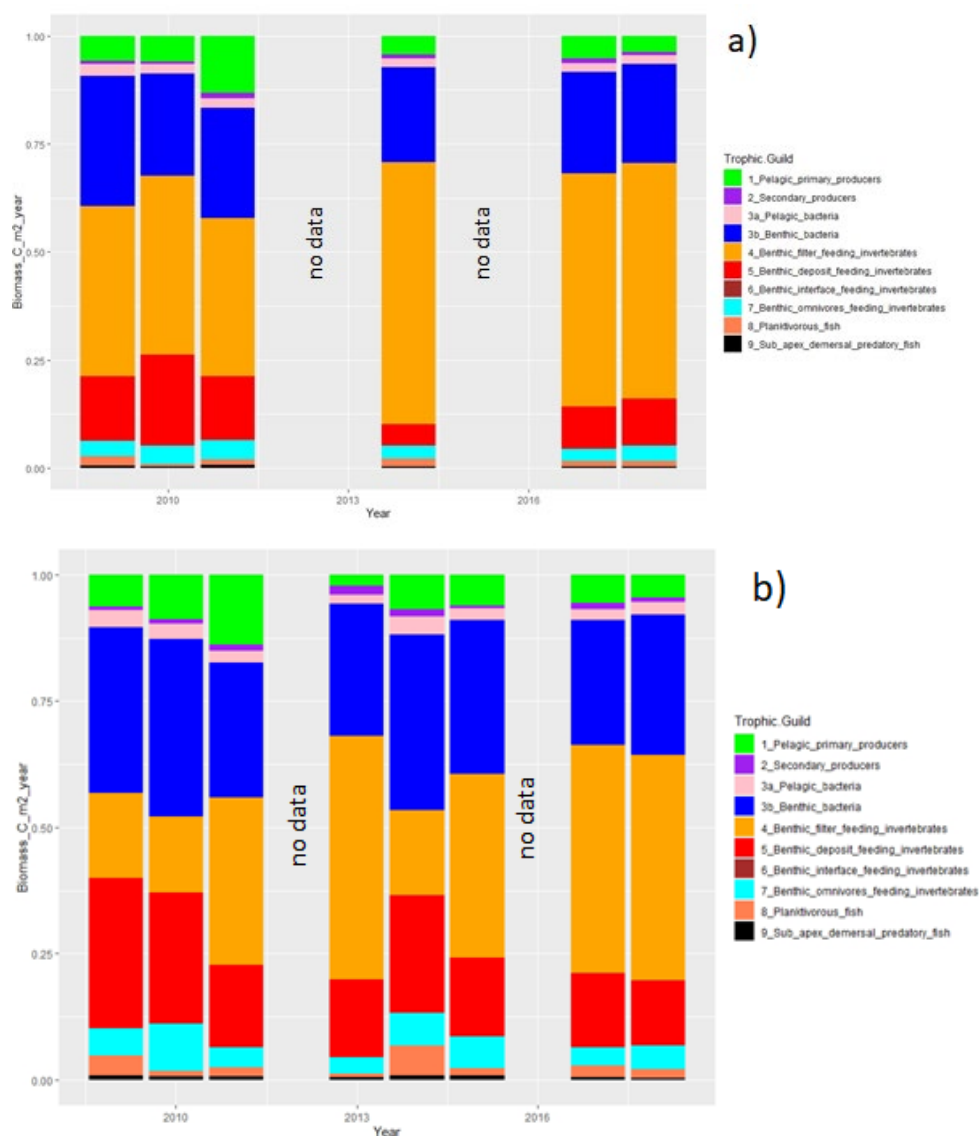
Figure l: Temporal variability of biomass (in Carbon) of trophic guilds between 2009 and 2018 for a) closed and b) trawled areas.

Most of the ENA indices showed no significant trends which is probably attributed to the short long-term series. The Detritivory to Herbivory ratio (**Figure m**) showed an increasing trend for the closed areas over time while no clear trend was obvious for the trawled areas.
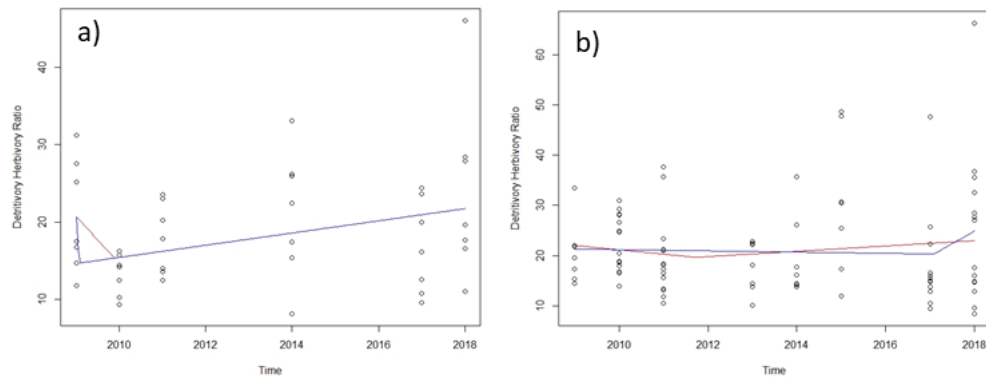
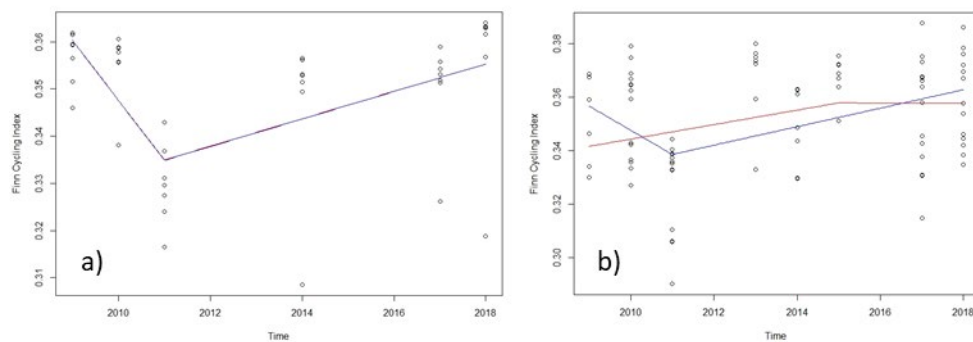**Figure m: Evolution of the Detritivory Herbivory Ratio between 2009 and 2018 in the Kattegat Sea food web models for a) closed and b) trawled areas. Time variation was fitted by two models: the piecewise linear (in red) and bent-cable (in blue).**

Figure n: Evolution of the Finn Cycling index between 2009 and 2018 in the Kattegat Sea food web models for a) closed and b) trawled areas. Time variation was fitted by two models: the piecewise linear (in red) and bent-cable (in blue).

The Finn Cycling Index (**Figure n**) showed in both areas a sharp decrease from 2009 to 2011 but increased since 2011. High values indicate that the system is more independent from imports. This trend follows the temporal variability of phytoplankton production which showed highest values in 2011.
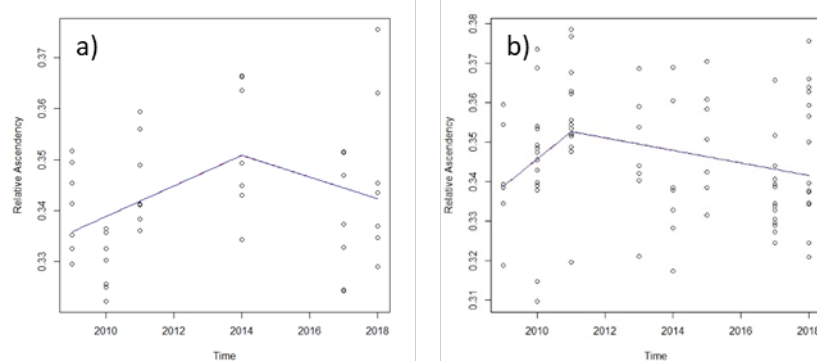

Figure o: Evolution of the Relative Ascendency index between 2009 and 2018 in the Kattegat Sea food web models for a) closed and b) trawled areas. Time variation was fitted by two models: the piecewise linear (in red) and bent-cable (in blue).

Ascendency represents both the size and organisation of the flows in a single measure. The increase in the Relative Ascendency from 2009 until 2011 in the trawled areas is probably attributed to the increase in phytoplankton production in 2011 (**Figure o**). Relative Ascendency increased from 2009 until 2014 in the closed areas following a decreasing trend from 2014 until 2018, which indicates changes in the structure of the food web.

OSPAR Region V:

Primary producers were estimated in Ecopath to form 15,4% of the total biomass while fish biomass contribute to 14,8%. The largest part of the total ecosystem estimated biomass is composed by the invertebrate and zooplankton groups, while the non-fish groups occupying the higher trophic levels (seabirds, dolphins, baleen whales and toothed whales) only contribute to 0,5% of the total estimated biomass.

The Azores Ecopath model included five trophic levels (TL) with toothed whales and deep-water sharks, representing the top predators in the ecosystem with TL of 4,63 and 4,52, respectively. Other functional groups with a TL > 4 included the large-size pelagic fish, large-size demersal and bathydemersal fish, dolphins, pelagic sharks, seabirds, tunas, rays and other sharks. The remaining fish groups have a TL ranging from 2,99 (small-size pelagic fish) to 3,90 (bathypelagic fish). Invertebrate functional groups were estimated to have a TL between 2,05 (benthic filter feeders) and 2,77 (shrimps), with the exception of cephalopods that were estimated to have a TL of 3,72. Zooplankton functional groups have a TL of 2,00 (small-size) and 2,58 (large-size and gelatinous). The food web and flow diagram demonstrated the complex structure of the ecosystem (see Morato *et al.*, 2016, **Figure d**).

A global decreasing trend of the biomass over trophic level was evident (Morato *et al.*, 2016).

An overview on the relative contribution of low and mid-trophic level groups that shape the biomass spectra on the diets of top-predators (TL > 4), suggests two distinct energy pathways reaching the top of the chain. The demersal pathway is dominated by small-sized demersal and bathydemersal fish and shrimps that constitute the main food source of deep demersal predators and deep-sea sharks. Cephalopods and forage fish that constitute the micronekton (small-sized pelagic fish and mesopelagic fish) seem to connect both pathways. Marine mammals, such as toothed whales, and large pelagic fish and sharks are the most reliant on the pelagic pathway that has few steps from primary producers to top-predators.

The groups playing important structuring roles in the food web form a remarkable pelagic energy pathway that involve two apex predators, toothed whales and pelagic sharks, and their main prey, cephalopods and small-size pelagic fish, respectively (Morato *et al.*, 2016). Bathypelagic fish also showed a high keystone value, suggesting an important role as a trophic linkage between pelagic and deep-sea niches. Zooplankton groups were identified as the dominant groups, considering their high overall effect in the ecosystem as forage fish consumers (e.g., small pelagic fish, meso and bathypelagic fish). Based on the mean trophic level of key functional groups and the overall contribution of top-down effects to the response process, it seems there is a balance between top-down or bottom-up mechanisms regulating the ecosystem (Morato *et al.*, 2016).

Ecological Network Analysis indices of food web's general community properties reflected moderate productivity of the system in relation to the total biomass and respiration and low biomass relative to the total flows of energy (Morato *et al.*, 2016). Thus, primary productivity is not fully utilised, leading important flows to end up as detritus. Most ratios involving system production and biomass (PP/R; PP/B; B/T) depict an immature state because of high PP and T in relation to low system respiration and biomass. In addition, underdeveloped ecosystem states are expected to minimise biomass cycling, as was found in our model where just 0,75% of T is cycled. When analysing the relation between Ascendency (A), Overheads (O) and Development Capacity (A/DC and O/DC ratios), we found an intermediate situation in terms of ecosystem development and resilience: the relative balance between A and O for the deep-sea would suggest that this system has a high level of development, is resilient, and has strength in reserve. System's A reached 53% of the DC, leaving an overhead of 47,26%.

Connectance Index (CI) showed that 21% of the possible links occurred in the food web. The System Omnivory Index (SOI= 0,22) reflected an intermediate degree of connectivity among groups. The values of

these indices are intermediate, suggesting a relatively simple food web (because there are a number of possible trophic links that were not realized), and consequently denote a system that is not fully mature. Model derived Ecological Network Analysis indices were estimated between 1997-2018 using the best fitted Ecosim models. Since we relied on an ensemble of models to represent temporal dynamics, the information provided by model-derived ecological indicators, must be based on convergence on common model predictions. The significance and correlation between our suite of ENA indices and time was assessed based on Spearman's rank correlation.

Ecosim derived results suggest no significant correlations between the great majority of temporal ecological indicators and time (**Figure p**). The only exception was found for the relative overhead indicator, which according to the biomass and catch-biomass models, it is significantly decreasing over time. This prediction is thus suggesting the resilience of the ecosystem is decreasing. However, the trajectory of the indicator predicted from the catch-model, suggests the indicator is increasing. From the inter-annual variability of the indicators, it is perceptible that the indicator´s trajectory is strongly influenced by the availability of primary production in the system. Despite the different trajectories, the information from the three fitted models converges in a specific point: the lowest values of the relative overhead indicator occurred under the most pronounced cumulative effect of high fishing pressure and lowest levels of primary production. Since the primary production anomaly was estimated by each model, the lowest values of the indicator were predicted to occur at different times. For example, according to the biomass and catch-biomass model they happened between 2008 and 2012. The primary production anomalies driving the dynamics in these two models have a similar shape. The primary production anomaly estimated by the catch-model was more constant over time, when compared to the function estimated by the other models, and predicted an increase of relative production in the end of the modelled period. This increase in primary production helps replicating the time-series of absolute catches. On the other hand, it explains the overall increasing trajectory of the indicator over time. It should be noted that flow-based indicators, estimated with the Azores models, are particularly sensitive to perturbations that grasp effects at the level of ecological variables as total production or biomass of the system. Conversely, they seem to be less sensitive to the effects of predator-release.
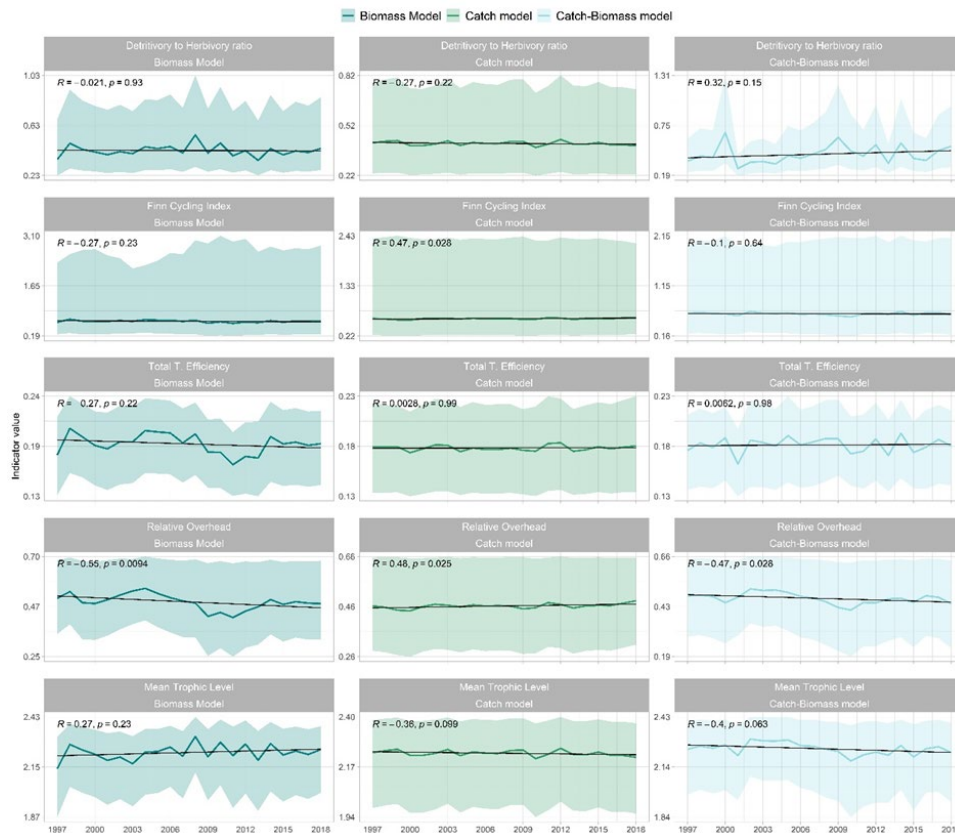
Figure p: Temporal variability of ENA indices, predicted by fitted Ecosim models. The black line indicates the linear relationship between indicators and time. R is statistical measure of the strength of a link or

relationship between indicator and time, while p is the value of critical probability. The shadow represents the 95% percentile and 5% percentile obtained through the Monte Carlo routine (100 preliminary interactions).

Conclusion

This pilot assessment illustrates the potential for the ENA indices indicator to show changes in the state of food webs at regional (COMP4 assessment areas) scale and provides key information on the temporal variability of food web structure and functioning. This pilot assessment demonstrates inter-annual variability within study sites and variability between them. The pilot assessments indicate that some ENA indices are strongly influenced by the availability of primary production in the food webs in OSPAR Regions II and V. However, most of the ENA indices showed no significant trend in OSPAR Region V. In contrast, significant trends of ENA indices focus on lower trophic levels such as Detritivory over Herbivory ratio or Finn Cycling Index were found in OSPAR Region II (Elbe Plume and Bay of Seine). However, results showed that the importance of trophic groups within food webs were different across COMP areas. Most of the ENA indices showed no significant trend in the Kattegat food web model probably attributed to the short time series.

Conclusion (extended)

This pilot assessment illustrates the potential for the ENA indices indicator to show changes in the food web structure and functioning at regional (COMP4 assessment areas) scale and provides information on the state of the food webs.

The pilot assessments indicate that ENA indices are influenced by the availability of primary production in the food webs in OSPAR Regions II and V. Recycling in the Bay of Seine model seems to be correlated to primary production. This is to be expected as primary production is the main energy source of trophic network, thus if primary production increases, recycling will increase due to the higher amount of energy in the system. This also means that the more productive a system is, the more certain it is that efficient trophic pathways will grow, reducing the redundancy of trophic pathways. Indeed, when primary production increases, zooplankton seem to grow rapidly, as well as planktivorous fish. This trophic pathway growth will thus reduce the overall redundancy of trophic pathways between phytoplankton and higher trophic groups in the Bay of Seine.

Changes in the Detritivory on Herbivory's index in OSPAR Region II seem to be the result of the shifting importance of the different trophic groups within the ecosystem. Groups playing a key role in herbivory and detritivory are zooplankton (secondary producers) and benthic filter feeding invertrebrate for herbivory, and benthic feeding invertebrates for detritivory. Increased in zooplankton biomass leading to a low values of the detritivory ratio, while increasing biomass of benthic feeding invertebrates increases the detritivory flow.

In OSPAR Region V, results suggest no significant correlations between most of the ENA indices over time. The only exception was found for the relative overhead indicator, which is significantly decreasing over time. This prediction is thus suggesting the resilience of the ecosystem is decreasing.

Knowledge Gaps

Further development of this indicator is needed: a) increase the spatial extent and including existing food web and ecosystem models, b) how to deal with ecosystem components that were not regularly monitored but important food resources in the food web (e.g., bacteria, meiofauna), c) harmonisation of functional groups among food webs, d) appropriate thresholds and assessment values, e) pressure-state relationships, f) inclusion of additional datasets and improvement of data calls and g) comparison between relevant food web (including other model based indicators, e.g. based on EwE (Ecopath with Ecosim), e.g. Piroddi *et al*., 2021) and other biodiversity indicators with FW9.

Knowledge Gaps (extended)

Further development and improvement of this indicator is needed, particularly on the following points. Biomass data on organisms < 1 mm (e.g., meiofauna, bacteria), essential food resources for benthic species and smaller demersal fish species, are mostly not available. In addition, good information on feeding interactions is strongly needed (similar to FW4 and FW7 food web indicators). Information from literature is used which introduces unknown uncertainty in the model. Therefore replicate models or an uncertainty analysis are necessary to assess the spatial or temporal variability and to test the statistical significance of the difference between two or more values for the same index.

We would also like to acknowledge the many food web and ecosystem models that have already been published. To increase the spatial extent and to include the remaining OSPAR Regions already existing ecosystem models will be included in future. Examples are the North Sea ecosystem model published by Mackison & Daskalov (2007) or the Bay of Biscay model published by (Chaalali *et al.*, 2016). Since several of these ecosystem models have not calculated ENA indices or topology of the food webs were not in accordance with the EU Guidance in terms of which trophic guilds are covered and the spatial scale, they were not applied to the assessment of FW9.

Further work is required to evaluate appropriate baselines and threshold values for this indicator. To improve the assessments, further work is required to investigate the influence of various (cumulative) anthropogenic stressors using pressure-state relationships that help to further define thresholds and assessments values by looking for commonalities. To do so, two major steps would need to be investigated in future work to cope with this knowledge gap. The various pressures effects on ENA indices need to be studied (1) separately, and (2) under multi- pressure conditions (the cumulative effects). A first approach was published for the Bay of Seine (Nogues *et al.*, 2020). To improve the assessment of the state of the food web, further work on the ENA indices indicator could include the inclusion of additional datasets to improve the confidence of indicator's result especially for remaining OSPAR Regions and additional COMP areas.

More discussion is needed to harmonise the topology (trophic guilds/functional groups) of food webs among OSPAR areas. During this pilot assessment it becomes clear that trophic guild definitions provided by ICES advice are not useful since several ecosystem components were grouped (e.g., fish and invertebrates) or missing (birds). More sub-guilds are relevant and need to be considered. This is also relevant in context of a comparison between relevant food web indicators and other biodiversity indicators (pelagic, benthic, fish, birds, mammals) which are assessed separately. This also includes consistency in temporal resolution across indicators. For example, similar primary production data were used for the FW2 and FW9 assessment. However, temporal observations were different (FW2 monthly resolution, FW9 yearly resolution) which can lead to different conclusions. For the next QSR it should also be discussed with the OSPAR Secretariat how to improve the data calls. Since several data of the assessed food webs were also used in other biodiversity indicator assessments joint data calls should be developed in future.

References

Baird, D. and R. E. Ulanowicz (1993). Comparative study on the trophic structure, cycling and ecosystem properties of four tidal estuaries. Marine Ecology Progress Series 99:221–237.

Baird, D., H. Asmus and R. Asmus (2012). Effect of invasive species on the structure and function of the Sylt-Rømø Bight ecosystem, northern Wadden Sea, over three time periods. Marine Ecology Progress Series 462: 43–161.

Borrett, S.R., Sheble, L., Moody, J., Anway, E.C. (2018) Bibliometric review of ecological network analysis: 2010-2016. Ecol. Model. 382: 62-82

Carvalho, N., Edwards-Jones, G., and Isidro, E. (2011). Defining scale in fisheries: small-size versus large-scale fishing operations in the Azores. Fish. Res. 109, 360–369. doi: 10.1016/j.fishres.2011.03.006

Chaalali, A., Beugrand, G., Raybaud, V., *et al.* (2016) Froms pecies distributions to ecosystem structure and function: A methodologica perspective. Ecological Modelling 334:78-90

Chrystal, R. A. and U. M. Scharler (2014). Network analysis indices reflect extreme hydrodynamic conditions in a shallow estuarine lake (Lake St Lucia), South Africa. Ecological Indicators 38:130–140.

de Jonge, V.N., Pinto, R. Turner, R.K. (2012) Integrating ecological, economic and social aspects to generate useful management information under the EU Directive´s Ecosystem Approch. Ocean and Coastal Management 68: 169-188

de Jonge, V.N, Schückel, U. (2019) Exploring effects of dredging and organic waste on the functioning and the quantitative biomass structure of the Ems estuary food web by applaying Input Method Balancing in Ecological Network Analysis. Ocean and Coastal Management 174: 38-55

de Jonge, V.N., Schückel, U., 2021. A comprehensible short list of ecological network analysis indices to boost real ecosystem-based management and policy making. Ocean & Coastal Management 208, 105582.

Fath, B. D., U. M. Scharler, R. E. Ulanowicz and B. Hannon (2007). Ecological network analysis: network construction. Ecological Modelling 208:49–55.

Fath F, Asmus H, Asmus R, Baird D, Borrett S, de Jonge VN, Ludovisi A, Niquil N, Scharler U, Schückel U, Wolff M (2019) Ecological Network Analysis Metrics: The need for an entire ecosystem approach in management and policy. Ocean, Coastal and Management

Fauconnet, L., Pham, C.K., Canha, A., Afonso, P., Diogo, H., Machete, M.,Silva,H.M.,Vandeperre, F. and Morato, T., 2019. An overview of fisheries discards in the Azores. Fisheries research, 209, pp.230-241.

Finn, J.T., 1976. Measures of ecosystem structure and function derived from analysis of flows. J. Theor. Biol. 56, 363–380.

Finn, J.T., 1980. Flow analysis of models of the Hubbard Brook ecosystem. Ecology 61, 562–571.

Gaspar, N. D. C. (2011). Sea to Shore: An Economic Evaluation of the Azorean Commercial Fisheries. PhD thesis, University of the Azores.

Guénette, S. and Morato, T., 2001. The Azores Archipelago, 1997. Fish. Cent. Res. Rep., 9(4), pp.241-270.

Halouani, G., Villanueva, C.-M., Raoux, A., Dauvin, J., Lasram, F., Foucher, E., Le Loc'h, F., *et al.* 2020. A spatial food web model to investigate potential spillover effects of a fishery closure in an offshore wind farm. Journal of Marine Systems, 212: 103434

Heymans, J.J., Shannon, L.J., Jarre, A. (2004) Changes in the northern Benguela ecosystem over three decades: 1970s, 1980s, and 1990s. Ecological Modelling 172: 175-195

Heymans, J. J., S. Guénette and V. Christensen (2007). Evaluating network analysis indicators of ecosystem status in the Gulf of Alaska. Ecosystems 10: 488–502.

Heymans, J.J., Tomczak, M.T., 2016. Regime shifts in the northern Benguela ecosystem: challenges for management, in: Ecopath 30 years - modelling ecosystem dynamics: beyond boundaries with Ewe. Ecol. Model. 331, 151–159.

Pilot Assessment of Ecological Network Analysis Indices

Jung, A.S., van der Veer, H.W., Philippart, J.M., Waser, A.M., Ens, B.J., de Jonge, V.N,, Schückel, U. (2020) Impacts of macrozoobenthic invasions on a temperate coastal food web. Marine Ecology Progress Series 653: 19-39

Kay, J.J., Graham, L.A., Ulanowicz, RE. (1989) A detailed guide to network analysis. In: Wulf, F. (Ed.), Network Analyis in Marine Ecology. Springer-Verlag Berlin Heidelberg 1989

Mackinson, S. and Daskalov, G., 2007. An ecosystem model of the North Sea to support an ecosystem approach to fisheries management: description and parameterisation. Sci. Ser. Tech Rep., Cefas Lowestoft, 142: 196pp

Margalef, R. (1968) Perspectives in Ecological Theory. University of Chicago Press, Chicago, 111 pp.

Morato, T., Bulman, C., and Pitcher, T.J. (2009). Modelled effects of primary and secondary production enhancement by seamounts on local fish stocks. Deep-Sea Research Part II, 56(25), 2713-2719. Elsevier. doi:10.1016/j.dsr2.2008.12.029.

Morato, T., Lemey, E., Menezes, G., Pham, C.K., Brito, J., Soszynski, A., Pitcher, T.J:, Heymans, J.J. (2016) Food-web and ecosystem structure of the open-ocean and depp-sea environments of the Azores, NE Atlantic. Front. Mar Sci 3:245

Niquil, N., E. Chaumillon, G. A. Johnson, X. Bertin *et al.* (2012). The effect of physical drivers on ecosystem indices derived from ecological network analysis: Comparison across estuarine ecosystems. Estuarine, Coastal and Shelf Science 108: 132–143.

Nogues Q, Raoux A, Araignous E, Chaalali A, Hattab T, Leroy B, Ben Rais Lasram F, David V, Le Loc'h F, Dauvin JC, Niquil N (2020). Cumulative effects of marine renewable energy and climate change on ecosystem properties: Sensitivity of ecological network analysis. Ecological Indicators, 121, 107128.

Odum, E.P. (1969) The strategy of ecosystem development. Sciende 164, 262-270

Pham, C. K., Canha, A., Diogo, H., Pereira, J. G., Prieto, R., and Morato, T. (2013). Total marine fishery catch for the Azores (1950-2010). ICES Journal of Marine Science,70 (3), 564-577. doi:10.1093/icesjms/fst024.

Piroddi, C., Akoglu, E., Andonegi, E., Bentley, J., Celic, I., Coll, M. *et al.* (2021) Effects of nutrient management scenarios on marine food webs: A Pan-European Assessment in Support of the Marine Strategy Framework Directive, Frontiers in M arine Science 8, doi: 10.3389/fmars.2021.596797

Preciado, I., Arroyo, N.L., González-Irusta, J.M., López-López, L., Punzón, A., Muñoz, I., Serrano, A., 2019. Small-scale spatial variations of trawling impact on food web structure. Ecological Indicators 98, 442-452.

Raoux, A., Tecchio, S., Pezy, J.-P., Lassalle, G., Degrear, S., Wilhelmsson, D., Cachera, M., Ernande, B., le Guen, C., Haraldsson, M., Grangeré, K., le Loc´h, F., Dauwin, J.-C., Niquil, N. (2017) Benthic and fish aggregation inside an offshore wind farm: Which effects on the trophic web functioning ? Ecological Indicators 72: 33-46

Raoux, A., Lassalle, G., Pezy, J.-P., Tecchio, S., Safi, G., Ernande, B., Mazé, C., le Loc´h, F., Lequesnw, J., Girardin, V., Dauvin, J.-C., Niquil, N. (2019) Measuring sensitivity of two OSPAR indicators for a coastal food web model under offshore wind farm construction. Ecological Indicators 96: 728-738

Safi, G., Giebels, D., Arroyo, L., Heymans, J., Preciado, I., Raoux, A., Schückel, U., Tecchio, S., de Jonge V.N., Niquil, N. (2019). Vitamine ENA: A framework for the development of ecosystem-based indicators for decision-makers. Ocean, Coastal and Management

Scharler, U. M. and D. Baird (2005). A comparison of selected ecosystem attributes of three South African estuaries with different freshwater inflow regimes, using network analysis. Journal of Marine Systems 56: 283–308.

Schückel, U., I. Kröncke and D. Baird (2015). Linking long-term changes in trophic structure and function of an intertidal macrobenthic system to eutrophication and climate change using ecological network analysis. Marine Ecology Progress Series 536: 25–38.

Schückel, U., de Jonge, V., Giebels, D., Horn, S., Niquil, N., Asmus, H., Asmus, R., Eulaers, E., Safi, G., Scharler, U. (2018) Use of coastal and estuarine food web models in policy making and management: the need for an entire approach, 26 pp

Sköld, M., Göransson, P., Jonsson, P., Bastardie, F., Blomquist, M., Agrenius, S., Hiddink, J.G., Nilsson, H.C., Bartolino, V. (2018) Effects of chronic bottwon trawling on soft-seafloor macrofauna in the Kattegat. Marine Ecology Progress Series 586: 41-55

Tecchio, S., A. T. Rius, J.-C. Dauvin, J. Lobry *et al*. (2015). The mosaic of habitats of the Seine estuary: Insights from food-web modelling and network analysis. Ecological modelling 312: 91–101.

Tecchio, S., Chaalali, A., Raoux, A., Tous Rius, A., Lequesne, J., Girardin, V., Lassalle, G., Cachera, M., Riou, P., Lobry, J., Dauvin, J-C., Niquil, N. (2016) Evaluating ecosystem-level anthropogenic impacts in a stressed transitional environment: The case of the Seine estuary. Ecological Indicators 61:833-845.

Ulanowicz, R.E. (1997) Ecology: the Ascendant Persepctive. Columbia University Press, New York, pp 201

Vega, C. de la, S. Horn, D. Baird, D. E. Hines *et al*. (2018a). Seasonal dynamics and functioning of the Sylt-Rømø Bight, northern Wadden Sea. Estuarine, Coastal and Shelf Science 203:100–118.

Vega, C. de la, U. Schückel, S. Horn, I. Kröncke, R. Asmus and H. Asmus (2018b). How to include ecological network analysis results in management? A case study of three tidal basins of the Wadden Sea, south-eastern North Sea. Ocean and Coastal Management 163: 401–416.

Vezina, A.F., Platt, T.C., 1988. Food web dynamics in the ocean. I. Best estimates of flow networks using inverse methods. Mar. Ecol. Prog. Ser. 42, 269–287.

Wolff, M., Stäbler, M., Kluger, L. C., Schückel, U. 2019 Network models for (socio-) ecological system assessment and management – comparing tropical and temperate contexts. Workshop report, ZMT, 2019. 26 pp.

Assessment Metadata

Field	Data Type	
Assessment type	List	Pilot Assessment
SDG Indicator	List	14.2 By 2020, sustainably manage and protect marine and coastal ecosystems to avoid significant adverse impacts, including by strengthening their resilience, and take action for their restoration in order to achieve healthy and productive oceans
Thematic Activity	List	Biological Diversity and Ecosystems
Date of publication	Date	2023-06-30
Conditions applying to access and use	URL	https://oap.ospar.org/en/data-policy/

OSPAR's Quality Status Report 2023

OSPAR Secretariat
The Aspect
12 Finsbury Square
London
EC2A 1AS
United Kingdom

t: +44 (0)20 7430 5200
f: +44 (0)20 7242 3737
e: secretariat@ospar.org
www.ospar.org

Our vision is a clean, healthy and biologically diverse North-East Atlantic Ocean, which is productive, used sustainably and resilient to climate change and ocean acidification.