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A trophic modelling framework: Key metrics for the ecological assessment of artificial structures

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Abstract :

As the global population expands, marine coastal ecosystems face mounting pressures from human activities, that have led to habitat deterioration and dwindling fishery resources. In this context, Artificial Reefs (ARs) have emerged as one of the promising solutions. They are generally implemented to provide habitat, to create a protective, physical boundary, to support sustainable fisheries and to facilitate ecosystem rehabilitation. Evaluating their ecological performance is crucial to ensuring they meet their objectives. Initially, assessment relied on comparing ARs to natural reefs using mainly ecological metrics which focused on fish assemblage and dynamics. Despite there being more research and documentation on effectiveness today, assessing ARs remains challenging due to the number of environmental factors that can affect the ecological systems. Moreover, ecological studies mainly used metrics that investigated the reef fish populations or ecological metrics such as fish assemblages or trophic structure that are often overlooked in studies that primarily focus on commercial fishery dynamics. Therefore, new ways of assessing artificial reef performance and the set-up of comprehensive metrics which integrate this level of complexity are needed. In this study, we focused on the "Rade de Cherbourg" in the English Channel, employing a trophic modeling approach using Ecopath with Ecosim (EwE). The study emphasizes the importance of Ecological Network Analysis (ENA) metrics for evaluating changes in the systems' properties—such as complexity, flow diversity, and recycling capacity— which result from AR implementation. Furthermore, we identified which metrics are suitable for assessing specific AR objectives. The proposed metrics serve as a command-and-control tool for AR site managers, enabling them to evaluate the performance of each AR objective effectively. With the anticipated increase in AR projects, especially those which compensate for human impact like the Cherbourg ARs, this research offers valuable insights and future perspectives to continuously improve the ecological performance of ARs.

Keywords : Ecosystem models ; Ecological Network Analysis ; Artificial Reefs ; Ecopath with Ecosim ; English Channel

1 Introduction

As the world population grows, coastal areas' attractivity and development will increase the pressures on marine coastal ecosystems. But our dependence on the services they provide will not abate (Selig et al., 2019). In fact, coastal artificialization, overfishing, pollution and other human activities have led to marine habitat deterioration and to the decline in fishery resources (Worm, 2016; IPBES, 2019). Marine and coastal planners' biggest challenges are to restore the natural ecosystem or enhance the ecological functioning while managing human activities.

Many effective tools exist that combine these ecological and social objectives including fishery management, marine spatial planning, and Artificial Reefs (ARs) (Carral et al., 2022; Chipaux et al., 2016). ARs - man-made underwater structures placed at the bottom of the sea to mimic a natural reef's physical and biological functions- were first installed to increase fishery production (Salaün et al., 2020a, b, 2023; Vivier et al., 2021; Taormina et al., 2021). But more recently and to a greater extent, ARs have been used to rehabilitate marine ecosystems and their functionalities, or to mitigate anthropogenic effects on marine coastal ecosystems (Seaman, 2019; Patranella et al., 2017; Pioch et al., 2011).

ARs can help achieve four major management objectives:

1. Production: enhance the fishery resource.
2. Protection: create a physical barrier in the coastal strip to protect a specific area.

3. Recreational: provide recreational fishing or scuba diving activities.

4. "Eco-functional": restore ecological functionalities after negative anthropic impact or to create a privileged area for the development of marine fauna (Salaün, 2022). The intended ecological functions are to provide permanent habitat or shelter, an area suitable for breeding or spawning, an area for juvenile development (nursery) and feeding areas. Feeding, nursery, and reproductive functions are key stages in the life cycle of fish and play a critical role in resource renewal (Lacroix et al., 2002). Within the new habitat the intended effect is an increase in biodiversity and abundance of specific species.

Evaluating ARs' ecological efficiency is crucial to prove they are meeting their objectives. Traditionally three types of assessment have been used. The first focuses on verifying colonization and development (Folpp et al., 2011). This process starts with checking if benthic fauna has colonized the structure and then measuring variation in the fish population. Historically, the primary goal of ARs was to enhance fish biomass, therefore this assessment focused mainly on metrics such as variations in fish assemblages, abundance, and the richness of species (Véron et al., 2008 ; Folpp et al., 2011; Neves dos Santos and Zalmon, 2015; Becker et al., 2018). Another type of assessment looks at whether ARs offer habitat quality which is similar to natural reefs (Page et al., 2007; Hallier and Gaertner, 2008). This requires natural reefs to exist and remain largely intact for comparison (Lemoine et al., 2019). The final type of assessment compares the efficiency between ARs to determine which design is most effective and what the general benefits tend to be (Dafforn et al., 2015, Firth et al., 2016).

The diversity of assessment processes provide contrasting results: while some studies have shown the direct effect of ARs on the increase in local primary production (Cresson et al., 2019, Véron et al., 2008; Folpp et al., 2011; Neves dos santos and Zalmon, 2015; Becker et al., 2018) others have observed major differences with natural reef systems (Becker et al., 2022; Bulger et al., 2019, Koeck et al., 2014 ; Folpp et al., 20113). Several factors can influence these efficiency results such as the design, the distance between the artificial and the natural reefs, the age of the reefs, the complexity of the habitats, etc. Together, these factors emphasize the difficulty in assessing the ecological impact of ARs and highlights the necessity in refining assessment methods and metrics to conduct a more specific analysis of their ecological functioning and efficiency (Lee et al*.,* 2018; Lima et al., 2020).

A well-known method, consisting in the modelling of trophic networks using Ecopath with Ecosim (EwE), was recently applied to coastal and marine systems to assess changes in their functioning, in response to anthropic pressure (Christensen et Pauly, 1992; Christensen and Walters, 2004). This method has already been applied to Offshore Wind Farms (OWF), marine aggregate exploitation, harbour construction and dumping of dredged materials (Raoux et al., 2017; Pezy et al., 2017), specific regulations for Marine Protected Areas (MPA) (Prato, 2016; Valls *et al.*, 2012; Wallmo and Kosaka, 2017; Fulton et al., 2015) and ARs (Salaün et al., 2023; Xu et al., 2019; Guan et al., 2016). This method offers a wide range of metrics- Ecological Network Analysis (ENA) -which are capable of detecting changes in an ecosystem and reflect the overall relationship between the functional compartments in the model (Safi et al., 2019; Fath et al., 2019).

In the study, we combined a before/after comparison with the trophic modelling approach to explore new metrics which assess ARs' ecological functioning and efficiency. We aimed to demonstrate that Ecological Network Analysis offers valuable metrics suitable for the different types of ARs and for the three major types of assessment, including intrinsic evaluation, comparisons with natural systems, and comparisons with other ARs. To illustrate this approach, we modelled the trophic network of a coastal ecosystem before and after the implementation of ARs. Specifically, we chose the site of Cherbourg harbour, known as the "Rade de Cherbourg", located along the French side of the English Channel. The Cherbourg area is considered one of the most anthropized seas (Halpern et al., 2008) making it an ideal case study to examine ARs implemented for restoration purposes. We chose to integrate in our models 25 functional compartments ranging from detritus to seabirds and marine mammals to represent the entire ecosystem. Although the ARs in our study are not expected to directly affect the higher and lower functional compartments, including them in the model allows us to propose a general approach that can help reveal potential effects on these compartments in other ecosystems.

2 Study area

The study area is located in English Channel, on the north of the Cotentin coast within the "Rade de Cherbourg" that is the largest artificial harbour in the world measuring about 1,500 ha with 3.7 km of dykes. The "Rade de Cherbourg" is delineated by three breakwaters allowing exchanges with open sea water by three channels (West Channel, East Channel and Collignon Channel)(Figure 1). The "Rade de Cherbourg" is a semi-enclosed ecosystem characterised by a sandy bottom surrounded by pebbles and hard substrates (Hamdi et al., 2010). The water volume is about 80,000 m^3 at spring tide, with a tidal range of 5.3 m (Merceron et al., 2002; Poizot et al., 2021).

The exchange with open sea is conditioned by the tidal cycle. On a rising tide, the harbour system fills from west to east through the West entrance. When the flows attempt the West entrance, the space shrinks and causes a venturi effect or an acceleration of the current (Poizot et al., 2021). During the rising tide, at the bottom of the East and West breakwater, the water is sucked in and generates an east-west current. When the tide ebbs, the current flow direction reverses and the harbour empties through the West entrance. The maximum current flow is 0.55 m s^{-1} in mid-water on an East-West axis. The European project RECIF (cooperation program INTERREG France/England, co-financed by the FEDER funds) implemented ARs in the "Rade de Cherbourg" in April 2015. This project used waste shells from oyster industry to make the concrete for ARs (Dauvin et al., 2021). The AR measures 3 m long, 2 m wide and 1.35 m high. The AR consists in 72 blocks, spread over 3 levels (8,1 m^3 each AR). Three clusters of four ARs forming a triangle measuring 0.46 ha were installed for a total volume of 97.4 m^3. They were placed in the harbour, 500 m away from the West breakwater and on 7 m deep water at low tide (Vivier, 2021). The triangle area is marked with buoys and is prohibited from any activity outside of scientific monitoring. The monitoring was carried out two years and four years after the ARs implementation by the University of Caen Normandy, the BOREA and M2C laboratories as part of the INTERREG MARINEFF project, and the SINAY study office.

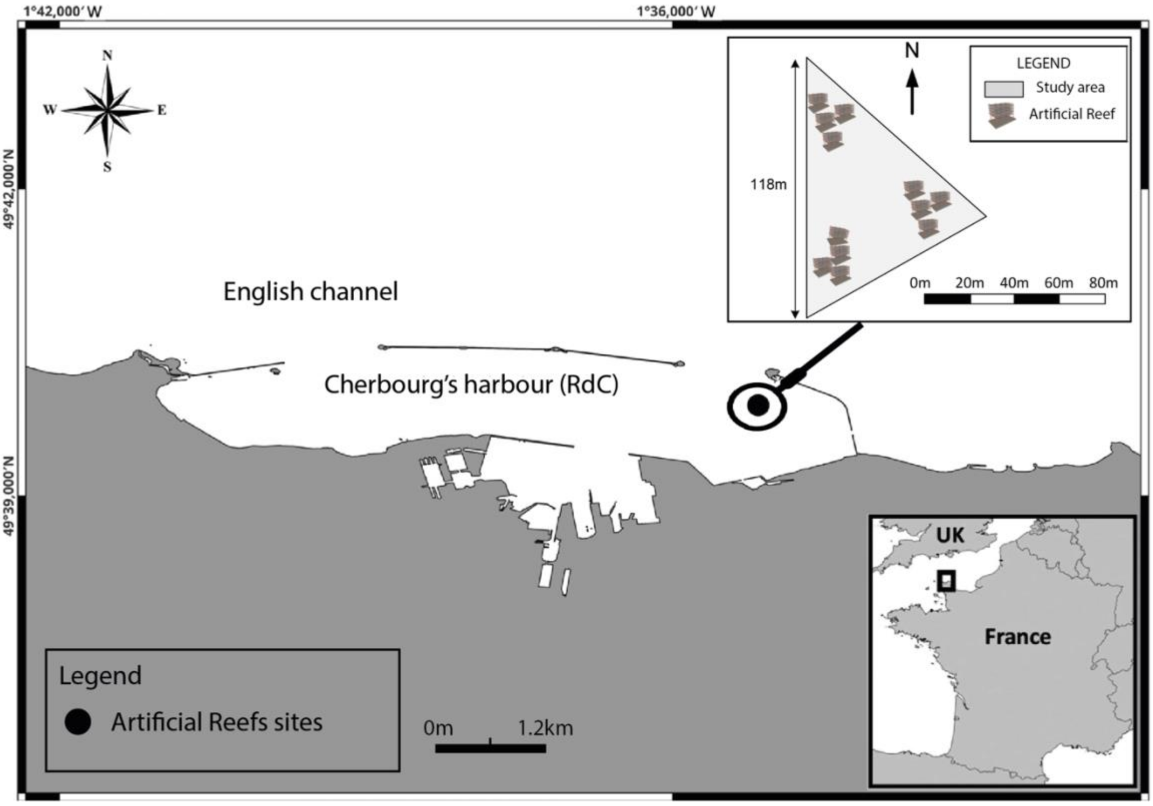

Figure 1: Location of Cherbourg'ARs site, along the French side of the English Channel. On the lower part, the figure shows the ARs location at the country scale, the main map represents the ARs position within the "Rade de Cherbourg" harbour and on the upper part the figure represents the triangle site forming by the 3 clusters of 4 ARs.

3 Trophic modelling approach: principle and steps of the method

The Ecopath with Ecosim (EwE) approach (Christensen and Pauly, 1992) was used to assess and perform Ecosystem Network Analyses (ENA) before and after the installation of ARs in the "Rade de Cherbourg". EwE is a trophic modelling approach developed to address ecological purposes and to investigate responsiveness of complex ecological systems (Patonai and Fábián, 2022; Christensen and Pauly, 1992). The model simulates mass-balance over a selected period in which the ecosystem is represented by functional compartments, which can be composed of species or group of species with ecological or biological similarities. Each compartment is characterized by biomass values, production

(P), consumption (Q), ecotrophic efficiency (EE) (the proportion of the production in the model) and diet data.

The Ecopath with Ecosim model is based on the assumption of mass conservation in the studied system, meaning there is a balance of incoming and outgoing flows. This hypothesis is described through two fundamental equations (Christensen and Pauly, 1992):

- The first equation (Eq. 1) describes the production of each i^{th} compartment (P_i) in the system as a function of the biomass (B_i) of its predators (M_i), the fishing mortality (Y_i, $gC \cdot m^{-2}$), the net migration (E_i; emigration – immigration, year−1), the biomass accumulation (BA_i, $year^{-1}$) and its natural mortality (1-EEi).

$$P_i = B_i M_i + Y_i + E_i + Ba_i + P_i(1 - EE_i) \text{ (Eq. 1)}$$

- The second equation (Eq. 2) ensures energy balance by calculating the consumption of the i^{th} compartment (Q) as the sum of its production, respiration (R), and excretion (Ui).

$$Q_i = P_i + R_i + U_i \text{ (Eq. 2)}$$

This modelling approach requires defining the study period and the functional compartments. Adapting this approach to our purposes (assessing changes in ecosystem due to ARs implementation in the harbour ecosystem), we chose to build a trophic model representing the ecosystem state of the Cherbourg harbour before ARs implementation, i.e. before 2015, to compare it with a second trophic model depicting the ecosystem state modified by the ARs. For both models, we selected 25 functional compartments ranging from detritus to seabirds and marine mammals. The distribution of species across these compartments provides a holistic view of prey-predator relationships, from primary producers to top predators.

The first step of the Ecopath with Ecosim approach consists of collecting the data for each species to characterize the functional compartments (Figure 2). The data needed are biomass (B), biomass-related production (P/B), biomass-related consumption (Q/B) and diet (expressed as percentage of species consumed). The data collected for both models are based on literature reviews and field data (Figure 2). Despite favouring on-site data, the spatial extent for both models depends on the availability of the data.

The second step consists of balancing the model to ensure trophic coherence and mass conservation. We used the Prebal diagnosis recommendations.

The last step is the data analysis. For this part, ecological metrics have been chosen to depict the different network levels: the node (functional compartment), the links between nodes (flows), the network (structure), and the functioning (activity).

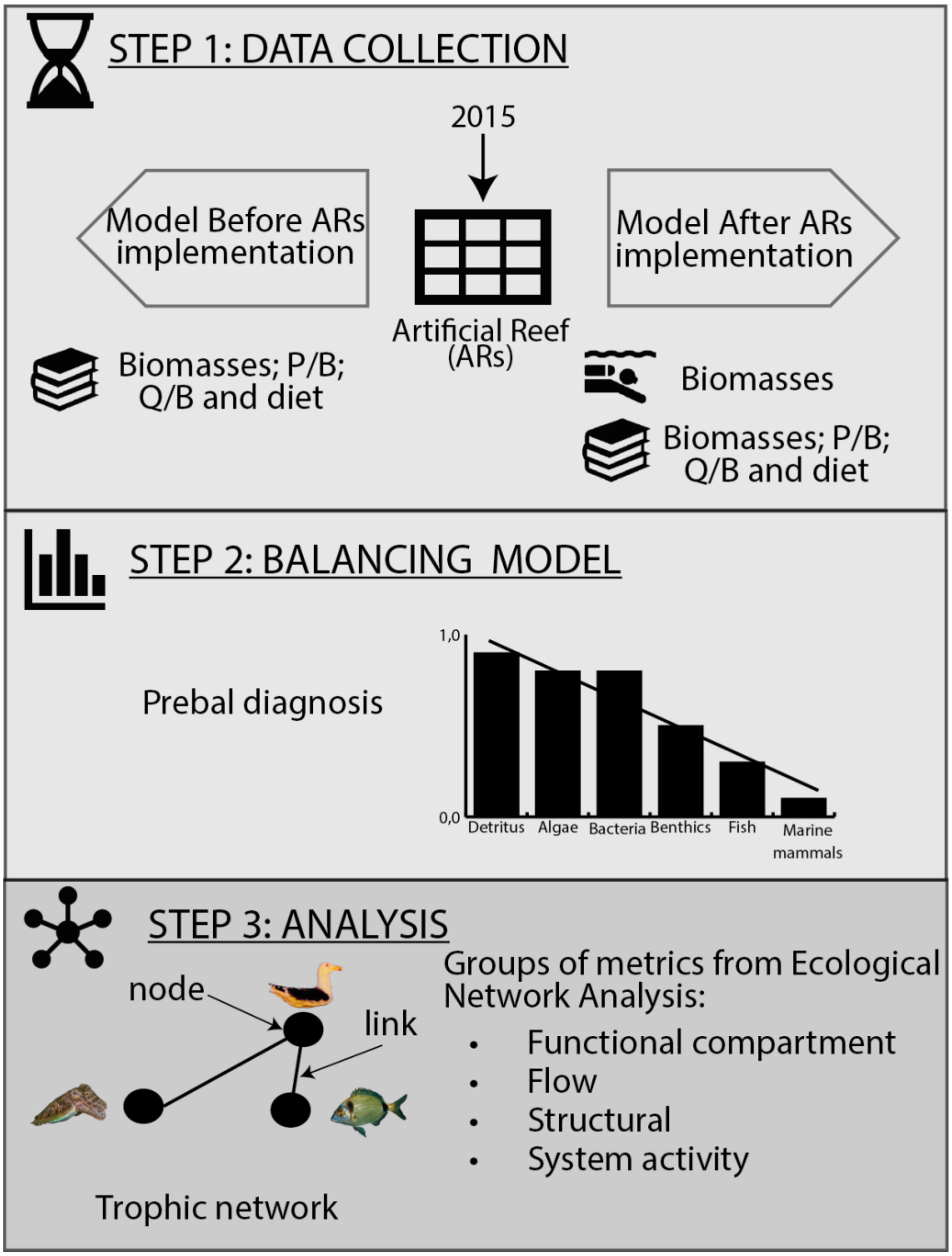

Figure 2: Adapting Ecopath with Ecosim modeling approach to our study in three major steps. The first step consists in collecting the data needed that are the biomass (B), the biomass-related production (P/B), the biomass-related consumption (Q/B) and diet. The second step consists in balancing the model using the Prebal diagnosis recommendations. Finally, the last step is the analysis of the network using Ecological Network Analysis that provides different metrics focusing on functional compartments, flows, structures and activities of the system.

4 Step 1: Data collection

The data collection used for modelling before and after the implementation of ARs was mostly based on sampling in the English Channel or estimated data (Figure 3). We prioritised in-situ data for both models. Where in-situ data were not available, we selected data from sites as close as possible to the study site and with a similar habitat (fine sediment).

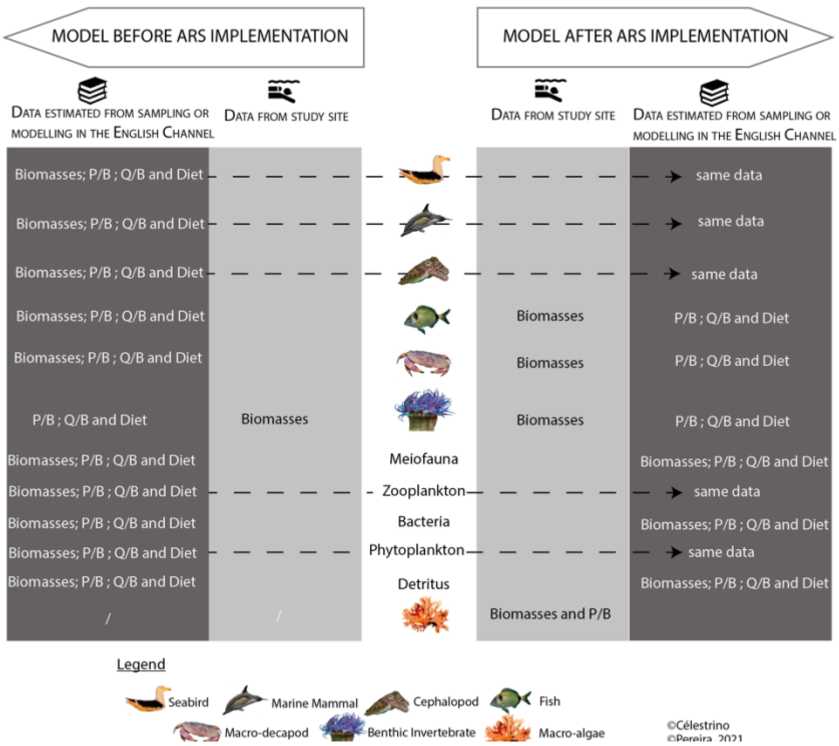

Figure 3 : Summary of Data Sources for Ecosystem Modeling Before and After AR Implementation. ***.*** *This figure illustrates the data sources used for modeling the ecosystem. The sources are categorized into two groups: data collected from the study site and data estimated or derived from other studies outside the study area. The essential data includes biomass (B), biomass-related production (P/B), biomass-related consumption (Q/B), and diet composition.*

4.1 Data collection for the model before ARs implementation

4.1.1 Primary producers and lower functional compartments

Zooplankton biomass was taken from trophic web modelling carried out in the eastern Channel by Garcia (2010), who used a single biomass value as an average of Channel estuary data (Rybarcyk and Elkaim, 2003) and estimated data for herbivorous zooplankton (Martin et al., 2009), zooplankton, mesozooplankton and microzooplankton (Vézina and Platt, 1988). Phytoplankton biomass was taken from the REPHY monitoring network carried out by Ifremer from 1992 to 2017 (Belin et al., 2021). For zooplankton and phytoplankton, it was assumed that AR implementation had no effect on their biomass and production (Miller and Falace, 2000). Therefore, we used identical data for both models and extended the data collection to both time periods. The meiofauna biomass corresponds to estimates made in the Bay of Saint-Brieuc (Bodin et al., 1989), and the bacteria and detritus biomasses are from simulations carried out in the Bay of Morlaix (Chardy and Dauvin, 1992). P/B, Q/B and diet were derived from modelling carried out in the eastern Channel by Garcia (2010) and from the simulation in the Bay of Morlaix (Chardy and Dauvin, 1992).

No macro-algae were described in the study area prior to the implementation of the AR. Consequently, the biomass of this compartment was approximately 0.

4.1.2 Benthic invertebrate

The benthic macrofauna of the soft-bottom communities of Cherbourg harbour was characterised between 2012 and 2015 by Baux et al. (2017). Fine sediment benthic macrofauna samples were collected in three campaigns: March 2012, February 2014 and 2015 (Baux et al., 2017). A total of thirty stations were sampled, nine of which, located in the eastern part of the harbour, were used to characterise the study area. For each station, three replicates were conducted using a Van Veen grab with a sampling area of 0.1 m², totalling 0.3 m² per station, and a fourth replicate was used for particle size analysis (Baux et al., 2017). The collected sediment was sieved using a 1 mm circular sieve. Samples were preserved in a 10% formaldehyde solution before sorting. Macrozoobenthic species were identified and counted at the lowest taxonomic level under a binocular microscope. The identified species were then placed in a drying oven at 60°C for the time necessary to dry the sample (approximately 48 to 96 hours). The samples were weighed to determine the dry weight (DW) before being placed in another oven at 500°C for 5 hours. The ash-free dry weight (AFDW) was obtained by subtracting the ash weight from the DW. Biomass was then calculated by converting the ash-free dry weight to carbon content using a conversion factor of 0.518 (Brey, 2001). Each biomass was applied to the sampled surface of the concrete blocks and then to the proportion of the colonisable surface in the study area (0.46 ha) to obtain the biomass value per m². P/B, Q/B and food were taken from Garcia (2010) and Pezy (2017).

4.1.3 Fish, cephalopod, and macro-decapod

No census of the mobile marine fauna (fish, cephalopods, and mcaro-decapods) was carried out in the area prior to the implementation of the AR. However, we calculated fish and cephalopod biomasses derived from the annual open sea bottom trawl survey, the Channel Ground Fish Survey (CGFS), to test whether these biomasses could characterise the system prior to the implementation of the AR.

The CGFS campaigns have been carried out by Ifremer in the English Channel since 1988 to estimate the abundance of the main demersal fish species (Le Roy et al., 1988; Baudrier et al., 2018). The gear used is a trawl with a high vertical opening. The data used cover the period 2012-2015. The selected trawl lines are located in the offshore settlement area, similar to the stocks in the study area (Carpentier, 2009). Biomasses were calculated from the catch data reported on the surface swept by the trawl lines, giving an average biomass value over the three years considered by species. The wet weight calculated for each species was converted to dry weight and then to carbon content using conversion factors of 0.35*0.35 for (fish and cephalopods) and 0.518 for macro-decapods (Brey et al., 2001; 2010).

The test results showed that the values extracted from these off-harbour campaigns were too high compared to the wildlife observed during the visual census campaign. Therefore, it was decided to apply a 30% reduction to the in-situ values of diving fish biomasses (described below). This rate is based on the results of a study in China where fish biomass was estimated to have increased by 30% with the implementation of ARs, corresponding to 2% of the surface area of the MPA (Pitcher et al., 2002). The ARs in the study area are represented in the same proportion in the study triangle.

For the cephalopod biomass, we estimated it using the fish/cephalopod biomass ratio from these campaigns and the fish biomass obtained using the -30%.

Mean biomasses were calculated for each functional compartment. Q/B and P/B ratios were taken from Mackinson and Daskalov (2007), who constructed the North Sea Ecopath model based on compilation from ICES stock assessment reports and estimation from mean daily rations. The diet matrix was constructed mainly using stomach contents from literature data from the eastern part of the English Channel (Carpentier et al., 2009; Lobry et al., 2008; Mackinson and Daskalov, 2007; Pezy, 2017).

4.1.4 **Marine mammal and seabird**

For the same reason as for fish, we were unable to use data from oceanographic campaigns. As a result, we decided to apply a ratio of marine mammals and seabirds to fish based on offshore species data, and then compare this to in-situ fish biomasses to determine the biomasses of top predators at our study site. Below we describe in detail how we established this ratio.

Marine mammal and bird densities were derived from the aerial strip transect of marine fauna (SAMM) campaigns conducted throughout metropolitan France in winter 2011 (November to February) and summer 2012 (May to August) across the English Channel (Pettex et al., 2014; Laran et al., 2017). The densities used correspond only to those of the continental shelf (26,682 km²). Biomass was calculated from these densities and the estimated mean weights for each species.

Mean weight, diet, P/B and Q/B data sources by species were derived from Spitz (2010); Trites et al. (1999); Certain et al. (2008); Nilsson and Nilsson (1976); Thompson et al. (1999); Hamer et al. (2000); Shamoun-Baranes and Camphuysen (2013); Bustnes et al. (2010); Spitz et al. (2018); and Pierrepont et al. (2003).

4.2 **Data collection for the model after ARs implementation**

Data collection for the model after the implementation of the ARs was based on estimated data, sampling in the English Channel and sampling at the study site (Figure 3).

4.2.1 Primary producer and lower functional compartments

From December 2019 to September 2020, sampling was carried out during four surveys: winter (5 February 2020), spring (2 June 2020), summer (1 July 2020) and autumn (29 September 2020). In each campaign, six concrete blocks from the AR top module were randomly sampled and transported to the laboratory using a zooplankton net (500 µm) to avoid any loss of fauna. Each concrete block was first incubated in a hermetic chamber with controlled lighting, kept in the large basin, to record photoactive radiation and calculate gross primary production (GPP) (see Vivier, 2021 for more details on primary production measurements). The production of macro-algae reported on the surface of the studied area (0.46 ha) results in 17.79.

After production estimation, the concrete blocks were conserved at -20 °C for macro-algal identification. Macro-algal species were classified into three main classes (Ulvophyceae, Rhodophyceae or Phaeophyceae) using Algaebase. The collected species were then placed in a drying oven at 60°C for the time necessary to dry the sample (approximately 48 to 96 hours). The samples were weighed to determine the dry weight (DW) before being placed in another oven at 500°C for 5 h. The ash-free dry weight (AFDW) was obtained by subtracting the ash weight from the DW. Ash-free dry weight biomass was converted to carbon content using a conversion factor of 0.35 (Surif and Raven, 1990; Pihl et al., 1996).

The macro-algae production has an influence on the biomass of detritus, bacteria and meiofauna. In fact, the macro-algae biomass export rate was estimated between 40% and 80% (Santos et al., 2021). The remaining biomass, which is not consumed, is recycled into detritus biomasses (Smale et al., 2020). The 80% export parameters were applied to the Cherbourg model resulting in an accumulation of 20% of dead macro-algae that were considered as supplementary detritus. Therefore, the detritus biomasses were calculated using the 20% ration and the value of the macro-algae production. With a similar trophic relationship, we used the ratio from Newell (1984) to calculate the biomasses of bacteria and meiofauna. For zooplankton and phytoplankton, ARs implementation was assumed to have no effect on their biomass and production (Miller and Falace, 2000).

We used the same value of P/B, Q/B and diet for both models (Figure 3).

4.2.2 Benthic invertebrate

The second parts of the concrete blocks coming from the sampling campaign were preserved in 10% formaldehyde solution before sorting. Macrozoobenthic species were identified and counted under a binocular microscope at the lowest taxonomic level. Biomass was then determined by AFDW and converted to carbon content using a conversion factor of 0.518 (Brey, 2001). Each biomass was divided by the area sampled on the concrete blocks and then extrapolated to the proportion of colonisable area in the study site (0.46 ha) to obtain a biomass value per m2.

We assumed that at the scale of a decade, the biomass of the fine sand community where the ARs were installed did not change much, so we used the fine sediment benthic macrofauna data from Baux et al. (2017), also for the post-implementation model, and added this biomass to the biomass of the hard substrate benthic macrofauna obtained from the in situ campaign described above.

We used the same value of P/B, Q/B and diet for both models (Figure 3).

4.2.3 Fish, cephalopod, and macro-decapod

The data were derived from underwater visual census observations from scientific campaigns carried out in August 2017, 2019 and July 2021 by the SINAY study office (Chevallier and Leroy, 2017, 2019) and the BOREA laboratory. The method included both stationary point counts and belt transect counts, allowing the recording of fast-moving species as well as benthic and cryptic species (Cresson et al., 2019; Lowry et al., 2011; Charbonnel et al., 1997; Labrosse et al., 2011). The three AR clusters were visited on each date for one count per site. The campaign followed the recommendations of Harmelin-Vivien et al. (1985): population abundances were counted individually up to 10 individuals, while larger populations were estimated using abundance classes (11-30; 31-50; 51-200; 201-500; 500-1000; >1000 individuals). The total length of fish was estimated in cm by trained scientific divers. The wet weight was obtained using the length-weight relationship W = a x TLb, where W is the wet weight in grams, TL is the mean total length of the size class in cm, while a and b are species-specific constants obtained from data available in Fishbase (Froese and Pauly, 2019) and selected in the vicinity of the study area. The calculated wet weight for each species (fish and macro-decapods) was converted to dry weight and then to carbon content using the conversion factors of 0.35 and 0.35 for fish and 0.518 for macro-decapods (Brey et al., 2001; 2010). Biomass was calculated using the mean abundance and wet weight for each species (fish and decapods) reported for the area of the study site (0.46 ha).

The absence of cephalopod observations during in-situ monitoring forced us to consider that these surveys were not adapted to estimate cephalopod biomass. As a result, in this study, the same data were used for these fauna compartments to model trophic networks before and after ARs implementation.

Mean biomasses were calculated for each functional compartment. Q/B and P/B ratios were taken from Mackinson and Daskalov (2007). Diet matrices were constructed mainly using stomach contents from literature data from the eastern English Channel (Carpentier et al., 2009; Lobry et al., 2008; Mackinson and Daskalov, 2007; Pezy, 2017).

4.2.4 Marine mammals and seabirds

For marine mammals and seabirds, due to the small surface area of ARs (72 m^2) compared to their predation areas (in km^2), ARs were assumed to have little impact on the biomass of these top predators (Ricart et al., 2014). Therefore, this study used the same data for these faunal compartments to model trophic networks before and after the implementation of ARs.

5 Step 2: Balancing the model

The original models were not balanced, as some Ecotrophic Efficiencies (EE) were greater than 1. The models were balanced according to the prebalance diagnosis (Link, 2001) and the quality of the sources. For example, the biomasses of the piscivorous, planktivorous, flatfish, macro-decapod and zooplankton trophic compartments were re-estimated by the model to match the EE estimated for them, which were found to be 0.6 for piscivorous fish (mainly conger eel with few predators), 0.875 for zooplankton and 0.9 for planktivorous fish, flatfish, macro-decapods (Christensen and Walters, 2004).

The Pedigree Index was used to assess the quality of the two models. This pedigree allows to categorise the origin of the inputs (B, P/Q, Q/ B, diet) and to indicate the likely uncertainty associated with these inputs (Christensen and Walters, 2004). The quality score ranges between 0 (low) and 1 (high) for each parameter according to its origin. An overall index of model quality can be obtained.

6 Step 3: Analysis

6.1.1 The metrics

When analysing trophic models, metrics characterized different levels of the network. At the first level, nodes are the focus, with metrics emphasising functional compartments. At the second level, attention shifts to fluxes, with relevant metrics capturing the connections between these compartments. The third level examines the structural aspects of the network. Finally, the fourth level focuses on network activity, its functionality and resulting output. Several metrics are available in the Ecopath plug-in to Ecosim (Christensen and Walters, 2004). From these, 18 were selected to describe changes in trophic network (Tomczac, 2013; Fath et al., 2019; Safi et al., 2019). Their definitions are presented in Table 1 (Tomzcak et al., 2013; Saint Béat et al.,2015, Gao et al., 2022; Safi et al., 2019).

Table 1 : Description of metrics categorized by their respective network levels: functional compartment, flows, structure, and activities.

	Metric	*Description*	*Reference*
Functional compartments *(level 1: node's characterization)*	Biomass	Biomass estimates are calculated in weight/area	Heymans et al., 2016
	Trophic Level (TL)	TL is the weighted average of the trophic levels of its prey (j),	Christensen et al., 2008
	Total living Biomass by trophic level	The biomass of each compartment in the system (without detritus) is distributed onto trophic levels	Christensen et al., 2008
Flows *(Level 2: link's characterization)*	Total System Throughput (T..)	Size of the whole ecosystem which corresponds to the sum of all flows occurring in the system	Ulanowicz 1986
	Sum of all respiratory flows	Measure all non-usable carbon that leaves compartment that represents the major parts of assimilated food	Christensen et al., 2008
	Sum of all Export	Measure of all carbon that leave the system	Christensen et al., 2008

Structural *(Level 3: network's characterization)*	Average Mutual Information (AMI)	Degree of system organization	Ulanowicz 1986
	Mean Path Length (MPL)	Average number of compartments that an inflow or outflow passes through	Finn, 1980
	Finn Cycle Index (FCI)	Fraction of the flows in the system that is generated by recycling	Finn, 1980
	System Omnivory Index (SOI)	Measure of the predator's trophic specialization of predators in terms of trophic levels and is an indicator of the structure of a trophic network	Libralato, 2008
	Connectance Index (CI)	Measure of the network complexity and is defined as the number of actual flows in the system by the total of possible number of flows	Christensen and Pauly, 2004
	Detritivory/Herbivory (D/H)	Distinguish the originated flows entering the trophic web	Ulanowicz 1992
Activity *(Level 4: functional's characterization)*	Total primary production/total biomass (PPt/Bt)	Measure of the system maturity.	Odum, 1969
	Total primary production/R (PPt /R)	Ratio of energy used for biomass production (total primary production) to energy used for maintaining system stability (total respiration). Measure of the system maturity.	Christensen et al., 2008; Odum, 1969
	Total biomass/total throughput (Bt/T..)	Measure of the system maturity.	Odum, 1969
	Relative Ascendancy (A/C)	Measure of the growth and the flow coherence of the system, integrating its size and its organization	Ulanowicz 1986
	Relative Redundancy (R/C)	System's energy in reserve that indicates possibility of alternative pathways for energy flows to get from one compartment to another	Ulanowicz, 1986
	Transfer Efficiency (TE)	Global efficiency of transfer through the network	Baird and Ulanowicz, 1989

6.1.2 The emergent properties

Several metrics allow us to describe the structural and functional properties of systems:

- The complexity of the network, derived from the Connectance Index (CI) and System Omnivory Index (SOI) metrics, provides information about the shape and structure of the network (Pauly et al. 1993). A complex network could be defined as one with a high flow capacity and a reticular shape.
- Flow diversity could be inferred using the herbivory (H) and mean path length (MPL) metrics. H and MPL are expected to be higher in systems with higher flow and cycling diversity (Christensen, 1995 in Fath et al., 2019).
- Recycling capacity is based on Detritivory/Herbivory (D/H). The D/H indicates the dependence of the system on algal or detrital materials (De jonge et al., 2021).

- Resilience is expressed by the Finn Cycle Index (FCI) metric. A high FCI value would mean that the system would recover more quickly from a perturbation or stress, while a lower FCI value would mean that the system would take longer to recover (Monaco and Ulanovitcz, 1997).
- Efficiency capacity is described by the transfer efficiency (TE) metric. The higher the TE value, the shorter the time it takes to transfer energy from low TL to higher TL (Safi et al., 2019).
- Robustness combines both efficiency and resilience of a system and refers to the sustainability of the system (Fath et al., 2015). It is calculated using the system's ascendency and capacity (A/C). The best value of system robustness is defined as an optimal peak balanced between efficiency and redundancy (Ulanowicz, 2009).
- The maturity of the system could be described by Odum's metrics: Total primary production/R (PPt/R), Total primary production/total biomass (PPt/Bt) and Total biomass/total throughput (Bt/T..) (Odum, 1969). When the system is growing, generally in a "young system", production exceeds respiration and the PPt/R index is higher. On the contrary, when the system is mature, it tends to balance the use of energy in terms of both production and consumption (Odum, 1969). The Bt/T.. ratio is an index that increases with the maturity of the system.

7 Results

The pedigree index in each model was 0.47 and 0.53 respectively for the models representing the system before and after ARs implementation, which is in the middle of the values obtained from more than 150 Ecopath models worldwide (Morissette et al., 2006). Thus, the models appear to be reliable for good quality input data.

7.1 Functional compartment characteristics

7.1.1 The biomasses

The results showed that the total biomass (with detritus) was higher after the installation of the ARs (Table 2). In fact, the biomass increased by about 32% between the two periods. This change was mainly due to the biomass of detritus, bacteria and macroalgae (dominated by Rhodophyceae (Sphaerococcus coronopifolius and Kallymenia reniformis), which increased by 3.6 gCm^{-2}, 0.53 gCm^{-2} and 0.62 gCm^{-2} respectively between the two periods (Table 2).

Detritus was the dominant functional compartment representing 61% and 64% of the biomass models. The other main compartments were (Table 2)

- the benthic invertebrate surface detritivore (sdF) (dominated by the polychaete Scoloplos armiger), which represented 10% of the total biomass of the system before the implementation of ARs and 7.7% of the total biomass of the system after the implementation of ARs,

- the benthic invertebrates, Sub Surface Detritic Feeder (SSDDF) (dominated by the polychaete Spio decorata), which represented 5% and 3.8% of the total biomass of the system before and after ARs implementation, respectively.

- The benthic invertebrates Scavenger/Omnivorous (scv/o) (dominated by the gastropod Tritia reticulatus), which represented 4.7% of the total biomass of the system before and 3.6% of the system after the implementation of ARs,

- the bacteria, which represented 5.7% of the total biomass of the system after the introduction of ARs.

*Table 2 : Biomass values (gC.m^{-2}, Trophic Levels, production over biomass (P/B) ratios, consumption over biomass (Q/B) ratios, in the two Ecopath models representing the system before and after ARs implementation. Major changes were highlighted in bold and dominant living functional compartment were indicated by *.*

	Biomass		**Trophic Level**		**P/B**		**Q/B**	
Functionnal compartment	**Before ARs implementation**	**After ARs implementation**	**Before ARs implementation**	**After ARs implementation**	**Before ARs implementation**	**After ARs implementation**	**Before ARs implementation**	**After ARs implementation**
Plunge and pursuit diver's seabirds	0.000006	0.000006	4.27	4.39	0.09	0.09	25.16	25.16
Surface feeder seabirds	0.000006	0.000006	4.31	4.48	0.09	0.09	39.84	39.84
Marine mammals	0.00006	0.00006	4.63	4.67	0.08	0.08	23.83	23.83
Benthopelagic cephalopods	0.000082	0.000082	4.27	4.37	2.80	2.80	15.00	15.00
Benthic cephalopods	0.000015	0.000015	4.25	4.34	3.50	3.50	15.00	15.00
Gadidae	0.000005	0.000007	3.91	4.16	1.72	1.72	5.87	5.90
Fish, piscivorous	0.000987	0.00128	3.76	4.31	0.41	0.67	3.50	3.80
Fish, benthos feeders	0.00194	0.00253	3.58	3.48	0.46	1.30	2.77	8.63
Labridae	0.00013	0.000169	3.42	3.52	1.32	1.32	10.38	10.38
Sparidae	0.000055	0.000071	3.25	3.45	0.55	0.55	2.54	2.54
Fish, flatfish	0.000126	0.000164	3.37	3.40	0.58	0.58	4.50	4.50
Fish, planktivorous	0.000464	0.000603	3.15	3.15	1.68	1.68	6.33	6.33
Macro-Decapods	0.000299	0.000699	3.12	3.16	1.18	1.18	5.90	5.90
Benthic invertebrates, Predators	0.55621	0.56546	3.23	3.29	2.22	2.80	11.10	14.00
Benthic invertebrates, Scavenger/Omnivorous (scv/o)	0.70462 *	0.70541 *	3.10	3.11	1.18	1.18	5.90	5.90
Benthic invertebrates, Filter feeders	0.34623	0.35331	2.27	2.29	2.83	2.83	11.32	11.32
Benthic invertebrates, surface Detritic Feeder (sdF)	1.51674 *	1.51713 *	2.22	2.34	2.08	2.08	10.40	10.40
Benthic invertebrates, sub surface Detritic Feeder (ssdF)	0.74343 *	0.74565 *	2.28	2.34	2.24	2.24	11.20	11.20

Benthic invertebrates, grazer	0.00086	0.00092	2.22	2.22	2.00	2.00	10.00	10.00
Macro Algae	0.00001	**0.62100**	1.00	1.00	28.65	28.65		
Meiofauna	0.42000	0.49294	2.26	2.31	15.00	15.00	60.00	60.00
Zooplankton	0.36331	0.36331	2.15	2.15	11.00	11.00	52.38	52.38
Bacteria	0.59000	**1.12370 ***	2.00	2.00	72.80	72.80	145.60	145.60
Phytoplankton	0.60000	0.60000	1.00	1.00	150.00	150.00		
Detritus	9.05000 *	**12.60800 ***	1.00	1.00				

7.1.2 Trophic level evolution

In each model, the trophic levels (TL) of the functional compartments ranged from 1 for primary producers and detritus to a maximum of 4.67 for marine mammals. Considering only the live biomass (that excluded detritus), the primary trophic level (TL1) consisted of two compartments (phytoplankton and macroalgae) representing 10% and 17% of the total live biomass in the system before and after the implementation of the ARs (Figure 4 and Appendix A).

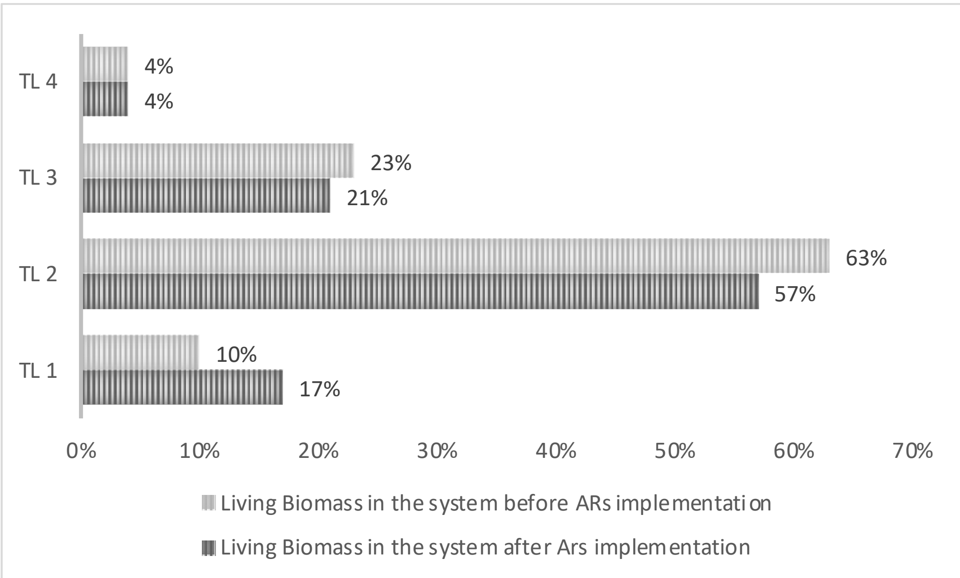

Figure 4: Proportion of each trophic level (from the primary trophic level -TL1-to the quaternary trophic level-TL4) in the total living biomass for the two ecopath models representing the system before and after the implementation of ARs. Proportion of each trophic level (TL) in the total living biomass for the two ecopath models representing the system before and after the implementation of ARs.

The secondary trophic level consisted of seven functional compartments (bacteria, zooplankton, meiofauna, sub-surface detritivores, grazers, surface detritivores and filter-feeding benthic invertebrates), representing 63% and 57% of the total living biomass in the systems before and after the implementation of ARs, respectively. The tertiary trophic level included most of the fish functional compartments (flatfish, benthivores, planktivores, Labridae and Sparidae). It was composed of nine functional compartments, representing 23% and 21% of the total living biomass for the systems before and after the implementation of ARs. Finally, the quaternary level consisted of five functional compartments. This level corresponded to the main predators and represented 4% of the total biomass in the systems before and after the implementation of ARs. This result showed that the secondary trophic level was the largest contributor to total living biomass in both models and that the ARs implementation in Cherbourg contributed little to biomass transfer to the upper trophic level.

7.2 **Flow characteristics**

The flux characteristics for each model are shown in Figure 5. The results showed a total system throughput value of 413.24 and 573.06 gC.m-2.year-1 before and after the implementation of ARs, respectively. Of these values, 41% and 44% were due to consumption (168.51 and 252.46 gC.m-2.year-1), 29% and 28% were due to detritus return (120.36 and 163.72 gC.m-2.year-1), 17% (69.88 and 96.72 gC.m-2.year-1) were due to respiration and 13% and 10% were due to export (54.48 and 60.16 gC.m-2.year-1). Thus, the total system throughput, consumption, detritus flux, respiration flux and export increased by approximately 39%, 49%, 36% and 10%, respectively, in the system after ARs implementation (Figure 5). The sum of all production was 151.12 and 209.93 gC.m-2.year-1 in the before and after model respectively (Figure 5 and Appendix A).

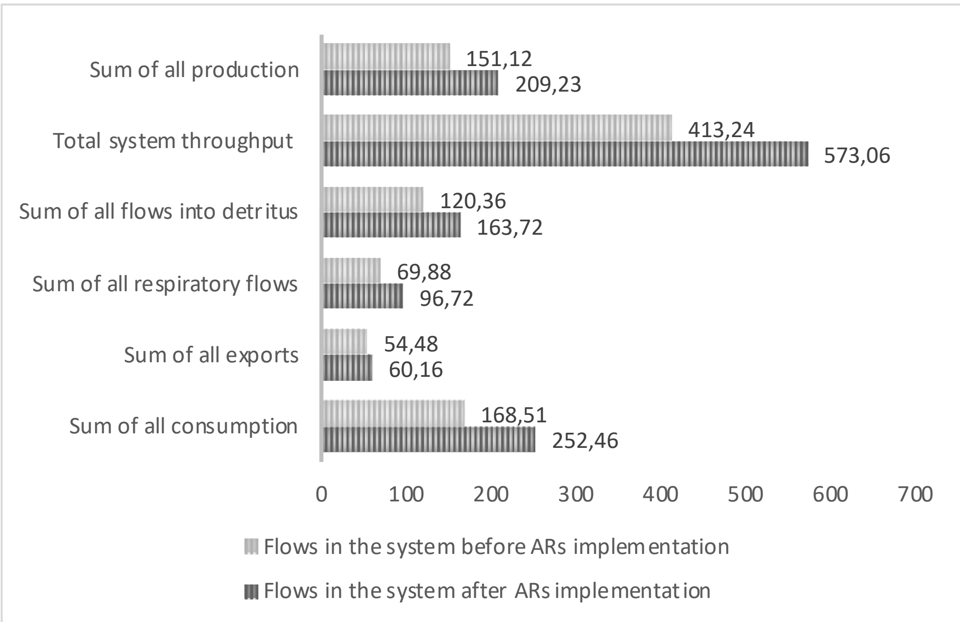

Figure 5: Flows for each model in gC.m-2 year-1

7.3 **Structural characteristics**

The indicators that described the flows in the network and the ecosystem structure for each model are listed in Figure 6. The structure of the trophic network evolved with the implementation of the ARs. The System Omnivory Index (SOI), Mean Path Length (MPL) and Finn Cycle Index (FCI) increased with ARs implementation (Figure 6). These change values expressed that the structural network became more complex (SOI +7.5%), with a higher diversity of flows (MPL +9.9%) and a better recycling system (FCI +42%). Meanwhile, the Average Mutual Information (AMI) and Detritivory/Herbivory (D/H) metrics showed little variation due to the implementation of ARs, meaning that the organisation in both models was based on detritus. All of these metrics of variation were consistent, showing a global change in the structural trophic network that became more focused on detritus fate.

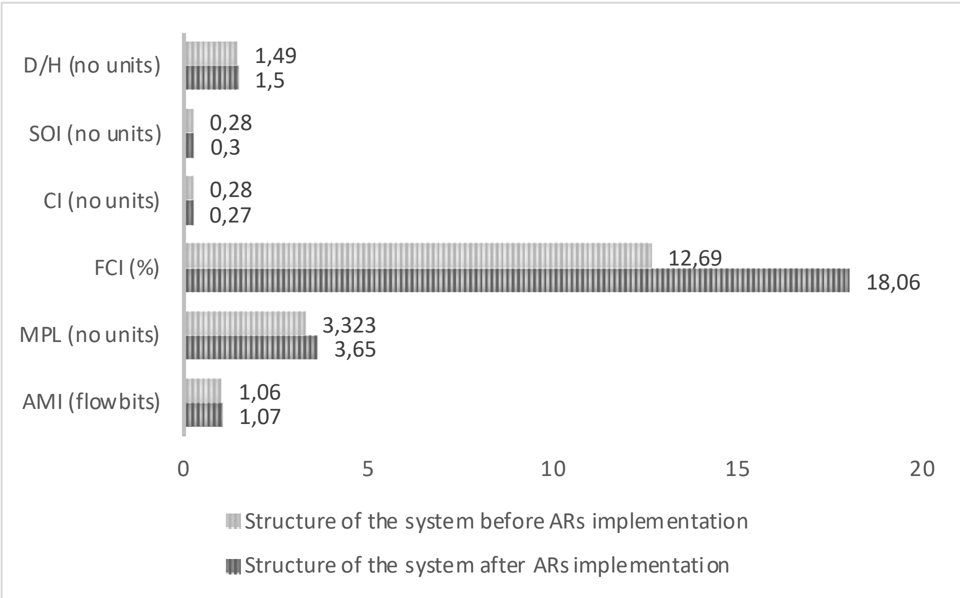

Figure 6: Structural metrics for each model that are Detritivory/Herbivory (D/H), System Omnivory Index (SOI), Connectance Index (CI), Finn Cycle Index (FCI), Mean Path Length (MPL), and Average Mutual Information (AMI)

7.4 System Activity characteristics

Results on system activity showed that total ecosystem activity (measured as the sum of all flows, T..) and relative ascent (A/C) increased between the two periods by about 38.7% and 5.5% respectively before and after ARs implementation (Figure 7). Relative Redundancy (R/C) values decreased with the implementation of ARs, indicating a reduction in the possible pathway to a higher trophic level (R/C -4.6%). Conversely, Transfer Efficiency (TE) increased with ARs implementation, showing a better capacity to transfer energy to high trophic level (TE+8.6).

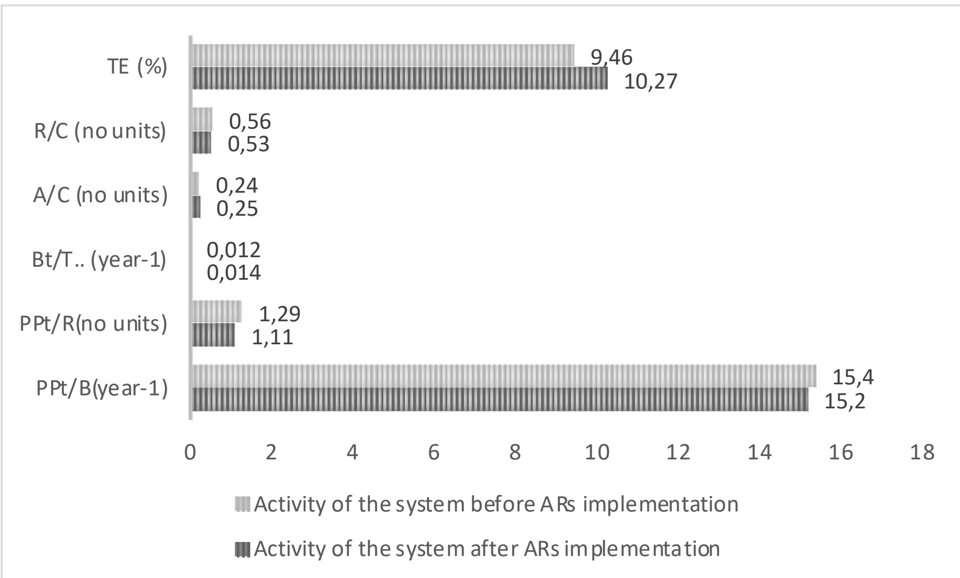

Figure 7: Activity metrics for each model that are Total primary production/total biomass (PPt/Bt), Total primary production/R (PPt /R), Total biomass/total throughput (Bt/T..), Relative Ascendancy (A/C), Relative Redundancy (R/C), and Transfer Efficiency (TE)

Regarding the metrics of the maturity of the ecosystem, the results showed that the ratios of total primary production/R (PPt/R), total primary production/total biomass (PPt/Bt) and total biomass/total throughput (Bt/T..) also varied between the two systems (Figure 7). In fact, the PPt/R decreased by about 13.5% in the system with ARs implementation (Figure 7). The Bt/T.. ratio showed an increase of about 12.5% with the implementation of ARs (Figure 7). Finally, the PPt/Bt showed little variation in the system with ARs implementation model by approximately 1.3% (Figure 7).

8 Discussion

In this section, we explore how Ecological Network Analysis metrics can serve as relevant indicators for evaluating the ecological efficiency of AR projects through three types of assessments: verifying colonization and development, assessing whether ARs provide a habitat comparable to natural reefs and determining trends in ecological benefits.

We first discuss (1) the limitations of the approach, then highlight the value of the metrics for each type of assessment (2 to 4). Finally, (5) we show which metrics could be used to assess each, specific AR objective.

8.1 Limitations of the trophic modelling approach

Trophic modelling relies on extensive biological data for each selected functional compartment. Diet, is a critical parameter. In our study, models were built with the most relevant data to the study area as possible, resulting in differences in sampling (e.g. between bottom trawl data and diver surveys) and temporal coverage. Consequently, our trophic models should be considered a first step which provides an overview of the AR's evolution.

The main difficulty in setting up a trophic analysis is the need for a large data set (biomass, production, consumption, diet). Monitoring all the necessary data can be challenging and costly for managers. It is therefore necessary to establish a sampling strategy to carry out the modelling and, more specifically, to measure AR efficiency metrics. Three main trophic levels are important to sample. Firstly, the first trophic levels (phytoplankton, bacteria, zooplankton, detritus) which are the least studied functional compartments. These functional compartments form the basis of the whole network and are involved in the primary production of the AR. Their monitoring could be carried out in a timely manner and would enrich the scientific knowledge and provide characteristic data for ARs. The next two essential compartments are the benthic fauna on hard substrate and the mobile fauna (fish, cephalopods, macro-decapods). In terms of trophic analysis, the monitoring of these functional compartments seems essential as they provide essential information on system properties such as maturity, complexity or resilience. In our study, we decided to focus on the main functional compartments that are affected by ARs (benthic invertebrates and fish), while fixing the biomasses of the upper and lower trophic compartments. This prevents direct biomass analysis for these compartments, but we kept them included in the trophic model to allow us to examine flux patterns within the whole system.

8.2 Using the metrics to verify colonization efficiency

In this part, we discuss the trends of metrics that show an interesting effect on the intrinsic functioning of ARs: regarding our main results on biomass production and system maturity.

8.2.1 Increase in macro-algae biomass and consequences for the ecological system

The main feature of the Cherbourg AR is the high development of macro-algae, which was significant and represented 9% of the system's total living biomass. Macro-algae on ARs is quite common worldwide, as they are known to provide new hard surfaces for attachment and by altering water flows around the structures, creating favourable conditions for their establishment (Schroeter et al., 2015; Young et al., 2015). Consequently, some ARs have been developed specifically to restore macroalgal forests, such as kelp forests or seagrass, such as artificial seagrass reefs (ASRs) (Jung et al., 2022). They provide shelter and food for herbivorous fish, leading to an increase in biomass (Heindeibin et al., 2006). In Cherbourg, all environmental conditions were favourable for the development of macro-algae after the AR was implemented: shallow depth (7 m on average) with a high amount of light penetration at this depth, proximity to a large natural hard substrate already colonised by various macro-algae including kelp, and new artificial hard substrate free for colonisation (Macreadie et al., 2017).

Macro-algae-formed habitats are known to be highly productive and therefore contribute to a substantial biomass input into the ecosystem, which is expected to be used by bacteria, detritivores, grazing benthic species and forage fish (Smale et al., 2020). Although macroalgal biomass production was well integrated by bacteria and detritivore species, few macroalgal grazing benthic species and forage fish were present on the Cherbourg AR. Therefore, the biomass produced was accumulated or exported through the phenomenon of out-welling, which has contributed to the transfer of carbon stocks to the deep offshore seabed (Kraufvelin et al., 2010). The age of the AR is also a potential explanation to the low transfer of production from macroalgae to higher trophic levels. Previous studies have shown that artificial structures become stable and have a significant impact on the ecosystem after 6 to 30 years (Cépralmar, 2015; Pitcher et al., 2002). Therefore, the Cherbourg AR may not have reached its stabilisation phase, which allows biomass transfer to higher trophic consumers.

8.2.2 Increase in the ecosystem maturity

According to Odum (1969), ecosystems evolve towards maturity in a process that involves structural changes that are orderly, directional and predictable. Various metrics can be used to identify the maturity of the system:

- The ratio of total primary production/R (PPt /R) is a functional index of ecosystem maturity and is expected to be greater than 1 in immature systems and tends towards 1 in mature systems (Odum, 1969).
- The ratio of total biomass to total throughput (Bt/T..) is higher in a mature ecosystem (Odum, 1971).
- The Ascendency (A) values are higher in a mature system, whereas low values indicate an immature system (Ulanowicz, 1997).
- Finn's Cycling Index (FCI) is the ratio of recycled flow to throughput (Finn, 1976). A high recycling index usually indicates a mature ecosystem (Holling, 1973).

Analysing the trends of these metrics in the models before and after ARs implementation in Cherbourg shows that:

-the estimated PPt/R values of the two Cherbourg models were greater than 1, indicating that they have not yet reached a mature stage. However, the PPt/R values were lower in the model representing the ecosystem after the implementation of ARs, indicating a more mature ecosystem.

- the B/T.. value after ARs implementation is higher than before, confirming that the ecosystem tends to be more mature after ARs implementation.

-the Ascendency trends and the FCI index show an increase, confirming that the ecosystem tends to be more mature after the ARs implementation.

All these results are in line with previous studies that also highlighted an increase in maturity after ARs implementation through the ecosystem metrics mentioned above (Guan et al., 2016; Xu et al., 2019; Wang et al., 2019; Gao et al., 2022; Yuan et al., 2022; Salaün et al., 2023). In addition, according to the scenarios simulated by Raoux et al. (2017), 30 years after the construction of the Courseulles-sur-Mer offshore wind farms, the total system throughput, ecosystem activity, recycling and maturity were expected to increase in response to the reef effect.

8.3 Using the metrics to verify AR habitat similarity to natural ecosystem

To test whether ARs provide similar habitat quality to the natural ecosystem, studies typically compare metric values from these two systems. As no relevant studies using Ecopath with Ecosim in a natural reef system were available, we illustrate how ENA metrics could benefit this type of assessment by comparing metrics from the system before AR, characterised by fine sand with other natural ecosystems with sand sediment (Appendix E; Table 3).

The structure and functioning of the Cherbourg harbour system before the implementation of its AR showed similarities with others, such as the Seine Bay or other French coastal systems, but it also revealed some specificities (Pezy et al., 2017; Pezy et al., 2020; Selleslagh et al., 2012; Selleslagh et al., 2012; Guan et al., 2016). The system prior to the implementation of its AR was characterised by a low number of flows and low living biomass, resulting in low system activity, similar to the Mont St Michel (Pezy et al., 2017). However, system omnivory index and recycling were not comparable and could be explained by the difference in hydrodynamic exchange (the Mont St Michel Bay has the second highest tidal range in Europe), and the high biomass of invasive species such as Crepidula fornicata, which had led to a trophic deadlock without transfer of primary production to a higher trophic level (Leloup et al., 2005). Despite that large number of the gastropod was also recorded in the centre of Cherbourg harbour (129 ind/0.3 m2), causing the deposition of fine particles at this station (Baux et al., 2017).,the Cherbourg system reflects more the Seine estuary system, in being known as a complex structure and a more mature system (Selleslagh et al., 2012). However, the Cherbourg site has a morphological peculiarity, that of being a semi-enclosed area and this feature could influence the trophic exchange with open waters.

It is clear that comparing different systems must be handled very carefully, because on the one hand, some metrics will be influenced by the number of functional compartments, such as the ascendency, the production ratio, the respiration ratio or the flow ratio (Baird and Ulanowicz, 1993), but on the other, some metrics will directly affect how others evolve. For example, the resilience has a higher value in a complex system (Rooney and Mccain, 2012), and a decrease in efficiency leads to a decrease in D/H FCI and then a decrease in assimilation (Safi et al., 2019). The maturity described by Odum (1969) is also given as being related to the complexity and the FCI of the system, suggesting that complexity and FCI increase with maturity (Christensen, 1995). Therefore, to avoid analysing a change in metric value due to model structure, it would be more relevant to compare the trends between two models built with the same number and type of functional compartments.

To this end, we examined two case studies comparing a natural system (N) with an artificial reef (AR) with the same model characteristics in the Yellow Sea, China. The metric trends generally show that ARs seem to have a similar positive influence on complexity, recycling capacity and resilience as natural ecosystems (Table 4). However, ARs seem to have no clear influence on the increase of flow diversity, efficiency, robustness and maturity. The robustness has to be considered from a 'system vitality' point of view, aiming for an optimal balance between resilience and efficiency that evolve in opposite directions: resiliency increases with diversity, while efficiency increases with specialisation that would weaken the system (Fath, 2015; Goerner et al., 2015).

The Robustness of the N and AR system in Haizhou Bay (China) shows differences (Appendix D, Figure 4): the ARs systems have a higher efficiency than the Ns one. This must be interpreted carefully, as the N and AR systems compared do not have the same environmental characteristics, with differences in hydrodynamics fishery management (Gao et al., 2022).

Table 3 : The evolution of properties after artificial implementation (+: increase; -: decrease; =: no variation; +/_: mixed variation some indicator showed an increase while others showed a decrease; /: no analysis), N: Natural ecosystem; AR: Artificial Reef. The table compares these changes between the natural ecosystem (denoted as 'N') and the artificial reef (denoted as 'AR'). This comparison aims to provide insights into how the artificial reef influences the surrounding ecosystem's properties, helping to assess the ecological impact of artificial reef implementations.

Model comparison	*N/ARs Haizhou Bay (China)*	*N/ARs Yellow Sea (China)*	*Before/After ARs Cherbourg (France)*	*Before/After ARs Landes (France)*	*Before/After ARs Layzhou Bay (China)*	*Before/ Simulation of After ARs Tianjia harbour (China)*	*Before/ After OWF Jiangsu coast (China)*	*Before/ Simulation of After OWF Courseulles-sur-Mer (France)*
Source	Gao et al., 2022	Zhang et al., 2022	This study	Salaün, 2023	Xu et al., 2019	Guan et al., 2016	Wang et al., 2019	Raoux et al., 2017
Complexity	+	+	+/-	-	-	+	-	+
Flows diversity	-	/	+/-	+	/	=	/	+
Recycling capacity	+	+	+	-	/	/	/	+
Resilience	+	+	+	=	+	=	+	+
Efficiency	+	=	+	+	+	=	-	=
Robustness	-	/	+	+	/	/	/	/
Maturity	+	-	+	+	+	+	+	+

Great care should be taken when analysing the results and conclusions of the N/AR comparison, as it is difficult to find two areas with comparable environmental conditions, and these differences may confound the results (Zhang et al., 2022). Nevertheless, when all the considerations discussed previously are taken into account, the ENA metrics are able to indicate whether ARs provide ecosystem properties that are similar to those of natural ecosystems.

8.4 Using the metrics to determine ecological benefit trends

This type of evaluation is similar to the previous one in that it uses comparisons to identify changes in system characteristics, but this time it focuses exclusively on comparing between different ARs.

We compared the Cherbourg AR with two other ARs models located in Landes (France) and China. The comparison showed that the structure and operation of the Cherbourg harbour system after the implementation of ARs had similar SOI functionalities to other models (Annex E, Table 3). The production ratios were also greater than 1 in three cases, including this study, in line with the characteristics of immature systems. The fluxes-within the systems- and the ascendency were slightly lower in the Landes system with 413 gC.m-2.year-1 and 465 for the ascendency. The main difference was the Landes system had a higher recycling value than Cherbourg (FCI=18%). This difference could be explained by the high production of macro-algae in the Cherbourg system, which is absent in the Landes. Indeed, the high production of macroalgae leads to an increase in detritus, which is recycled and used within the system (Newell, 1984). From an all-round point of view, the Cherbourg and the Landes systems show some immature characteristics because of the low flows and the low upwelling compared to other coastal systems - in the case of China, the upwelling was not mentioned and the flows in the system were calculated differently, making the value comparison impossible (Appendix E, Table 3).

However, these results should be viewed from the initial characteristics point of view before the implementation of the ARs. In fact, ARs are a coastal development tool that are used more often in ecological engineering projects, with an aim to restore fragile environments and to provide services (fishing, diving...), as well as to regulate flows in the system (storage of CO2 water cycle...) (Pioch et al., 2018). For this reason, their preferred location is in coastal areas with little or noecological functionality (few food sources, little shelter, few suitable nursery habitats). As was the case for the Cherbourg site (before AR), which was characterised by low maturity and low trophic flux. Therefore, in order to incorporate this into the assessment, we propose to use a before/after comparison.

Several studies have investigated the before/after evolution using a trophic network model and have focused on artificial structures such as ARs or offshore wind farms. As the metrics clearly depend on the model's structure, it's preferential to use similar models to assess the effect or evolution of artificial structures within the ecosystem, i.e. with the same number of compartments and the same composition of these compartments (Prato 2016; Fath et al., 2019; Christensen et al., 2008). By excluding potential structural differences and concentrating on the detail in the comparisons it is possible to highlight the similarities which emerge from the implementation of ARs.

Looking at the before/after studies, artificial structures seem to have no clear influence on the increase of system complexity, flow diversity and recycling capacity, while resilience and system efficiency seem to increase in the majority of our selected studies (Table 4). These characteristics have to also be considered from a 'system vitality' point of view (Annex E, Figure 4). In the studies before and after the implementation of ARs in Cherbourg and Landes, their implementation increases the robustness of the system in both. In Cherbourg, despite the large increase in resilience (+42%), the increase of efficiency (+9%) seems to be more valuable to make the system more robust (Supp Mat D, Figure 4). The last characteristic to analyze is the maturity of the system and in all cases the maturity of the system increases with the implementation of ARs (Table 4).

All results indicate that the implementation of ARs induce an increase in system efficiency and maturity but that is not strictly related to an increase in system complexity. Several reasons could explain these contradictions which future research could improve. Firstly, the metrics used to assess system maturity

(Odum's metrics) are numerous and unweighted. This results in a prior selection of metrics to describe system maturity by scientists. Secondly, the metrics trend, used in this study, is relative and only describes the rise or fall of the metric values without giving any precision to their sensitivities which effectively highlight even the smallest variations. Finally, every parameter input into the trophic modelling can influence the results. It is rarely possible to observe all the data in the field and in some cases large amounts of data can only be estimated. The system is therefore a mixture of model and simulation. The before/after analysis has the advantage of providing two similar systems for comparing trends in metrics, and trophic modelling offers the possibility of representing complex interactions in a simplified way (Ulanowicz, 1986). Therefore, this analysis provides useful metrics for understanding how the ecosystem responds to the implementation of ARs (Prato et al., 2014).

8.5 **Identifying Appropriate Metrics for Various AR objectives**

The metrics highlight which functional and structural properties have been changed by the implementation of ARs. These metrics can be directly used by AR managers and stakeholders to control the ecological efficiency through "feedback" (Denis and Henocque, 2001).

For the record, ARs are installed mainly to support fishery production, to develop coastal activities such as diving and to increase ecological functions (Salaün et al., 2020a,b). Production ARs are installed to develop fishery resources or specific target species, often of high trophic level (i.e. tuna, carangidae, etc.). In order to ensure ARs performance, the efficiency and complexity metrics can be followed. The efficiency metric will allow feedback on how biomass is transferred to higher trophic levels, while the complexity metric will monitor that the system provides a variety of prey for the fish. The AR manager may also be interested in monitoring the sustainability of the fishing area (Table 5). Thus, they can control the resilience and robustness characteristics of the ARs system.

Recreational ARs focus on developing biodiversity and increasing fish abundance for scuba divers. Therefore, the ARs managers could assess the efficiency of the ARs using the system properties of complexity and flow diversity metrics. They can also use the maturity characteristics to assess the health of the system.

Finally, ARs installed for the purpose of developing or restoring ecological functions could be monitored using metrics that show variations in system properties such as resilience, robustness, maturity, etc. In this case, it seems interesting to monitor all the properties in order to have a general idea and to allow the stakeholder to adapt the management measures where necessary.

Table 4 : Keyboard of metrics for each AR objective (production, recreation, protection and "Eco-functional":). This table presents a selection of metrics tailored to specific artificial reef (AR) objectives, including production, recreation, production, and ecology. Each metric is evaluated for its relevance to these objectives. The table is structured to highlight which metrics provide the most informative insights into the system properties of each AR objective, indicated by a 'Yes' in the respective columns. Metrics listed within brackets are additional options that could be included to enhance the analysis or provide supplementary information. This structure aims to assist users in selecting the most appropriate metrics for assessing and monitoring artificial reef performance in relation to their specific goals

General ARs objective	*Production*	*Recreational*	*Protection (Increase ecological functionalities by protection)*	*"Eco-functional": Restore ecological functionalities*
Example of ARs location	Haizhou Bay (Gao et al., 2022)/ Layzhou Bay (Xu et al., 2019)	North and South America, Australia and Israël (Spanier, 2000; Baine, 2001; Lee et al., 2018)	Landes (Salaün et al, 2022, 2023)	Cherbourg (this study)/ Tianjia harbour (Guan et al., 2016)/Yellow Sea (Zhang et al., 2022)
Complexity (SOI, CI)	Yes	Yes	Yes	Yes
Flows diversity (MPL, H)		Yes	Yes	Yes
Recycling capacity (D/H)			Yes	Yes
Resilience (FCI)	(Yes)		Yes	Yes
Efficiency (TE)	Yes		Yes	Yes
Robustness (A/C)	(Yes)		Yes	Yes
Maturity (PPt/R; PPt/B; B/T..)		(Yes)	Yes	Yes

These metrics would allow managers in charge of ARs to have a command-and-control keyboard to identify what has worked, what has not yet been achieved or what has failed, in a targeted and precise manner.

In this study, we have tried to propose the most appropriate metrics for each ecological objective in order to assess their efficiency. However, there exists a multitude of ecological metrics (AZTI Marine Biotic Index, Bentix, Benthic Quality Index, Benthic Opportunistic Polychaetes Amphipods ratio etc) that are also available depending on the analysis, but few of them have been adapted to hard bottom and ARs (Taormina et al., 2022). The plethora of metrics makes the choice complex for managers and can hinder the clarity of results for administrations and funders. This study provides an initial overview of how the metrics from Ecological Network Analysis could be used by AR managers. They would then need to be agreed and incorporated into a national or international strategy in a balanced and satisfactory way to capture trends that could be adjusted if necessary.

9 Conclusion

Artificial reefs have been identified as a potential solution to mitigate or repair environmental degradation, and their number will increase mechanically in the coming years due to coastal development and pressure. Their assessment is an essential step to evaluate their efficiency and verify their effectiveness in addressing these issues and maintaining healthy ecosystems. As ARs aim to restore or supplement natural habitats, their assessment must include a wide range of ecosystem components to accurately determine the potential impact of these structures on the ecosystem.

Using the case of the Cherbourg ARs, this study demonstrates that trophic network modelling (EwE) provides relevant metrics for assessing the net contributions of ARs to ecosystem structure and function. We selected EwE metrics, based on studies highlighting effective metrics for detecting ecosystem change, and applied them to the ARs. These metrics reveal the overall impact of ARs on ecosystem properties such as complexity, flow diversity, recycling capacity, resilience, efficiency, robustness and maturity. The results show that these metrics can identify intrinsic changes in the ecosystem; for example, the Cherbourg ARs increased the efficiency and maturity of the system. In addition, while ENA metrics are useful for comparison with natural ecosystems, caution is required due to their dependence on model construction. We recommend comparing metrics before and after AR implementation to identify and overcome structural differences between systems, and to highlight emerging and similar ecological characteristics.

These metrics provide a comprehensive set of tools for managers to evaluate the effectiveness of each AR objective under different environmental management measures. The evaluation process could enable more precise 'command and control' actions that in turn influence ecosystem outcomes. In addition, these metrics provide broad insights into the potential uses of ARs, and offer a pathway for continuous improvement in ecological monitoring and the evaluation and comparison of large-scale ecological restoration.

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