Élaboration d'un diagnostic robuste de mise en place d'aires marines protégées, dans un contexte d'incertitudes

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RÉSUMÉ

La Manche orientale est une zone stratégique supportant de nombreuses activités humaines, notamment la pêche, le transport de marchandises, l'extraction de granulats et le tourisme. Plusieurs projets de champs d'éoliennes sont également à l'étude. Il est capital de bien comprendre les interactions entre ces activités afin d'évaluer, prévenir et gérer les risques de conflits entre acteurs. De plus, leurs impacts potentiels sur les populations marines sont encore mal connus. Plusieurs Aires Marines Protégées (AMPs) sont en cours de création en Manche orientale afin de protéger des habitats ou populations clés. Il apparait nécessaire de prévoir dans quelle mesure ces AMPs peuvent contribuer à une exploitation durable des ressources marines, et quel sera leur effet sur l'organisation des secteurs d'activité qui en dépendent. La modélisation est une approche pertinente pour représenter ces activités et mesures de gestion, ainsi que leurs interactions. Un problème majeur rencontré lorsqu'on souhaite modéliser un écosystème marin est les fortes incertitudes auxquelles on est confronté : les populations marines et leur niveau exploitation sont très variables, et leur état est souvent mal connu. Ces incertitudes augmentent les risques de non atteinte des objectifs de gestion écosystémiques. Des mesures de gestion robustes aux incertitudes et minimisant ces risques doivent donc être trouvées. Des méthodes ont été développées dans le cadre de la Théorie de la Décision afin de permettre la prise de décision dans un contexte de forte incertitude. Une de ces méthodes est la théorie du fossé de l'information (info-gap). Cette méthode a commencé à être appliquée à des modèles d'écosystèmes, en particulier pour la protection d'espèces menacées. Néanmoins, ces applications pratiques ont jusqu'ici été limitées à des modèles simples ayant des solutions analytiques. Nous mettons ici en place une approche non probabiliste d'évaluation de la robustesse des mesures de contrôle basée sur la théorie de la décision, pour un modèle complexe non analytique. Cette méthode est appliquée à la gestion de la pêcherie de poissons plats de Manche Est. Les populations de sole et de plie de cette zone sont représentées au moyen de la plate-forme de modélisation ISIS-Fish. Une large gamme de valeurs des paramètres de gestion est testée, afin de déterminer lesquelles peuvent permettre d'atteindre le rendement maximum durable (RMD) avec la plus grande robustesse possible aux incertitudes. Ce modèle est ensuite enrichi pour évaluer les impacts de différentes activités humaines sur plusieurs compartiments de l'écosystème de la Manche orientale, dans un contexte de gestion spatialisée. Les effets possibles des AMPs sur les activités de pêche et les populations sont modélisés selon les informations trouvées dans la littérature. Les impacts de l'extraction de granulats marins sur les communautés benthiques sont déterminés grâce à des données obtenues sur une zone d'extraction expérimentale située en baie de Seine. Cette étude s'est déroulée de 2007 à 2012 et porte sur les guildes trophiques présentes sur la zone et consommées par la sole et la plie. Grâce à ces données, on détermine la dynamique des guildes trophiques avec extraction et sans extraction. Ces dynamiques sont incorporées dans le modèle ISIS-Fish de la zone. Finalement différents scénarios de gestion des pêches sont testés, avec divers niveaux de limitation de l'accès aux AMPs. Parmi ces scénarios, on tente d'identifier ceux permettant d'atteindre les objectifs de gestion, malgré nos incertitudes potentiellement fortes sur des paramètres environnementaux, le comportement des pêcheurs ou la biologie des espèces modélisées.

Mots-clés : Incertitudes, Modèle Complexe, Manche, Sole, Plie, Benthos, Mesures de Gestion Spatialisées

ABSTRACT

The Eastern Channel is a strategic area that supports many human activities, in particular fishing, freight, aggregates extraction and tourism. Many zones have also been identified that are suitable to host windfarms. A good understanding of interactions between these activities is necessary in order to estimate, forestall and manage risks of major spatial conflicts. Besides, their potential interactions with marine populations have not yet been well established. Several Marine Protected Areas (MPAs) have been created in the Channel in the past few years, to protect key habitats or populations, or are still being discussed. It is necessary to predict to what extent these MPAs can be an asset for a sustainable exploitation of marine resources, and what their effects will be on the organisation of sectors that depend on these resources. Thus, modelling becomes a relevant approach. One major issue when trying to model marine ecosystems is that they have high uncertainties: marine populations and fishing intensity are very variable, and their state is often poorly known. These uncertainties increase risks that ecosystem management goals may not be reached. Therefore, management measures that are robust to uncertainties and minimise these risks have to be found. Methods have been developed within Decision Theory that aim at allowing decision making under severe uncertainty. One of these methods is the information-gap decision theory. The info-gap method has started to permeate ecological modelling, with recent applications to conservation. However, these practical applications have so far been restricted to rather simple models with analytical solutions. Here we set up a nonprobabilistic management measures robustness assessment approach based on decision theory for a complex, non-analytical model. This approach is applied to the management of the Eastern Channel flatfish fishery. Using the ISIS-Fish modelling platform, we model populations of sole and plaice in this area. A wide range of values of management parameters is tested, so that we can determine which values allow reaching maximum sustainable yield (MSY) with the highest robustness to uncertainties. This model is then further enhanced in order to evaluate the impacts of human activities on various elements of the Eastern Channel ecosystem, in a context of developing spatial management. Effects of MPAs on fishing activities, fish and benthic populations are modelled according to information found in the literature. Impacts of sediment extraction on benthic communities can be determined by means of data obtained on experimental dredging sites located in the Bay of Seine. This study took place from 2007 to 2012, the analysis being performed at the level of the trophic guilds that appear in the diets of sole and plaice. Based on these data, we determine the dynamics of benthic populations at the trophic guild level on dredged sites and on reference sites. These dynamics are then included in the ISIS-Fish model of the area. Finally, we test various management scenarios corresponding to different levels of access limitations to Marine Protected Areas. Among all scenarios tested, we search for those that allow reaching management goals, despite potentially large uncertainties on several parameters linked to the environment, human behavior, or fish biology.

Key-words: Uncertainties, Complex Models, English Channel, Sole, Plaice, Benthos, Spatial Management Measures

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Chapitre 1

Introduction Générale

La nécessité d'une approche écosystémique des pêches est reconnue au niveau international depuis le début des années 70, avec une très forte augmentation du nombre de déclarations et de conventions depuis le début des années 90 (1). A une échelle plus restreinte ce processus de prise en compte et de gestion des écosystèmes se traduit par des textes tels que la Directive Cadre Stratégie pour le Milieu Marin (DCSMM, (2)) qui demande l'atteinte du bon état écologique des écosystèmes marins d'ici à 2020.

L'atteinte du bon état écologique d'un écosystème passe par l'atteinte de ce bon état pour les différents composants de cet écosystème. Afin de définir si le bon état est atteint ou non, la DCSMM propose plusieurs descripteurs de l'écosystème. Pour chacun de ces descripteurs plusieurs critères sont définis afin d'évaluer son état. Ces descripteurs peuvent aller de l'état du fond marin (3) au niveau d'eutrophisation de l'écosystème (4), en passant par l'état des populations de poissons, mollusques et crustacés (5) ou du réseau trophique dans son ensemble (6). Les critères appliqués à chacun de ces descripteurs peuvent être variés. En ce qui concerne le descripteur sur l'état des populations, qui revêt une grande importance pour la gestion des pêches, ces critères sont par exemple : (i) le niveau de pression de l'activité de pêche, (ii) la capacité de reproduction du stock et (iii) la structure en âge et en taille des populations.

Améliorer l'état d'un compartiment d'un écosystème passe par la gestion des activités humaines impactant ce compartiment. Ces activités sont le plus souvent gérées de manière individuelle, sans prendre en compte les interactions qui peuvent subvenir entre elles. La nécessité de tendre vers le bon état écologique devrait pousser la gestion à évoluer vers une approche plus écosystémique (EBM, Ecosystem Based Management) prenant en compte les impacts possibles à tous les niveaux et toutes les échelles de l'écosystème, ainsi que les interactions entre impacts. Ces approches de gestion demeurent très diverses et leur perception varie entre les différents acteurs impliqués dans la gestion (7). Améliorer cette gestion implique de définir ce qu'est un écosystème, puis de comprendre le fonctionnement des écosystèmes que l'on souhaite gérer.

Un écosystème peut généralement être défini comme un ensemble d'êtres vivants en interaction entre eux et avec leur environnement. Ce terme est apparu dès 1935 et résulte de l'idée que les organismes, même s'ils sont le principal centre d'intérêt d'une étude, ne peuvent pas être séparés de leur environnement avec lequel ils forment un système particulier (8). Cette définition relativement sibylline a peu évolué jusqu'à aujourd'hui. Par exemple, l'Evaluation des Ecosystèmes pour le Millénaire (9) définit un écosystème comme “un complexe dynamique composé de plantes, d’animaux, de micro-organismes, et de la nature morte environnante agissant en interaction en tant qu’unité fonctionnelle”. Ce texte place l’être humain au coeur des écosystèmes car il est en interaction dynamique avec ces derniers. Cette notion d’interaction est capitale car elle met en avant le fait qu’en modifiant les écosystèmes l’Homme altère également les services qu’il peut en tirer. Cette définition peut s’appliquer au milieu terrestre comme au milieu marin, à des échelles très variées.

On voit donc qu’un écosystème est un objet pouvant être extrêmement complexe. Comme le mettait en avant Tansley une étude se focalise en général sur une portion uniquement d’un écosystème. De ce fait, lorsqu’on les étudie les écosystèmes sont en général découpés en différents compartiments et seuls certains compartiments sont étudiés. Il en résulte qu’aucune approche de gestion écosystémique ne se fait réellement à l’échelle de l’écosystème ; elles se concentrent sur quelques aspects de l’écosystème jugés comme étant déterminants dans la dynamique du processus que l’on souhaite gérer.

Une approche fréquemment utilisée pour mieux comprendre les systèmes complexes est de les modéliser. De la même manière qu’il n’existe pas dans les faits une unique approche écosystémique, il existe plusieurs types de modèles d’écosystèmes. Ces modèles semblent être fréquemment classés selon des critères techniques, selon la manière dont ils représentent un système d’intérêt (modèles simples ou complexes, individus-centrés, bioénergétiques, etc.) (10; 11). On peut également tenter une brève classification de ces modèles selon leur but et les compartiments de l’écosystème sur lesquels ils se focalisent en particulier. Dans le cas des écosystèmes marins, on semble pouvoir distinguer quatre grands types de modèles. Les modèles physiques représentent le climat et les courants, mais aussi les échanges d’énergie, de nutriments et d’individus au sein de l’écosystème. Ils permettent en général de définir les éléments de base sur lesquels s’appuie le réseau trophique de l’écosystème. Ce réseau trophique est modélisé par des modèles trophiques, qui s’intéressent à la propagation de l’énergie depuis les producteurs primaires jusqu’aux prédateurs supérieurs, puis à son recyclage. Ces modèles représentent de différentes manières les relations prédateurs/proies au sein de l’écosystème ainsi que les différentes pertes et dépenses d’énergie qui ont lieu lors du passage d’un niveau trophique à l’autre. Ils peuvent parfois représenter la pêche comme un autre prédateur supérieur. Le troisième type de modèles est destiné à représenter la dynamique des activités humaines (en particulier la pêche) et les interactions entre ces activités et les populations de poissons. Ces modèles peuvent également permettre de représenter la gestion des pêches. Un quatrième groupe de modèles pourrait être ajouté à cette liste : les modèles permettant de représenter les conséquences économiques et sociales de la pêche pour les régions qui la pratiquent. Il existe également des modèles ayant vocation à représenter l’écosystème dans son intégralité, comme le modèle ATLANTIS (12) qui est en fait composé de plusieurs sous-modèles de parties de l’écosystème.

De tels modèles sont développés car ils répondent à des besoins. Ils visent en particulier à apporter la connaissance nécessaire à une transition depuis une gestion des pêches monospécifique au niveau du stock vers une gestion écosystémique.

Dix principes généraux peuvent être identifiés comme facilitant la transition vers une approche plus écosystémique des pêches (13). Ils impliquent en particulier la caractérisation spatiale de la structure des stocks de poisson et des habitats, le maintien de la structure en âge des populations et de la résilience des écosystèmes ou encore la prise en compte des dynamiques temporelles. En termes de modélisation ces principes peuvent être traduits en trois points où des avancées sont nécessaires afin de progresser vers une approche plus écosystémique : (i) la représentation de plusieurs espèces en interaction, (ii) la représentation spatialisée des populations et des activités humaines et (iii) la meilleure prise en compte des différentes sources d'incertitudes dans l'évaluation de l'état des écosystèmes et la quantification de la robustesse de ce dernier.

Tenter de représenter plus finement différents compartiments d'un écosystème dans un modèle revient à augmenter la complexité de ce modèle. L'augmentation de cette complexité s'accompagne souvent d'une augmentation de l'explicitation des incertitudes (14). Les incertitudes sur certaines variables pouvant être fortes, certaines sorties du modèle peuvent rapidement devenir incertaines. Une méthode courante et souvent recommandée (15) de prise en compte des incertitudes est l'approche Bayésienne, qui ne se concentre pas sur une évaluation ponctuelle de la valeur d'un paramètre mais tente de définir la loi de densité de probabilité de ce paramètre à partir de l'information disponible. De ce fait, cette approche permet d'obtenir une estimation des incertitudes autour des variables calculées par un modèle, à partir des incertitudes sur les paramètres du modèle. L'approche Bayésienne est également utilisée dans le cadre de la théorie de la décision (16), qui permet de confronter plusieurs actions possibles à plusieurs états de la Nature afin de tenter de trouver la meilleure solution. Cette approche semble particulièrement intéressante dans le cadre de la gestion des pêches car elle permet de confronter différents scénarios de gestion à nos incertitudes sur l'état de l'écosystème afin de déterminer comment atteindre les objectifs de gestion. Elle peut toutefois montrer certaines limites dans les cas où les sources d'incertitudes explicitement prises en compte dans un modèle sont nombreuses. En effet, les lois de probabilités jointes peuvent se révéler difficiles à traiter, en particulier quand elles impliquent un grand nombre de variables. L'incertitude devient en quelque sorte trop complexe pour être prise en compte correctement. Les cas où les sources incertitudes sont nombreuses peuvent de plus être des cas où les incertitudes sont fortes, ce qui justifie la construction de modèles complexes visant à les prendre en compte explicitement. Dans les cas où les incertitudes sont fortes, ou les conséquences d'un échec trop graves, la meilleure action peut ne pas être celle permettant de maximiser un gain, mais celle permettant de maximiser un gain minimum (17). Ceci peut être une incitation à privilégier une approche de type minimax (18; 19; 20) à une approche Bayésienne. Dans ce cas précis, on ne recherche donc plus la meilleure solution mais l'ensemble des actions permettant de garantir que le gain ne descendra pas sous une valeur critique. Cette approche nous semble particulièrement adaptée dans le cadre de la gestion des pêches où des limites à ne pas dépasser, que ce soit en terme mortalité par pêche ou de biomasse féconde des stocks, sont définies par les groupes d'évaluation. Il est d'autant plus important de ne pas atteindre ces limites que des changements brusques et potentiellement irréversibles du régime de fonctionnement des écosystèmes peuvent survenir si elles sont dépassées, comme ce fut le cas pour l'écosystème du Nord Benguela (21).

On souhaite ici contribuer à la meilleure prise en compte de certains des principes évoqués précédemment dans les représentations de l'écosystème de la Manche Orientale, pour aider au choix de réglementations permettant d'atteindre les objectifs de gestion malgré les incertitudes sur ce système. Cela passe tout d'abord par l'évaluation de la robustesse de la gestion actuelle aux incertitudes. On teste ensuite la pertinence de mesures de gestion spatialisées en représentant explicitement dans l'espace les populations de poissons et les activités humaines, ainsi qu'en incluant dans le diagnostic les communautés benthiques.

Pour cela, on développe un modèle de la Manche Est représentant les interactions entre populations de poissons plats, activités de pêche et gestion (22). Une des limites identifiées des modèles développés sur la Manche Est est que les populations benthiques n'y sont pas prises en compte. Or, il est connu que les activités humaines telles que la pêche ou l'extraction de granulats marins ont des effets négatifs sur les fonds marins et les populations benthiques qui y vivent (23; 24). Ceci peut revêtir une grande importance dans le cadre d'une approche écosystémique. En effet, le benthos se trouve à la base de nombreuses chaînes trophiques et est en particulier une source d'alimentation capitale pour les poissons plats (25; 26). Le benthos contribue également au développement de la structure tridimensionnelle des fonds marins, qui fournit des habitats aux populations de poissons (27). On comprend donc aisément que son interaction avec les activités humaines peut avoir des conséquences indirectes sur les populations de poissons et les activités de pêche.

De ce fait, on souhaite ajouter le benthos à un modèle de dynamique de pêche modélisant les interactions entre activités humaines et stocks de poissons plats. Parmi les activités humaines qui interagissent avec les populations benthiques on trouve la pêche, mais aussi l'extraction de granulats. La question des impacts de cette activité sur le benthos est assez récente et le nombre d'études s'y consacrant augmente rapidement. Néanmoins, les informations contenues dans ces articles peuvent être limitées ou ne pas être directement utilisables (pour des raisons d'échelle, de nature des paramètres étudiés) dans le modèle que nous souhaitons développer. La meilleure méthode pour réduire les incertitudes d'un modèle est de n'y inclure que les informations sur lesquelles on a de faibles incertitudes. Cela peut passer par la collecte de nouvelles informations. De ce fait, avant de tenter de modéliser les effets de l'extraction de granulats sur les communautés benthiques, nous tentons de mieux comprendre et quantifier ces effets au moyen de l'étude de données recueillies sur un site expérimental situé en Baie de Seine.

Le modèle utilisé se base sur la plateforme de modélisation ISIS-Fish (28; 29) qui permet de représenter explicitement les interactions spatiales et temporelles entre les stocks de poisson, l'activité de pêche et les mesures de gestion. La flexibilité du modèle nous permet également d'ajouter les populations de benthos et de nouveaux usages tels que l'extraction de granulats à notre modèle de l'écosystème Manche. L'utilisation de ce modèle nous permet également d'affiner la représentation actuelle des activités humaines et des populations de poissons dans cette zone. En particulier, les différentes zones de nourricerie des espèces représentées et leurs zones de reproduction sont représentées explicitement dans l'espace, ainsi que les échanges entre ces zones. Les communautés benthiques sont également modélisées à des échelles fines afin de mettre en évidence de possibles déplétions locales et les interactions avec des mesures de gestion spatialisées. L'activité de pêche est quant à elle représentée de manière aussi réaliste que possible en se basant sur une étude récente analysant les diverses sources de données disponibles à l'heure actuelle (30).

Dans un premier temps nous procédons donc à l'analyse de données recueillies à petite échelle sur un site expérimental afin de quantifier les effets de l'extraction de granulats sur les communautés benthiques. On tente en particulier d'établir un lien entre intensité d'extraction et baisse de l'abondance de benthos sur une zone donnée au moyen de modèles linéaires. Nous proposons ensuite une méthode de prise en compte des incertitudes dans les modèles complexes. Cette méthode est est appliquée au modèle simple de la Manche Orientale que l'on souhaite faire évoluer par la suite. Cette analyse d'incertitude se base sur les méthodes d'analyse de sensibilité pour explorer l'espace des paramètres du modèle puis sur la réalisation d'arbres de classification afin de trier les résultats obtenus. Enfin, on procède à la complexification du modèle afin de représenter de manière plus réaliste les populations de poissons, le benthos et les activités humaines. Différents scénarios de gestion sont testés et la méthode de prise en compte des incertitudes proposée précédemment est appliquée à ce modèle afin de déterminer si les objectifs de gestion peuvent être atteints de manière robuste aux incertitudes.

Afin de faciliter la compréhension de ce travail et de fournir davantage d'éléments de contexte deux chapitres introductifs font suite à cette introduction générale. Le premier de ces chapitres se focalise sur les activités humaines présentes en Manche Est, les impacts connus de ces activités sur le reste de l'écosystème et la manière dont celles-ci sont gérées. Le second chapitre tente de brosser un bref historique des méthodes de prise de décision dans un contexte incertain, puis de montrer comment ces méthodes communément appliquées à des modèles analytiques peuvent être utilisées sur des modèles complexes.

Chapitre 2

Pêcherie Manche et Aide à la décision

Sommaire

2.1 La Manche Est

La Manche Orientale s'étend de -2 à +2 degrés de longitude et de 49 à 51 degrés de latitude Nord, la portion située à l'ouest de la pointe du Cotentin étant exclue. La bathymétrie y est d'environ 70 m maximum, et diminue progressivement vers l'est et en allant du large vers les côtes. La profondeur est d'environ 40 m au centre du détroit du Pas-de-Calais. Cette zone se caractérise également par des marées importantes générant de forts courants (pouvant atteindre 4 noeuds, en particulier sur les côtes françaises). Les courants peuvent également être importants au niveau du détroit du pas de calais (31). Ces courants peuvent causer de fortes tensions de cisaillement sur les fonds et être à l'origine du déplacement de flux sédimentaires notables (32) pouvant expliquer en partie la nature des fonds en Manche Est (sédiments grossiers dans les zones de forts courants et plus fins dans les zones abritées).

Le type de sédiment impacte fortement la nature des communautés benthiques (33) : (i) sur les cailloutis des zones à forts courants on trouve la communauté à forte épibiose sessile ; (ii)sur les zones de graviers et sables grossiers la communauté à Amphioxus (Branchiostome) ; (iii) sur les sables moyens dunaires la communauté à Ophelia borealis (ver polychète) ; et (iv) sur les sables fins plus ou moins envasés des zones à faible courant la communauté à Abra alba.

La nature des sédiments a également une grande importance pour les juvéniles de poissons, et en particulier de poissons plats, qui ont en général besoin de sédiments meubles pour pouvoir s'enfouir (34). Du fait de ses nombreuses baies aux sédiments meubles la Manche Est possède un grand nombre frayères et de nourriceries vitales pour de nombreuses populations de poissons. Certaines de ces espèces ont une forte valeur commerciale et supportent des activités de pêche intenses à l'échelle Européenne (33).

2.1.1 Principales activités humaines

2.1.1.1 Activités actuelles

La pêche est une activité très structurante à l'échelle régionale en Manche Est. De 2000 à 2004 ce sont 460838 tonnes de poisson provenant de Manche Orientale qui ont été débarqués par des navires de plus de 10 mètres (35), soit plus de 92000 tonnes par an en moyenne. Ces débarquements peuvent être très fluctuants à de courtes échelles de temps (77206 tonnes déclarées en 2000, 113587 tonnes en 2003). Ils sont essentiellement le fait de cinq pays proches de la zone : la France, le Royaume-Uni, la Belgique, les Pays-Bas et l'Allemagne. La France réalise environ la moitié des débarquements en provenance de cette zone, suivie par les Pays-Bas qui en réalisent environ un quart. L'Allemagne et le Royaume-Uni représentent moins de dix pourcent des débarquements et la Belgique moins de cinq. L'Angleterre et la France semblent concentrer l'essentiel de leur effort sur leurs côtes respectives (35; 36). Si l'on prend en compte toutes les tailles de navires ce sont 641 navires français qui opéraient en Manche Orientale en 2005 contre 59 britanniques. Cela représente des captures totales de 90763 tonnes pour la France, d'une valeur de 218 millions d'euros.

A l'échelle de la Manche Est les espèces pélagiques constituent l'essentiel des tonnages débarqués, avec en tête le hareng, le chinchard, le maquereau et la sardine qui représentent à eux seuls près de 70% des débarquements en tonnage. Les espèces démersales et benthiques représentent des tonnages beaucoup plus faibles mais revêtent une grande importance du fait de leur prix élevé. A titre d'exemple, à l'échelle du territoire français en 2010 un kilo de sole se négociait sous halle-à-marée à 12.07 euros en moyenne, un kilo de morue à 3.59 euros, un kilo de plie à 1.42 euros, contre 1.04 euros pour le maquereau ou 0.6 euro pour la sardine (37). La coquille Saint-Jacques est une autre espèce de forte valeur, elle soutient une activité de pêche relativement spécialisée et très contrôlée. La coquille est essentiellement capturée à la drague tandis que les poissons plats sont plutôt capturés au trémail ou au chalut de fond à perche ou à panneaux.

Une autre activité extrêmement importante en Manche Orientale est le trafic de marchandises et de passagers. En effet cette zone de faible surface concentre à elle seule près de 20% du trafic maritime mondial, ce qui représente environ 800 navires empruntant quotidiennement le détroit du pas-de-Calais (38). Ce sont ainsi 560 millions de tonnes de marchandises qui ont transité par le rail du pas-de-Calais en 2010. Il faut de plus noter que 30% de ce trafic concerne des matières dangereuses pouvant avoir des conséquences notables sur les écosystèmes en cas d'accident. A ce trafic est-ouest de marchandises essentiellement destinées à alimenter les grands ports de Mer du Nord il faut ajouter un trafic transmanche intense permettant de relier l'Angleterre au continent. Ce sont ainsi près de 20000 rotations qui ont lieu annuellement entre Douvres et le continent et permettent d'acheminer environ 13 millions de passagers et 5 millions de véhicules.

Ce transport de passagers intense peut en partie être relié aux activités touristiques localement importantes dans certaines parties de la Manche Orientale. C'est par exemple le cas des côtes des départements français du pas-de-Calais et de la Somme où le tourisme génère un chiffre d'affaires de 5 milliards d'euros et 15000 emplois (38). Ces activités de tourisme font fortement fluctuer la population des villes côtières, essentiellement en période estivale. En effet, de nombreuses villes de cette côte relativement peu peuplée (quelques centaines de milliers d'habitants essentiellement concentrés autour de Calais et Boulogne-sur-Mer) ont une capacité d'accueil touristique plus de cinq fois supérieure à leur population résidente. Ce tourisme étant essentiellement tourné vers la mer, ce sont ponctuellement plusieurs milliers de navires de plaisance qui peuvent venir s'ajouter au trafic déjà intense de la zone à partir des nombreux ports de plaisance français de taille moyenne qui s'étalent le long de la côte, et de quelques ports anglais (38).

Ces activités touristiques peuvent localement être en compétition avec d'autres activités économiques. Cette compétition peut être une compétition directe pour l'espace ou une compétition plus indirecte due à la dégradation de la qualité de certains milieux ou à l'altération des paysages.

Une de ces activités est la culture de coquillages, en particulier la mytiliculture. Si les gisements naturels peuvent constituer un atout d'un point de vue touristique les concessions peuvent avoir l'effet inverse en modifiant le paysage et en occupant de vastes surfaces au niveau de l'estran. Ce sont par exemple plusieurs dizaines de kilomètres de concessions de moules de bouchot qui se trouvent à proximité directe de la zone touristique de la baie de Somme. Il est estimé que la longueur totale de bouchots sur les côtes françaises donnant sur la Manche était de plus de 600 kilomètres en 1995 (39). A la même époque la surface de parcs ostréicoles sur cette zone était approximativement de 400 hectares. La production conchylicole se concentre essentiellement en baie de Seine et au niveau de la pointe du Cotentin, elle est plus modeste en baie de Somme. La production d'huîtres de la zone Nord-Normandie était en 2010 de 16200 tonnes, pour 84100 tonnes à l'échelle du pays (40). La production de moules était elle de 12000 tonnes pour une production nationale de 73900 tonnes.

L'extraction de granulats marins semble avoir des interactions plus modérées avec les autres activités présentes dans la zone et rencontrer moins d'opposition. Sur les côtes françaises de la Manche Orientale cette activité semble être relativement récente, la plupart des permis d'exploration ou d'exploitation ayant été délivrés après 2005 (41). Entre 2001 et 2009 la production de granulats marins en Manche Est par la France a été relativement constante à environ 1.6 millions de tonnes par an. Six concessions ont à l'heure actuelle été attribuées côté français en Manche Orientale, toutes ne font pas l'objet d'une exploitation. Ces concessions peuvent être composées de plusieurs polygones proches et leur superficie est généralement de l'ordre de quelques kilomètres carrés, la plus grande d'entre elles faisant environ 61 kilomètres carrés. La tendance actuelle semble être d'éloigner ces concessions des côtes, en particulier au delà de la bande des trois milles nautiques qui est un espace reconnu comme devant être protégé au regard de son importance halieutique et benthique (42). Chaque concession est attribuée à une groupement pour une durée de trente ans et un volume maximal pouvant être extrait annuellement est associé à chacune d'elles (43; 44; 45; 46; 47; 48). Ce volume est à l'heure actuelle de 3 millions de mètres cubes (environ 5.4 millions de tonnes) pour deux des plus grandes concessions (Saint-Nicolas et Manche Orientale) (49). A titre de comparaison la production française totale de granulats marins était de 7 millions de tonnes de 2011 (50) tandis que la production de roches alluvionnaires et autres sables en milieu terrestre s'élevait à 142 millions de tonnes.

La production anglaise de granulats marins en Manche Est s'élevait à environ 8 millions de tonnes en 2012. Cette production est issue de deux zones distinctes : une zone "Est" située entre 0 et 1 degrés Est et une zone "Sud" entre 0.15 et 1.45 degrés Ouest (51). En 2007 la surface de la zone "Est" était de 80 kilomètres carrés environ et celle de la zone "Sud" de 210 kilomètres carrés environ. Chacune de ces zones est composée de nombreux polygones, tous n'étant pas exploités. Les autorisations d'exploitation sont allouées pour une durée maximale de 15 ans. Comme les concessions françaises les concessions anglaises demeurent exploitées en dessous de leur niveau maximal, 16.5 millions de tonnes peuvent en effet être extraites au total des deux zones qui sont exploitées à environ la moitié de leur capacité.

A ces activités on peut ajouter d'autres facteurs de pression sur le milieu marin plus indirects. La Manche Orientale est ainsi bordée de nombreuses centrales nucléaires et est parcourue de plusieurs câbles sous-marins importants (et traversée par le tunnel sous la Manche) pouvant limiter le développement des autres activités dans ces zones. Des fleuves drainant d'importants bassins versants ont également leur embouchure en Manche Est. C'est par exemple le cas de la Seine sur les côtes françaises qui draine le bassin parisien et peut être une importante source de pollution.

2.1.1.2 Activités futures

Une autre activité très décriée, en particulier en raison des fortes modifications du paysage qu'elle peut causer, est l'implantation de champs d'éoliennes. La Manche Orientale, avec ses fonds meubles et peu profonds et ses zones ventées présente de nombreux sites propices à l'installation de champs d'éoliennes (52). Trois zones ont ainsi été identifiées sur les côtes françaises de Manche Est comme susceptibles d'accueillir des champs d'éoliennes offshores : le site de Courseulles-sur-Mer pour un champ de 450MW et celui de Fécamp de 498MW lors du premier appel d'offre éolien en mer. Il a été proposé lors du second appel d'offre éolien en mer d'ajouter à ces deux sites un troisième au niveau du Tréport, d'une puissance équivalente à celle du site de Fécamp (53; 54; 55; 56). Chacun de ces sites couvre une surface supérieure à 100 kilomètres carrés. A titre de comparaison la puissance installée estimée au premier Avril 2011 en Haute Normandie était de 182MW et celle installée en Nord-Pas de Calais de 382MW (57). Ces champs d'éoliennes représenteraient donc une hausse extrêmement forte de la production d'énergie éolienne de ces régions. En ce qui concerne la côte anglaise de la Manche Orientale deux zones ont été identifiées comme propices à l'installation de champs d'éoliennes. Ces deux sites, également d'une puissance d'environ 500MW chacun se trouvent au large d'Hastings et à l'ouest de l'île de Wight (58). Ces deux sont relativement modestes en termes de taille et de puissance par rapport à ceux envisagés sur d'autres côtes du Royaume-Uni.

A l'heure actuelle aucune hydrolienne n'est installée en Manche Est, toutefois l'implantation de champs expérimentaux est à envisager d'ici 2016. En effet, un Appel à Manifestation d'Intérêt (AMI, (59)) est ouvert depuis Octobre 2013 afin "d ?accompagner la réalisation de fermes pilote hydroliennes en mer" au niveau du Raz Blanchard, à la frontière entre Manche Est et Manche Ouest.

2.1.2 Conséquences connues de ces activités sur l'écosystème

Toutes les activités présentées précédemment vont avoir des impacts forts ou modérés, directs ou indirects sur l'écosystème Manche. Tous ces impacts et leur interactions possibles ne sont probablement pas encore connus, néanmoins de nombreuses études ont été faites afin de tenter de mieux les connaître et de les quantifier. Nous nous focalisons ici uniquement sur les impacts les plus directs sur l'écosystème, c'est-à-dire ceux causés par la pêche, l'extraction de granulats et l'implantation de champs d'éoliennes.

Les conséquences les plus notables de la pêche sont sur les populations de poissons ciblées par cette dernière. Le premier effet observé de la pêche est une chute rapide et importante de l'abondance des stocks ciblés. De nombreux exemples existent où la pêche à réduit l'abondance des stocks à une infime partie de leur abondance avant exploitation (60; 61; 62). Dans ces cas il est fréquent que ces faibles abondances ne permettent pas la récupération du stock même après arrêt de la pêche, du fait par exemple de la concurrence avec d'autres espèces de poissons (63). Il arrive même que la pêche cause des basculements irréversibles dans la composition des écosystèmes en favorisant certaines espèces au détriment d'autres. Cela peut venir du prélèvement d'un prédateur laissant ses proies se développer de manière inhabituelle ou au contraire du prélèvement d'une proie privant un prédateur peu opportuniste d'un partie importante de ses sources d'énergie (64). Lorsque plusieurs espèces remplissent le même rôle écologique et que seules certaines sont exploitées commercialement cela libère des niches écologiques pour les espèces non exploitées. En plus de favoriser les espèces non exploitées, la pêche exerce également une pression de sélection sur les espèce qu'elle cible (65). En effet les engins de pêche sont faits de telle sorte qu'ils sélectionnent essentiellement les individus de plus grande taille (meilleure valeur marchande, désir de laisser les juvéniles atteindre leur âge à maturité). Ceci tend donc à sélectionner les individus à croissance lente et à maturité précoce qui arrivent à se reproduire avant d'être capturés (66; 67). Dans certains cas cela se traduit par de très fortes baisses de l'âge à maturité d'espèces fortement exploitées. Si la pêche capture essentiellement des individus de grande taille, elle ciblerait également en priorité les espèces de prédateurs supérieurs situées en haut de la pyramide trophique. De ce fait elle aurait tendance à éroder cette pyramide par le haut (fishing down marine foodwebs (68)) et à faire baisser le niveau trophique moyen des écosystèmes exploités.

La pêche peut également avoir de forts effets sur les espèces qu'elle ne cible pas (espèces accessoires) mais qui sont capturées avec les espèces ciblées du fait de leurs similarités (taille, morphologie, position dans la colonne d'eau, comportement, cycle de vie). Ces captures accessoires peuvent représenter une proportion élevée du total des captures (source) et sont généralement rejetées à l'eau mortes où elles contribuent à l'alimentation des oiseaux et de certaines espèces de poissons ou de benthos nécrophage. Ceci peut tendre à favoriser des voies trophiques initialement peu importantes dans l'écosystème. La nature et la quantité des captures accessoires dépendent du type d'engin de pêche utilisé mais aussi de la manière dont ce dernier est manié. Les engins dits trainants (dragues, chaluts) sont généralement présentés comme moins sélectifs que les engins dits dormants (filets, casiers), même si le niveau de sélectivité dépend également de la taille des mailles (69) et de la présence de dispositifs d'échappement (pour les juvéniles ou les espèces accessoires) sur l'engin (70). Les engins de pêche peuvent également avoir des actions physiques marquées sur les fonds marins s'ils sont en contact avec ces derniers. Là encore il semble que les engins trainants aient des impacts plus forts. Les facteurs déterminant l'impact des engins sur l'environnement sont leur poids, la vitesse à laquelle ils sont tirés, la nature des sédiments et la force des courants de marée (71). On peut ajouter que la largeur des engins est sujette peut être variable et impacte directement la surface couverte par ces derniers. Ceci se traduit tout d'abord par une réduction de la complexité des habitats et un lissage des reliefs du fond. Ainsi, le modèle d'Auster (27) prédit une réduction linéaire de la complexité des habitats avec la hausse de l'effort de pêche, les habitats les plus complexes (en termes de structure tridimensionnelle) étant les plus fortement impactés. Les impacts des chaluts à perche, des dragues et des panneaux des chaluts de fond semblent être relativement similaires (72). La profondeur à laquelle s'enfoncent ces engins dépend du type de sédiments (73) et varie de quelques centimètres (74) à une trentaine de centimètres pour les panneaux latéraux des chaluts de fond (71). Cette action physique sur les fonds se traduit par la remise en suspension de sédiments. Le nuage soulevé peut augmenter significativement la turbidité de l'eau jusqu'à dix mètres au dessus du fond (75). Les particules mises en suspension vont retomber vers le fond à des vitesses dépendant de leur taille. Les particules les plus fines formant des agrégats la vitesse de sédimentation n'est jamais inférieure à 0.5 mètres par jour (75). Cette hausse de la turbidité peut causer des phénomènes d'évitement par les juvéniles de poissons (76) et augmenter la mortalité des larves. Les poissons qui détectent leurs proies par la vue comme le turbot ou le maquereau sont les plus affectés par les hausses de la turbidité (77) et donc les plus susceptibles d'éviter la zone. Les impacts des engins trainants sont également très forts sur les communautés benthiques, même si le niveau d'impact réel en fonction de l'effort de pêche reste difficile à déterminer. En effet, toutes les études ne sont pas réalisées au même niveau taxonomique et sont souvent des observations avant/après passage d'un engin sur une zone donnée réalisées à petite échelle. Parmi les effets à court terme on observe une baisse significative du nombre d'individus sur la zone ainsi qu'une baisse du nombre de taxons observables (74). Après le passage d'un engin les crabes et étoiles de mer qui n'ont pas été endommagés ainsi que certains poissons sont très actifs et se nourrissent de la macrofaune endommagée (72). L'utilisation de chaînes sur les chaluts, destinées à augmenter la turbidité afin de faciliter la captures de certaines espèces de poissons, augmente la quantité d'espèces benthiques capturées de manière linéaire en fonction du nombre de chaînes. Les effets du chalutage sur les espèces benthiques semblent être relativement similaires à ceux sur les poissons en éliminant les espèces à croissance lente et à longue durée de vie (espèces de type K comme les mollusques ou les crustacés) au profit des espèces à croissance rapide de type r comme les vers polychètes (71). Dayton et al. (73) mentionnent des taux de mortalité, pour un unique coup de chalut, de l'ordre de 10 à 30% pour les pour les étoiles de mer, de 10 à 50% pour de nombreux mollusques et de 40 à 60% pour les crabes. Une méta-analyse (23) tente de quantifier de manière plus globale les effets des engins trainants sur les espèces benthiques. Cette étude indique qu'en moyenne le passage d'un chalut réduit de 46% le nombre d'individus sur un site, et de 55% si l'on étudie les effets taxon par taxon. Ceci est associé à une baisse moyenne du nombre d'espèces de 27%. Cette étude s'accorde également avec (78) pour mettre en avant le fait qu'une perturbation forte mais relativement ponctuelle dans les temps et l'espace va avoir des effets relativement plus faibles qu'une perturbation chronique et/ou à grande échelle. Le temps de récupération suite au passage d'un engin est très variable. Il peut aller de 100 jours sur des substrats sableux fréquemment perturbés aux populations opportunistes (23) jusqu'à 15 ans pour des fonds composés de coraux et d'éponges (78). Ce temps de récupération semble être en moyenne de l'ordre de 500 jours (23). Cette récupération se fait en plusieurs phases distinctes (24) avec colonisation de l'espace laissé libre par des espèces opportunistes. Si le sédiment a été trop altéré par rapport à son état d'origine la colonisation par les espèces opportunistes peut se révéler pérenne, ou des espèces différentes de celles présentes à l'origine peuvent coloniser ce milieu qui leur est devenu favorable.

Les impacts des activités d'extraction de granulats sur les fonds sont relativement similaires à ceux des activités de pêche. Une des différences principales est qu'une partie du sédiment est enlevé, ce qui est susceptible d'accélérer la disparition des couches meubles de surface. L'extraction ayant lieu sur des surfaces relativement réduites, de nombreux passages sont effectués au même endroit. De ce fait la zone d'extraction peut se retrouver localement creusée de plusieurs mètres (79). Ce creusement est d'autant plus important que la tête d'élingue est capable de pénétrer de 50 centimètres dans le sédiment. En France il est interdit d'effectuer un tamisage des granulats extraits sur le bateau. Cela signifie que tout le sédiment extrait est ramené au port. Lorsque ce tamisage est autorisé, il cause le rejet en mer des tailles de granulats non désirées. Ceci crée un nuage turbide important derrière les bateaux d'extraction. Les sédiments grossiers vont sédimenter rapidement mais les sédiments fins peuvent rester en suspension plus longtemps et ainsi se répartir sur une plus grande surface. Ces sédiments fins peuvent causer le colmatage de sédiments plus grossiers (76). Ceci augmente la mortalité des oeufs et larves présents sur la zone en réduisant leur respiration ou leur capacité à s'alimenter. Des impacts négatifs peuvent également être observés sur les espèces de benthos fouisseuses si ces dernières se retrouvent recouvertes par une couche trop importante de sédiments et étouffent avant d'avoir réussi à atteindre la surface du sédiment. La plupart des poissons semblent migrer hors de la zone durant l'extraction (77) pour y revenir dès que les sédiments fins sont retombés, même si les adultes sont capables de survivre dans des eaux très turbides. La récupération après extraction se fait de la même manière que celle après pêche sur le modèle classique de la succession écologique (24), cette récupération pouvant prendre plusieurs mois à plusieurs années selon le type de sédiment et les populations associées.

Les impacts des éoliennes sur les éoliennes peuvent être divisés en trois étapes : la phase de construction, la phase d'opération et la phase de déconstruction à la fin de la période de vie des éoliennes. Les phases de construction et de déconstruction on des conséquences similaires et ponctuelles sur l'écosystème. A l'inverse la phase d'opération du champ d'éolienne peut être une source de perturbations chroniques. Il apparait globalement que les perturbations chroniques causées par les éoliennes sont relativement modérées et que les perturbations ponctuelles en début et fin de vie du champ vont avoir des conséquences plus marquées, mais réversibles à plus ou moins long terme (80; 81). Ces observations ont en particulier eu lieu dans les eaux Danoises où l'implantation d'éoliennes en mer depuis de nombreuses années permet d'avoir des observations sur les différentes phases de vie du champ d'éoliennes

(82; 83). L'implantation d'un champ d'éolienne cause tout d'abord une modification du substrat, introduisant sur un fond généralement sableux des structures dures (béton, pierres, cables). L'introduction de ces structures dans le sol génère une hausse de la turbidité et des niveaux sonores très élevés susceptibles de causer l'émigration de nombreuses espèces mobiles, et en particulier de mammifères (mcConnell et al., 2012). Il apparait toutefois que ces effets négatifs sont assez faibles que le retour des espèces affectées prend de quelques mois à deux ans pour les plus sensibles (83). La modification du substrat cause une modification des communautés benthiques associées, avec en particulier l'apparition de grandes quantités de coquillages fixés (moules) sur les structures. Ces structures vont aussi modifier la circulation d'eau à leurs abords, mais sur seulement quelques mètres, et créer des abris supplémentaires. Ces modifications ne semblent toutefois pas avoir d'effets significatifs sur la nature ou l'abondance des populations de poissons présentes (83). En phase d'opération les éoliennes sont génératrices de bruit et de champs magnétiques, essentiellement au niveau des câbles qui les relient à la terre (81). Le bruit produit s'ajoute au bruit ambiant et son niveau ainsi que les fréquences produites ne semblent pas avoir d'effets significatifs sur la faune avoisinante. Les champs magnétiques produits par les câbles peuvent être détectés par les poissons utilisant habituellement ces derniers pour migrer. Peu d'effets des champs magnétiques des câbles ont pu être mis en évidence (81), il semblerait toutefois que les anguilles, espèce particulièrement sensible, puissent voir leur migration gênée par le passage des câbles (83). Hors de l'eau, les collisions avec des oiseaux sont extrêmement rares, ceux-ci ayant tendance à modifier leur trajectoire de vol afin d'éviter les champs d'éoliennes, de 1 à 5 kilomètres environ (83).

Les hydroliennes ne semblent pas être une source de mortalité directe notable pour les poissons. Toutefois, à la manière des éoliennes, elles modifient le comportement des poissons qui tendent à éviter les rotors, en particulier en cas de fort courant (84).

Les conséquences de ces différentes activités sur l'écosystème sont loin d'être négligeables. De plus, la Manche Orientale étant un espace de petite taille il est probable que plusieurs de ces activités doivent cohabiter dans des zones relativement réduites. Cela ouvre la porte à des interactions possibles entre ces activités, aux effets potentiellement difficiles à prévoir et devant être pris en compte dans les mesures de gestion appliquées à la zone.

Il apparait donc comme nécessaire de prendre en compte les impacts de ces activités sur l'écosystème. Dans le cas des activités en développement ou à venir le peu de recul dont on dispose rend le rôle des modèles encore plus important. En effet, ces derniers peuvent permettre de prédire dans une certaine mesure les conséquences possibles de ces activités sur l'écosystème. En particulier, il peuvent permettre de tester différents scénarios de développement de ces activités et de les confronter à une possible variabilité des conditions environnementales. Les modèles peuvent également avoir le rôle d'intégrateurs de la connaissance pour les activités dont les conséquences sont étudiées. Au fur et à mesure de l'intégration de la connaissance dans le modèle on va ainsi mieux pouvoir comprendre les interactions entre ces activités et leurs possibles effets cumulés.

2.1.3 Gestion actuelle et gestion possible

La gestion des pêches en Manche se base sur le principe de l'approche de précaution (85). Les espèces commerciales ont une taille minimale de débarquement visant à leur laisser atteindre leur âge de première reproduction avant d'être pêchés. Cette taille minimale est fréquemment associée à une limitation de la quantité totale de captures annuelle (86). Ce niveau de TAC (Total Autorisé des Captures) est actualisé annuellement sur la base de recommandations faites par les groupes d'évaluation des stocks du Conseil International pour l'Exploration de la Mer (CIEM). L'objectif premier de ces TACs est de maintenir la mortalité par pêche à un niveau suffisamment faible pour que les populations demeurent dans un état où le risque que ces dernières s'effondrent est faible. A cet objectif s'ajoute désormais celui de l'amélioration de l'état des stocks. Ceci passe par la définition d'une nouvelle cible de gestion : le Rendement Maximum Durable (RMD). Cela revient à contrôler l'effort de pêche afin de ne prélever que la quantité maximale pouvant être produite annuellement par le stock de poisson ciblé. Cet objectif est donc avant tout un objectif en termes de production. Toutefois un haut niveau de production est associé à des niveaux de biomasse assez élevés, ce qui implique de ce fait une nécessaire récupération de l'état des stocks. La transition depuis une gestion basée sur l'approche de précaution vers une gestion au RMD se fait de manière progressive, à l'échelle de plusieurs années (5 ans pour les stocks de poissons plats en Manche Est). Cette transition est contrôlée par une règle de gestion ajustant la baisse du TAC en fonction de diverses contraintes sur l'état du stock et la variabilité inter-annuelle du TAC. Le principe de la gestion au RMD est parfois contesté (87; 88) mais demeure un élément central de la gestion de nombreux stocks. L'effort de pêche peut également être géré directement, essentiellement au moyen de limitations du temps de pêche, de la définition de saisons de pêche, ou encore de limitations de la puissance des navires de pêche. Cette dernière mesure peut rejoindre les mesures dites techniques qui contraignent également l'activité de pêche. Ces mesures sont par exemple des interdictions de certains engins jugés trop destructeurs pour l'environnement, des limitations d'accès aux zones côtières ou encore des tailles de maillage limites et l'obligation d'utiliser des dispositifs d'échappement afin de ne sélectionner que certains individus.

La notion de stock de poisson illustre le caractère largement monospécifique et axé vers la production de la gestion actuelle. Il est désormais reconnu que cette gestion devrait être plurispécifique voire écosystémique afin de prendre en compte les interactions pouvant avoir lieu à différentes échelles de l'écosystème (1). Néanmoins cette approche de la gestion demeure difficile à mettre en place, d'où la construction de nombreux modèles à l'heure actuelle afin de mieux comprendre le fonctionnement des écosystèmes marins.

Un outil de la gestion écosystémique est les Aires Marines Protégées (AMPs) dont il existe de nombreux types répondant à divers objectifs (89). Sur les côtes françaises ces aires visent à protéger des fonctions écosystémiques particulières dans des zones données (par exemple un type d'habitat particulier et les populations associées), et ne sont pas réellement à l'heure actuelle des outils de gestion des pêches. L'efficacité des aires marines protégées pour la gestion des pêches demeure d'ailleurs très discutée (90; 91; 92). Ces zones ont aussi vocation à être des espaces de concertation permettant de prendre en compte tous les acteurs dans la gestion.

La gestion peut également se faire au moyen de leviers économiques, par exemple en attribuant des quotas de pêche individuels (QI) aux pêcheurs. Il est également possible d'autoriser l'achat et la vente de ces quotas de pêche entre pêcheurs ou organisation de pêcheurs. On parle alors de quotas individuels transférables (QITs, (93; 94)). De tels scénarios de gestion on été testés en Manche Est pour des stocks de poissons plats grâce au modèle de (22), qui représente le point de départ des modèles utilisés dans cette thèse. La gestion peut également se faire grâce à une taxe devant être payée par les pêcheurs pour toute capture au delà du TAC accordé (95). Ces moyens de gestion possibles s'ajoutent en général aux mesures communément appliquées de gestion de l'effort et de la nature des captures telles que des tailles minimales au débarquement, des restrictions sur les engins utilisables et la durée d'utilisation de ces engins. Dans le cas français une méthode mise en place afin de réduire l'effort de pêche est de subventionner la sortie de flotte de certains navires tout en versant des primes afin de moderniser ou de modifier les navires restants.

L'extraction de granulats marins et la production d'énergie au moyen d'éoliennes sont gérées par le biais de concessions attribuées pour une durée déterminée et associées à des limites sur les quantités maximales produites. Ces activités ayant une emprise spatiale pouvant limiter la présence d'autres activités (par exemple la pêche dans les champs d'éoliennes) il est envisagé des les utiliser comme des mesures de gestion spatiales (Blyth-Skyrme, 2010, Blyth-Skyrme, 2011).

Il apparait que la gestion appliquée à l'heure actuelle demeure très éloignée d'une gestion écosystémique, malgré ses évolutions récentes. Cela peut tenir à plusieurs facteurs : (i) la gestion monospécifique actuelle est déjà très complexe, (ii) on se focalise en priorité sur les impacts les plus visibles, en particulier d'ordre économique, et (iii) on connait mal les interactions directes et indirectes pouvant survenir entre les activités humaines et les différentes composantes de l'écosystème. L'approche proposée ici peut permettre certaines avancées vers une gestion plus écosystémique. Tout d'abord, on modélise conjointement plusieurs compartiments d'un écosystème ainsi que différentes mesures de gestion appliquées ou applicables, et leurs impacts. On peut de ce fait observer assez finement des interactions qui pourraient être noyées dans un "bruit ambiant" et difficiles à observer hors d'un cadre expérimental. De plus le modèle étant complexe il est possible de voir émerger des interactions difficilement prévisibles dont la recherche des causes peut améliorer notre connaissance du fonctionnement de l'écosystème. L'application des méthodes d'analyse de sensibilité au modèle construit permet d'identifier les paramètres les plus importants de ce modèle. De ce fait, il devient possible de tester les conséquences des différentes mesures de gestion et d'identifier celles ayant le plus d'impact sur l'écosystème. On peut alors tenter d'éliminer celles ayant le moins d'influence afin de tester des pistes pour une gestion simplifiée.

2.2 Recherche de mesures de gestion efficaces malgré de fortes incertitudes

"There are only two classes of forecasters : those who don't know and those who don't know they don't know." (John Kenneth Galbraith)

Les incertitudes sont un problème majeur pour les scientifiques et les prévisionnistes en général, et font partie intégrante de toute approche visant à acquérir des connaissances sur un sujet donné. Les économistes furent parmi les premiers à tenter de prendre en compte les incertitudes (96) et leurs effets possibles. D'autres disciplines suivirent et tentèrent d'appliquer leurs méthodes à des cas différents. Une de ces disciplines, dont l'essor s'est fait en parallèle de celui de l'informatique, est la modélisation. La hausse des capacités de calcul permet de créer et simuler des modèles de plus en plus complexes, intégrant un nombre croissant de paramètres afin de mieux représenter la réalité. Un des effets pervers de cette hausse des capacités de calcul peut être une course à la complexité, c'est à dire la création de modèles de plus en plus complexes sans justification à priori. Ceci peut causer une perte progressive du lien avec la réalité des processus modélisés et avec les finalités premières du modèle. Cette augmentation de la complexité part de l'intention à priori louable de réduire nos incertitudes sur les systèmes modélisés. En effet, augmenter la complexité d'un modèle peut en partie permettre de réduire les incertitudes, en prenant en compte explicitement des processus incertains (97). Le problème est qu'augmenter le nombre de processus pris en compte cause l'introduction de nouveaux paramètres mal connus dans le modèle, d'où une hausse des incertitudes (98). On peut aussi dire qu'augmenter la complexité du modèle augmente la complexité de l'incertitude, en rendant ses diverses sources plus difficiles à identifier. En effet les incertitudes peuvent influencer positivement ou négativement chaque paramètre d'un écosystème, le type et la magnitude de cette influence dépendant des caractéristiques du système lui-même, causant des réactions complexes et difficilement prévisibles (99).

2.2.1 Risque et sources d'incertitude

L'objectif quand on tente de réduire des incertitudes est de réduire le risque associé. Ce risque est généralement défini comme la probabilité d'occurrence d'un évènement non désiré (100). Cette probabilité peut être associée à une fonction de densité de probabilité, ainsi le risque peut être quantifié.

Le risque peut également être défini comme le produit de la probabilité d'occurrence d'un évènement non désiré et la perte qui serait subie dans le cas où cet évènement se produirait. Cette définition est en particulier privilégiée en théorie de la décision, où un choix entre plusieurs actions doit être fait selon les conséquences possibles de ces actions. Dans ce cas, un compromis est fait entre probabilité d'occurrence de l'évènement et ses conséquences possibles. Si l'évènement est très probable mais ses conséquences négligeables, le risque est très faible et peut ne même pas être pris en compte. Si à l'inverse les conséquences de l'évènement seraient catastrophiques mais la probabilité d'occurrence est infime, le risque est là encore très faible.

Les limites de cette approche apparaissent rapidement, en particulier pour les évènements "extrêmes" qui ont des probabilités d'occurrence faibles mais des conséquences potentiellement désastreuses. Ces évènements étant très rares ils ont été très peu observés et il est par conséquent très difficile de choisir une loi de densité de probabilité permettant de bien les prédire. Ceci est d'autant plus problématique que les évènements exceptionnels se trouvent systématiquement dans la queue de distribution de la fonction de densité de probabilité utilisée pour les prédire. Or, d'une fonction à l'autre la probabilité d'occurrence d'un évènement peut varier d'un facteur 10, 100, ou plus, en se plaçant à un même endroit de la fonction ("fat tails" (101)). De ce fait, les évènements que l'on souhaiterait pouvoir prévoir avec la plus grande précision sont en fait les plus imprévisibles.

Un troisième cas apparait alors : lorsque les conséquences de l'évènement indésirable seraient si graves qu'il doit absolument être pris en compte. Dans ce cas précis, on fait abstraction de la probabilité d'occurrence de l'évènement pour se focaliser sur ses conséquences. Deux lignes de conduite complémentaires peuvent alors être adoptées. La première consiste à mettre en place les conditions permettant de rendre impossible (idéalement) la survenue de l'évènement indésirable. La seconde consiste à altérer le système à protéger de sorte à minimiser l'impact de l'évènement extrême lorsqu'il surviendra. L'impact d'un évènement extrême dépend lui-même de nombreux paramètres incertains et peut être très difficile à modéliser.

Les incertitudes sont la source des erreurs que l'on peut observer dans un modèle (102). Il est désormais reconnu que les incertitudes et leurs sources doivent être prises en compte en gestion des pêches et de nombreux auteurs s'attachent à proposer des solutions à ce problème (97; 103; 104; 105; 106; 107; 99; 108; 109; 110; 111). Il apparait que le nombre de ces sources, ainsi que leur définition, varie d'un auteur à l'autre en fonction du niveau de détail nécessaire et du domaine étudié. En règle générale, au moins trois sources d'incertitude sont identifiées (112; 113; 114), voire six en gestion des pêcheries (102) ou beaucoup plus si on s'intéresse à l'écologie en général (115). En ce qui concerne l'exercice de modélisation réalisé ici, trois sources d'incertitude semblent prévaloir : (i) les erreurs d'observation, causant une mauvaise estimation de certains paramètres ; (ii) la variabilité des processus naturels (variabilité autour de la valeur estimée) ; (iii) les erreurs dans la structure du modèle (causées par la mauvaise connaissance des processus modélisés ou de mauvais choix de modélisation). Il est à noter que l'interaction de ces différentes sources d'incertitude peut faire apparaitre des sources d'incertitude secondaires. Ici, nous nous intéressons en particulier aux deux premières sources d'incertitude et tentons de comprendre comment elles altèrent le diagnostic de l'état d'un écosystème et comment s'en prémunir grâce à l'adaptation des mesures de gestion.

2.2.2 Info-gap et Théorie de la Décision : quelques repères

Une première approche visant à permettre la prise de décision dans un contexte d'incertitude a été proposée par Abraham Wald dès 1939. Récemment, des approches de ce type ont commencé à être appliquées à la modélisation écosystémique et à la protection des espèces. Ces approches (116; 117) sont en particulier de type info-gap (118; 119) qui peut être vue comme une adaptation simplifiée de la Théorie des Jeux et de la Théorie de la Fonction de Décision Statistique de Wald.

2.2.2.1 La théorie des jeux

La Théorie des Jeux (120) prend en compte une large gamme de situations où des joueurs "s'affrontent" dans le but d'obtenir quelque chose. Si l'on se place dans une situation aussi simple que possible, un jeu à somme nulle (donc non coopératif) entre deux joueurs et joué en un seul coup (chaque joueur fait un et un seul choix). L'objectif de chaque joueur est de maximiser son gain et de minimiser ses pertes. Si l'on nomme K le gain, comme le jeu est à somme nulle l'un des joueurs obtiendra K tandis que l'autre obtiendra $-K$. Chaque joueur a le choix entre deux stratégies : une stratégie poure ou une stratégie mixte. Adopter une stratégie pure revient à choisir une action (τ) parmi toutes celles disponibles. Choisir une stratégie mixte revient à attribuer une probabilité (ρ) à chaque action possible puis laisser un tirage dans cette loi de probabilité décider de l'action effectivement adoptée. La question centrale dans ce genre de jeux est celle du niveau d'information dont dispose chaque joueur, car cela va grandement influer sur leur comportement. Si chaque joueur connait exactement toutes les options dont dispose son adversaire, la probabilité que celui-ci a de choisir une option plutôt qu'une autre et le gain associé à chaque action, on dit que le jeu se fait en information complète. Si une de ces conditions n'est pas remplie on est alors en situation d'information incomplète.

	Probability	$\rho_{2,1}$	$\rho_{2,2}$	$\cdots$	$\rho_{2,n}$
Probability	**Player 2** **Player 1**	$\tau_{2,1}$	$\tau_{2,2}$	$\cdots$	$\tau_{2,n}$
$\rho_{1,1}$ $\rho_{1,2}$ $\vdots$ $\rho_{1,m}$	$\tau_{1,1}$ $\tau_{1,2}$ $\vdots$ $\tau_{1,m}$	$K(\tau_1, \tau_2)$			

TABLE 2.1 – Mixed Strategy under Risk

La différence entre la prise de décision dans un contexte de risque et la prise de décision dans un contexte d'incertitude est que dans un contexte de risque toutes les probabilités ρ sont supposées connues, alors qu'elles ne le sont pas dans un contexte d'incertitude (121). Il peut être difficile de prédire le comportement des joueurs lorsque ces derniers disposent de beaucoup d'information sur le jeu. En effet, dans ce cas leur comportement va pouvoir être influencé par leur tendance personnelle à prendre des risques, le comportement qu'ils attendent de la part de l'autre joueur, le niveau de gain ou de perte correspondant à chacune des actions disponibles, etc. A l'inverse, si peu d'information est disponible et que chaque joueur connait uniquement le niveau de gain (ou de perte) associé à chaque combinaison d'actions (la sienne et celle se son adversaire) alors ils vont avoir tendance à utiliser des stratégies pures. Dans ce cas où l'information manque il devient intéressant de ne pas choisir l'action qui pourrait donner le plus fort gain, car le comportement de l'adversaire est inconnu, mais l'action qui garantit le meilleur niveau de gain. Cette approche revient pour chaque joueur, selon la situation, soit à maximiser son gain minimal (maximin) soit à minimiser sa perte maximale (minimax). Cette approche est très restrictive et peut conduire à ne pas prendre en compte toute l'information disponible (20). Néanmoins, cette action a l'avantage d'offrir une certaine robustesse aux incertitudes : quelle que soit l'action choisie par son adversaire un joueur adoptant cette stratégie se garantit que son gain ne sera pas inférieur à un certain seuil (ou que sa perte ne sera pas supérieure à un certain seuil).

2.2.2.2 La Fonction de Décision Statistique de Wald

La théorie de la décision de Wald (122; 123; 124; 17) vise à permettre la prise de décision dans un situation de forte incertitude, situation que nous avons commencé à illustrer précédemment. L'approche de Wald consiste à chercher la stratégie qui minimiserait la perte. Un aspect important de l'approche de Wald est l'utilisation

Player 2 / **Player 1**	$\tau_{2,1}$	$\tau_{2,2}$	$\cdots$	$\tau_{2,n}$
$\tau_{1,1}$				
$\tau_{1,2}$				
$\vdots$			$K(\tau_1, \tau_2)$	
$\tau_{1,m}$				

Table 2.2 – Various Pure Strategies under Uncertainty

d'une fonction de poids qui modifie la valeur de la perte, en fonction de critères fixés par le décideur. (125) présente la théorie de wald de la manière suivante :

- La fonction de poids $W_{\alpha,\beta}(\theta, \bar{\theta})$ est définie de telle sorte que :

$$\left\{ \begin{array}{lcll} W_{\alpha,\beta} & = & 0 & if\ d(\theta,\bar{\theta}) \leq \beta \\ W_{\alpha,\beta} & = & [d(\theta,\bar{\theta})]^{\alpha} & if\ d(\theta,\bar{\theta}) > \beta \end{array} \right.$$

  ,$d(\theta, \bar{\theta})$ étant la distance euclidienne entre le point θ et son estimation $\bar{\theta}$.
- Le risque $r_{\alpha,\beta}(H, S)$ est calculé comme :

$$r_{\alpha,\beta}(H,S) = \int [d(\theta, \bar{\theta_S})]^{\alpha} dH(\theta)$$

- L'objectif étant de trouver la stratégie optimale pour $W_{\alpha,\beta}$ et un H donné, notée $S^{(\alpha,\beta)}(H)$, c'est à dire la stratégie minimisant $r_{\alpha,\beta}(H, S)$.

Dans son article de 1945, Wald ajouté un aspect fondamental à sa théorie : il suggéra un ensemble d'indicateurs qui ne tenteraient plus de minimiser une moyenne pondérée du risque mais plutôt le risque maximum, posant ainsi les bases de la théorie minimax. Cette inclusion d'une approche minimax dans les travaux de Wald est d'un grand intérêt pour au moins deux raisons :

- Elle fourni une solution au problème de la prise de décision dan sun contexte d'incertitudes, même si c'est une solution très conservatrice.
- Cela permit à Wald de démontrer que le problème d'inférence statistique de sa méthode peut être interprété comme un jeu à somme nulle à deux joueurs dans la théorie de von Neumann et Morgenstern. Le joueur 1 est alors le gestionnaire, qui doit choisir entre plusieurs actions possibles, et le joueur 2 est la Nature. L'hypothèse est faite que la Nature va systématiquement donner la pire issue possible pour l'action choisie par le gestionnaire. De ce fait, le gestionnaire n'a d'autre choix que celui de l'action avec la meilleure pire issue, c'est à dire de minimiser une perte maximale ou de maximiser un gain minimal, selon la variable étudiée et les objectifs du gestionnaire.

Les différents états de la Nature peuvent être vus comme autant de valeurs possibles pour un même paramètre, représentant l'incertitude autour de ce paramètre.

	Probability	η_1	η_2	$\cdots$	η_n
Probability	**Nature** **Manager**	F_1	F_2	$\cdots$	F_n
ξ_1	$\mathfrak{D}_1 = d(x)_1$	$K(\xi,\eta) = r(\xi,\eta) = \int_{\mathcal{Q}} \int_{\Omega} r(F,\mathfrak{D}) d\xi d\eta$			
ξ_2	$\mathfrak{D}_2$				
$\vdots$	$\vdots$				
ξ_m	$\mathfrak{D}_m$				

Table 2.3 – Statistical Decision Functions Theory (mixed strategy)

Si on fait le choix de ne pas donner de probabilité à chaque état de la Nature (ce qui revient en définitive à leur attribuer des probabilités égales) alors on peut calculer l'issue de la combinaison de chaque option de gestion avec chaque état de la Nature et déterminer quelles combinaisons donnent des résultats acceptables selon les critères du gestionnaire.

Nature **Manager**	F_1	F_2	$\cdots$	F_n
$\mathfrak{D}_1 = d(x)_1$	$K(F,\mathfrak{D}) = r(F,\mathfrak{D}) = \int W_{\alpha,\beta}(F,\mathfrak{D}) dF(d(x))$			
$\mathfrak{D}_2$				
$\vdots$				
$\mathfrak{D}_m$				

Table 2.4 – Statistical Decision Functions Theory (pure strategy)

Wald pousse l'analogie encore plus loin dans son article de 1950, en suggérant l'utilisation d'une fonction de poids simplifiée ne pouvant prendre que les valeurs 0 et 1 (0 correspondant à une décision correcte et 1 à une décision fausse). Ainsi, on voit que Wald suggère que β constitue le seuil entre une bonne décision et une mauvaise décision (toujours relativement à des objectifs de gestion), β étant une valeur de la variable de sortie étudiée.

2.2.2.3 La théorie du Fossé de l'Information

La théorie du Fossé de l'Information (information-gap, (118; 119)) vise elle aussi à permettre la prise de décision malgré la présence de fortes incertitudes. Elle est basée sur deux fonctions : une fonction de robustesse ayant pour rôle de garantir l'immunité aux échecs et une fonction d'oportunité déterminant les conditions d'accession à un gain dépassant largement les objectifs de base. En règle générale la fonction de robustesse est utilisée seule, l'objectif prioritaire étant de se prémunir face aux incertitudes. De la même manière que les méthodes précédentes, la théorie info-gap permet de comparer l'efficacité de plusieurs actions (q) en fonction de divers états de la Nature (u). Il est intéressant que les états de la NAture sont aussi nommés "incertitude ambiante", ce qui illustre bien le fait simuler plusieurs valeurs pour un même paramètre environnemental revient à faire une analyse d'incertitude autour de ce paramètre. La fonction de gain $R(q, u)$ correspond à la fonction K dans la Théorie des Jeux ou dans l'approche proposée Wald. Cette fonction est communément obtenue grâce à un modèle mathématique simple permettant de la calculer à partir d'un nombre réduit de paramètres d'entrée. Dans le cas où l'on souhaite prendre en compte un plus grand nombre de paramètres, des modèles complexes peuvent aussi être utilisés pour calculer $R(q, u)$. Si ces modèles sont déterministes (comme ISIS-Fish) alors une simulation correspond à la combinaison entre une valeur de chaque paramètre de l'environnement et une valeur de chaque paramètre déterminant la stratégie de gestion testée. Si le modèle utilisé est stochastique un nombre beaucoup plus élevé de simulations est nécessaire, chaque valeur de $R(q, u)$ étant alors un résultat moyen issu de plusieurs simulations réalisées avec les mêmes valeurs de paramètres d'entrée.

	Uncertainty Model
Decision scenario	System Model + Performance Criterion

TABLE 2.5 – Structure of an Info-Gap model

Ambient Uncertainty **Decision Vector**	u_{-n}	$\cdots$	u_{-1}	u_0	u_1	$\cdots$	u_n
q_1	$R(q, u)$						
q_2							
$\vdots$							
q_m							

TABLE 2.6 – Taking the Analogy Further

En fonction de ses objectifs le gestionnaire choisit une valeur critique (r_c) de la variable de sortie étudiée, qui est le seuil sous lequel on ne veut pas que la variable descende. A partir de ce seuil on peut définir la fonction de robustesse (notée $\hat{\alpha}(q, r_c)$), qui correspond au plus grand horizon d'incertitude α pouvant être toléré sur les états de la Nature tout en garantissant que la fonction de gain (la variable de sortie) ne descend pas sous le seuil r_c. Pour pour chaque action testée le niveau d'incertitude pouvant être toléré sur les paramètres environnementaux est ainsi déterminé et le gestionnaire peut choisir l'action qui lui semble être la plus appropriée. Cette analyse est par nature locale car elle est réalisée autour de la valeur de référence des paramètres du modèle (126), la taille de l'espace exploré pouvant toutefois varier selon les besoins du modélisateur.

Un exemple de mise en place d'une approche info-gap peut être trouvée dans (116) qui appliquèrent cette méthode à un modèle simple afin d'étudier l'impact de l'intensité de marquage des nids sur la survie du Pluvier montagnard. La procédure mise en place peut être résumée de la manière suivante :

1. Choix d'un effort de marquage (la seule variable pouvant être controlée).
2. A partir de cette valeur, calcul de la valeur de référence des autres paramètres.
3. Perturbation des valeurs de référence de chaque paramètre et utilisation de ces valeurs pour calculer la valeur de la variable de sortie (le taux de croissance d'une population dans ce cas).
4. Sélection des groupes de paramètres donnant uniquement des résultats compatibles avec le critère de sélection.
5. Retour à l'étape 1 avec changement de la valeur d'effort.

Cette approche peut être représentée sous la forme d'un tableau, en représentant uniquement un paramètre environnemental par soucis de simplicité :

Si le modèle utilisé représente n paramètres représentant l'environnement, alors le tableau présenté ci-dessus devient n+1 dimensionnel. Cela signifie que le nombre de calculs (simulations) nécessaires pour explorer cet espace de paramètres augmente extrêmement rapidement avec le nombre de paramètres testés.

Appliquer une telle méthode revient donc à explorer l'espace des paramètres d'entrée d'un modèle, jusqu'à trouver la frontière entre la zone de l'espace des paramètres d'entrée donnant des résultats acceptables et celle conduisant à des échecs 2.1. Cette méthode devant être appliquée à un modèle ayant des temps de simulation potentiellement longs, cela revient à : (i) trouver d'une part une méthode efficace d'exploration de l'espace afin de pouvoir obtenir une image correcte de cet espace en un temps limité et (ii) une fois cet espace exploré trouver une méthode permettant de le découper afin d'isoler les zones d'intérêt.

Nous proposons ici une approche basée sur les techniques d'analyse de sensibilité afin d'explorer l'espace des paramètres d'entrée d'un modèle. Cette méthode peut être couplée à l'utilisation d'arbres de classification afin de découper cet espace. Cette méthode permet ainsi de déterminer quels scénarios de gestion sont les plus robustes à la variabilité des paramètres de l'écosystème, et quel niveau d'incertitude ces scénarios permettent de tolérer.

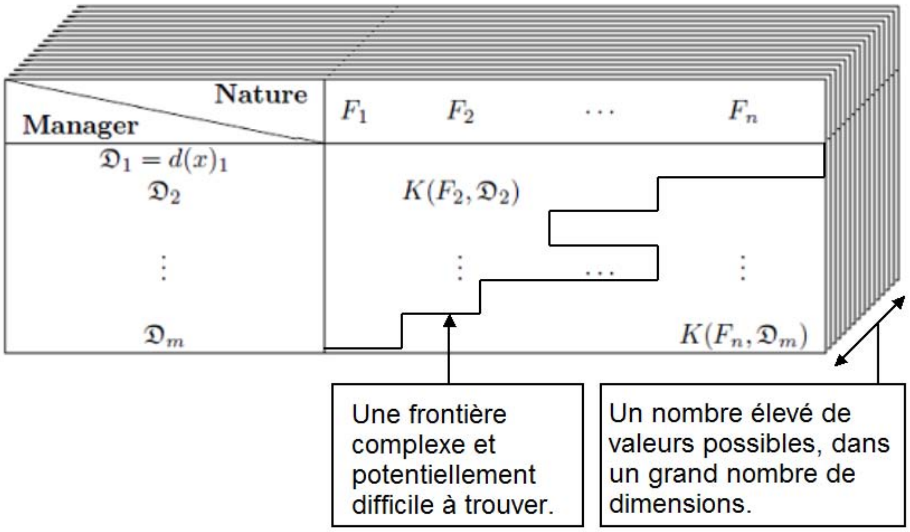

FIGURE 2.1 – **The two issues to be solved.** The first issue is that of the exploration of the input parameter space, that can be high-dimentional. The second issue is to find the boundary between the zone leading to successes and that leading to failures.

Action \ Parameter value	$u_{-p,1}$	$\cdots$	$u_{-1,1}$	$u_{0,1}$	$u_{1,1}$	$\cdots$	$u_{n,1}$
E_1	$f(u_{-p,1}, E_1)$	$\cdots$	$f(u_{-1,1}, E_1)$	$f(u_{0,1}, E_1)$	$f(u_{1,1}, E_1)$	$\cdots$	$f(u_{n,1}, E_1)$
Action \ Parameter value	$u_{-p,2}$	$\cdots$	$u_{-1,2}$	$u_{0,2}$	$u_{1,2}$	$\cdots$	$u_{n,2}$
E_2	$f(u_{-p,2}, E_2)$	$\cdots$	$f(u_{-1,2}, E_2)$	$f(u_{0,2}, E_2)$	$f(u_{1,2}, E_2)$	$\cdots$	$f(u_{n,2}, E_2)$
$\vdots$				$\vdots$			
Action \ Parameter value	$u_{-p,m}$	$\cdots$	$u_{-1,m}$	$u_{0,m}$	$u_{1,m}$	$\cdots$	$u_{n,m}$
E_m	$f(u_{-p,m}, E_m)$	$\cdots$	$f(u_{-1,m}, E_m)$	$f(u_{0,m}, E_m)$	$f(u_{1,m}, E_m)$	$\cdots$	$f(u_{n,m}, E_m)$

TABLE 2.7 – Using info-gap in computer simulations

Chapitre 3

Impacts de l'extraction de granulats marins sur les communautés benthiques de Manche Est - Gros plan sur les espèces consommées par la sole et la plie.

Sommaire

Impacts of aggregate extraction on benthic communities in the English Channel - A zoom on species consumed by sole and plaice.

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3.1 Abstract

We aim to assess the effects of sediment extraction on benthic species consumed by sole and plaice in the Eastern English Channel. A functional approach based on benthic species biological traits (trophic and mobility guilds) is used as i) groups of traits respond to human pressure in a more predictable way than individual species and ii) it also favours the transferability of our results.

The density of benthic species was measured for six years on an experimental dredging area located in the Bay of Seine. The extraction area was divided into several sites displaying different levels of extraction at different periods of time. We tested the significance of relationships between aggregates extraction and benthos density by means of mixed-effects models.

When studying species grouped according to their mobility significant negative impacts of dredging can be observed on burrowers, whereas mobile species seem to be less impacted. When grouping species according to their diet, impacts of dredging particularly show on carnivorous species. Detritus feeders seem to be positively impacted by dredging, at least for a few months after dredging, which may underline their opportunistic behaviour. In several cases, increases in abundance after dredging cannot be linked to a recovery time and are linked to strong inter-annual variability in benthos abundance.

3.2 Introduction

Benthic species (macro and mega fauna) contribute substantially to the energy flows in marine ecosystems (127; 128; 129; 130), recycling organic matter and reworking sediments. They occupy several key functions and respond quickly to ecosystem changes (131; 132).

These species are however strongly impacted by fishing activities. Known effects on benthic communities include strong decreases in their abundance and biomass (72; 73; 133; 134), loss of diversity (74; 23), increased patchiness (135; 136) and biological shifts (137) or colonisation by opportunistic species. Fishing activities also affect indirectly benthic communities by altering sediments that form the habitats where they live (71). Known effects on sediments include loss of tridimensional structure (78), levelling of dunes and mounds, loss of some sediment sizes, siltation by fine deposits (27; 138; 139). All these indirect effects are combined with direct effects, thereby affecting benthic communities in multiple ways.

Another human activity that impacts benthic communities is aggregate extraction. The effects of sediment dredging are often deemed to be similar to those of fishing, and fewer studies actually focus on sediment extraction (140; 77; 76). Nevertheless, this activity is becoming increasingly studied, especially in areas such as the English Channel or the North Sea (141; 142; 143; 144; 145; 146; 147; 148; 24; 149), where it may compete with other human activities. It appears however that sediment extraction differs from fishing on various aspects : (i) sediment is not just ploughed, part of it is also taken away and the grading changes ; (ii) the furrows left by the suction dredge are not as wide as those left by a clam dredge or a trawl, but they are deeper as the dredge digs 50 cm below the sediment surface ; (iii) the area where sediment is extracted is usually much smaller than a fishing area (150) and the dredging intensity can vary a lot ; (iv) sediment is taken to the surface and sediment sizes that are not wanted can be discarded, which may lead to bigger plumes than those caused by trawling where the turbid cloud is usually only a few meters high. This may lead to important modifications of the dredged area, with possible depth variations of few meters and the disappearance of some sediment sizes from dredged areas.

In this study we investigate such processes. To do this, we analyse data from an experimental dredging site located in the Bay of Seine where densities of benthos were measured with and without aggregate extraction. From this small-scale study we aim to evaluate the consequences of aggregate extraction on benthic communities at different time- and spatial-scales.

The diversity of benthic species found on the seafloor can be high, even at limited spatial scales. In order to make our results as general as possible the studied species are grouped according to their trophic or mobility guild so as to determine if particular feeding behaviours or ways of moving are more impacted than others. Two groups of benthic taxa are studied : on the one hand those eaten by sole and on the other hand those eaten by plaice, in an attempt to get insights of possible consequences of changes in benthic communities on these important fish species. These groups are defined based on the diets of sole and plaice found in (151; 25; 152; 153; 154; 155).

This article is part of a wider project that aims at taking benthic guilds into account in a complex ecosystem model of the Eastern English Channel. To do this, we need to quantify relationships between aggregates extraction activities and the state of the benthic communities themselves. This is a new step taken in comparison to other studies made in nearby areas (145; 146; 147; 156) where, even if extracted volumes were known in some detail, no attempt was made to directly link impacts on the benthic guilds to the intensity of the aggregates extraction activity. It is known that the relationship between a dredging intensity and a level of perturbation is not linear (73; 78) but to depend on other factors such as the frequency of the dredging events, the length of the dredging period or the size of the dredged area. To take these aspects into account we link several variables defining the aggregates extraction activity to densities of benthic species in the experimental area. Other variables defining the state of benthic communities such as biomass, species richness or guild structure are not studied here. This is done by means of simple linear mixed-effects models adjusted to the available data, both at a short term that corresponds to direct impacts of aggregates extraction and at a longer term corresponding to recovery.

3.3 Material

The data used in this paper come from annual scientific surveys set up jointly by the University of Rouen and the Cellule de Suivi du Littoral Normand (CSLN). The surveys took place from 2007 to 2012 on an experimental extraction area located in the Bay of Seine (Fig. 3.1). The studied area was 20 metres deep before extraction and some zones were up to five metres deeper in 2012 due to the extraction activity. The sediment is essentially made of coarse sand and gravels, with mud and shells in lower proportions.

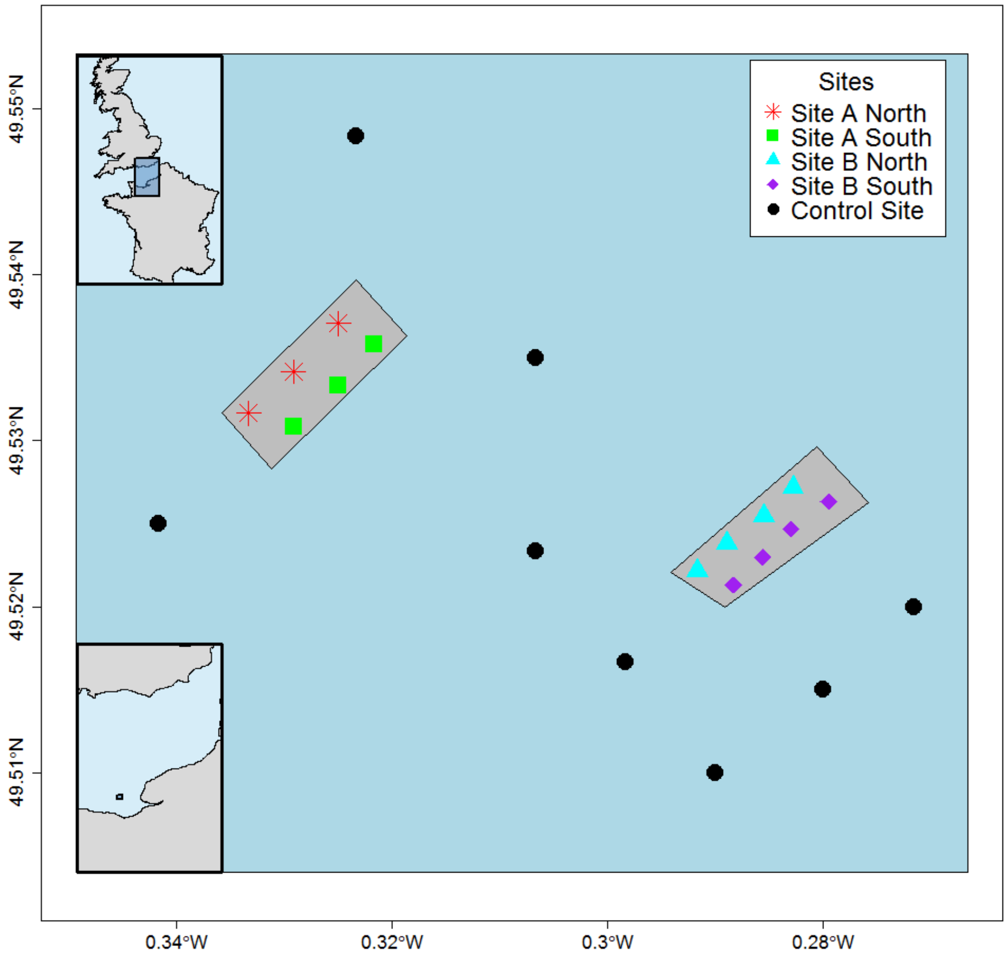


FIGURE 3.1 – **Sites and theoretical location of the sampling stations.** Not all sites were sampled every year. In particular sites of zone B were not sampled in 2007 and 2009. For years when zone B is not sampled, the three control stations closest to this zone (located at the South of zone B) are not taken into account in the analysis.

The experimental area contains two extraction zones (A and B). Extraction zones A and B are each further divided longitudinally into two sites (ANorth and ASouth for zone A and BNorth and BSouth for zone B). The control site with no extraction is mostly located between zones A and B (Fig. 3.1). Sites in zones A and B display different extraction intensities and periods. Indeed, sediment extraction started earlier on sites belonging to zone A and only later on those of zone B, where it also lasted longer (Fig. 3.2). Therefore extraction stopped much later on zone B compared to zone A (no extraction on sites ANorth and ASouth from 2009 onwards) and sediment volume extracted on zone B was much higher. As a consequence, the recovery of zone A could be investigated for several years whereas only a couple of years could be observed on zone B after extraction stopped.

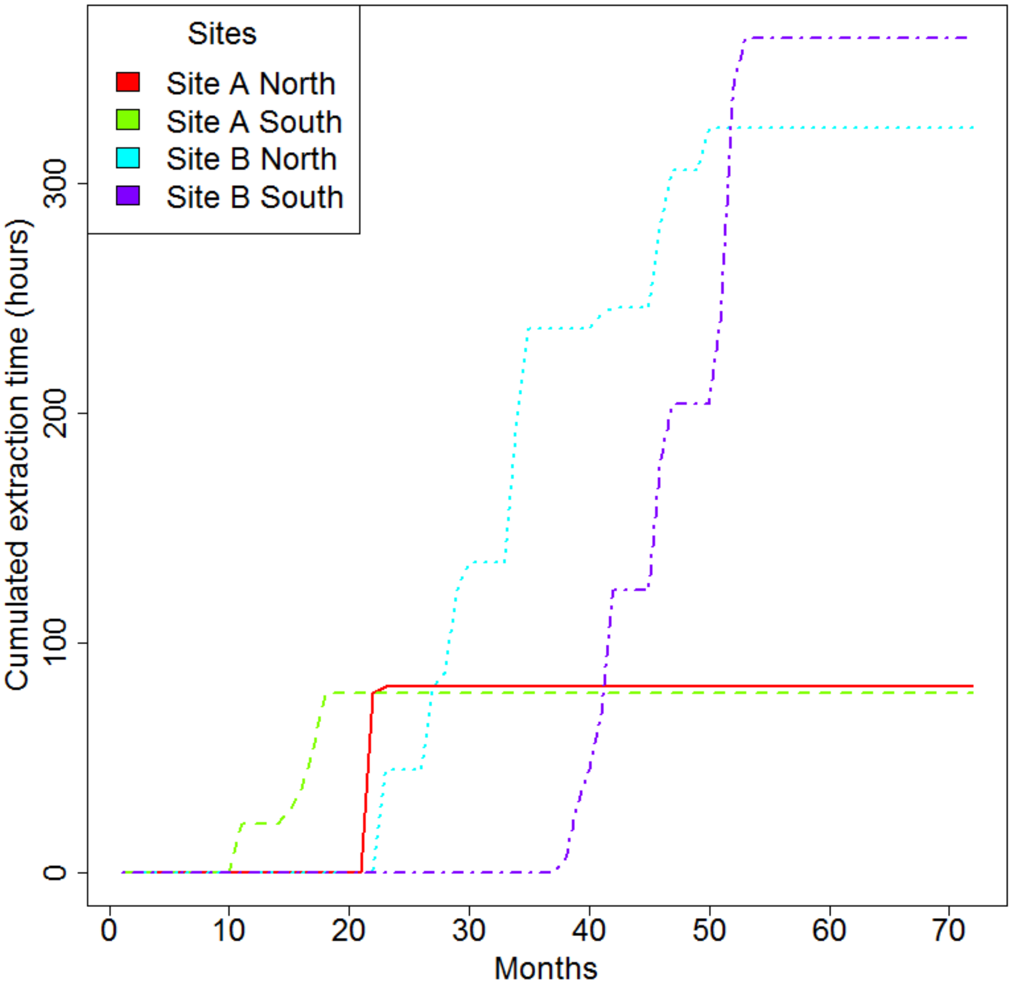


FIGURE 3.2 – **Cumulated extraction time (hours) per site.**

Each of the four sites was sampled on at least three stations by means of a mini-Hamon grab ($0.1m^2$). At each station, three replicates were made for benthic identification and a fourth one for sediments. The sampling surveys took place at the end of February / beginning of March. Not all sites and stations were sampled every year. In particular, no sampling was undertaken on sites of zone B in 2007 and 2009 (Table 3.1).

Site	Station	2007	2008	2009	2010	2011	2012
ANorth	A1	X	X	X	X	X	X
	A3	X	X	X	X	X	X
	A5	X	X	X	X	X	X
ASouth	A2	X	X	X	X	X	X
	A4	X	X	X	X	X	X
	A6	X	X	X	X	X	X
BNorth	B101		X		X	X	X
	B102		X		X	X	X
	B103		X		X	X	X
	B104		X		X	X	X
BSouth	B1		X				X
	B2		X				X
	B3		X				X
	B4		X				X
Control	R1	X	X	X	X	X	X
	R2	X	X	X	X	X	X
	R3	X	X	X	X	X	X
	R4	X	X	X	X	X	X
	R5	X	X	X	X	X	X
	R6		X		X	X	X
	R7		X		X	X	X
	R8		X		X	X	X

Table 3.1 – Stations sampled.

The sediment is dominated by gravels and fine sands and stations are very similar, with no noticeable gradient (79). The amount of fine sediments, in particular muds, increases along time on stations belonging to the reference site. This can be linked to an increase in turbidity caused by the aggregates extraction occurring in adjacent areas. However, this increase is not significant (79). On extracted sites, the proportion of gravels tends to increase while those of fine sediments and muds slightly increase. However, these changes are not marked enough to be significant (79). Therefore it can be said that observed effects of aggregates extraction on benthic communities in this area are more likely to be direct ones than indirect ones caused by changes in the sediment composition. This agrees with conclusions of (146) and experimental studies by (156) and (147).

Before extraction the experimental area was largely dominated by worms. The five species with the highest densities, over the 639 species observed, represented 68% of the average total density measured over all stations. The most important species in terms of density was the sedentary worm *Pomatoceros triqueter* with 481 individuals per square meter. Then two mobile worms (*Lumbrineris gracilis* and *Protodorvillea kefersteini*) were found, with densities of 102 and 80 individuals per square meter respectively. *Echinocyamus pusillus* is a small sea urchin with a density of $33 inds.m^{-2}$ and is the only species among the five most abundant that is not a worm. The last species of this list is the sedentary worm *Caulleriella alata* that had an average density of $27 inds.m^{-2}$ before the beginning of extraction.

3.4 Methods

3.4.1 Grouping Taxa

Our goal is to determine the consequences of sediment extraction on benthic communities. The study area being small (2 * 0.6 km^2 zones ; 8 km^2 in total, including the control site) and benthic species diversity being high, even at small spatial scales, results obtained at the species level would be difficult to interpret. Furthermore, (157) showed that groups of species respond to environmental changes in a more predictable way than individual species. We thus assume that the response of benthic species to extraction depends more on their morphology, their ability to move (or not), or their feeding habits than on their taxonomic classification. Therefore, we classify species according to their feeding habits and mobility.

Species are grouped in the trophic guilds developed by the French Benthic Network (REBENT) in 2002 (158). These guilds distinguish how species feed and what they feed on, and were elaborated using 1200 species from an area close to our study site.

In total, 639 different taxa were identified on the sampling stations from 2007 to 2012. Many of the benthic organisms found could not be identified at the species level and were identified at the genus or higher taxonomic level. For benthic organisms identified at the genus level, or species that did not appear in the Rebent classification, we made the following hypothesis : if two or more species with that same genus appear in the Rebent classification and share the same trophic index, then this index is attributed to the species or genus missing a trophic index. Using this method, we managed to classify 350 taxa found during sampling.

We also grouped the species (species and genus) depending on their mobility, using the Biological Traits Information Catalogue (BIOTIC) of the Marine Life Information Network (MarLIN) (159). Most information available in the MarLIN database is at the genus level. Assuming that all the species belonging to a given genus belong to the same mobility group, we classified 250 of our taxa.

Therefore, using the two guilds, we obtained information on 193 of the taxa found in the area (out of 639) which represent 62% of the total mean initial density of benthos in our data. Furthermore, we simplified the mobility groups by keeping only three categories (Mobile, Burrower and Attached) and making combinations of these categories.

3.4.2 Variables

3.4.2.1 Density per benthic group

The studied variable is the density of benthos per square meter. Densities per group (trophic or mobility guild) can be computed at the station or at the site level. Here, we are interested in the density of benthic groups per station because analyses are performed on all stations at once, notwithstanding the notion of site.

To obtain densities per group at the station level, densities of all species belonging to the same group are summed for each station. Let s be a station of site S, y the density and N an abundance. Let M be the area sampled for a given station, usually $0.3m^2$ because three replicates are made for each station and the mini-Hamon grab is $0.1m^2$. With G a given group and g a species belonging to this group, the density per group per station for a given year i can be computed as :

$$Y_{s,i}^{G} = \frac{\sum_{g \in G} N_{s,i}^{g}}{M_{s,i}} = \frac{\sum_{g \in G} \sum_{grab=1}^{3} N_{s,grab,i}^{g}}{\sum_{grab=1}^{3} M_{s,grab}} \tag{3.1}$$

This allows us to remove intra-station variability and account for inter-stations variability in the analyses, so that the patchy distribution of most of the studied species can be taken into account.

3.4.2.2 Extraction Intensity

Each of the four studied sites has a particular dredging period, that can be defined by an initial month m_i and a final month m_f. Thus for a given site S this period is : $[m_{i,S}; m_{f,S}]$, with m varying between 1 and 72 because six years have been observed. The duration of this period, in months, can be computed as :

$$D_s = m_{f,S} - m_{i,S} + 1 \tag{3.2}$$

Sites are composed of several stations. Therefore, all stations belonging to site S have the same initial and final months and the same duration of extraction : $\forall s \in S, m_{i,s} = m_{i,S}$, $m_{f,s} = m_{f,S}$ and $D_s = D_s$.

For a given month $m \in [m_{i,S}; m_{f,S}]$ both the extracted volume of sediment and the corresponding extraction time are known. We used the extraction time as the volume is determined by the power of the suction dredge and the depth at which it digs whereas the extraction time is directly linked to the surface of sediment disturbed and thus to the impacts on benthic species. This amounts to making the hypothesis that all dredge types dig deep enough to excavate all burrowing species. The monthly time of extraction for a given site S is $t(S, m)$ (in hours). Thus, the monthly time of extraction for a station s belonging to site S is $t(s, m) = t(S, m)$.

Therefore the cumulated extraction time (in hours) over all the extraction period for station s can be computed as :

$$\forall s \in S, T_s([m_{i,s}; m_{f,s}]) = \sum_{m=m_{i,s}}^{m_{f,s}} t(s,m) \tag{3.3}$$

Consequences of aggregates extraction do not depend only on punctual dredging events but also on how these events interact with one another. For instance, a high dredging intensity on a small period of time and a low intensity over a longer period may have very different effects on benthic communities (73; 78). In order to try to take this into account, a third variable I_s is defined in addition to D_s and T_s. This variable is the mean monthly time of extraction during the extraction period :

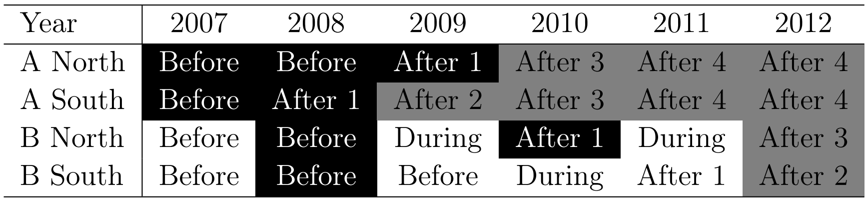
$$I_s = \frac{T_s}{D_s} \tag{3.4}$$

3.4.3 Quantifying the effects of extraction and recovery

The effects of extraction can be divided into two parts : (i) short- and medium-term effects that will cause densities to differ from their initial level and (ii) longer-term effects where densities tend to go back to their initial level. We divided the period after extraction into four periods (Table 3.2). The first period corresponds to a six-months period after extraction (After 1) during which peaks in abundances of opportunistic species tend to be observed (24). We first compare the Before and After1 periods in order to quantify the effects of aggregates extraction. Then all periods where a recovery can occur (After2 to After4) are compared to the After1 period to search for a significant recovery (Table 3.3).

Period	Before	During	After 1	After 2	After 3	After 4
Time	Before extraction	During extraction	$1 \leq t < 6$ months after extraction	$6 \leq t < 12$	$12 \leq t < 24$	$t \geq 24$

Table 3.2 – Periods defined to perform the analyses and corresponding number of months.



Year	2007	2008	2009	2010	2011	2012
A North	Before	Before	After 1	After 3	After 4	After 4
A South	Before	After 1	After 2	After 3	After 4	After 4
B North	Before	Before	During	After 1	During	After 3
B South	Before	Before	Before	During	After 1	After 2

Table 3.3 – Periods corresponding to each year for the four study sites. Periods used in both analyses are in black, additional periods used to quantify recovery are in gray and periods not used in any analysis are in white.

3.4.3.1 Effects of extraction

We concluded to an effect of extraction when a significant negative relationship was found between one of the three extraction variables (D_s, T_s and I_s) and the density of benthos, using simple linear mixed-effects models. The asset of linear mixed-effects model is that they allowed us to define sampling stations and years as sources of variability impacting the level of density but not the trend coming from the dredging intensity ((1|Station) and (1|Year) in the formula). This seems appropriate in our case as densities show high inter-stations and inter-years variability, even within a given site (not shown). This agrees with what is commonly observed in benthic populations, a patchy and variable distribution (160; 161) transposed into a zero-inflated dataset and thus a high variability.

As the three extraction variables are highly correlated ($\tau = 0.96$), we built three models, testing the effects of these variables individually. The three models are built for each studied guild G :

$$Y_{s,i}^{G} = \alpha T_{s,i} + b_s + c_i + \varepsilon_{s,i} \tag{3.5}$$

$$Y_{s,i}^{G} = \alpha D_{s,i} + b_s + c_i + \varepsilon_{s,i} \tag{3.6}$$

$$Y_{s,i}^{G} = \alpha I_{s,i} + b_s + c_i + \varepsilon_{s,i} \tag{3.7}$$

The studied variable $Y_{s,i}^{G}$ is the density of benthos on each station, with :

- s the station : $s \in \{1, \ldots, 18\}$,
- i the year : $i \in \{1, \ldots, 4\}$ i.e. $i \in \{2007, \ldots, 2010\}$,
- α the fixed linear effect coefficient,
- b_s and c_i random effects ($b_s \sim N(0, \sigma_{b_s}^2)$ and $c_i \sim N(0, \sigma_{c_i}^2)$),
- $\varepsilon_{s,i}$ the independent error term ($\varepsilon_{s,i} \sim N(0, \sigma_{\varepsilon_{s,i}}^2)$),
- $G \in \{1, \ldots, 14\}$ i.e. $G \in \{Attached, \ldots, SuspensionFeeders\}$

b_s and c_i being random effects means that we make the hypothesis that the spatial variability between site is captured by the variability between stations (homogeneous zone) and that we deem that the year effects are only a source of variance.

In these models the periods compared are that before extraction and a six months period after extraction, so that densities of benthos are not impacted by recovery (24). We compare situations where extraction equals zero (all years on the control site or years before extraction on the study sites) to situations when extraction has occurred (six-months period after extraction on all study sites). We assume that all random effects and residuals of the models built have normal probability density functions.

Mixed-effects models are built using the lmerTest package (162) in R (163) that provides p-values for the lmer function from the lme4 package (164).

3.4.3.2 Recovery

In the second stage, we use mixed-effects models to quantify recovery rates after dredging. The After1 period is used as a reference and compared to the three other periods after extraction to test for a trend in the evolution of benthos density. Densities after extraction are also compared to those before extraction to determine whether recovery allows reaching initial abundance levels at the end of the time series.

The corresponding model is :

$$Y_{s,i}^{G} = \beta_{p,s,i} + b_s + c_i + \varepsilon_{s,i} \tag{3.8}$$

$Y_{s,i}^{G}$ is the density of the studied benthic guild on each station, with :
- s the station : $s \in \{1, \ldots, 22\}$,
- i the year : $i \in lbrace 1, \ldots, 6\}$ i.e. $i \in \{2007, \ldots, 2012\}$,
- p the period : $p \in \{1, 3, 4, 5, 6\}$ i.e. $i \in \{before, After1, \ldots, After4\}$,
- β the fixed linear effect coefficient,
- b_s and c_i random effects ($b_s \sim N(0, \sigma_{b_s}^2)$ and $c_i \sim N(0, \sigma_{c_i}^2)$),
- $\varepsilon_{s,i}$ the independent error term ($\varepsilon_{s,i} \sim N(0, \sigma_{\varepsilon_{s,i}}^2)$),
- $G \in \{1, \ldots, 14\}$ i.e. $G \in \{Attached, \ldots, SuspensionFeeders\}$

Similarly to the previous models, we make the hypothesis that years are only a source of variability around the effect coming from the progressive recovery of benthic groups. We also make the hypothesis that recovery is independent of the amount of sediment extracted and did not include this variable in the model. The various periods are compared by means of t tests with Satterthwaite approximations for degees of freedom, with the hypothesis $H_0 : \beta_{p1} = \beta_{p2}$ and "treatment" contrasts (all periods are contrasted to the After1 period that has 0 value).

The hypothesis is that in the case of a recovery a trend in values of β obtained for the various periods will appear. This trend could be a long- or short-term one depending on when recovery really starts and when it ends. An example of trend might be : ($\beta_{A1} < \beta_{A2} < \beta_{A3} < \beta_{A4}$ or $\beta_{A1} > \beta_{A2} > \beta_{A3} > \beta_{A4}$ in the event of an opportunistic behaviour).

3.5 Results

3.5.1 Exploratory analysis

Densities shown in Table 3.4 and Table 3.5 correspond to those observed on the control site, averaged over the observed time series. Standard deviations indicate annual variability. Variability between stations is high for a given year (not shown) and the standard deviation often equals the mean density. The total average density per year also varies a lot, between 1000 and 4000 individuals per square meter for taxa with a trophic group. Densities for mobility groups are lower because a smaller number of taxa were classified in that guild.

Trophic guild	Number of taxa	Mean density ($.m^{-2}$)	Standard Error
Carnivores	134	647	325
Carrion Feeders	8	15	8
Herbivores	2	30	16
Detritus feeders	23	40	18
Suspension Feeders	86	767	640
Selective Deposit Feeders	61	308	205
Non Selective Deposit Feeders	21	323	190
Micrograzers	15	82	34

Table 3.4 – Number of taxa, mean density and standard error for each trophic group. Density is computed over the whole time series on the control site.

Mobility group	Number of taxa	Mean density ($.m^{-2}$)	Standard Error
Mobile and Burrower	76	106	59
Burrower	79	153	50
Mobile and Attached	4	1	1
Mobile and Burrower and Attached	9	16	9
Mobile	63	397	239
Attached	16	645	529

Table 3.5 – Number of taxa, mean density and standard error for each mobility group. Density is computed over the whole time series on the control site.

Analyses of the species contribution for each trophic guild on the extraction site before extraction occurs indicate that these guilds tend to be dominated by one species with high density (Table 3.6). This is particularly the case for suspension feeders, both guilds of deposit feeders and micrograzers. Species density is more even for other trophic guilds.

Species in the mobility guilds contribute more evenly to the total guild density (Table 3.7), except for the Attached guild, that is strongly dominated by *Pomatoceros triqueter*. Two mobility guilds appear to be dominated by species that also dominate trophic guilds (Table 3.7). In particular *Pomatoceros triqueter* is the species with highest density for both the Suspension feeder and the Attached guilds, so it is likely that these guilds behave in partly similar ways. Other guilds dominated by similar species are the Micrograzer and the Burrower guilds that are dominated by *Echinocyamus pusillus*. The Mobile and Carnivore guilds also very strongly overlap, with several high-density species in common.

Comparison of the sole and plaice diets suggest that plaice has a more diverse diet than sole. In total, we identify 122 taxa that can be eaten by plaice (105 for which trophic and mobility guilds are known) and 94 that can be eaten by sole (73 with trophic and mobility guilds). Diets of sole and plaice do not totally overlap : over the 105 guilds eaten by plaice, only 64 are also prey of sole and 41 are exclusive to plaice. This means that there are also nine taxa that can be eaten by sole and not by plaice. The main difference between diets of sole and plaice is that plaice eats more suspension feeders than sole. Indeed, 20 taxa belonging to this guild are exclusive to the diet of plaice while sole and plaice have 12 such taxa in common and only one is exclusive to the diet of sole.

When plotting densities of benthic groups on the control site on a graph along with dredging intensity, negative effects of dredging clearly stand out for some of the studied groups such as carnivores, burrowers, or non selective deposit feeders. From these graphs, it appears that the duration of the negative impact as well as the recovery rate differ greatly depending on the studied group.

Qualitative exploration of the data shows notable negative effects for at least two trophic groups. The first group is the one containing carnivorous species (Fig. 3.3). This group has a high density, up to 600 individuals per square meter. Before dredging occurs, density on the control site and on the impact site are similar. After dredging, density on the impact site is much lower than that of the control site, especially for sites A North and A South. It is much harder to observe an increase in density corresponding to a recovery to density levels corresponding to those of the control site. Usually, the most noticeable effects can be observed on site A North, therefore only graphs corresponding to this site will be shown to illustrate the analyses. However, analyses are performed on all sites. Results obtained are quite similar for non-selective deposit feeders (Fig. 3.4) eaten by plaice. This group also has high densities (up to 800 individuals per square meter) that are strongly negatively impacted by dredging. For these two groups negative effects can be observed for the two sampling surveys that follow the end of dredging : these groups are still strongly negatively impacted one year and a half after dredging. Even four years after the dredging stopped the densities remained low.

The mobility guild most impacted by dredging is the “Burrower” group. Trends observed on site A North are a lot like those observed for impacted trophic groups, with strong negative impacts that can still be observed one year and a half after dredging stopped (Fig. 3.5). However, effects of sediment extraction are much less obvious on other sites.

3.5.2 Quantifying the effects of extraction

3.5.2.1 Trophic guilds

Results obtained from models built tend to agree with those obtained in the exploratory analysis. In particular models show significant effects of extraction on carnivores and non selective deposit feeders, groups that already stood out in the exploratory analysis. No clear effects could be observed for selective deposit feeders in the exploratory analysis but models suggest that they are also impacted by aggregates extraction.

All estimated effects ($\hat{\alpha}$) are negative : aggregates extraction negatively impacts benthos density, at least shortly after the extraction period. In all cases, standard deviations are low enough not to impact the sign of these values (i.e. the direction of the variation, Table 3.8.

Results obtained for selective deposit feeders and carnivores eaten by sole and plaice are similar. Selective deposit feeders are the group with the most significant effects in this study. All three variables tested with the three models are significant for that guild, intensity being the least significant one. All three variables are equally significant for carnivores, whether eaten by sole or plaice.

There is a discrepancy between results obtained for sole and those obtained for plaice concerning non selective deposit feeders. Indeed, all three variables are significant for non selective deposit feeders eaten by plaice, but the monthly extraction intensity is not significant for those eaten by sole. Besides, variables tested are less significant for non selective deposit feeders by sole than for those eaten by plaice. Only micrograzers eaten by plaice are significantly impacted, and only by the monthly extraction intensity.

3.5.2.2 Mobility guilds

Two mobility groups appear to be significantly and consistently impacted by sediment dredging : “Mobile”, “Burrower”. Results obtained for sole are almost identical to those obtained for plaice (Table 3.9).

For both groups all three variables cause significant decreases in the density of benthos. I_s is the variable most significantly impacting the density of burrowers. On the contrary, I_s is the least impacting variable for the “Mobile” group. Therefore it can be inferred that interactions between T_s and D_s are less impacting for mobile benthos than for burrowers, which may come from a better capacity for mobile groups to colonize dredged areas once extraction stops.

3.5.3 Quantifying Recovery

Recovery is hard to evidence due to high spatial and temporal variability in benthos abundance. Only results that can be analysed are discussed here. The only group showing a significant recovery after extraction is that corresponding to detritus feeders eaten by plaice. Trends can be observed for several other groups but are not significant because of variability. The Mobile guild shows an interesting behaviour that may illustrate its opportunism and is also discussed here.

All periods after extraction appear to be significantly different from the reference period (After1, Table 3.10), that corresponds here to the six-months period following extraction. Only the model intercept is positive, and all other periods correspond to lower densities of detritus feeders. This may indicate an opportunistic behaviour of this group, whose density increases shortly after extraction and then decreases. Densities for periods After 2 to After 4 are rather similar, which indicates a quick recovery. However, comparison of the density levels of these periods indicates that they are significantly different from that of the Before period (not shown here). Therefore no recovery to levels close to the initial level of density can be observed for detritus feeders eaten by plaice, even more than two years after extraction.

The density of the Mobile guild peaks during the After2 period (Table 3.11) and then decreases to levels higher but not significantly different from that of the

After1 period. This may illustrate the fact that mobile groups are able to colonize an impacted area after dredging, but not as quickly as other opportunistic guilds. This can be due to higher reproduction rates or higher dispersal capabilities than those of other guilds, but that remains to be verified. This guild has very high densities (up to 1500 individuals per square meter), mostly due to a strong year effect, which makes results trickier to further interpret.

For suspension feeders and attached taxa the increase in density seems to last more than a year before the decrease occurs, but these trends trend are not significant.

3.6 Conclusion

In this paper we wanted to assess the effects of marine aggregate extraction on benthic communities eaten by sole and plaice, despite the high natural variability of their densities. The order of magnitude of the levels of density observed in our analysis as well as the high variability seem to agree with observations made by (165) in several bays of the Eastern English Channel.

We showed significant effects of extraction on species grouped by diet and mobility. In particular carnivores, selective and non selective deposit feeders appear to be particularly negatively impacted by sediment dredging. Burrowing and mobile species showed a high sensitivity to sediment dredging. Recovery appears much more difficult to identify than the impact of dredging itself and was significant only for detritus feeders. It should be noted that the only recovery where we managed to observe a significant trend is a negative one. This step suggests that detritus feeders are opportunistic, with densities that peak shortly after extraction stops and then decrease. Similar trends seemed observable for some other trophic groups but were not significant, which calls for further study. We could not identify any recovery for guilds that are negatively impacted by dredging, i.e. any positive trend after extraction where density would increase to get back to its initial level. This, in addition to strong yearly variability, could be the sign of slow recoveries for these guilds that could not be quantified at the time scales used in our analyses. Such recovery times were not expected at first, several papers indicating that recovery should be expected within two to three years in sandy gravel habitats (140; 145; 24). Our study is coherent with that of (141) which predicts prolonged recovery times, in particular for areas dredged repeatedly such as our study area.

Our analysis was performed on broad benthic guilds and is not very sensitive to the predator species studied. Therefore it could have been performed on a broader diet corresponding to those of both species studied, or only on the diet of one of them because sole and plaice have large parts of their diets in common.

Even if our exploratory analysis showed that some of the guilds defined were dominated by similar species, such guilds appeared to have different behaviours. This tends to indicate that the influence of a single, even highly abundant, species is limited at the guild level. The Mobile and Carnivore guilds have several high-density species in common at the beginning of the time series. Variables T_s and D_s are equally significant for these two guilds, but I_S is less significant for mobile guilds. So even when two guilds have large proportions of their total density in common they do not behave in totally similar ways. Such results may be dependent on how guilds are built. Indeed, we built guilds based on a limited number of sources and thus did not manage to allocate a guild to all the taxa found on the experimental site. Adding more taxa to the studied guilds is not likely to deeply change the results obtained but may alter them enough to change their level of significance or make significant results that were previously almost significant.

We demonstrated here that impacts of aggregates extraction on benthic communities are significant, even if they do not always seem to be very strong. These seemingly low impacts should nonetheless be interpreted carefully. Indeed, the decreases in density evidenced occurred despite the very high productivity of benthic species and can be seen as direct decreases of the carrying capacity of the studied area. Besides, even if the studied variables show strong correlations they are not identical and may interact, further increasing the rate of decrease in benthos density. Units of the variables studied also have to be taken into account and compared to real-life aggregates extraction efforts : even if the effects of a single hour of extraction may seem negligible, hundreds of hours may have dire consequences on the dredged area.

3.7 Discussion

3.7.1 Sampling procedure and methodology

We evidenced significant impacts of aggregates of extraction on benthos and recovery trends despite the rather small size of the dataset at hand. Small, patchy datasets containing potentially biased data are rather usual in ecology. This is mostly due to the high cost of collecting samples in the field, that limits the number of stations sampled, reduces the frequency at which samples are collected and shortens the duration of the studied time series. This is worsened by potentially bad weather conditions, variability in the methods and tools used to collect the samples and the fact that the sampling procedure is sometimes not elaborated with the statistical analyses in mind. Such issues may interact with characteristics inherent to benthic communities such as their patchiness and the high variability of their density to limit the power of the analyses that can be performed.

In our case, the studied time-series is rather short (2007 to 2012) and, depending on the site and the length of the dredging period, we often had very few stations to estimate the initial state or the recovery. Besides, relatively few stations were sampled in each site : three stations for both A sites and four stations per B site. The sampling procedure was also affected by weather conditions and currents. This means that the real position of the sampling stations changes between years. For some years, in particular 2010 and 2011 the position of the stations shifted so much that some stations outside a dredged area were sampled on a dredged site while some stations normally belonging to the dredged area were sampled outside. Benthic species displaying patchy distributions, changing the position of a station may cause the sampling of a different patch and deeply modify the characteristics of the station in the analysis. A last element to be taken into account is that not all stations studied are sampled every year, and that the sampling procedure changed during the sampling period to take into account an unforeseen dredging dynamic on site B.

Another aspect of the sampling procedure that causes it to greatly impact our results is that, the "catchability" of mobile species is much lower than that of burrowers or attached species when using a mini-Hamon grab. The mini-Hamon grab samples the substratum with a scoop that takes time to close, which favours the escapement of mobile benthos. Besides, when using a mini-Hamon grab the scooping action is dependent of the pulling action of the winch cable (166), which may create local perturbations and increase the escapement. This may explain the rarity of shrimps in the data whereas shrimps belonging to the Pandalina genus are dominant in the diet of sole in this area (167). This may also be one of the reasons why a recovery in the density of mobile species cannot be observed. This recovery may also be partly masked by the very strong increase in the density of crabs belonging to the Pisidia genus during the studied period whose density is much higher than that of any mobile group at the end of the time series. Another element that might prevent us from observing recoveries is that shifts may occur in the species composition of the guilds without major changes in their overall density. Indeed, not all taxa within a guild have identical characteristics and some may be more affected by dredging by others. Therefore some taxa may take advantage of the decrease in density of other taxa to spread to new areas or increase their density in a given area.

Facing this, we had to adapt our statistical analysis to get as much statistically sound information as possible out of the studied dataset. First, we chose to focus on simple models so that correlated variables were not tested together. This also helps us deal with the fact that the data we use is little contrasted, so only few parameters can effectively be tested in a single model. To increase the number of points studied in a model, we do not take variability between sites into account (potentially coming from their geographical position within the study area) but only variability between stations in the area and study all stations at once. Another point to be taken into account in these analysis is the fact that yearly changes in density are not taken into account as a "year" effect but only as a source of variability. This amounts to making the strong hypothesis that there is no trend in density along years during the studied period. We also tackled with variability caused by bad weather conditions by taking into account the real position of the sampling stations every year. Therefore, only stations that were effectively within the dredging area were taken into account in the analysis.

The analyses would be further improved by an improved sampling procedure with a better controlled dredging activity. Indeed, sampling only occurred once a year at the end of February. This period makes sense because benthos productivity is at its lowest, but one sample collection a year does not seem to be enough to make a proper analysis. From a statistical point of view it would have been better to collect samples at least twice a year, on randomly chosen months (similar months for all stations) ; but this kind of approach is difficult to implement in reality. Such a sampling procedure would have been all the more interesting since the aggregates extraction activity is highly variable and leads to little contrasted situations. If such an experiment is to be made again it is our opinion that the main goal should be to apply constant and contrasted (in terms of total duration as well as monthly intensity) aggregates extraction efforts. Such a pattern would allow us to compare long/short periods with high/low efforts and make sure that several months stops in the extraction activity do not appear in the studied time series. The studied time series should also be adapted so that recovery can be studied for long enough on all studied sites. The aim should also be to have a balanced design, which could be achieved with the increase in the sampling effort. This increase could also occur in space, with a higher number of sampled stations. An increased density of sampling stations may be the key to deal with benthos small-scale variability and heterogeneity and help reduce inter-stations variability in the analyses.

3.7.2 Perspectives

In this article we focus on guild density but not changes in communities or guild structure and thus make the hypothesis that structural changes are less important than overall changes in abundance. This hypothesis seems to agree with trends observed in the exploratory analysis where it can be observed that guilds tend to be dominated by similar species in the Before and After1 periods (Tables 3.6 and 3.7), but with lower abundances. However, our analysis may not have been as valid had the Before and During periods been compared. Indeed, changes in the guild structure can be observed for the During period, previously dominant species being replaced by species that were absent or had low abundances. It would be interesting to further study this short-term structural shift to determine if it is statistically significant or just an artefact in the exploratory analysis. If this event appears to be significant it would indicate that recovery occurs in several stages : first a quick structural recovery where guild structure reverts to its initial structure and then an longer recovery where abundances tend to increase to their initial level. This two-stage recovery has already been observed on other study sites (145; 143; 147). Analyses of such processes may help us better understand consequences of anthropogenic activities on benthic guilds and how to limit these impacts.

We chose to group benthic taxa into guilds and studied trophic and mobility guilds. Our analyses could be performed on other types of guilds, for instance guilds better taking into account life cycles and characteristics of the reproduction of benthic taxa. Such guilds may allow us to better underline and take into account opportunistic behaviours and competition between guilds. These aspects may help further understand the recovery process. Guilds could also be built based on other benthos characteristics such as the amount of energy a given taxa represents for fish species. Results obtained in this article are to be used in a complex model of the Eastern English Channel to try to better understand consequences of aggregates extraction on benthic communities in this area. If a similar study was performed on guilds built based on an amount of available energy, it may become possible to understand and model how anthropogenic activities indirectly impact the trophic network through benthic species.

The next step of this analysis is to determine how marine aggregate extraction impacts populations of sole and plaice through benthic populations and their trophic network. This could be done by means of estimates of the proportion of each group in the diet of these species. More complex indices could also be used. In particular, studies have focused on estimating the amount of energy corresponding to several benthic species. If such indices could be computed at the level of the trophic or mobility group then we could compute the loss of energy available to fish caused by sediment dredging.

However, it would still be difficult to extrapolate impacts of aggregates extraction on fish from impacts on benthos observed by means of sampling performed with a grab. Indeed, a fish and a grab are two totally different types of samplers. Grabs are not mobile and passive and only sample small areas of the substratum, with possible escapements of some benthic groups. Conversely fish are active predators that can forage on larger areas and select their feeding area and their prey. These phenomena may be lower for benthic species like sole and plaice that graze at the surface of the substratum, do not rely on sight to feed and tend to cover smaller areas than demersal or pelagic fish species. Fish can also, to some extent, adapt to changes in prey availability. Thus, even if benthic communities change sole should be less impacted than plaice because of its better ability to adapt. We tried to define diets as wide as possible for both species so as to take adaptation into account, but it is still possible that these species may be able to feed on benthic groups not taken into account in the analysis. Therefore in addition to benthic surveys fish surveys also seem to be very important as they allow to determine whether fish diet changed, but also whether a change in the fish diet caused changes in the condition of the fish. For instance fish size or mean weight at some periods of the year may change and results in lower reproduction rates or higher mortality rates.

Benthic taxa are usually taken into account in trophic models, building in predator/prey relationships within ecosystems. Some of these models (168) focus on the biological aspects of the ecosystem, without including the humans and their activities. In other approaches, anthropogenic activities (e.g. fishing) are modelled as another top-predator that impacts part of the foodweb (169) and can increase recycling (e.g. by discarding part of the catch). But these approaches usually focus on the direct impacts of fishing on its target species (and sometimes on bycatch) and do not take technical interactions between species into account. In such models, benthic groups would only be impacted through the bottom up control of high trophic level species on their prey.

One way to get insights in the effects of human activities on fish species through benthic guilds would be to couple a trophic model to complex spatially explicit model representing human activities. The latter can be built with the ISIS-Fish modelling platform (28; 14; 29) that allows explicitly modelling interactions between several fleets and various target and non-target species in a given area. An ISIS-Fish model of the Eastern English Channel has already been developed (170) in order to search for management measures that would allow robustly reaching management goals. It focuses mainly on two flatfish species : sole and plaice. Sole is the target of several fishing activities in this area because its high value (37) and plaice is both a target and a bycatch species. Results obtained in the present article will be used to represent benthic species in addition to fish species in this model, in order to model consequences of fishing and extraction activities on benthic guilds. In the event of the coupling with a trophic model, the addition of this benthic compartment could greatly modify model results concerning the robustness of management measures tested so far.

Guild	Taxa	Density ($inds.m^{-2}$) Before	During	After1	After2	After3	After4
Carnivores	Protodorvillea kefersteini	121.9	61.8	84.1	166.5	127.4	101.7
	Pholoe inornata	111.2	6.8	55.6	39.5	30	29
	Amphipholis squamata	99.5	19.2	29.3	48.3	8	44.6
	Sphaerosyllis bulbosa	72.7	3.2	18.3	0.5	1.3	0.3
	Odontosyllis gibba	26.1	0	11.3	17.8	19.2	11.2
	Phyllodoce mucosa	0.4	46.5	0	90.2	5.2	0.3
	Syllidia armata	21.7	30.8	21.6	49.8	27.7	22.7
	Glycinde nordmanni	17.2	13.2	15.7	21.7	23.6	3.9
	Eumida sp	0.2	9.8	0	23.3	0	23.1
Non Selective Deposit Feeders	Lumbrineris gracilis	215.8	116.8	130	116	76.6	171.5
	Antalis entalis	3.4	0	0	0	0	0
	Scalibregma inflatum	2.6	0	1	1.7	0	3.9
	Heteromastus filiformis	2.2	0	0	0	0	0
	Pectinaria koreni	1.4	0	0.6	0	0.3	0
	Notomastus latericeus	1.1	3.2	0.6	2.7	4.9	6.4
	Orbinia cuvieri	0.2	2.5	0.3	3.3	0	9.3
	Scoloplos armiger	0.9	1.8	0.3	0.5	0.7	2.2
	Capitellides giardi	0	0	16	0	0	0
	Lumbrineris latreilli	0	0	6	0	0	0
	Capitella capitata	0.4	0	2	0.5	0.3	0
	Mediomastus fragilis	0	0	0	17.7	0.3	15.7
	Lagis koreni	0	0	0	3.8	0	0.3
	Echinocardium cordatum	0.5	0	0.6	0	0.7	0
	Lumbrineris sp	0	0	0	0	0	7.3
Selective Deposit Feeders	Caulleriella alata	259	94.2	128.7	235.7	105	50.5
	Caulleriella bioculata	29.8	0.8	17.3	36.7	5.4	1.3
	Antalis vulgaris	25.3	6	11.3	8.5	3.5	7.4
	Caulleriella sp	22	0		0		0
	Pista cristata	9.5	10	15.5	24	14.6	44
	Polycirrus sp	2.2	15.5	7.6	15	16.3	14.6
	Polydora ciliata	0	5	0	0	0.3	0
	Euclymene oerstedii	0	1.8	8	17.8	3.9	8.7
	Abra alba	0	0	0	17.7	2.6	1
	Macrochaeta clavicornis	0	0	4.1	0	5.9	2.3
	Flabelligera affinis	2.5	0	5.9	3.3	5.7	3.9
Detritus Feeders	Maera othonis	19.1	0	19	2.2	2	0
	Atylus vedlomensis	7.4	2.5	6.7	5.5	3.6	3.3
	Anapagurus hyndmanni	1.9	0	1.3	6.7	0.3	1.3
	Photis longicaudata	1.9	1.8	7.4	28.3	4.4	12.8
	Amphilochus neopolitanus	1.3	0	0	0.5	0.7	0.6
	Abludomelita obtusata	0	3.2	5.3	3.3	1.9	0
	Abludomelita gladiosa	0	0	0	0.5	0	0
	Megamphopus cornutus	0.4	0	1.9	0	0.6	2.3
	Gammaropsis sp	0	0	1	9.3		
	Cheirocratus intermedius	0	0	1.7	5.5	3.7	0
	Gammaropsis maculata	0	0	0.9	0.5	11.3	1.7
	Cheirocratus sp	0	0	0	1.2	2.3	0
	Athanas nitescens	0	0	0	2.2	1.3	2
Herbivores	Psammechinus miliaris	33.1	1.8	18.9	15.5	5	6.2
	Leptosynapta inhaerens	18	2.5	8.1	7.8	4.3	6
Micrograzers	Echinocyamus pusillus	42.4	6.8	16	10.5	0.9	0.9
	Leptochiton asellus	3.5	6.8	12	13.8	3.6	46.2
	Lepidochitona cinerea	1.4	0	2	0	0	0
	Gibbula tumida	0.8	0	1.3	1.7	0.6	0.3
	Leptochiton cancellatus	0.4	0	1.7	0	0	0
	Calliostoma zizyphinum	0	0	0	0	0.3	0
Carrion Feeders	Tryphosella sarsi	5.7	0	2.6	4.3	0	0
	Pagurus berhnardus	3.8	0	0	0	0	0
	Tryphosella nanoides	2.4	0	0	0	0	0
	Pagurus cuanensis	1.6	0.8	0.3	0.5	0	1
	Orchomenella sp	0.6	0	1.6	0	0	0
	Buccinum undatum	0	0	0	0.5	0	0
	Orchomenella nana	0	0	3	2.2	2.3	4.3
Suspension Feeders	Pomatoceros triqueter	397.4	0	298.6	759.8	304.6	0
	Ampelisca typica	7.4	7.5	4.7	15	6	4.6
	Galathea intermedia	5	4.2	22.1	14.2	16.7	10.6
	Ampelisca spnipes	4.7	4.2	12.2	1.7	5.3	3.7
	Corophium sextonae	3.8	0	1	3.3	0.3	0
	Pisidia longicornis	1.5	30.8	9.6	488.8	183.4	139.3
	Ericthonius punctatus	0	3.5	5.4	17.7	61	2.7
	Jasmineira elegans	0	0	0.7	8.8	10.3	1
	Moerella donacina	0	0	0	2.2	1.3	5.1

TABLE 3.6 – Most important taxa of each trophic guild for each studied period, averaged over all stations belonging to a dredged site. The five most important taxa of each period are in black.

Guild	Taxa	Density ($inds.m^{-2}$) Before	During	After1	After2	After3	After4
Attached	Pomatoceros triqueter	397.4	0	298.6	759.8	304.6	0
	Pomatoceros lamarcki	75.4	0	93.5	117.7	156	0
	Pomatoceros sp	16.8	0		48.3	77.3	0
	Serpula vermicularis	0.6	0	1	1.7	2.2	1
	Ascidia sp	0	0	0.3	0	0	0
	Balanus crenatus	0	899.2	0	21.7	7	0.3
	Ascidiacea sp	0	1.5	1.7	6.7	14.4	36.6
	Mytilus edulis	0	0.8	0	2.2	0	0
	Ascidia obliqua	0	0	0	0	0	0
Burrower	Echinocyamus pusillus	42.4	6.8	16	10.5	0.9	0.9
	Leptosynapta inhaerens	18	2.5	8.1	7.8	4.3	6
	Ampharete finmarchica	16.4	9	21.4	31.2	8.9	23.3
	Tellina donacina	14.8	1.8	6.6	5.5	4	2
	Cerianthus lloydii	14.2	2.5	9.6	6.5	0	6.6
	Polydora ciliata	0	5	0	0	0.3	0
	Aonides oxycephala	0.9	4.8	0.6	2.2	3.6	4.7
	Owenia fusiformis	0.2	3.2	0.3	1.7	0.3	3.6
	Abra alba	0	0	0	17.7	2.6	1
	Mediomastus fragilis	0	0	0	17.7	0.3	15.7
	Parvicardium minimum	0.4	0	0.6	10	0	0.3
	Scalibregma celticum	0	1.5	2.9	6	5.3	9.5
	Cheirocratus intermedius	0	0	1.7	5.5	3.7	0
	Orbinia cuvieri	0.2	2.5	0.3	3.3		9.3
Mobile	Pholoe inornata	111.2	6.8	55.6	39.5	30	29
	Amphipholis squamata	99.5	19.2	29.3	48.3	8	44.6
	Sphaerosyllis bulbosa	72.7	3.2	18.3	0.5	1.3	0.3
	Psammechinus miliaris	33.1	1.8	18.9	15.5	5	6.2
	Lepidonotus squamatus	13.5	0	13.6	17.2	6.7	12.9
	Pisidia longicornis	1.5	30.8	9.6	488.8	183.4	139.3
	Pholoe baltica	0	20	17	34.5	15	26.7
	Eumida sp	0.2	9.8	0	23.3	0	23.1
	Leptochiton asellus	3.5	6.8	12	13.8	3.6	46.2
	Sphaerosyllis hystrix	5.9	5.8	7	28.8	30	12.6
	Gammaropsis maculata	0	0	0.9	0.5	11.3	1.7
Mobile and Burrower	Odontosyllis gibba	26.1	0	11.3	17.8	19.2	11.2
	Glycera capitata	22.9	0	4.8	0	0	0
	Kurtiella bidentata	19.8	5.8	6.6	1.7	3.6	2.2
	Eulalia bilineata	19.4	0	0.7	0	0	0
	Maera othonis	19.1	0	19	2.2	2	0
	Urothoe elegans	0	7.5	0.6	19.8	16.7	0.6
	Eurydice pulchra	0.7	5.8	2.6	2.3	3	1.4
	Nucula hanleyi	2.2	4.2	1.4	1.5	1.7	4.6
	Glycera lapidum	1.4	4	1.3	13.3	2.5	6.6
	Photis longicaudata	1.9	1.8	7.4	28.3	4.4	12.8
	Abludomelita obtusata	0	3.2	5.3	3.3	1.9	0
	Lanice conchilega	0	0.8	0.7	6	4.7	0.3
	Nephtys kersivalensis	7.7	1.5	3.7	4.3	3.6	2.3
Mobile and Burrower and Attached	Ampelisca typica	7.4	7.5	4.7	15	6	4.6
	Ampelisca spnipes	4.7	4.2	12.2	1.7	5.3	3.7
	Eusyllis blomstrandi	3.4	0	0	1.2	4	2.2
	Eusyllis assimilis	0.7	0	0	0	0	0
	Ampelisca tenuicornis	0.4	0	0.3	0	0.3	0
	Eusyllis lamelligera	0	1.8	0	0.5	1.3	1.3
	Ampelisca brevicornis	0	0	0.3	0	0.6	0
	Ampelisca sp	0	0	0.3	0.5	0	0.3

Table 3.7 – Most important taxa of each mobility guild for each studied period, averaged over all stations belonging to a dredged site. The five most important taxa of each period are in black.

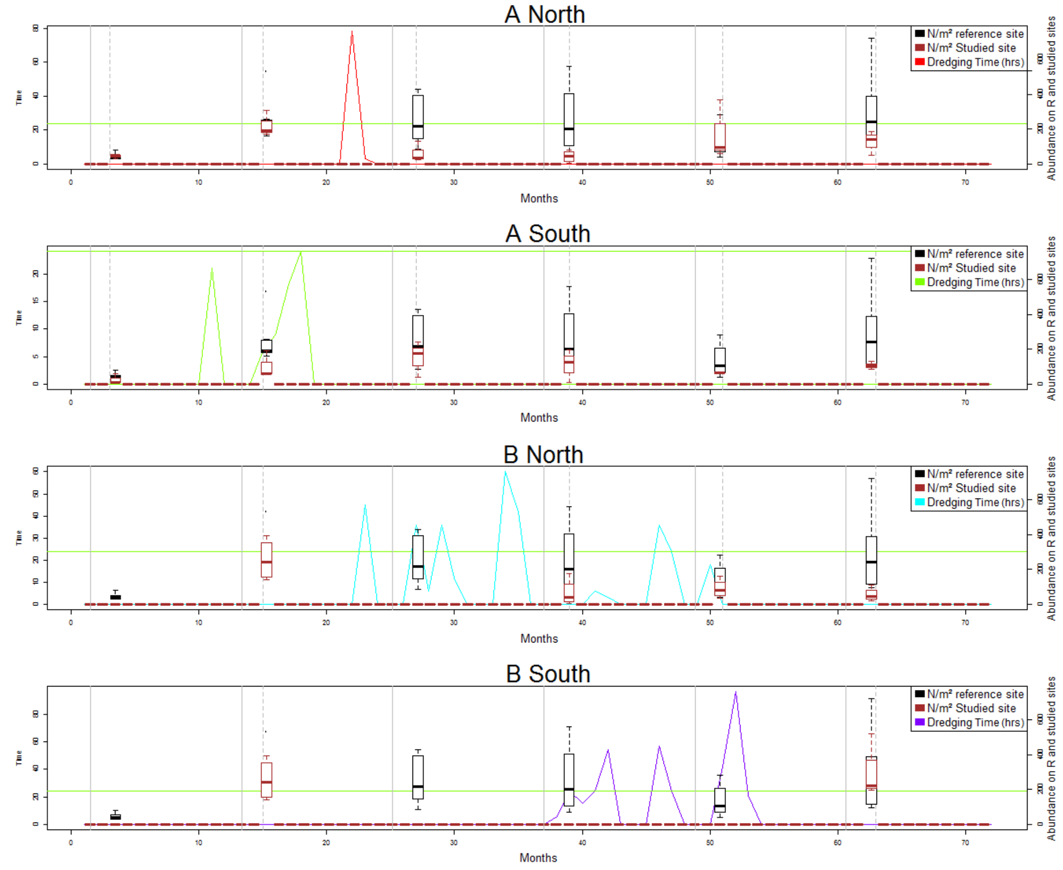


Figure 3.3 – **Density of carnivores eaten by sole on the four studied sites, compared to that on the control site.** The curve in the back of the graph indicates the number of hours that were dredged for each month on the studied site. The horizontal line indicates the maximum extraction time on the less impacted site (A South) so that sites can be compared more easily.

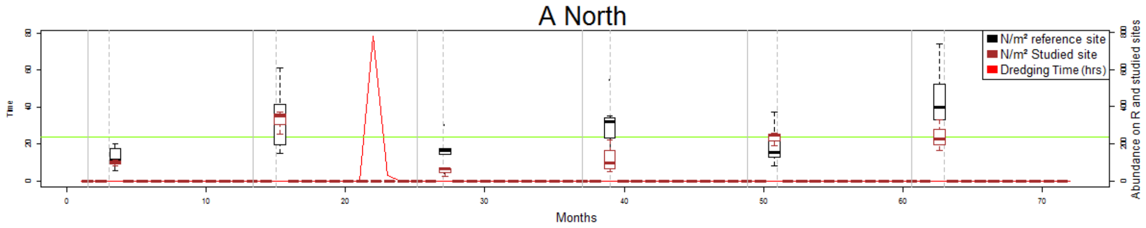


FIGURE 3.4 – **Density of non selective deposit feeders eaten by plaice on site A North for plaice.** The curve in the back of the graph indicates the number of hours that were dredged for each month on the studied site. The horizontal line indicates the maximum extraction time on the less impacted site (A South) so that sites can be compared more easily.

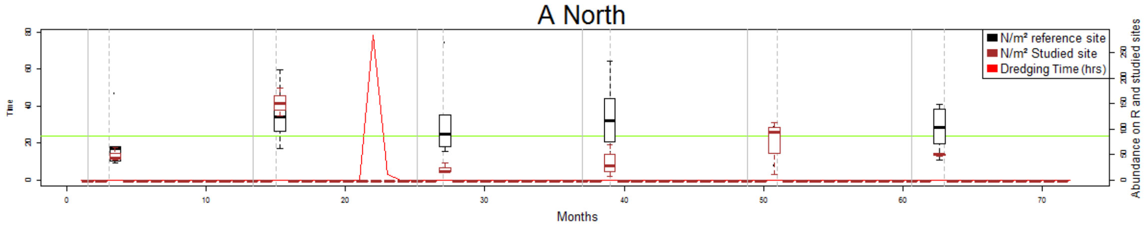


FIGURE 3.5 – **Density of burrowers eaten by sole on site A North.** The curve in the back of the graph indicates the number of hours that were dredged for each month on the studied site. The horizontal line indicates the maximum extraction time on the less impacted site (A South) so that sites can be compared more easily.

Predator	Trophic Group	Variable	$\hat{\alpha}$	% of mean N	Standard Error	p-value	Signif.
Sole	Selective Deposit Feeders	T_s	-0.23	-0.07%	0.06	0.0002	***
		I_s	-1.19	-0.39%	0.39	0.0034	**
		D_s	-3.54	-1.15%	0.93	0.0004	***
	Carnivores	T_s	-0.94	-0.15%	0.29	0.0024	**
		I_s	-5.81	-0.90%	1.83	0.0027	**
		D_s	-15.84	-2.45%	4.70	0.0016	**
	Non Selective Deposit Feeders	T_s	-0.05	-0.02%	0.02	0.0109	*
		D_s	-0.73	-0.23%	0.31	0.0221	*
Plaice	Selective Deposit Feeders	T_s	-0.41	-0.13%	0.09	5e-05	***
		I_s	-1.96	-0.64%	0.63	0.0034	**
		D_s	-5.96	-1.94%	1.51	0.0003	***
	Carnivores	T_s	-0.92	-0.14%	0.29	0.0025	**
		I_s	-5.80	-0.90%	1.81	0.0025	**
		D_s	-15.26	-2.36%	4.70	0.0022	**
	Non Selective Deposit Feeders	T_s	-0.77	-0.24%	0.26	0.0046	**
		I_s	-4.23	-1.31%	1.70	0.0162	*
		D_s	-11.37	-3.52%	4.21	0.0099	**
	Micrograzers	I_s	-0.76	-0.93%	0.36	0.0407	*

Table 3.8 – Effects of aggregate extraction on trophic groups eaten by sole and plaice. Signif. codes : “***” $< 0.001 <$ “**” $< 0.01 <$ “*” < 0.05.

Predator	Motility Group	Variable	$\hat{\alpha}$	% of mean N	Standard Error	p-value	Signif.
Sole	Burrower	T_s	-0.29	-0.19%	0.13	0.0344	*
		I_s	-2.42	-1.58%	0.82	0.0048	**
		D_s	-4.57	-2.99%	2.22	0.0449	*
	Mobile	T_s	-0.86	-0.22%	0.31	0.0089	**
		I_s	-4.41	-1.11%	2.04	0.0354	*
		D_s	-14.16	-3.57%	5.13	0.0082	**
Plaice	Burrower	T_s	-0.34	-0.22%	0.15	0.0275	*
		I_s	-2.71	-1.76%	0.90	0.0045	**
		D_s	-5.18	-3.39%	2.44	0.0391	*
	Mobile	T_s	-0.83	-0.21%	0.31	0.0094	**
		I_s	-4.42	-1.11%	2.00	0.0317	*
		D_s	-13.79	-3.47%	5.04	0.0088	**

Table 3.9 – Effects of aggregate extraction on mobility groups eaten by sole and plaice. Signif. codes : “***” $< 0.001 <$ “**” $< 0.01 <$ “*” < 0.05.

Period	$\hat{\beta}$	Standard Error	p-value	Signif.
(Intercept)	16.62	4.81	0.0042	**
After 2 (6-12)	-14.86	5.82	0.0139	*
After 3 (12-24)	-10.64	4.84	0.034	*
After 4 ($>$ 24)	-15.00	6.18	0.0212	*

Table 3.10 – Results given by the mixed-effects model for detritus feeders eaten by plaice. Signif. codes : “***” $< 0.001 <$ “**” $< 0.01 <$ “*” < 0.05.

Period	$\hat{\beta}$	Standard Error	p-value	Signif.
(Intercept)	131.42	120.97	0.2926	
After 2 (6-12)	428.18	171.35	0.0169	*
After 3 (12-24)	45.90	139.56	0.7451	
After 4 ($>$ 24)	33.95	163.41	0.8384	

TABLE 3.11 – Results given by the mixed-effects model for mobile species by plaice. Signif. codes : "***" $< 0.001 <$ "**" $< 0.01 <$ "*" < 0.05.

Chapitre 4

Prise en compte des incertitudes dans un modèle complexe d'écosystème marin

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Sommaire

Supporting fisheries management by means of complex models : can we point out isles of robustness in a sea of uncertainty ?

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4.1 Abstract

Ecosystems are usually complex, nonlinear and strongly influenced by poorly known environmental variables. Among these systems, marine ecosystems have high uncertainties : marine populations in general are known to exhibit large levels of natural variability and the intensity of fishing efforts can change rapidly. These uncertainties are a source of risks that threaten the sustainability of both fish populations and fishing fleets targeting them. Appropriate management measures have to be found in order to reduce these risks and decrease sensitivity to uncertainties.

Methods have been developed within decision theory that aim at allowing decision making under severe uncertainty. One of these methods is the information-gap decision theory. The info-gap method has started to permeate ecological modelling, with recent applications to conservation. However, these practical applications have so far been restricted to simple models with analytical solutions. Here we implement a deterministic approach based on decision theory in a complex model of the Eastern English Channel. Using the ISIS-Fish modelling platform, we model populations of sole and plaice in this area. We test a wide range of values for ecosystem, fleet and management parameters. From these simulations, we identify management rules controlling fish harvesting that allow reaching management goals recommended by ICES (International Council for the Exploration of the Sea) working groups while providing the highest robustness to uncertainties on ecosystem parameters.

4.2 Introduction

The Eastern Channel is a very important area because of its high biodiversity and the many fisheries it sustains (171), the most emblematic being the flatfish fishery. Sole (Solea solea) is one of the most economically valuable flatfish species in this area (37). Fishing mortality applied to sole being high (172), there are risks that the sole population may be over-harvested. This may have critical consequences for the sole population, bycatch species and the economic viability of fishing vessels. The level of risk is highly dependent on the level of fishing effort, but also on our level of knowledge on environmental and biological parameters. For instance if natural mortality (a parameter that is usually very difficult to determine) is higher than what is commonly deemed to be the correct value, then fishing mortality is overestimated and fishing regulations will not have the expected impact.

The ICES (International Council for the Exploration of the Sea) stock assessment working groups have traditionally dealt with uncertainties by means of a precautionary approach. When possible a limit spawning biomass and/or fishing mortality are defined, beyond which the risk of recruitment impairment is high (86). In addition, more conservative reference points have also been defined, based on a precautionary approach (PA points). These PA points aim to prevent reaching the critical limit, despite uncertainty in the ecosystem state or in the fishing effort (86). Total Allowable Catches (TACs) should be adjusted yearly depending on the estimated state of the stock so that these limit reference points are not reached.

Even if flatfish populations in this area have been studied in depth, many uncertainties remain concerning their biology, their dynamics, or the fishing pressure they are subject to. Therefore, current management cannot guarantee that management goals will be reached. Indeed, the method currently used by ICES working groups does not explicitly take uncertainties into account and is only based on past ecosystem states, not anticipating situations that have not been observed yet. One possible way to circumvent these limitations is to model the ecosystem of interest and our uncertainties on ecosystem parameters, then to test the performance of management measures. If management measures can be found that always allow reaching management goals, then such management measures can be considered robust to uncertainties. In this paper, we propose to determine whether simple changes made to current management measures can allow reaching management goals with a higher robustness to uncertainties.

Uncertainties and risks have been increasingly taken into account in fisheries management since the beginning of the 1990s (102; 112) and now pervade modern fisheries management (173). Many methods dealing with risks in fisheries management have been developed and have been reviewed in (107) and (173). Most quantitative methods are based on a Bayesian approach, as advocated in (15; 174; 175; 176). This probabilistic approach is based on the use of available a priori information on the shape of probability density functions that are attributed to model parameters.

However, using probability density functions is possible only if enough knowledge is available to estimate them precisely ; otherwise it only amounts to adding more uncertainties to the model. This is especially the case with complex ecosystem models where sources of uncertainty (177; 112; 178; 115) impacting model outputs are even more difficult to identify, as the number of modelled processes increases. In addition, ecosystems are usually complex, nonlinear and strongly influenced by poorly known ecological variables (179; 180). Among these systems, marine ecosystems have enormous biological uncertainty (15) and exploited populations in general are known to exhibit large levels of natural variability (108). Therefore, in some cases it may not be possible or desirable to give probability density functions to model parameter values when dealing with such ecosystems.

Choosing to not define probability density functions makes the use of methods commonly used to deal with risk less tractable. However, methods have been developed within Decision Theory that aim at allowing decision making under severe uncertainty. One of these methods is the information-gap decision theory (118; 119). The info-gap method has started to permeate ecological modelling, with recent applications to conservation (117; 181; 116). However, these practical applications have so far been restricted to rather simple models with analytical solutions and many limitations of this approach have been evidenced (182; 126).

To determine whether management measures can allow robustly reaching management goals for sole and plaice (Pleuronectes platessa) in the Eastern Channel, we build an ISIS-Fish (183; 184; 28; 29) model of ICES area 7D. This spatialized fisheries dynamics model allows us to represent both fish populations and fleets targeting them, and model management scenarios. From this model, we apply a method based on decision theory so as to find out if management goals on these species can be reached despite uncertainties. First, the input parameters space of our model is explored by means of sensitivity analysis techniques. This allows us to identify and rank parameters that most influence model outputs and whose uncertainty should be tested against management measures in priority. Once enough model runs have been performed, we split combinations of model parameters between those giving output variables equal to or above our management goals and those that do not allow reaching management goals. So as to identify combinations of management parameters and ecological parameters needed to reach management goals, supervised classification is performed by means of classification trees on the dataset obtained with sensitivity analysis. This classification allows us to identify management parameter values that are most desirable so as to reach management goals, and what level of uncertainty on environmental parameters can be tolerated without compromising the achievement of management objectives.

4.3 Material

4.3.1 ISIS-Fish

ISIS-Fish was designed to simulate and evaluate policies in the context of mixed fisheries (multi-species multi-fleet fisheries) and to take into account the spatial and seasonal heterogeneities in the distribution of resources and fishing activities (28).

This fishery model is based on three submodels : (i) a fishing activity dynamics model, (ii) a population dynamics model and (iii) a management dynamics model. Each submodel is spatially and seasonally explicit, with a monthly time step. The three submodels interact only if they overlap in space and time. The modelled area is represented by a grid, the resolution of which, in latitude and longitude, is chosen with respect to the dynamics being described and the available knowledge of the studied fishery. Within this region, zones (i.e. sets of contiguous grid cells) are defined independently for each population, each fishing activity, each management measure. Seasons are defined as sets of successive months. It is also possible to take into account fish price as well as fixed and variable costs in ISIS-Fish (184) to better model fishers behaviour. In our model, fish price is the only economic variable needed to determine the choice of fishing areas.

4.3.2 The Eastern Channel

Twenty-six exploited species can be found in the Eastern Channel, but also feeding, spawning and nursery grounds, as well as migration routes. Most catches come from the French and the English fleets, the English fishing activity having decreased a lot in the past decades with only a few ports maintaining a fishing fleet on the South-Eastern coast of England. On the contrary, the French fleet in this area still comprised 641 ships in 2005 that landed more than 90000 tons of fish, worth 218M euros (185). The harbour of Boulogne-sur-Mer is the biggest fishing harbour in this area (ICES area 7D) with 171 active fishing vessels in 2009 (186). The majority of landings are demersal species, especially common sole, scallops and whiting. Plaice is an important bycatch of fishing vessels targeting sole and is also directly targeted by fleets from the Netherlands and Belgium. Sole and plaice are mostly caught by beam-trawlers and netters.

Populations of sole and plaice are managed by means of TACs which build to some extent on catch limits recommended by ICES. Until the end of 2010 these catch limits aimed to keep the fishing mortality below precaution fishing mortality ($F_{pa} = 0.4$ for sole and $F_{pa} = 0.45$ for plaice). In 2011 a transition framework to maximum sustainable yield (MSY) was implemented. This transition framework is based on a harvest control rule (HCR) spanning a 5-year period. The goal is to reduce the fishing mortality from current levels to the fishing mortality providing the maximum sustainable yield (F_{MSY}). So as to reach F_{MSY} by the end of the HCR, the level of fishing mortality tolerated (and therefore the associated catch limit) is progressively decreased from F_{2010} to F_{MSY}. Year-to-year variation in catch limits is bounded to 15% (86).

4.3.3 The Eastern Channel model

This study is based on an ISIS-Fish model of the English Channel by (22). This model is deterministic : a given set of parameter values always gives the same values for the output variables. We performed 10-year simulations so as to model a

2008-2017 period that encompasses the 2010-2015 period of the ICES transition framework to MSY. This allows us to force some input parameter values to their estimated value for the first three years modelled (2008-2011) and then test transition scenarios and determine their consequences on the ecosystem.

4.3.3.1 Exploitation

This model focuses on the French flatfish fishery. Only French gillnetters and English or Belgian beam trawlers are explicitly taken into account and the modelling assumption is made that the travelling time from their home harbour to fishing grounds is negligible. It is considered that they fish all year long in ICES areas 7D and 7E and fleet (i.e. group of boats with same characteristics belonging to the same harbour) parameterisation is the same for both areas. They target sole (Solea solea) and their main bycatch is plaice (Pleuronectes platessa). An important notion when defining fishing activities in ISIS-Fish is that of métiers. For a given fishing vessel and a given month of the year, the métier practised by the vessel is defined by the gear used, the target species and the fished area (187; 188). Some fishing units have the same métier all year long, others change of métier (i.e. change of fishing area and/or gear and/or target species) depending on the season. A succession of métiers in a year defines a fishing strategy. Métiers, as well as entire fishing strategies can be common to groups of vessels. Fishing units with the same fishing trip duration, the same number of trips per month and belonging to the same harbour belong to the same fleet (28). Notions of fleet and métier are not totally correlated ; a given métier can be common to ships belonging to various fleets. Métiers, fishing strategies as well as fleets are defined based on real data (e.g. commercial logbook data, fishers interviews, observer data, etc.), the level of detail depending on the available information and the modeller's needs (29). In our model métiers parameterisation differs between ICES area 7D and ICES area 7E, but in general trawlers target both species with almost the same intensity while netters clearly focus on sole. These two species are the only ones explicitly represented in the model, other species caught by this fishery being grouped together in a single group. Our goal being to study flatfish fisheries, we are mostly interested in the Eastern Channel (ICES area 7D). However, fishing activities and fish stocks in the Eastern and Western part of the Channel being linked, we chose to model both areas and emphasize results obtained on the Eastern Channel. In our model, fishers select their métier dynamically by means of a gravity model. The attractivity of each choice is estimated yearly from fishing habits and past outcomes of the fishery (184). Once the catch limit has been reached for a species a set of conditions are applied in an attempt to realistically model fishers' behaviour : for a given métier i) if the species only is a bycatch species then fishing goes on and this species is discarded, ii) if the species is the target species then the métier stops and fishers look for another métier for the remaining months of the year. The choice of an alternative métier depends on the ease of implementation of the métier : métiers within the same strategy (i.e. monthly choice of métiers for a fleet during the year) using the same gear are preferred to métiers where a change in fishing gear is needed or métiers outside the strategy (that can correspond to no fishing activity). Discarded fish have a chance to survive that is species-dependent but age-independent and discarded fish that survive are returned to the abundance of their year class.

4.3.3.2 Populations

Both species are assumed to be distributed homogeneously over the whole modelled area. Each species is split in two populations, one for area 7D and one for area 7E. Biological parameters, and in particular weight-at-age, maturity, initial fish abundance, correspond to those estimated for year 2008 by the ICES Working Group on the Assessment of Demersal Stocks in the North Sea and Skagerrak (WGNSSK) (189). Fish catchability was calibrated so that, for each population, fishing mortality at age for year 2008 in our ISIS-Fish model corresponds to that estimated by the working group for year 2008 in the 2011 stock evaluation. In our model, the spawning biomass of sole in the Channel seems to be within acceptable biological limits, but with a high fishing mortality. No reliable stock recruitment relationships could be fitted to these stocks. We used spawning biomass precautionary thresholds (as suggested by (178)) that were defined by the ICES. We forced reference values of recruitment for years 2008, 2009 and 2010 (Table 4.1) to values estimated by ICES (189). From year 2011 onwards, recruitment was fixed as the geometric mean of past recruitment values (189).

Recruitment Population	2008	2009	2010	after
Sole 7D	2.395e7	5.298e7	2.817e7	2.353e7
Plaice 7D	1.157e7	2.343e7	1.498e7	1.216e7
Sole 7E	2.379e6	2.885e6	4.301e6	4.301e6
Plaice 7E	5.560e6	1.006e7	5.007e6	5.007e6

TABLE 4.1 – **Recruitment (in number) values used for the first three simulated years and after.**

4.3.3.3 Management

We chose to focus only on the current transition scenario to MSY and to test a wide range of values for parameters defining this management scenario. This corresponds to situations where managers have already chosen how to manage a resource or an ecosystem, but where uncertainties remain on the best way to apply the chosen scenario. This allows us to find which particular range of parameter values gives maximum efficiency to the management measure. It also permits determining whether a range of parameter values allows reaching some robustness to uncertainties on biological parameters.

For the first three years of simulation, populations are managed using TACs. TAC values for these years correspond to those that were applied in 2008, 2009 and 2010 (Table 4.2). As a single TAC level was used to manage plaice in areas 7D and 7E, this TAC was split so that a TAC level could be attributed to each population. Plaice TAC in area 7D was set to 3500t for years 2008 and 2009 and to 3400t for year 2010, as recommended in ICES stock evaluations (190; 191; 172). The remaining part of the total TAC was attributed to plaice in 7E. This allowed us to get 2010 TAC levels for all simulated populations and simulate harvest control rules thereafter. From 2011 onwards TAC values are determined for each population by a harvest control rule (HCR) that controls the transition towards MSY. Every year during the transition period a value of maximum fishing mortality to be applied to the ecosystem is computed as a combination of the 2010 fishing mortality and the fishing mortality that would give the maximum sustainable yield. Transition duration being 5 years the proportion of F_{2010} in the computed F decreases by 20% every year and the proportion of F_{MSY} increases by 20%. The TAC level computed by the HCR is determined by other conditions in addition to those on fishing mortality : (i) Spawning biomass has to be above a minimum level ($MSYB_{trigger}$), (ii) TAC value cannot change by more than 15% from one year to the next, and (iii) the computed fishing mortality has to be below F_{pa}. A minimum landing size is also implemented for each species : 27cm for plaice and 24cm for sole.

TAC Population	2008	2009	2010
Sole 7D	6593t	5274t	4219t
Plaice 7D	3500t	3500t	3400t
Sole 7E	765t	650t	618t
Plaice 7E	1550t	1146t	874t

Table 4.2 – **TAC values used for the first three simulated years**

The database used in this paper, including HCR and gravity model java code, can be downloaded from the ISIS-Fish website (http ://www.isis-fish.org/download.html), as well as the latest version of the ISIS-Fish model.

4.4 Methods

4.4.1 Decision Theory

The info-gap decision theory (118; 119) aims at allowing decision making under severe uncertainty. This theory allows comparison between various courses of action (q ∈ Q), depending on states of Nature (u ∈ U). "u" is called the "ambient uncertainty" in the info-gap theory. The reward function $R(q, u)$ gives the expected outcome for a given course of action q and a given state of Nature u. When using a model to compute $R(q, u)$, the model gives the value of $R(q, u)$ associated to parameter values used to perform the simulation. The decision maker has to choose a critical value (r_c) below which the reward function should not drop (in case a high value of the selected output variable is desirable, for instance a high fish biomass). The robustness function ($\hat{\alpha}(q, r_c)$) is the greatest horizon of uncertainty α that can be tolerated (on Nature's state or on variables controlled by human activities) while being sure that the reward function did not cross r_c. The Info-Gap theory proposes several functions of uncertainty to compute the robustness function. In practise, the most popular one consists in testing parameter values around the reference parameter values set in the model that increasingly differ from these reference values, until a combination of values is reached for which $R(q, u) < r_c$. The distance between the last parameter values tested for which $R(q, u) \geqslant r_c$ and the reference values corresponds to $\hat{\alpha}(q, r_c)$. As a result, the decision maker knows for each possible action and state of Nature the level of uncertainty that can be tolerated. Thus, it is possible to choose the action that seems most appropriate to the situation, depending on management goals, local knowledge of the fishery and the environment, and the level of risk that is accepted by stakeholders. If the probability of occurrence of the various ecosystem states is not known, then a minimax approach (17) or other aspects of the Theory of Games (120) can be applied.

Many limitations of the info-gap approach have been underlined by Sniedovitch (182; 126) and are reviewed in (192). One of these is that it is performed around a given reference point and therefore is inherently local and not suitable to situations of severe uncertainty. This particularly is an issue in fisheries science where reference model parameterisations (corresponding to our knowledge of the state of an ecosystem) often poorly meet management goals. When this happens, then $\hat{\alpha}(q, r_c)$ is very small, and many other values of the parameters could lead to acceptable reward R. So this approach seems to be very conservative and cautious and does not fully allow the exploration of the input parameters space and the identification of an area in the input parameters space that gives acceptable results. Therefore, we choose to define a priori the window in the input parameters space that we wish to explore and perform the exploration by means of exploration techniques from sensitivity analysis.

4.4.2 Applying Info-gap to a complex model

To our knowledge, the Info Gap Theory has only been applied to analytical models. In the context of complex models (with no analytical solution), deriving the robustness function is a great challenge. The ISIS-Fish model belongs to this family of complex models. The model can be used to simulate the reward function $R(q, u)$ for each selected couple (q,u) in QxU. Many model runs have to be performed to explore the robustness of R to uncertainty in q and u. Therefore, we propose a pragmatical approach to apply Info-Gap theory to complex model, following two main steps : 1) a sensitivity analysis of the simulation model performed on the input parameters space UxQ ; 2) a classification trees analysis fitted to the model's outputs (simulated for the previous step) to discriminate the sensitive parameters and their range of variation accounting for reward $R(q, u)$ above the critical value r_c.

4.4.2.1 Perform sensitivity analysis

A good exploration of the input space can rely on the powerful tools provided by sensitivity analysis techniques. Many sensitivity analysis techniques are available to modellers (193; 194).

Most sensitivity analysis techniques can be divided into two parts : a method to explore the parameter space and a method to rank parameters according to their levels of sensitivity. Following (195; 196) we chose to focus on a global sensitivity analysis method associated to a variance decomposition method, instead of a one-at-a-time (OAT) method. The difference between local and global sensitivity analysis techniques is that global techniques study variations of the output variable over the entire range of values of the input parameters (197; 198). The main asset of global sensitivity analysis is that it allows us to measure interaction effects, which can be of great importance in complex ecosystem models.

We identified 81 parameters from our model on which to perform sensitivity analysis (Fig. 1,Step 0). These parameters can be split into three groups : biological parameters, technical parameters and management parameters. Biological and technical parameters (Table 4.3) correspond to states of Nature with various levels of uncertainties and management parameters (Table 4.4) allow us to test various management scenarios.

We chose to explore a window corresponding to +/- 50% of the reference value of each parameter (Fig. 4.1,Step 1). This range of values agrees with observations from (106) who noted that managers can rarely measure stock levels accurately and typically use confidence intervals of 50%.

From a biological point of view, exploring the same range of values for all parameters makes little sense. This window probably does not allow us to take into account the total variability of all parameters, as some may naturally vary within a greater range, but should be wide enough to contain most variability. On the other hand, this window may be too wide for some well-known parameters with little natural variability. This has to be taken into account when studying sensitivity analysis results. Indeed, some parameters may be identified as important because an unrealistically wide range of values was tested for them. By contrast, other parameters may be identified as little impacting only due to a too narrow range of values tested in the analysis.

The selected window in the input parameters space was explored by means of Latin Hypercube Sampling (LHS, (199)) using the "sensitivity" package (200) from R (163) (Fig. 4.1,Step 2). Latin Hypercube Sampling is a probabilistic sampling procedure that incorporates many of the desirable features of random sampling and stratified sampling (201). Then a a variance decomposition method gave us for each input parameter X_j a coefficient S_{Tj} ,corresponding to $S^{tot}_{(j)}$ in (202), indicating whether X_j is a sensitive input parameter or not for the output variable studied.

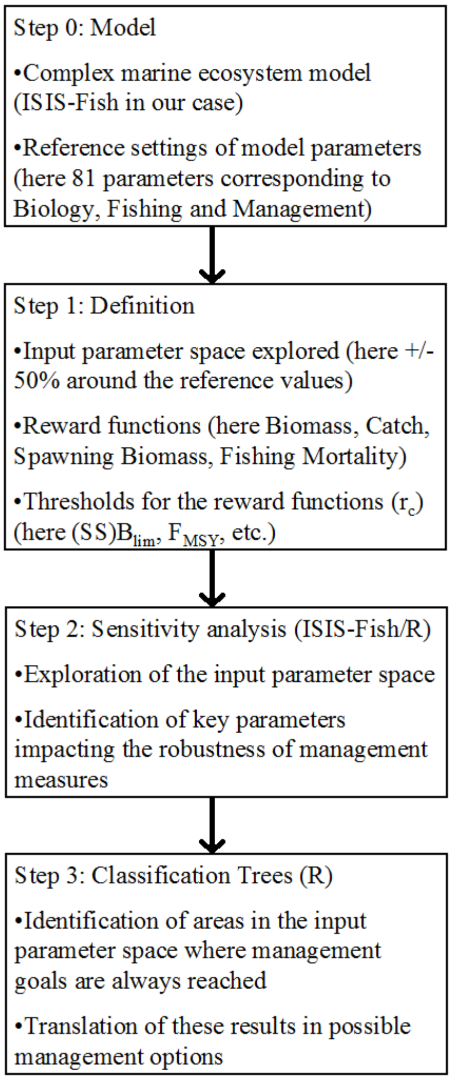

FIGURE 4.1 – **The general approach used to identify areas of interest in the input parameters subspace.** Once the model has been built and parameters identified (step 0) output variables to be studied and thresholds corresponding to these variables can be chosen and the input space to explore defined (step1). Then the input parameter is explored and important parameters identified (step 2). Classification trees are used to classify input parameters values depending on the output values they gave when exploring the input parameter space (step 3).

Parameters with S_{Tj} close to or equal to zero can be removed from the uncertainty analysis as they do not influence the model output. Model outputs we studied are biomass, spawning biomass, fishing mortality and catch. Spawning biomass and fishing mortality were studied in priority as they are variables commonly studied by

Parameter Name	Abbreviation
Catchability	Q
Mean Weight	MW
Recruitment	RE
Natural Death Rate	NDR
Growth Rate	K
Asymptotic Length	Linf
Time at the Origin	T0
Price	P
Selectivity Beam Trawl	SBT
Selectivity Net	SN
Selectivity Other Gears	SO
Target Factor Beaming	TFB
Target Factor Netting	TFN
Target Factor Other Métiers	TFO
Proportion of effort allocation coming from habits	habit

Table 4.3 – **Biological and technical parameters tested, for every population, métier or gear.** 50% variations were tested around the reference value. Each biological parameter exists in five versions, one for each population : sole 7D (S7D), plaice 7D (P7D), S7E or P7E and one for the "Other" group. Technical parameters are either defined at the population scale (S7D, etc.) or at the area scale (7D, 7E, or both : 7DE).

ICES working groups. Classification trees were then built on management parameters and natural parameters most influencing model outputs.

4.4.2.2 Point out isles of robustness with classification trees

Once the input parameters space has been explored, one has to find the boundary between input parameter values leading to acceptable outputs (relative to management goals) and input values leading to failures (Fig. 4.1,Step 3). As we use a complex model to compute values of $R(q, u)$ this boundary cannot be found analytically, but has to be identified from a limited number of simulations. Very powerful methods have been developed for machine learning that allow identifying the hyperplane separating input values from a dataset in two (or more) groups depending on resulting outputs, but these method are either black boxes or provide results that are too difficult to interpret for our needs. Therefore we chose to focus on classification trees (203), that allow for simpler representations of results by means of successive univariate splits of the set of input parameter values.

A classification tree is built step by step. At each step, a split is performed on a parameter belonging to the set of parameters on which the tree is built. A split separates the dataset (values of $R(q, u)$ simulated by the model and associated

Parameter Name	Abbreviation	Reference Value
Minimum Landing Size for Sole	MinSizeS7DE	24cm
Minimum Landing Size for Plaice	MinSizeP7DE	27cm
Duration of the transition framework	Trans	5yrs
Survival rate of discarded fish	PropSurv	0.25
Maximum yearly TAC variation	varTAC	0.15
Targeted fishing mortality at MSY, Sole 7D	FmsyS7D	0.29
Targeted fishing mortality at MSY, Plaice 7D	FmsyP7D	0.23
Targeted fishing mortality at MSY, Sole 7E	FmsyS7E	0.27
Targeted fishing mortality at MSY, Plaice 7E	FmsyP7E	0.19
Precautionary fishing mortality, Sole 7D	FpaS7D	0.4
Precautionary fishing mortality, Plaice 7D	FpaP7D	0.45
Precautionary fishing mortality, Sole 7E	FpaS7E	0.4
Precautionary fishing mortality, Plaice 7E	FpaP7E	0.45
HCR Trigger Biomass, Sole 7D	MsyBtS7D	8000t
HCR Trigger Biomass, Plaice 7D	MsyBtP7D	8000t
HCR Trigger Biomass, Sole 7E	MsyBtS7E	2800t
HCR Trigger Biomass, Plaice 7E	MsyBtP7E	2500t

TABLE 4.4 – **Management parameters tested.** 50% variations were tested around the reference value, except for the survival rate of discarded fish. Survival rates from 0 (no survival of discarded fish) to 0.5 (survival of 50% of discarded fish) were tested because of large uncertainties on this parameter and because the reference model value is 0. Parameters "Trans","PropSurv" and "varTAC" have similar reference values for all populations so they are only given once.

parameters) into two parts according to values of this parameter. When building a tree, the rule to perform a split is that some measure of discrepancy between the two datasets given by the split is maximized. Therefore the tree is built from the most important node (i.e. combination of a splitting parameter and a splitting value for this parameter) to the least important one. Classification stops when some criteria defined by the user are reached. For instance, classification can stop when the information gain given by a split (i.e. the increase in discrepancy between the two datasets obtained) is lower than a defined threshold. The terminal nodes of a tree are small datasets that are parts of the initial dataset on which the tree was built. They are called leaves. For a given tree, the path leading from a node (usually the first node) to a leaf is called a branch of the tree (Fig. 4.2).

Our goal being to determine the extent to which management measures can allow reaching management goals robustly, we perform classification in two successive steps for each output variable (Fig. 4.2). Let $\{R_1, R_2, \ldots, R_M\}$ be the set of all output variables studied. Let Q be the set of all management parameters in the model : $Q = \{X_{q,1}, X_{q,2}, \ldots, X_{q,n}\}$. Let U be the set of parameters corresponding to important states of Nature (fish biology and fleet characteristics) identified at step

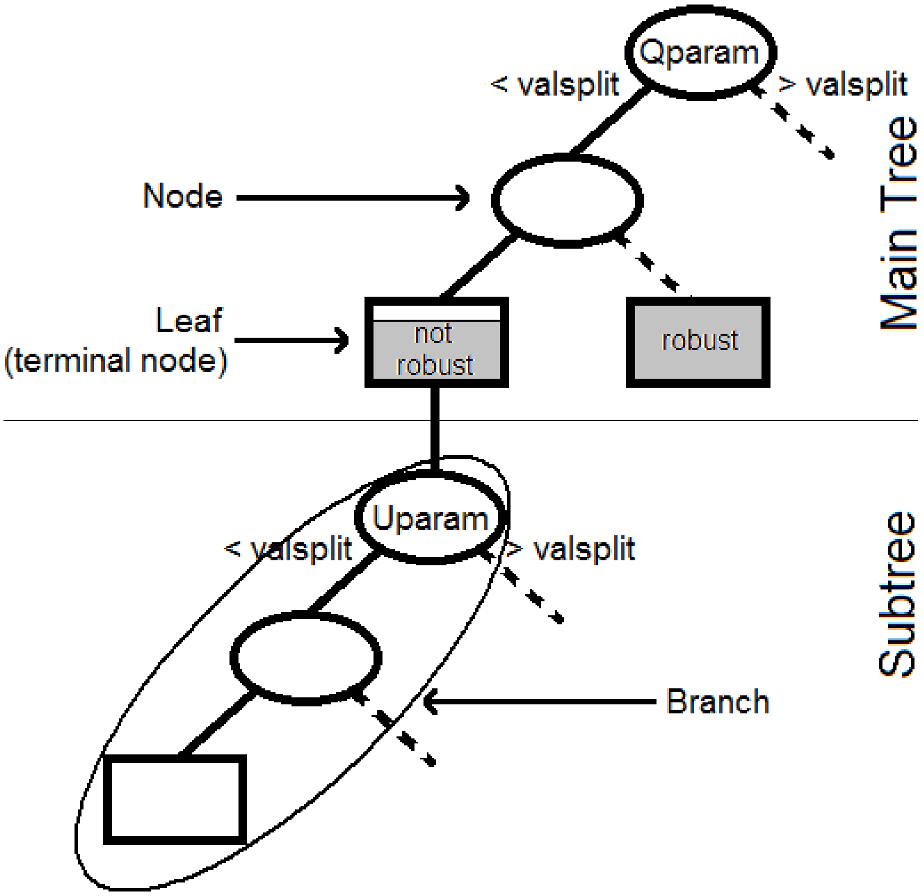

FIGURE 4.2 – **Successive use of trees for input parameters space exploration.** The first tree (called the main tree) is built on management parameters only, as they are the main concern in the study. When management parameters do not allow robustly reaching management goals, trees (called subtrees) are built for each leaf of the main tree. The second set of trees is built on parameters identified as important by the sensitivity analysis. Results of interest are parameters values corresponding to branches (either of the main tree only or both the main tree and a subtree) leading to robust leaves.

$1 : U = \{X_{u,1}, X_{u,2}, \ldots, X_{u,m}\}$.

1. A classification tree (the main tree) is built based only on management parameters belonging to Q. Each leaf of the tree may contain either successes (values of $R(q, u)$ above some threshold) or failures, or a mix of both. A leaf can be considered "robust" when it contains only successes, i.e. all realisations belonging to this leaf correspond to successful management configurations, whatever the state of Nature. In practice, as the classification relies on model simulations only robust nodes with a high weight (i.e. containing many

simulations) should be considered. The concept of robustness can be adapted, depending on the willingness of managers to tolerate risk, so that a node may be considered to be robust if its proportion of successes is above some threshold. Here, we fixed that threshold to 99% of successes as node boundaries can sometimes be difficult to identify very accurately and therefore a few failures can be included in a node that would otherwise be robust. At the end of this first step, leaves are classified into two classes : “robust leaves” if leaves are identified within the tree based on management that allow always reaching management goals or “not robust leaves” for others. Leaves that are not robust are used to perform a second stage of classification.

2. For each leaf that is not robust, we performed a second classification on parameters belonging to U. This allows us to grow subtrees in a limited amount of time while knowing that important parameters are tested. If a robust terminal node is identified within a subtree associated to a leaf of the main tree, then we know within which range of values of $\{X_{u,1}, \ldots, X_{u,m}\}$ management parameter values corresponding to that leaf will allow reaching management goals.

This approach has two assets : (i) if management parameters tested are consistent with current management measures then we know what level of uncertainty or variability on natural parameters can be tolerated while still reaching management goals at the end of the period ; (ii) if current management parameters do not correspond to those identified by the tree then we know how (and to what extent) current management should be altered to have a chance to reach management goals considering the uncertainty on management parameters.

The method we chose to build classification trees is that of conditional trees (204) that allows overcoming usual problems of possible overfitting, selection bias, or input parameters scaling.

We assessed tree and subtrees instability (i.e tree structure changing when slightly modifying the dataset used to build it) by means of re-sampling techniques. For the main tree and subtrees corresponding to leaves of the main tree, we built sets of 500 trees with subsets containing 95% of the dataset. From all these trees, we identified the tree appearing most often and focused on it, making the hypothesis that all trees would converge to this tree type provided the dataset is big enough. We also compute average splitting values and standard deviations from the 500 replicates so as to have clear indicators of tree variability (see Supporting Information S1 for more details about this method). If tree variability is too high (it is especially the case for subtrees, as they are built from a subset of the main dataset corresponding to their associated leaf), it makes little sense to focus only on one particular tree type. In this case, results from the most common tree types can be provided.

4.5 Results

4.5.1 Sensitivity analyses

Results from the sensitivity analysis are presented in Fig. 4.3, where each column stands for an output variable and each row for a different input parameter. Only rows should be compared as the intersection of a row and a column represents the amount of the total variance of a particular output variable explained by an input parameter. The naming of the various input parameters appearing in Fig. 4.3 is detailed in Table 4.3 and Table 4.4. Parameters are presented individually in Fig. 4.3 whereas parameters with similar values for all populations appear only once in Table 4.3 and Table 4.4. Besides F_{MSY} and F_{pa} were varied jointly in the sensitivity analysis and appear as a single parameter $F_{MSY}F_{pa}$ in Fig. 4.3.

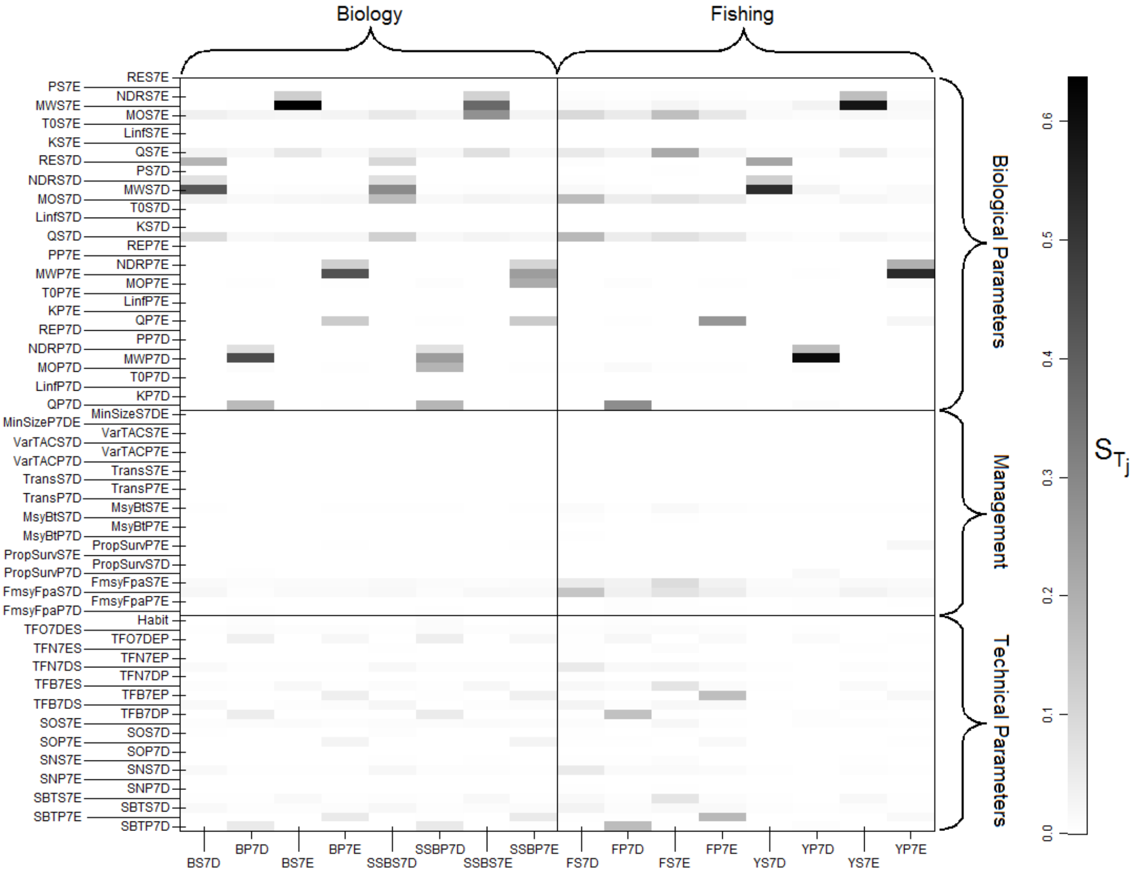

Figure 4.3 – **Results of the sensitivity analysis.** Each row corresponds to one of the 81 input parameters tested and each column to an output variable. Output variables are biomass (B), Spawning Biomass (SSB), Fishing mortality (F) and Catch (Y). Results have to be studied in columns, black cells indicating important parameters for a given output.

The goal being to reduce the model, we focus only on the most sensitive parameters that really stand out compared to the others. These input parameters are listed from the most important to the least important in Table 4.5 and Table 4.6. First order interactions between parameters were also tested. Interactions are pretty straightforward, the most sensitive parameters producing the strongest interactions. Interactions do not especially drive output variables in our model. Fishing mortality is the output variable most impacted by interactions, but even in this case the most sensitive interaction is always less than 0.1 times as impacting as the most important parameter. We therefore focus only on main effects when presenting our results. It is interesting to note that the most important parameters impact the four studied populations whereas less important parameters are specific to only some of these populations. Also, small interactions between areas 7D and 7E appear, as sometimes populations from one area can be slightly impacted by parameters defined for the other area.

Biomass				**Spawning Biomass**			
Mean Weight (MW) Catchability (Q) Natural Death Rate (NDR)				Mean Weight Maturity Ogive (MO) Catchability Natural Death Rate			
S7D	P7D	S7E	P7E	S7D	P7D	S7E	P7E
RES7D MOS7E	SBTP7D TFB7DP	MOS7E QS7D	SBTP7E TFB7EP	RES7D	SBTP7D	QS7D	SBTP7E
0.751	0.742	0.825	0.716	0.676	0.669	0.786	0.666

TABLE 4.5 – **Input parameters most impacting output variables.** Parameters are sorted from the most impacting to the less impacting, and the proportion of total output variance explained by these parameters is given at the bottom of each column.

Fishing Mortality				**Catch**			
Catchability				Mean Weight Natural Death Rate			
S7D	P7D	S7E	P7E	S7D	P7D	S7E	P7E
MOS7D FmsyFpaS7D MOS7E QS7E	SBTP7D TFB7DP MOS7E QS7D	MOS7E FmsyFpaS7E QS7D FmsyFpaS7D	SBTP7E TFB7EP MOS7E QS7E	RES7D MOS7E FmsyFpaS7D	MWS7E MWS7D PropSurvP7D	QS7E FmsyFpaS7D QS7D	QP7E PropSurvP7E TFB7EP
0.571	0.637	0.542	0.629	0.847	0.805	0.799	0.747

TABLE 4.6 – **Input parameters most impacting output variables (continued).** Parameters are sorted from the most impacting to the less impacting, and the proportion of total output variance explained by these parameters is given at the bottom of each column.

The first thing that stands out from these results is that all studied output variables except fishing mortality are mostly impacted by biological parameters, technical parameters having a lower impact on the outputs. Management measures have little impact on all output variables, even those directly related to fishing. The parameter of our harvest control rule that most influences output variables is the target value of fishing mortality F_{MSY} (and the associated F_{pa}). However the effects of the F_{MSY} parameter are limited to fishing mortality and effects of management on biomass or spawning biomass are low.

Interestingly, while fishing mortality is driven by catchability and technical parameters, catch appears to be impacted almost only by biological parameters. Fish mean weight-at-age alone explains up to 60.0% of the total variance of the catch which leaves little variance to be explained by other parameters. This importance of fish weight may be linked to the fact that we study biomass and catch in tons, and not abundances or catch in numbers. Besides fish mean weight-at-age is a parameter with relatively small variability, and testing 50% around this parameter's reference value may artificially give it an important weight.

Mean weight is the input parameter most impacting model output variables, and is followed by natural death rate and catchability (Table 4.5 and Table 4.6). Recruitment has noticeable impact only for sole in area 7D, and this impact is moderate.

In our model, beam-trawling is the most impacting fishing activity, with high S_js for both its target factor (TFB) and selectivity (SBT). It impacts both populations of plaice, as well as fishing mortality of Sole in area 7E. The effects of netting are limited to sole in area 7D and can be noted only for fishing mortality (moderate S_js for SNS7D and TFN7DS). Biomass and spawning biomass of plaice in area 7D are also impacted by the target factor of the "other métiers" group of our model, that bundles together all the lesser operated métiers.

4.5.2 Conditional Tree analysis

Conditional trees can be built according to the previously presented method for all output variables studied in the sensitivity analysis. However, only the most important variable for conservation, spawning biomass, will be discussed here for populations of sole and plaice in area 7D. The management goal, for both sole and plaice, is that spawning biomass must remain above 8000t.

4.5.2.1 Sole 7D

The tree structure obtained when building the main tree (Fig. 4.4) on the entire training dataset corresponds to that identified when creating a large number of trees on a smaller subset (see Supporting Information S2 for more details about the tree-building process). Therefore we can infer that our dataset is big enough to ensure good tree stability and that all trees built from subsets would converge to that particular tree provided we had enough data. As no leaf of this tree is robust, i.e they all contain some failures, a subtree was built for each of the six leaves of the main tree to determine if management measures allow reaching management goals within a certain range of values of natural parameters, as shown in Fig. 4.2. Among all subtrees corresponding to a leaf of the main tree, only some subtrees are of interest (robustness + high weight), we therefore decided to focus on those. For each subtree, we compute mean splitting value and standard deviation around these values because there can be variations in the splitting values at each inner node.

Branches of interest can be identified within subtrees that lead to leaves that are robust and have a high weight. Besides, it appears that these particular branches are much more stable than other branches of subtrees. Table 4.7 shows for each leaf of the main tree built on management measures parameters and split values corresponding to interesting subtree branches that would allow robustly reaching management goals. A very interesting thing to observe is that for all subtrees all important branches identified correspond to boundaries on natural parameters that contain the reference model parameterisation (which corresponds to value 0.5). Therefore all robust nodes identified in Table 4.7 can be reached by means of changes made to management measures. The distance between the reference model parameter value (supposed current "real" state of the system) and the split parameter value identified by the classification tells us how much uncertainty or variability can be tolerated around the reference parameter value to ensure reaching management goals.

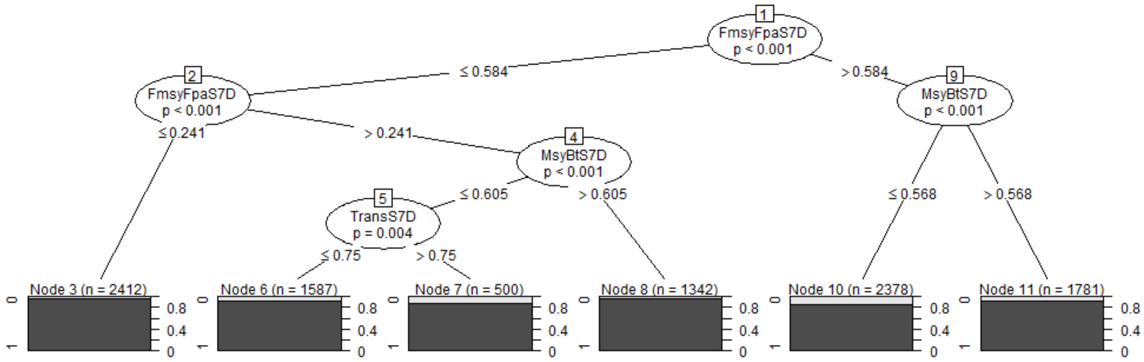

FIGURE 4.4 – **Main tree built on management measures for sole in subarea 7D.** The criterion separating successes from failures is a 8000t spawning biomass threshold. Splitting variables (and values) are sorted by importance, from the top to the bottom of the tree (the most important corresponding to node 1). Values appearing on branches of the tree are the splitting values. The black and grey squares at the bottom of the tree are the terminal nodes or leaves.

Leaf 6 of the main tree (top center of Table 4.7) is of particular interest as both management and natural parameter values leading to the robust node are compatible with reference model parameterisation (Table 4.8). This means that current management measures should allow reaching a spawning biomass of sole above 8000t by 2018, provided reference model parameterisation correctly represents the environment. As our model cannot correctly represent the environment, we look at conditions imposed on environmental parameters by subtree branches. In both cases, only two conditions on environmental parameters are imposed : one on recruitment and the other on mean weight-at-age. The first branch identified tells us that it is possible to reach management goals if (i) mean weight-at-age of sole in area 7D is no more than 32% lower than reference model sole mean weight-at-age and (ii) recruitment until 2018 is no more than 12% lower than recruitment used to perform simulations in our model (which is the geometric mean of recruitment values estimated for the previous years). The second option identified corresponds to a mean weight no more than 24% lower than reference and to a recruitment recruitment no more than 22% lower than reference recruitment value. If a 22% variation around a mean recruitment and a 24% variation in mean weight are deemed sufficient to encompass both natural variability and our uncertainties, then management measures can be left the way they are. If a greater margin is needed on natural parameter values, then management parameters have to be modified.

Moving from leaf 6 to leaf 3 imposes reducing $F_{msy}F_{pa}S7D$. Results for leaf 3 (Table 4.9) allow greater uncertainties on recruitment than those for leaf 6. In this case, reducing the target fishing mortality of the management measure by 26% or more will allow tolerating recruitments up to 37% lower than the reference recruitment of our model, but lower uncertainties can be tolerated on mean weight-at-age, and natural death rate now also has to be taken into account.

Results for other leaves of the main tree are rather similar, and correspond to different combinations of the most important management parameters (target F, MSY Btrigger and transition duration). There can be variability in results obtained between leaves, and sometimes stricter conditions on environmental parameters are not observed in leaves where they could be expected (e.g. when the value of FmsyFpaS7D is increased). This illustrates the fact that some results obtained may be too conservative and that some uncertainties remain about the position of the boundary separating acceptable parameter combinations from unacceptable parameter combinations.

4.5.2.2 Plaice 7D

Table 4.10 shows that, similarly to what was observed for sole, it is the value of fishing mortality targeted by the harvest control rule that mostly determines whether management goals will be reached or not. For plaice, this parameter is associated to the duration of the transition period and to the survival rate of discarded fish.

Leaf 4 (see S3 for more information about trees built for plaice) if of great interest as it is the only one that contains values of current management parameters, and natural parameters that are compatible with our perception of plaice life cycle. The two options identified in leaf 4 are quite similar, except for the uncertainty that can be tolerated on fish mean weight. This shows that when building trees both split value were identified as having rather similar abilities to split a node into two other nodes, and we cannot say that one value really is better than the other. The goal being to be robust to uncertainties, only the value of 0.366, corresponding to a maximum 13% variability on fish mean weight should probably be considered (Table 4.11). Uncertainties on parameters such as fish mean weight-at-age or age at maturity usually being lower than uncertainties on recruitment, management goals may still be reachable with the current management (even if the 0.501 threshold on FmsyFpa leaves no room for variation on this parameter, the reference value being 0.5).

Values of FmsyFpa above 0.501 (leaf 12 and leaf 13) are associated to survival rates of discarded fish above 0.251, which means that more than 13% of discarded plaice have to survive. Even if this were true, other conditions on natural parameters are not fulfilled except for one branch of leaf 13. This combination of parameters appearing in leaf 13 but not in leaf 12 where management is tighter makes little sense and illustrates tree instability and the need for a bigger training set and a more thorough tree exploration.

Other leaves correspond to other combinations of management and natural parameters. Even if many natural parameters do not agree with our reference model parameterisation, split values for catchability and sometimes fish mean weight are close to model values, and little changes in these parameters could make management goals reachable for a wider range of management scenarios.

4.6 Discussion

4.6.1 Management Implications

For sole as well as plaice, no combinations of management measures could be identified that always allow reaching management goals accounting for “Nature uncertainty”. However, the sole population in area 7D is in a good enough state to make the 8000t spawning biomass goal recommended by working groups reachable for a rather wide variety of management parameters values and states of Nature. In particular, management goals on spawning biomass can be reached with current management, provided mean weight-at-age and recruitment of sole do not vary too much. This seems acceptable for mean weight, as it is not a highly variable parameter. On the contrary, variations in recruitment higher than 22% seem likely to happen for sole in the Eastern Channel (205). In this case, the model suggests strongly reducing the target fishing mortality (division by more than two of F_{MSY} and F_{pa}) so as to be able to withstand much stronger variations in recruitment.

For plaice in area 7D the spawning biomass threshold chosen by the working group is also Bpa=8000t, which corresponds in the evaluation to a fishing mortality threshold Fpa=0.45. However, this goal is harder to reach for the plaice population in 7D than it was for sole. This can be seen with the much smaller ranges of values of management parameters that allow reaching management goals. However, tolerable ranges of values of natural parameters are rather broad, and these parameters are not known for their high variability (in particular, recruitment is not one of them, and could have been an issue otherwise (206)). Therefore, management goals could be reached, provided plaice stocks are carefully managed. Trees built for plaice are also less stable than those of sole. This instability may have two causes : (i) too few model simulations reach management goals, i.e. the state of the stock is so bad that only a fraction of parameter values tested yield acceptable results or (ii) uncertainties in the life traits of the modelled species are so high that parameters importance and split values cannot be assessed correctly. These two aspects may be linked, bad stock state possibly leading to more variability.

These results are coherent with what is known of the history of sole and plaice stocks in the English Channel. Indeed, mean fishing mortality estimated by working groups for sole remained between 0.3 and 0.6 since 1989 whereas plaice fishing mortality evolved between 0.45 and more than 1.2 (most values being equal to or above 0.6) during the same period (207). Target and precautionary fishing mortalities for both species being rather similar, it can be said that plaice was more overexploited than sole. Management goals being difficult to reach for plaice our model cannot correctly predict successes, which increases tree instability. As the state of the stock improves, tree stability and therefore our ability to make correct predictions will increase. It is nonetheless worth noting that despite model simplicity and uncertainties our results are coherent with what is known about the studied fish stocks.

Results presented in this article only concern spawning biomass, but similar analyses were performed for the other output variables mentioned previously. The goal here is not to look for the most robust management method, which makes little sense if the analysis is not multivariate. For instance, the best way to maximize spawning biomass is to stop fishing, and there is no need for a complex model to determine this. The interest here is to find management measures that allow reaching management goals on various (and possibly conflicting) output variables and determine how all these constraints on management can be combined. In particular, the key point is to find management measures that allow keeping biomass to acceptable levels while guaranteeing a high enough income to fisheries. A first insight can be obtained by looking for conditions ensuring that catches of flatfish species do not go below a certain level. As it is possible to model fishing costs and fish price in the ISIS-Fish model, it would be possible to look for conditions allowing reaching given economic goals, provided these criteria can be found. Here, we chose to study output variables separately and then look for similarities or discrepancies in management measures manually. But multivariate classification methods exist, and would be an ideal choice here provided hypotheses they are based on and types of results they provide are compatible with our needs. These multivariate techniques could also be used to perform multi species analyses, so that we make sure changing management measures on a species does not negatively impact another species. Here, this problem did not arise since sole is only impacted by "sole" parameters and plaice by "plaice" parameters (Table 4.8, Table 4.9, Table 4.11).

4.6.2 Caveats

The main limit of techniques we used is that many model simulations are needed. Otherwise reliability of sensitivity analysis results decreases, as large parts of the input space can be left unexplored, and results obtained with classification trees can become highly unstable if they cannot be trained properly. Here, we could perform many simulations because we used a "simple" model (one run takes about one minute on one core), but this method could be harder to apply to a more complex model with longer simulation times. However, the quick increase in available computing power opens very interesting prospects concerning the exploration of complex models input parameters spaces.

Another limit of the method we used is that we built classification subtrees from the main tree obtained when using the whole training dataset. Even if the main tree's type corresponds to the most common tree type identified, it could be interesting to grow subtrees from the leaves of a mean tree that would account for tree variability. This would allow us to take variability on management parameters into account when building subtrees. As standard deviations can also be computed, this would allow us to determine margins of uncertainty around split values. Depending on the needed level of robustness to uncertainties this would permit testing hypotheses and finding areas considered safe at the needed level of confidence. Another possible concern about classification trees is the size of the sub areas they can distinguish in the input parameters space. Would they be able to find "isles of robustness in a sea of non-robustness"? No extensive tests were run so as to answer this question but we think the method would still be appropriate in this case, even if it may not be the most suitable one. In this case, no constraints should be applied to the tree building process, so simulation time may increase a lot and trees may become much larger than those presented in this paper. This would also make results a lot harder to interpret and translate into applicable management measures.

Concerning the model itself, the weight of the "Other Métiers" group (i.e. métiers not explicitly taken into account in the model) in the sensitivity analysis shows that additional information could have been obtained, had the fishing activity been modelled in more details. This may illustrate the fact that our model is overly simplified and does not take into account enough parameters (at least when modelling the fishing activity). Indeed, we only model explicitly two species and three fleets corresponding to two gear types. Besides, fishing, management and population zones are defined at the scale of the ICES area. Therefore, only large-scale changes can be tested. As few ecosystem compartments are modelled, we only focus on a limited number of processes. Adding information to the model may greatly change its behaviour, especially because of interactions between parameters that cannot be observed in the current simple model. Changing the way in which the model represents the studied processes could also have notable impacts. Indeed, (192) state that not taking model structural uncertainty into account is one of the main flaws in the info-gap approaches that have been implemented so far. The same hypothesis is made in our study, i.e. that parameter variability is more impacting than structural uncertainty. We made this hypothesis mostly because our ISIS-Fish model is based on equations that are commonly used by stock assessment working groups, thus our results should at least be coherent with those given in stock evaluations. Nonetheless, structural uncertainty of our model should (at least partly) be assessed in order to determine whether model structure strongly impacts conclusions drawn, or if they depend (as we supposed in this study) on variations of input parameters.

We use a three-years "forcing" period at the beginning of our model, so as to use available information to better represent the studied ecosystem. This allows us to set management parameters to their real value during that period. We also chose to set recruitment to values estimated by working groups for that period. This could partly explain why recruitment little impacts values of output variables in the sensitivity analysis. Besides, we deal with long-lived species, so there can by a many-years delay between changes in recruitment and the observation of impacts on other parameters caused by these changes. For other biological parameters this forcing period was not used and they therefore are fixed to the calibrated 2008 value, with a 50% level of uncertainty. So on the one hand no uncertainties are associated to recruitment for the first tree years, whereas there is uncertainty in the estimation made by working groups, and on the other hand other biological parameters are given a fixed value with high uncertainties during that period. As all past values can be estimated with some level of certainty, it would seem more logical to force all biological parameters for the first tree years to their estimated values and give them low uncertainty levels corresponding to the accuracy of the estimation. Then for values from 2011 onwards either calibrated or mean values can be used, with much higher levels of uncertainty depending on the level of natural variability and uncertainty corresponding to these parameters.

The question of the range of parameter values to explore is linked with these levels of variability and uncertainty. We chose to perform sensitivity analysis on a 50% window around our model reference parameterisation, while 20% variations are commonly tested when performing sensitivity analyses (208; 209). As variability can be very high on some parameters and very low on others, this method could be improved. Indeed, it gives too high a weight to parameters the variability of which is overestimated and lessens the weight of parameters with higher variability. A next step in the modelling process is to perform sensitivity analysis on the domain of variation of parameters tested.

Model simplicity and modelling choices we made prevented us from testing spatial management measures. In addition to model changes that we discussed, our goal now is to better represent fish populations and fleets targeting them in our model. In particular, decreasing the scale at which processes are modelled may allow us to test spatial management measures. Combining these measures to those already included in our model, we hope to find out if spatial management measures such as MPAs can allow reaching management objectives with a higher robustness to uncertainties on the state of the ecosystem.

4.6.3 Management Strategy Evaluation

The approach we used is very close in essence and in goals to that implemented when performing a management strategy evaluation (MSE). Indeed, when performing MSE, the goal is to assess the consequences of a set of management procedures against key performance measures (210). Similarly to what we wanted to do, the MSE approach does not seek to prescribe an optimal strategy, but rather to provide decision makers with sound information on which to base their decisions. Providing sound information implies identifying strategies that are robust to uncertainty and natural variation (211), as was done in our approach. MSEs can be based on various interacting models, from very simple ones to very complex ones.

The main difference between our approach and common MSE is that we did not just test management scenarios but let parameters controlling the Harvest Control Rule vary within a chosen range of values. Therefore, despite strong environmental drivers in our model (possibly coming from choices made when performing sensitivity analysis), our method is able to find a range of management parameters and values that allow reaching management goals. Therefore, instead of determining whether a scenario is robust or not, we can identify a subspace of management parameters values that are robust to uncertainties. Besides we can also define ranges of variation on natural parameters that can be tolerated and still allow reaching management goals, an information of potentially great importance. Therefore, within the range of management parameters values that can yield robust outcomes, managers can choose the combination of values that seems best depending on their goals and their willingness to take risks.

Such results could be discussed with stakeholders so as to determine the opportuneness of various management measures and to better perceive which situations would lead to failures to reach management or economic goals, and potential solutions to avoid them. This method also allows identifying particular input parameters on which uncertainties should be reduced in priority to allow for better forecast. If the cost of a reduction in uncertainty on some parameters is known (e.g. the price to pay to get additional samples) then it can be balanced against the cost of not robustly reaching management goals and choices can be made depending on managers' priorities. Once these important parameters have been identified an adaptive management procedure (212; 213) targeting them in particular could be set up. An interesting feature of the method is that there is no need to identify a priori sources of uncertainty impacting input parameters. Uncertainty is treated as a whole and identification of sources of uncertainty is only an optional step that can be made by the user.

Leaf 3		**Leaf 6**				**Leaf 10**	
$F_{MSY}F_{pa}STD$<0.241		**0.241**<$F_{MSY}F_{pa}STD$<**0.584** $MSYB_{trigger}STD$<**0.605** TransitionSTD<**0.75**				$F_{MSY}F_{pa}STD$>0.584 $MSYB_{trigger}STD$<**0.568**	
Subtree 1		Subtree 1		Subtree 2		Subtree 1	
Mean Split Value	Standard Deviation	Mean Split Value	Standard Deviation	Mean Split Value	Standard Deviation	Mean Split Value	Standard Deviation
$MWSTD$>**0.317**	0.0006	$MWSTD$>**0.172**	0.0009	$MWSTD$>**0.260**	0.02	$MWSTD$>**0.248**	0.005
$RESTD$>**0.131**	0.003	$RESTD$>**0.383**	0.002	$RESTD$>**0.279**	0.02	$RESTD$>**0.097**	0.009
$NDRSTD$<**0.801**	0.007						

TABLE 4.7 – **R**esults of the classification performed with conditional trees for sole. Each block corresponds to a leaf of the main tree and gives conditions on management parameters that are needed to reach it. The lower part of each block corresponds to branches identified from subtrees and gives conditions on environmental parameters that are to be added to those on management to reach a robust terminal node. Cells that are compatible with the reference model parameterisation (i.e. containing value 0.5) are in bold type.

Management
$0.22 < F_{MSY}S7D < 0.31$ $0.30 < F_{pa}S7D < 0.43$ $MSYB_tS7D <$ 8840t TransitionS7D < 7.5yrs
Environment
$MWS7D > MWS7D_{ref}$ - 24% $RES7D > RES7D_{ref}$ - 22%

Table 4.8 – **Management parameters values identified from leaf 6 and environmental variability they allow dealing with.** For sole in subarea 7D.

Management
$F_{MSY}S7D < 0.22$ $F_{pa}S7D < 0.30$
Environment
$MWS7D > MWS7D_{ref}$ - 18% $RES7D > RES7D_{ref}$ - 37% $NDRS7D < NDRS7D_{ref} + 10\%$

Table 4.9 – **Management parameters values identified from leaf 3 and environmental variability they allow dealing with.** For sole in subarea 7D.

Leaf 4				**Leaf 13**			
$F_{MSY}F_{pa}P7D<\mathbf{0.501}$ PropSurvP7D<**0.759** TransP7D<**0.52**				$F_{MSY}F_{pa}P7D>0.86$ PropSurvP7D>**0.251**			
Subtree 1		Subtree 2		Subtree 1		Subtree 2	
Mean Split Value	Standard Deviation	Mean Split Value	Standard Deviation	Mean Split Value	Standard Deviation	Mean Split Value	Standard Deviation
$MWP7D>\mathbf{0.258}$	0.0005	$MWP7D>\mathbf{0.366}$	0.0006	$MWP7D>\mathbf{0.445}$	0.003	$MWP7D>\mathbf{0.452}$	0.004
$QP7D<\mathbf{0.720}$	0.0003	$QP7D<\mathbf{0.720}$	0.0001	$QP7D<0.447$	0.002	$QP7D<\mathbf{0.549}$	0.0001
$MOP7D>\mathbf{0.259}$	0.02	$MOP7D>\mathbf{0.254}$	0.02	$MOP7D>\mathbf{0.170}$	0.02	$MOP7D>\mathbf{0.206}$	0.02

Leaf 12					
$0.501<F_{MSY}F_{pa}P7D<0.86$ PropSurvP7D>**0.251**					
Subtree 1		Subtree 2		Subtree 3	
Mean Split Value	Standard Deviation	Mean Split Value	Standard Deviation	Mean Split Value	Standard Deviation
$MWP7D>0.665$	0.005	$MWP7D>0.667$	0.006	$MWP7D>0.529$	0.002
$QP7D<0.459$	0.006	$QP7D<\mathbf{0.587}$	0.08	$QP7D<0.458$	0
$MOP7D>\mathbf{0.253}$	0.002	$MOP7D>\mathbf{0.240}$	0.02	$MOP7D>\mathbf{0.310}$	0.02

TABLE 4.10 – **R**esults of the classification performed with conditional trees for plaice. Each block corresponds to a leaf of the main tree and gives conditions on management parameters that are needed to reach it. The lower part of each block corresponds to branches identified from subtrees and gives conditions on environmental parameters that are to be added to those on management to reach a robust terminal node. Cells that are compatible with the reference model parameterisation (i.e. containing value 0.5) are in bold type.

Management
$F_{MSY}P7D < 0.23$
$F_{pa}P7D < 0.45$
TransP7D $<$ 5.1yrs
PropSurvP7D $<$ 0.38
Environment
$MWP7D > MWP7D_{ref}$ - 13%
$QP7D < QP7D_{ref} + 22\%$
$MOP7D > MOP7D_{ref}$ - 25%

Table 4.11 – **Management parameters values identified from leaf 4 and environmental variability they allow dealing with.** For plaice in subarea 7D.

4.7 Supporting Information S1 : Dealing with conditional trees instability

Among methods designed to build classification trees, conditional trees (204) allow overcoming problems of possible overfitting, selection bias, or input parameters scaling. Variable selection and splitting are performed in two successive steps by means of hypothesis testing. For variable selection, the null hypothesis $H_0^j : P(Y|X_j) = P(Y)$ is tested by means of permutation tests for each parameter X_j. This amounts to making the null hypothesis that knowing X_j does not increase our knowledge of Y. If this hypothesis can be rejected for at least one X_j, parameter X_j with strongest association to Y is selected for splitting. Then various split values are tested for parameter X_j that split its domain of variation into two subsets A and $\bar{A}$. The split value selected is that maximizing a test statistic measuring the discrepancy between $Y|X_j \in A$ and $Y|X_j \in \bar{A}$. This method ensures that the tree built is optimal relative to the criteria chosen and so no step of tree pruning is needed once the tree has been built.

One of the greatest issues when using classification trees is their instability. Indeed, tree structure as well as split values can change greatly when slightly modifying the dataset used for classification. Methods such as bagging (214; 215), random forests (216) or gradient boosting (217; 218) allow dealing with tree instability and increasing prediction accuracy. These methods build an average tree from a high number of simulated trees, but characteristics of each individual tree are lost.

To estimate tree stability but keep tree characteristics we first build 500 replicates of the tree, using samples containing 95% of the total number of simulations contained in our training set (i.e. 9500 simulations). From these 500 trees we assess similarity between trees by computing the mean distance between all 500 trees built, using the method from (219). Distance $D(A_1, A_2)$ between two trees is :

$$\begin{cases} D(A_1, A_2) = 1 & \text{if } A_1 \text{ and } A_2 \text{ have different structures} \\ D(A_1, A_2) = \sum_{t=0}^{T} d(A_1, A_2)_t \cdot \dfrac{1}{T} & \text{if } A_1 \text{ and } A_2 \text{ have similar structures} \end{cases}$$

where t is a given node of the tree, T the total number of inner nodes and $d(A_1, A_2)_t$ the distance between tree A_1 and tree A_2 for inner node t. Two trees A_1 and A_2 have similar structures if they have the same set of nodes at similar locations. For trees with similar structure, distance between nodes is computed as :

$$\begin{cases} d(A_1, A_2)_t = 1 & \text{if } A_1 \text{ and } A_2 \text{ do not split on the same parameter at node t} \\ d(A_1, A_2)_t = \dfrac{|\delta_1 - \delta_2|}{range(X_k)} & \text{if } A_1 \text{ and } A_2 \text{ split on the same parameter at node t} \end{cases}$$

where δ_1 is the split value for parameter X_k at node t for tree A_1 and δ_2 is the split value for parameter X_k at node t for tree A_2. The range of values that parameter X_k can take is denoted range(X_k).

If mean distance between trees is small then tree stability can be seen as good.

This measure of distance between trees gives information about tree stability but does not allow determining whether a particular tree structure can be selected to build subtrees. So as to determine whether the tree built from the entire training dataset is stable, we identify among the 500 trees built how many are of similar type with this particular tree. Here, we mean by tree type, trees with same structure and same split parameter for each node. If the tree built from the whole dataset corresponds to the most common tree type identified from the set of 500 trees, then we deem it is stable enough to build subtrees. One of the assets of sorting trees by tree type is that, for each inner node of a particular tree type, the mean and standard deviation of the splitting value can be computed. We can therefore obtain a mean tree for each tree type and asses its variability, so as to estimate which splitting values would ensure a high robustness to uncertainties.

4.8 Supporting Information S2 : Building conditional trees for sole in 7D

Trees built on management measures for spawning biomass of sole in area 7D are very stable. Indeed, over the 500 trees built, 434 trees share the same type. This is linked with a small mean distance between trees (0.24) and a standard deviation of 0.38. The high standard deviation comes from the fact that 20 other tree types can be identified in addition to the most common tree type. There are few of these trees, but as two trees with different structures are attributed a distance of 1 (the maximum distance between trees) these trees act as outliers that increase variance, while mean remains low.

The most common tree type identified has 5 inner nodes and 6 terminal nodes, with no robust leaf. The first split (corresponding to node 1) is performed on the target fishing mortality of the harvest control rule. Another split is performed on this target F, two splits are performed on the value of trigger biomass and the last split is performed on transition duration (Table B 4.12). No terminal leaf of this tree is robust. This tree structure is similar to that of the tree built from the entire dataset. There is variability in the value selected at each node to perform the split. For some nodes different values are selected almost the same number of times to perform the split. However, these values remain rather close one to another, which leads to moderate variations around a mean splitting value. Splitting values for the main tree built on the entire training dataset are summarized in Table B 4.13. All split values in the main tree correspond to those identified as most common except for node 5 where the most common splitting value is 0.82.

Node	Parameter	Mean split value	Standard deviation
1	FmsyFpaS7D	0.633	0.062
2	FmsyFpaS7D	0.234	0.030
4	MsyBtS7D	0.598	0.011
5	TransS7D	0.764	0.060
9	MsyBtS7D	0.571	0.023

TABLE 4.12 – **Summary of inner nodes characteristics for the most common tree type for sole in 7D.** The domain of variation of each parameter is normalised between 0 and 1 ; 0.5 corresponding to the reference value, 0 to the reference value minus the chosen width of the explored window and 1 to the reference value plus the width of the explored window.

This main tree seems to be stable enough and a good starting point to grow subtrees so as to look for robust nodes. Subtrees grown from leaves of the main tree are in general more unstable than the main tree itself, probably because the number of model simulations available to build them is much smaller (equal the weight of the terminal leaf they correspond to). The number of different tree types for a given leaf is high and sometimes two or three different types have a rather similar number

Node	Parameter	Split value
1	FmsyFpaS7D	0.584
2	FmsyFpaS7D	0.241
4	MsyBtS7D	0.605
5	TransS7D	0.750
9	MsyBtS7D	0.568

Table 4.13 – **Summary of inner nodes characteristics for the base main tree for sole in 7D.**

of occurences. This is the reason why mean distance between subtrees is generally high (Table B 4.14).

Leaf	Number of different tree types	Mean distance between subtrees	Standard deviation
3	85	0.793	0.371
6	40	0.636	0.454
7	22	0.660	0.440
8	10	0.412	0.485
10	9	0.198	0.392
11	18	0.812	0.382

Table 4.14 – **Differences between subtrees for each leaf of the main tree.**

Every version of the subtree of a given leaf has an interesting characteristic : one terminal node among the N terminal nodes of the subtree has a much higher weight than any other node and is robust. As this node has a high weight, the information it provides can be considered as rather reliable. Besides, even if tree structure changes from subtree to subtree, the branch of the subtree leading to that particular node appears to be more stable than the subtree as a whole. So split values corresponding to that particular branch of the subtrees can be extracted and provide interesting information about conditions on states of Nature that allow reaching management goals, for various management measures.

When building a subtree from all training data belonging to a given leaf, this subtree does not always correspond to the most common subtree identified from the set of 500 subtrees built for that leaf. Besides, for some leaves, one single tree type does not clearly dominate in terms of number of occurences but instead a few tree types dominate. As many different tree types are available for a given leaf, we chose to focus only on tree types occuring in more than 10% of the total number of subtrees grown for a given leaf.

It is interesting to note that different branches often correspond to different combinations of split values for the same parameters. Branches can also be identical from one subtree type to another. In this case, the branch is only given once, even if it appears in various subtree types.

4.9 Supporting Information S3 : Building conditional trees for plaice in 7D

Tree instability is higher for plaice in area 7D. When simulating 500 trees based on management parameters, 36 different tree types can be identified and the most common type corresponds to only 157 trees. This higher instability is shown by the higher distance between trees (0.83) and standard deviation (0.35). The most common tree structure is shown in Table C 4.15. This tree has no robust leaves.

Node	Parameter	Mean split value	Standard deviation
1	FmsyFpaP7D	0.51	0.01
2	TransP7D	0.52	0.0001
3	PropSurvP7D	0.44	0.2
6	PropSurvP7D	0.44	0.01
9	PropSurvP7D	0.25	0.003
11	FmsyFpaP7D	0.82	0.03

Table 4.15 – **Summary of inner nodes characteristics for the most common tree type for plaice in 7D.** The domain of variation of each parameter is normalised between 0 and 1 ; 0.5 corresponding to the reference value, 0 to the reference value minus the chosen width of the explored window and 1 to the reference value plus the width of the explored window.

The main tree obtained from the entire training dataset is of the tree type identified as most common. Split values are close to the mean split values of the most common tree type, except for inner node 3 (where two very different splitting values coexist) where the split value is much higher than the mean value (Table C 4.16). This main tree was nonetheless selected to build subtrees.

Node	Parameter	Split value
1	FmsyFpaP7D	0.501
2	TransP7D	0.520
3	PropSurvP7D	0.759
6	PropSurvP7D	0.445
9	PropSurvP7D	0.251
11	FmsyFpaP7D	0.860

Table 4.16 – **Summary of inner nodes characteristics for the base main tree for plaice in 7D.**

Instability also is higher for subtrees built for plaice in 7D, with a very high number of different tree types and a high distance between trees (Table C 4.17).

For each leaf, the number of tree types is so high that sometimes no tree type occurs in more than 10% of cases. Therefore, for plaice in 7D, we chose to study the

Leaf	Number of different tree types	Mean distance between subtrees	Standard deviation
4	195	0.951	0.199
5	258	0.947	0.208
7	103	0.749	0.409
8	153	0.787	0.373
10	290	0.987	0.107
12	180	0.931	0.241
13	211	0.969	0.166

TABLE 4.17 – **Differences between subtrees for each leaf of the main tree for plaice in 7D.**

3 most common tree types for each leaf regardless of their number of occurrences among the 500 simulated trees. Similarly to what we did for sole in 7D, we focus on various versions of the branch leading to the node with highest weight. In this case, however, this node is sometimes not identified as robust, depending on the criterion used to define robustness (but it still contains an extremely high proportion of successes).

4.10 Autres méthodes de gestion du risque

On trouve des applications de de la théorie de la décision dans de nombreux domaines d'étude, depuis l'économie jusqu'aux sciences de la vie ou à l'ingénierie. C'est par exemple dans l'ingénierie que se sont développées les méthodes FORM (First Order Reliability Method) et SORM (Second Order Reliability Method) (220; 221; 222) qui permettent d'évaluer les contraintes auxquelles une structure peut résister. Cette approche pourrait être intéressante pour évaluer le risque qu'un écosystème a de connaître un changement de régime brutal, en fonction du niveau de pressions anthropiques et des variations naturelles. Néanmoins, cette approche est basée sur une approche probabiliste et nécessite de définir au moins deux fonctions de densité de probabilité : (i) une fonction décrivant le niveau de perturbation que l'écosystème peut supporter ; (ii) une fonction décrivant les perturbations appliquées à l'écosystème. Le niveau de risque correspond à l'aire de la zone où les deux courbes se superposent, et l'objectif de la méthode est de trouver le point dans l'espace des paramètres d'entrée ayant la plus forte probabilité de conduire à un échec (223). Le choix d'une fonction de densité de probabilité appropriée semble pouvoir être assez direct dans les cas simples, mais doit vite se complexifier et demander un haut niveau d'information à priori lorsqu'on tente de représenter un écosystème.

Un des avantages des méthodes telles que FORM et SORM est que le système d'intérêt est modélisé de manière très simple. En effet, il peut sembler paradoxal d'utiliser un modèle complexe dans une situation de forte incertitude. Ceci a été souligné par (98), qui met en avant le peu de sens d'une approche incluant explicitement un paramètre ou un processus mal connu dans un modèle. De ce fait, il est généralement recommandé d'utiliser des modèles simples dans des situations de forte incertitude. Nous avons tout de même fait le choix d'un modèle complexe car notre approche est exploratoire et vise à identifier les situations où notre manque de connaissances pourrait fortement impacter les sorties du modèle. De ce fait, il devient intéressant de modéliser même les processus mal connus car on peut déterminer si notre manque de connaissances sur ces processus serait dommageable ou pas.

Parmi les méthodes développées pour explorer des modèles complexes, la RSA (Regionalised Sensitivity Analysis) permet l'identification des valeurs de paramètres d'un modèle permettant un comportement acceptable du modèle et de celles ne le permettant pas. Ceci est réalisé au moyen de tests de Smirnov qui comparent $F_m(X_i|B)$ and $F_m(X_i|\bar{B})$. Néanmoins, (193) et (224) identifient plusieurs limites dans cette approche : (i) elle ne prend pas en compte les interactions entre paramètres, (ii) pour les modèles de grande taille, seule une petite fraction des résultats correspondent à des résultats acceptables (Behavioural : B), d'pù un manque de puissance statistique et (iii) la RSA ne permet pas d'identifier les paramètres importants qui ne causent des variations que dans la gamme des sorties du modèle acceptables. Lorsqu'on construit un modèle complexe prenant en compte de nombreux paramètres, un des objectifs principaux est de rechercher des propriétés émergentes résultant d'interactions entre parmètres. De ce fait, les méthodes ne faisant varier qu'un facteur à la fois (One at A Time : OAT) telles que la RSA ou la Sequential Bifurcation (SB), et ne prenant donc pas en compte les interactions, sont d'un intérêt limité dans notre cas. La méthode de Tree Structured Density Estimation (TSDE, (224)) correspondrait mieux à nos besoins. Elle permet en effet d'identifier les zones dans un espace de paramètres donnant les meilleures chances de succès. Le problème demeure d'appliquer cette méthode à un modèle complexe et de réussir à identifier et délimiter un sous-espace ne contenant que des réussites.

Chapitre 5

Test de scénarios de gestion spatialisés au moyen d'un modèle complexe de la Manche Est.

Sommaire

Can spatial management measures help robustly managing human activities in the eastern English Channel ?

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Abstract

In this article we build a complex, spatially explicit, model of two flatfish populations and three benthic groups in the eastern English Channel using the ISIS-Fish modelling platform. Fish species modelled are sole (*Solea solea*) and plaice (*Pleuronectes platessa*), two species undergoing high fishing mortalities in the eastern English Channel. Benthos is modelled as three groups gathering several taxa, based on their mobility. Fish populations interact with the fishing activity in the modelled area. Modelled fleets correspond to five different fishing gears with various boat sizes. Aggregates extraction is also modelled and only impacts benthos. Benthic groups are also impacted by fishing fleets that use towed gears. Simulations are run for twelve years to encompass a 2008-2020 period. 2020 corresponds to the year when good ecosystem state should be reached in E.U marine ecosystem according to the Marine Strategy Framework Directive. We model conventional management measures as well as possible spatial management scenarios in the Eastern English Channel so as to determine how these measures can help ecosystem recovery. Then the model behaviour is explored by means of sensitivity analysis techniques. Classification trees are built from these simulations to perform an uncertainty analysis that helps assess the robustness of our diagnostic to variability on values of the input parameters used. Results obtained suggest that a Harvest Control Rule can help the recovery of strongly depleted biomasses for both fish species modelled. The uncertainty analysis indicates that the current ecosystem state should allow reaching management goals defined for sole and plaice in this ecosystem in terms of spawning biomass. However, natural variability that can be tolerated on environmental parameters is low. We evidence no 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. On the contrary, 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.

5.1 Introduction

The main challenge marine management has to face is to reach a good ecosystem state by 2020, as defined by the Marine Strategy Framework Directive (MSFD (2)). This definition of a good ecosystem state is made of several definitions of good states that have to be reached at various levels and on various components of the ecosystem (5; 6; 4; 3)). This goal 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, it is known that fishing decreases the abundance of its target species. In the eastern English Channel this can be observed from the yearly stock assessments (190; 191; 172; 189; 207) that recommend actions to be taken in order to maintain target species biomass above precautionary levels. Fishing activities often use towed fishing gears, that are known to impact the substratum and benthic species living on it (72; 71; 73; 74; 27; 23). These species are important sources of food for several fish species living close to the bottom, in particular sole and plaice (225; 25; 26; 151). Therefore, if one wants to accurately predict the effects of fishing at the level of the ecosystem in an attempt to implement an ecosystem-based management(EBM) all these components of the ecosystem should be taken into account.

So far, management of flatfish populations in the English Channel is not at the ecosystem scale, as assessments by the ICES (International Council for the Exploration of the Sea) working groups are made at the stock level (one fish species for a given management area) and do not take interactions with other fish species or human activities other than fishing into account (207; 86). Conventional management of fishing activities is done by means of Total Allowable Catches (TACs) that are computed by means of a Harvest Control Rule (HCR) that adjusts the level of TAC so that Precautionary Approach (PA) points (in terms of biomass or fishing mortality) are not reached and fishing mortality decreases to that allowing to fish at the Maximum Sustainable Yield (MSY). The MSY is the maximum amount of fish that can be produced in a year by a targeted stock and caught without durably decreasing its biomass (86). It can be seen as a compromise between fish production, level of income and state of stocks as MSY is linked to biomass levels higher than precautionary points used in conventional fisheries management (86).

Human activities other than fishing may impact fish species and benthic taxa in this area. One of these activities, that is still developing in the Eastern English Channel, is aggregates extraction. Several articles (140; 77; 76; 141; 142; 143; 144; 147; 148; 24; 149) have shown that this activity has negative effects on the bottom structure and benthic species.

This activity 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 (92). In particular, benthic species seem to be strongly positively impacted when a no-catch or limited-access zone is created (90). These positive impacts on benthic species may have positive consequences on fish species because of trophic interactions between them. Several types of MPAs exist (89) and few of them aim at managing fishing or protecting fish resources. One of their asset is that they can serve to manage any human activity they overlap with, and not only fishing. MPAs are also seen as a tool to increase populations robustness by protecting parts of them from the detrimental effects of fishing.

Increasing populations robustness to uncertainty can also be done by means of management scenarios based on conventional management measures only. For instance (170) developed a method based on sensitivity analyses of a complex model and classification trees that allows to test a wide range of parameter values for the HCR and identify those that allow robustly reaching management goals. This analysis was performed on a simple model of the eastern English Channel representing impacts of the fishing activity on two flatfish populations (22). It showed that under certain conditions management scenarios based on conventional management measures allow 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 issue of this article is to determine whether MPAs would allow reaching management goals more easily and in a more robust way. Management goals mostly concern fish species so far but their definition can be widened to take into account definitions given by the MSFD. In this paper, we attempt to make a step towards a more ecosystem assessment of the effects of human activities at sea. We model 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 do this, we build a model of this area using the ISIS-Fish modelling platform. This model is based on that created by (22) that was used in (170). The model is improved to address the issue of the spatial management of mixed fisheries and better describe fish populations and fishing activities. The fishing activity is modelled at the level of the ICES statistical rectangle. Sixteen fleets corresponding to five different gears and various boat sizes are modelled, using information from (30). Another fleet is added to represent the aggregates extraction activity. Therefore, the consequences of spatial management measures can be tested and their local impacts on populations observed. Three management scenarios are tested. All use the current HCR as base management and test different rules concerning access to MPAs, from full access for all fishing gears to no fishing at all.

5.2 Material and Methods

The ISIS-Fish modelling platform (28; 29) 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. Models built with ISIS-Fish have the ability to make the 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 representing 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 (226; 14). A major feature of the ISIS-Fish model is that the fishing activity can be modelled 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 statistical rectangle. Therefore several métiers 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 metiers to obtain a total fishing mortality. This total 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 (226; 14) for more details).

We chose to focus on ICES area 7D, corresponding to the Eastern English Channel. Characteristics of this area as well as reasons why it is deemed important to model it can be found in (171; 170). Several changes were made to the model compared to that used in (170). 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 populations 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 (227) and on maps of preferential habitats from (33). 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, this is the reason why they are modelled as one population at the scale of the Eastern English Channel.

- The fishing activity corresponds to that used in (30) and is defined at the scale of the ICES statistical rectangle. It explicitly models sixteen fleets that correspond to four main fleets that are further divided based on boat length and the harbour they belong to. For a given main fleet different boat sizes and harbours correspond to different technical efficiencies and different métiers, i.e. different fishing areas given the fact that a main fleet is mostly defined based on the gears used. The four main 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 modelled as populations in ISIS-Fish (similar to a fish population) and correspond to three important mobility groups identified in (228) : 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 (229). Their mean abundance is computed as the mean abundance on reference stations in (228) and their accessibility calibrated in the model so that the loss of benthos on aggregates extraction sites is coherent with that computed in (228).
- We added a fleet corresponding to the suction dredges used for aggregates extraction in the Channel. This fleet is particular in the sense that it only fishes benthos ; it was parametrized to be consistent with other fleets modelled. 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. 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 in (72).

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

5.2.1 Modelling 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. (227). This allows us to define three sub-populations in the Eastern English Channel, each corresponding to a particular geographic area : (i) the Somme sub-population corresponds to that 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.

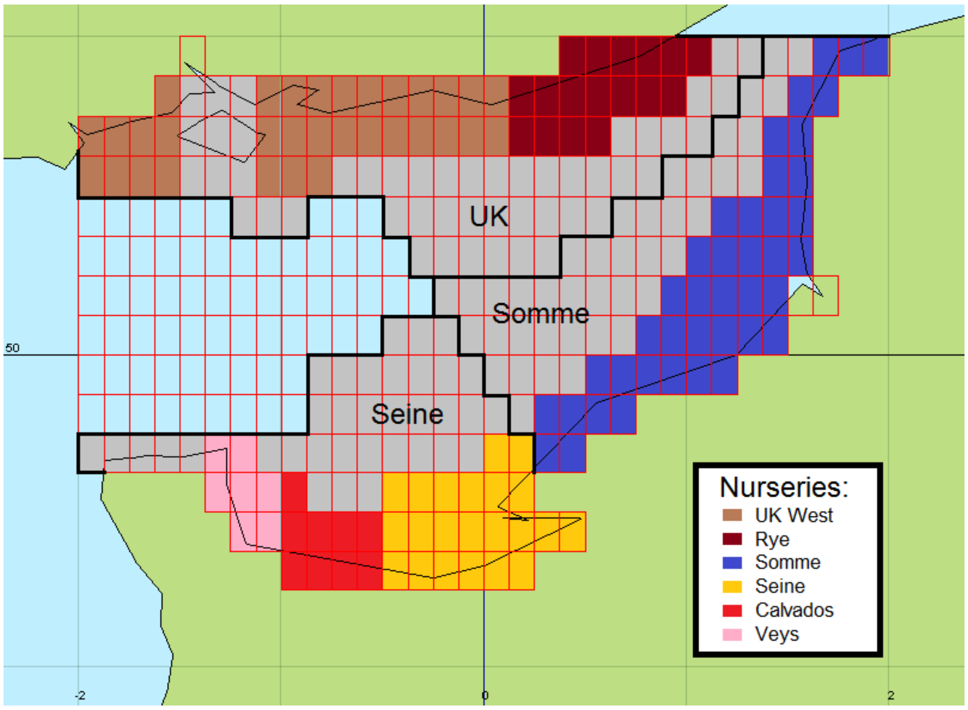

Figure 5.1 – **Map of nursery areas and population zones modelled.** Surfaces in grey in each of the three zones correspond to either a population or a reproduction area, or both as these two types of areas can overlap.

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 (33). There is a two-months gap between reproduction and recruitment on the nurseries to take into account the duration of the larval drift that is approximately 60 days in (227). Fish migrate to their reproduction area on the month before the beginning of reproduction and leave the reproduction area after reproduction. Sole leave nurseries to reproduction areas at age 3 and sole leave nurseries between age 2 and age 4, depending on the proportion of matures. 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.

5.2.2 Modelling benthic taxa

We could find no stock-recruitment relationships for benthos, especially at the level at which it is modelled 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 (230; 231; 232; 233; 234; 235), depending on factors such as taxonomic group, temperature, substratum, etc. It appears that Production over Biomass (P/B) ratios for several common groups are between 1 and 2. Our model covering an area wider than that at which studies are made and groups we chose to make gathering species with different production rates, we chose a rather moderate recruitment. Indeed, we assume that the 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 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 susceptible to impact benthos occur (i.e. aggregates extraction or fishing using towed gears) one zone corresponds to one model cell, so that local depletions can be modelled ; (ii) a last zone corresponds to all model cells where none of these activities occurs (for instance within the 3 nautic 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 forever in zones where it is not impacted by human activities. It also is 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 the "Granulats Havrais" area, that is the aggregates extraction area closest to the dredging site where measures studied in (228) were made.

5.2.3 Modelling human activities and management

The fishing activity sub-model is largely based on that built by (30). Relationships between a fishing fleet and a population in ISIS-Fish depend on several technical parameters (29; 226). Among these parameters are the selectivity, the target factor and a standardisation factor. We had to define these parameters for the additional aggregates extraction fleet and to add information to the existing ones so that benthos is impacted in a coherent way by the fishing activity. The target factor is defined at the métier (187) level and quantifies the strength with which the species is sought for by the métier (28). This target factor depends on species attractivity and on the savoir-faire of the fishers (29). 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 so as to “bypass” this term in the fishing mortality. 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 standardisation factor is used to standardise 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 extraction zones in the Eastern Channel : three on the French side and two on the English side (Fig. 5.2). Aggregates extraction zones being small those that are close to one another were grouped so as to model surfaces that are coherent with reality. Aggregates extraction is modelled by a fleet with a fishing strategy, similarly 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 behaviour is modelled 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), de default hypothesis being that both have equal weight in the decision-making process. Fishers reactions to management measures were also modelled. If a TAC on a species 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 goes 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 (170) the TAC level is set to its actual level for years 2008 to 2010, then it is computed by a Harvest Control Rule. The HCR decreases the TAC by at most 15% per year for five year to reduce fishing mortality applied to each population from its 2010 level to the F_{MSY} level. Other conditions apply so that the TAC can be further reduced if fishing mortality is above F_{pa} or biomass is below some trigger level. A minimum landing size also applies to fish populations.

The lower model resolution allows us to test for spatial management measures. Not all marine protected areas (Fig. 5.2) 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.

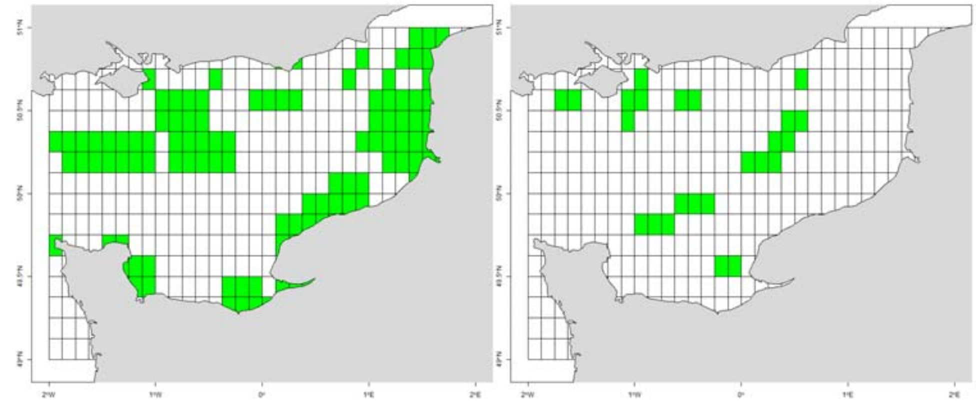

FIGURE 5.2 – **Maps of MPAs simulated in the model (left) and aggregates extraction areas (right).** The size, shape and position of these areas is constrained by the model structure and resolution.

5.2.4 Simulations

Simulations are run for twelve years, from 2008 to 2020 when a good ecosystem state should be reached.

Once scenarios were tested we performed a sensitivity analysis so as to determine how changes made to the model changed its behaviour. 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 in (170) 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 modelled. These parameters are those that had been identified as 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. 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 (170). 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.

5.3 Results

5.3.0.1 Fish species

Results obtained with the three management scenarios tested are rather similar in terms of values and identical in terms of trends (Fig. 5.3). 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.

Because of this increase due to management the spawning biomass at the end of the simulation if higher than 10000t for sole and higher than 5000t for plaice. This means that sole spawning biomass is well above the 8000t threshold recommended by the ICES WGNSSK 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 of 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 (170) 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 on the second year (Fig. 5.3). 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 stabilise. The fishing mortality stays almost constant once the optimal situation has been found by the gravity model. This stabilisation 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 modelling artefact 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 on 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 simulation for both species. This agrees 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 metiers, a metier 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 metiers 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. 5.4) 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 (185) (30). 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 that at the beginning of the simulation, and the fishing season only lasts 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.

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 on the "Somme" area than on the "UK" and the "Seine" area (Fig. 5.5). Indeed, sole spawning biomass decreases strongly for the first two years of the simulation on the "Somme" area (by more than 3000t per year for sole, 900t for plaice) whereas the decrease is very moderate on the other two areas. Therefore the behaviour of the Eastern Channel 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 behaviour 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 on the "Somme" area does not show on Fig. 5.5 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 on the "Somme" area increases back to its initial level when management measures are applied. Nonetheless this SSB still is 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 on 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 on the "Seine" area.

The evolution of the spawning biomass over the whole simulated period for the three zones can be summed-up by Fig. 5.6. 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.

5.3.0.2 Benthic species

In most model cells the effects of fishing and aggregates extraction appear to be low, recovery occurring 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. 5.7, 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 stabilises 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 on all the year when not managed. These peaks in effort leave some time to 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 modelled 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 modelled. 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 tend 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 5.4).

Consequences of spatial management measures on benthic taxa seem to be very straightforward in our model : 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. 5.7). 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. Only after the HCR begins and effort becomes very low for some of the year does the abundance of burrowers increase. 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 5.7. 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.

5.3.0.3 Sensitivity analyses

Results obtained are similar to those of (170) where more parameters of a simpler model were tested (Fig. 5.8. 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.

Globally, F_{MSY} mostly impacts fishing mortality and biomass but not the level of catch. This agrees 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.

5.3.0.4 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 over 500 (Fig. 5.9). Variables, mean split values and standard deviations corresponding to each node of this tree type are given in Table 5.1. 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 normalised 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 higher than the reference recruitment minus ten percent. Similarly, the age at maturity must not decrease more than 16.9% below the reference age at maturity of sole and the mean weight-at-age must not decrease more than 11.9% below the reference mean weight-at-age.

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 5.1 – **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 no 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 on input parameters values.

An interesting point is that if mean weight is too low to reach the robust node (15) 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 5.2 – **Inner nodes of the most common tree type, for plaice.** 0.5 corresponds to the reference model value.

5.4 Discussion

5.4.1 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 like 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. This in particular was observed here 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 modelling 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, that 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 very short-term decrease in catch. Then biomass increases and catches increase accordingly. The decrease in the fishing effort is associated to a change in its temporality, effort becoming increasingly concentrated at the beginning of the year. Indeed, biomass being high at the end of the simulated period these relatively low efforts are enough to reach the TAC in a few month. Nonetheless this pattern in catch seems little 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 altered a lot 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 bycatch species of sole in the modelled 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 attractivity (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' behaviour. 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 increase, that applied to sole increasing to much higher levels. 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 in (170) even though model complexity increased. The sensitivity analysis again puts to light the fact that management only has 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 parametrisation) 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 modelled accordingly. Therefore impacts of aggregates extraction appear 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 of the French side of the Channel. Indeed, efforts of aggregates extraction are much lower than fishing efforts. Consequences of fishing really show only in 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 very 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 slowing down the recovery process. This may also lead to increased vulnerability of the taxa and colonisation 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 to these rules, that there is no lag in time between decision and implementation of a rule, that they are not subject to political discussion, etc. In this ideal case it appears that management using TACs is efficient and manages to help 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 the in case of bat 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 metiers remains the same but is applied to parts of the metier area outside the closed area. Only if the metier area is fully within the closed area does the metier stop. The hypothesis being made in ISIS-Fish that fish species are mobile at the scale of their population area it is enough that one cell of a metier area overlaps one cell of a population area for the whole population in this area to be fully impacted by the metier. Therefore in our model Marine Protected Areas tend to modify the spatial structure of the effort, which may lead to "local" depletions if populations areas are small enough, but do not really modify the level of catch.

These effects would however be limited in our model in its current version because we seem to have underestimated benthos accessibility when calibrating our model. This is due to the fact that contrary to fish species few catch data is available for benthos, the impacted benthos not necessarily being caught in the fishing gear and the benthos caught being discarded. We therefore tried to calibrate benthos accessibility using data obtained from experimental dredging sites and linking extraction intensity to decreases in mean abundance (the method used is explained in more details 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 observations made in the field (23), even when taking into account the fact that not all the surface of an exploited zone is directly impacted. This calls for changes in the way relationships between benthos and human activities are modelled, and in particular an update of benthos accessibility in ISIS-Fish.

5.4.2 Modelling perspectives

Assets and limits of the method used have already been discussed in (170) and were clearly put to light in this modelling approach. Indeed, the model used is much more complex than that of (170) 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 less values for each parameter. We chose to make compromises on 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 behaviour 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 few parameters known to be important we chose not to split the input parameter space with the method of (170) 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 modelled and how they are modelled. Here we chose to group benthic taxa according to their swimming capacity because it seems to affect their productivity (229) and may also be linked to their sensitivity to fishing gears (for instance determining their ability to evade 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 (236) that seem to be good surrogates for species identification in community analyses.

MPAs should be modelled 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 so far 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 (237) allows optimizing the size and position of a management zone as well as the rules that apply in this zone. Attempts are currently being made (238) 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 (90; 92) 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 positive effects of MPAs on benthos cannot propagate to fish species.

Therefore it seems necessary to try to take these aspects into account. Our model structure may not be the best to take these pieces information into account, but other models have been developed for such tasks. We may be able to learn a lot by making our ISIS-Fish model interact the OSMOSE (239; 240) 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 modelled 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 a great added value in terms in terms of fisheries management as they would allow testing rather realistic measures and model their consequences at the ecosystem scale.

5.5 Acknowledgments

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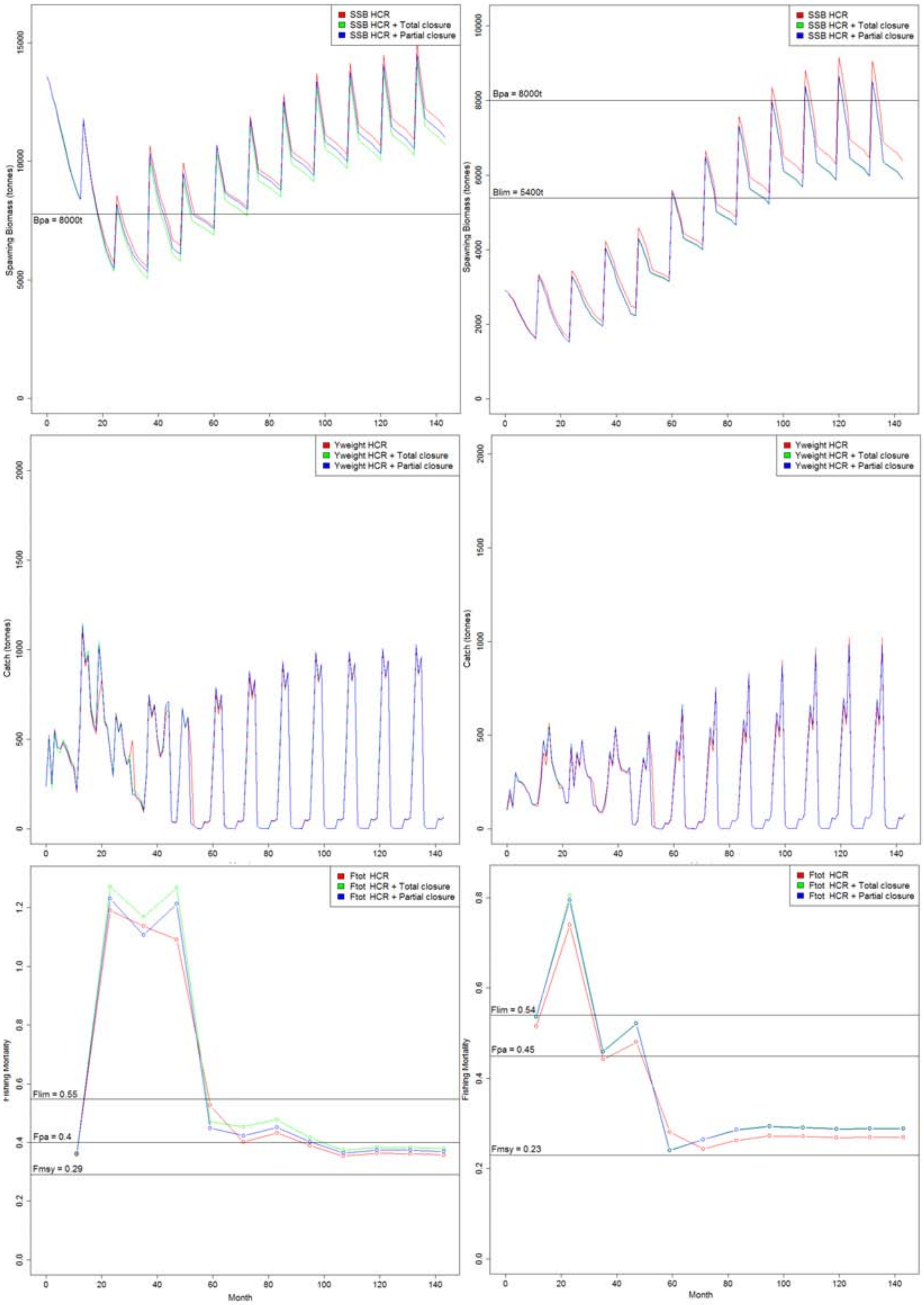

FIGURE 5.3 – **Total Spawning Stock Biomass (top), catch (middle) and fishing mortality (bottom) for sole (left) and plaice (right) in the Eastern English Channel.**

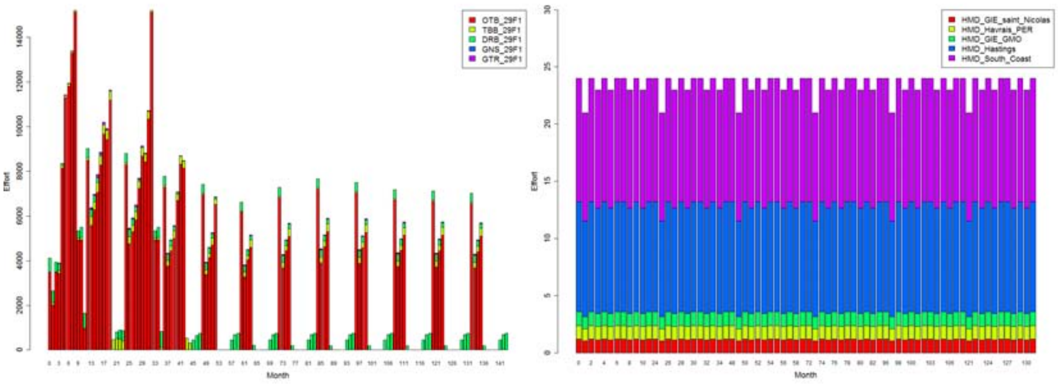

FIGURE 5.4 – **Fishing Effort per gear in ICES rectangle 29F1 (left) and extraction effort on the various aggregates extraction areas (right).** The fishing effort is at the scale of a whole ICES statistical rectangle whereas the extraction effort is at the scale of the surface of the extraction areas. Rectangle 29F1 is one of the most heavily fished rectangles in the Eastern English Channel.

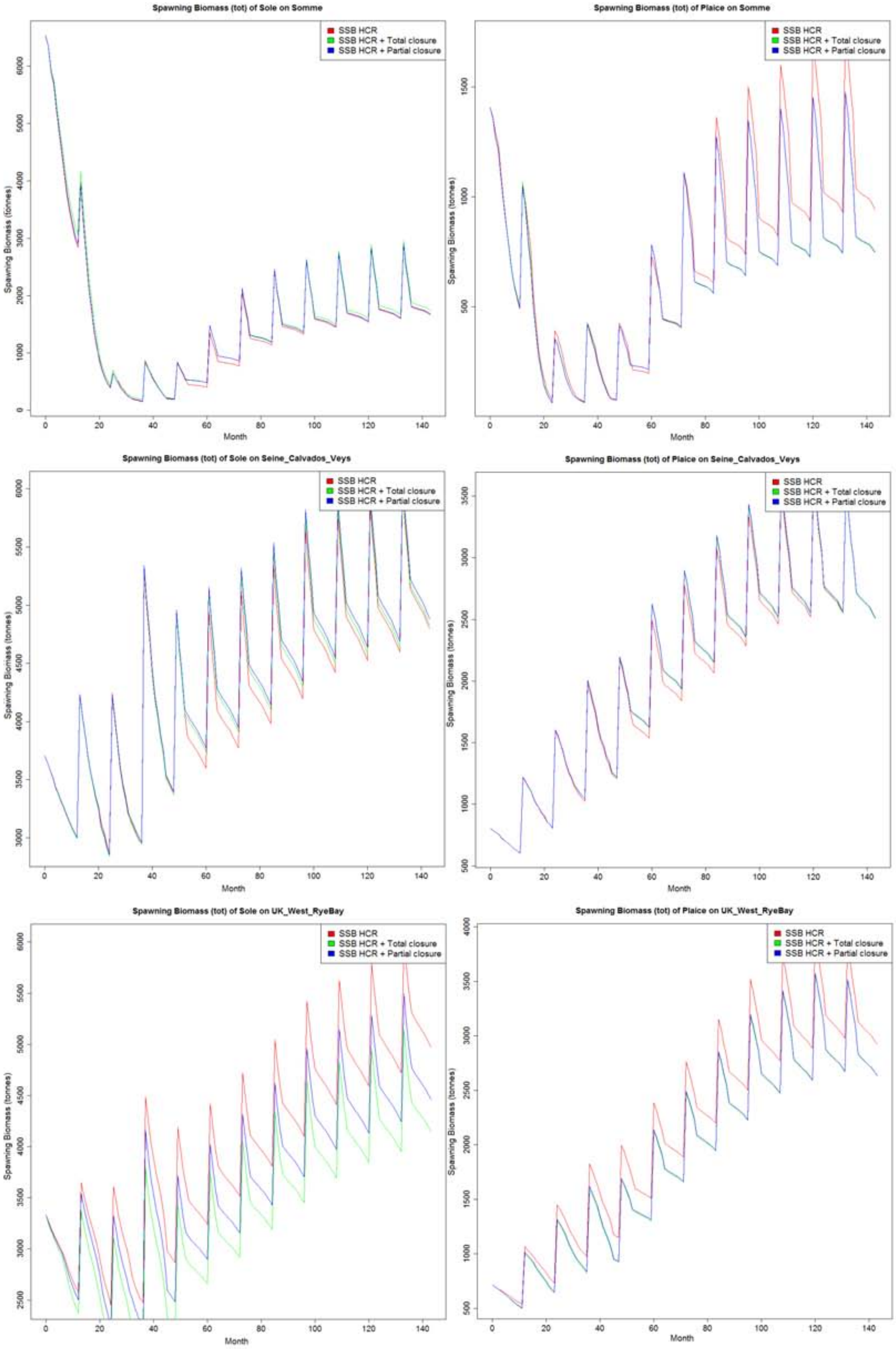

FIGURE 5.5 – **Spawning Stock Biomass for sole (left) and plaice (right).** For each of the three modelled sub-areas : “Somme” (top), “Seine” (middle) and “UK” (bottom).

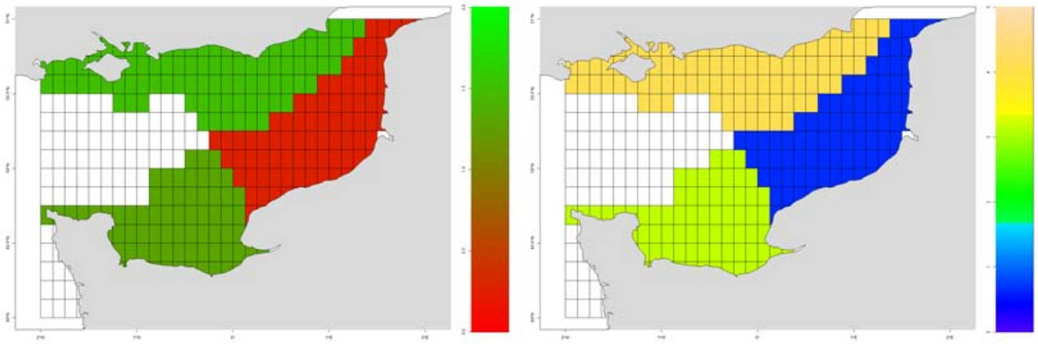

FIGURE 5.6 – **Maps summarising the evolution of the spawning biomass of sole (left) and plaice (right) in the three studied sub-areas.** Values given are a ratio between the SSB for the last month and that of the first month of the simulation.

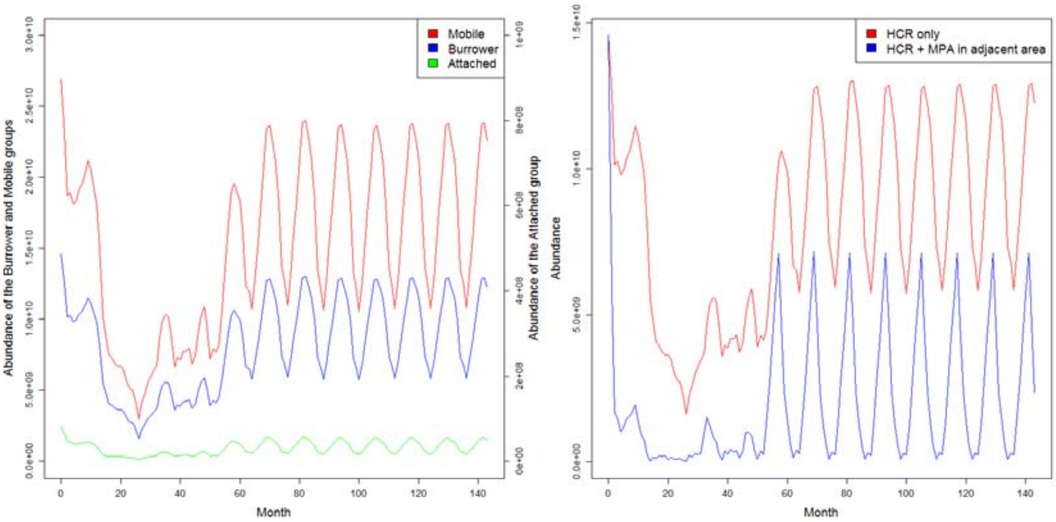

FIGURE 5.7 – **Abundances of the three modelled benthic taxa in a strongly fished area (left) and abundance of burrowers in a model cell adjacent to a possible MPA (right).**

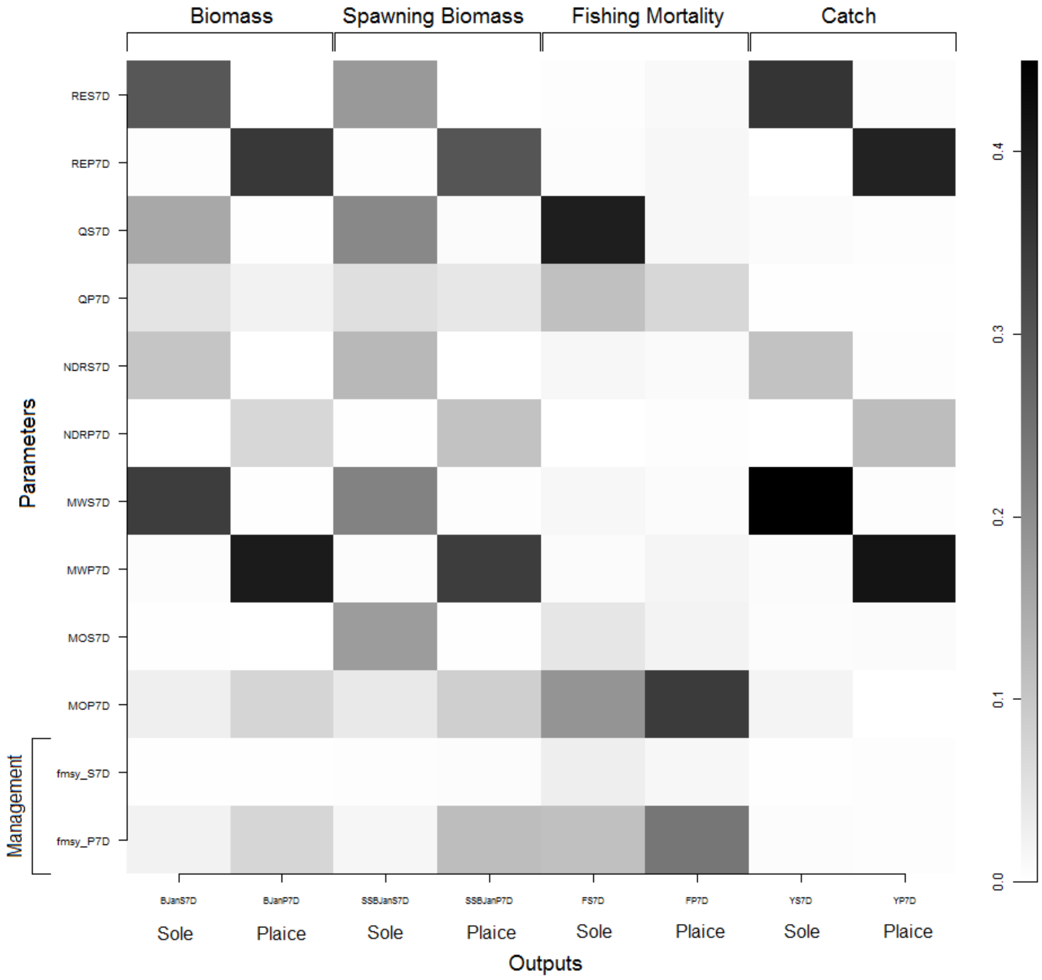

FIGURE 5.8 – **Sensitivity analysis results for scenario fishing + HCR.** Tiles in black correspond to parameters most impacting the variance of the output variables. Results have to be studied vertically, columns sum to one.

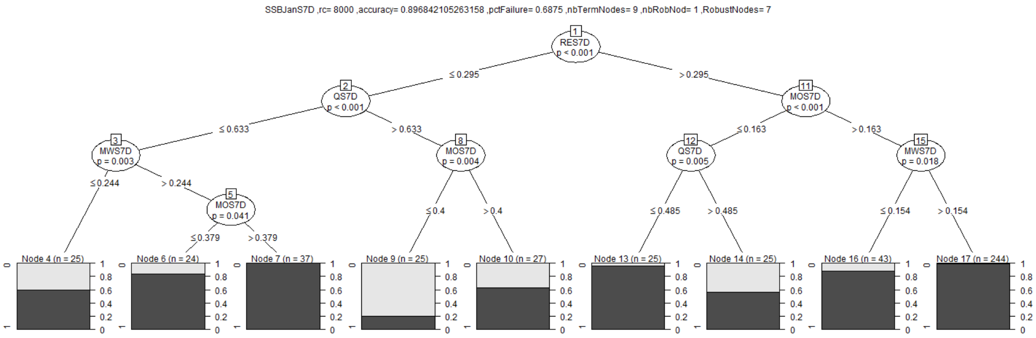

FIGURE 5.9 – **Classification tree corresponding to the most common tree structure, for sole.** Split values may differ from average split values computed from the 172 trees of that type.

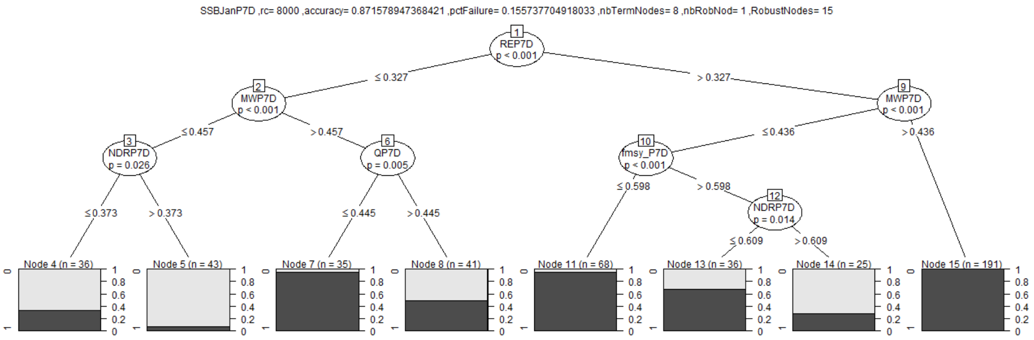

FIGURE 5.10 – **Classification tree corresponding to the most common tree structure, for plaice.** Split values may differ from average split values computed from the 172 trees of that type.

5.6 Supporting Information S1 : 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 (187) level and quantifies the strength with which the species is sought for by the métier (28). This target factor depends on species attractivity and on the savoir-faire of the fishers (29). 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 is attributed to each population caught. As we have three populations of benthos, one selectivity had to be defined for each of them. This is somewhat problematic as benthic groups we defined 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 (23). 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.

Standardisation is another parameter defined at the gear level. One gear is chosen as the standard gear and given the value 1. Standardisations for other gears are defined relative to that standard gear. The standardisation 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 standardisation 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 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 (241). This allows us to make the hypothesis that the volume extracted at the scale of the Eastern English Channel is an acceptable proxi 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 (242; 47; 45; 43; 44; 46; 48). 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 (51; 243). 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 Channel (244; 245; 246; 247; 248; 249). 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 influences a lot the level of catch of a species is the accessibility parameter in ISIS-Fish. It is a biological parameter that is very close in reality to an accessibility parameter. 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 (228) 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 can seem to be quite high but seems to be coherent as it was obtained from 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 km^2 over 80 km^2 were dredged in 2007, and 26 km^2 over 220 km^2 for the South Coast extraction area (51). 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 the "Granulats Havrais" area, that is the aggregates extraction area closest to the dredging site where measures studied in (228) were made.

CHAPITRE 6

Conclusion

Sommaire

6.1 Chronologie des travaux réalisés et résultats marquants

L'objectif de cette thèse était le développement et l'application d'une méthode de prise en compte des incertitudes dans les modèles complexes d'écosystèmes marins. Le développement de cette méthode devait s'inscrire dans le contexte plus large de l'atteinte du bon état écologique des écosystèmes et de la gestion écosystémique. Cette méthode devait en particulier permettre de tester divers scénarios de gestion à l'échelle de la Manche Orientale afin d'identifier ceux permettant d'atteindre les objectifs de gestion avec la plus grande robustesse possible.

Ce contexte global a imposé un certain nombre d'objectifs “intermédiaires” devant être remplis afin de répondre à la question principale. La recherche de mesures de gestion plus robustes passant en particulier par la simulation de mesures de gestion spatialisées, il a fallu développer un modèle représentant explicitement dans l'espace les populations de poissons et les activités de pêche. Les objectifs étant d'ordre écosystémique, d'autres populations et activités susceptibles d'avoir des impacts notables à l'échelle de l'écosystème devaient être prises en compte. Elles devaient dans la mesure du possible être décrites avec un niveau de détail similaire aux populations et activités déjà décrites.

Cette tentative de prise en compte des incertitudes s'inscrivait dans le cadre du projet VECTORS, qui vise à comprendre les vecteurs de changement à l'échelle de tout l'écosystème de la Manche Est. C'est dans ce cadre que les analyses réalisées par l'université de Rouen et le CSLN ont permis de mettre en avant le fait que l'extraction de granulats peut avoir des effets négatifs notables sur les populations de benthos. Or, l'extraction de granulats est à l'heure actuelle une activité relativement limitée dans l'espace et ayant des niveaux d'effort, en terme de temps passé et de surface de fond couverte, sans commune mesure avec ceux de la pêche. Il est alors apparu que quantifier les effets conjoints de ces deux activités sur le benthos pouvait se révéler intéressant et prospectif dans le cadre d'une extension probable de cette activité.

Il est ressorti de l'étude bibliographique portant sur les effets de l'extraction de granulats que ce sujet d'étude est globalement plus récent que l'étude des effets de la pêche. Il est aussi apparu que l'information disponible dans ces articles est relativement parcellaire et peu adaptée pour être utilisée dans un modèle comme ISIS-Fish. En effet beaucoup de données sont issues d'observations avant/après le passage d'une drague, à très petite échelle. Lorsque des informations sont données sur le temps de récupération des zones draguées, il apparait que celui-ci est très variable et corrélé à la nature du sédiment dans la plupart des cas. Afin d'inclure l'extraction de granulats dans un modèle, nous avions besoin de comprendre les causes de la baisse d'abondance de benthos suite au passage d'une drague, et de quantifier le lien entre intensité d'extraction et impacts sur le benthos. Afin d'accéder à ces résultats il a été décider de procéder à une nouvelle analyse des données de l'université de Rouen, recueillies en partenariat avec l'association CSLN (Cellule de

Suivi du Littoral Normand), qui avaient déjà fait l'objet d'un rapport courant 2012 (79).

6.1.1 Effets de l'extraction de granulats sur le benthos

Cette nouvelle analyse a permis de mettre en évidence des **effets négatifs de l'extraction sur certains groupes de benthos**, en fonction de leur type d'alimentation ou de leur mode de déplacement. On met en particulier en évidence une baisse significative de l'abondance du benthos enfoui et attaché sous les effets de l'extraction de granulats. Trois variables étudiées apparaissent comme ayant des effets négatifs sur l'abondance de benthos : le temps d'extraction cumulé sur toute la période d'extraction, la durée de cette période d'extraction et le nombre d'heures d'extraction moyen par mois. En revanche, aucune récupération des groupes impactés négativement par l'extraction n'a pu être observée une fois celle-ci terminée. Les tendances positives observées dans l'abondance de certains des groupes étudiés dans les analyses exploratoires n'ont en effet pas pu être reliées à une réelle récupération et semblent dues à des tendances annuelles non corrélées avec l'extraction. On observe en revanche pour certains groupes comme les détritivores des abondances élevées juste après extraction et qui décroissent rapidement. Ces groupes semblent donc présenter un caractère opportuniste. Suite à cette étude les populations de benthos ont été ajoutées à notre modèle de la Manche Est.

6.1.2 Sensibilité et Robustesse

Le modèle ayant connu deux versions successives, deux analyses de sensibilité ont été réalisées. 81 paramètres ont pu être testés dans la première analyse de sensibilité. Ces paramètres peuvent être séparés en trois groupes : les paramètres biologiques, qui définissent les traites de vie des espèces modélisées, les paramètres techniques permettant de représenter les activités de pêche et enfin les paramètres des mesures de gestion. Cette première analyse révèle que **la dynamique des populations représentées dépend essentiellement des paramètres biologiques** tels que le recrutement, le poids moyen aux âges, ou encore le taux de mortalité naturelle. Les paramètres techniques ont des effets plus modérés. Viennent enfin les paramètres des mesures de gestion, dont les effets sont peu perceptibles dans cette analyse. Une exception concerne la valeur de mortalité par pêche ciblée par la gestion qui apparait comme étant le paramètre de gestion le plus important, malgré ses effets très modestes en comparaison des paramètres cités précédemment.

Une analyse d'incertitude a ensuite été réalisée sur les résultats des simulations réalisées dans l'analyse d'incertitude. Ces premiers résultats ont notamment permis de déterminer que **les mesures de gestion conventionnelles (TAC) seules ne sont pas robustes aux incertitudes**. Néanmoins il apparait que seul un petit nombre de paramètres de l'écosystème déterminent l'atteinte ou non des objectifs de gestion, ce qui devrait être une incitation forte à l'acquisition de plus de connaissances sur ces paramètres. Nous avons en effet montré que seul le poids moyen aux âges et le recrutement contraignent réellement la gestion de la sole en zone 7D. Nous avons également mis en évidence le fait que les règles de gestion actuelles devraient permettre d'atteindre les objectifs de gestion à l'horizon 2018 si le recrutement et le poids moyen ne sortent pas d'une certaine gamme de valeurs.

L'analyse des résultats du modèle final a été orientée par celle des premiers résultats obtenus avec sa version plus simple. En effet, les temps de simulation ayant fortement augmenté au cours de la complexification du modèle il était impossible matériellement de réaliser une analyse aussi complète que précédemment. Le choix a donc été fait de se focaliser sur les paramètres identifiés comme importants dans la première analyse afin de déterminer si la hausse de la complexité du modèle avait affecté leur influence sur ce dernier. Il apparait que ce n'est pas le cas, les paramètres les plus importants restant les mêmes et leur ordre d'importance n'évoluant pas.

Les résultats du découpage de l'espace des paramètres d'entrée du modèle au moyen d'arbres de régression sont également consistants avec ceux obtenus précédemment. Il apparait en particulier que **les objectifs de gestion peuvent être atteints à l'horizon 2020**, tant que les conditions environnementales ne sortent pas d'une certaine gamme. Cette gamme de valeurs tolérables est par ailleurs beaucoup plus restreinte pour la plie que pour la sole. Ceci est lié à la biomasse globalement plus faible de la plie dans cet écosystème.

6.1.3 Intérêts de la spatialisation du modèle

Le développement du modèle a permis de prendre en compte explicitement la dynamique spatiale des populations modélisées. Cette spatialisation du modèle s'est basée principalement sur les matrices de connectivité entre zones de reproduction et nourriceries côtières (227). Ces matrices montrent que trois zones indépendantes peuvent être identifiées en Manche Est entre lesquelles les échanges de juvéniles sont quasiment nuls. Ainsi, trois sous-ensembles de zones ont été définis en Manche Orientale, chacun correspondant à une zone de reproduction, une zone de "population" pour les adultes hors reproduction et une ou plusieurs nourriceries. ces zones correspondent grossièrement à la baie de Somme, à la baie de Seine et à la côte anglaise. Les migrations des populations sont définies de sorte à reproduire les migrations décrites dans la littérature et les ogives de maturité utilisées par les groupes d'évaluation. La dynamique de l'effort de pêche a également été spatialisée en se basant sur une analyse récente (30). Cette spatialisation du modèle permet de représenter les effets de la pêche sur les populations à une échelle inférieure à celle de toute la Manche Est.

Le modèle montre en particulier une **très forte décroissance des populations de sole et de plie en baie de Somme sous l'effet de la pêche**, au début de la série temporelle modélisée. Cette décroissance semble résulter directement des forts efforts de pêche appliqués à cette zone. **Les effets des mesures de gestion spatialisées sont en revanche nuls ou négatifs sur les populations de poissons** modélisées, par rapport à la gestion actuelle non spatialisée. Ceci vient du fait que les efforts de pêche se reportent en intégralité sur les zones non affectées par les mesures de gestion. De ce fait on peut observer des impacts locaux potentiellement plus marqués mais l'impact global sur les populations de poissons reste de même. Les effets de ces reports d'effort peuvent en revanche être observés sur les groupes de benthos modélisés car ceux-ci sont représentés à des échelles très fines et ne sont pas mobiles à l'échelle d'une cellule du modèle, contrairement aux populations de poissons. Il apparait alors que **les reports d'effort peuvent être la cause de fortes baisses d'abondance de benthos** dans certaines cellules du modèle. Ce phénomène est assez extrême et n'est observé que dans un petit nombre de cellules appartenant à un rectangle statistique subissant de forts efforts de pêche et étant très largement fermé lors de la mise en place des mesures de gestion spatialisées dans le modèle. Cela conduit à des reports d'effort particulièrement marqués. Lorsque l'effort de pêche se réduit du fait de la gestion on observe que les populations de benthos se maintiennent dans un état stable, avec une forte variation de leur abondance au cours de l'année, du fait de leur forte productivité. Dans les cas où le report d'effort entraîne une hausse locale de ce dernier l'abondance de benthos se stabilise à des niveaux plus faibles que lorsque l'HCR seule est appliquée. Des différences notables apparaissent entre ces deux scénarios en début de simulation. En effet, l'abondance de benthos diminue jusqu'à des niveaux proches de zéro, ce qui serait susceptible d'ouvrir la porte à des modifications voire à des shifts dans la nature des communautés qui ne sont pas modélisés ici. **Les effets de l'extraction de granulats ne ressortent presque pas de l'analyse aux échelles modélisées**. Ceci peut être expliqué par le fait que cette activité s'effectue sur de petites surfaces, inférieures à une cellule de notre modèle. De ce fait, d'éventuels effets locaux ne peuvent être observés, et ceux-ci se retrouvent vite gommés aux échelles représentées par le modèle.

6.2 Obstacles à la mise en place de l'approche

6.2.1 Hétérogénéités

Deux types d'hétérogénéités ont dû être pris en compte dans le modèle : d'une part des hétérogénéités dans les connaissances disponibles et d'autre part des hétérogénéités spatiales, potentiellement liées au manque de connaissances. Ceci a imposé diverses contraintes dans la réalisation du modèle et nécessité de faire un certain nombre d'hypothèses.

Une première contrainte est que les espèces de benthos sont beaucoup trop nombreuses, trop diverses et trop mal connues pour être représentées dans le modèle à ce niveau taxonomique. De ce fait, les groupes de benthos sont décrits assez grossièrement dans notre modèle. Ces groupes sont définis selon leur niveau de mobilité et peuvent donc regrouper des taxons aux traits de vie potentiellement très différents. De ce fait, seule une dynamique globale (et donc imprécise) peut leur être appliquée. De plus, on définit une abondance de benthos uniforme à l'échelle de toute la Manche. Cette hypothèse simple résulte de l'extrapolation de données issues d'une étude locale n'ayant pas permis d'appréhender la variabilité des populations benthiques à l'échelle de la Manche. Cette étude a de plus été réalisée sur une zone de profondeur relativement constante. L'abondance du benthos et sa composition spécifique étant connues pour être hétérogènes spatialement il est très probable que nous sous-estimions l'abondance de benthos (et sa productivité) dans certaines zones et la surestimions dans d'autres. Une manière d'améliorer la représentation du benthos dans le modèle pourrait être de faire des regroupements par morpho-espèces (236) plutôt que par guilde trophique ou type de déplacement.

Les zones de nourricerie et de reproduction des espèces modélisées ont été définies à partir de cartes de connectivité. En revanche la zone de "population" peuplée par les populations adultes en dehors des périodes de reproduction est moins bien connue. Nous avons donc fait face à cette hétérogénéité du niveau de connaissance des populations en définissant la position et la taille de cette zone à partir de la littérature. La position de cette zone est donc plus imprécise et ne recoupe que partiellement la zone de reproduction. Les migrations entre ces deux zones étant supposées totales dans le modèle, une fraction de chacune des zones modélisées peut se retrouver privée d'une ou des deux espèces représentées pendant une partie de l'année. De ce fait, une partie de l'effort de pêche est susceptible de s'appliquer à une zone sans poisson, ou en tout cas sans son espèce cible. Ceci explique certaines pertes d'effort observées dans le modèle. Même si ces phénomènes sont marginaux et ne changent pas le diagnostic posé sur l'écosystème ils sont le révélateur de comportements du modèle qui pourraient être améliorés. Il est en effet peu réaliste que les pêcheurs exercent leur effort sur une zone temporairement privée de poisson.

Ce mismatch entre activité de pêche et populations peut aussi s'expliquer par les hétérogénéités d'échelle entre la description des populations de poissons et celle des activités de pêche. En effet, l'information concernant ces dernières n'est disponible qu'au niveau du rectangle statistique, tandis que les populations de poisson sont décrites a des échelles beaucoup plus fines. De ce fait, même en définissant parfaitement les zones de population et les zones de pêche avec l'information disponible, on est susceptibles d'appliquer de l'effort de pêche à des parties de rectangle statistique où aucun effort n'a en fait lieu.

Les cartes utilisées pour définir la position des zones de reproduction et de nourricerie sont celles de la sole. La matrice de connectivité liant ces zones entre elles est également celle obtenue pour la sole, même pour la population de plie. Ceci illustre des hétérogénéités dans la quantité d'information disponible pour chacune des deux espèces représentées. De ce fait la position des zones correspondant à la plie est susceptible d'évoluer si on utilise les données de dérive larvaire de cette espèce. Ceci peut faire évoluer sensiblement les résultats obtenus concernant la population de plie.

6.2.2 Représentation du comportement des pêcheurs

Une partie importante des résultats obtenus semble résulter directement de la manière dont est décrit le comportement des pêcheurs, que ce soit dans le choix de l'activité à pratiquer ou dans la réaction face aux mesures de gestion.

Le choix de la proportion de chaque métier au sein de la stratégie adoptée par un pêcheur est déterminé chaque année en utilisant un modèle de gravité. Ce modèle détermine le choix des métiers en prenant en compte d'une part la VPUE de chaque métier pour l'année précédente et d'autre part leurs habitudes (la proportion des métiers pratiqués l'année précédente). Le poids de chacun de ces paramètres dans la décision prise dépend d'un paramètre α variant entre 0 et 1. A zero la décision va dépendre uniquement des habitudes et la stratégie pratiquée demeurer la même pendant toute la simulation. A un la répartition des métiers dans la stratégie dépendra uniquement de la VPUE de chacun l'année précédente. La valeur choisie par défaut est 0.5, la décision repose donc à la fois sur la VPUE de l'année précédente et la stratégie pratiquée cette année là. La VPUE étant une valeur, elle est calculée à partir du prix de vente du poisson. Or, ce dernier est défini de manière constante dans le modèle, ce qui va limiter les variations de la VPUE et donc du choix des métiers.

Le comportement des pêcheurs face aux mesures de gestion dépend des espèces qu'ils ciblent. Si le TAC est atteint pour une espèce étant l'espèce principale ciblée par un métier alors ce métier s'arrête. Par contre si cette espèce est secondaire pour ce métier et que le TAC n'est pas atteint pour sa cible principale alors le métier va continuer à pêcher les deux espèces mais rejeter l'espèce secondaire pour laquelle le TAC est atteint. ceci est une des raisons pour lesquelles il apparait que la mortalité par pêche se stabilise légèrement au dessus de la valeur cible en fin de simulation.

Lorsqu'une aire marine protégée est mise en place l'effort d'un métier utilisant un engin interdit dans cette zone ne devient nul que dans le cas ou la zone métier est intégralement contenue dans l'AMP. Dans le cas contraire l'effort est reporté dans son intégralité sur la partie de la zone métier hors de l'AMP. Cela peut conduire à des reports d'effort très marqués aux conséquences visibles dans le modèle. Une meilleure prise en compte des coûts de la pêche et des revenus générés permettrait probablement une meilleure représentation de la dynamique de l'effort de pêche. En effet, ici un métier n'est jamais arrêté s'il n'est plus ou pas assez profitable. Ce type de situation est susceptible de se produire dans le cas où une AMP reporterait un effort dans une zone trop éloignée, de trop petite taille ou plus simplement sur une portion d'une zone métier ne se superposant plus à la zone de présence de la population ciblée. Sans aller jusqu'à un arrêt total d'un métier en réaction à la gestion, la quantité d'effort alloué à ce métier pourrait être ajustée dynamiquement en fonction des mesures de gestion. Une telle modification du modèle serait en particulier visible sur les populations de benthos subissent de plein fouet les reports d'effort à de petites échelles du fait de leur absence de mobilité.

Une meilleure connaissance du comportement des pêcheurs, de leurs motivations et habitudes permettrait de représenter ceux-ci de manière plus réaliste et moins tranchée. Ceci aurait pour conséquence de limiter les modifications brusques et massives de l'activité de pêche qui peuvent survenir dans le modèle mais ne sont que peu probables en réalité.

6.2.3 Complexité

La gestion par TACs et l'objectif du RMD, bien que parfois contestés (87), on contribué à la récupération de certains stocks subissant des pressions de pêche trop fortes (250). Toutefois, les échecs de cette gestion montrent qu'elle n'est pas suffisante dans certains cas. C'est la raison pour laquelle une approche écosystémique des pêches est promue au niveau international depuis près de 40 ans (1). Ceci impose en parallèle une évolution des modèles utilisés pour acquérir de la connaissance sur cette activité.

L'augmentation de la complexité du modèle est une conséquence directe de nos besoins de modélisation. Elle nous a permis de représenter les populations benthiques jusqu'ici absentes du modèle et de spatialiser la représentation des populations et des activités humaines, tout en les représentant plus en détail. Nous avons ainsi pu tester des mesures de gestion spatialisées en supplément de la gestion conventionnelle, et observer leurs conséquences à l'échelle des grandes baies de la Manche Est. Cette complexification du modèle apparait donc comme une nécessité si on souhaite se diriger vers une gestion à l'échelle des écosystèmes.

La force des modèles complexes est de pouvoir mettre en évidence des processus émergents (251; 252). Les caractéristiques émergentes, c'est à dire qui sont plus que la somme de leurs parties, des écosystèmes sont le fruit des interactions entre différents paramètres qui définissent ces derniers. Elles sont donc par nature impossibles à prédire avec des modèles simples, ce qui est une incitation supplémentaire à se diriger vers des modèles plus complexes.

Pousser le modèle vers un degré de réalisme supérieur implique toutefois une hausse extrêmement rapide des temps de simulation, comme cela peut être observé avec le second modèle créé. Les processus simulés sont pourtant fondamentalement les mêmes que dans le premier modèle, la structure du modèle n'ayant pas changé ; ils sont simplement décrits de manière plus complète et plus fine. Ce modèle ne représente malgré tout qu'une fraction de la complexité de l'écosystème étudié, et seul le couplage avec d'autres modèle semble pouvoir permettre de commencer à aborder réellement le fonctionnement de cet écosystème. Le problème est que cet ensemble de modèles risque d'être une complexité telle que qu'il sera impossible à utiliser en l'état actuel de nos capacités de calcul.

Les solutions communément proposée pour réduire ces temps de simulation sont de réduire le nombre de paramètres étudiés, en se focalisant par exemple sur les paramètres identifiés comme étant les plus sensibles. Le problème est que les techniques d'analyse de sensibilité permettant l'identification des paramètres importants d'un modèle nécessitent de faire un grand nombre de simulations avec ce modèle, afin de pouvoir explorer l'espace des paramètres d'entrée de manière suffisamment fine et suffisamment large à la fois. Le risque demeure donc de ne pouvoir appliquer dans les faits ces méthodes qu'à des modèles suffisamment simples.

Il ressort de nos modèles successifs de la Manche Est que la manière dont est construit et exploité le modèle peut influer de manière non négligeable sur les temps de simulation. Le modèle étant spatialement explicite il apparait également que le nombre de zones distinctes représentées dans le modèle doit se limiter à un minimum. Ainsi, il est possible d'agréger en une zone unique toutes les zones ayant des caractéristiques similaires afin de limiter le nombre d'itérations réalisées par le modèle. Cela impose une prise de recul par rapport au modèle créé afin de faire le lien entre les processus biologiques que l'on souhaite représenter et la manière dont ils peuvent être représentés dans le modèle afin d'identifier des points pouvant être simplifiés. L'augmentation de la complexité des modèles doit donc être associée de manière croissante à la mise en place de bonnes pratiques de modélisation permettant d'éliminer tout élément inutile pour répondre aux questions posées, et à une rigueur croissante dans le développement et l'utilisation des modèles.

6.3 Perspectives

6.3.1 Evolutions Méhodologiques

Les méthodes d'analyse de sensibilité sont associées à des méthodes efficaces d'exploration de l'espace des paramètres d'entrée d'un modèle, permettant de ne pas réaliser de simulations inutiles pour l'exploration du modèle. Dans les cas des modèles complexes les temps de simulation élevés empêchent la plupart du temps de réaliser un grand nombre de simulations. L'espace des paramètres d'entrée est donc exploré de manière très lacunaire du fait du grand nombre de dimensions à explorer. Des approches existent également où ce n'est pas le modèle qui est exploré mais un méta-modèle plus simple. Utiliser ce modèle plus simple permet de réaliser un plus grand nombre de simulations, et donc une meilleure exploration de l'espace. La création d'un méta-modèle implique toutefois une bonne connaissance du modèle original et donc une exploration préalable de ce dernier. Une autre piste est construire d'un méta-modèle représentant à la fois le comportement moyen du modèle mais également sa variabilité (253). Dans tous ces cas on se réduit toutefois à explorer une portion plus ou moins grande de l'espace des paramètres d'entrée du modèle, puis à tenter de découper celle-ci en différentes zones en fonction des objectifs fixés sur les valeurs des sorties du modèle. Il apparait lors de ces découpages, réalisés ici au moyen d'arbres de régression conditionnels, qu'en réalité seules les frontières entre les zones acceptables et celles non acceptables présentent un intérêt. Le reste de l'exploration, correspondant en réalité à l'essentiel des points simulés, a donc été réalisé à pure perte mais on ne pouvait pas le savoir à priori.

Une solution afin de limiter les temps de simulation pourrait donc être de coupler les étapes d'exploration de l'espace et d'identification des frontières, afin de se focaliser sur ces dernières. Cela passerait par exemple par l'utilisation de méthodes d'échantillonnage, soit aléatoires soit plus ou moins contraintes afin de suivre des gradients, permettant d'explorer l'espace des paramètres d'entrée par itérations successives et comparaison avec une valeur cible à atteindre sur les variables de sortie étudiées. De telles méthodes, qui ne sont pas sans rappeler la méthode de Monte-Carlo par chaînes de Markov (MCMC), permettraient ainsi dans un premier temps de progresser pas à pas vers un point de la frontière de modèle. Cette première recherche d'un point de la frontière peut se faire de manière grossière. Une fois un élément de la frontière trouvé on peut penser adapter la méthode d'échantillonnage (méthode type adaptative MCMC) afin d'explorer plus finement la zone proche de ce point de la frontière afin de trouver un autre point de celle-ci. Ainsi par itération on devrait arriver à reconstituer les frontières entre les zones de l'espace d'entrée des paramètres du modèle sans avoir eu à explorer en détail l'intérieur de ces zones. Cela revient en quelque sorte à adopter le comportement d'une personne cherchant à tâtons les bords d'une pièce dans le noir. Cette exploration plus fine au niveau des frontières du modèle semble présenter au moins deux autres avantages. Premièrement, à petite échelle on peut approximer des processus non linéaires par des processus linéaires. Deuxièmement, on sait également qu'il est inutile (toujours à petite échelle) d'explorer la direction depuis laquelle on arrive car on y est déjà passés en cherchant un point précédent de la limite. La limite du modèle pouvant être franchie cette démarche devrait se traduire par des sauts de puce multidimensionnels de part et d'autre de la frontière du modèle (relativement à une valeur seuil).

6.3.2 Inclusion dans le projet VECTORS

Ce travail s'inscrit dans le cadre du projet européen Vectors of Change in Oceans and Seas Marine Life, Impact on Economic Sectors (VECTORS)(254). L'objectif principal du projet VECTORS est d'acquérir de meilleures connaissances sur les différentes pressions actuelles et potentielles et les vecteurs, directs et indirects, de changement de l'environnement marin. Les vecteurs pris en compte peuvent être de nature climatique, physique, biologique, écologique, économique, sociale, afin de couvrir tous les aspects possibles des écosystèmes étudiés. Ce projet s'intéresse à trois mers régionales à l'échelle européenne : la Baltique, la Mer du Nord et l'ouest de la Méditerranée. Chacune de ces mers régionales est subdivisée en plusieurs zones d'étude de plus petite taille, comme la Manche Est qui est l'objet de cette étude.

Dans le cadre de ce projet la Manche Est est représentée au moyen de plusieurs modèles qui sont développés en parallèle. Les courants sont modélisés au moyen d'un modèle MARS 3D (255; 256), qui est également utilisé pour représenter la dérive des oeufs et larves (227). Un de ces modèles utilise l'approche OSMOSE (239; 240), qui permet de modéliser les relations trophiques entre espèces de l'écosystème en faisant l'hypothèse d'une prédation opportuniste basée sur des relations de taille entre elles. Un troisième modèle utilise le cadre ATLANTIS (12; 257) afin de fournir une vision plus globale de l'écosystème. Ce modèle est déjà utilisé dans certains cas d'étude pour la gestion (258) et est la preuve que des modèles complexes peuvent être appliqués dans des cas concrets. Le modèle ISIS-Fish développé ici s'inscrit dans une démarche similaire d'approche globale de l'écosystème basée sur le couplage avec les autre modèles développés. En particulier, les résultats issus du modèle MARS sont déjà utilisés dans le modèle afin de définir les différentes zones de population et les matrices de connectivité permettant de les relier. Le couplage avec le modèle OSMOSE peut être vu comme l'étape suivante du processus de modélisation. Simuler les interactions trophique permettrait de prendre en compte les interactions entre poissons, mais surtout les interactions entre poisson et benthos. Ces interactions semblent particulièrement intéressantes à prendre en compte, car elles devraient permettre de faire le lien entre les évolutions à petite échelle des populations de benthos sous et la dynamique des populations de poisson à plus large échelle. On pourrait ainsi modéliser des boucles de rétro-contrôle de la pêche sur elle-même du fait de la propagation de ses impacts le long de la chaîne trophique. Ceci modifierait probablement le diagnostic fait sur les conséquences de l'implantation d'aires marins protégées sur les populations de poissons.

Prendre en compte les interactions entre groupes de benthos peut également présenter un intérêt. Les relations trophiques au sein du benthos sont très certainement trop complexes pour être prises en compte dans un modèle tel que celui développé ici. On pourrait toutefois tenter de mieux prendre en compte les relations de compétition (pour l'espace, la nourriture) entre les différents groupes représentés. Il pourrait également être intéressant de prendre en compte le niveau d'opportunisme de chacun de ces groupes, afin de mettre en évidence de possibles shifts dans la composition des espèces en relation avec l'intensité de pêche.

Pour aller plus loin, on peut également imaginer représenter les modifications du comportement des poissons et des pêcheurs en relation avec l'altération du benthos par les activités humaines. En effet, il a été observé que si certaines espèces de poissons fuient les zones d'extraction de granulats d'autres profitent de la mise en suspension du benthos pour venir se nourrir. Il en résulte des phénomènes de concentration locaux à court terme dont les pêcheurs cherchent parfois à tirer profit. Même si ces phénomènes sont locaux leur répétition pourrait avoir des conséquences à plus large échelle que l'on pourrait tenter de modéliser. D'autres aspects du comportement des poissons pourraient être pris en compte, comme la réduction de la taille de leur zone de présence avec la réduction de leur abondance.

6.3.3 Un modèle pour la gestion ?

La pêche est avant tout une activité économique. Les aspects économiques ont été peu pris en compte dans le modèle développé ici, seule la quantité de captures permettant de faire un lien avec les activités en aval de la pêche. En particulier, les différents coûts de la pêche n'ont pas été pris en compte dans la détermination du comportement des pêcheurs : seul le prix du poisson a fait office d'indicateur économique. De plus, lors de l'étude de la robustesse du modèle on s'est focalisés sur des critères biologiques (biomasse féconde) alors que cette robustesse pourrait également être recherchée sur des critères socio-économiques (maintien à plus ou moins long terme d'un niveau de capture ou de revenu, baisse des coûts de la pêche, etc.).

Cette piste semble d'autant plus intéressante que la gestion actuelle tend à être de plus en plus participative. Une des forces de la méthode proposée ici est de pouvoir tester des hypothèses, des scénarios, afin de mieux comprendre leurs conséquences. Elle peut donc fournir une base de discussion entre acteurs dans la recherche de mesures de gestion et éventuellement dans la réalisation de compromis entre différents objectifs de gestion quand cela est nécessaire. Cela pourrait ainsi faciliter l'interaction entre des acteurs avec des intérêts, des connaissances et des perceptions divers. Le modèle et la méthode proposés peuvent fournir un cadre général permettant d'intégrer ces différents aspects et faciliter les échanges entre acteurs autour d'une base commune. Cette démarche irait dans le sens de celle de (238) qui vise à coupler le modèle ISIS avec Marxan afin de tester des scénarios de gestion spatiale optimaux. On pourrait alors réellement imaginer d'utiliser ce modèle comme un outil d'aide à la gestion.

Un des inconvénients majeurs du modèle (des modèles complexes en général) et de la méthode développés est qu'ils sont difficilement utilisables de manière opérationnelle sur de courtes périodes de temps, comme cela est souvent nécessaire. En effet, l'utilisation d'un modèle complexe associée à la nécessité de faire un grand nombre de simulations fait que les temps de calcul sont élevés. Les groupes d'évaluation des stocks de poisson se réunissant sur des périodes de quelques jours seulement, il serait nécessaire de faire cette évaluation en deux temps : un premier pou définir les scénarios à tester et un second pour analyser les résultats obtenus après simulation. De ce fait notre modèle se positionne plus à l'heure actuelle comme un outil d'intégration de la connaissance et de production d'information destinée à soutenir la prise de décision, et non comme un outil de prise de décision à part entière.

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A spatial model of the mixed demersal fisheries in the Eastern Channel

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Abstract

A major reform process in the Common Fisheries Policy (CFP) led to proposals for new management measures. These new measures aim at overcoming the shortcomings of the previous management regime especially regarding the issues related to mixed fisheries, spatial dynamics of the stocks, discarding behaviour and data-poor species. These new measures include enforcement of long-term management plans potentially multi-specific, the generalization of Harvest Control Rules (HCR), and the adoption of a regional approach to fishery management and the progressive ban of discards potentially through the enforcement of catch quotas.

These measures are expected to have major impacts on the fleets operating in the Eastern English Channel, which face many of the problem targeted by the new CFP: strong regional structuration, multi-species catch, high level of discards, and harvesting of species not currently managed. To allow the projection of the ecological and economic impacts of the new CFP on these fisheries, we developed a Management Strategy Evaluation framework. The framework needed to take into account the specificities of the Channel fisheries and fish populations. We used the simulation framework ISIS-Fish which embodies a spatialized operational model to describe the dynamics of the seven species of major economic interest including their spatial dynamics, driven by habitat models and the dynamics of the fleets dictated by a fishing behaviour model. The model is then validated against diverse time series of observations and its appropriateness for the evaluation of management measures of the new CFP is assessed.

Introduction

The French demersal fisheries in the Eastern Channel are mixed fisheries which dominant target species are sole, seabass and scallops. However these fleets also land a large diversity of other species, either because they target it when their main species is not accessible - as for the dredgers who go for plaice and sole in summer, or because they caught it simultaneously.

In this context the evaluation of the impact of management measures of this fishery requires to take into account the behavior of the fleets and the multi-specific aspect of the fishery. In particular, report of effort in space or time when TACs are reached or to avoid by-catch, are likely to occur with unknown impact on fish communities. A spatialized model with intra annual temporal resolution was thus necessary.

ISIS-Fish is a simulator of mixed fisheries, developed especially to assess economic and ecological impacts of management measures (including spatial measures) (Pelletier et al. 2009).

The parameterization and validation of the model represent major challenges in term of the analysis of the fishery dynamics and the assessment of sensible parameter values which allow projecting the fishery under new regulation.

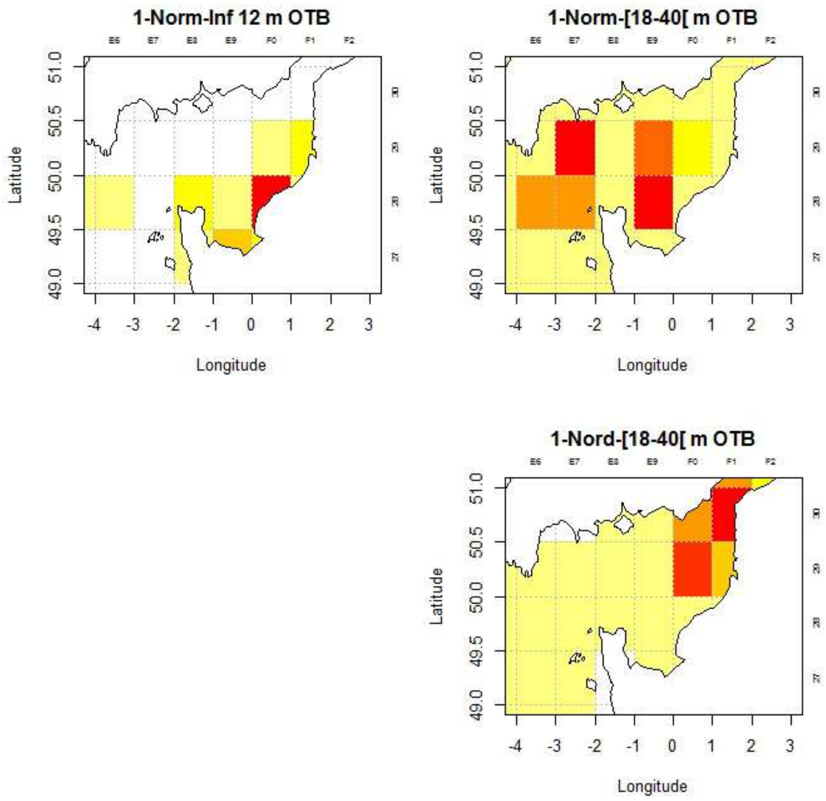
Materiel and Methods

The complexity of the fishery structure in the Channel, in particular, the diversity of gear, home harbor, métier and fishing area, required to develop a rigorous procedure for the analysis and the modeling of the activity of fishing fleets.

Selection of fleets and species explicitly modeled

The objective of fleet description is to model as accurately as possible the structuration of effort in space and time in order to derive fishing mortality. However, simulation times for the model increase with complexity and a tradeoff needs to be found between accuracy and level of description. To limit complexity, we did not start our analysis of fishing activity at individual boat level but rather used the segmentation created by the French Fishery Information System (SIH), which groups French vessels based on the main, or two main, gear practiced during the year (hereafter denominated IFR-fleet). We also crossed with home region (North of France and Normandy) and vessel length class (<10m, 10-12m, 12-18m, 18-24m, 24-40m and >40m) to create fleet segments because these characteristics potentially create heterogeneity in effort distribution (Figure 1).

Figure 1. Annual map of effort distribution for 3 different fleets when they operate métiers using bottom trawls.



The two most valuable species landed by French fleets in the Eastern Channel are sole (Solea solea) and scallops (Pectens maximus). The latter is mainly landed by the fleet of trawlers-dredgers which also targets sole. The majority of sole landings come from netters, bottom trawlers and mixed trawlers (pelagic and bottom trawls). We thus focused on these four IFR-fleets and pooled the others into an inexplicit fleet “OTHER”. The rest of the value landed by these four fleets is mainly made on squid, seabass, whiting, cod, and plaice, which consequently were included in the list of species explicitly accounted for, while the rest of the species caught, are pooled together into a species OTHER.

Definition of métiers and strategies

Métiers are defined as the combination of a gear and an area of practice. For the purpose of the study, all the effort spent outside area VIId is considered a unique métier (OTH-OTH). Within VIId, effort per gear per fleet is computed to identify the main gears used and the others are pooled together in a métier OTH-VIId. Within the fishing trips using the main gears, a threshold of 3% of the species of interest in the catch is used to filter fishing trips that do not target these species and these trips are attributed to métier OTH-VIId as well. Maps of effort helped identifying the main ices rectangles of practice for each métier and one métier per rectangle is created (e.g. OTB-27E9, Fig. 1). ICES rectangles with low effort for a given gear are pooled together in a unique métier (e.g. OTB-VIId).

Some fleet segments contain less than 3 vessels which prevent access to confidential economic data at the fleet scale. In order to reduce the number of fleet segments modeled without losing information, we looked for possible grouping of small segments. A Hierarchical Ascendant Classification was performed on fishing time per métier (gear x statistical rectangle) within a fleet (IFR-fleets and harbor) which represent fleet strategy. If the classification revealed similarities between the strategy of a small segment and the strategy of another segment, vessel length in the two segments are studied to determine if grouping would be relevant. If it is, the vessels from the small segment are added to the other one (e.g. the 2 bottom trawlers from Normandy of size class [10-12[m were 10m long and thus merged to the 14 trawlers smaller than 10m). The small segments that could not be grouped with others were not explicitly described and pooled with the “OTHER” fleet bringing the number of fleets considered to 17, hereafter referred to as “strategies” (table 1).

Table 1. Strategies described in the model, with indication of their main gears, home harbor, vessel length class, average number of boats participating between 2008-2010 and estimated technical efficiency.

IFR-Fleet	Harbor	Vessel size	Average number of boats (2008-2010)	Technical efficiency
Bottom trawlers	Normandy	<12m	16	2.64
Bottom trawlers	Normandy	18-40m	17	0.71

Bottom trawlers	North	18-40m	15	1
Mixed trawlers	Normandy	18-40m	7	1.22
Mixed trawlers	North	18-24m	11	0.83
Mixed trawlers	North	24-40m	8	1.19
Dredgers-trawlers	Normandy	<10m	9	1.72
Dredgers-trawlers	Normandy	10-12m	50	2.03
Dredgers-trawlers	Normandy	12-18m	103	1.59
Dredgers-trawlers	Normandy	18-24m	5	1.82
Dredgers-trawlers	North	10-12	14	4.02
Dredgers-trawlers	North	12-18m	9	1.37
Netters	Normandy	<10m	22	1
Netters	Normandy	10-12m	12	2.28
Netters	North	<10m	7	2.13
Netters	North	10-12m	51	5.01
Netters	North	12-18m	12	3.78

Standardization of effort:

ISIS-Fish decomposes catchability into multiplicative effects linked to fish accessibility, gear, ability to specifically target a species and technical efficiency. Technical efficiency is related to vessel characteristics, often linked with vessel length, it is thus assumed identical within the vessels of a strategy. In the case of the Channel fisheries, we assume that targeting is not realized by operating the gear in a certain way but rather by choosing a gear and a fishing ground where a species has good chances to be caught. We also assume that spatial differences in catch composition observed between métiers practiced with the same gear, result from the heterogeneity in species distribution rather than from differences in fishing practices between areas. The model thus only assumes an effect of gear (written as a main effect (Fstd) x a specific effect (interactions species:gear)) and an effect of the strategy as a proxy for technical efficiency. The heterogeneity in species distribution being explicitly modeled in ISIS-Fish, no area effect is added. Generalized linear models are used to assess these effects using logbook data transformed from catch in weight into catch in numbers (see following section)(equation 1). We use catch in numbers because ISIS-Fish applies the Baranov equation to Abundance rather than biomass. Given the high frequency of occurrence of zeros in the dataset (each trip generates 7 observations, one for each species, with a value of zero if the species was not caught during the trip), individual trips were aggregated at the monthly scale and a negative binomial distribution is used (log link). Results are presented in tables 1 and 2.

$$C_{month\,,\,year}(sp\,,gear\,,m\acute{e}t\,ier\,,strategy)$$ gear + gear:species + strategy + offset(Effort) (1)

Table 2. Estimates of the parameters of effort standardization

	TBB	DRB	GNS	GTR	OTB
Fstd	0.3	1.0	17.6	2.2	20.7
			Interactions gear:species		
COD	1.21	0.80	4.08	5.92	1.66
PLE	498.49	12.97	0.40	16.54	2.63
SCE	879.30	3320.77	0.00	0.03	3.81
SOL	791.59	13.91	2.44	68.76	1.48
SQZ	0.81	7.36	0.01	0.02	11.88
WHG	6.06	3.90	0.41	1.99	36.25

Conversion of catch in kg to catch in numbers at age:

Since 2012, the length composition of commercial catch elevated from sampling is stored in the ICES database Inter-catch and was made available for the species assessed (cod, ple, sol, whg). 2012 was assumed representative of catch composition per quarter and gear. Von Bertalanffy parameters were used to convert length into age and derive quarterly age composition by gear for the French fleets. Weight at age was then used to compute mean weight in the catch and convert landing data in kilograms into numbers and numbers at age (equation 2 and 3).

$$w(gear',quarter)=\sum_{a} ¿(a\,,gear\,,quarter)*weight(a\,,quarter) \qquad (2)$$

$$C_N(gear\,,quarter)=\frac{C_W(gear\,,quarter)}{w(gear',quarter)} \qquad (3)$$

C_W catch in weight and C_N catch in numbers.

For seabass, length composition in the catch were available for 2008-2010 in the assessment report (WGNEW 2012) by year and gear (nets and trawls).

Length frequency in the catch not available for scallops and squid and an average weight in the catch was assumed.

Fishing mortality from other fleets

Part of the catches of these species is not explicitly modeled here because they come from foreign fleets or French fleets left apart from the analysis. This extra source of mortality is accounted for by applying an additional fishing mortality (FOTH) to each age class. The ratio of modeled catch over total catch is used as a proxy for the ratio of modeled fishing mortality over total fishing mortality (eq. 4).

$$\text{Ratio}(a,y) = 1 - (\text{Cfr}(a,y) / \text{Ctot}(a,y)) \qquad (4)$$

Where y is the year, a is the age class, Cfr(a,y) is the catch in numbers at age a in year y by the French fleets explicitly modeled, computed as explained above; Ctot(a, y) are the total catch in numbers at age *a* in year y reported in the assessment report.

Then FOTH(a,y) is derived as follows:

$$\text{FOTH}(a,y) = \text{Ftot}(a,y) * \text{Ratio}(a,y) \qquad (5)$$

With Ftot(a,y) the fishing mortality at age age in year y reported in assessment report. In prediction, the ratio is assumed constant but Ftot is derived from the management measure.

Toward a dynamical approach for systematic conservation planning of Eastern English Channel fisheries

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Abstract. In the past decade, systematic conservation planning tools have been increasingly and successfully used to set spatial conservation plans that meet quantitative protection targets while minimizing enforcement and socio-economic costs. However, when applied to fisheries, systematic conservation planning fails to account for (i) changes in fleet dynamics induced by new conservation constraints and their associated feed-backs on conservation costs, or (ii) their influence on fish population dynamics and distributions, which may in turn alter the achievement of conservation targets. Such a static approach may therefore lead to short- or medium-term misestimates in forecasted costs and target achievements. In order to circumvent such limitations of systematic conservation planning we present a first attempt to couple a conservation planning tool (Marxan with Zones) with a mixed-fisheries dynamics simulation model (ISIS-Fish), applied to the Eastern English Channel fisheries. Broad principles and perspectives are discussed and anticipated future challenges of such approach are presented.

Keywords: MPAs, systematic conservation planning, mixed fisheries dynamics, model coupling, Eastern English Channel, Marxan with Zones, ISIS-Fish.

1 Introduction

The Eastern English Channel is an important ecological area that is experiencing growing human pressure, mainly exerted by the bordering countries, France and the United Kingdom, and is subjected to a wide range of uses such as fishing, sediment extraction and transport [1–3]. Spatial regulations, including Marine Protected Areas (MPAs) are now increasingly introduced to legislate these perturbations [4–7]. MPAs,

adfa, p. 1, 2011.

to be successfully implemented, need to balance conservation objectives and socioeconomic features, such as fisheries [8]. Within this context, France and the United Kingdom are under the obligation to create a consistent Marine Protected Area (MPA) network that complies with several conventions, especially the Convention on Biological Diversity [review in 9] and the Bird and Habitat European directives, whilst ensuring a viable future for the wide range of uses within this area.

In the past decade, systematic conservation planning tools have been increasingly and successfully used to develop spatial conservation plans – involving MPAs – which meet quantitative targets (*e.g.* a given protected percentage of each species distribution or habitat area) while minimizing enforcement and socio-economic costs [10–13]. This approach thus provides a framework that is deemed suitable to design consistent MPA networks that are cost-effective and minimize social costs, hence increasing their likelihood of effective implementation. However, systematic conservation planning applied to fisheries accounts neither for (*i*) changes in fleet dynamics induced by new conservation constraints and their associated feed-backs on conservation costs, nor (*ii*) their influence on fish population dynamics and distributions, which may in turn alter the achievement of conservation targets. Such a static approach may therefore lead to short- or medium-term misestimates in forecasted costs and target achievements.

Mixed fisheries simulation models are increasingly used to predict changes in fleet and fish population dynamics under various fishery management scenarios (*e.g.* [14, 15]), but lack, in most of case, the methodology to transpose the results into advices to support spatial conservation measures.

In this context, coupling systematic conservation planning tools with mixed-fisheries models (or other types of simulation models, in accordance with the type of issue tackled) seems a promising approach to test scenarios and build advices for management of highly dynamic and complex systems such as coastal areas under intense human use.

2 Proposed approach

2.1 Overview of selected tools

Systematic Conservation Planning. Several tools exist that are dedicated to systematic conservation planning (*e.g.* Marxan, Zonation) and can help to design MPAs. Most of them however rely on a binary and often unrealistic full protection strategy, therefore missing the complexity of management strategies which can be deployed through multiple types of MPAs. A recent tool, Marxan with Zones (MwZ) [16], allows this limitation to be overcome in an optimal way by extending the Marxan methodology (based on the minimum-set principle aiming to achieve given quantitative representation level of species and habitats at minimal cost) to multiple zone types. It affords the possibility to consider multiple, possibly concurrent, resource uses which are managed in different ways, while taking into account a variety of costs. It has been shown to be able to provide management scenarios that – compared to a standard Marxan analysis – ensure more equitable impacts among different uses while lowering the overall economic and social impact, and still meeting conservation targets, thus increasing the likelihood of an effective implementation [17]. However, unlike Marxan, feedback on MwZ effectiveness remains scarce.

Marxan with Zones uses a simulated annealing algorithm to work as an optimization tool which meets complex constraints such as combinations of overall percent and/or absolute values of each feature (often species or habitats) to be protected in each type of "zone" (with varying protection levels corresponding to which human use are maintained). The objective function it minimizes has the form:

$$\sum_{PUs} Cost + \underbrace{BLM \times \sum_{Z} \sum_{PUs} BoundaryCost}_{Connectivity\cos ts} + \sum_{Ft} FPF \times FeaturePenalty \qquad (1)$$

Where *Cost* represents the sum of various costs associated with the selection of a "protection unit" (*PU*, the smaller spatial unit). These costs can be of any kind found to be relevant to each case study, for instance surface area, enforcement or socioeconomic costs. Additionally to these inherent costs, connectivity costs – of which the boundary length modifier (BLM) controls the overall contribution to the objective function value – allow to control the level of aggregation/fragmentation of conservation zones or increase the co-selection of connected PUs. The "connectivity" between zones is also controlled in this term by zone boundary costs, which are formally multipliers of PU boundary costs between each combination of adjacent zones[1]. The last term of eq. (1) refers to penalties for failing to achieve targets, summed over features (*Ft*; or species) and is controlled through feature penalty factors (*FPF*); the higher the *FPF*, the most likely the fulfillment of the target.

MwZ has been selected for this study owing to its ability to both reproduce complex management scenarios and include use-specific costs in a flexible way, which more accurately reflect mixed-fisheries properties than other existing tools.

Mixed fisheries simulation model. ISIS-Fish has been chosen because it is a modelling tool suitable for investigating the consequences of alternative policies on the dynamics of fish resources and fisheries [18]. This spatially explicit model allows quantitative policy screening for fisheries with mixed-species harvests [18, 19]. It may be used to investigate the effects of combined management scenarios including a variety of policies: total allowable catch (TAC), licenses, gear restrictions, effort controls but also alternative ones such as the introduction of marine protected areas [20, 21, 15] or individual quotas [22], etc. Fisher's response to management may be accounted for by means of decision rules based on population and exploitation parameters or explicit dynamic model with endogenous (*e.g.* fixed fish prices and variable costs, that can be explicitly modelled) or exogenous variables (not affected by the model). This fishery model is based on three submodels (i) a fishing activity

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[1] Note that a PU can pertain to only one zone at a time.

dynamics model, (ii) a fish population dynamics model and (iii) a management dynamics model.

Each submodel is spatially and seasonally explicit, with a monthly time step. The three submodels interact only if they overlap in space and time. The modelled area is represented by a grid, the resolution of which, in latitude and longitude, is chosen with respect to the dynamics being described and the available knowledge of the studied fishery. Within this region, zones (i.e. sets of grid cells) are defined independently and delimit the spatial scope for each population, each fishing activity and each management measure. Finally, bioeconomic outputs can be simulated and their properties (including uncertainties) statistically analyzed to produce indicators of the relevance of management strategies [23].

2.2 Models scopes and implementations

Here we present the first highlights of an ongoing study which aims at coupling a systematic conservation planning software with a mixed-fisheries model to evaluate the relevancy for fisheries management of the MPA network being implemented in the Eastern English Channel (**Fig. 1**). And, where relevant, provide advice for management strategies.

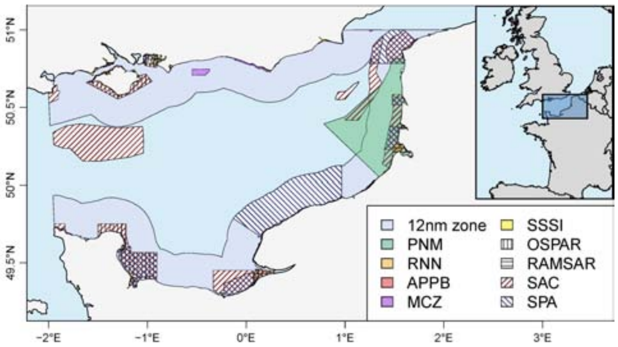

Fig. 1. MPAs (actual or planned) in the ICES VIId zone. PNM: natural marine park (France); RNN: national natural reserve (Fr.); APPB: prefectural biotope protection (Fr.); MCZ: marine conservation zone (UK); SSSI: site of special scientific interest (UK); OSPAR: OSPAR convention zone; RAMSAR: RAMSAR convention zone; SAC: special area of conservation (Natura 2000, habitat convention); SPA: special protected area (Natura 2000, bird convention).

This study therefore focuses on ecosystem and socioeconomic features which are linked to fishing activities, restricted to professional fishing owing to data availability.

Marxan with Zones features to protect are:

- abundance distributions (mean over 1990-2012) of two of the main targeted species (those accounted in the fleet dynamics model), common sole (*Solea solea*; **Erreur ! Source du renvoi introuvable.a**) and plaice (*Pleuronectes platessa*;

Erreur ! Source du renvoi introuvable.b) assessed from the Channel ground fish surveys (CGFS) data.

- eighteen benthic habitats, on which towed gears can have a negative impact. Data used are those from Delavenne *et al.* [11].
- thirteen pelagic habitats, which contains communities that can be affected by most pelagic gears. The typology used is the one defined by Delavenne *et al.* [24].

Among feature types, only exploited species distributions are planned to be dynamically linked to the simulation model, habitats being handled as a static part of the system.

As for costs, hours fished by type of gear and zone, estimated from VMS [25] are used as proxies of value losses when a protection unit is selected for a type of zone which bans or limits some uses (**Erreur ! Source du renvoi introuvable.c & d**). As a first approach, they will not be processed in a dynamic way because the selection of a PU would eliminate its cost on the next step (no or less fishing).

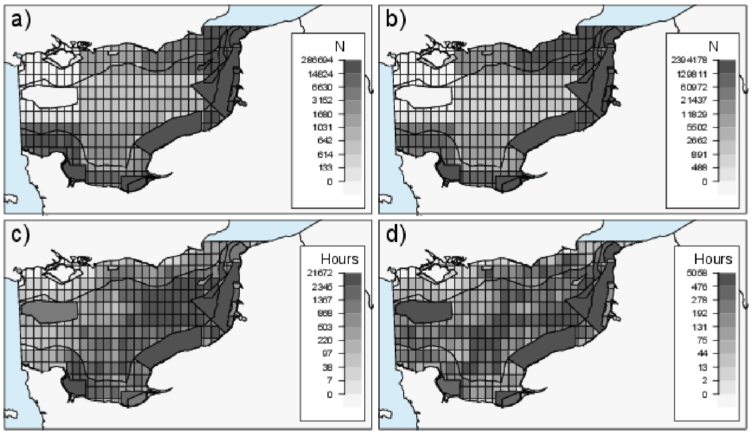

Fig. 2. Mean abundances (N, decile scale) by PU of *Solea solea* (a) and *Pleuronectes platessa* (b) estimated from the CGFS survey data and aggregated efforts by PU (hours fished, decile scale) from French (2008) and English (2007) VMS data for ground-towed gears (c) and all other types of gears (d).

The spatial grid was defined so that each MPA was divided into as many subareas of unique administrative status and that remaining available areas were separated according to the same grid as the ISIS-Fish model (1/32nd of ICES statistical rectangle; **Erreur ! Source du renvoi introuvable.**) and further divided according to the 12 nautical miles zone.

A scenario was tested with only two kinds of protection zones – no ground-towed gears and no-take (all activities prohibited) – and where the 12 nautical miles zone was considered already contributing to conservation (limited access to vessels >24m). Already planned MPAs were constrained to always apply one of the two protection levels. Zone contributions to conservation (**Table 1**) and costs multipliers (**Table 2**;

formally corresponding to proportion of effort reduction within protected zones) were set arbitrarily, although displaying sensible orders of magnitude.

The results of this drastic scenario (few choices for management strategies, all of which prohibited ground-towed gears) are reported in **Fig. 3**.

Table 1. Zone contributions to protection to varying types of features. These are proportions of feature actually protected by type of zone.

Zones Features	Unprotected	12nm zone	No ground-towed	No-take
Benthic habitats	0.0	0.1	1.0	1.0
Pelagic habitats	0.0	0.0	0.0	1.0
Target species	0.0	0.1	0.5	1.0

Table 2. Cost multipliers by zone and gear type. These are also the proportion of effort reduction applied by type of zone and gear for the two type of MPA management strategies tested.

Zones Gears	Unprotected	12nm zone	No ground-towed	No-take
Towed	0.0	0.1	1.0	1.0
Other	0.0	0.0	0.0	1.0

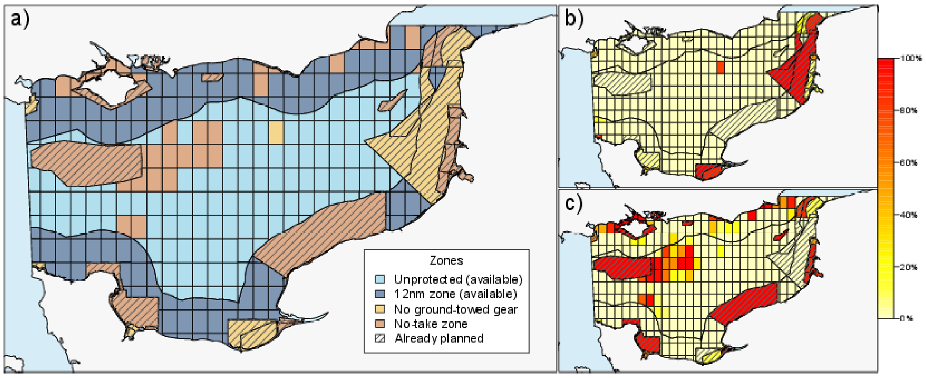

Fig. 3. Marxan with Zones results on 100 runs: best solution (a) and selection frequencies of "no ground-towed gear" (b) and "no-take" (c) zones. Available cells are those not selected for any purpose.

ISIS-Fish. The model used is an improved version of the one developed by Gasche *et al.* [26], characterizing population and exploitation dynamics of sole and plaice, with:

1. a finer spatial resolution of 0.125° (latitude and longitude) that allows for a more realistic depiction of biological and exploitation processes.

2. better account of populations distributions across life stages – with feeding grounds, nurseries and reproduction zones for three populations by species (*i.e.* nine zones for both plaice and sole) – according to [27, 28].

3. fishing activities updated according to Lehuta *et al.* ([29]).

The model was tested over 12 years under two different scenarios *(i)* one with only total allowable catches (TAC) as management measure (forced by 2008-2011 actual TACs, then dynamically set by an harvest control rule that aims at reaching F_{MSY} in five years following ICES advices) and *(ii)* another with additional spatial conservation measures (**Fig. 4**) consistent with MwZ outputs (**Fig. 3**).

Results show very little differences between scenarios with TACs and TACS+MPAs for abundances (**Fig. 5.a**) and landings (**Fig. 5.b**) of both Plaice and Sole during the transition period towards management at F_{MSY} (until year seven). However, differences become more substantial afterward (when the TAC is less constraining), although no clear difference appears in trends.

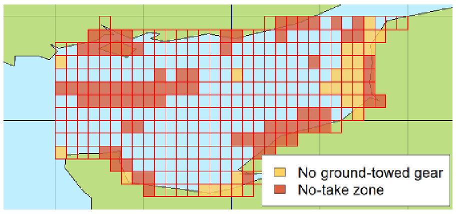

Fig. 4. ISIS-Fish grid and management zones defined in the model. These are an attempt to broadly reproduce MwZ outputs (**Fig. 3**).

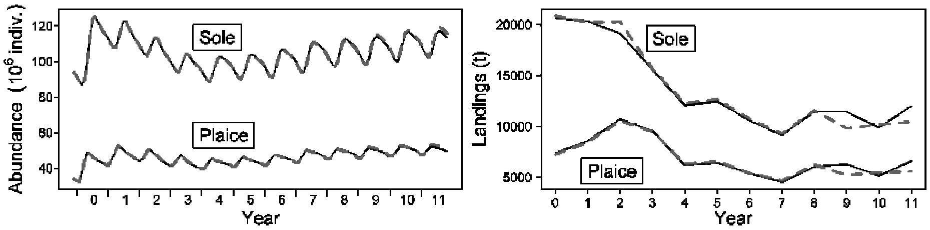

Fig. 5. ISIS-Fish model outputs of monthly abundances (left) and yearly landings (right) compared between TAC only (black solid lines) and TAC+MPAs (grey dashed lines) scenarios.

2.3 Explicit model coupling in practice

Marxan with Zones is a command-line software which works with text input files, all listed in a main input file and with fairly simple and documented structures [30]. It is therefore easily controlled through any platform which allows running commands and is able to handle data and text files (*e.g.* we have easily controlled all the Marxan with Zones analysis sequence, from data formatting to output representation, with R; `http://cran.r-project.org`). Principal Marxan input files which would be subject to dynamical updates are:

- the protection unit file, and more particularly the costs given to each planning unit (PU).
- the features (here exploited species) versus PUs file which gives amount of each feature for each PU.

Files such as those containing boundary lengths/connectivities between protection units, or "zone boundary costs" pertain to the original design and are unlikely to be modified by iterative runs (except for sensitivity analysis).

As for ISIS-Fish, control from a third-party tool seems more difficult since most parameters are stored in embedded data-bases, with internal referencing of objects, as spatial units or populations for instance. Therefore, even though ISIS-Fish simulations themselves can be run from command-line calls, the parameters of the model cannot be directly controlled through text files. However, ISIS-Fish is an open-source modeling platform, with an active development team, hence highly extendable.

For instance, concerning the translation from Marxan with Zones outputs to ISIS-Fish management strategies, the ISIS extensive scripting (Java script) abilities will be used to:

1. load formatted MwZ outputs (preferably preprocessed by R scripts for easy handling).
2. define as many management zones as different effort reductions by gear, calculated from MwZ outputs (pre-simulation script).
3. calculate an overall effort reduction by métier from the overlap between the métier zone and management zones for the gear used (new management rule) at each time step.

The second point raises the issue of transferring costs, features and optimized spatial management measures between two possibly different spatial scales. Indeed, there is no requirement for the spatial grid in MwZ to be regular, as it is the case for the one in ISIS-Fish. In fact, it is even convenient to keep existing – intricately shaped – MPAs as separated PUs (*e.g.* **Fig. 3**) for MwZ analyses. Therefore, even under simple homogeneity assumption regarding amounts within grid cells, the transfer of data from a grid to another requires extensive calculations, among which assessment of cell surface overlap between the two model spatial grid layers (that only needs to be calculated once) and pro rata reallocation rates from one grid to the other. This is easily done with R scripts which can notably calculate an accurate effort reduction in ISIS spatial unit (cell) by gear, from a given MwZ solution, since gears limitations are given for each kind of protection zone.

On the other side, two options emerge for the automation of the coupling:

1. to iteratively run both of the tools and operate the data format conversion between them, a first step will be to use an external third-party tool. As suggested above, R

is a good candidate because of its extended data-handling abilities and its capacity to interact with other software packages through so-called “system calls”.

2. it is nevertheless considered to further develop a module within ISIS-Fish to control MwZ directly from within simulation iterations, hence getting quick cost assessments and zoning optimizations to enhance the dynamical properties of the whole system. The existing ability of ISIS to connect to R could in particular be used to run scripts controlling the whole MwZ loop, from data preparation to processing of output to provide management zones that can be handled by ISIS. This way, costs and feature amounts could be directly updated from ISIS-Fish to MwZ and, as a feedback, zoning in ISIS-Fish could be updated according to MwZ outputs.

2.4 Types of scenarios to be tested

Coupling these two tools would open the opportunity to test a wide range of scenarios regarding the dynamics of spatial conservation plans. The following propositions are far from exhaustive but focus on types of analyses already planned in the context of this study.

A first and fairly obvious type of analysis would consist in testing what would be the dynamics of the main fleets and fish populations under different conservation scenarios, and how would they influence targets achievement. This would only require a one way coupling from Marxan with Zones to ISIS-Fish that does not really require explicit and automated translation of a common Marxan with Zones analysis protected zones outputs to management scenarios within ISIS-Fish. Achievement of targets would be easily assessed from ISIS-Fish outputs in terms of species abundance distributions. More realistic economic costs, although completely independent at this stage from those used in Marxan with Zones, could also be assessed from comparison of simulated landings at the beginning and the end of the simulation period.

From the previous analysis, it may be possible to test whether any proposed MPA network with particular set of conservation measures is suitable to ensure medium to long-term viability of fleets and of the fish populations they harvest. At present, in the Eastern English Channel, enforcement measures are still to be defined in most proposed MPAs and the use of Marxan with Zones on its own, to provide near optimal management scenarios within the already planned MPA network may be very relevant. The addition of the ISIS-FISH simulations would enable to evaluate under which management strategy the fishing fleets will remain viable. For that purpose, the methodology proposed by Lehuta *et al.* [23], based on bioeconomic indicators and their uncertainty to evaluate management strategies, could be used.

Other types of analyses would require a more intricate and fully dynamic coupling of the two tools than the preceding ones. It is for instance planned to test the effect of the chronological sequence of enforcement, and how it could be optimized. Indeed, it would involve running ISIS-fish over a given interval of time (*e.g.* one year), then testing which would be the best enforcement addition to the network and running

ISIS-Fish again from where it stopped, etc. Such an approach would have to be tested over various time lags.

All these types of analyses would benefit by also testing whether different near-optimal solutions, which differ noticeably in term of selected sets of PU-zone pairs but not in term of cost, would lead to different dynamics and viabilities of fleets and harvested populations. Such analyses would have to account for confidence in data. From a more general point of view, a sensitivity analysis on optimization and model parameters would be necessary.

3 Perspectives and future challenges

Challenges raised by this coupling approach pertain to (i) finding a relevant design and level of detail regarding processes embodied in each model, (ii) getting a proper parameterization of both models and (iii) keeping an overall consistency, instead of technical issues regarding the coupling itself.

First of all, the methodology promoted here will be extended to a more representative set of the Eastern English Channel fisheries. For that purpose, the MwZ number of features will be extended to include seven target species, in order to be coupled with the ISIS-FISH model developed by Lehuta *et al.* ([29]) and which includes five additional target species: Cod *Gadus Morhua*, Whiting *Merlangius merlangus*, European seabass *Dicentrarchus labrax*, Squids *Loligo spp.* and Scallop *Pecten maximus*.

Additional features such as birds or seals colonies, highly sensitive habitats or endangered species known locations may also be added to increase the spatial constraint on the MwZ solutions and to better represent the local biodiversity and not only the exploited species.

The definition of management scenarios that are likely to be implemented is also of high importance. This raises the need for identification of more realistic management measure regarding either the administrative status of zones (e.g. is it desirable to ban ground-towed gears from large zones designated under the bird directive only?) or the local characteristics of an MPA itself (*e.g.* in the case of particularly discrete natural bivalve beds where dredging cannot be banned without deep economic impacts). In that context, a consultation of managers is underway to move towards more realistic constraints on MwZ inputs.

Contributions of different zone types to the preservation of various features will also have to be characterized in a more explicit way, based on observed and quantified gear impacts, proposed effort reduction (by gear within the zone) and overall effort partition among gears.

This approach could also be improved by moving fishing activities from costs to features that also have to be protected (*e.g.* with a 90% target on value landed by each type of activity, that is no more than 10% loss). This would ensure a better equitability of the conservation effort among métiers by applying a high penalty to scenarios inducing a larger loss than the one set by target for at least one type of activity.

These improvements should all together enhance our capacity to provide valuable diagnostics and advices regarding the effectiveness of MPA networks for management of mixed fisheries.

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