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Structural and functional changes in Artificial Reefs ecosystem stressed by trophic modelling approach: Case study in the Bay of Biscay

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Abstract :

Artificial Reefs (ARs) are commonly cited as a tool used for increasing fishery production and reinstating ecosystem functionalities. The assessment of AR deployment is mostly based on analysis of the evolution of fish communities. Recently, studies have focused on trophic modelling to understand the functioning of such ecosystems in a more holistic approach. Trophic models are able to address this concern by describing the interaction between species at different trophic levels and based on the quantification of energy and matter flows through ecosystems. These models allow the application of numerical methods - also called Ecological Network Analysis (ENA) - to characterize emergent properties of the ecosystem. Usually, ENA indices are proposed as indicators of ecosystem health as they are sensitive to different impacts on marine ecosystems. In the present study, the Ecopath software is used to build an ecosystem model composed of 23 compartments, from detritus and phytoplankton to mammals, to describe the situation “before” and “after” the deployment of ARs in the south of the Bay of Biscay along the Landes coast. In addition, ENA indices are calculated for two periods, “before” and “after” the deployment of artificial reefs, to compare network functioning and the overall properties of the structural trophic network. Our results show little structural change in the ecosystem, with a rearrangement of the trophic levels and a simultaneous increase in biomass and system maturity. These preliminary results evidently need to be confronted with other environmental factors such as, for instance, substrate composition, proximity to natural reefs and larval supply... Nonetheless, we consider that the maturity index could be used as a new indicator to assess the evolution of ARs with specific management objectives.

Keywords : Trophic network, Before/After approach, Maturity indicator, Eopath with Ecosim software

1 Introduction

Artificial Reefs (ARs) are human made voluntarily submerged structures created “to mimic certain functions of a natural reef such as protecting, regenerating, concentrating, and/or enhancing populations of marine resources” (FAO, 2015). Despite some controversial debates about the effects of production and concentration (Osenberg *et al.,* 2002; Pickering et Whitmarsh, 1997), ARs are commonly considered to be relevant tools for increasing fisheries production and supporting commercial or recreational fishing activities, if they are properly managed (Seaman *et al.,* 2000; Santos et Monteiro, 1997). Therefore, they are used worldwide with fish production being the main objective (Lacroix *et al.*, 2002; Jensen, 2002; Baine*,* 2001). Recently and to an increasing extent, ARs are also deployed to rehabilitate marine ecosystems (coral, rocky or algae substrata) and their functionalities (*e.g.* nursery, feeding or reproductive), or to mitigate the effects of anthropogenic impacts (Seaman, 2019; Patranella, et al., 2017; Pioch et al., 2011). Although the general objectives of AR projects are frequently defined in terms of production, protection or recreational activities, there is a lack of information on the precise objectives with specific indicators (Becker *et al.*, 2018, Claudet *et al.*, 2006). Thus, suitable criteria and quantitative indicators need to be developed to assess the attainment of AR objectives (Hammond *et al.*, 2020).

Despite the worldwide deployment of ARs and the increasing research on their design, performance and management, knowledge of their efficiency remains largely insufficient regarding the production and protection aspects (Lee *at al.,* 2018; Lima *et al.*, 2020). As their main goal is to enhance fish biomass, studies have focused predominantly on the variation of certain ecological components such as fish assemblages, abundance and species richness (Véron *et al.*, 2008; Folpp *et al.*, 2011; Neves dos Santos and Zalmon, 2015; Becker *et al.*, 2018). Moreover, ARs create new hard substrates to be colonized by sessile fauna and consequently provide new food resources that are non-pre-existent on soft bottoms (Baine, 2001). The feeding relations have been recently explored to demonstrate the real contribution to fish production as a function of the attraction effect of ARs using stable isotopic ratios to characterize the trophic network (Cresson *et al.*, 2019). These results open new perspectives using trophic analysis as a tool to understand the overall functioning of AR systems from the primary producers to the top predators, while providing original new metrics to improve the effectiveness assessment of ARs.

Trophic analyses were firstly developed to evaluate ecosystem-based management of fisheries (Polovina, 1984; Christensen and Pauly, 1992; Gascuel, 2019). For this purpose, models using the Ecopath with Ecosim software (EwE) have been intensively used and developed over the last three decades (Colléter *et al.,* 2015; Drouineau *et al.,* 2006; Chouvelon, 2011; Moullec, 2015; Guénette and Gascuel, 2012; Halouani, 2016). These joint trophic approach have been recently applied to coastal and marine systems to assess changes in their functioning in response to environmental perturbations such as Offshore Wind Farm (OWF), marine aggregates exploitation, harbour construction and dumping of dredged materials (Raoux *et al.,* 2017; Pezy *et al.*, 2017) as well as specific regulations for Marine Protected Areas (MPA) (Prato, 2016; Valls *et al.*, 2012; Wallmo and Kosaka, 2017; Fulton *et al.,* 2015). These studies based on Ecological Network Analysis (ENA) provide metrics that could be used to define the state of marine ecosystems and assess the effectiveness of conservative management tools. Hence, this approach has led to the development of indicators for stakeholders and decision makers, allowing them to build up and enrich Ecosystem Based Management (Safi *et al.*, 2019: Fath *et al.*, 2019). Applied to ARs, this would be an innovative approach to assess positive or negative changes in ecosystems associated with the deployment of ARs.

Other engineering infrastructures can act as ARs, such as shipwrecks, oil platforms or Offshore Wind-Farms (OWF). These artificial structures also induce an increase of fish biomass, species diversity and provide shelter against predators. Several surveys on fish and macro-invertebrates indicate that these structures also give rise to reef effects (Glarou *et al.*, 2020; Ajamian *et al.*, 2015; Picken *et al.*, 2000). Therefore, this innovative approach, which consists of comparing the state of an ecosystem before and after a few years of AR deployment by using trophic modelling, could provide an effective overview of the ecological effects of ARs and other artificial structures on the marine ecosystem (Conner *et al.,* 2016; Raoux *et al.,* 2017).

The present study tries to apply the trophic framework approach to validate that ARs can contribute to enhancing an ecosystem. In this study, we build Ecopath ecosystem models, composed of 23 compartments, ranging from detritus and phytoplankton to marine mammals, to describe the situation "before" and "after" the deployment of ARs in the southern part of the Bay of Biscay along the Landes coast. These ARs were deployed by an association named "Atlantique Landes Récifs" with the aim of creating a protected area for growing fishes after observing a decrease of catches along the coast (ALR, 1998). For this purpose, all marine activities are prohibited on the site and three types of AR have been deployed to offer refuges, habitats and food supply for demersal and pelagic fishes. The hypothesis tested in this study is that ARs modify the structure and functioning of the trophic network. Moreover, our study focuses on the identification of emergent properties that evolve with the deployment of ARs and proposes the use of ecological indicators to monitor the progress of AR projects in reaching their objectives.

2 Materials and methods

2.1 Study area

In France, ARs were deployed since 1968 throughout more than fifty sites, firstly in the aim to develop or protect fisheries resources and, since a decade, to restore marine ecosystems (Salaün *et al.*, 2022b). The study area is located in the south of the Bay of Biscay, a large gulf on the French Atlantic coast characterized by a continental shelf that decreases in width from the North (150 km) to the South (12 km) (Borja *et al.*, 2019). The south of the continental shelf is incised by the Capbreton Canyon whose head is situated only 250 m off the Landes coast (Mazières *et al.*, 2015). The study area is exposed to strong swell that transports around 1,000 m^3 per year of sediment to the south (Abadie *et al.*, 2006). Spring upwelling occurs in the area (Planque *et al.*, 2004) and the sediment habitat correspond to fine sand with wave influence (Borja *et al.*, 2019) associated with *Nephtys cirrosa* benthic communities (after Monbet, 1972). Boreal and subtropical fish species are distributed across this rich ecosystem (Authier *et al.*, 2018) and top predators such as marine mammals and seabirds are attracted (Planque *et al.*, 2004; Sanchez and Santurtun, 2013).

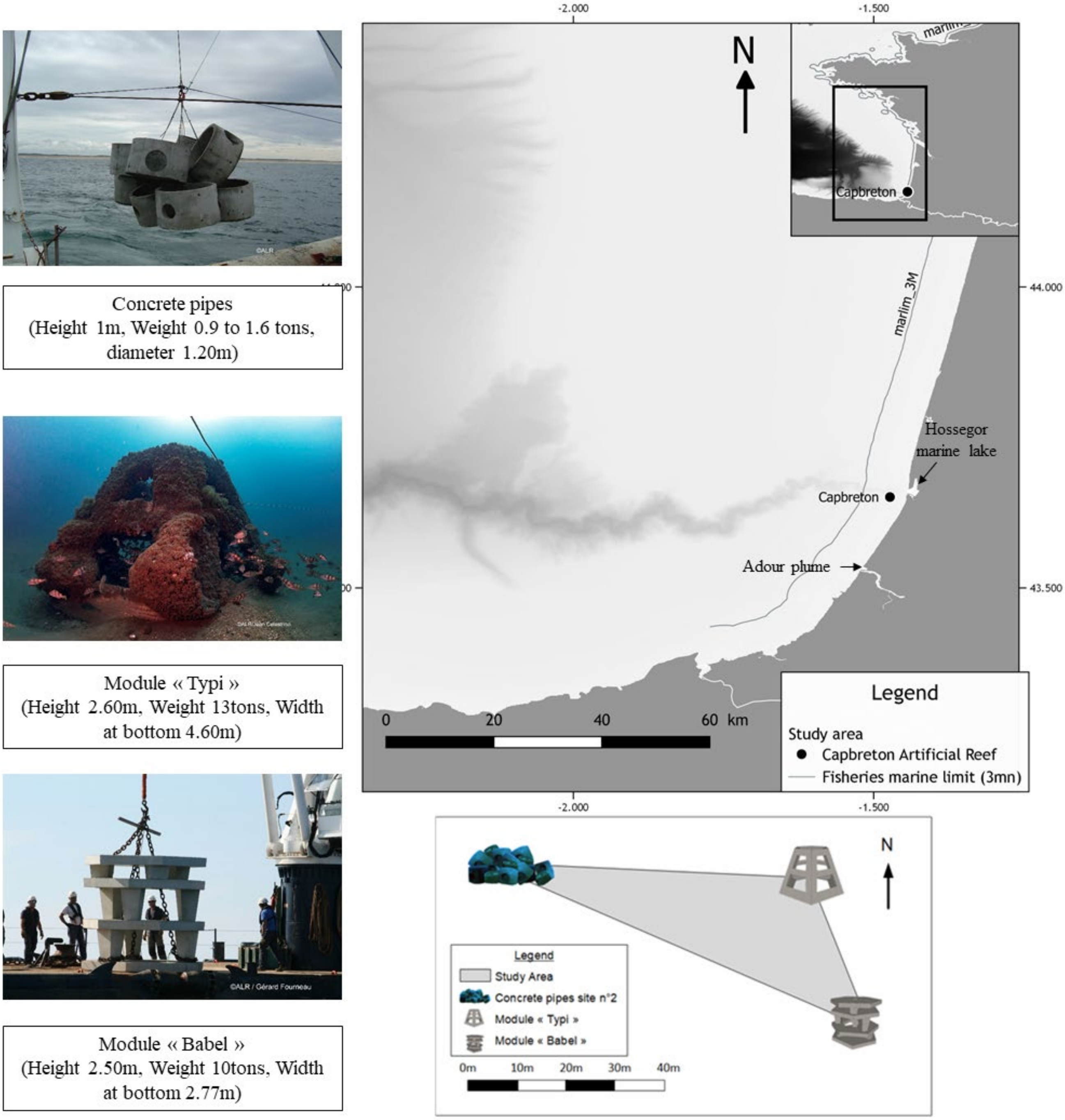

Figure 1 : Location of the Capbreton Artificial Reef study site and photographs of the three artificial reefs modules forming a triangle

ARs have been implemented off Capbreton to create hard bottom habitats for fisheries production (Salaün *et al.*, 2022); this location was chosen for many reasons, including the proximity of Capbreton harbour and the coastline (2.2 km offshore on a sandy bottom at 20 m depth), the gentle slope of the continental shelf (<0.8 %) and the supply of organic matter from the Hossegor marine lake and the Adour plume (Biosub, 1999; Mazière *et al.*, 2015) (Figure 1). Three types of ARs were deployed by the association "Atlantique Landes Récifs", with clusters of concrete Bonna® pipes being emplaced in 1999 at three sites of around 200 m² each. The "Typi" modules were deployed in 2010 with a 11 m² footprint and the "Babel" modules in 2015 with a 5 m² footprint (Figure 1). Rapidly, two ARs sites with Bonna pipes were buried. The study focused on the three remaining ARs sites that forms a triangle covering an area of 900 m², with one peak corresponding to site n°2 with clusters of concrete pipes, and the other two peaks corresponding to the sites with Typi and Babel modules. Also, the total ARs surface footprint covered 102 m², the total surface colonized represented 3 656 m² and the volume formed were 830 m^3.

2.2 Trophic network modelling framework

In this study, we use the Ecopath with Ecosim (EwE) approach and software (Polovina, 1984; Christensen and Pauly, 1992) to model the trophic network before and after the placement of three ARs in the Capbreton Artificial Reef study site. Thus, the two Ecopath models called BAR (Before AR deployment) and AARs (After AR deployment) based on data collected on the three artificial reefs of the Capbreton study site were created. To ensure statistical robustness a minimum of five years' data for each model were addressed. Therefore, the BAR model covered the period from 1997 to 2002 and the AAR model covered for ten years from 2010 to 2020. Despite the overlapping of the BAR period and the first AR deployment occurring in 1999, the data considered to represent this period were carefully selected to minimize the possibility that ARs deployment had an influence on it.

Ecopath modelling is based on functional groups that constitute the trophic network units. A functional group could include several species or individual ones that have similar habitat and ecological characteristics (growth rates, diets, predators, consumption). Whereas detritus group is essential, the number of groups are not limited (Heymans *et al.*, 2016). The Ecopath model requires inputs for each functional group that are information on the biomass, the diet composition and other ecological parameter such as the Production-Biomass ratio, the Consumption-Biomass ratio and the Ecotrophic Efficiency. The inputs are based on the individual species information (Step 1 in Figure 2) that are weighted by their relative biomass to calculate a single functional group parameter (Step 2 in Figure 2). Then, the models are analysed and compared using ecological index (Step 3 in Figure 2). However, the index is related to the model structural parameters, such as the number of groups, that could make comparisons between models difficult to interpret (Pinnegar *et al.,* 2005).

To make the static Ecopath models more comparable, we chose to fix the number and the type of the functional groups, but the species including in these groups could be different over models. Thus, we selected 23 functional groups based on heightened species interest referenced in our study site (e.g., commercial species, cultural value species, reef species) that range from detritus and phytoplankton to top predators such as marine mammals or diving seabirds. Whereas ecological parameters are relatively easy to estimate with literature values, the biomass is more space related. We chose to focus our sampling strategy on the local biomass data acquisition considering our allocated time and funds.

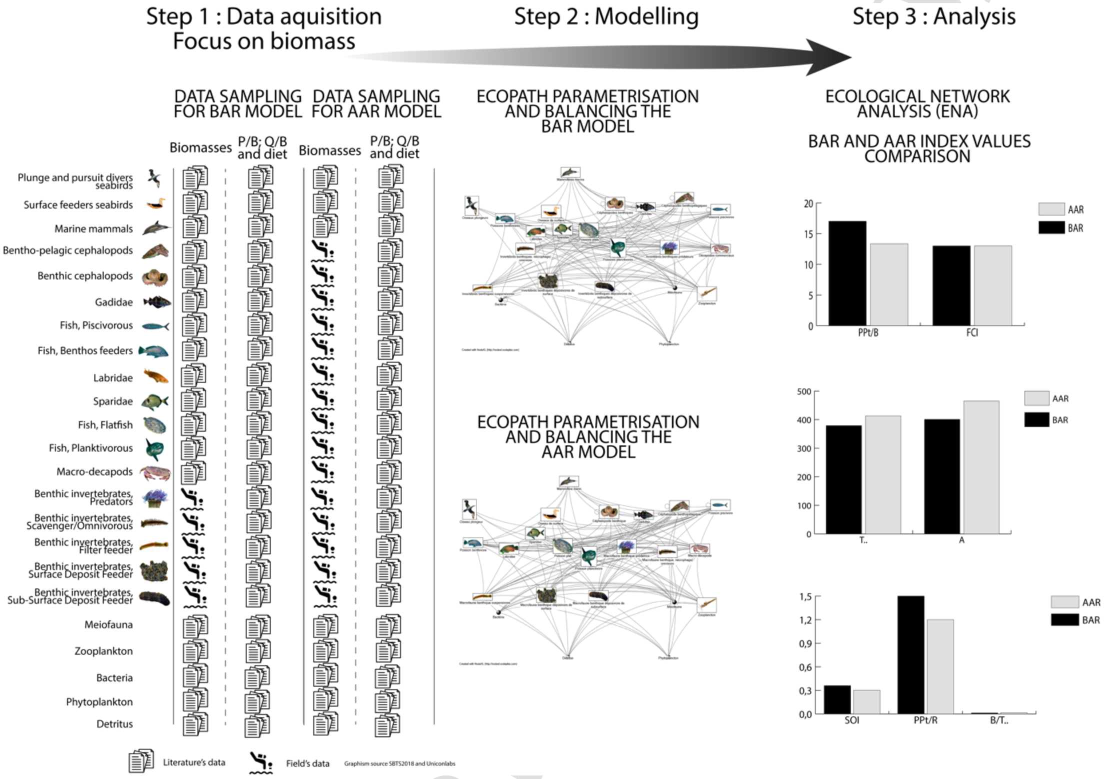

Figure 2: Trophic network modelling framework apply to model before and after the ARs deployment in Capbreton

2.3 Biomass sampling strategy

The biomass data intended to represent the stable state of relationships in the ecosystem before and after ARs deployment and overview their ecological effect. The biomass sampling strategy were made to this end: (1) on-site collection of fish and benthic invertebrates of soft bottom and hard substrate was privileged, as they are the faunal communities the most impacted by ARs deployment (Fabi *et al.*, 2006); (2) the biomass data for the other fauna groups were taken from the literature (details given on Table 1); (3) the same biomass data were used for meiofauna, zooplankton, phytoplankton and bacteria groups before and after ARs deployment because ARs deployment was assumed to have no effect on their biomass and production (Miller and Falace, 2000) and (4) the same biomass data were used for marine mammals and birds groups due to the low surface-area of ARs (in m²) compared to their predation areas (in km²), ARs were assumed to have little influence on the biomass of these top predators (Castège and

Milon, 2018). As a result, other external factors that could influence the parameter fluctuation such as temperature growth were limited and top-down and bottom repercussion due to ARs deployment were strengthened.

The biomass sampling was carried out using multi-tool surveys:

- Soft bottom benthic fauna sampling (Before and After AR deployment);
- Hard bottom benthic fauna sampling (on Artificial Reef);
- Underwater visual census of fishes, macro-decapods and cephalopods (only After AR deployment).

The annual average representation of exchanges in the ecosystem was made with available data covering a minimum of five years as possible (Table 1). To be able to compare data from various numbers of samples and different method, all data were express into mean annual biomass by square meter implemented on the surface triangle (900 m²).

Table 1: The periods and locations of the data collected for the functional groups included in the two models (BAR: Before Artificial Reef and AR: After Artificial Reef)

	BAR	**AAR**
	Periods and location for biomass data	Periods and location for biomass data
Plunge and pursuit divers' seabirds	1999-2004 (Bay of Biscay surveys)	
Surface feeders seabirds		
Marine mammals		
Benthopelagic cephalopod	1997-2002 (South of Bay of Biscay surveys)	2010-2020 (this study coupled with older ARs surveys)
Benthic cephalopod		
Gadidae		
Fish piscivorous		
Fish, benthos feeders		
Labridae		
Sparidae		
Fish flatfish		
Fish planctivorous		
Macro-decapods		
Predators	2000 (Capbreton survey)	2019-2020 (this study)
Scavengers /omnivorous		
Filters feeders		
Surface Deposit Feeders		
Sub surface Deposit Feeders		
Meiofauna	1981 (Galicia survey)	
Zooplankton	1999-2006 (Bay of Biscay surveys)	
Bacteria	1994-2002 (Bay of Biscay surveys)	
Phytoplankton	1999-2000 (Bay of Biscay surveys)	
Détritus	1994-2002 (Bay of Biscay surveys)	

2.3.1 Soft bottom benthic fauna sampling

Hand box-corer samplers and a grab of 0.02 m² were used to sample the soft bottom benthic fauna. The corer used is 25 cm long and has a diameter of 16 cm, with a cap to limit loss of material. The opening of the grab is 20 cm long and 10 cm wide.

The first soft bottom benthic fauna sample campaign was conducted by scuba-divers in May 2000, just after the first deployment of ARs, on 12 stations located along the four cardinal directions at distances of 1 m, 5 m and 10 m from the AR site (Ferrou, 2000; Figure 3). Two replicates of 0.02 m² each were sampled with the hand box-corer at each station. A grab was operated for the furthest stations at 30 m from site n°2 for one sample (Ferrou, 2000). To be able to use these data in the BAR model, we selected only those stations furthest from the AR placement site (5m, 10m and 30 m). This allows us to avoid considering the recent influence of ARs on the soft bottom benthic community mainly concentrated around the AR footprints (Créocéan, 2008). Therefore, the total sampling effort used for the BAR model corresponds to a coverage of around 0.4 m².

The second soft bottom benthic fauna sample campaign was conducted by scuba-divers in September 2019 and 2020 on 12 stations located along each cardinal direction 20 m from the three ARs (Figure 3). Between three to six replicates of 0.02 m² were collected at each station with the hand-box corer. The total sampling effort was around 0.9 m² and the collected samples were used for the After Artificial Reefs model (AAR).

To ensure comparison, the data collected for BAR and AAR model were expressed into mean annual biomass by square meter implemented on the surface triangle (900 m²) using the method described below.

The sediment collected was sieved through a 0.1-mm mesh. All samples were preserved in 10 % formaldehyde solution before being sorted. The species were counted under a binocular microscope and identified at the lowest taxonomic level needed to classify them into functional groups. The biomass of species collected for the BAR model was calculated based on the abundance of the benthic species present and their individual weight referenced by Pezy (2017). The species collected for the AAR model were placed in a drying oven at 60°C for the duration needed to ensure drying of the sample (around 48h to 96h). Then, the samples were weighed to determine dry weight (DW) before being put into another oven for 5 hours at 500°C. The ash-free dry weight (AFDW) was obtained by subtracting ash weight from DW. For benthic invertebrates, the biomass was converted from AFDW to carbon content using a conversion factor of 0.518 (Brey, 2001). Then the biomass was reported on the studied triangle site (900 m²).

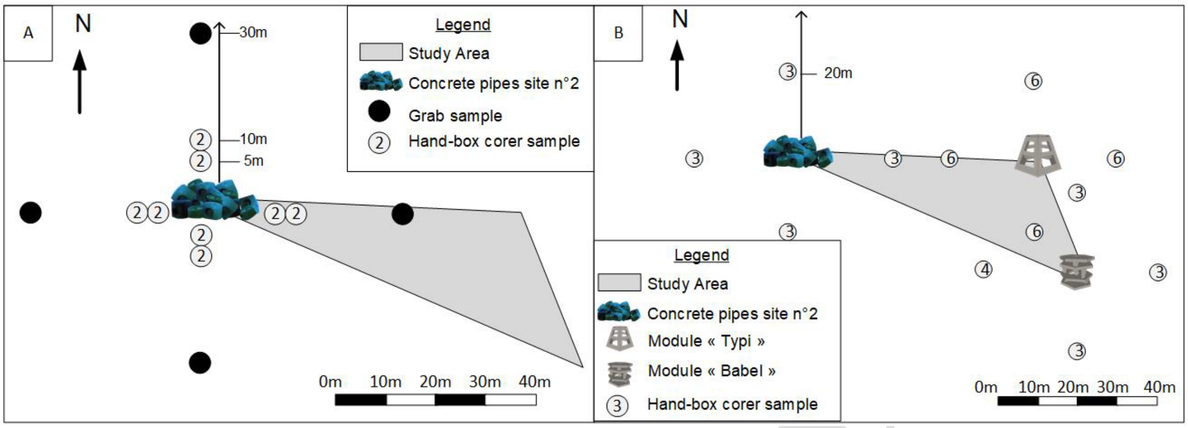

Figure 3: Number of soft bottom benthic fauna sampling in 2000 (A) and 2019-2020 (B)

2.3.2 Hard bottom benthic fauna sampling

The Scraping method is commonly used to analyse benthic fauna communities on hard substrates (FAO, 2015) and was applied in this study to ARs. The difficulty of this technique is to be able to collect all organisms, especially those of small size, when there is underwater current (FAO, 2015).

Samples of benthos were collected in summer 2019 and winter 2020. Six scrape samples at different AR positions (inside, below, North, East, West and South) were collected by divers on each AR module type, using a quadrat sampler of 20 x 20cm for 2019 and 30 x 30cm for 2020, a putty knife and a net bag (<1 mm mesh size). A total area of 0.553 m² was sampled in summer 2019 and 0.819 m² in winter 2020, thus providing mean annual biomass for the After Artificial Reefs model (AAR).

All samples were preserved in 10 % formaldehyde solution before being sorted. The species were counted under a binocular microscope and identified at the lowest taxonomic level needed to classify them into functional groups. The mean annual biomass was then obtained using the AFDW determined from the sampled surface for each species extending to the entire ARs colonisable surface (3 656 m²) and was converting to carbon content using a conversion factor of 0.518 (Brey, 2001). To ensure a comparable data on two dimensional, the biomass obtained was then reported on the total ARs surface footprint (102 m²) and then on the studied triangle site (900 m²).

2.3.3 Fishes, macro-decapods and cephalopods underwater visual census

Underwater visual census (UVC) is commonly used as a non-destructive survey method for assessing fish assemblages (Kulbicki and Sarramégna. 1999). This method was adapted to ARs using both stationary point and belt transect counts to record fast moving species and then benthic and cryptic species (Cresson *et al.*, 2018; Lowry *et al.*, 2011, Charbonnel *et al.*, 1997; Labrosse *et al.*, 2011). The survey covered the surface of ARs footprints. UVC campaigns were conducted when possible each year between May to September on each ARs. For the AAR model, we selected 120 counts carried out during the last ten years: 58 on the concrete pipes at site n°2 (2010, 2011, 2012, 2013, 2015, 2016, 2018 and 2019), along with 49 on the Typi (2010, 2011, 2012, 2013, 2015, 2016, 2018, 2019 and 2020) and 13 on the Babel module types (2015, 2016, 2018, 2019 and 2020). This selection that covers several years allows a better representation of the mean annual biomass of fishes, macro-decapods and cephalopods. The data was derived from underwater visual census observations of scientific divers and trained volunteer divers following the recommendations of Harmelin-Vivien *et al.* (1985). Abundances of populations were counted individually up to 10 individuals, whereas larger populations were estimated using abundance classes reviewed in the literature (11–30; 31–50; 51–200; 201–500; 500-1000; >1000 individuals). The total length of fish was evaluated in cm. The wet weight was then obtained using the length-weight relationship $W = a \times TL^b$, where W is the wet weight in grams, TL is the average total length of the size class in cm, while a and b are species-specific constants obtained from the data available in Fishbase (Froese and Pauly, 2019) and selected in the vicinity of the study area. As the survey covered only the surface of ARs footprints, the average biomass was calculated for each species (fish, cephalopods and decapods) using the footprint of ARs. Conversion factors of 0.192 and 0.402 were used to convert cephalopod wet weights into dry weights and then into carbon contents, respectively, while 0.35 was used for fishes and 0.518 for decapods (Brey *et al.*, 2010; Brey, 2001). To maintain the proportion between the surface covered by ARs and the triangle studied surface, the mean annual biomass obtained was then reported on the studied triangle site (900 m²) with no extrapolation.

2.3.4 Collected data from literature

According to the literature, two trophic networks on the continental shelf of the Bay of Biscay have already been established by Lassalle *et al.* (2011) and Moullec *et al.* (2017). Data were extracted from these models by preferentially selecting coastal data in the vicinity of the study area.

For the BAR model, the biomasses of fishes, macro-decapods and cephalopods were estimated from bottom-trawl surveys carried out by IFREMER since 1997 in the Bay of Biscay in the context of the EVHOE cruises for the West Europe fisheries evaluation (Evaluation Halieutique Ouest de l'Europe; Devreker and Lefebvre, 2018; Mahé and Poulard, 2005). Only the southeast coastal surveys were selected. In order to ensure a better statistical overview, the period selected for representing the state before ARs were extending to 2002. The distance from the trawl surveys selected after the first ARs deployment in 1999 and the study sites (around 65 km) allowed to consider the fish macro-decapods and cephalopods biomass were not influenced by the 1999 ARs deployment. The data captured from these thirteen selected trawl transects during the period 1997-2002 were averaged with respect to the survey surface-area over the five years selected (1997, 1998, 1999, 2001 and 2002). Conversion factors of 0.192 and 0.402 were used to convert cephalopod wet weights into dry weights and then into carbon contents, respectively, with 0.35 used for fishes and 0.518 for decapods (Brey *et al.*, 2010; Brey, 2001). The collection of data for top predators is derived from aerial strip-transect surveys, named ROMER and ATLANCET, conducted from 2001 to 2004 in the Bay of Biscay (Certain *et al.,* 2008). Annual average abundances were converted into biomass using weight referenced by species for sea birds and marine mammals (Spitz *et al.*,2018; Anonymous, 2008; ICES, 2000; Hunt *et al.*, 2005). Conversion factors of 0.3 and 0.4 were used to convert wet weights into dry weights and then into carbon contents, respectively, for sea birds (Lassalle *et al.*, 2011) and a coefficient of 10% is used to convert directly wet weights into carbon contents for marine mammals (Bradford-Grieve *et al.*, 2003).

Meiofauna data were selected from a site located in Galicia, Spain, where benthic habitat characteristics are closely similar to the Capbreton site (Tenore *et al.*, 1984). The biomasses of the benthic macrobenthic invertebrates were converted from AFDW to carbon content using a conversion factor of 0.518 (Brey, 2001).

Zooplankton data were taken from BIOMAN campaigns conducted in the Bay of Biscay from 1999 to 2006 (Irigorien *et al.*, 2008). The phytoplankton data used were acquired in the south of the Bay of Biscay as far as the 100 m isobath (zone known as “Gironde Interne”) for the period 1999-2000 (Lampert *et al.*, 2001). Then, the data were normalized to the depth of the study (20 m) and the chlorophyll-a were converted into carbon content using a factor 40 (Chardy and Dauvin,1992). The bacteria and detritus biomass were derived from the Ecopath model of the Bay of Biscay built by Lassalle (2011).

2.4 Trophic network modelling

2.4.1 Ecopath equation-based modeling

Ecopath is a mass-balance single-solution model that uses linear equations to estimate flows between a number of functional groups established *a priori* (Christensen and Walters, 2004). The parameterization of an Ecopath model is based on satisfying two equations. The first equation (Eq. 1) describes the production of each compartment in the system as a function of the consumption to biomass ratio (Q/B) of its predators (j), the fishing mortality (Yi, gC·m−2), the net migration (Ei; emigration – immigration, year−1), the biomass accumulation (BAi, year−1) and its natural mortality (1—EEi). The Ecotrophic Efficiency (EE) is the fraction of total production consumed in the system (by fishing activities or by predators). Its value can never exceed unity. (1-EEi) represents the fraction of mortality not explained by the model, such as mortality due to old age or diseases.

$$B\left(\frac{P}{B}\right)_i = \sum_j B_j \left(\frac{Q}{B}\right)_j DC_{ij} + Y_i + E_i + BA_i + B_i\left(\frac{P}{B}\right)_i (1 - EE_i) \qquad \text{(Eq. 1)}$$

The second equation (Eq. 2) ensures energy balance, calculating consumption of the ith group (Q) as the sum of its production, respiration (R), and excretion (U)

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

2.4.2 Ecopath model parametrisation

The two models developed in this study are composed of 23 functional groups or compartments ranging from seabirds and mammals to detritus. Seabirds are divided into two groups, according to their feeding strategies. The "plunge and pursuit divers" group is mainly composed of gannets and the "surface feeders" are mainly composed of gulls and kittiwakes. Marine mammals (*Delphinus delphis, Stenella coeruleoalba* and *Tursiops truncatus*) are placed together in one group. Cephalopods are divided into two groups: the benthopelagic group mainly composed of *Loligo vulgaris* and the benthic group mainly composed of *Sepia officinalis*. The model also comprises seven groups of fish (Gadidae, piscivorous, benthos feeders, Labridae, Sparidae, flatfish and planktivorous). Gadidae, Labridae and Sparidae are not aggregated with the other compartments to allow a detailed analysis of the potential impact of the reef effect on these three groups which also include commercial species. Benthic invertebrates are divided into six groups (macro-decapods, predators, scavengers/omnivores, filter feeders, surface deposit feeders and subsurface selective feeders). Finally, the model also comprises one group of zooplankton, one group of bacteria, one group of phytoplankton and one group of detritus.

The source data used for obtaining the model parameters (Biomass, P/B, Q/B, diet and conversion factor) are listed in Supplementary material (Table 1). The dietary preferences for multi-species groups are weighted by the relative biomass contribution of each species (Supplementary material, Tables 2 and 3). Besides, considering the study site as an open system, the diet import was added in proportion of time spent outside the system (Christensen and Walters, 2004).

The two models comparison aimed to reveal what modification on the structure and functioning of the trophic network ARs modify. The differences in the fishing activity between BAR and AAR model (all activities were restricted) conduct to add a MPA effect on the system. Therefore, to highlight mostly ARs effect, the fishing mortality in the BAR model represented by Y was not considered.

2.4.3 Balancing the Ecopath model

To equalize the mass balances, the input data to the models had to be manually and slightly calibrated. The balancing approach was top-down, starting modifications from top predators down to the lowest trophic levels. Balancing was performed taking into account the quality of the diet data source. Due to a lack of data, the biomass of planctivorous fish and flatfish were left to be estimated by the model after setting their Ecotrophic Efficiency at 0.95 (Christensen and Walters, 2004). In the same way, the biomass of macro-decapods and benthic invertebrate filter feeders were also estimated by Ecopath using an EE of 0.95. The consistency of the model was checked with the Ecopath PREBAL tool (Link, 2010).

2.5 Analysis of the ecosystem organization and maturity

Ecological network analysis (ENA) was performed to reveal the emergent properties of the trophic network using the plug-in included in EwE software (Christensen and Walters, 2004). Thus, for both models, we made use of the Total System Throughput (T..), which corresponds to the sum of all flows occurring in the system (Latham, 2006), and the System Omnivory Index (SOI), which provides a measure of the trophic specialization of predators in terms of trophic levels and an indicator of the structure and complexity of a trophic network (Libralato, 2008). We also calculated the Finn's Cycling Index (FCI), which represents the fraction of the flows in the system generated by recycling (Finn, 1980) and the Ascendency, which is a measure of the growth and the flow coherence of the system, integrating its size and organization (Ulanowicz and Abarca-Arenas, 1997; González *et al.,* 2016; Nogues *et al*., 2021). In addition, the maturity status of the ecosystems (Odum, 1969) was also assessed using the following ratios: the total primary production/total respiration (PPt/R), the total primary production/total biomass (PPt/B) and the total biomass/total system throughput (B/T..) (Christensen *et al.*, 2005).

The trophic level (TL) of each functional group (i) was calculated as the weighted average of the trophic levels of its prey (j), according to the following equation:

$$TL_j = 1 + \sum_{i=1}^{N} DC_{ji} TL_j$$

where DC_{ji} is the fraction of prey i in the diet of predator j.

It is noteworthy that the EwE software is a single solution model and statistical comparisons between models are not possible (Christensen and Walters, 2004).

3 Results

3.1 Functional group biomass profiles and trophic levels

Results show that, before the deployment of the ARs, Phytoplankton is the dominant functional group in the biomass, representing approximately 28 % of the total living biomass of the system (Table 2). The other major groups of the system are benthic invertebrates, scavengers/omnivores and bacteria, making up approximately 17 % and 8 % of the total living biomass, respectively (Table 2).

After the deployment of the artificial reef, the phytoplankton remains the dominant functional group of the total living biomass of the system, followed by the benthic invertebrate filter feeders (mostly composed of the barnacles *Balanus* spp.) and the predators of benthic invertebrates (mostly composed of the gastropod *Natica*), representing approximately 22 %, 19 % and 10 % of the total living biomass, respectively (Table 2). Results show that the total living biomass is higher after deployment of the artificial reefs. In fact, the total living biomass increases by approximately 28 % after deployment of the three artificial reefs on the Capbreton site. This increase of biomass is mostly due to the macro-decapods and benthic invertebrate filter feeders, whose biomass increases by a factor of approximately 5 and 4, respectively, after the installation of the ARs. From another perspective, the Labridae and benthos feeder fish are the functional groups that experienced the greatest proportional increases (increase by a factor of 10 and 7 respectively).

Three notable changes in the species composition of functional group occurred with the ARs deployment and modified consequently the Q/B data input. The fish piscivorous functional group biomass was dominated by 93% of *Trachurus trachurus* (Linnaeus, 1758) in the BAR model while in the AAR it was dominated by 72% of *Conger conger* (Linnaeus, 1758). The fish benthos feeder's functional biomass group was mainly represented by 47% of *Trachinus draco*, (Linnaeus, 1758), 24% of *Chelon ramada* (Risso, 1827) and 16% of *Mullus surmuletus* (Linnaeus, 1758) while with ARs the functional group biomass was composed by 90% of *Umbrina canariensis* (Valenciennes, 1843). The biomass of Sparidae functional group evolved from 67% of *Boops boops* (Linnaeus, 1758), to 27% of *Diplodus sargus* (Linnaeus, 1758), 29% *Diplodus vulgaris* (Geoffroy Saint Hilaire, 1817) and 37% *Spondyliosoma cantharus* (Linnaeus, 1758).

*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 ("before" (BAR) and "after" (AAR) the construction of the artificial reef). Major changes were highlighted in bold and dominant living functional group were indicated by *.*

	Biomass		**Trophic Level**		**P/B**		**Q/B**		**EE**	
Functional groups	**BAR model**	**AAR model**	**BAR model**	**AAR model**	**BAR model**	**AAR model**	**BAR model**	**AAR model**	**BAR model**	**AAR model**
Plunge and pursuit diver's seabirds	0.0001	0.0001	4.06	4.11	0.09	0.09	70.4	70.4	0	0
Surface feeders seabirds	0.0001	0.0001	3.93	3.94	0.09	0.09	74.94	74.94	0	0
Marine mammals	0.0018	0.0018	4.85	4.83	0.08	0.08	42.5	42.5	0	0
Benthopelagic cephalopods	0.0096	0.0037	4.31	4.17	2.71	2.71	14.54	14.54	0.86	0.95
Benthic cephalopods	0.0240	0.0371	4.10	4.06	3.5	3.5	15	15	0.68	0.95
Gadidae	0.0487	0.0409	4.03	4.00	0.75	0.75	5.1	5.1	0.86	0.98
Fish, piscivorous	0.1758	0.3065	3.98	4.20	0.6	0.55	**5.9**	**4.1**	0.90	0.98
Fish, benthos feeders	0.0246	**0.1705**	3.78	3.43	0.93	1.17	**7.71**	**3.96**	0.94	0.99
Labridae	$3{,}9 \times 10^{-6}$	$\mathbf{4 \times 10^{-5}}$	3.51	3.31	1.3	1.3	10.38	10.38	0.92	0.95
Sparidae	0.0263	0.0089	3.56	3.62	0.55	1.38	**2.45**	**6.05**	0.93	0.99
Fish, flatfish	0.0208	0.0672	3.53	3.36	0.78	0.88	3.01	3.28	0.95	0.95
Fish, planktivorous	0.2231	0.2670	3.15	3.15	1.09	1.092	7.73	7.73	0.95	0.95
Macro-Decapods	0.0322	0.1543	3.13	3.12	1.18	1.01	5.9	5.05	0.95	0.95
Benthic invertebrates, Predators	0.2920	0.6161*	3.16	3.22	2.35	2.1	11.75	10.5	0.96	0.95
Benthic invertebrates, Scv/O	0.8228*	0.2827	3.14	3.20	0.55	0.66	2.75	3.3	0.95	0.98
Benthic invertebrates, Filter feeders	0.3073	1.2196*	2.28	2.32	2.46	2.8	9.84	11.2	0.95	0.91
Benthic invertebrates, sDF	0.2690	0.3366	2.44	2.36	2.8	3.11	14	15.55	0.94	0.95
Benthic invertebrates, ssDF	0.2883	0.4126	2.22	2.22	3.12	2.6	15.6	13	0.92	0.91
Meiofauna	0.2642	0.2642	2.29	2.21	15	15	60	60	0.86	0.92
Zooplankton	0.3600	0.3600	2.15	2.15	11	11	52.38	52.38	0.96	0.90
Bacteria	0.3940*	0.3940	2.02	2.02	115	125	230	250	0.26	0.20
Phytoplancton	1.3800*	1.3800*	1	1	61.2	61.2			0.44	0.53

Detritus	2.8467	2.8467	1	1		0.47	0.52

In both models (BAR and AAR), the trophic levels (TL) of the functional groups range from 1 for primary producers and detritus to a maximum of 4.8 for marine mammals that could be considered as top predators in this area (Table 3; Figure 4). As mentioned below, TL 1 is composed of two groups (primary producers and detritus, as imposed by the model structure) and represents approximatively 54 % and 46 % of the total biomass in the BAR and AAR models, respectively. TL 2 is composed of six functional groups (bacteria, zooplankton, benthic invertebrate subsurface and surface deposit feeders and filter feeders) making up approximately 23 % and 29 % of the total biomass in the BAR and AAR models, respectively. TL 3 incorporates the major part of the fish functional group (such as flatfish, benthos feeders, planktivorous and Sparidae) and is composed of nine functional groups. It represents 16 and 18 % of the total biomass in the BAR and AAR model, respectively. Finally, TL 4 is composed of five functional groups in the BAR model and six functional groups in the AAR model. TL 4 corresponds to top predators and represents only 8 and 7% of the total biomass in the BAR and AAR models, respectively. Thus, TL2 is the trophic level contributing most to the total living biomass in both models.

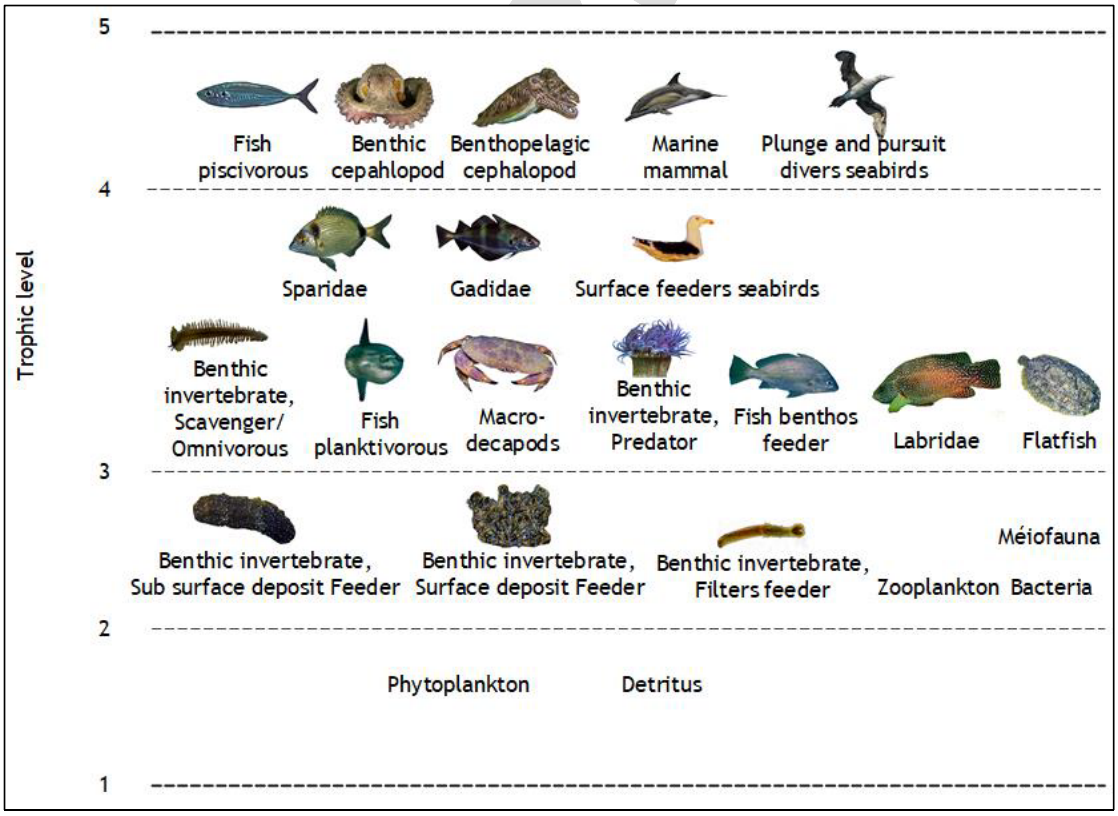

Figure 4: Trophic levels of the 23 groups in the AAR models

Table 3: Percentage of the biomass for each Trophic Level in the two models Ecopath models: "before" (BAR) and "after" (AAR) the construction of the artificial reef.

Trophic Levels	BAR	AAR
>TL 4	8 %	7 %
TL 3	16 %	18 %
TL 2	23 %	29 %
TL 1	54 %	46 %

3.2 Ecological Network Analysis and time-evolution of ecosystems

The results obtained from Ecological Network Analysis (ENA) show that the activity of the ecosystems studied here, as indicated by the Total System Throughput and Ascendency, is relatively lower than other estuarine French models as well as other coastal models (Table 4). The System Omnivory Index shows that, in both models, the trophic networks have a complex "web-like" structure before and after AR deployment (Libralato, 2008). Finn's Cycling Index obtained for both models points towards a medium recycling system and the PPt/R ratio suggests that the systems are immature. Any comparison between the ENAs of different models should be performed with caution because some indicators are specific to the topology of the model, such as the number of functional groups and the distribution of species (Heymans *et al.*, 2016). Table 4 presents the ENA of similar ecosystems characteristics, i.e. coastal and sandy sediment in order to place the results in context.

Table 4: Comparison of indices of network analysis for various French estuary ecosystems reef deployment put in context with other Ecopath models. N (number of functional group), Total System Throughput (T.., gC.m-2. Year-1); Ascendency (A, flowbits); System Omnivory Index (SOI, %), Finn Cycle Index (%), Biomass total (excluding detritus) (Bt, gC.m-2. Year-1), Total primary production/total respiration (PPt/R), Total biomass/total throughput (B/T..) and Total primary production/total biomass (PPt/B)

Ecosystem	BAR Landes Coast	AAR Landes Coast	Seine estuary (France)	St Michel bay (France)	Loire estuary (France)	Gironde estuary (France)	Galicia coast (Spain)	Lithuanian coast
Reference	This study	This study	Selleslagh *et al.*, 2012	Selleslagh *et al.*, 2012	Selleslagh *et al.*, 2012	Selleslagh *et al.*, 2012	Paradell *et al.*, 2020	Tomczak et al., 2009
N	23	23	15	19	19	18	23	12
T..	379	413	3603.22	376.00	635.35	744.30	/	900
A	401	465	3944.3	451.6	647.0	939.5	1239	1084
SOI	0.36	0.302	0.11	0.06	0.12	0.12	0.205	0.059
FCI	13	13	16.10	0.64	0.19	3.99	19.4	2.55
Bt	5	6.33	22.30	7.20	4.14	2.12	/	20.98

PPt/R	1.5	1,2	1.37	6.10	139.59	1.05	1.758	3.397
B/T..	0.013	0.015	0.01	0.02	0.01	0.003	0.017	0.023
PPt/B	17.01	13.35	38.26	24.60	76.26	21.52	18.34	18.351

4 Discussion

4.1 Structural comparison with natural reef

ARs have been mainly deployed to mimic the ecological functionalities of natural reefs and/or sustain artisanal fisheries (Pioch, 2008, Salaün *et al.*, 2022a). Over the past decade, studies of ARs have been focused on demonstrating their real contribution to the production of commercial fishes in the context of fish assemblage analysis; their similitude to natural reef assemblages has been defined as a goal to reach (Simon *et al.*, 2013; Perieira *et al.*, 2016; Streich *et al.*; 2018, Wu *et al.*, 2019). The results of this study showed that fish assemblages of ARs are dominated by piscivorous, planktivorous and benthos feeder fish (representing 86 % of the total biomass). This range of results is similar to other studies conducted in the Mediterranean Sea (Cresson *et al.*, 2017; Koeck *et al.*, 2014; Leitao *et al.*, 2013) but also in the Yellow sea, South-West Atlantic and the Pacific (Wu *et al.*, 2019; Hackradt *et al.*, 2011; Smith *et al.*, 2016; Hylkema *et al.*, 2020). However, despite the similarity with other fish assemblages in ARs system, it seems weak to use this indicator to assess the efficiency of ARs.

Firstly, in this study, the comparison with natural reef were not possible because the study site is surrounded by soft-bottom and the first natural reef is 20km away and represents a rocky shore ecosystem (Castège *et al.*, 2016).

Secondly, there is no consensus among the scientific community about using fish assemblages to indicate whether ARs are successful in acting as natural reefs. Some studies highlight the performance of ARs in providing the same assemblage as a natural reef over a short period of time (Wu *et al.*, 2019). However, other studies conclude that equivalence cannot be achieved on a long time scale (100 years) (Simon *et al.*, 2013). Besides, this criterion of performance seems to be influenced by other parameters such as the size of the ecosystem, as well as the localisation, substrate features and roughness of the habitat (Lopez de Oliveira, 2016). But remarkably, it seems that the distance between ARs and natural reefs does not impact the fish assemblage of ARs. (Simon *et al.*, 2013).

Therefore, to provide robust indicators, scientific studies highlight the need to use functional approaches to provide indicators to assess the effects of ARs on communities (Cresson *et al.*, 2014). In this way, trophic network studies, isotopic analysis and modelling approaches provide functional description of ARs system based on biomass evolution and this criterion could be used to assess the ARs productivity to support fisheries (Roa-Ureta, *et al.*, 2019; Cresson *et al*, 2019; Smith *et al.*, 2016; Mavraki *et al.*, 2021).

4.2 Functional evolvement using biomass indicators

The deployment of three types of ARs along the Landes coast offered new hard substrates for sessile fauna, notably invertebrate filter feeders, thus promoting their development within the ecosystem (Raoux, 2017; Cresson, 2013). The benthic community of the BAR system is mainly composed of benthic detritivorous species (41 %) with a small proportion of filter-feeder organisms (16 %). The fish assemblage is dominated by planktivorous and piscivorous fish. With the deployment of ARs, the total biomass of the system is increased by 14 %. Filter feeders become the predominant benthic taxa in the system (62 %). This result needs to be qualified by the fact that the biomasses of filter feeders is calculated by the model.

The presence of filter feeders and grazer communities on ARs is considered essential to transfer the energy from the water column to the macro-invertebrates and fish communities (Bortone *et al.*, 2000). Their dominance in the benthic community has been demonstrated by various studies on artificial structures (Cresson, 2013; Boaventura *et al.*, 2006; Wetzel *et al.*, 2014). The dominance of filter feeder species such as barnacles and mussels has been described as an initial condition for the colonization of artificial structures before the establishment of a more heterogeneous community (Boaventura *et al.*, 2006; Wetzel *et al.*, 2014, Cresson, 2013, Monteiro and Santos, 2000). A similar benthic composition was expected for the ARs on the Capbreton site.

We find a major difference in the composition of the benthic community compared to other artificial structures such as offshore wind farms (OWF). On OWF foundations located in the Baltic Sea, the biomass of blue mussels is totally predominant and accounts for more than 97 % of the total biomass of the benthic fauna (Maar *et al.*, 2009). The enhanced concentration of blue mussels observed on a pillar near the surface is about 7 to 18 times higher than on scour protection (Maar *et al.*, 2009). A very different composition is found on the studied ARs along the Landes coast, where the mussel biomass is very low and represents less than 1 % of the invertebrate filter feeders. This major difference could be explained by the distinct difference in size between the two structures and the different environmental context (Degraer *et al.,* 2020). While OWFs make use of monopiles placed on a soft bottom habitat and which reach up to the sea-surface, ARs do not reach the surface and have little influence on the water column (in this study, ARs have a height of around 2.6 m). Besides, the Baltic Sea bottom is covered by extensive blue mussel beds, whereas, along the Landes coast, the nearest mussel beds are located at distances of 3 km and then 20 km from the Capbreton Artificial Reef study site (Figure 1). Therefore, the larval flow of mussels is reduced. Compared to OWFs, the small effect on filter feeder biomass is due to the light colonization of this species (blue mussel) from the water column (Degraer *et al.,* 2020). Instead of blue mussels, another trophic competitor has colonized the Capbreton ARs: barnacles account for around 52 % of benthic community on ARs. Despite of the presence of barnacle, in comparison with other similar ARs, the filter feeder biomass still is lower by a factor of 10 (Wetzel *et al.*, 2014).

The benthic fauna represents the primary prey of reef fishes. As the benthic fauna increases, it is expected that fishes will come to feed on the ARs and thus contribute to increase in production around ARs (Fabi *et al.,* 2006). However, we need to analyse gut contents to confirm this hypothesis. Indeed, some species such as planktivorous fish, do not feed on the ARs benthic fauna, but can nevertheless be attracted onto ARs by the zooplankton exposure due to the ocean (Cresson *et al.*, 2019). A similar study conducted in Hong Kong simulated the reef effect after the implantation of ARs (Pitcher *et al.*, 2002). With ARs covering 3 % of the Marine Protected Areas of Hong Kong, the fish biomass is estimated to have increased by 30 % corresponding to 247 t (Xu *et al*., 2019). In the current study, ARs represent 11 % of the studied area that could be taken as equivalent to an MPA because of the restricted access established over the entire area. The fish biomass has increased by 67 %, but this represents only 460 kg. Based on the biomass evolution between BAR and AAR system, the increase of benthic fauna and fish community could be interpreted as a success of ARs biomass production. Nevertheless, ARs are known to have attraction function, and their contribution to biomass production may be local (Cresson *et al.,* 2019). Exploring the bottom trawl survey data carried out by IFREMER in the Bay of Biscay in 2016, using a proximal trait to the study site (7km), and compare it to the BAR data (from the same IFREMER survey but in 1999-2002), the results showed little biomass variation that could be an increase or decrease depending on the species (e.g. Trachinus Draco: -3%). This comparison, using only one data campaign, supports the local trends in ARs contribution.

The comparison with other ARs systems could give a scale of effectiveness but need to integrate local characteristics. Several factors could influence the biomass production of fish and epifauna: reef shape, size, volume, relief, roughness, substrate composition, kelp density, invertebrate density, reef age, proximity to natural reefs and larval supply (Granneman *et al.*, 2015, Moschella *et al.*, 2005; Baine, 2001; Abelson *et al.,* 2002). The complexity of a module is a function of its shape, roughness, porosity and the size of cavities that it contains (Riera, 2020). An indicator was used to classify modules in function of their objective and the fauna characteristics. Two of the three ARs deployed in the present study can be characterized as follows (following Bouchard, 2018):

1) The Bonna pipe module is described as a “box” structure with a large hole on the top and a small hole at the side. The surface specific deployed seems sufficient for settled benthic fauna and to provide wide shelter for demersal fishes.

2) The Typi module is a « cage » type structure that is not suitable for demersal fishes because of the lack of shelters. The size and the volume of these modules deployed in the studied area (102 m² and 830 m^3) may be too small to sufficiently enhance biodiversity and biomass (Hackradt *et al.*, 2011). Environmental criteria also influence the efficiency of ARs. In fact, the diversity of fauna communities depends on the larval flow and is affected by ocean dynamics and the connectivity with other hard substrates (Svane and Petersen 2001; Koeck *et al.* 2011, De Bie *et al.* 2012). As already highlighted, the studied sector is 20 km away from a natural rocky habitat. But the shipwrecks and harbour channels near the studied area act as transitory hard substrates which could ensure the connectivity between these

features (Pastor, 2008). The site is not only subject to local upwelling carrying primary producers but also intensive storms that damage reefs by smoothing (Hylkema *et al.,* 2020). All these local factors make it difficult to compare the productivity based on the biomass indicator between different ARs at various localities (Baine, 2001).

4.3 Ecological Network Analysis provide new indicators to assess ARs effectiveness

The changes in ecosystems over time can only be described when ecosystem topologies remain similar. The Before/After analysis used in our study has the advantage of providing two similar ecosystem topologies for the Ecological Network Analysis (ENA). ENA provides indicators that enable us to link the ecosystem structure and its functionalities (Ulanowicz, 1986). Then, these evolving trends can be compared to other types of ecosystems. In our study, the trends in Total ecosystem activity and Ascendency (A) between the two periods show an increase of approximately 9 and 16 %, respectively (Table 5). These rising rates are similar to those simulated in the English Channel for a system before and after OWF deployment (Raoux *et al.*, 2017). Conversely, the System Omnivory Index (SOI) decreases between the two periods, and this trend was also observed in Laizhou Bay following AR deployment (Table 4). Finally, the results also highlight that Finn's Cycling Index remains mostly stable between the two periods.

Results concerning the other ecosystem attributes show that the ratios PPt/R, PPt/B and the B/T.. vary between the two systems, but this is not the case for the simulation of ARs deployment in Bohai Bay (Table 5). In fact, the PPt/R ratio decreases between the BAR model and the AAR model by approximately 20 %. This trend is also observed in Laizhou Bay as well as in the English Channel, but is the opposite of the change occurring in the ecosystem of the Yellow sea with OWF deployment (Table 5). A similar pattern is observed for the PPt/B ratio, which shows a decrease of approximately 22 % between the BAR model and the AAR model (Table 5). By contrast, the B/T.. increases between the BAR model and the AAR model by approximately 15 %, in accordance with the change in the ecosystem of the English Channel (Table 4).

The maturity of a system can be assessed using several indices. The PPt/R index is the ratio between the energy used for biomass production (total primary production) and the energy used for maintaining stability of the system (total respiration) (Christensen *et al.*, 2005). When the system is growing, generally in a "young system", production exceeds the respiration and the PPt/R index is higher than unity. On the contrary, when the system is mature, the system tends to balance the use of energy related to both production and consumption (Odum, 1969). The B/T.. ratio is an index that increases with the maturity of the system. Regarding these indicators used by Odum (1969), there is a good correlation between the decrease in PPt/R, PPt/B, net community production and the increase of B/T (Table 5). Since there are identical input data of primary producers in both models, the PPt/R and PPt/B metrics inevitably decrease, because the primary production stays the same whereas biomass increases. The

B/T.. ratio, which is not directly related to primary production, could better be used in this study to describe the change in maturity of the system. Thus, the B/T.. ratio shows an increase in system maturity with the deployment of ARs. Mature and young systems have been described by Odum (1969) as extreme opposites of an ecosystem. While young systems are characterized by production in terms of growing and abundance, mature systems yield indicators, such as B/T... but also A and SOI index, that reflect the stability of a complex web-like system. The increase of system maturity showed by B/T.. is confirmed by the Ascendancy increase (Ulanowicz, 1997) and the SOI index trend that indicates evolution to a more complex system (Libralato, 2008). Thus, in our case, the deployment of ARs changes the structure of the ecosystem towards a more complex system and its functionality towards a more stable system.

However, the study is based on observation of the last two years of the benthic community and ten years of fish assemblages. By averaging ten years of surveys, we can smooth out the inter-annual variations in biomass. While communities associated with ARs could rapidly become a stable system (Scarcella *et al.*, 2015), the ARs of Capbreton could have been a production system during the initial period before becoming more mature. Compared to other trophic modelling simulations on ARs (Guan *et al.*, 2016; Xu *et al.*, 2019) or other artificial structures such as OWFs, (Raoux *et al.*, 2017), the increase of maturity seems to be a criteria of reef effect based on the B/T.., PPt/R or PPt/B metrics (Table 5). In addition, Wang *et al*, (2019) used the System Omnivory Index to measure the increase of maturity with OWF deployment in the Yellow Sea. This index describes the complexity of the system and also provides the characteristics of a mature stage (trophic food chains represented as a web-like system).

Références for Ecopath models	**This study**			**Guan et al 2016**			**Xu et al., 2019**			**Wang *et al.* 2019**			**Raoux *et al.*, 2017**		
Models	BAR	AAR		BAR in Bohai Bay	Simulation of AR in Bohai Bay		BAR in Laizhou*	AAR in Laizhou*		Before OWF in Yellow sea*	OWF in Yellow sea *		Before OWF in English channel	Simulation of OWF in English channel	
Number of group	23	23	/	13	13	/	13	17	/	14	14	/	37	37	/
T..	379	413	+9 %	/	/	/	4721,2	3924,49	-17 %	/	/	/	1607,62	1831,93	+13,95 %
A	401	465	+16 %	/	/	/	/	/	/	7 195	8677	+21%	1869,1	2156,9	+15,40%
SOI	0,36	0,302	-16%	0,0379	0,0379	0%	0,23	0,188	-18%	/	/	/	0,173	0,199	+15,03%
FCI	13	13	0%	0,25	0,25	0%	9,035	16,84	+86%	3,899	8,448	+117%	/	/	/
PPt/R	1,5	1,2	-20%	3,1088	3,1087	0%	1,724	0,665	-61%	1,47	1,784	+21%	1,72	1,12	-34,88%
B/T..	0,013	0,015	+15%	0,0105	0,0105	0%	0,01	0,045	+350%	/	/	/	0,03	0,04	+33,33%
PPt/B	17,01	13,35	-22%	42,399	42,3987	-0,001%	27,54	3,145	-89%	52,281	67,737	+30%	/	/	/

*Table 5: Comparison between ENA indicators of Before/After analysis with other Ecopath models. Total System Throughput (T.., gC.m-2. Year-1; * t.km-2. Year-1); Ascendency (A, flowbits); System Omnivory Index (SOI, %), FCI (%), Total primary production/total respiration (PPt/R), Total biomass/total throughput (B/T..) and T otal primary production/total biomass (PPt/B)*

Others model parameters also help us understand the functioning of AR systems. Ascendency represents the level of the system activity and its organization (Ulanowicz, 1986). The increase of Ascendency also indicates a higher activity in the system, which is characteristic of a maturity stage (Ulanowicz, 1997). Regarding the modelling of Wang *et al.*, (2019) and Raoux *et al.*, (2017), this parameter increases respectively after eight and thirty years of OWF deployment. This result should be qualified by the unchanged value of the FCI (percentage of all flow in the system) before and after AR deployment (Finn, 1980). Thus, the low boosting of activity corroborates the local effect of ARs on the Capbreton site, without any strong modification in the system structure and functioning.

The indicators suggested to detect changes in ecosystems in this study are based on studies conducted to highlight the relevant ENA indicators (Safi *et al.*, 2019; Fath *et al.*, 2019). By analysing the ecosystem functioning and structure, ENA provides holistic indicators to assess the impact of human activities and environmental management measures such as ARs deployment. In fact, the ecological effects expected from the deployment of ARs were listed by Claudet and Pelletier (2004), but no details were given about the quantified objectives to be attained. As a result, only indicators showing the trend of the system towards the general objectives could be used by managers to monitor the performance of ARs. Coupled before/after analysis with trophic modelling approach allows indicators that reveal structural and functional changes in the ecosystem with ARs deployment and could be used by managers to assess the effectiveness of ARs.

The growth in the use of the trophic modelling approach reflects the emerging need for indicators for managers (Heymans *et al.*, 2016; Pezy *et al.*, 2017; Raoux *et al.*, 2017; Guan *et al.*, 2016; Wang *et al.* 2019; Xu *et al.*, 2019; Prato *et al.* 2016; Valls *et al.* 2012; Hermosillo-Núñez *et al.*, 2018). The current study is embedded in this approach, with the aim of highlighting the effect of AR deployment on ecosystems.

4.4 Limitations of the trophic modelling approach

Trophic modelling is based on large amounts of biological data for each functional group chosen. Besides, diet is a key parameter in the trophic modelling approach. In this study, the BAR model is largely inspired by the data selected from coastal areas of the southern part of the Bay of Biscay (Lassalle *et al.*, 2011) and information on diet is drawn from the literature. In this study, the models were based on available data as proximate to the study as possible that lead to differences in the sampling efficiency (such as between bottom trawl data and scuba diver surveys) and the period covered up. As a consequence, our trophic models should be considered as a first approach to providing an overview of the evolution of AR systems. Artificial reefs are known to attract a high abundance of fish, which could potentially increase the local production. Thus, there is a need to investigate the feeding ecology and trophic diet of fish that occur abundantly on artificial reefs by analysing stable isotopes and stomach contents to examine the short- and long-term trophic diet composition (Bentorcha *et al.*, 2017). The difficulty and cost of such extensive data collection could be an obstacle when applying the trophic approach to coastal management. Prato *et al.* (2014) suggested carrying out a prior survey of the most important and less documented functional groups. In this how, we chose to focus on benthic invertebrates and fish biomass surveys and fixed the upper and lower trophic groups biomasses. Consequently, the direct biomass trends of these groups could not be analysed. However, they still were integrated in the trophic modelling as a part of the system, and the flows tendencies were investigated. Finally, ENA is clearly dependent on the model structure and comparisons between trophic models could be hazardous (Prato 2016; Fath *et al.*, 2019; Christensen *et al.*, 2005). Equivalent models need to be favoured to assess the effect or evolution of coastal management tools within the ecosystem, i.e. models based on the same number of functional groups and the same composition of these groups.

5 Conclusion

The ARs assessment still remains a challenge for marine managers who are required to monitor the objectives of maintaining or enhancing fisheries production, with the aim of readjusting human pressures on the ecosystems (Salaün *et al.*, 2022a). At the same time, trophic modelling has been developed over many decades and applied to monitor various marine ecosystems around the world. This approach has been used to understand the effect of fisheries on the entire ecosystem. Recently, it was extended to other research domains such as the management of MPAs (Hermosillo-Núñez *et al.*, 2018) and the simulation of the effect of OWFs on the ecosystem (Raoux *et al.,* 2017; Pezy *et al.*, 2017). Our study represents a new investigation of the use of trophic modelling, based on a comparison of the system before and after the deployment of ARs.
Like OWFs, ARs are mostly deployed on soft bottom habitats. So, they create hard substrates that become colonized by various communities. With the deployment of ARs, the total biomass of the system increases and the dominant fauna changes from detritivores to invertebrate filter feeders. However, the reef effect is restricted to its vicinity and the low increase in biomass should rather be linked to the environmental context of the studied area (a sandy coast with low connectivity with hard substrates). In this case, the deployment of ARs has little influence on the ecosystem structure and biomass production. By using ENA metrics on AR systems, it is possible to highlight the trophic modifications linked to the introduction of hard substrates on soft habitats. Our study highlights a positive effect with an increase in system maturity through ARs deployment; this finding has emerged by using ENA indicators, such as B/T.., PPt/R, PPt/B and SOI. In accordance with other studies, this related change in maturity seems to be a criterion reflecting the effect of artificial structures. Thus, our results demonstrate the interest of using a large set of ENA indicators to characterize different trophic functioning attributes. This is

essential for an effective overview of the induced changes. By the end, ENA provides indicators that could be used by managers to monitor the temporal colonization and evolution of ARs, and assess performances objectives, to appreciate the pertinence of their deployment.

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Author Statement

J.S, A.R. and J-C D. designed the networks. J.S gathered the data. A.R. modelled the networks. J.S analyzed the data and A.R. and J-C D. helped in interpreting the results. J.S wrote the paper with input from all authors.

Declaration of interests

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

☐ The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: