



VECTORS of Change in Oceans and Seas Marine Life, Impact on Economic Sectors

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Author(s):	Loïc Gasche (IFREMER), Sigrid Lehuta (IFREMER), Stéphanie Mahévas (IFREMER), Paul Marchal (IFREMER), Marie Savina-Rolland (IFREMER), Loïc Baulier (IFREMER)		
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VECTORS Overview

'VECTORS seeks to develop integrated, multidisciplinary research-based understanding that will contribute the information and knowledge required for addressing forthcoming requirements, policies and regulations across multiple sectors.'

Marine life makes a substantial contribution to the economy and society of Europe. In reflection of this VECTORS is a substantial integrated EU funded project of 38 partner institutes and a budget of €16.33 million. It aims to elucidate the drivers, pressures and vectors that cause change in marine life, the mechanisms by which they do so, the impacts that they have on ecosystem structures and functioning, and on the economics of associated marine sectors and society. VECTORS will particularly focus on causes and consequences of invasive alien species, outbreak forming species, and changes in fish distribution and productivity. New and existing knowledge and insight will be synthesized and integrated to project changes in marine life, ecosystems and economies under future scenarios for adaptation and mitigation in the light of new technologies, fishing strategies and policy needs. VECTORS will evaluate current forms and mechanisms of marine governance in relation to the vectors of change. Based on its findings, VECTORS will provide solutions and tools for relevant stakeholders and policymakers, to be available for use during the lifetime of the project.

The project will address a complex array of interests comprising areas of concern for marine life, biodiversity, sectoral interests, regional seas, and academic disciplines and especially the interests of stakeholders. VECTORS will ensure that the links and interactions between all these areas of interest are explored, explained, modeled and communicated effectively to the relevant stakeholders. The VECTORS consortium is extremely experienced and genuinely multidisciplinary. It includes a mixture of natural scientists with knowledge of socio-economic aspects, and social scientists (environmental economists, policy and governance analysts and environmental law specialists) with interests in natural system functioning. VECTORS is therefore fully equipped to deliver the integrated interdisciplinary research required to achieve its objectives with maximal impact in the arenas of science, policy, management and society.

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Executive Summary

- The main objective of Deliverable D5.1.4 was to evaluate the conservation performance of management strategies developed to mitigate the cumulative effects of fishing and aggregate extraction activities on key Eastern English Channel ecosystem components (flatfish species and benthic taxa).
- To that purpose, a complex, spatially explicit, model of two flatfish populations and three benthic groups in the Eastern English Channel was built using the ISIS-Fish modeling platform.
- Fish species modeled were sole (*Solea solea*) and plaice (*Pleuronectes platessa*), two species which are undergoing high fishing mortalities in this area. Benthos was modeled as three groups combining several taxa, based on their mobility.
- Five fleets were modeled by combining different fishing gears and boat sizes. Fish populations interact with the fishing activity in the modeled area. Aggregates extraction was also modeled and was assumed to impact only the benthos. Benthic groups were also impacted by fishing fleets that use towed gears.
- Simulations were run for twelve years from 2008-2020. 2020 corresponds to the year when Good Environmental Status should be reached in EU marine ecosystems according to the Marine Strategy Framework Directive.
- Conventional fisheries management measures were modelled as well as possible spatial management scenarios in the Eastern English Channel in order to determine whether these measures can help ecosystem conservation. The model robustness was then explored by means of advanced sensitivity analysis techniques. Classification trees were built from these simulations to perform an uncertainty analysis and assess the robustness of our diagnostic to input parameters used.
- Results obtained suggest that a Harvest Control Rule can help the recovery of strongly depleted biomasses for both fish species modeled. The uncertainty analysis indicates that the current ecosystem state should allow attainment of management goals defined for sole and plaice in this ecosystem in terms of spawning biomass. However, the level of natural variability in environmental parameters that can be tolerated is low.
- There was no evidence of positive effects of MPAs on fish populations, either at the scale of the eastern English Channel or at smaller scales gathering several bays on the French and English coasts.
- In contrast, effects of MPAs were very important for benthic taxa because they are not mobile at the level of a model cell. In particular, MPAs can efficiently maintain high benthos abundances in protected areas, but at the cost of severely depleted abundances in adjacent areas that are not protected.

Deliverable 5.1.4 is linked to:

- WP2.2, by integrating the spatially-explicit Eastern English Channel flatfish larvae trajectories and sole life cycle models (Deliverables 2.2.3 & 4.2.1; Rochette et al. 2012) in the ISIS-Fish modeling platform
- WP4.2, by integrating the analyses of the impact of aggregate extractions on Eastern English Channel benthic populations (summarized in Deliverable 4.2.1 and detailed in Gasche et al. (in prep.)) in the ISIS-Fish modeling platform
- WP6, by using (and expanding) the ISIS-Fish model structure and risk assessment methodology detailed in Deliverable D6.0.6 (Production of policy and governance synthesis as a tool for stakeholders and policy-makers) and in Gasche et al. (2013); the results obtained in D5.1.4 will also contribute to the general VECTORS synthesis dedicated to stakeholders (D6.7).

Introduction

The main challenge marine management has to face is to reach a Good Environmental Status (GES) by 2020, as defined by the Marine Strategy Framework Directive (MSFD 2008). The overall GES is defined and adapted to different components of the ecosystem (Vinchon et al. 2011; Biseau and Lorance 2012; Feral et al. 2012; Lefèbvre et al. 2012). This GES can prove hard to reach because ecosystem state is altered by human activities. Impacts of human activities are not always known and can be difficult to predict because they can occur at several, possibly interacting, levels of the ecosystem.

For instance, fishing decreases the abundance of fish species targeted in the Eastern English Channel (ICES 2008; ICES 2009; ICES 2010; ICES 2011; ICES 2012a). In addition, fishing activities using towed fishing gears adversely impact the substratum and benthic species living on it (de Groot 1984; Jones 1992; Dayton et al. 1995; Thrush et al. 1995; Auster 1998; Collie et al. 2000). These benthic species are important sources of food for several fish species living close to the bottom, in particular sole and plaice (Braber and de Groot 1973; Molinero and Flos 1991; Rijnsdorp and Vingerhoed 2001; Tous Rious 2009). Therefore, if one wants to accurately predict the effects of fishing at the level of the ecosystem, to implement ecosystem-based management (EBM), all these ecosystem components should be taken into account.

At present, the assessment and management of flatfish populations in the English Channel is conducted on a single-species basis, and does not take into account interactions with other fish species or human activities other than fishing (ICES 2012a; ICES 2012b). Conventional management of fishing activities is done by means of single-species Total Allowable Catches (TACs). ICES provides TAC advice increasingly based on Harvest Control Rules (HCR), which adjust the level of TAC depending on whether or not target (and/or limit) reference points (e.g., biomass, fishing mortality, CPUE trends) are breached. For most stocks assessed by ICES, target reference points are defined based on the Maximum Sustainable Yield (MSY), i.e., the maximum amount of fish that can be produced in a year by a targeted stock and caught without durably decreasing its biomass (ICES 2012b).

Human activities other than fishing may impact fish species and benthic taxa in the Eastern English Channel. One of these activities, which is still developing in the Eastern English Channel, is aggregates extraction. Several articles (de Groot 1979a; de Groot 1979b; Messieh et al. 1991; Kenny et al. 1996; Newell et al. 1998; Boyd and Rees 2003; Boyd et al. 2004; Newell et al. 2004; Boyd et al. 2005; Le Bot et al. 2010; Briffault et al. 2012) have shown that this activity has negative effects on the bottom structure and benthic species.

Aggregates extraction is managed by means of spatial management measures that determine the size of the exploited areas. Fishing activities can also be managed spatially, for instance with temporary or permanent closures restricting access to some areas. A general term to define such areas is Marine Protected Areas (MPAs). MPAs are sometimes seen as more efficient than conventional management measures to help ecosystem recovery. In reality, results are more contrasted and highly variable, successes generally corresponding to large MPAs and species with low mobility (Halpern 2003). In particular, benthic species seem to be strongly positively impacted when a no-catch or limited-access zone is created (Lester et al. 2009). These positive impacts on benthic species may have positive consequences on fish species because of trophic interactions between them. Several types of MPAs exist (Klein et al. 2010) and few of them aim at managing fishing or protecting fish resources. One advantage of MPAs is that they can serve to manage locally any human activity they overlap with, not just

fishing. In this way, MPAs usefully complement more conventional management measures building on, e.g., TACs.

A major challenge for decision-makers is to develop management strategies that are robust to the inherent ecosystem uncertainty. This is a topic which has been subject to scientific advances in recent years. For instance, Gasche et al. (2013) developed a method based on sensitivity analyses of a complex model and classification trees that allows testing of a wide range of parameter values for the HCR and identification of those reaching management goals robustly. This analysis was performed on a simple model of the Eastern English Channel representing impacts of the fishing activity on two flatfish populations (Marchal et al. 2011). It showed that under certain conditions management scenarios based on conventional management measures allow us to reach management goals of the studied species. The notion of robustness means that management goals are reached for any level of variability on environmental parameters.

The objective of this study is to determine whether MPAs would allow us to reach management goals more robustly if used in replacement of, or in complement to, conventional fisheries management. Management goals mostly concern fish species exploited by fishing fleets, but these could be expanded to account for other ecosystem compartments (e.g., benthic invertebrates) and/or human activities (e.g., aggregates extraction), consistent with the MSFD integrated and cross-sectorial guidelines. In this paper, we attempt to make a step towards a more holistic ecosystem assessment of the effects of human activities at sea. We consider benthic groups as well as flatfish populations in the Eastern English Channel so as to determine how and to what extent they are impacted by human activities. To that purpose, we build a model of this area using the ISIS-Fish modeling platform. This model is based on that created by Marchal et al. (2011) and further developed by Gasche et al. (2013) (also in Deliverable D6.0.6). The model is improved to address the issue of the spatial management of mixed fisheries and better describe fish populations and fishing activities (by integrating processes modeled under Deliverables D2.2.3 and D4.2.1, e.g., modeling of flatfish larvae trajectories and of the full sole life cycle). The fishing activity is modeled at the level of the ICES statistical rectangle. Sixteen fleets corresponding to five different gears and various boat sizes are modeled, using information from Lehuta et al. (in prep.). Another "pseudo-fleet" is added to represent the aggregates extraction activity and its impact on benthic populations, building on the outcomes of Gasche et al. (in prep.), summarized in Deliverable D4.2.1. Therefore, the consequences of spatial management measures can be tested and their local impacts on populations observed. Three management scenarios are tested. All use HCRs as base management and test different rules concerning access to MPAs, from full access for all fishing gears to no exploitation (by fishing activities or aggregates extraction) at all.

Core Activity

Material and methods

The ISIS-Fish modeling platform (Mahévas and Pelletier 2004; Pelletier and Mahévas 2005) was chosen to model the Eastern English Channel ecosystem. This platform represents marine ecosystems using three sub-models: (i) a fish populations' sub-model, (ii) a fishing activity sub-model and (iii) a management sub-model. Those three sub-models explicitly interact in time and space. Therefore they are particularly suitable to test spatial interactions between populations and human activities, or between human activities. The fish populations sub-model is flexible enough to allow representation of any population. Therefore it is also used here to model benthic populations that are defined with a lower level of detail than fish populations because of the smaller amount of information available on benthos. The model has already been largely described in (Mahévas 2009; Lehuta et al. 2010). A major feature of the ISIS-Fish model is that the fishing activity can be modeled in a very detailed way. In particular, the link between the fishing mortality applied to a given age group of a population in a given population zone and the effort applied to this zone by a given métier depends on three parameters. Here, a métier is defined as a gear being used in a particular ICES statistical rectangle of dimensions 30' (latitude) X 60' (longitude). Therefore, several métiers may occur in a given statistical rectangle if several gears are used in it, and several métiers corresponding to the same gear can impact one population zone if the population zone overlaps several statistical rectangles. The effort is first multiplied by an accessibility parameter that depends on the studied population only (and can vary between age groups) but not on the type of gear used. Then the effort is multiplied by a selectivity coefficient that depends on the gear used and can be specific to each age group of each population. Finally a target factor, defined for each age group of each population targeted by each métier, is applied. The fishing mortality obtained is summed over months and métiers to obtain a total fishing mortality. This total fishing mortality is used to compute a total catch rate using the Baranov equation and the natural death rate. This total catch rate then serves to compute a catch rate per métier, a catch rate per métier and per zone and a catch rate per métier per cell. The catch rate per métier at the cell level is then multiplied by the average abundance of fish per cell in the studied zone to compute a level of catch (see Mahévas (2009) and Lehuta et al. (2010) for more details).

We chose to focus on ICES Division 7D, corresponding to the Eastern English Channel. General characteristics of this area can be found in (Carpentier et al. 2009; Gasche et al. 2013). Several changes were made to the model compared to that used in (Gasche et al. 2013).

These changes can be summarized in four main points:

- The model is now spatially explicit and models populations as well as fishing activities and management at a smaller scale. The spatial resolution is much finer, each cell of the model now being 0.125° wide, in both latitude and longitude. This means that each ICES statistical rectangle corresponds to 32 of these cells. This allows us to model fish population zones and spatial management measures in a much more realistic way. Fish population zones are defined based on results from an updated version of the larval drift model from (Rochette et al. 2012) and on maps of preferential habitats from (Carpentier et al. 2009). Modelled populations (sole and plaice) were split into three sub-populations, each with one reproduction area and one or more nurseries. Apart from their location these sub-populations share the same biological parameters, and these have been modeled as one single population at the scale of the Eastern English Channel.

- The fishing activity corresponds to that used in Lehuta et al. (in prep.) and is defined at the scale of the ICES statistical rectangle. It explicitly models sixteen fleets that correspond to four main gear-based fleets that are further divided based on boat length and the harbor they belong to. For the main fleets, different boat sizes and harbors correspond to different technical efficiencies and different métiers, i.e., different fishing areas given a main fleet is mostly defined based on the gears used. The four main gear-based fleets were defined from an analysis of the French landings over the 2001-2010 period. These fleets correspond to: (i) bottom trawlers that use beam trawls (TBB FAO code) and otter trawls (OTB), (ii) mixed exclusive trawlers that only use otter trawls, (iii) dredgers that use dredges (DRB) for part of the year in addition to bottom trawls and beam trawls, and (iv) exclusive netters using gillnets (GNS) and trammel nets (GTR).
- Benthos dynamics are taken into account in the model. We added three benthic groups to the model that are modeled as populations in ISIS-Fish (similar to a fish population) and correspond to three important mobility groups identified in Gasche et al. (in prep.): mobile taxa, burrowing taxa and attached taxa. Mobility groups seemed to be the best option to globally describe benthic populations because productivity is linked to swimming capacities for several benthic species (Cartes et al. 2002). Their mean abundance is computed as the mean abundance on reference stations in Gasche et al. (in prep.) and their accessibility calibrated in the model so that the loss of benthos on aggregates extraction sites is coherent with that computed in Gasche et al. (in prep.).
- We added a pseudo-fleet corresponding to the suction dredges used for aggregates extraction in the Eastern English Channel. This fleet is particular in the sense that it only causes mortality for benthic populations. It was parameterized to be consistent with fishing fleets modeled. Each month, we allocate 85% of the extraction time to the two English extraction zones and 15% to the French zones. Volumes extracted are almost always higher in the South Coast region than in the East English Channel region, therefore 45% of the time spent was allocated to the South Coast and 40% to the East English Channel. French extraction times were split evenly between the three areas we defined. This allocation of extraction time is the same every month. Other fleets that impact benthos are trawlers and dredgers. The hypothesis was made that beam trawls and dredges impact benthos in similar ways, as suggested by de Groot (1984).

More details about changes made to the model can be found in the following two subsections, related to fish and benthic communities.

Modeling fish populations

Fish populations are structured by age. Nursery and Reproduction Zones are defined based on maps of updated data based on the work of Rochette et al. (2012). This allows us to define three sub-populations in the Eastern English Channel (Fig. 1), each corresponding to a particular geographic area: (i) the Somme sub-population corresponds to those whose nursery is the Bay of Somme, it also has one reproduction zone and one "population" zone that corresponds to that of adults outside the reproduction period and partly overlaps the reproduction zone; (ii) the Seine sub-population corresponds to three nurseries, Seine, Calvados and Veys, and also has one reproduction and one population area and; (iii) the UK sub-population corresponds to two nurseries (West of the UK coast and Bay of Rye) associated with one reproduction and one population area. A connectivity matrix defines the proportion of the recruitment of each sub-population that reaches each nursery of that sub-population.

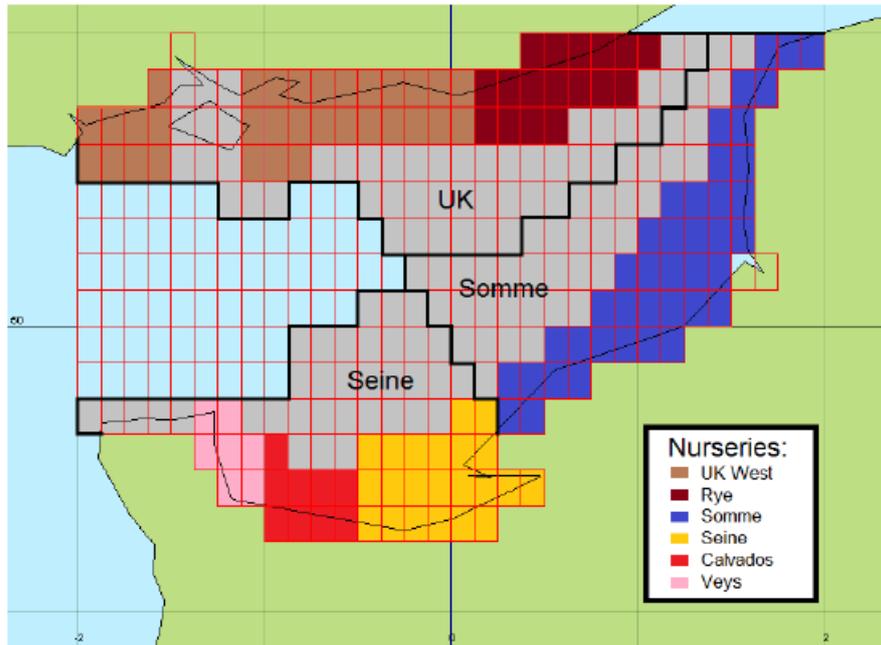


Figure 1. Map of nursery areas and population zones modeled. Surfaces in grey in each of the three zones correspond to either a population or a reproduction area, or both as these two types of areas can overlap.

Reproduction occurs from February to June with a peak in April for sole, and from December to March for plaice, with a peak in February (Carpentier et al. 2009). There is a two-month gap between reproduction and recruitment in the nurseries to take into account the duration of the larval drift that is approximately 60 days in Rochette et al. (2012). Fish migrate to their reproduction area in the month before the beginning of reproduction and leave the reproduction area after reproduction. Sole leave their nurseries to move to their reproduction areas at the age of 3 and plaice leave their nurseries between the age of 2 and 4, depending on the proportion of mature adults. So as to be as consistent as possible with working groups evaluations one age 0 group was added for each population so that larvae become age-1 fish in January. Group changes for both sole and plaice occur in January.

Modeling benthic taxa

We could not find stock-recruitment relationships for benthos, especially at the level at which it is modeled here. Therefore, we fixed the recruitment to a certain amount of the abundance at the beginning of the simulation (that corresponds to a pristine abundance), in order to approximately match known production levels of benthos. Productivity of benthic communities is generally high, and highly variable (Migne and Davoult 1997; Thatje and Mutschke 1999; Migne et al. 2004; Cusson and Bourget 2005; Spilmont et al. 2006; Migne et al. 2009), depending on factors such as taxonomic group, temperature, substratum. It appears that Production over Biomass (P/B) ratios for several common groups are between 1 and 2. Our model covers an area wider than that at which studies are usually made, and the groups we chose gather species with different production rates. As a result, we chose a moderate level of recruitment for the benthic groups. Indeed, we assume that recruitment is equal to the initial abundance of benthos, which amounts to assuming that P/B equals 1. Recruitment for benthos is not similar to that of fish: it occurs almost all year long. Therefore we spread the yearly recruitment over a period ranging from March to December, assuming that water temperatures are too low in January and February for reproduction to occur.

Population zones defined for benthos depend on human activities: (i) in areas where human activities occur that are likely to impact the benthos (i.e. aggregates extraction or fishing using towed gears) one zone corresponds to one model cell, so that local depletions can be modeled; (ii) a last zone corresponds to all model cells where none of these activities occur (for instance within the 3 nautical miles limit) and benthos should therefore be little impacted. We also make the hypothesis that benthos is immobile at the scale of a model cell. Therefore there is no migration of benthos and recruitment in a given zone is directly linked to reproduction in that zone. We gave no natural mortality to benthic groups, so we added a condition to recruitment so that it occurs only when benthos abundance is below the initial abundance. This condition prevents benthos abundance from increasing uncontrollably in zones where it is not impacted by human activities. This assumption is also coherent with the idea that the productivity of a stock that is not impacted tends to be lower than that of an impacted one as recruitment only needs to balance natural mortality. The accessibility of benthos was calibrated so that the abundance of benthic taxa decreases by 8% per month on a cell belonging to the “Granulats Havrais” area, which is the aggregates extraction area closest to the dredging site investigated by Gasche et al. (in prep.).

Modeling human activities and management

The fishing activity sub-model is largely based on that built by Lehuta et al. (in prep.). Relationships between a fishing fleet and a population in ISIS-Fish depend on several technical parameters (Pelletier and Mahévas 2005; Lehuta et al. 2010). Among these parameters are the selectivity, the target factor and a standardisation factor. We had to define these parameters for the additional aggregates extraction pseudo-fleet and to add information to the existing ones so that benthos is impacted in a coherent way by fishing. The target factor is defined at the métier (Biseau 1998) level and quantifies the strength with which the species is sought for by the métier (Mahévas and Pelletier 2004). This target factor depends on species commercial appeal and on the savoir-faire (know-how) of the fishers (Pelletier and Mahévas 2005). Benthic species studied in this analysis are deemed not to be targeted by fishing vessels, so the target factor of all benthic groups was set to 1, a neutral value in the fishing mortality multiplicative equation. Selectivity in ISIS-Fish is computed as the amount of fish caught in the fishing gear over the amount of fish exposed to the fishing gear. A selectivity of 0.5 was attributed to mobile and burrower groups for all gears and of 0.75 to attached groups that seem to be more damaged by fishing gears. We make the hypothesis that any benthic individual caught by the suction dredge is either destroyed or removed, thus selectivity for this gear was set to 1. A standardization factor is used to standardize fishing efforts between gears. This factor is adjusted for benthos so that the impact on benthos is not driven by a parameter initially defined for fish populations.

We chose to focus on five aggregate extraction zones in the Eastern English Channel: three on the French side and two on the English side (Fig. 2). As the aggregate extraction zones are small, those that are close to one another were grouped so as to model surfaces more realistically. Aggregates extraction is modeled by a pseudo-fleet with a strategy similar to the fishing fleets. The total effort allocated to that fleet was set to match the volume of aggregates extracted each year. More details about this fleet and how it impacts benthos can be found in Appendix 1.

Fishers' behavior is modeled with a gravity model. This model takes into account fishers' habits (i.e. the proportion of each métier in their strategy on the previous year) and the value per unit of effort (VPUE) of their catch to compute their strategy. New strategies are computed on a

yearly basis, based on the VPUE of the previous year. Several hypotheses can be tested where fishers are either opportunistic (choose their métiers based on past VPUE) or more risk averse (choose their métiers based on their habits), the default hypothesis being that both contribute equally to the fishers' decision-making process. Fishers' reactions to management measures were also modeled. If a species TAC is reached two scenarios are possible: (i) if the species is the main target species of a métier then the métier stops and, if possible, the effort is allocated to another métier; (ii) if the species is not the main target species then fishing continues and the species is discarded. The effort of a métier using a gear that is not allowed in a reserve that overlaps with the métier zone is reallocated to the part of the métier zone where this gear is allowed. If the whole métier zone is included in a fishing reserve where the corresponding gear is forbidden then this métier stops.

In our model fishing is first managed by means of TACs. Similarly to what was done in (Gasche et al. 2013) the TAC level is set to its actual level for years 2008 to 2010, then it is computed using a Harvest Control Rule. The HCR decreases the TAC by at most 15% per year for five years to reduce fishing mortality applied to each population from its 2010 level to the F_{MSY} level (target reference level). Other conditions apply so that the TAC can be further reduced if fishing mortality is above F_{pa} (precautionary fishing mortality) or biomass is below some trigger level. A minimum landing size also applies to fish populations.

The higher model spatial resolution allows us to test for spatial management measures. Not all marine protected areas (Fig. 2) could be added to the model because of the small size of some of them. In total 19 MPAs, corresponding to various kinds of management areas, were added to the model, some of them overlapping.

Three scenarios were tested :

- HCR and minimum landing sizes only
- previous rules + MPAs closed to all fishing gears
- previous rules + MPAs opened only to non-towed gears (i.e. closed to scallop and suction dredges and all types of trawls).

They correspond to two extreme scenarios that have to be tested in the VECTORS project, and to an intermediate scenario. The scenario with the HCR only can be seen as a "business as usual" scenario while that with total closures is a very restrictive one. We added the scenario with partial closures so as to determine whether the type of MPA implemented impacts fish populations.

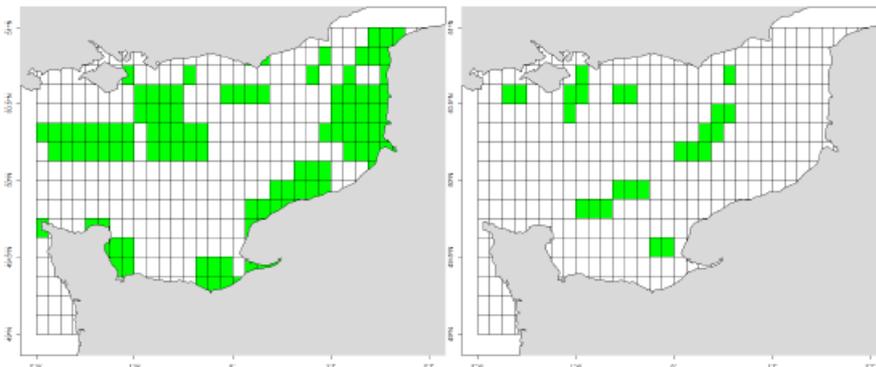


Figure 2. Maps of MPAs simulated in the model (left) and aggregates extraction areas (right). The size, shape and position of these areas are constrained by the model structure and resolution.

Simulations

Simulations are run for twelve years, from 2008 to 2020 when GES should be reached. Once scenarios were tested we performed a sensitivity analysis so as to determine how changes made to the model changed its behavior. This sensitivity analysis was performed with only “conventional” management measures so that results obtained can, to some extent, be compared to the sensitivity analysis made by Gasche et al. (2013) with similar management measures. In total 500 simulations were performed using random Latin hypercube Sampling (LHS) to explore the input parameters space. The low number of simulations performed is linked to the simulation time and resources needed due to the model complexity.

Six parameters were tested for both populations modeled. These parameters are those that had been identified as the most important in our previous analysis: recruitment (RE), natural death rate (NDR), mean weight-at-age (MW), accessibility (Q), age at maturity (MO) and target level of fishing mortality (F_{MSY}). A 25% range of values was explored around the reference value of each parameter. Output variables studied are the biomass in January of the last simulated year (BJan), spawning stock biomass for the same month (SSBJan), fishing mortality for the last simulated year (F), and catch of the last year (Y).

Once the input parameter space has been explored we identify simulations where management goals are reached in 2020 and those where they are not achieved. We use classification trees to determine which combinations of input parameters lead to successes and try to identify combinations of parameter values that always lead to successful management measures, in an attempt to identify scenarios providing the highest robustness to uncertainties on input parameters. 500 trees are built from subsets of the dataset obtained with LHS to assess tree variability. The most common tree type is identified and mean split values as well as variability of these splits are computed for this particular tree type. Due to the low number of parameters tested, no subtree was grown from the main tree obtained, so this approach is simpler than that applied in Gasche et al. (2013). We finally identify terminal leaves of the tree that contain only successes and determine which splits are needed to reach these values. This allows us to identify combinations of parameter values leading to these leaves and therefore the range of situations where management goals can be reached robustly.

Results

Fish species

Results obtained with the three management scenarios tested are rather similar in terms of values and identical in terms of trends (Fig. 3). For sole the studied time series can be divided into two parts. For the first three years spawning biomass decreases, then it starts to increase on the fourth year (2011) and keeps on increasing until the end of the simulation. At its lowest at the end of the third year the sole spawning biomass in the eastern English Channel is 5000t. It is interesting to note that the fourth year of the simulation corresponds to the first year when TAC levels are defined by a HCR. It is also the first year when the TAC level really constrains fishing, as can be observed with the slower decrease in SSB at the end of the year (that only corresponds to natural mortality). Spawning biomass of plaice at the beginning of the simulation is much lower than that of sole (3000t for plaice, 14000t for sole). This may explain why there is almost no decrease in the spawning biomass of plaice at the beginning of the time series (the minimum SSB is 1700t) and why SSB starts to increase one year earlier than for sole.

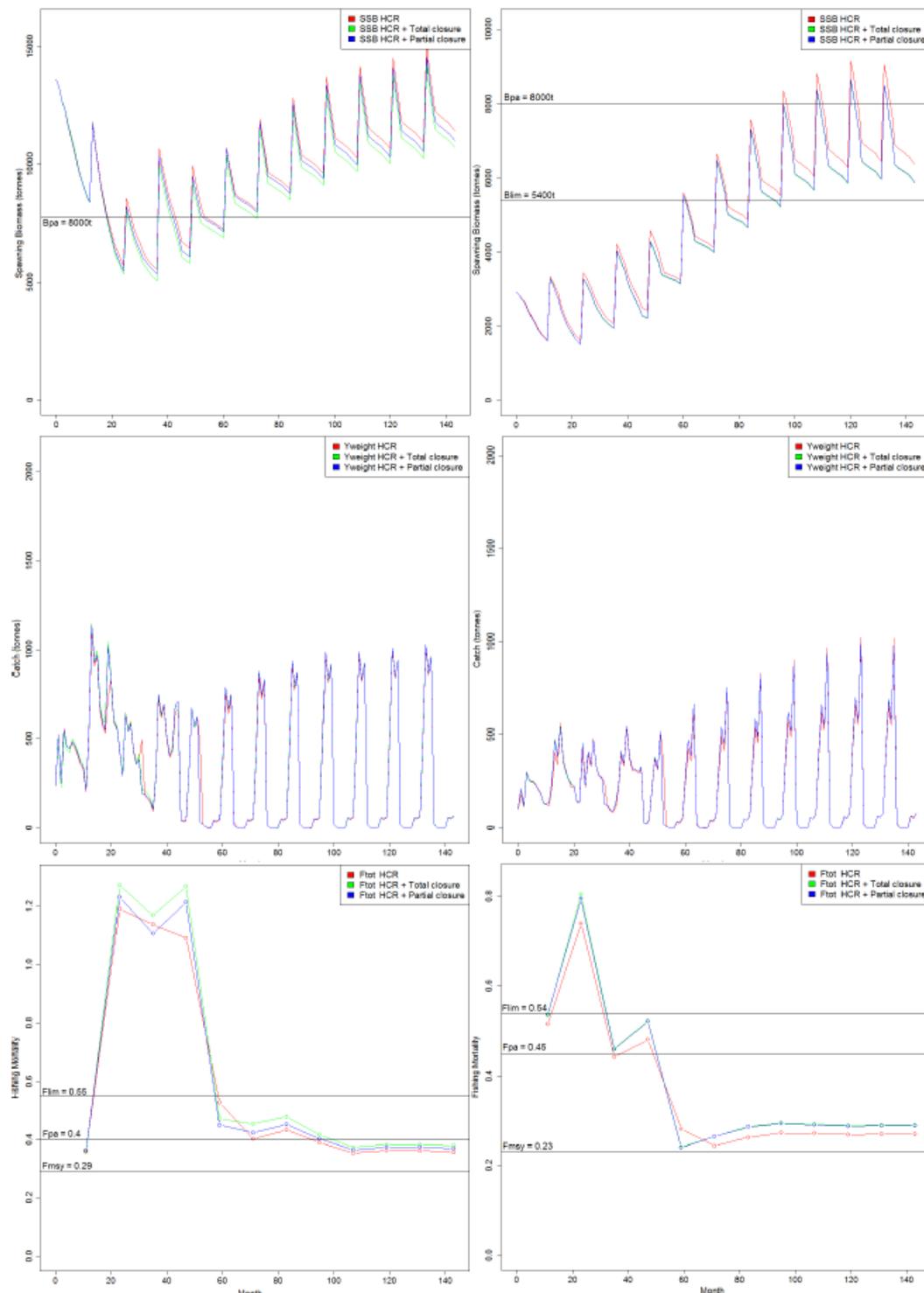


Figure 3. Total Spawning Stock Biomass (top), catch (middle) and fishing mortality (bottom) for sole (left) and plaice (right) in the Eastern English Channel.

Because of this increase due to management the spawning biomass at the end of the simulation is higher than 10,000t for sole and higher than 5,000t for plaice. This means that sole spawning biomass is well above the 8,000t threshold recommended by the ICES WGSSK working group but that plaice SSB is not. Given the fact that the plaice SSB at the end of the simulation is almost stable it seems unlikely that plaice may reach this threshold. It is interesting to note that for both species the best scenario is that with no area partially or totally

closed to fishing. Therefore, a conventional management with only a harvest control rule (HCR) determining a TAC level and a minimum landing size seems to be more efficient at maintaining high SSBs than a conventional management coupled with area closures. The scenario with a total closure is the least efficient one (stronger decreases in SSB and lower overall SSB).

These decreases in spawning biomass at the beginning of the time series can be linked with the amount of fish caught in the area. For both species the starting point of the increase in SSB corresponds to the beginning of a shift in the fishing period. This corresponds to a progressive increase in catches at the beginning of the year and a decrease in catches at the end of the year. This leads to a very strong monthly variability in the level of catch at the end of the time series. For instance the catch of sole peaks at 1000t per month for three months and is almost zero for the rest of the year. The pattern is similar for plaice (Fig. 5.3 right) with catch that only peaks at 1000t for one month and is 600 to 700t for the other two important months. The increase in the value of the peak in catch can be linked with the increase in biomass of each species. Globally, it can be observed that the level of catch decreases in the short term. This seems to be necessary from a management point of view given the drop in sole and plaice spawning biomass at the beginning of the time series. In the long run the level of catch tends to increase and catches peak at levels close to those the beginning of the simulation, but for fewer months. This increase in the monthly level of catch shows that management makes the fishery increasingly efficient because fish biomass increases. This is a factor worth taking into account as it may help reduce fishing costs.

The level of catch depends on biomass, but also on the level of fishing mortality that is applied to fish species. At the level of a whole fish population fishing mortality is computed yearly in the model using the Baranov equation. Similarly to what was done in Gasche et al. (2013), total fishing mortality is computed on only some age groups so as to match the way it is computed by stock assessment working groups. Fishing mortality applied to sole is higher than that applied to plaice, which agrees with the stronger decrease in spawning biomass observed for sole at the beginning of the time series. For both species, total fishing mortality peaks in the second year (Fig. 3). This peak can be explained by the use of a gravity model that modifies the fishing strategy of fishers depending on their habits and the value per unit of effort (VPUE) yielded by the various métiers on the previous year. The hypothesis is made that fishers try to maximize their income. After this peak values of F tend to stabilize. The fishing mortality stays almost constant once the optimal situation has been found by the gravity model. This stabilization is quicker for plaice where the peak at year 2 is much lower (0.8 for plaice versus 1.2 for sole). Fishing mortality is higher at the beginning of the simulation for scenarios with closures. This is a modeling artifact that comes from the way migration between zones is defined in our model. In fact fishing mortality is underestimated for scenarios without closures. Therefore the real fishing mortality in year 2 is equal to or slightly above that of the scenario with total closures. After the peak in fishing mortality F quickly converges for both species to a much lower level. For both species this level is reached at the fifth year of simulation after a strong decrease in F that corresponds to the second year of the implementation of the Harvest Control Rule. For sole the total fishing mortality converges to approximately 0.35 whereas it decreases to 0.25 for plaice. The target fishing mortality in the Eastern English Channel is 0.29 for sole and 0.23 for plaice. Management fails to maintain fishing mortality below these levels for both species during the simulated period. The scenario with no closure is that with the lowest fishing mortality during almost all the simulations for both species. This is in agreement with trends observed for spawning stock biomass where this scenario consistently has higher SSB.

This decrease in fishing mortality is linked to a decrease in effort by several métiers, a métier being defined here by the intersection between a fishing activity and an ICES statistical rectangle. Not all results concerning fishing efforts can be shown due to the number of métiers and statistical rectangles studied. One pattern that appears consistently over all rectangles studied is that the fishing effort applied using trawls becomes concentrated on a few months of the year and decreases. This phenomenon cannot be observed as easily on all statistical rectangles and for all métiers because some correspond to very low efforts. It can be easily observed for otter trawlers, beam-trawlers and trammel netters on rectangles 28E9, 29F0 and 29F1 (Fig. 4) that have high fishing efforts. Rectangle 28E9 is located in the middle of the Bay of Seine while rectangles 29F0 and 29F1 belong to the Somme area. These rectangles are known to be among the most intensely exploited in the Eastern English Channel (Carpentier et al. 2009; Lehuta et al. in prep.). At the beginning of the simulation fishing occurs almost continuously and effort peaks at high values. Then the effort decreases to lower values under the effect of management and is limited to only a fraction of the year. The maximum yearly peak in effort at the end of the simulation is three times lower than at the beginning of the simulation, and the fishing season only lasts for four months. These peaks in effort can be linked to the peaks in catch observed for both sole and plaice at the end of the simulated period. Catches increase during the second half of the simulated period whereas efforts remain rather stable after a strong initial decrease. This indicates that catches per unit of effort increase, a clear indicator that stocks are recovering under the effects of management.

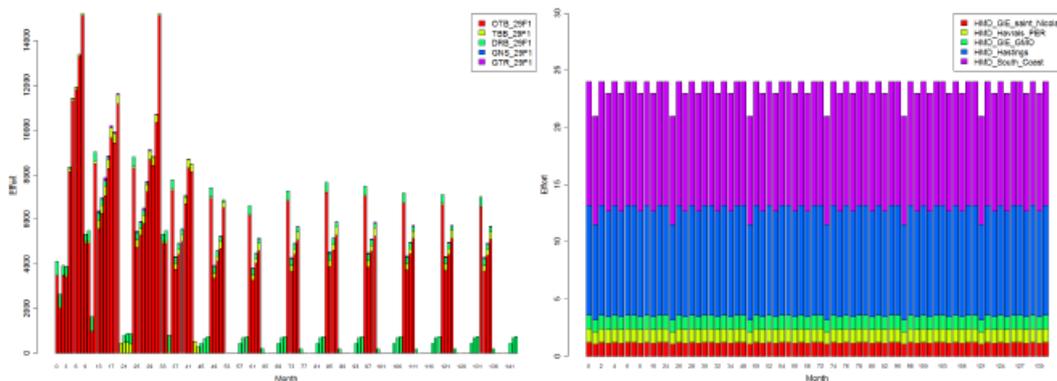


Figure 4. Fishing effort per gear in ICES rectangle 29F1 (left) and extraction effort on the various aggregates extraction areas (right). The fishing effort is at the scale of a whole ICES statistical rectangle whereas the extraction effort is at the scale of the surface of the extraction areas. Rectangle 29F1 is one of the most heavily fished rectangles in the Eastern English Channel.

One may wonder whether trends observed at the scale of the Eastern Channel mimic changes occurring at lower spatial scales or are only an average trend that may hide local depletions. It appears that spawning biomass behaves differently in the “Somme” area than in the “UK” and the “Seine” area (Fig. 5). Indeed, sole spawning biomass decreases strongly for the first two years of the simulation in the “Somme” area (by more than 3000t per year for sole, 900t for plaice) whereas the decrease is very moderate in the other two areas. Therefore the behavior of the Eastern English Channel population at the beginning of the simulation appears to be mostly driven by that of the “Somme” area, even if this decrease is dampened by the behavior of the two other areas. Paradoxically the “Somme” area is that with the highest spawning biomass at the beginning of the simulation and that with the lowest SSB at the end of the simulation because the strong decrease observed cannot be compensated by recruitment over the simulated period. Besides the yearly loss of biomass on this area seems to be high and the

recovery caused by management measures is very slow; it even seems unlikely that the 2008 initial level could be reached again. Results obtained for plaice are even more contrasted as plaice SSB strongly decreases for the first two years in the "Somme" area but increases in the two other areas. This decrease in spawning biomass in the "Somme" area does not show on Fig. 5 representing the SSB at the scale of the Eastern Channel ecosystem because it is compensated by the increases that occur in the two other areas. Contrary to what was observed for sole, the plaice spawning biomass in the "Somme" area increases back to its initial level when management measures are applied. Nonetheless this SSB is still the lowest of the three, the spawning biomass strongly increasing in scenarios where fishing is managed. The scenario with no closure has much higher spawning biomasses for both species in the "UK" area. However, this pattern is not constant over all areas and the SSB corresponding to this scenario on the contrary tends to be slightly lower in the "Seine" area.

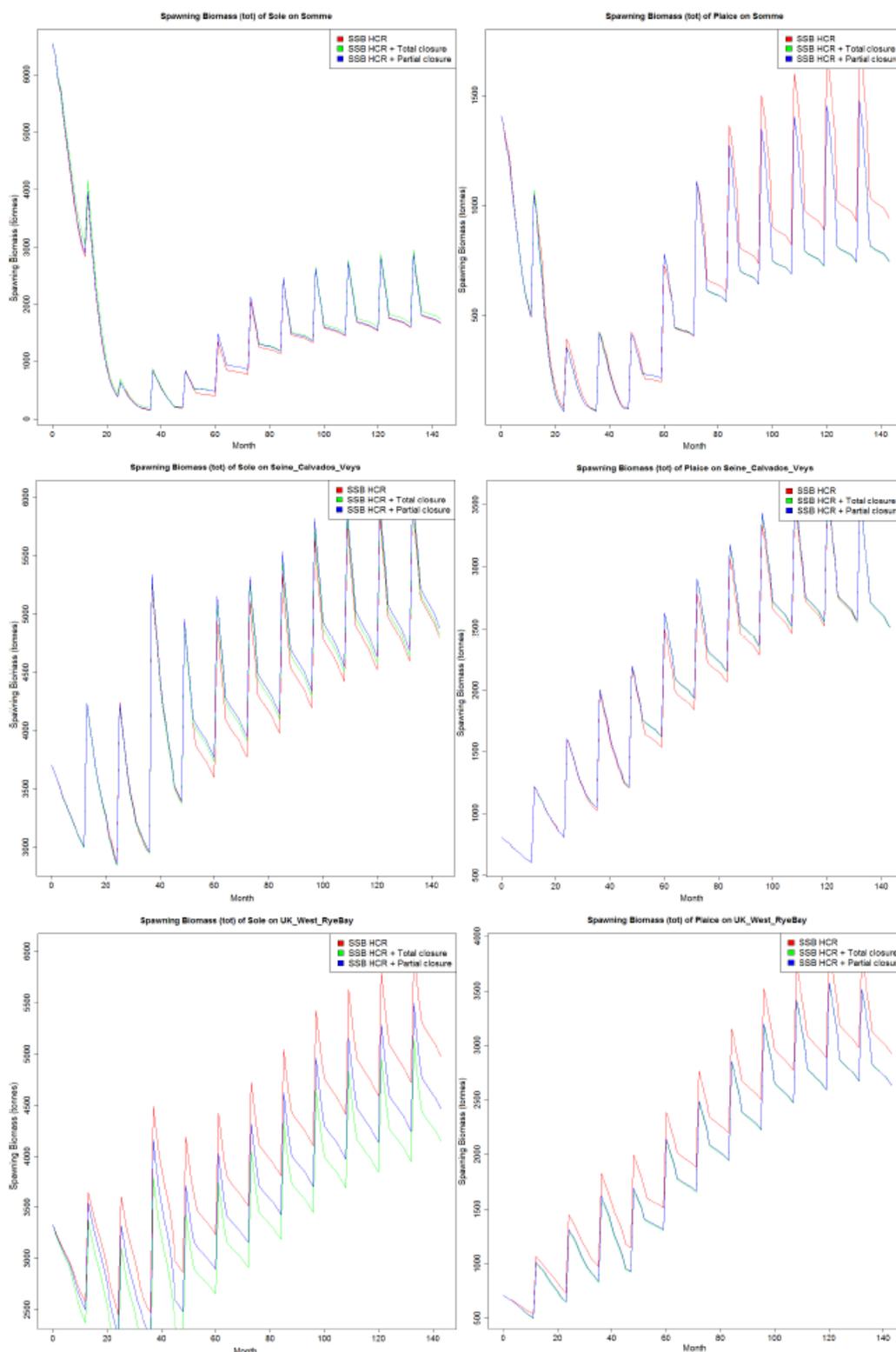


Figure 5. Spawning Stock Biomass for sole (left) and plaice (right). For each of the three modelled sub-areas : “Somme” (top), “Seine” (middle) and “UK” (bottom).

The evolution of the spawning biomass over the whole simulated period for the three zones can be summed-up by Fig. 6. These maps only show the spawning biomass ratio between the first and last time step of the simulation, therefore information concerning what happens in between is lost. Only the scenario corresponding to the HCR only is shown because the choice of a

management scenario does not affect the observed trends and only has little effect on the final values.

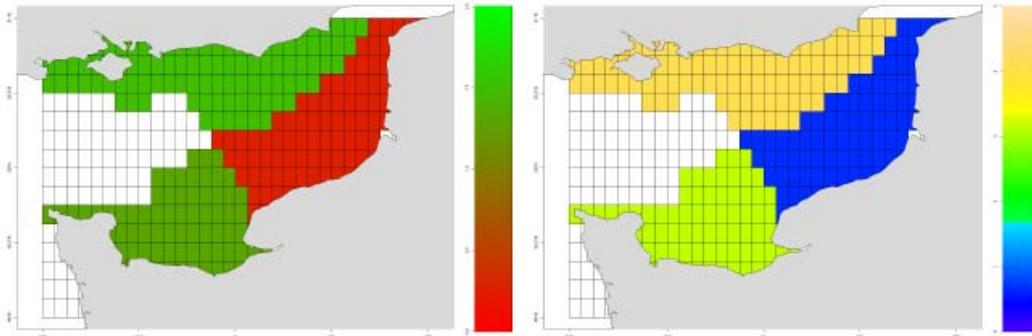


Figure 6. Maps summarizing the evolution of the spawning biomass of sole (left) and plaice (right) in the three studied sub-areas. Values given are a ratio between the SSB for the last month and that of the first month of the simulation.

Benthic species

In most model cells the effects of fishing and aggregates extraction appear to be low, recovery occurs quickly after benthos is impacted because recruitment occurs almost all year long. Nonetheless in some areas, such as the model cell belonging to ICES rectangle 29F1 (where high fishing efforts were observed previously) shown in Fig. 7, human activities maintain benthic populations in a permanently altered state. In the scenario where benthos is impacted its abundance decreases strongly at the beginning of the simulation, similarly to what could be observed for fish populations in the Somme area (to which the studied cell belongs). The abundance of benthic groups increases a lot after the decrease of the first two years and stabilizes at values on average one third lower than their initial abundance. This “stable” situation is associated with a high variability, the benthic population loosing and then recovering about half of its abundance yearly. These patterns can be explained by those of fish (and benthos) catches in the area where it was observed that the HCR causes peaks in fishing effort (and catch) for some months while the effort is spread across the year when not managed. These peaks in effort leave some time for benthic communities to recover so their abundance increases. It can be inferred that a more constant (and higher) effort would not allow this short-term recovery to occur and would cause much stronger decreases in benthos abundance. The effects of partial or total closures are similar because only towed gears impact benthos in the model. In this case the abundance of benthic taxa remains at its initial level. Patterns observed are similar for the three benthic groups modeled because they are defined in very similar ways. The only biological parameter that changes between them is their initial abundance. It is their sensitivity to fishing that really differentiates the three groups modeled. The issue is that the most sensitive group (Attached) is also that with the lowest abundance. Even if the abundance of all groups decreases strongly at the beginning of the simulation, that of the Attached group seems to decrease the most. In particular, the abundance of attached taxa almost reaches zero in the studied cell at the end of the second year. It then recovers in our model because there is no stock-recruitment relationship and recruitment is a fraction of the initial abundance. The impacts of aggregates extraction on benthos cannot be observed at the model scale. This is mostly due to the fact that aggregates extraction efforts are very low compared to fishing efforts (Fig. 4).

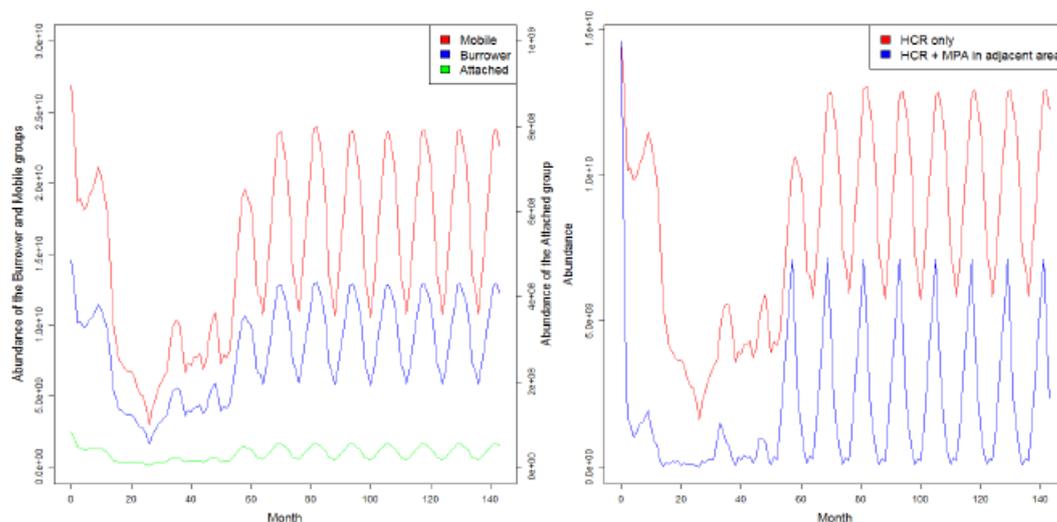


Figure 7. Abundances of the three modeled benthic taxa in a strongly fished area (left) and abundance of burrowers in a model cell adjacent to a possible MPA (right).

The consequences of spatial management measures on benthic taxa seem to be rather straightforward: if no towed gears are used then abundance stays stable at its initial level. However, closing an area to fishing has important consequences on benthos abundance in adjacent areas (Fig. 7). The figure shows a strong decrease in the abundance of the Burrower group in a model cell when a MPA is created in a nearby cell. In particular, the abundance of burrowers reaches almost 0 about a year after the beginning of the simulation and remains in this extremely altered state for three years. The abundance of burrowers increases only after the HCR begins and effort becomes very low for some of the years. This abundance increases on average and peaks at about half the initial abundance but is also very variable and decreases to almost 0 at the end of the fishing period. This demonstrates that closing some areas to fishing can have positive consequences at a local scale but very negative ones in areas close to the MPA. In particular, this can cause the shift from a situation where the abundance of benthos is only reduced to a situation where this abundance is almost zero. These very low local abundances may have more consequences at the ecosystem scale than reduced abundances on a wider area. The studied cell belongs to the 29F1 statistical rectangle that is studied in both graphs of Fig. 7. A large part of this rectangle can be made into a MPA, which leads to strong reports of effort to cells of that rectangle that are not closed to fishing.

Sensitivity analyses

Results obtained are similar to those of Gasche et al. (2013), where more parameters of a simpler model were tested (Fig. 8). Fish biomass is mostly impacted by recruitment and fish mean weight. Accessibility and natural death rate also stand out but have less impact. The target fishing mortality at MSY has little influence on the value of biomass observed at the end of the simulation. Patterns observed for spawning biomass are similar to those observed for biomass, with age at maturity also having a high importance and thus decreasing that of the other important parameters. Fishing mortality is strongly impacted by accessibility. Patterns for catches are very similar to those of biomass. They are also impacted by natural mortality. Accessibility does not have much influence on the level of catch. Thus, it seems that the level of catch is more driven by the level of biomass in the ecosystem than by the fishing effort applied to that biomass.

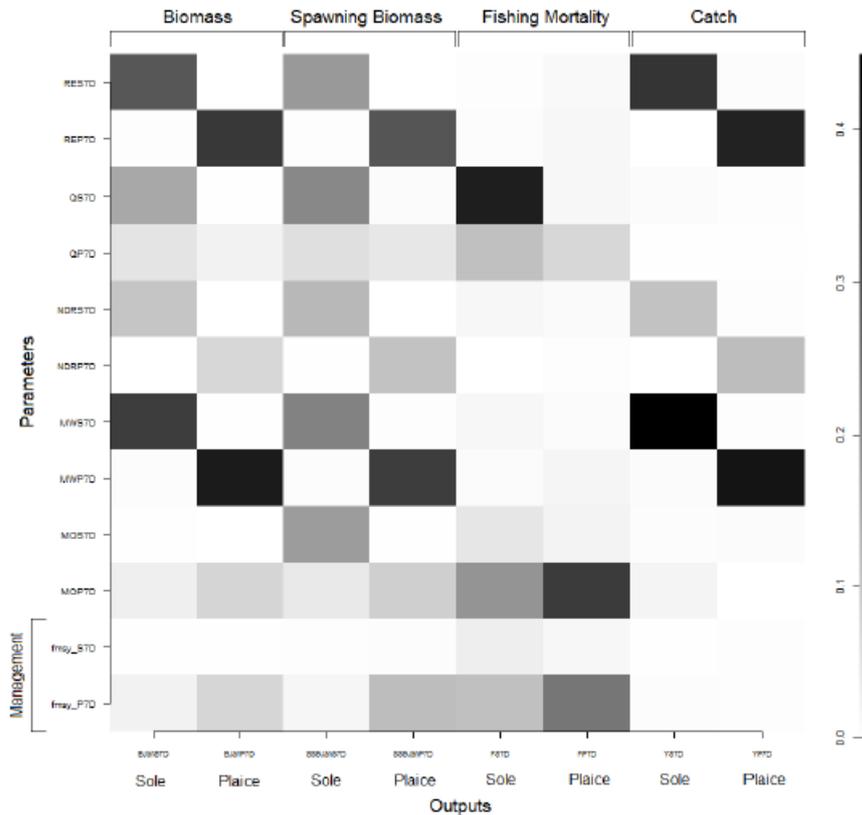


Figure 8. Sensitivity analysis results for scenario fishing + HCR. Tiles in black correspond to parameters most impacting the variance of the output variables. Results have to be studied vertically, columns sum to one.

Globally, F_{MSY} mostly impacts fishing mortality and biomass but not the level of catch. This is in agreement with previous observations and tends to show that management by TAC is able to decrease fishing mortality and help biomass recovery while maintaining, at least in the long run, constant levels of catch. Other parameters with some weight for F are those impacting biomass. Accessibility and maturity ogive of plaice in the area appear to also impact output variables of sole. In particular, they seem to impact sole biomass and fishing mortality. On the contrary these parameters do not impact the level of catch of either plaice or sole. The impact of plaice accessibility on sole may be caused by the use of the gravity model that can cause part of the effort to be redirected to sole if plaice accessibility is too low for instance (the contrary may also happen). Fishing mortality of plaice is also strongly impacted by the maturity ogive and less by accessibility.

Classification trees

We chose to focus on the spawning biomass of both studied populations. For sole and plaice the target spawning biomass in the Eastern English Channel is 8000t. For sole, one tree type clearly stands out and is obtained 172 times out of 500 (Fig. 9). Variables, mean split values and standard deviations corresponding to each node of this tree type, are given in Table 1. It can be observed that variability around the mean split value is very low: this tree structure is very stable. One leaf of that tree (node 17 on the right) seems to be particularly interesting. Indeed, this node is not totally robust, but almost, and has a high weight (it contains 244 simulations out of 500). To reach node 17 it is necessary to go through nodes 1, 11 and 15; in all cases the condition is to be above the split value corresponding to the node. The split values given correspond to values normalized between 0 and 1; 0.5 corresponding to

the reference model value, 0 to this value -25% and 1 to this value +25%. Therefore, being above 0.294 for recruitment means that recruitment has to be higher than the reference recruitment minus ten percent. Similarly, the age at maturity must not decrease by more than 16.9% below the reference age at maturity of sole and the mean weight-at-age must not decrease by more than 11.9% below the reference mean weight-at-age.

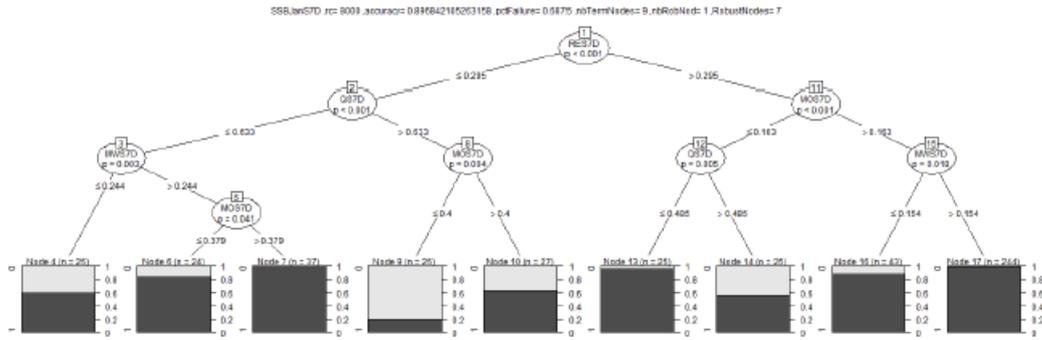


Figure 9. Classification tree corresponding to the most common tree structure, for sole. Split values may differ from average split values computed from the 172 trees of that type.

Globally, this analysis shows that the management goal in terms of spawning biomass can be reached robustly for sole in the Eastern English Channel. However, conditions on three parameters; recruitment, mean weight at age and recruitment, have to be fulfilled in order to reach this goal. The level of variability that can be tolerated on these parameters is not very high: from ten to seventeen percent depending on the parameter. The F_{MSY} parameter does not appear in this tree, even though it was one of the variables tested. It does not appear to be a strong driver of sole biomass in the Eastern English Channel when compared to biological parameters.

Inner Node	Parameter	Mean Split Value	Standard Deviation
1	RES7D	0.294	0.001
2	QS7D	0.651	0.013
3	MWS7D	0.243	0.002
5	MOS7D	0.396	0.019
8	MOS7D	0.400	0.013
11	MOS7D	0.163	0.000
12	QS7D	0.489	0.0196
15	MWS7D	0.262	0.042

Table 1. Inner nodes of the most common tree type, for sole. The reference model value is 0.5, therefore values higher than 0.5 are above the reference value and those lower than 0.5 are below the reference.

Tree instability is higher for plaice than for sole. For plaice 152 different tree types can be observed whereas only 84 different tree types are created for sole. Besides, the most common tree type for plaice occurs only 42 times, and stands out much less than the most common tree type of sole. Indeed, the second most common tree type for plaice appears 34 times and the third most common 25 times. However, the two most common tree types are similar, but for one node on their left part. The interesting robust terminal node being on the right part of the tree (Fig. 5.10), the branch leading to that leaf appears to be stable and is worth studying. Only two conditions are imposed to reach that node: (i) recruitment must not decrease below 7.1% under the reference recruitment and (ii) fish mean weight-at-age must not decrease below

3.2% under the reference. Globally, conditions leading to successful management are harder to reach for plaice than for sole and less variability can be tolerated in input parameters values.

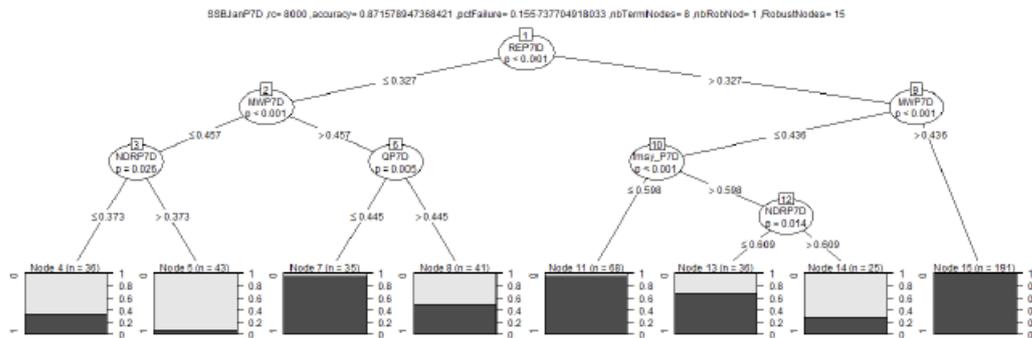


Figure 10. Classification tree corresponding to the most common tree structure, for plaice. Split values may differ from average split values computed from the 172 trees of that type.

An interesting point is that if mean weight is too low to reach the robust node (Hilborn 1997) then the next split is performed on the value of F_{MSY} for plaice. If F_{MSY} is below 0.598, that is no higher than 5% above the model reference F_{MSY} , a fairly robust terminal node can be reached. This leaf is also the second in terms of weight (number of simulations) after the previous leaf. Nodes corresponding to situations where F_{MSY} is above 0.242 (reference F_{MSY} for plaice is 0.23) contain fewer successes. This demonstrates that management, even if not the main driver of ecosystem spawning biomass, can increase chances to successfully reach management goals in situations where environmental conditions are not optimal.

Inner Node	Parameter	Mean Split Value	Standard Deviation
1	REP7D	0,327	0,000
2	MWP7D	0,457	0,001
3	NDRP7D	0,511	0,154
6	QP7D	0,449	0,015
9	MWP7D	0,436	0,000
10	$F_{MSY}P7D$	0,630	0,018
12	NDRP7D	0,579	0,0261

Table 2. Inner nodes of the most common tree type, for plaice. 0.5 corresponds to the reference model value.

Discussion

Results obtained in relation to model hypotheses and structure

Results obtained show that fishing can have considerable impacts on the two studied flatfish species. In particular when fishing is not managed or when TACs are too high, such as in 2008 and 2009, the biomass of a species in an area can be reduced drastically in a very short period of time.

Patterns observed at the scale of the entire Eastern English Channel for both species are the combination of patterns observed at lower spatial scales. These diagnostics, made at lower scales than that at which management measures are defined, seem to be important, as stability at a large scale can hide high variability at lower scales. In particular, this was observed for plaice with biomass being constant or slightly increasing at the scale of the Eastern English

Channel but strongly decreasing when only focusing on the “Somme” area. This shows the usefulness of explicitly modeling populations and activities at low enough spatial scales, so that spatial dynamics can be taken into account. These spatial aspects also allowed us to test spatial management measures, which are one possible tool to help ecosystem recovery and management.

The implementation of management measures strongly reduces the fishing effort in most studied areas. This decrease in fishing effort causes a slight decrease in short-term catch. Then biomass increases and catches increase accordingly. The decrease in the fishing effort is associated with a change in its seasonal pattern, with effort becoming increasingly concentrated at the beginning of the year. Indeed, when biomass is high at the end of the simulated period, these relatively low efforts are sufficient to reach the TAC within a few months. Nonetheless this pattern in catch does not seem realistic for an activity interacting with a market with year-long demands. If economic dynamics were taken into account more finely in this model the fishing pattern would certainly be adjusted so as to provide fish all year long and thus maintain high fish price and better match demand. Such a scenario could be more realistic for an industrial fishery where all fish is frozen or processed when caught where it would help reduce operating costs.

Plaice is mostly a by-catch species of sole in the modeled fishery. Therefore, even if biomass of sole in this ecosystem is higher than that of plaice, sole also seems to be more vulnerable because of its high attractiveness (higher price). This can be seen with the strong peak in fishing mortality applied to sole from year 2 onwards when a gravity model, partly based on value per unit of effort, is used to determine fishers' behavior. Indeed, fishing mortality for the first year is higher for plaice than for sole. Then the gravity model changes fishers' strategies and fishing mortality applied to sole and plaice increases, especially for sole. Paradoxically, classification trees show that management goals are easier to reach for sole than for plaice, in terms of spawning biomass.

Sensitive parameters are similar to those identified by Gasche et al. (2013) even though model complexity increased. The sensitivity analysis again highlights the fact that management has only marginal effects on ecosystem behaviour. However, classification trees built show that management can help maintain a slightly higher fish biomass when the ecosystem state is not good. Considering the current ecosystem state (the reference model parameterization) both sole and plaice reach management goals in terms of spawning stock biomass at the end of the simulated period. However, little variability around ecosystem parameters can be tolerated, in particular for plaice.

It appears that consequences of fishing on benthic communities can be important at a local scale. On the contrary, we did not manage to show clear effects of aggregates extraction. This is probably because these activities occur at smaller scales than fishing and are modeled accordingly. Therefore impacts of aggregates extraction appears to be almost negligible compared to those of fishing at the scale of the model cell. The low impact of aggregates extraction is coherent with the fact that this activity is a growing one that has not yet reached its full capacity, in particular on the French side of the Channel. Indeed, the effort of the aggregates extraction pseudo-fleet is much lower than true fishing efforts. Consequences from fishing are only really shown in a few of the most heavily fished areas. Fishing reduces the abundance of benthos until a point is reached where the average abundance of benthos remains constant, with strong monthly variations. Area closures can have detrimental effects on benthos in adjacent cells because the fishing effort shifts to cells that are not closed to fishing

when a MPA is implemented. Area closures focus the fishing effort on a limited number of zones that thus undergo high fishing pressures. When the abundance of a given taxa reaches low levels such as those observed it is possible that stock-recruitment relationships apply again and recruitment is reduced. This would greatly slow down the recovery process. This may also lead to increased vulnerability of the taxa and colonization by more opportunistic and/or less sensitive groups.

Management measures simulated here are ideal ones: it is deemed that we know the exact amount of catch and landings, that all fishers comply with management rules, that there is no time lag between decision and implementation of a rule, etc. In this ideal case it appears that management using TACs is efficient and helps stocks recover provided these TACs really limit the amount of fish caught. This result is however obtained for a scenario where all input parameters values are fixed. Sensitivity analyses show that TAC values would hardly be able to help ecosystem recovery in poor environmental conditions. On the other hand we fail to put to light assets of closures, either partial or total ones, even when applied to large areas. This is mostly caused by hypotheses made in the model. Indeed, the hypothesis is made that when an area is closed to fishing the effort of the impacted métiers remains the same but is applied to parts of the métier area outside the closed area. Only if the métier area is fully within the closed area does the métier stop. Since fish species are assumed fully mobile at the scale of their population area, it is enough that one cell of a métier area overlaps one cell of a population area for the whole population in this area to be fully impacted by the métier. Therefore in our model Marine Protected Areas tend to modify the spatial structure of the effort, which may lead to "local" depletions if population areas are small enough, but does not really modify the level of catch.

These effects would however be limited in our model in its current version because benthos accessibility to fishing and aggregates extraction might have been underestimated. This is due to the fact that contrary to fish species few data are available for benthos. We therefore tried to calibrate benthos accessibility using data obtained from the literature and experimental dredging sites and linking extraction intensity to decreases in mean abundance (the method used is explained in more detail in Appendix 1). It appears that the abundance of benthic species in our model remains close to its unaltered level in most model cells, only those with the highest fishing efforts being notably impacted. These scenarios seem to be too optimistic when compared to field observations (Collie et al. 2000), even when taking into account the fact that not all of the surface of an exploited zone is directly impacted. This calls for changes in the way relationships between benthos and human activities are modeled, and in particular an update of benthos accessibility in ISIS-Fish.

Modelling perspectives

Assets and limits of the method used have already been discussed by Gasche et al. (2013) and were clearly put to light in this modeling approach. Indeed, the model used is much more complex than that of Gasche et al. (2013) and now needs about three hours to run, when only a few minutes were needed previously. This means that the model exploration performed by means of a sensitivity analysis is much more costly now and requires either (i) to run the analysis longer; (ii) to increase our computing power (use of more cores); (iii) to test fewer parameters; (iv) to test fewer values for each parameter. We chose to make compromises for all of these aspects. We focused on a limited number of parameters and a narrower range of values (thus fewer values) for each parameter. We first chose to focus on parameters identified as important in the sensitivity analysis performed on the simpler model of the Eastern English

Channel. Only 500 simulations were run on six cores for the sensitivity analysis, which took twelve days. This analysis allowed us to roughly compare the behavior of the new model to that of the previous version. It appears that even if populations are described differently, in particular with several areas between which they migrate depending on the season, parameters that most impact output variables remain the same. Due to the limited number of simulations and to the fact that we focused on a few parameters known to be important we chose to not split the input parameter space with the method of Gasche et al. (2013) because results obtained would have been rather uninformative and highly uncertain. The next step of this analysis is to run sensitivity analyses of our model on a dedicated cluster so that the input parameter space can be explored in more details and a higher number of parameters tested. Only once this exploration has been performed, on biological parameters as well as management, will the splitting of the input parameter space be able to provide interesting results concerning which scenarios appear to be best to reach management goals.

Possible improvements of the model include which benthic taxa are modeled and how. Here we chose to group benthic taxa according to their swimming capacity because it seems to affect their productivity (Cartes et al. 2002) and may also be linked to their sensitivity to fishing gears (for instance determining their ability to escape the gear or to search for less impacted patches). Taxa could also be grouped according to their diet or to their body shape and size that may affect their interactions with the fishing gear. It seems necessary to group benthic taxa in such a model that represents several aspects of an ecosystem at large spatial and temporal scales. One interesting grouping method would be to use morphospecies (Brind'Amour et al. 2014) that seem to be good surrogates for species identification in community analyses.

MPAs should be modeled in a more refined and detailed way. Indeed, even MPAs that may not be implemented are tested so the surface of MPAs in our model is very high and is unlikely to correspond to the real surface of MPAs in the Eastern English Channel. Besides, our scenarios only correspond to partial or total closures to fishing. However, rules that are to be applied in most MPAs in the Eastern English Channel remain unknown. It is highly unlikely that fishing would be restrained so much in these areas. Indeed, in most cases MPAs do not aim at restricting fishing but at protecting some particular ecosystem features. In cases where MPAs serve to manage fishing, they appear more like zones where discussions between stakeholders are encouraged to better manage resources than zones where fishing is partly or totally forbidden. Interactions with other models could help test much more realistic situations. The Marxan model (Ball et al. 2009) allows optimization of the size and position of a management zone as well as the rules that apply in this zone. Attempts are currently being made (Reecht et al. in prep.) to couple this model to the ISIS-Fish model so that more realistic management scenarios can be tested and spatial management can adapt dynamically to the state of fish stocks. In our model we take into account the effects of closures on the fishing effort but do not model the numerous biological effects (Halpern 2003; Lester et al. 2009) that can be expected or observed in MPAs. For instance we model benthic species that are expected to be positively impacted by MPAs due to their low mobility, but not trophic links with fish species so the positive effects of MPAs on benthos cannot propagate to fish species.

Our model structure may not be the best to integrate trophic links, but other models have been developed for such tasks. We may be able to learn a lot by making our ISIS-Fish model interact with the OSMOSE (Shin and Cury 2000; Shin and Cury 2001) model of the area that focuses on trophic interactions between species in this ecosystem; either by means of forcing or coupling. In particular a coupling of this ISIS-Fish model with the OSMOSE model that takes into account trophic dynamics in the area would allow us to understand how trophic

relationships may modify reactions of the various modeled species to human activities. Therefore we would be able to determine how far in the ecosystem effects of fishing extend and whether they could be worsened by control loops coming from interactions between species at several levels of the ecosystem. It seems unlikely that a good ecosystem state could be reached without understanding these interactions. Results from such a group of models would certainly provide significant added value in terms of fisheries management as they would allow testing of realistic measures and could model their consequences at the ecosystem scale.

Conclusion

This study is, in our best knowledge, the first to model the cumulative effects of fishing activities and aggregates extraction on several ecosystem compartments. It is also the first to evaluate the conservation performance of management strategies combining spatial restrictions and harvest control rules. Although this approach did not include all human pressures and ecosystem interactions, it provides an advanced step towards the provision of a fully holistic and cross-sectorial modeling framework supporting ecosystem-based.

Our results suggest that a harvest control rule can help the recovery of strongly depleted fish biomasses. The uncertainty analysis indicates that the current ecosystem state should allow us to reach spawning biomass targets, although the natural variability that can be tolerated on environmental parameters is low. We found no evidence of positive effects of MPAs on fish populations, either at the scale of the eastern English Channel or at smaller scales gathering several bays on the French and English coasts. In contrast, effects of MPAs are very important on benthic taxa because they are not mobile at the level of a model cell. In particular, we show that MPAs can efficiently maintain high benthos abundances in protected areas, but at the cost of severely depleted abundances in adjacent areas that are not protected.

While the modeling framework developed in this study was applied to one specific area, the Eastern English Channel, it is generic enough to be parameterized and transposed to other case studies where fishing activities interact/compete with other sectors, the effects of which could be mitigated through spatial management. This applies in particular to other VECTORS case studies where fisheries, aggregates extraction, but possibly also windfarms and maritime traffic, operate on the same maritime domain (e.g., Baltic Sea, Dogger Bank, Southern North Sea).

Appendix 1: Linking benthic populations and aggregates extraction in ISIS-Fish

Relationships between a fishing fleet and a population in ISIS-Fish depend on several technical parameters. Among these parameters are the selectivity, the target factor and a standardisation factor. The target factor is defined at the métier (Biseau 1998) level and quantifies the strength with which the species is sought for by the métier (Mahévas et al. 2004). This target factor depends on species attractiveness and on the savoir-faire of the fishers (Pelletier and Mahévas 2005). Benthic species studied in this analysis are deemed not to be targeted by fishing vessels, and no information is available about it. Therefore, the target factor of all benthic groups was set to 1 for all métiers deemed likely to catch them. Choosing this value allows us to “bypass” the target factor term in the equation when computing fishing mortality, as this term multiplies other parameters used to compute fishing mortality.

Selectivity in ISIS-Fish is defined at the fishing gear level and can vary between populations and age groups. It is computed as the amount of fish caught in the fishing gear over the amount of fish exposed to the fishing gear. For one given gear, one selectivity value is

attributed to each population caught. As we have three populations of benthos, one selectivity value had to be defined for each of them. This is somewhat problematic as benthic groups gather many different taxa with varying levels of sensitivity to fishing gears. To try to get a broad idea of selectivity for various mobility groups we decided to define selectivity based on the meta-analysis by Collie et al. (2000). A selectivity of 0.5 was attributed to mobile and burrower groups for all gears and of 0.75 to attached groups, which seem to be more damaged by fishing gears. We make the hypothesis that any benthic individual caught by the suction dredge is either destroyed or removed, thus selectivity for this gear was set to 1.

Standardization is another parameter defined at the gear level. One gear is chosen as the standard gear and given the value 1. Standardizations for other gears are defined relative to that standard gear. The standardization factor given to each gear is an estimate of the amount of catch obtained in one hour with that gear compared to the amount of fish caught with the reference gear over the same period of time. Selectivity defined for benthic groups is divided by their standardization factor that was defined relative to fishing, so that catches of benthos are not driven by a parameter defined for fish. Gear selectivity is further modified to take into account the surface covered by the gear per unit of time. For instance, considering the speed of the boat and the width of the furrow left by a suction dredge it was estimated that it covers a surface ten times lower than that covered by a scallop dredge (the reference gear) in the same amount of time. Concerning bottom trawls, we make the hypothesis that only the panels on the sides of the trawl significantly damage benthos.

Strategy in ISIS-Fish defines for a given fleet how exploitation time is split between the various métiers accessible to this strategy. Concerning aggregates extraction, we make the hypothesis that fleet characteristics are similar on all extraction areas. This is unlikely to be totally true as two different types of dredgers, with different sizes, can be identified (Kemp 2008). This allows us to make the hypothesis that the volume extracted at the scale of the Eastern English Channel is an acceptable proxy of the time spent dredging. We chose to focus on five extraction zones in the Eastern Channel: three on the French side and two on the English side. Coordinates of French extraction sites were found in (JORF 2010a; JORF 2010b; JORF 2011; JORF 2012; Anon. 2013; JORF 2013a; JORF 2013b). The size of these sites being smaller than that of our model cells, these areas were not easy to include in our model. We therefore chose to create only three dredging areas on the French side, each corresponding to one or more dredging areas in reality. Two of these areas are located in the Bay of Seine, one near Le Havre and one offshore, and one area is in the Bay of Somme and corresponds to several extraction zones near Dieppe and offshore. Aggregates extraction on the English side of the Eastern Channel occurs in two areas: the East English Channel region and the South Coast region (TCEBMAP 2007; TCEBMAP 2012). These regions each contain several smaller areas that are grouped together, similarly to what was done for the French extraction zones. The amount of aggregates extracted by France represents only about 15% of the tonnage extracted each year in the Eastern English Channel (Kalaydjian 2012; Crown Estate 2008; Crown Estate 2009; Crown Estate 2010; Crown Estate 2011; Crown Estate 2012). Therefore, each month we allocate 85% of the extraction time to the two English extraction zones and 15% to the French zones. Volumes extracted are almost always higher in the South Coast region than in the East English Channel region, therefore 45% of the time spent was allocated to the South Coast and 40% to the East English Channel. French extraction times were split evenly between the three areas we defined. This allocation of extraction time is the same for every month.

Another parameter influencing a lot the level of catch of a species is the accessibility. This parameter is usually very poorly known and is therefore used to calibrate the model so as to

adjust fishing mortality in the model to levels estimated by working groups. This calibration is rather straightforward for fish species as fishing mortality is estimated in yearly stock assessments, but no such data is available for benthos, especially at the grouping level we used. In order to compute accessibility we used results from Gasche et al. (in prep.) linking an extraction intensity to a decrease in benthos abundance. Extraction intensity was defined as a mean monthly extraction time. It was computed that one hour of extraction per month reduced the abundance of burrowers by 2.98 individuals per square meter, that of mobile taxa by 4.97 and that of attached taxa by 0.1 individual per square meter. The initial abundance of burrowers is 117 individuals per square meter, that of mobile taxa of 216 and that of attached taxa of 0.65 individual per square meter. Therefore the observed loss of abundance corresponds to a 2.5% loss per hour per month for burrowers, a 2.3% for mobile taxa and a 10% loss for attached taxa. Abundance of attached taxa being very low we chose to focus on the quite similar results corresponding to the two other groups. Thus we chose to make the hypothesis that one hour of extraction per month decreases the abundance by 2.5%. We make the hypothesis that each additional hour decreases the remaining abundance and not the initial abundance so that abundance is not reduced to zero for intensities higher than 40 hours per month. In our model we make the hypothesis that extraction occurs only once per day, which means that the daily extraction time on a given area is about 2.5 hours. We also defined extraction as taking place on average 23 days per month so that monthly and yearly extraction times in the model are coherent with those needed to extract the volume of aggregates extracted yearly in the Eastern English Channel. So in total the extraction intensity is on average 57.5 hours per month. Applying the 2.5% coefficient this means that at this level of intensity the abundance of benthos on an extraction area should decrease by 77% per month. This value seems to be high, although it is coherent with measures made right in the furrow left by a dredge on a small experimental dredging site. Nevertheless such a value seems to make little sense at the scale of a whole extraction area. Indeed, only a fraction of a licensed area is really dredged each year. For instance in the East-English Channel dredging area only 8 km² over 80 km² were dredged in 2007, and 26 km² over 220 km² for the South Coast extraction area (TCEBMAP 2007). Therefore, it is only about 10% a licensed extraction area that are dredged each year. At the scale of a whole extraction area the decrease in abundance, even if locally strong, should therefore be about 8% per month with the way extraction activities are currently defined in the model. Therefore, accessibility was calibrated so that the abundance of benthic taxa decreases by 8% per month on a cell belonging to the "Granulats Havrais" area, which is the aggregates extraction area closest to the dredging site where measures studied in Gasche et al. (in prep.) were collected.

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