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Spatial evaluation of ecological indicators performance to track fishing impacts using an ecosystem modelling approach

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SUMMARY

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1. Introduction

People always believed that resources of the sea were infinite. It took the great stock collapses of the 20th century to realise that marine resources were not limited. The dramatic socio-economic and environmental consequences that followed justified the implementation of current coordinated management measures (Gascuel 2019).

There are three main types of measures to manage fisheries: 1/ control of catch levels, 2/ control of access to the resource and 3/ technical measures. All of these measures aim to control fishing mortality (abbreviated as F) (Ifremer 2018a). The level of catches is controlled at European level by TACs (Total Allowable Catches) decided by the European Commission for each stock (Biseau 2020), following recommendations made by scientists in the framework of the International Council for the Exploration of the Sea (ICES) groups (Commission Delegated Regulation 2020). The management advice adopted in Europe is the MSY approach (Maximum Sustainable Yield) based on biomass and fishing mortality reference points. The reference points used are respectively B_{MSY} the biomass that enables a stock to deliver the maximum sustainable yield (that usually corresponds to the maximum production of biomass of the stock) and F_{MSY} the maximum rate of fishing mortality resulting eventually on a long frame in stock size of B_{MSY} and a catch of MSY. At European scale, all these management measures have led to a halving of the F/F_{MSY} ratio in 20 years (STECF 2021). During this period, stock sizes increased by 50% (Worm, Branch 2012).

For the Eastern English Channel (the EEC), these measures are applied at three different scales, from the regional to the European scale (ANNEX I). At European level, the main measures are TACs, mesh size, the implementation of European Fisheries Authorisation for demersal species and specifically for cod. On a national scale, there are measures to control spatial and temporal access to the resource, such as marine protected areas or a ban on fishing in sole nursery areas. However, on a regional scale, there are few derogations for fishing within some nursery areas during certain periods.

Despite these measures, many stocks are still overexploited in the EEC. Indeed, the horse mackerel, sole and red mullet stocks are overexploited. Cod has collapsed since the 2000s with no recovery of the stock today (Biseau 2020). In 2019, about 40% of stocks exploited by France (in number) are overexploited (Biseau 2020). Therefore, the F_{MSY} objective for all stocks, set during the World Summit on Sustainable Development in Johannesburg in 2002, is still far from being achieved. The management of European fisheries resources is mainly done on a stock-by-stock basis, and does not take into account the ecosystem effects of fishing, either by impacting the habitat or the non-target species (Bentley et al. 2019, García 2003). But this management method is limited. It is impossible to achieve single-species MSY for all stocks simultaneously because fisheries are generally mixed and species interact with each other. Furthermore, it does not take into account the impact of the environment on MSY.

Indeed, fishing is considered to be the anthropogenic factor that has the greatest impact on marine ecosystems (Worm et al. 2006; Dayton et al. 1995). It is therefore necessary to use an ecosystem-based management method to measure the impact of fishing in a more integrated way. This method is called the Ecosystem Approach to Fisheries (EAF) and aims to establish a holistic method for managing fisheries sustainably while preserving the ecosystem (Pikitch et al. 2004; Szaro, Sexton, Malone 1998; Larkin 1996).

However, to implement this management method and track fishing impacts it is necessary to develop reliable and user-friendly indicators for managers (Rice, Rochet 2005). An indicator is a tool developed to improve our knowledge of the ecosystem. It has two main purposes, it measures the human impact or sustainability of a system and it reduces the complexity of the real world into a tool that is easy to understand and facilitating decision-making (Boldt et al. 2014). Over the past 15 years, researchers have developed methods to select indicators that can measure the impact of fisheries on ecosystems by measuring structure, composition or function, but also the state of ecosystems for management and conservation purposes (Reed et al. 2017; the IndiSeas Working Group et al. 2012; Link 2005; Rice, Rochet 2005). A single indicator is not sufficient to define the state of an ecosystem. Therefore, for each ecosystem, a set of indicators must be defined jointly and complement each other according to the management objective. (Rice, Rochet 2005; Fulton, Smith, Punt 2005).

Given the multiplicity of indicators, it is necessary to be able to compare them with each other and choose the best performing indicators. Each indicator is evaluated according to criteria defined for each study. Boldt et al. (2014) summarised the main criteria for evaluating ecosystem indicators to multiple stressors, such as sensitivity and responsiveness. Generally, the importance weight of the criteria is different for each criterion. Then by knowing score value of an indicator for each criterion and their weight, it is possible to establish a ranking of the best indicators (Rice, Rochet 2005).

Many of these indicators are calculated using ecosystem models. These complex models make it possible to quantify the interactions and flows within an ecosystem, from primary production to the exploitation of resources by humans (Fulton et al. 2011). Nevertheless, structural uncertainties might emerge due to the use of models based on different assumptions. The multi-model approach could help to overcome this problem in order to ensure the robustness of the results.

Previous studies on indicators performance for detecting the impact of fishing have already been carried out, such with the Indiseas project (the IndiSeas Working Group et al. 2012). However, in these studies indicators were calculated as an average value over the whole study area. However, we do not know whether the average value is representative of the whole area or whether, on the contrary, indicators present spatial heterogeneities. Hence the interest of analysing the spatial performance of indicators to assess fishing impacts.

Our study is part of the European project SeaWise, whose objectives are to set up a network of stakeholders, solidify knowledge on marine environments, develop predictive models and provide management advice. More specifically, it is part of the task 4.4, which focuses on the effects of fishing on food webs and community diversity. All these objectives are in line with the overall goal of implementing ecosystem-based fisheries management at the European level. Furthermore, it keep up with the IndiSeas project (the IndiSeas Working Group et al. 2012) by focusing on a new, unstudied indicator feature : the spatial performance. The main objective of this study is to investigate the performance of spatialized indicators in capturing the impacts of fishing in the Eastern English Channel ecosystem. We will carry out a comparative approach using two ecosystem models: Osmose (Travers-Trolet et al. 2019) and Atlantis (Girardin et al. 2018). We will test different types of fishing scenarios to evaluate the performance of indicators in assessing the state of the ecosystem.

2. Materiel and methods

2.1 Presentation of the case study

Our study area is the VIId area (ICES), commonly called the Eastern English Channel (EEC), a semi-enclosed sea. It has a surface area of around 35,000km² (Girardin et al. 2018). It is characterized on the French side by significant river inflow from the Seine and the Somme, which results in low salinity and turbid water (Figure 1). Along the English coast the freshwater input are lower so the salinity is higher (Vaz, Carpentier, Coppin 2007). The seafloor is mainly composed of pebbles, gravels and gravely sands sediments (Rochette et al. 2010). This ecosystem is also characterised by its shallowness. This particular environment strongly drive the structure of the EEC fish community. Indeed, when the depth is shallow there is a stronger benthic pelagic coupling (Cresson et al. 2020; Giraldo et al. 2017). The demersal fish communities are distributed along both depth and sediment gradients (Vaz, Carpentier, Coppin 2007). In addition, EEC estuaries are important nurseries for flatfish, such as the Sole (Rochette et al. 2010).

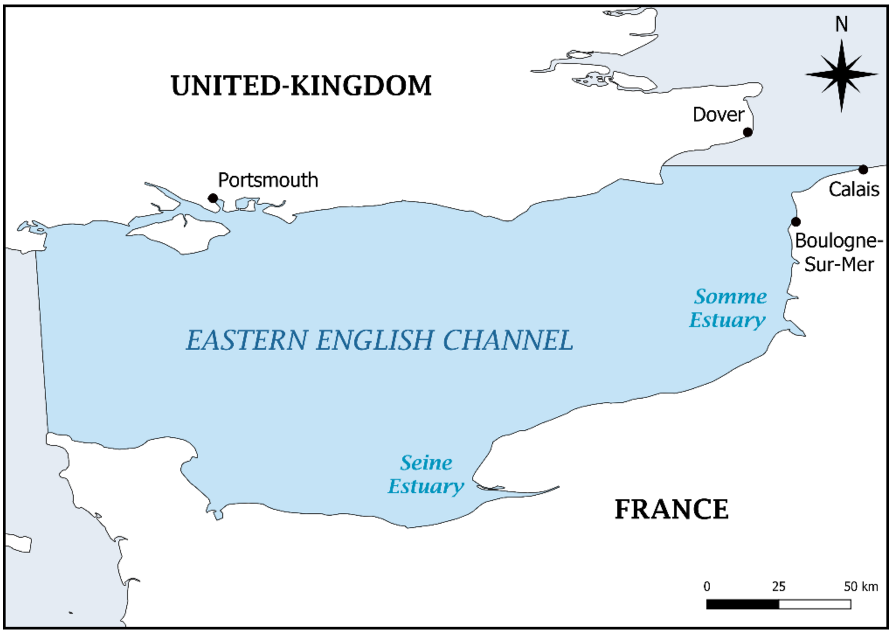

Figure 1: Map of EEC (Package oceanmap) (lines in black are the delimitation of the VIId ICES zone)

The EEC is an important fishing area with 125000T of captures in 2019 (ICES 2021), for 127 million of revenues for the French fisheries, mainly unload in the largest French port, Boulogne-Sur-Mer (FranceAgriMer 2021). In 2019, the main fished species are herring (*Clupea harengus*), mackerel (*Scomber* spp.), horse mackerel (*Trachurus* spp.), whiting (*Merlangius merlangus*), red mullet (*Mullus surmuletus*), lesser spotted dogfish (*Scyliorhinus canicula*) and common sole (*Solea solea*), with king scallop (*Pecten maximus*), whelk (*Buccinum undatum*), cuttlefish (*Sepia officinalis*), common european bittersweet (*Glycymeris* glyceymeris) and squid (*Loligo vulgaris*) (ICES 2021) for non-fish species. Fishers mainly use beam trawl and bottom trawl, with trammel net for coastal and small-scale fisheries (CRPMEM Hauts de France 2015). This fishing area is shared by several European countries (from those which catch the most to those which catch the least in 2019): France, United Kingdom, Netherlands, Belgium, Ireland, Germany, Spain, Denmark and Lithuania (ICES 2021). Recently the Brexit has led to a reorganization of the total allowable catch (TAC) allocated between the EU and the United Kingdom in area VIId from the year 2022. The EU and the United Kingdom receive a defined percentage, after negotiations, for each TAC stock (Directorate-General for Maritime Affairs and Fisheries 2021).

2.2 Multi-model approach

In this study, we use two ecosystem models Osmose and Atlantis modelling the same ecosystem: the EEC. For the same ecosystem, the two models will have different structures and assumptions, which may influence the results and add uncertainty. The multi-model approach will allow us to compare our results for the same ecosystem and thus observe the impact of model assumptions on our results and improve the robustness of our results (Shin et al. 2018).

2.3 Atlantis: model overview

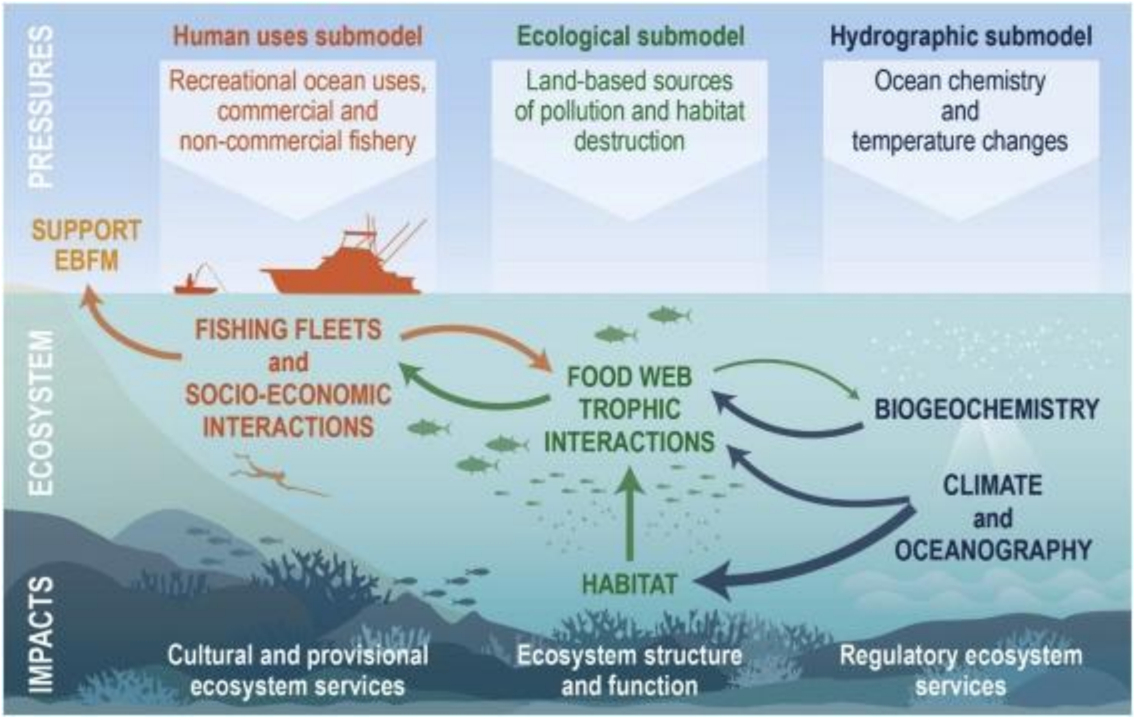

Figure 2: Major processes in Atlantis (Pethybridge et al. 2019)

Atlantis is an end-to-end model. It models the entire ecosystem to be used in fisheries evaluation and management (Fulton et al. 2011). Atlantis is separated in different submodels: biophysical, industry and management (Figure 2).

- Biophysical

This submodel is a deterministic model. It is a 3D model cut into polygons of different depth layers. It allows the calculation of biological parameters such as consumption, production, waste production, movement, migration, predation, recruitment, habitat dependency, mortality, but also physical parameters such as temperature, pH, dissolved oxygen, salinity by a coupling with an oceanographic transport model.

- Industry

This submodel represents the human uses of the system, mainly fisheries but it could add other human impacts such as pollution or coastal development. It is possible to implement several fisheries characterised by their fishing gear, target and effort. It is possible to add more features to the model to take into account of the behaviour of fishers according to evolution of markets and fuel prices.

- Management

This part includes monitoring, assessment and management decision processes. Monitoring data are coming from the biophysical and industry models. This simulated data allow calculating ecological indicators. Moreover, it can be used in assessment submodel to evaluate

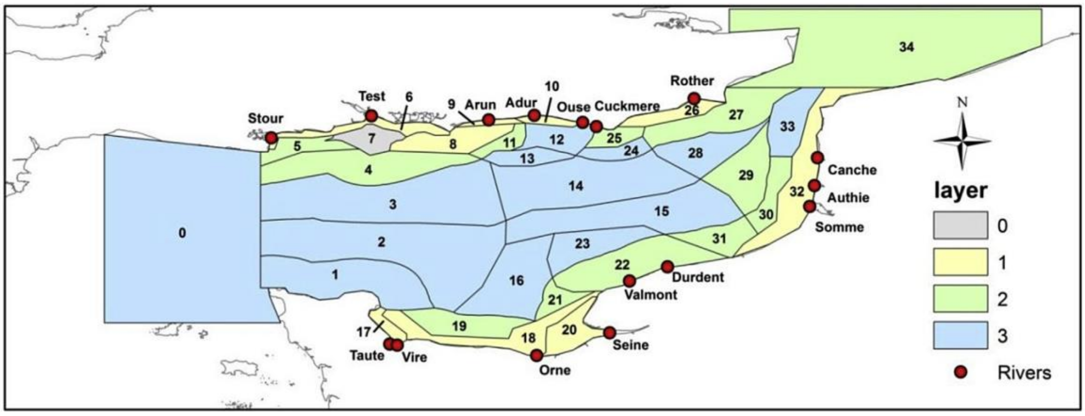

Figure 3: Spatial structure of Atlantis EEC (Girardin et al. 2018)

the state of the ecosystem and later with management submodel to applied decisions according to the assessment. There is a lot of decision possible in Atlantis based on classic fishery management measures such as gear restrictions, quotas, temporal zoning.

Today Atlantis is applied for more 30 systems all around the word. It covers diverse ecosystems from temperate ecosystem to polar ecosystem and at different scale level, from estuaries to ocean regions (CSIRO 2020). In this study, we present an application of the Atlantis model in the Eastern English Channel ecosystem.

2.4 Atlantis: application to the English Channel

The Atlantis EEC (Girardin et al. 2018) model has been developed to simulate the functioning of the EEC ecosystem with a focus on two commercial species: common sole and plaice (*Pleuronectes platessa*). This model is implemented for the period between 2002 and 2011.

- The spatial structure

The model is divided into 35 polygons on 3 depth layers (0-15m, 15-30m and over 30m) including a sediment layer (Figure 1). The maximum depth is 60m. The polygons were delineated from biogeographic data of the area and take account the administrative separations between France and the UK (12 miles, EEZ). Polygons 0 and 34 at the extremities of the zone represent the Western English Channel and Southern North Sea respectively. Polygon 7 is an island.

- Physical forcing in the Atlantis EEC model

The MARS3D hydrodynamic model (Bailly du Bois, Dumas, Solier 2005) reproduces the water flows, salinity and temperature in the model. The freshwater flows are very important due to the presence of numerous estuaries such as the Seine.

- Biological functional groups implementation

40 functional groups were implemented in the model. Species in each group were grouped according to their habitat, growth, migration behaviour and feeding. There are 21 vertebrate groups including seabirds, marine mammals, fish and 16 invertebrate groups including plankton. Finally, there are three detritus groups including labile, refractory and fisheries discards.

Vertebrates are subdivided into 10 age classes, each representing 1/10 of the group total lifespan. For invertebrates, cephalopods are structured in stages, while all other groups are considered as single biomass pools.

- Fisheries in Atlantis-EEC model

Unlike the original model (Girardin et al. 2018), we use a simplified version of the model with only one fishery with a constant fishing mortality for each functional group (Bracis et al. 2020).

2.5 Osmose: model overview

Osmose (Object-oriented Simulator of Marine ecOSystem Exploitation) version 4.3.2 is an Individual-based model (IBM), it represents fish individuals grouped in schools (super-individual) defined by their size, weight, age, taxonomy and spatial position on a 2D regular grid (Shin, Cury 2004). The model is based on the hypothesis of size-based opportunistic predation (i.e. size adequacy and spatial co-occurrence between a predator and its preys). Osmose model could be forced by a biogeochemical model in a one-way coupling to set up an end-to-end approach. Low trophic levels sources (LTL) (phytoplankton and zooplankton groups) obtained from the biogeochemical model are used as biomass prey fields for high trophic level species (HTL) during the process of predation. The main biological processes occurring in each time step are distribution, local interactions (predation and other sources of mortality), growth and reproduction (Travers-Trolet, Shin, Field 2014; Travers-Trolet et al. 2019) (Figure 4).

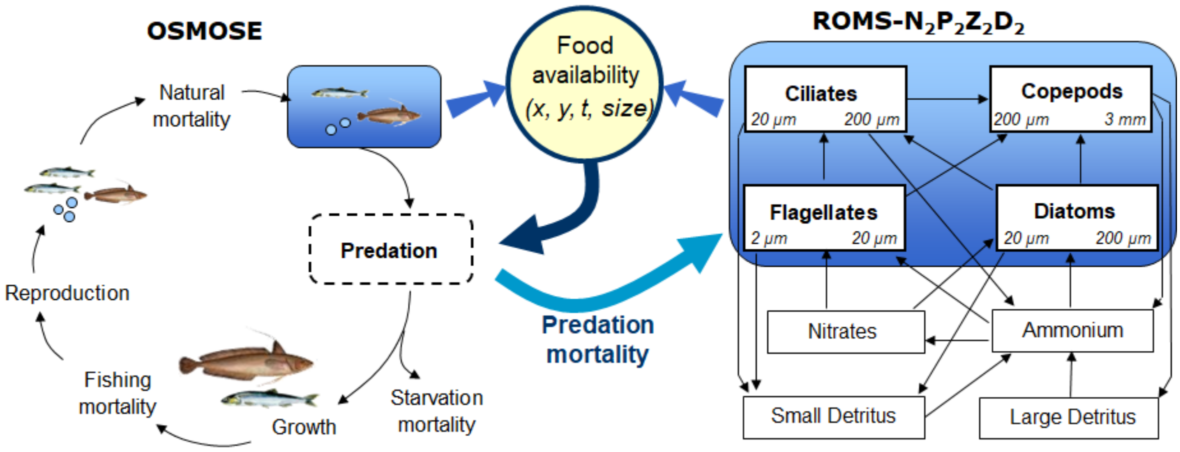

Figure 4: Major processes in Osmose at right and at left NPZ model coupled to Osmose (source: Osmose 2022; Travers-Trolet, Shin, Field 2014)

- Movement

Species in Osmose move following a random walk restricted to their distribution area set up as a presence/absence map.

- Local interactions

Local interactions, take in account the different sources of mortality namely predation, starvation mortality, fishing mortality and mortality from other sources.

For predation, each super individual is associated with an ingestion rate. This ingestion rate is compared to the availability of food locally and their assimilation rate. Indeed, preys must correspond to the size range that predators can consume. According to prey consumption, each species is associated with a predation efficiency rate. This predation efficiency rate is compared to the predator satiety (food requirement for maintenance). In the model, if the predation efficiency is below satiety then there is a starvation mortality. Other sources of mortality are fishing mortality, mortality at the larval and egg stages and other mortality due to not explicitly modelled predators and/or disease-related mortality. In this configuration, the fishing mortality is species specific; it was parameterized by providing a fishing mortality rate F by species and stage (juvenile/adults). However, in recent versions of Osmose, it become possible to specify several fleets targeting different species.

- Growth

The growth of individuals can only take place when the individuals have passed satiety. In this case, individuals will grow according to Von Bertanlaffy's law weighted by predation success.

- Reproduction

The number of eggs released in the system depends on the relative fecundity of females (the number of eggs emitted per gram of mature female) and the biomass of mature females.

2.6 Osmose: application to the English Channel

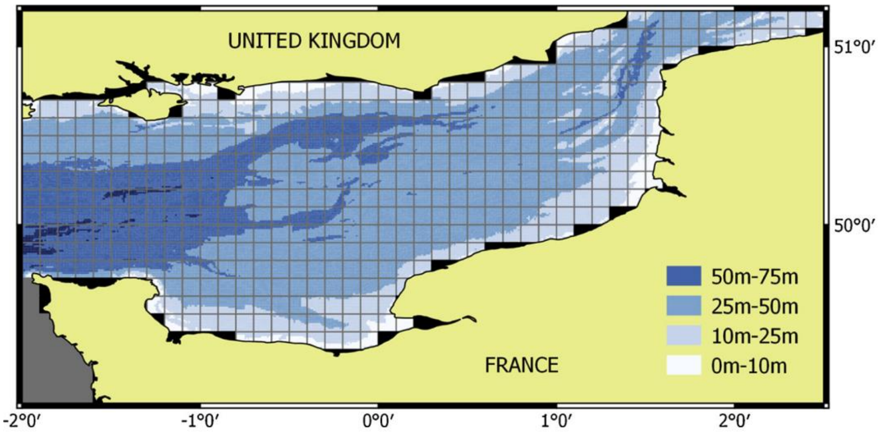

Figure 5: EEC Osmose grid (0.6° x 0.6° cells) with indication of depth (from GEBCO website) (Travers-Trolet et al. 2019)

We use in our study the Osmose model applied to this EEC ecosystem (Travers-Trolet et al. 2019). This model is implemented with data from 2000 to 2009. It is composed by 14 species, 13 fish and 1 squid group. These species represent 80% of the international landing in EEC.

The grid is composed of 445 cells of 0.6x0.6 degree from 49° N – 2°W to 51.5°N – 2.5°E (Figure 5). The biogeochemical ECO-MARS3D applied to the EEC was used to model hydrodynamics and biogeochemical the model ECO-MARS3D is used (Vanhoutte-Brunier et al. 2008; Le Goff et al. 2017). In addition, there is a matrix of species accessibility for predation to represent vertical interaction between benthic and pelagic communities.

2.7 Selection of indicators

One of the objectives of this work is to evaluate the spatial performance of a set of ecological indicators to detect the effects of fishing on the ecosystem. These indicators have to be concrete, easy to understand and facilitate decision-making since they are intended to be used by different stakeholders (e.g fishery managers, policymakers, etc.). In addition, they have to be easy to calculate spatially from model outputs. Thus, we have defined a list of potential indicators based on the previous study Indiseas and different other studies (Halouani et al. 2019; Reed et al. 2017; Bourdaud et al. 2016; the IndiSeas Working Group et al. 2012; Shin et al. 2010; 2005; Rice, Rochet 2005; Fulton, Smith, Punt 2005), separated into three themes: fishing pressure, trophic level, ecosystem structure indicators. The selected indicators (Table 1) were computed using Osmose and Atlantis outputs except for the length at age, the maximum length and the life span. The two first indicators were not provided by Osmose (since individual species sizes were averaged in each cell) and the life span indicator is not computed in Atlantis.

To compute API, % Predator, and ratio P/D, species/functional groups in both model were characterised as predator, and demersal/pelagic fish based on FishBase and SeaLifeBase (Froese, Pauly 2022; Palomares, D. Pauly. 2022) (ANNEX II & ANNEX III). Predators were defined according to Indiseas definition: “*Predatory fish are considered to be all surveyed fish species that are not largely planktivorous (i.e. phytoplankton and zooplankton feeders should be excluded). A fish species is classified as predatory if it is piscivorous, or if it feeds on invertebrates that are larger than the macrozooplankton category (> 2cm). Detritivores are not classified as predatory fish.*”(the IndiSeas Working Group et al. 2012)

We calculate all our indicators following the formula in Table 1.

Table 1: List of all the indicators calculated, their calculation and their objectives

Indicators	**Name**	**Theme**	**Calculation**	**Range**	**Objectives**	**Sources**
IFP	Inverse fishing pressure	Fishing pressure indicator	$\frac{\sum_c B_c}{\sum_c Y_c}$ No unit	]0, +∞[	Maintaining Resource Potential	(the IndiSeas Working Group et al. 2012; Shin et al. 2010)
API	Apex Predator Index	Trophic Level of ecosystem	$\frac{\sum_c B_{c,TL\geq 4}\ *}{\sum_c B_{c,predator}}$ No unit	[0,1]	High level trophic	(Bourdaud et al. 2016)
HTI	High Trophic level Indicator	Trophic Level of ecosystem	$\frac{\sum_c B_{c,TL\geq 4}\ **}{\sum_c B_c}$ No unit	[0,1]	High level trophic	(Bourdaud et al. 2016)
LFI	The Large Fish Indicator	Trophic Level of ecosystem	$\frac{\sum_c B_{c,size\ \geq l,g}}{\sum_c B_c}$ $l \in (20,30,40,50)$ $g \in (\text{pelagic, demersal})$ No unit	[0,1]	Size composition in fish communities	(OSPAR - FW3 2015)
*Lifespan****	Mean lifespan	Structure of the ecosystem	$\frac{\sum_{c,s}(B_{c,s} \times Age_{\max})}{\sum_c B_c}$ Year	]0, +∞[	Maintaining ecosystem Stability and Resistance to perturbations	(the IndiSeas Working Group et al. 2012; Shin et al. 2010)
L_{age}****	Mean Length at age in population	Structure of the ecosystem	$\frac{\sum_{c,a} L_{c,a} \times N_{c,a}}{\sum_{c,a} N_{c,a}}$ cm	]0, +∞[	Reflects size and age structure of population + differential growth rate caused by density-dependent effects and environmental conditions	(Shin et al. 2005)
L_{max}****	Maximum Length in Population	Structure of the ecosystem	$Maximum\ observed\ length$ cm	]0, +∞[	Quantify depletion of large fish within a population	(Shin et al. 2005)
Mean Size	Mean length of fish in the community	Structure of the ecosystem	$\frac{\sum_c L_c \times N_c}{\sum_c N_c}$ cm	]0, +∞[	Reflects size structure of community	(the IndiSeas Working Group et al. 2012; Shin et al. 2010)
TL_c	Trophic level of catches	Trophic Level of ecosystem	$\frac{\sum_c Y_c \times TL_{c,catch}}{\sum_c Y_c}$ No unit	]0, +∞[	Change in average trophic level of catches	(the IndiSeas Working Group et al. 2012; Shin et al. 2010)
% Predator	Proportion of predatory fish	Structure of the ecosystem	$\frac{\sum_c B_{c,predator}}{\sum_c B_c}$ No unit	[0,1]	Conservation of biodiversity	(the IndiSeas Working Group et al. 2012; Shin et al. 2010)
Ratio P/D	Pelagic to demersal ratio	Structure of the ecosystem	$\frac{\sum_c B_{c,pelagic}}{\sum_c B_{c,demersal}}$ No unit	[0, +∞[	Energy flow and community structure	(Large et al. 2015)

*B:Biomass in T, Y:Yield in T, L:Length in cm, N:Abundance in number, Age max: Lifespan in years, TL:trophic level, c:cell or polygon, a:age, l: thereshold length for large fish in cm, Age max: Lifespan in years, g: group (pelagic or demersal) * TL ≥ 4: we select species with high trophic level (≥4) inside the predatory group using Fishbase. For Atlantis there are Atlantic cod, whiting, Pollack, sharks, and large bottom fish and for Osmose, there are whiting, Atlantic cod and lesser-spotted dogfish. ** TL ≥ 4: We filter the species with high trophic level (≥4) calculated during extraction ***Only calculated from Osmose outputs ****Only calculated from Atlantis outputs*

2.8 Indicators performance criteria

We evaluated the performance of the indicators based on their suitability to detect fishing effects with two criteria sensitivity and responsiveness (Rice, Rochet 2005).

The **sensitivity**: Does the indicator respond significantly to fishing? (i.e. indicators with a significant linear decreasing response to fishing pressure).

The **responsiveness**: the reactivity of the indicators to a disturbance in the ecosystem. Does the indicator respond shortly to changes in fishing pressure?

2.8.1 Sensitivity calculation

The perfect response of an indicator to the fishing mortality is a linear decreasing. We tried to fit a general linear model (GLM) to indicator values in function of fishing pressure. We choose laws depending of theoretical range of each indicators. For indicators with values from 0 to 1, we fitted a binomial law. For strictly positive indicator, we tried gamma and lognormal law. In this case, the best one was selected by AIC (

ANNEX V). AIC were calculated using the following formula for lognormal law.

$$AIC.\log = AIC(model) + 2 \times \sum log.value$$

Table 2 : Link function corresponding for each law and its trend

Law	Link function	Trend
Gamma	reciprocal	monotonic decreasing
LogNormal	identity	monotonic increasing
Binomial	logit	monotonic increasing

In case of binomial law, we observed that parts of our data are under dispersed. We assumed data to be under dispersed if the sum of the squares of the Pearson residuals from the binomial model was less than the 5^{th} percentile of Chi² distribution. In this case, we used a quasibinomial law instead to fit data.

We fitted a GLM for each cell or polygon in the model. We extracted the slope of the GLM in the case when results are significant. With a student test, we tested if slope is significantly different from zero. We also calculated the deviance explained by our model from our results using residual deviance.

The fitted model is linear through the link function for each chosen law, however the relationship between the response variable and the explanatory one is only monotonic (Table 2). Therefore, we could express curves trend (increasing/decreasing). Indeed, if the link function is a monotonic increasing function curve, it is conserved.

Sensitivity is calculated for each cell/polygon of models and for each indicators.

2.8.2 Responsiveness calculation

To calculate the responsiveness, we estimated Δy which corresponds to the number of year after a perturbation to observe indicators variation (Figure 5). We increased the fishing mortality and assumed after a period of time, that indicators will stabilize to a new value. To estimate the change point 1 we used the MCP Bayesian method, the most recommended method to estimate change point (Lindeløv 2020). At first, we indicated for each segment if there is a change point and a change of intercept. For example, the Figure 6 shows: i/no change point in the first segment and no change of intercept, ii/on the second segment, there are one change point and one change of intercept and iii/on the third segment, there are one change point and no change of intercept. With this method, we have the possibility to modify the automatic priors, which is recommended (Lindeløv 2020). We also tested the segmented and the BCP method but they were not adapted. We have chosen for both change point priors a uniform law from first year of perturbation to the last year, in order to avoid that the model fits a change point value before the perturbation. We calculated responsiveness for each cell/ polygon of models and for each indicators.

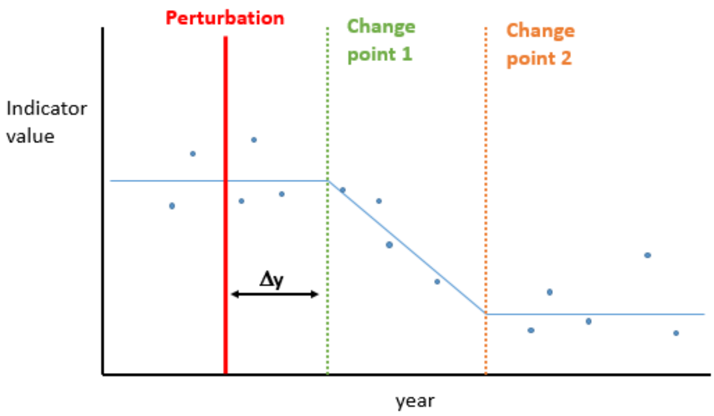

Figure 6: Figure explaining the calculation of the responsiveness Δy fitting MCP model (blue line) to our indicators data (blue points)

2.8.3 Extractions of model outputs

In our simulation plan, only exploited species were kept to facilitate the comparison between two models (ANNEX II & ANNEX III).

- Atlantis outputs

Using the Atlantis tool library we extract biomass in T, abundance, and structural and reserve nitrogen in mgN/ind.

The gross output from Atlantis is grouped by 5-day time step, species, polygon, layer, age class (only for vertebrate species). The outputs are then averaged by year, for the migratory species of the model we keep the data only on the period of the year when they are present (Annex 2). The layers are not kept either. Then from the diet matrices, we calculate the trophic level. From the structure and reserve nitrogen, we obtain the size of the individuals using the following conversion. The a and b values for each species are in ANNEX IV.

$IndMass = mean_{s,a,p,la}\left(\frac{ResN+StructN}{1000} \times 5.7 \times 20\right)$ in g (Audzijonyte et al. 2017)

$Indlength_{s,a,p,la} = \left(\frac{IndMass}{a}\right)^{\frac{1}{b}}$ in cm (Ricker 1973; Beverton, Holt 1993)

ResN and StructN : the reserve and the structure nitrogen
s: species, a: age class, p: polygon, la : layer

- Osmose outputs

Osmose simulations generates a large number of spatial outputs, (e.g. biomass, abundance, size and trophic level indicators) by species. For each scenario, 10 simulations were carried out due to the stochasticity of the model. All simulation outputs are averaged in one to calculate indicators.

2.8.4 Sensitivity scenario

For the sensitivity, we run 11 scenarios using different F_{MSY} fishing mortality multipliers from 0 to 2 by 0.2. Each simulation run for 60 and 70 years, respectively for Atlantis and Osmose, the last 20 years of the simulation were used to compute indicators, allowing a stabilisation period for each model (Figure 7). In our models input, one F_{MSY} is entered by species. This F_{MSY} is different from the monospecific F_{MSY} and the multispecies F_{MMSY}. It is calculated by gradually varying the F of the target species until the optimum catch is reached, while keeping the fishing mortalities of the other species at a constant). This calculation takes account of trophic interactions in the model. The difference with the calculation of an F_{MMSY} are that technical interactions are not taken into account and catches are not optimised over all the exploited compartments of the ecosystem (Travers-Trolet et al. 2020).

Figure 7: Chronology of the sensitivity scenario

2.8.5 Responsiveness scenario

For the responsiveness, we create one scenario with a change of fishing pressure to measure indicators response time (Figure 8). For each model, we begin the scenario with the model specific stabilisation period followed with 20 years with a F status quo. Then we apply a disturbance, we multiply by two the F statu quo during the last 20 years of the scenario.

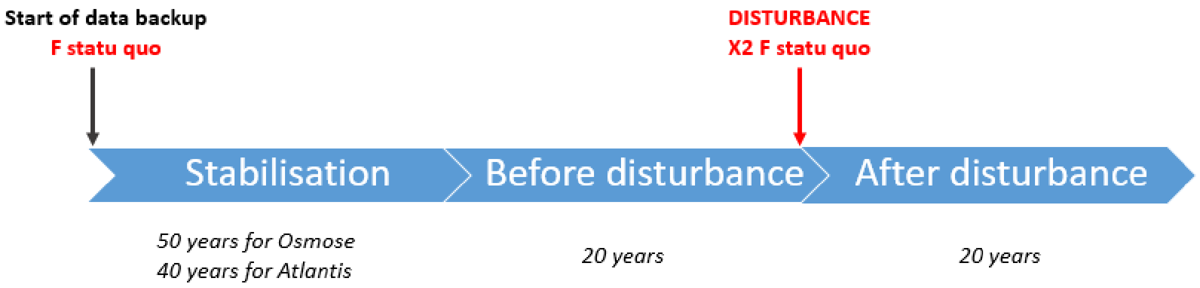

Figure 8: Chronology of the responsiveness scenario

3. Results

3.1 Raw outputs of spatial indicators from sensitivity scenario

Atlantis results show for most indicators an opposition between coastal area and central area (**Erreur ! Source du renvoi introuvable.**). Two main patterns are observed among all the indicators. One is characterized by lower value at coast than at central area such as Mean size, LFI, % Predator, API here (**Erreur ! Source du renvoi introuvable.**). The other one is characterized by high values at coastal areas such as IFP, Ratio P/D here (**Erreur ! Source du renvoi introuvable.**). These patterns observed are less visible as F increases. Other indicators results at F = F_{MSY} are in annex (ANNEX VI).

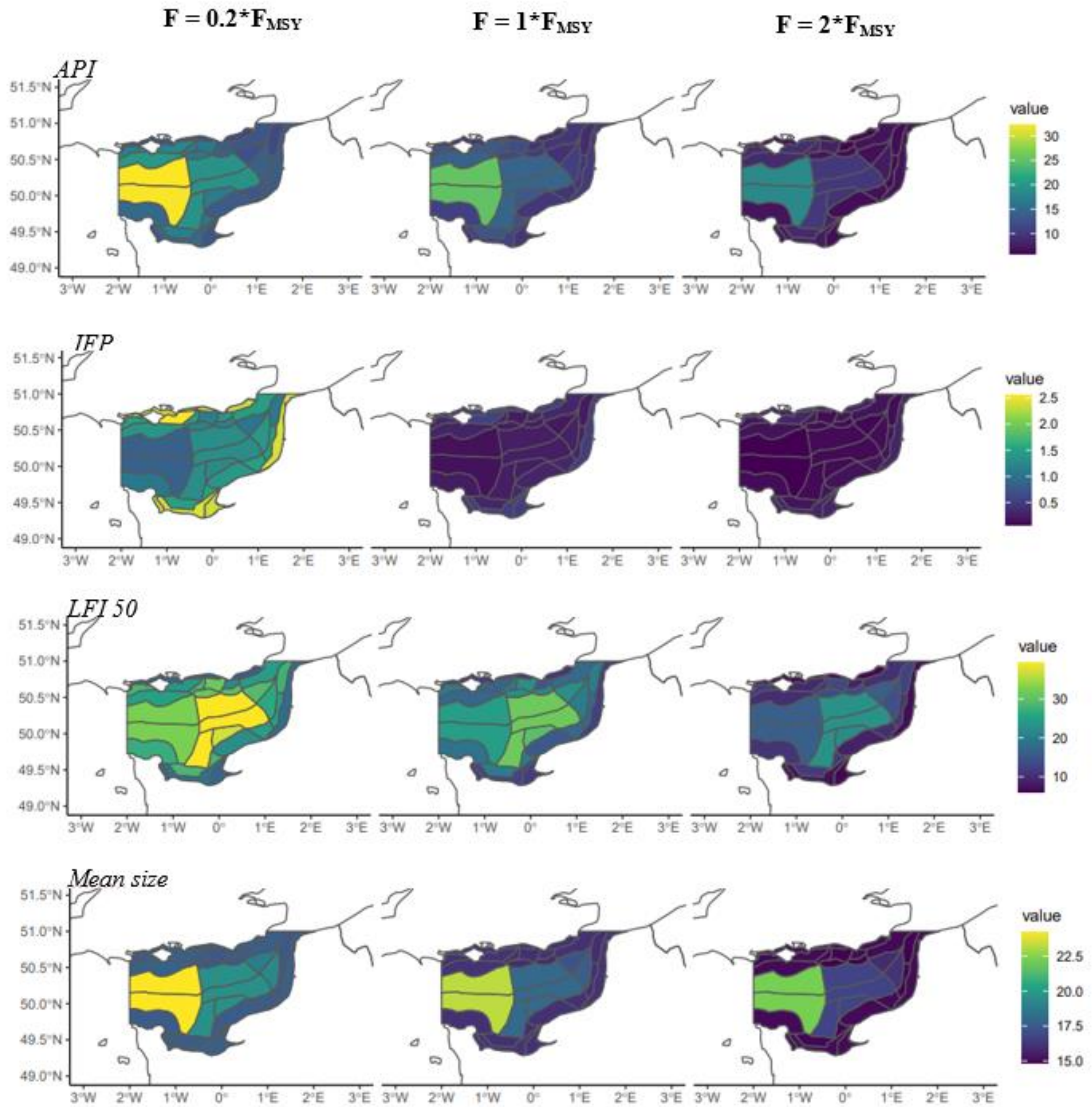

Figure 9: Map of indicator values at different F scenario with Atlantis

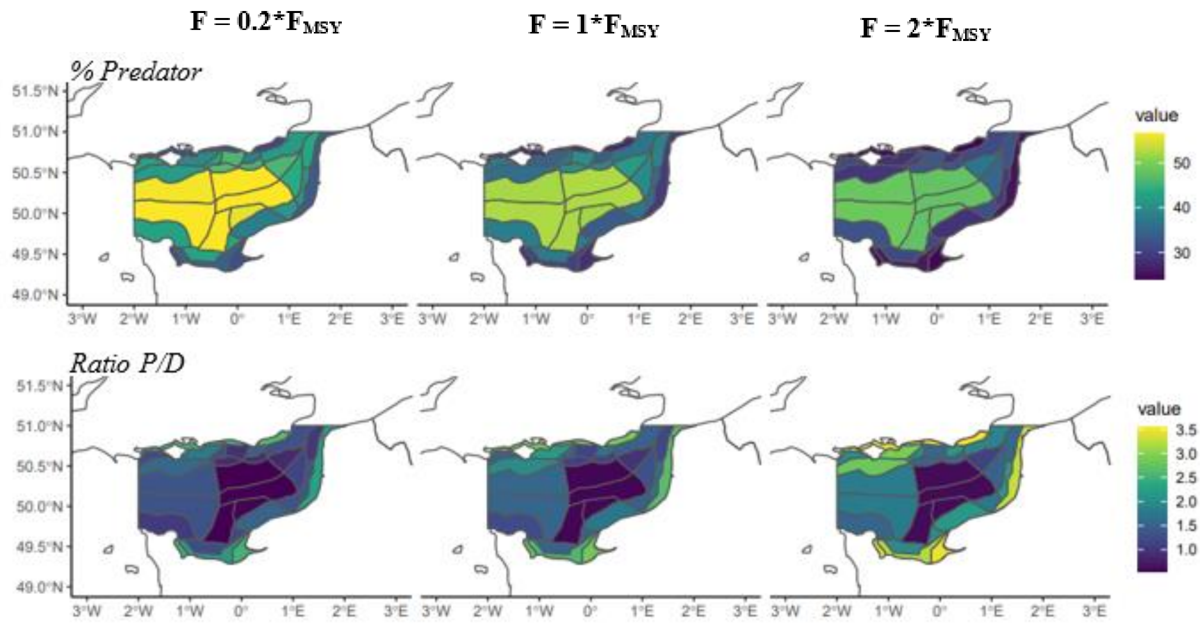

Figure 9: Map of indicator values at different F scenario with Atlantis

With Osmose, we observed homogenous maps for most of indicator (ANNEX VII). For two indicators % Predator and IFP we observe two ranges of values separated by the axis NE/SW. There is more predator and less fishing pressure in the north part. With API we observed more apex predator at north east of the EEC and HTI is characterized by high value at central area and no observation of high trophic level at coast. These patterns observed are less visible as F increases for IFP and API (Figure 10).

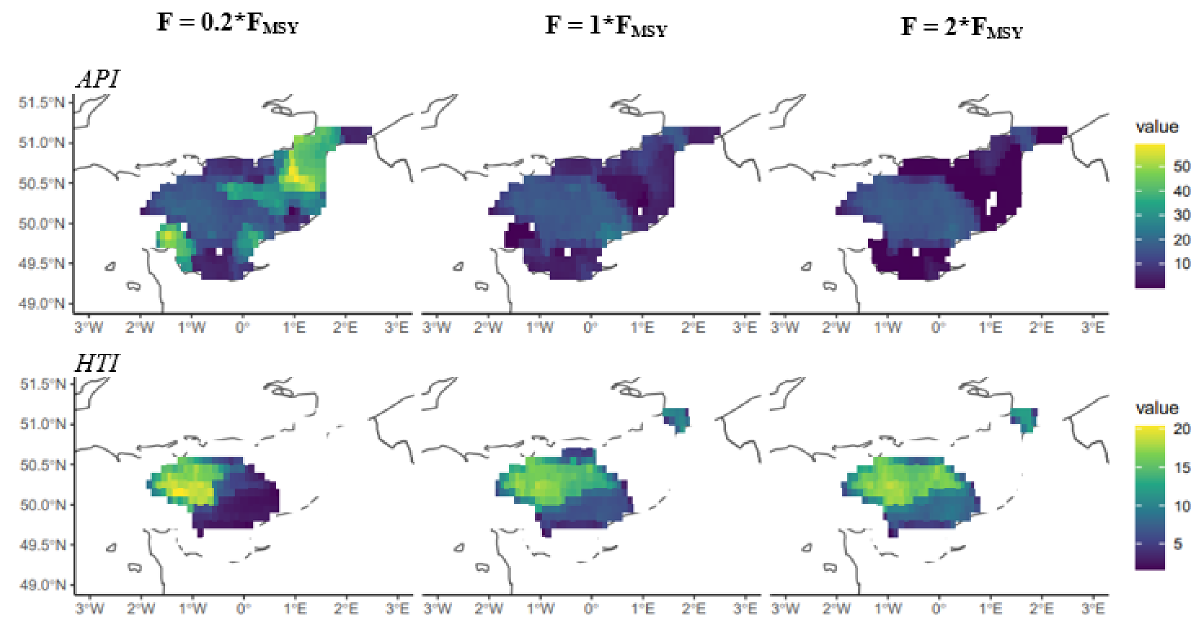

Figure 10: Map of indicator values at different F scenario with Osmose

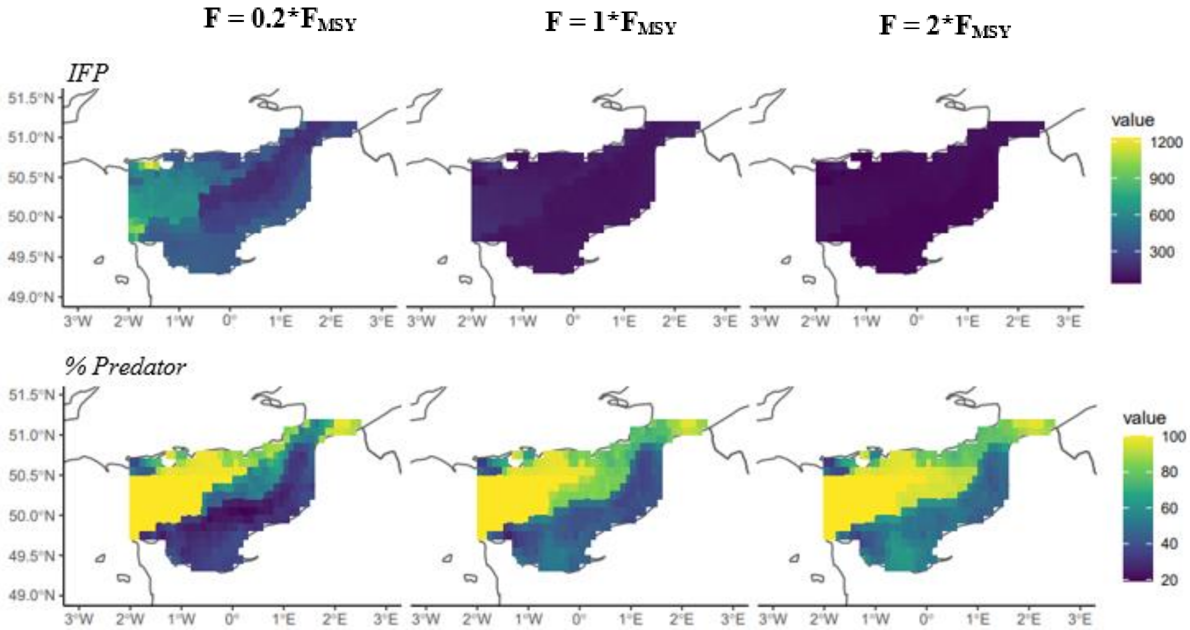

Figure 10: Map of indicator values at different F scenario with Osmose

Comparison of the results between both models

The comparison of spatial pattern between the two models revealed some differences (Figure 9 & Figure 10). With Atlantis patterns are more contrasted than with Osmose. Moreover, there is less indicators with recognizable patterns with Osmose, indicators are more homogeneous, than with Atlantis. With the same indicators such as API, % Predator, IFP, their patterns are different between both models. Furthermore, indicator values seems to diminish with the rising of F at local and global scale (ANNEX VIII & ANNEX IX).

3.2 Spatial indicators performance to detect fishing effects

3.2.1 Sensitivity

We focused on the sign of the slope since slope values are difficult to interpret through the link function (ANNEX X & ANNEX XII). For both model, explained deviance is high when the slope is significant, between 75% and 100%, except for TLc in Altantis and Lifespan / mean size in Osmose where explained deviance was between 50% and 75% (ANNEX XI & ANNEX XIII).

For Atlantis, almost all indicators decrease when the fishing pressure increases (negative slope). Three indicators present a spatial variation of slope sign: Lage increases with the increase of F in central area while it decreases in coastal areas, in contrast to TLc and Ratio PD which show an inversed spatial pattern (Figure 11).

For Osmose, there were more spatial variations (Figure 12). Only IFP, mean size and API significantly decrease with fishing pressure (with a few exception) while TLc and HTI seem to increase with F in the whole area. For the remaining indicators we observed two spatial patterns, an opposition between central and coastal areas for LFI and another pattern characterized by a South/North opposition for % Predator, Lifespan and Ratio P/D. Moreover, lot of GLM did not fit for mean size and % Predator indicator. This were due to large amount of missing values for LFIs and HTI and not to non-significant models. When looking at both model, the most sensitive indicators are IFP, mean size and API.

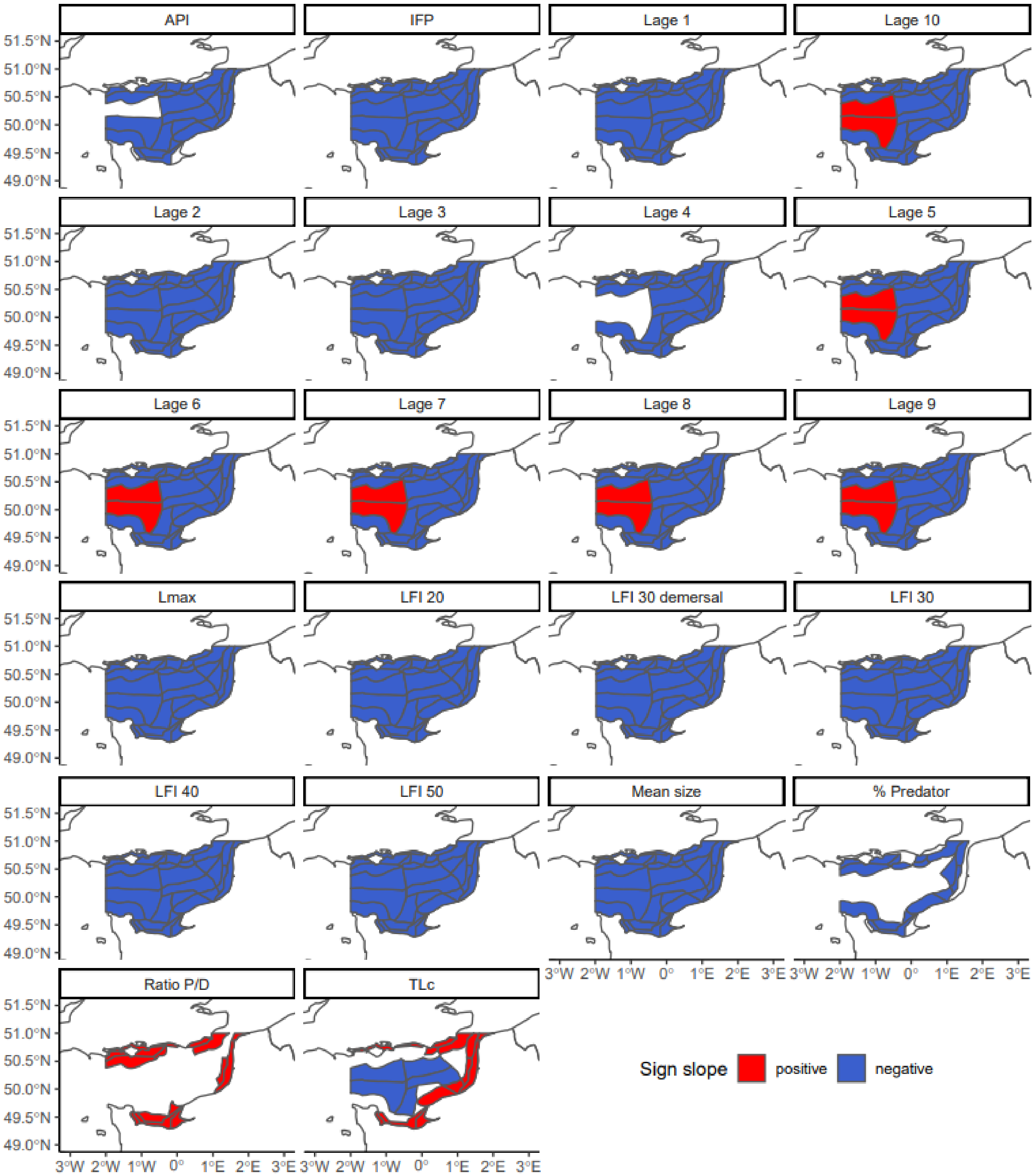

Figure 11: Map of Atlantis indicator sensitivity results, slope sign of our fitting models of indicator value in function of F

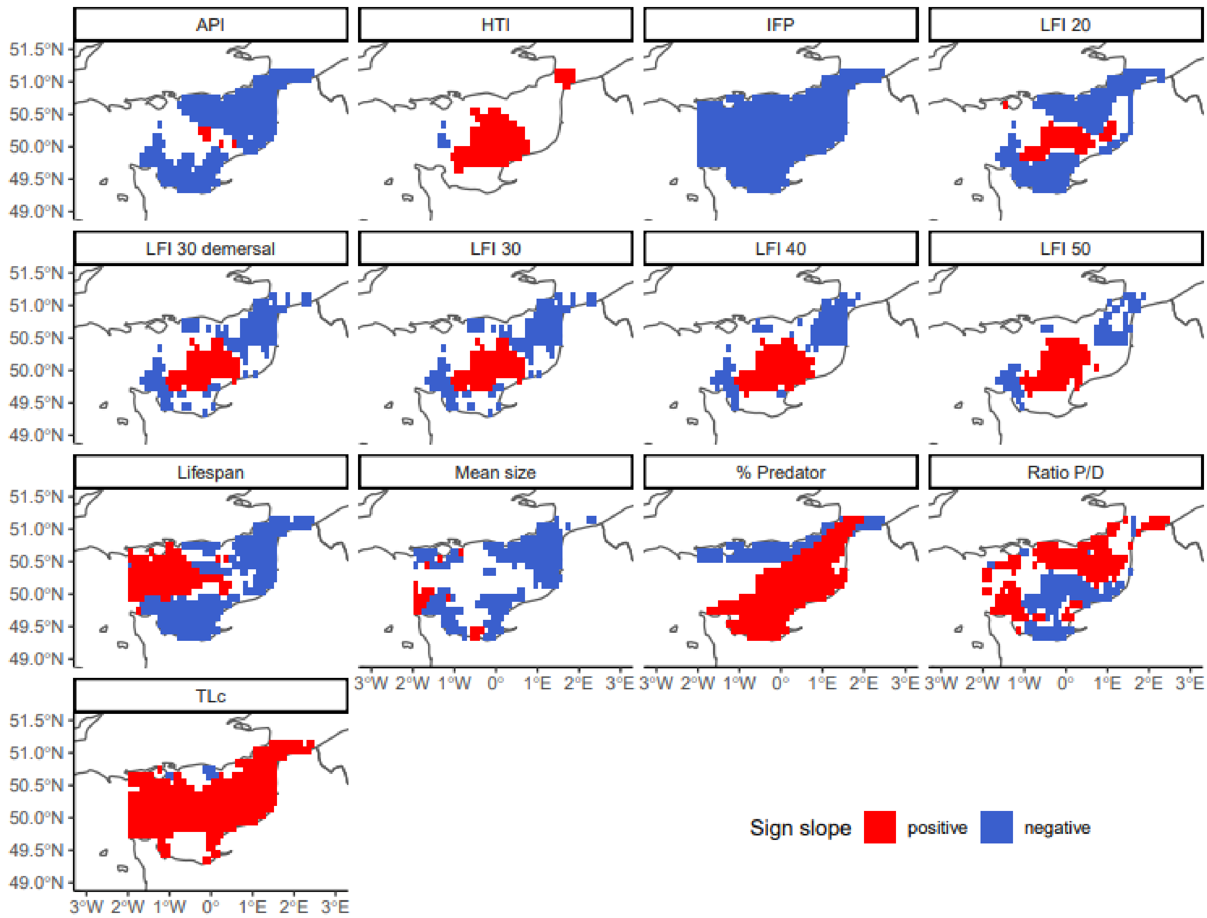

Figure 12: Map of Osmose indicator sensitivity results, slope sign of our fitting models of indicator value in function of F

3.2.2 Responsiveness

To obtain responsiveness maps, we grouped change point value in five categories from '1' which corresponds to a response in one year to '>10' when the response time is more than ten years.

In Atlantis, we see that almost all indicators are very reactive, less than three years to detect the change in the fishing pressure except for three indicators TL_c, L_{max} and L_{age}. The response of TL_c is faster in center area while L_{max} and L_{age} are slower. (Figure 13)

For Osmose, IFP was the most reactive indicator followed by the lifespan. In general, the responsiveness was spatially heterogeneous for the all indicators except for IFP, with a response time superior to 5 years. (Figure 14)

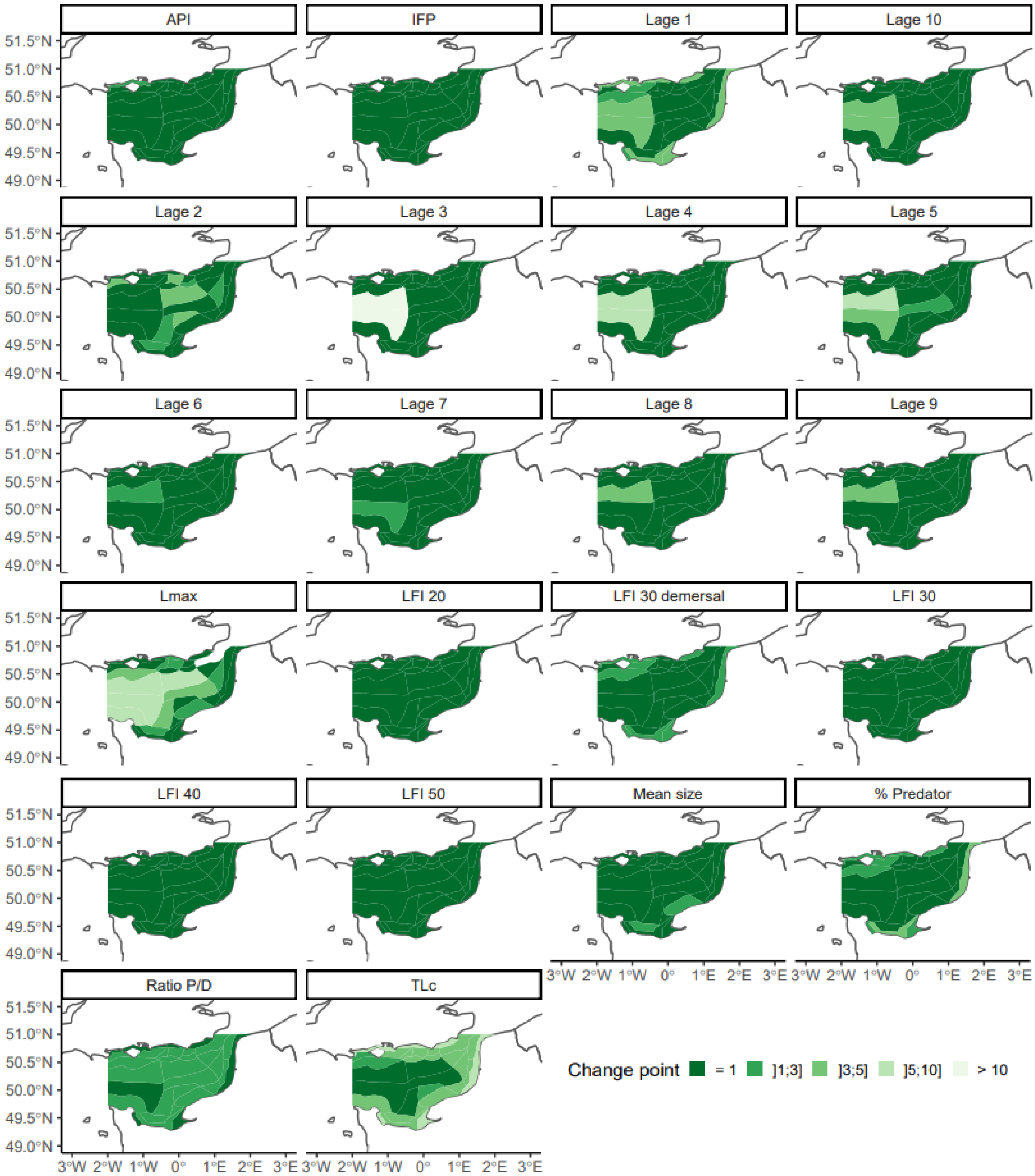

Figure 13: Responsiveness value after F perturbation of indicators with Atlantis model. The intervals represent the number of years after the change of fishing pressure.

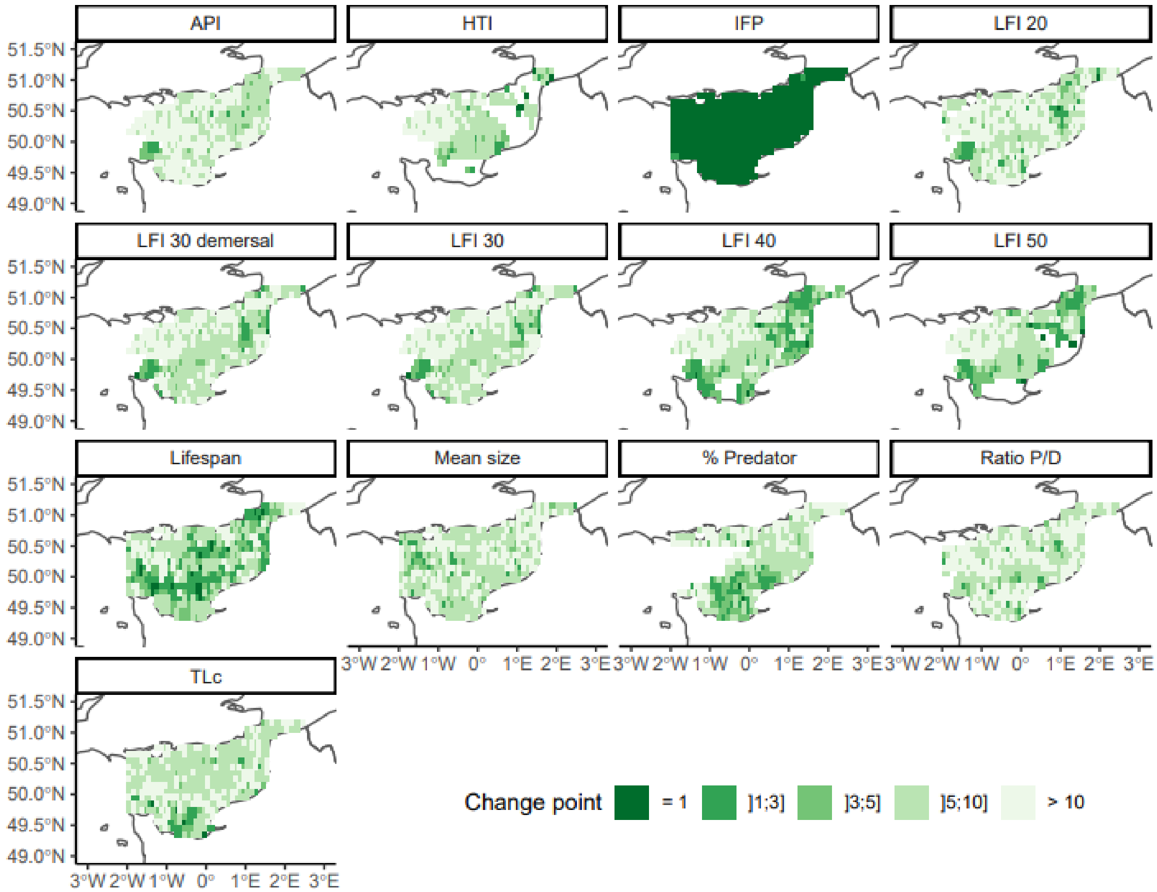

Figure 14: Responsiveness value after F perturbation of indicators with Atlantis model

To explore further responsiveness results, we search if we can identify species assemblage that could explain faster response of our indicators. In this objective, we compare which was initial value for each responsiveness response category. We consider the initial value as mean value of the last 20 years before disturbance for each cell and each indicator. We resume data in one box plot of initial value for each responsiveness category and each indicator (Figure 15 & Figure 16).

With Osmose, we observed that with the increase of predators and large fish with longer lifespan, the responsiveness of indicators is lower, which could suggest an ecosystem more robust to a perturbation at short time scale. We had the same analysis with the Altantis Lmax for large fish and with Ratio P/D with demersal fish but for the other indicator, responsiveness is too fast to analyse boxplot (only one or two category of responsiveness).

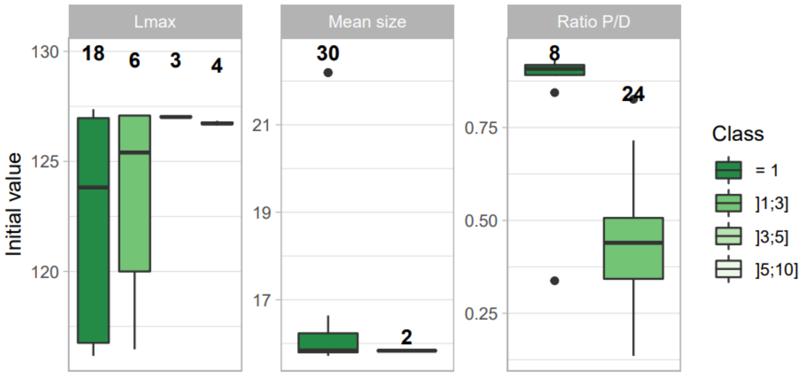

Figure 15: Responsiveness categories in function of initial value (before perturbation) for three indicators of the Atlantis model, count of observation by boxplot in bold

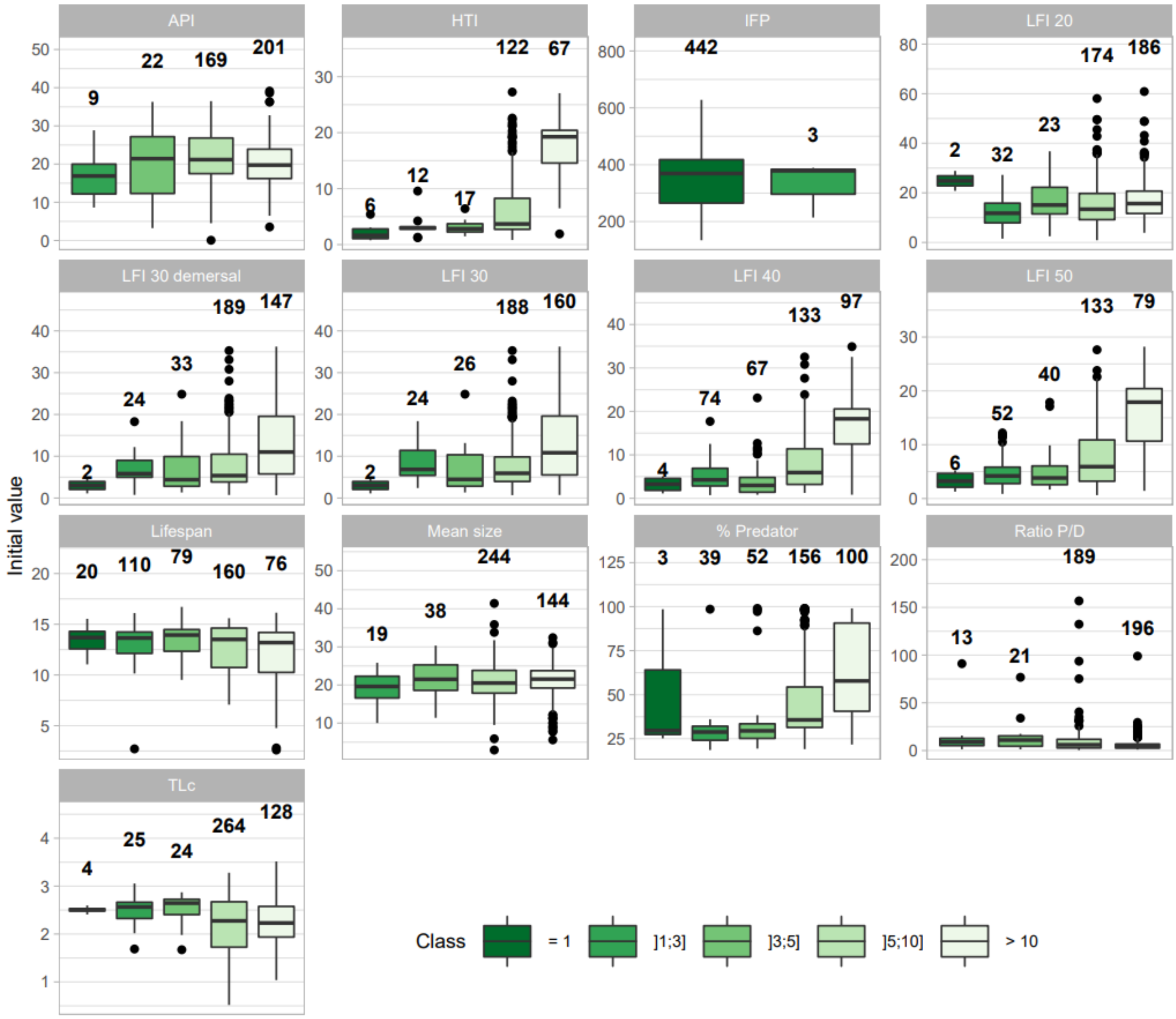

Figure 16: Responsiveness categories in function of initial value (before perturbation) for indicators of the Osmose model, count of observation by boxplot in bold

4. Discussion

Previous studies on indicators performance for detecting the impact of fishing have already been carried out in the EEC, such with the Indiseas project (the IndiSeas Working Group et al. 2012). However, in these studies the indicator’s performance were calculated as an average value over the whole study area.

The objective of our study was to study the potential spatial heterogeneity of the indicators and their performances, by spatializing them using Osmose and Atlantis models.

First, we discuss the results without considering spatialization to contextualize our results within previous studies. This is mainly focused on performance and in particular the sensitivity of the indicators to fishing which has been widely studied in contrast to responsiveness.

4.1 Indicators sensitivity to fishing

For both models we found that IFP, mean size and API were the most sensitive indicators to fishing impact, in coherence with the literature on high trophic level indicators (Bourdaud et al. 2016). Indeed, high trophic species have a smaller productivity and a longer turn over, so they are more impacted by fishing impact (Bourdaud et al. 2016; Gascuel et al. 2016). Moreover, size based indicators are also proven to be sensitive to fishing impact (Shin et al. 2005) as largest fish are specifically targeted by fishing. The main fishing impact lead to a decrease in the biomass of large individuals and therefore in the average size of individuals. Moreover studies show that on a longer scale, the selection pressure on large individuals due to fishing could induce changes in the gene pool and genetic diversity. Younger reproducing individuals with slower growth will be favored (Shin et al. 2005; Jørgensen et al. 2007; Conover, Munch 2002).This was observed with Northeast artic cod, which mature size decreased by about ten centimeters in 70 years and its maturation age arrives two years earlier (Borrell 2013). Furthermore, it has been shown that % predator and lifespan indicators are less sensitive to fishing pressure than indicators derived directly from the catch (Bundy et al. 2010) such as IFP.

However, size indicators should be taken with caution, as they are not specific to the effect of fishing (Shin et al. 2005). As explained above, we expected to see a decrease in the size of individuals due to fishing. However, a decrease in the average size of individuals may also be linked to environmental effects that favour very strong recruitment, or impact negatively growth and large individuals survival (Bundy et al. 2010).

One of the results that emerges from our sensitivity analysis was that the TL_c indicator is the least sensitive to fishing and is even increasing with Osmose model over the whole area. However, according to Pauly's theory of “fishing down the marine food web”, species trophic level caught are expected to decreases over time. In fact, after having exploited the large predators, fisheries are expected to successively switch to smaller species with a lower trophic level. In the study by Bourdaud, it was also shown that this indicator was less effective than the API and HTI indicators (Bourdaud et al. 2016). It is also possible that this increase was an artefact of the Osmose model, linked to the crash of small species.

4.2 Indicators Responsiveness to disturbance

For Atlantis, all indicators were very responsive following the scenario’s disturbance set up. For Osmose, there was a big difference between the IFP, which was very responsive than the other indicators. This is not surprising because when we increased F in our scenarios the catches are directly impacted. Due to the construction of the models, Atlantis would be more responsive to disturbances; it is a deterministic model with each species homogeneous in the same cohort, unlike Osmose, which is stochastic.

4.3 Indicators spatialization

We sorted the indicators into three cases using a decision tree (Figure 17) according to the patterns of their response.

Case 1: the spatialized indicator is homogeneous over the whole study area with both models. In this case, we consider that spatialization of the indicator does not add any additional information to the average value computed over the whole area. This is the case here for the TL_c indicator.

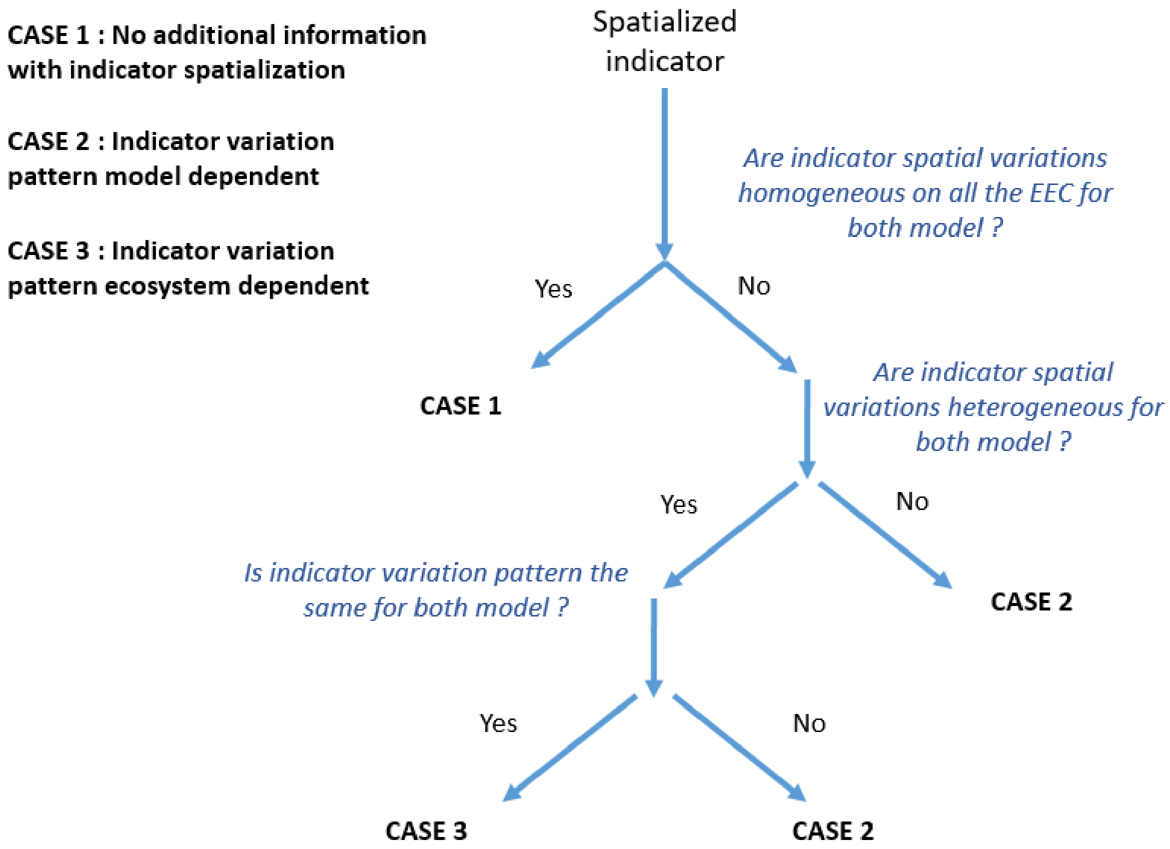

Figure 17: Decision tree to determine the different cases of spatial variation of the indicators with its implications for each case

Case 2: indicator is heterogeneous on the area for both models but with a different pattern or is homogeneous with one model and heterogeneous with the other. The majority of our indicators are in this case, such as IFP, Mean size, % Predator, P/D Ratio. These indicators will be categorised as model dependent, i.e. spatial variation are partly driven by model structure and assumptions. The main patterns observed with Osmose were an opposition on the NE/SW axis for IFP and % Predator when we mainly observe an opposition between the central zone and the coasts with Atlantis for almost all indicators. For Osmose, the input maps that constrain the distribution of species could explain this clear separation. Indeed, this opposition was found for two pelagic species that are very abundant in the ecosystem, herring and sardine (ANNEX XIV). For Atlantis, this opposition between the coast and the central zone is explained by the fact that the majority of juveniles were on the coast, which will influence the average size, and that the two central polygons (2 and 3) are those with the least species diversity.

Case 3: indicator present a similar spatialization pattern between the two models. In this case, the ecosystem is considered to be what led to this pattern or at least both models capture the ecosystem characteristics the same way. The LFIs and API indicators are found in this case, with the same pattern of opposition between the central zone and the coast. This difference in the quantity of large predators and large individuals between the coasts and the center of the EEC is not surprising. Indeed, the smallest and lowest individuals on the trophic scale, such as juveniles, are mainly found on the coasts, particularly near estuaries (Rochette et al. 2010). Moreover, the pattern observed for API is very close to the sediment distribution map (Larsonneur, Bouysse, Auffret 1982). API is highest where there are pebbles.

The last three indicators are Lmax Lage and Lifespan which were calculated for only one model each time, so the decision tree cannot be used. However, according to our results the response of Lmax is homogeneous over the whole area for each scenario, so we can assume that this indicator does not have to be spatialized. For L_{age} and Lifespan there is heterogeneous distribution of values, we cannot conclude between case 2 or 3.

With Osmose, we observe that the indicators are generally less sensitive in the central zone of the EEC except for the API and IFP indicators. This could be an indication that this part of the ecosystem these indicators have a lower performance in this area. In the case of Atlantis, L_{age} also performs worse in the central area, while the reverse is true for Ratio P/D and TL_c, which perform worse at the coast. The rest of the indicators are uniformly sensitive on the EEC.

With Atlantis, we observe that most of the indicators are very reactive uniformly over the whole areas. TL_c, Lmax and Lage whose show lower and heterogeneous performance, following the opposition central area and coast pattern. With Osmose, the indicators are less performant than with Atlantis as we have seen previously; they vary heterogeneously over the whole of the EEC without any discernible pattern. These results are driven by our model assumptions. Indeed, Atlantis is a mechanistic model with a lower resolution so reactivity will be fast and spatially contrasted while Osmose is a stochastic model with a higher resolution and higher interactions between species so it will react slower (stochasticity will smooth disturbance reaction) and homogeneously. In the exploratory part for responsiveness, we found that the responsiveness is longer when predators are larger with longer lifespan, which could suggest an ecosystem more resistant to a disturbance at short time scale. Indeed, it has been shown that diversified systems with high functional redundancy (apex predator in our case) gain in resilience (Downing et al. 2012).

4.4 Limits

This study has some limitations. Indeed, we compared two models that did not have the same species and we tried to restrict this impact by limiting ourselves to the study of the exploited species. Moreover, if we consider our models calibration, no hindcast has been done only a climatological calibration was performed. There is also a lack of data on certain component of the ecosystem (benthos components are not well represented). Even if the EEC is a well documented and studied area (Travers-Trolet, Shin, Field 2014), the models are not completely up-to-date as the Osmose model is based on data from 2000 to 2009 and the Atlantis model is based on data from 2002 to 2011. It is possible that the parameter values have changed in ten years, especially with the impacts of climate change and the impact of human activities in the EEC (Dauvin 2012).

Moreover, we observed that for the IFP indicator there is a large difference in magnitude between the two models. The IFP indicator for Osmose appear to be abnormally high, in the order of a hundred on average (ANNEX IX). With Osmose, the outputs accumulate all the individuals older than 0.5 years, so there are many juveniles taken into account in the biomass of the stock even though they have not yet been recruited, which would favour high IFP values.

Finally, the most limiting part of our study is that our models do not model fisheries fleet separately but only one F per species which is not realistic in EEC where fisheries are mostly multi-species and multi-gears (Lehuta, Vermard, Marchal 2015). It would have been possible to compensate for this problem in our scenarios by varying the multiplier for the first scenario differentially between species. However, we did not have enough time to implement this type of scenario. If fisheries were to be implemented in the models, the distribution of fishing effort should be studied in a spatialized manner in order to be able to identify some of the most vulnerable areas to fishing. In reality, depending on species distribution and their port of origin, fishers will not fish uniformly throughout the area.

Furthermore, it would be interesting to deepen our scenarios by carrying out spatialized management scenarios. For example, it would be possible to carry out a scenario that proposes area closures to fishing or limitations and see what spatial impact this might have on our indicators.

5. Conclusion

During this study our objective was to investigate the performance of spatialized indicators in capturing the impacts of fishing in the Eastern Channel with an ecosystem management objective using a multi model approach. This study is part of 4.4 tasks of SeaWise project focusing on food web and community diversity.

We have shown the importance of the spatial study of the indicators to study their performance, in particular with the multi-model approach which has made it possible to show at what extent certain indicators could be dependent on the construction of the model and its hypotheses. The most relevant indicators will be those that do not depend on the construction of the model. Indeed, indicators that are very efficient but very dependent on the model will be precise but the results will not necessarily be accurate, such as IFP, whereas indicators that are ecosystem dependent but less efficient will be less precise but more accurate. In this category, two indicators stand out, API and LFIs. Among the LFIs, LFI at 40 cm and at 50 cm are the most performant as the other LFIs. Thus, in the remainder of this study we recommend the use of these two indicators.

However, to assess human impacts on ecosystems a single indicator is not sufficient; a complementary set of indicators (the IndiSeas Working Group et al. 2012) is needed to take management decision. Furthermore, after identifying these indicators, specific thresholds for each indicator must be found to allow managers to assess the good ecological status of ecosystems. These are difficult to find because thresholds vary according to each ecosystem and threshold values change with climate change (Travers-Trolet et al. 2020). For example, overexploited cold-water species have reference points that decline with climate change (Travers-Trolet et al. 2020).

In order to continue this study, it would be interesting to carry out new scenarios with more developed models that allow fishing effort to be spatialized and fisheries to be taken into account. It would be interesting to carry out the same study on other ecosystems to compare indicators' responses.

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ANNEX

ANNEX I: Resume of the main management measures to conserve fisheries resources and to protect marine ecosystems in the Eastern English Channel

Regulation level	**Type of control**	**Target species**	**Management measures**	**Objectives**	**Source**
European Union	Control mortality rate	Most of commercial species*****	TACs and Quotas	Stocks within biologically sustainable levels	(Biseau 2020)
		All species with TACs	Landing Obligation	To eliminate discards	(Commission Delegated Regulation 2020)
		Nephrops, Sole, Plaice, Skates, red seabream	Landing exemption based on survival rate	Provide flexibility for the landing obligation	
		Megrim, whiting, mackerel, horse mackerel, blue whiting	Landing exemption de minimis	Provide flexibility for the landing obligation	
	Control resource access	Demersal species	AEP Autorisation Européenne de pêche	Protect and conserve community dynamics	Règlement (CE) n°1954/2003 & Arrêté du 28 décembre 2012
		Cod	AEP Autorisation Européenne de pêche	Stock rebuilding and recovery	Règlement (CE) n°1342/2008 & Arrêté du 27 mai 2016
	Technical measures	All commercial fish	Mesh size + type of mesh (square mesh…)	Improving selectivity	Regulation (EU) 2019/1241 of the European Parliament and of the Council of 20 June 2019
		All commercial fish	Type of gear (beam trawl, gillnets, …)	Improving selectivity	
		Most of commercial fish******	Minimal size	Improving selectivity	
National	Control resource access	Sensitive species of an ecosystem	MCZ, MPA (EPMO)***	Protect and conserve species and habitat	(Ifremer 2018b; CRPMEM Hauts de France 2016)
		Sole	Maximum fishing effort for trawl and gillnets	Protect and conserve species	article 3.1 et 3.2 §VII annexe IV arrêté ministériel 27 mai 2016
		Sole	Fishing prohibition in Sole nurseries***	Protect and conserve species and habitat	article 3.1 et 3.2 §VII annexe IV arrêté ministériel 27 mai 2016
Regional	Control resource access	Sole, Mackerel	Derogatory measures : authorization to fish with trawl in 3 miles area***	Economic viability of the fishery	arrêté 58/2007
		Plaice	Derogatory measures : authorization to fish in Sole nursery at 1st April to 30th June***	Economic viability of the fishery	arrêté préfectoral n°79/2020

* Cod (*Gadus morhua*), Whiting (*Merlangius merlangus*), Sea bass (*Dicentrarchus labrax*), Red mullet (*Mullus barbatus*), Lemon sole (*Microstomus kitt*), Witch flounder (*Glyptocephalus cynoglossus*), Grey gurnard (*Eutrigla gurnardus*), Lesser spotted dogfish (*Scyliorhinus canicula*), Blonde ray (*Raja brachyura*), Thornback ray (Raja clavata), Spotted ray (*Raja montagui*), Small-eyed ray (*Raja microocellata*), Undulate ray (*Raja undalata*), Brill, Sole (*Solea* spp.), Plaice (*Pleuronectes platessa*)

** Cod (*Gadus morhua*) Haddock (*Melanogrammus aeglefinus*) Saithe (*Pollachius virens*) Pollack (*Pollachius pollachius*) Common hake (*Merluccius merluccius*) Megrim (*Lepidorhombus* spp.) Sole (*Solea* spp.) Common plaice (*Pleuronectes platessa*) Whiting (*Merlangius merlangus*) Northern ling (*Molva molva*) Blue ling (*Molva dypterygia*) Norway lobster (*Nephrops norvegicus*) Mackerel (*Scomber* spp.) Herring (*Clupea harengus*) Horse mackerel (*Trachurus* spp.) Anchovy (*Engraulis encrasicolus*) Sea bass (*Dicentrarchus labrax*) Sardine (*Sardina pilchardus*) Red seabream (*Pagellus bogaraveo*) Lobster (*Homarus gammarus*) Spider crab (*Maja squinado*) Lapwing (*Chlamys* spp.) Clam (*Ruditapes decussatus*) Clamshell (*Venerupis pullastra*) Japanese clam (*Venerupis philippinarum*) Clam (*Venus verrucosa*) Red clam (*Callista chione*) Razor clam (*Ensis* spp.) Solid clam (*Spisula solida*) Sea olive (*Donax* spp.) Ceratisole pod (*Pharus legumen*) Whelk (*Buccinum undatum*) Octopus (*Octopus vulgaris*) Spiny lobster (*Palinurus* spp.) Pink shrimp (*Parapenaeus longirostris*) Edible crab (*Cancer pagurus*) Scallop (*Pecten maximus*)

***Spatialized management measures, MCZ: Marine Conservation Zone, MPA: Marine Protected Area, EPMO: Parc naturel marin Estuaires picards et de la mer d'Opale

ANNEX II: Species characteristics (predation, group, migration) in Atlantis model A :List of Atlantis exploited species and their characteristics source : (Raphaël Girardin et al. 2018; Froese, Pauly 2022; Palomares, D. Pauly. 2022)B: Migrating species and period of migration in Atlantis

A

Species/Group of Species	**Predator**	**Group**
Atlantic Cod	Yes	Demersal
Bivalves	No	Benthic
Cephalopod	Yes	Pelagic
Clupeidae	No	Pelagic
Common Dab	Yes	Demersal
Common Sole	No	Demersal
Crabs	Yes	Benthic
European Seabass	Yes	Demersal
Gurnards	Yes	Demersal
Large Bottom fish	Yes	Demersal
Lobsters	Yes	Benthic
Mackerels	Yes	Pelagic
Mugilidae	Yes	Demersal
Other flatfish	Yes	Demersal
Other Gadoids	Yes	Demersal
Plaice	Yes	Demersal
Pollack	Yes	Demersal
Rays and Dogfish	Yes	Demersal
Scallops	No	Benthic
Sharks	Yes	Pelagic
Shrimps	No	Benthic
Small demersal fish	No	Demersal
Sparidae	Yes	Demersal
Whelks	No	Benthic
Whiting	Yes	Demersal

B

Species/Group of Species	**Period of migration**
Cephalopod	Septembre-Mars
Whiting	Avril-Octobre
European Seabass	Septembre-Mars
Clupeidae	Février-Septembre
Sparidae	Octobre-Avril

ANNEX III: List of Osmose exploited species and their characteristics (source: Travers-Trolet et al. 2019; Froese, D. Pauly. 2022; Palomares, D. Pauly. 2022)

Species	**Latin name**	**Group**	**Predator**	**Lifespan**
Cod	*Gadus morhua*	Demersal	Yes	25
Herring	*Clupea harengus*	Pelagic	No	11
HorseMackerel	*Trachurus* spp.	Pelagic	Yes	15
Lesser Spotted Dogfish	*Scyliorhinus canicula*	Demersal	Yes	10
Mackerel	*Scomber* spp.	Pelagic	Yes	17
Plaice	*Pleuronectes platessa*	Demersal	Yes	15
Pouting	*Trisopterus luscus*	Demersal	Yes	4
Red Mullet	*Mullus surmuletus*	Demersal	Yes	11
Sardine	*Sardina pilchardus*	Pelagic	No	15
Sole	*Solea* spp.	Demersal	No	20
Squids	*Lolignidae*	Pelagic	Yes	2
Whiting	*Merlangius merlangus*	Demersal	Yes	20

ANNEX IV: a,b parameters from size-weight relationship for Atlantis species and groups (Audzijonyte et al. 2017)

Species	**a**	**b**
Atlantic Cod	0.00835	3.0532
Clupeidae	0.00564	3.0576
Common Dab	0.00547	3.2211
Common Sole	0.00391	3.2639
European Seabass	0.01244	2.9529
Gurnards	0.00528	3.1407
Large Bottom fish	0.03328	2.7659
Mackerels	0.00338	3.1085
Mugilidae	0.00756	3.0574
Other flatfish	0.01018	3.0514
Other Gadoids	0.00728	3.1333
Plaice	0.0103	3.0169
Pollack	0.00613	3.1153
Rays and Dogfish	0.003048	3.1783
Sharks	0.00273	3.1533
Small demersal fish	0.0123	2.8092
Sparidae	0.00982	3.1414
Whiting	0.00621	3.1028

ANNEX V: Chosen distribution law to Atlantis model indicators (A) and Osmose model indicators (B) with range values for each indicators

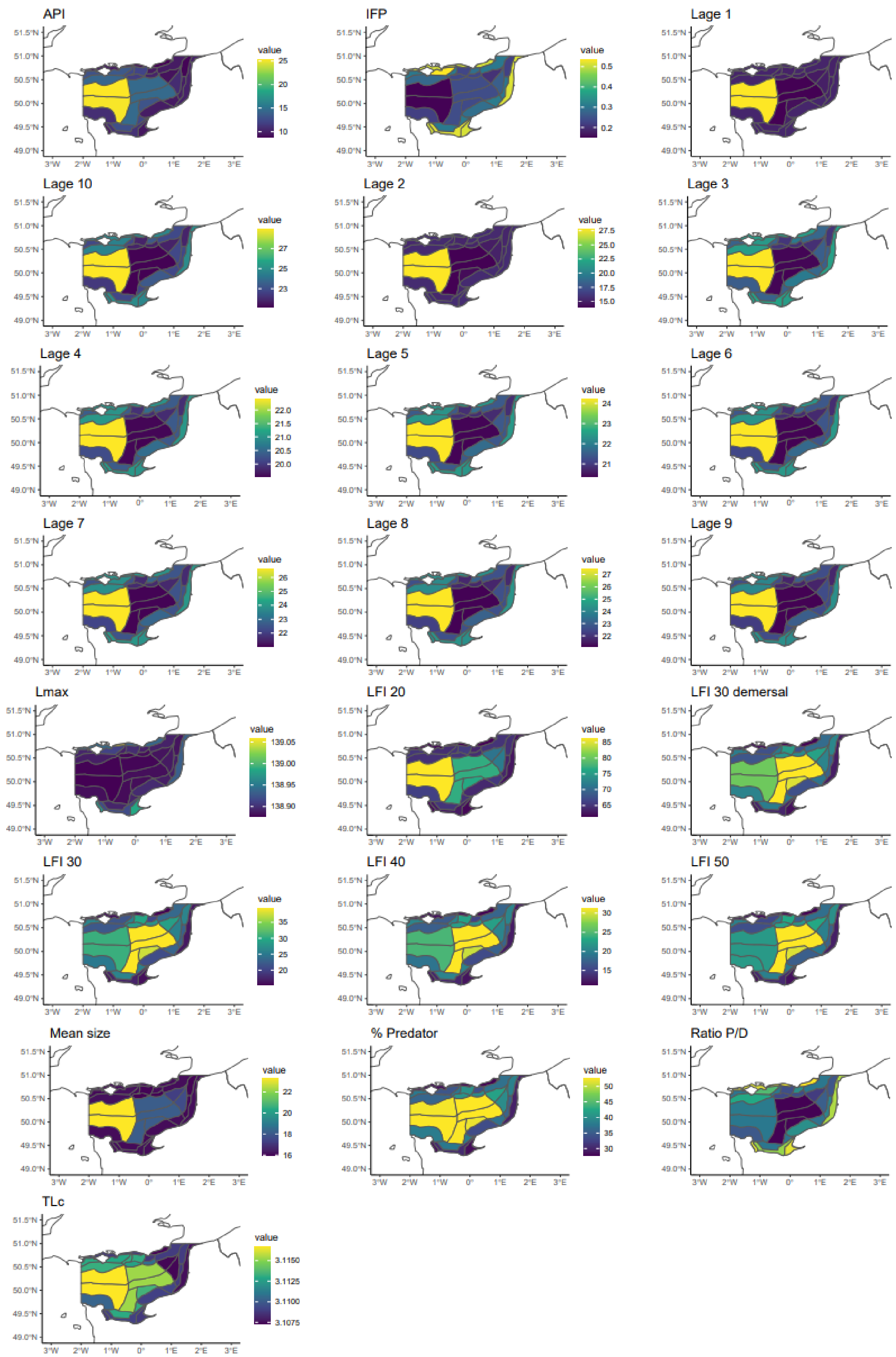
A

Indicators Atlantis	**Range**	**Law**
API	2.86E-02-3.23E-01	Binomial/Quasibinomial
HTI	NA	Binomial/Quasibinomial
IFP	6.85E-02-2.55E+00	Gamma
Lage	1.09E+01-3.06E+01	Gamma
LFI 20	4.53E-01-9.32E-01	Binomial/Quasibinomial
LFI 30	1.69E-01-5.63E-01	Binomial/Quasibinomial
LFI 30 demersal	1.06E-01-4.73E-01	Binomial/Quasibinomial
LFI 40	9.75E-02-4.71E-01	Binomial/Quasibinomial
LFI 50	6.10E-02-3.93E-01	Binomial/Quasibinomial
Lmax	1.23E+02-1.72E+02	Gamma
Mean size	1.49E+01-2.44E+01	Gamma
% Predator	7.24E-02-5.66E-01	Binomial/Quasibinomial
Ratio P/D	5.48E-01-3.54E+00	Gamma
TL_c	3.11E+00-3.13E+00	Gamma

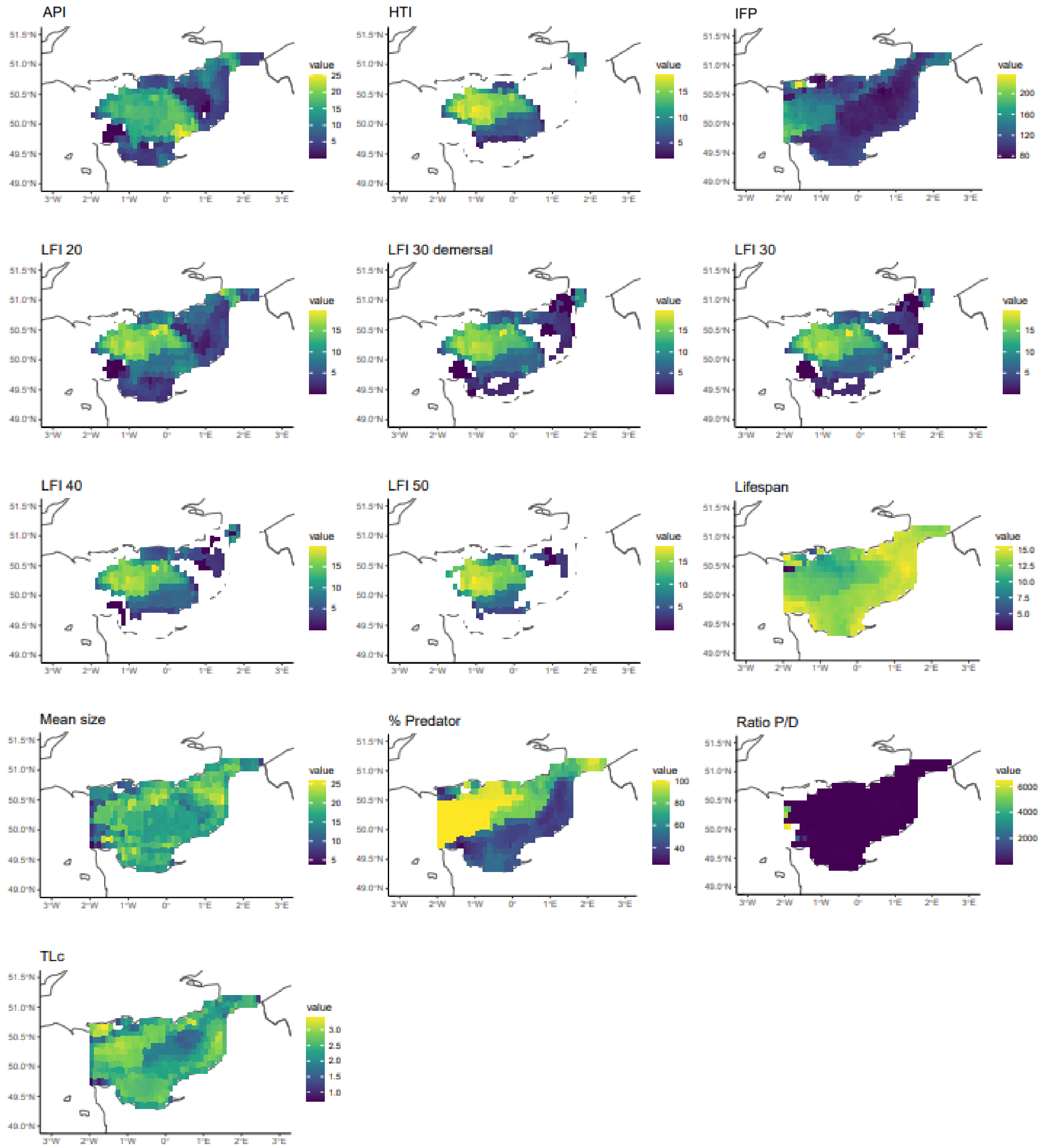
B

Indicators Osmose	**Range**	**Law**
API	1.63E-04-7.37E-01	Binomial/Quasibinomial
HTI	5.89E-03-2.58E-01	Binomial/Quasibinomial
IFP	4.06E+01-1.23E+03	Gamma
LFI 20	1.80E-04-7.49E-01	Binomial/Quasibinomial
LFI 30	1.79E-04-7.49E-01	Binomial/Quasibinomial
LFI 30 demersal	1.79E-04-7.49E-01	Binomial/Quasibinomial
LFI 40	1.95E-04-7.49E-01	Binomial/Quasibinomial
LFI 50	3.13E-04-4.92E-01	Binomial/Quasibinomial
Lifespan	2.40E+00-2.18E+01	Gamma
Mean size	1.63E+00-4.01E+01	Gamma
% Predator	1.52E-01-1.00E+00	Binomial/Quasibinomial
Ratio P/D	1.52E-01-1.15E+04	LogNormal
TL_c	3.89E-01-3.46E+00	Gamma

ANNEX VI: Map representing indicator values at F= F_{msy} scenario for Atlantis



ANNEX VII: Map representing indicator values at F= F_{msy} scenario for Osmose



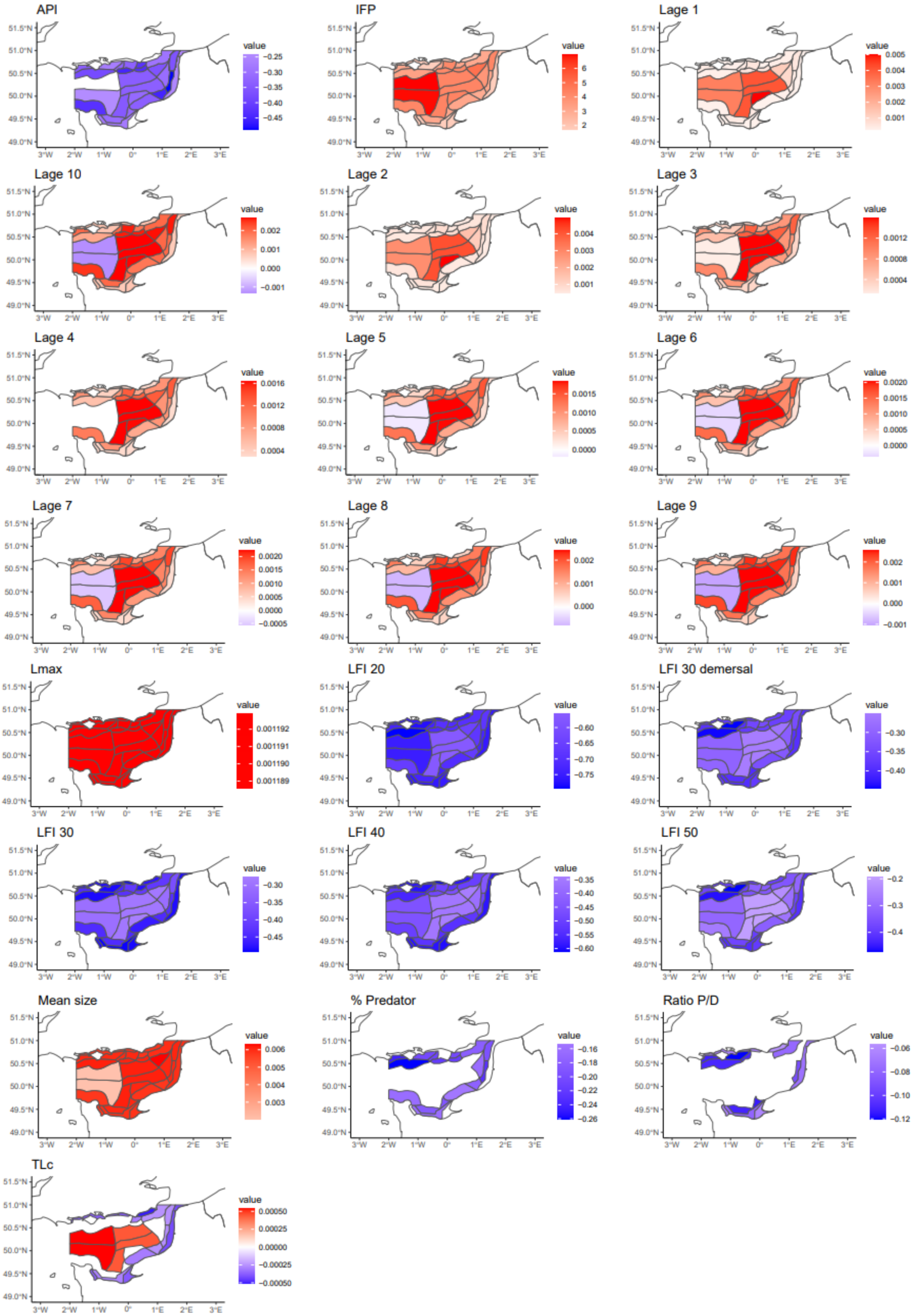
ANNEX VIII: Mean value of indicators for each scenario with Atlantis

F	0	0,2	0,4	0,6	0,8	1	1,2	1,4	1,6	1,8	2
API	11,23	16,2	15,1	14,05	13,08	12,07	11,2	10,35	9,54	8,86	8,19
IFP	NA	1,59	0,8	0,54	0,41	0,33	0,28	0,24	0,21	0,19	0,18
Lage	21,67	20,87	20,78	20,7	20,62	20,54	20,49	20,44	20,39	20,34	20,29
LFI 20	172,36	164,31	157,08	150,27	144,18	138,91	135,06	131,54	128,4	125,57	123,15
LFI 30	82,51	79,14	76,47	73,58	70,63	67,56	64,53	61,48	58,47	55,73	53
LFI 30 dem	38,53	41,1	39,4	37,46	35,87	33,91	32,38	30,93	29,49	27,48	25,77
LFI 40	28,45	31,89	30,09	28,37	26,6	24,85	22,93	21,41	19,71	18,44	17,19
LFI 50	24,24	26,26	24,5	22,82	21,19	19,21	17,78	16,39	14,28	13,15	12,09
Lmax	27,8	32,44	31,12	29,45	28,06	26,27	24,89	23,57	22,24	20,35	18,78
Mean size	19,2	18,36	17,94	17,56	17,19	16,87	16,59	16,34	16,12	15,91	15,73
% Predator	31,95	42,63	41,48	40,35	39,24	38,04	37,03	35,97	34,8	33,9	32,89
Ratio P/D	1,92	1,45	1,5	1,55	1,61	1,69	1,75	1,84	1,94	2,02	2,11
TL_c	NA	3,12	3,11	3,11	3,11	3,11	3,11	3,11	3,11	3,12	3,12

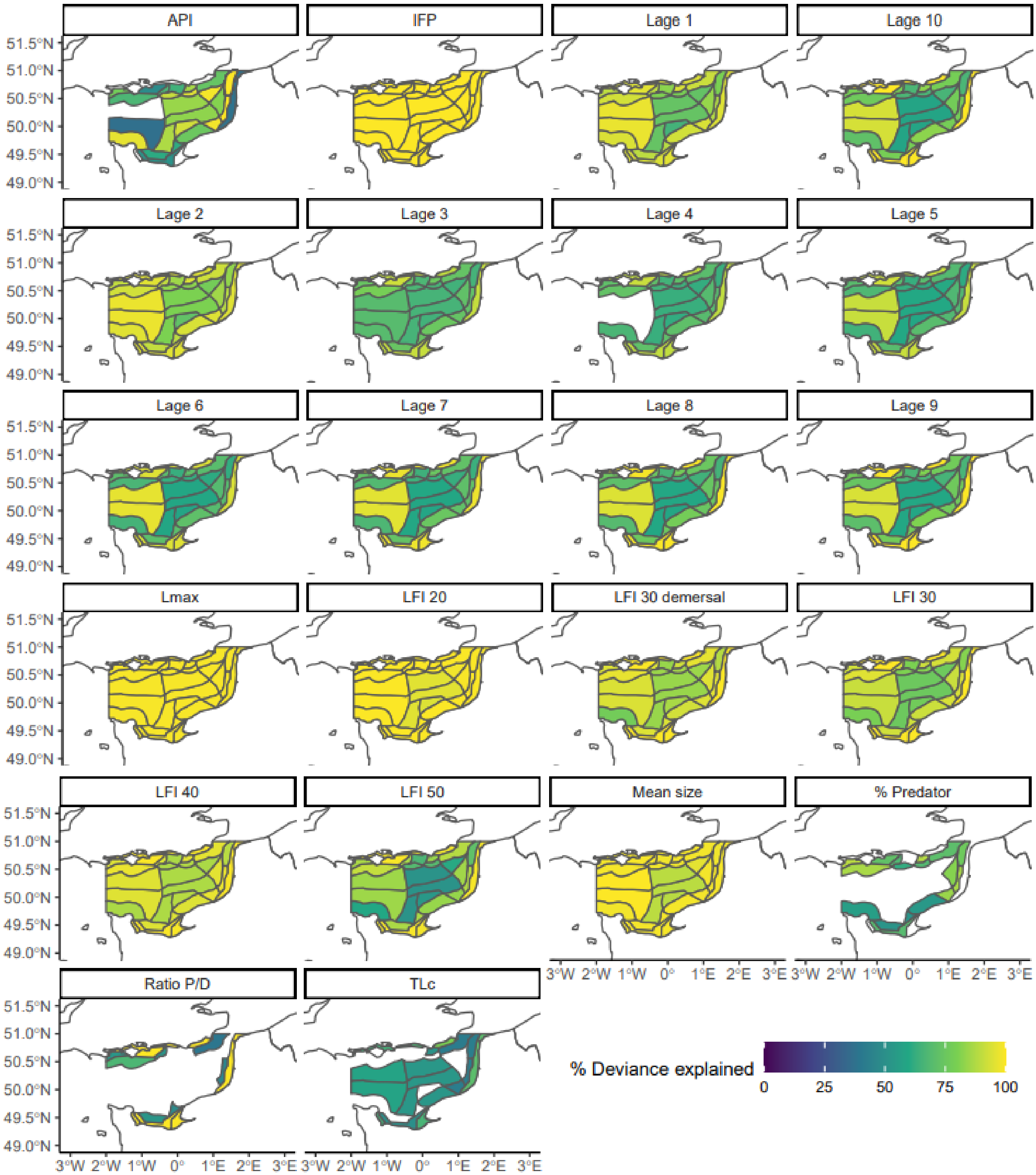
ANNEX IX: Mean value of indicators for each scenario with Osmose

F	0	0,2	0,4	0,6	0,8	1	1,2	1,4	1,6	1,8	2
API	29,1	22,09	16,47	13,21	12,01	10,27	9,66	8,82	7,94	7,63	7,83
HTI	13,96	8,75	6,94	9,22	10,15	10,41	10,49	10,78	11,03	11,03	11,62
IFP	NA	398,82	228,69	166	131,95	113,09	98,67	87,64	79,8	71,84	67,22
LFI 20	20,98	14,48	11,95	10,03	9	7,87	7,2	6,77	8,92	11,22	12,3
LFI 30	14,27	10,88	8,62	7,52	7,44	7,52	8,35	8,56	8,81	9,4	10,98
LFI 30 demersal	14,27	10,88	8,62	7,52	7,44	7,52	8,35	8,56	8,81	9,4	10,98
LFI 40	12,59	11	8,73	7,99	7,68	8,66	9,09	9,44	10,37	11,13	11,68
LFI 50	12,01	10,65	7,93	7,86	7,75	9,98	11,05	11,25	8,8	6,62	5,81
Lifespan	13,33	13,12	13,26	12,99	13,1	12,99	12,94	12,93	12,98	12,77	12,96
Mean size	17,12	17,83	19,73	19,63	19,17	17,29	16,67	16,39	15,96	15,71	15,4
% Predator	59,89	60,5	64,84	66	66,75	68,9	69,62	70,3	71,51	72,92	73,59
Ratio P/D	6,89	8,69	12,77	19,07	32,22	80,38	16,92	23,32	31,31	48,71	9,14
TL_c	NA	1,99	2,08	2,26	2,34	2,4	2,43	2,47	2,51	2,5	2,6

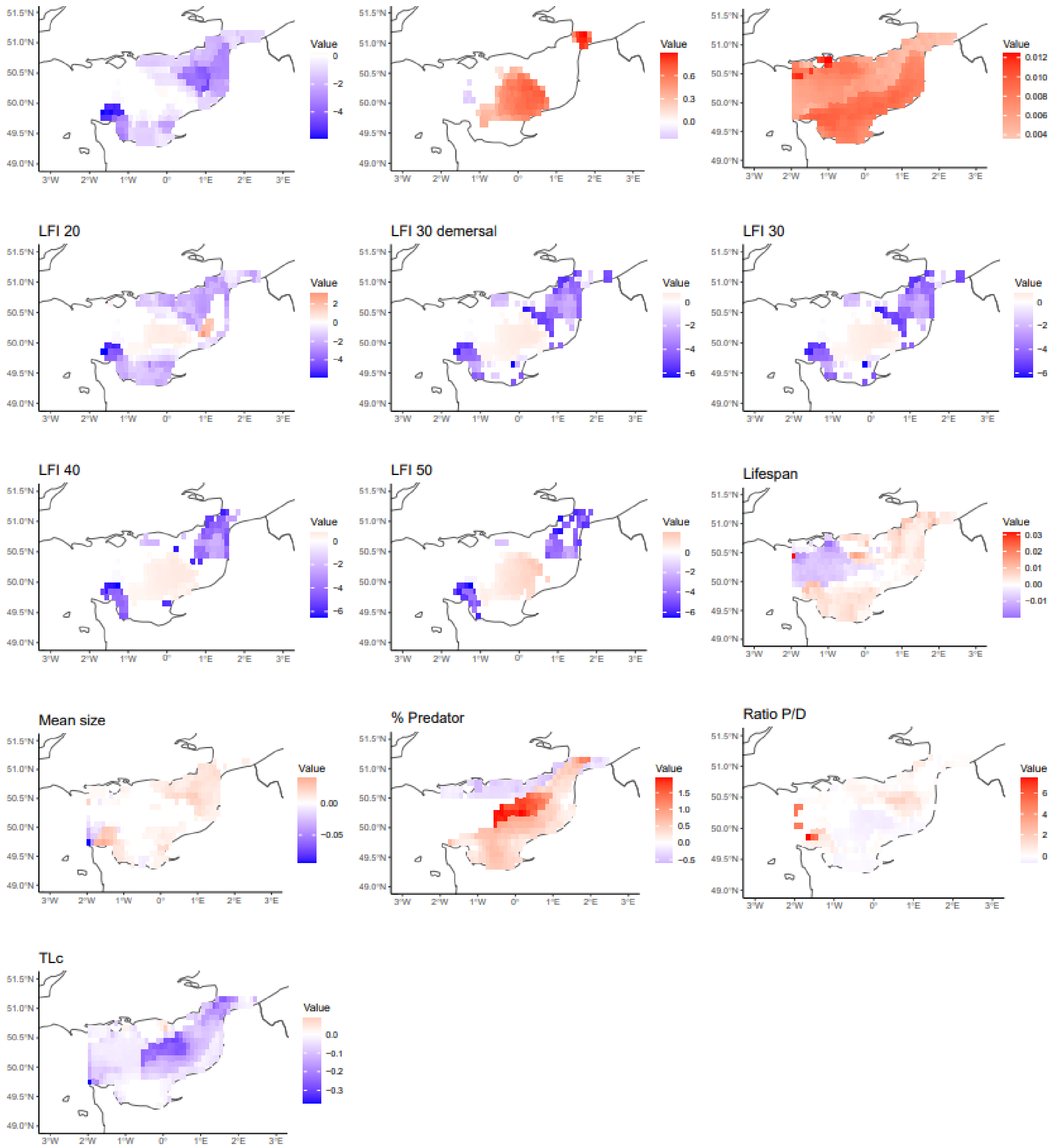
ANNEX X: Map of Atlantis indicator sensitivity results, row slope value of our fitting models of indicator value in function of F



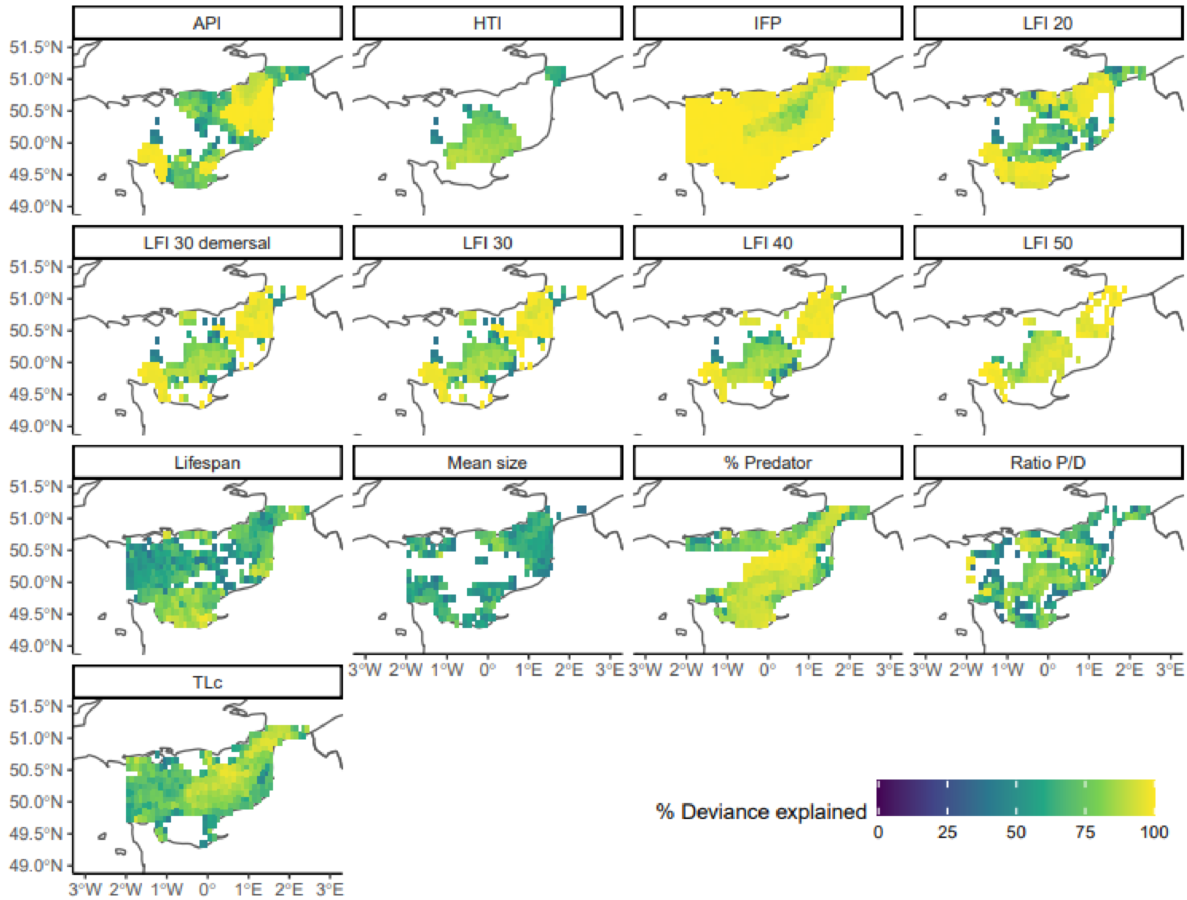
ANNEX XI: Map of Atlantis indicator sensitivity results, deviance explained of our fitting models of indicator value in function of F



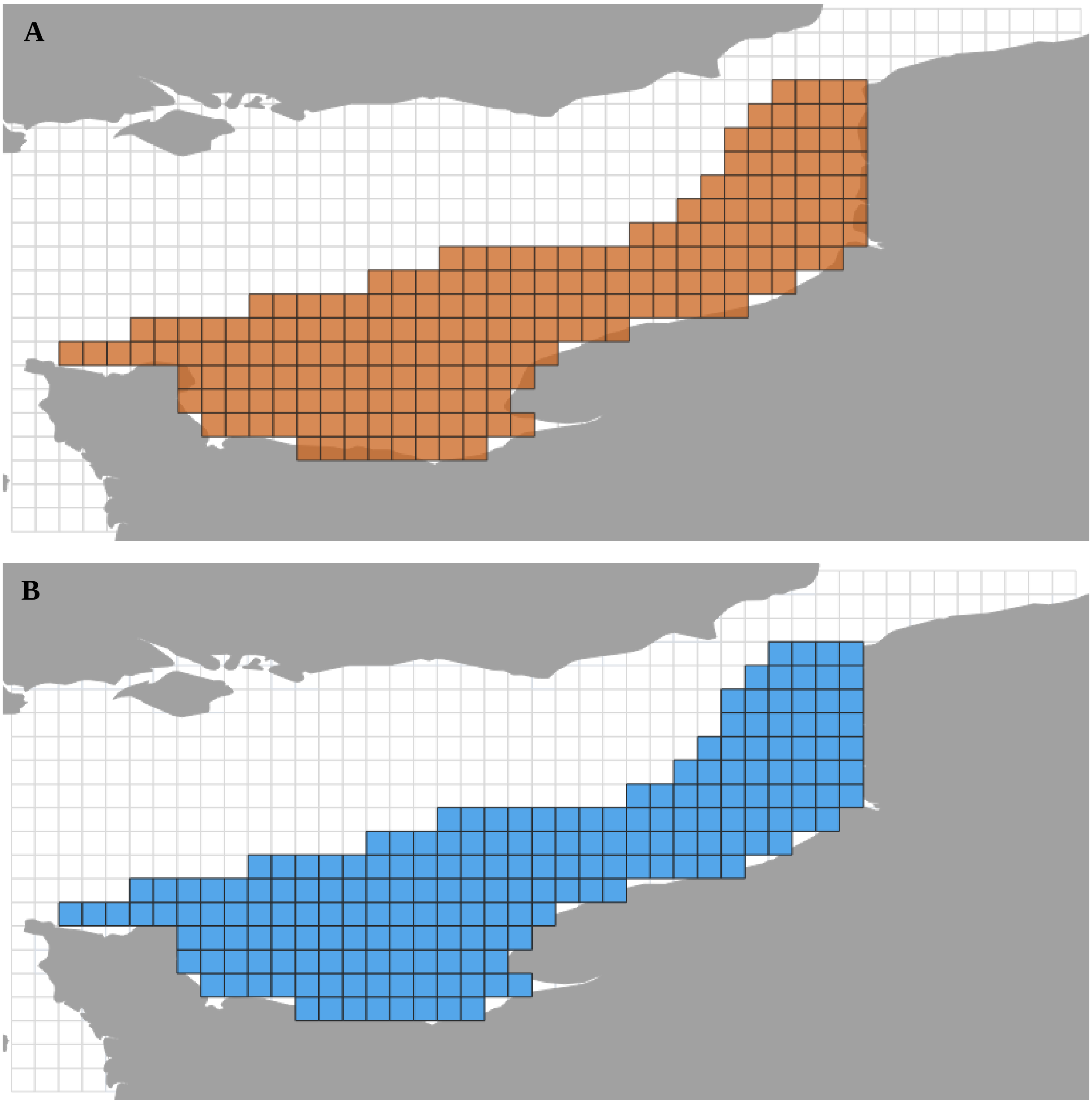
ANNEX XII: Map of Osmose indicator sensitivity results, row slope value of our fitting models of indicator value in function of F



ANNEX XIII: Map of Osmose indicator sensitivity results, deviance explained of our fitting models of indicator value in function of F



ANNEX XIV: A- Input of the Herring spatial distribution allowed in Osmose (orange: presence allowed, white: presence not allowed), B- Input of the Sardine spatial distribution allowed in Osmose (blue: presence allowed, white: presence not allowed)



L'INSTITUT agro Rennes Angers	Diplôme : Ingénieur Agronome Spécialité : Sciences Halieutiques et Aquacoles Spécialisation / option : Ressources et écosystèmes aquatiques Enseignant référent : Olivier Le Pape

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Titre français : Evaluation spatiale de la performance d'indicateurs écologiques à détecter les impacts de la pêche à travers une approche de modélisation écosystémique

Titre anglais: Spatial evaluation of ecological indicators performance to track fishing impacts using an ecosystem modelling approach

Résumé (1600 caractères maximum) :

La pêche est l'une des pressions qui impactent le plus les écosystèmes marins. Pour mieux comprendre ces impacts, il est important de considérer la complexité des interactions trophiques ainsi que les dynamiques spatiales et temporelles. L'objectif de notre étude est d'évaluer la performance d'indicateurs écologiques à évaluer les impacts de la pêche dans l'espace en termes de sensibilité et de réactivité. Une approche multi-modèles est appliquée à l'écosystème de Manche Orientale en utilisant deux modèles end-to-end, Osmose et Atlantis. Un ensemble d'indicateurs ont été calculés sous différents scénarios de mortalité par pêche en exploitant les sorties spatialisées des modèles afin de d'étudier leurs réponses à la pression de pêche. Nos résultats ont montré l'importance de la spatialisation des indicateurs pour évaluer les impacts de la pêche sur des échelles locales et mieux comprendre l'effet de la structure du modèle sur l'estimation des indicateurs. En effet, la spatialisation a révélé que certains indicateurs bien que sensibles et réactifs à la pression de pêche présentent une forte incertitude liée au choix modèle à l'exemple de l'Inverse Fishing Pressure. La comparaison de la performance des indicateurs a permis d'identifier le Large Fish Index pour les tailles de 40 cm et 50 cm et l'Apex Predator Index comme les indicateurs ayant le meilleur compromis entre sensibilité, réactivité et robustesse à la structure des modèles. Ces indicateurs semblent donc adéquats pour une gestion écosystémique à différentes échelles spatiales (locale et régionale) en Manche Orientale.

Abstract (1600 caractères maximum):

Fishing is one of the pressures that most impact marine ecosystems. To understand better these impacts, it is important to consider the complexity of trophic interactions as well as spatial and temporal dynamics. The objective of our study is to evaluate the performance of ecological indicators to assess the impacts of fishing in space in terms of sensitivity and reactivity. A multi-model approach is applied to the Eastern English Channel ecosystem using two end-to-end models, Osmose and Atlantis. A set of indicators were calculated under different fishing mortality scenarios using the spatialized outputs of the models to study their responses to fishing pressure. Our results showed the importance of indcators spatialization to assess the impacts of fishing on local scales and to better understand the effect of the model structure on the estimation of indicators. Indeed, the spatialization revealed that some indicators, although sensitive and reactive to fishing pressure, present a high uncertainty linked to the choice of model, such as the Inverse Fishing Pressure. The comparison of the performance of the indicators made it possible to identify the Large Fish Index for sizes 40 cm and 50 cm and the Apex Predator Index as the indicators with the best compromise between sensitivity, reactivity and robustness to the structure of the models. These indicators therefore appear to be suitable for ecosystem-based management at different spatial scales (local and regional) in the Eastern English Channel.

Mots-clés : SeaWise, indicateur spatial, Manche orientale, approche multi-modèle, gestion écosystémique

Key Words: SeaWise, spatial indicator, Eastern English Channel, multi-model approach, ecosystemic management