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Ecosystem and fishers' behaviour modelling: two crucial and interacting approaches to support Ecosystem Based Fisheries Management in the Eastern English Channel.

Modélisation des écosystèmes et des comportements de pêche: deux approches liées et essentielles en appui à la gestion écosystémique des pêches en Manche Orientale.

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Résumé

La mise en place de l'approche écosystémique des pêches (AEP) requiert une amélioration de nos connaissances sur la complexité des écosystèmes. Comprendre la réaction de l'écosystème à des mesures de gestion est essentiel pour atteindre les objectifs de conservation. La modélisation écosystémique a amélioré nos connaissances sur le fonctionnement des écosystèmes et leurs interactions avec les usages du domaine maritime; et est de plus en plus utilisée pour évaluer l'impact de mesures de gestion. Le comportement de pêche des flottilles démersales françaises en Manche Orientale a été analysé. Les résultats montrent que les pêcheurs conservent leurs habitudes de pêches et que le trafic maritime peut impacter leurs décisions. Une analyse globale des résultats d'études menées au cours des trente dernières années démontre l'influence des habitudes et des espèces ciblées sur le comportement de pêche. L'exploration de la dynamique de l'écosystème a nécessité l'utilisation du modèle Atlantis, en focalisant sur deux espèces commerciales, la sole (Solea solea) et la plie (Pleuronectes platessa). L'importance des zones estuariennes est révélée, ainsi que le rôle joué par les rejets et par deux espèces clés, la morue (Gadus morhua) et le merlan (Merlangius merlangius). La sole et la plie ont peu d'influence sur le réseau trophique excepté sur la dynamique des invertébrés benthiques. Nous évaluons les conséquences de l'application de fermeture de zones et d'une réduction d'effort sur le comportement de pêche et l'écosystème et mettons en évidence un bénéfice de l'application combinée de ces mesures sur la biomasse des espèces commerciales et sur la valeur débarquée par unité d'effort.

Abstract

The implementation of the ecosystem approach to fisheries management (EAFM) requires an enhancement of our knowledge of ecosystem complexity. Understanding the ecosystem reaction to management regulation is a key to achieve conservation objectives. Ecosystem modelling improves our knowledge on ecosystem functioning in interaction with human activities, and it is now widely used to evaluate management strategies. The fishers' behaviour of the French demersal fisheries in the Eastern English Channel (EEC) has been investigated. Results showed that fishers tended to adhere to past annual fishing practices and maritime traffic may impact on fishing decision. A global analysis of the fisheries science literature during the last three decades evidenced the influence of tradition and species targeting in fishers' behaviour. The exploration of ecosystem dynamics required the use of the ecosystem model Atlantis with a focus on two commercial flatfish species, sole (*Solea solea*) and plaice (*Pleuronectes platessa*). The importance of estuary areas and of nutrient inputs has been revealed as well as the role of discards and of two key species, cod (*Gadus morhua*) and whiting (*Merlangius merlangius*). Sole and plaice did not have a strong influence on the trophic network excepted on benthic invertebrate dynamics. Finally, we investigated the consequences of area closure and effort reduction on fishers' behaviour and the ecosystem impacted. We observed a noticeable benefit of combining area closure and effort reduction on the biomass of most commercial species and on the total value landed per unit effort.

Résumé étendu

L'implémentation de l'approche écosystémique des pêches (AEP) nécessite une amélioration de nos connaissances sur la complexité des écosystèmes, de leur fonctionnement, et de leurs interactions avec les activités de pêche. Comprendre les réactions de l'écosystème et des pêcheurs aux mesures de gestion mises en place est cruciale pour atteindre les objectifs de conservation et durabilité des ressources marines. Depuis ces dernières années, le développement de la modélisation écosystémique a contribué de façon significative à l'amélioration de nos connaissances sur le fonctionnement des écosystèmes et sur leurs interactions avec les usages du domaine maritime. Cette approche de modélisation est de plus en plus employée pour évaluer l'efficacité des plans de gestion.

Anticiper le comportement des pêcheurs est un élément clé de la réussite de la mise en place de d'une AEP. L'un des principaux défis rencontrés par les gestionnaires de la pêche est leur capacité à anticiper comment les pêcheurs réallouent leur effort de pêche suite à une fermeture de zone de pêche, permanente ou saisonnière, en prenant en considération la compétition avec d'autres usages pour l'accès au domaine maritime et à la ressource. Premièrement, nous appliquons un « Random Utility Model » (RUM) pour déterminer comment l'effort de pêche est alloué dans l'espace et le temps par les pêcheries françaises mixtes démersales opérant en Manche Orientale. Les variables explicatives utilisées pour décrire le comportement des pêcheurs étaient l'effort de pêche passé représentant leurs expériences ou habitudes de pêche, les captures et la valeur de débarquement par unité d'effort précédemment enregistrées pour représenter le profit attendu, l'effort de pêche d'autres flottilles de pêche, ainsi que la proportion de la zone de pêche occupée par les réglementations spatiales et par le trafic maritime. Les pêcheurs conservent en grande partie leurs habitudes de pêches d'une année sur l'autre, à l'exception de la flottille ciblant les coquilles Saint-Jacques (*Pecten maximus*) qui présente durant l'année un comportement saisonnier. De plus, nous avons montré que les coquillards français et anglais partageaient principalement la même zone de pêche et que le trafic maritime suivant les flottilles influençait les décisions de pêche. Finalement, le modèle a été validé avec succès en comparant au moyen de deux approches différentes sa capacité à prédire l'allocation d'effort avec des données observées.

Dans un deuxième temps, nous avons cherché à synthétiser et analyser les connaissances acquises sur l'utilisation de RUM dans la littérature halieutique durant les trois dernières décennies, pour différents cas d'étude répartis à travers le monde. Une méthodologie a été développée pour standardiser l'information fournie par différentes études et comparer les résultats obtenus. Six facteurs impactant le comportement de pêche ont été identifiés : la concentration d'autres navires, l'habitude, le revenu espéré, le ciblage d'espèces, le coût et le risque encouru. Nous avons utilisé

trois analyses basées sur des modèles linéaires pour comprendre dans quelle mesure ces facteurs influencent le comportement de pêche. Premièrement, une analyse binaire nous a montré que les pêcheurs sont attirés par leurs revenus espérés, leurs habitudes, les espèces cibles et la concentration d'autres utilisateurs du domaine maritime, et évitent les choix impliquant un coût élevé. Deuxièmement, nous avons mis en évidence que les flottilles démersales actives privilégiaient des modes saisonniers et étaient moins influencés par les informations plus récentes. Finalement, il a été montré que les flottilles démersales sont généralement peu enclines à prendre des risques, et également que l'habitude et le ciblage d'espèces sont plus importants que le revenu espéré dans le processus de décision. Le coût et la concentration d'autres usages ont la même importance que le revenu espéré. Les flottilles pélagiques semblent accorder une importance égale à tous les facteurs, mais ce résultat doit d'être nuancé et considéré avec précaution en raison du manque de données disponibles pour ces flottilles.

Pour pouvoir représenter dans sa totalité les interactions et l'impact du comportement de pêche sur l'écosystème de la Manche orientale, une meilleure compréhension de la structure de l'écosystème et de son fonctionnement est nécessaire. Durant la dernière décennie, nous avons vu une augmentation de l'intérêt accordé à la modélisation écosystémique et le développement de plusieurs applications permettant de mieux comprendre le fonctionnement des écosystèmes marins, ce qui est crucial pour améliorer les stratégies de gestion futures des ressources marines. Nous avons utilisé le modèle écosystémique Atlantis dans notre étude pour explorer la dynamique et les processus dominants de l'écosystème de la Manche Orientale. Nous nous sommes plus particulièrement intéressés à deux espèces commerciales de poissons plats, la sole (*Solea solea*) et la plie (*Pleuronectes platessa*). Ce modèle complexe a été paramétré avec des données collectées, à partir de plusieurs sources d'information (littérature, données de campagnes scientifiques, données de débarquement, résultats d'évaluation de stocks et d'autres modèles) et ajusté pour que les captures et les biomasses simulées reflètent les observations collectées entre 2002 et 2011. Ici, nous présentons principalement les sorties pour les deux espèces cibles et quelques espèces jouant un rôle essentiel au cœur du réseau trophique. La calibration du modèle a révélé l'importance des zones côtières, et des apports de nutriments provenant des estuaires en Manche Orientale. Il émerge que le manque de nutriments provenant des rivières décroit la productivité des nourriceries et affecte négativement la production de sole et de plie. Le rôle des rejets a également été mis en évidence sur le réseau trophique. Même si la sole et la plie n'avaient pas d'influence significative sur le réseau trophique des vertébrés, ils sont d'importants prédateurs pour les invertébrés benthiques et sont en compétition pour la nourriture avec les crustacés, le merlan (*Merlangius merlangius*) et d'autres poissons démersaux. De plus, deux espèces clés ont été mises en évidence dans notre représentation de l'écosystème, la morue (*Gadus morhua*) et le merlan. L'utilisation d'un modèle « end-to-end » améliore de façon substantielle notre compréhension de la dynamique de l'écosystème de la Manche Orientale, et du rôle de la sole et de la plie dans cet écosystème.

Dans la dernière partie de notre étude, nous avons évalué les conséquences de la mise en place de mesures de management sur l'écosystème de Manche orientale et sur les activités de pêche qui en dépendent. Nous avons tout d'abord couplé le modèle Atlantis avec différents modèles de dynamique de flottilles afin d'identifier le modèle de dynamique de flottilles permettant le meilleur ajustement des efforts de pêche prédits aux données. Le modèle intégrant les facteurs ciblage de la sole et habitude comme déterminants du comportement de pêche était le plus adapté. Nous avons alors évalué les conséquences de l'application de fermeture de zones à la pêche et/ou d'une réduction d'effort sur l'écosystème de la Manche Orientale et sur la flottille française de fileyeurs ciblant la sole. Nous avons analysé à la fois les modifications du comportement de pêche et du fonctionnement de l'écosystème après 50 ans d'application de ces mesures de gestion. Nous avons observé un bénéfice notable de mesures de gestion combinant une fermeture de zones et une réduction d'effort sur la biomasse de la plupart des espèces commerciales, incluant la sole et la plie, et sur la valeur débarquée par unité d'effort. Une diminution de la biomasse des proies a été observée, bien que cette diminution ait été limitée par le changement de comportement de prédation dans le modèle. En effet, avec la diminution des biomasses des espèces de proie et l'augmentation de celles des prédateurs, les prédateurs consomment une proportion plus importante de juvéniles d'espèces prédatrices. La réponse de l'écosystème était variable suivant les métiers et espèces considérées. Dans notre étude, le coût d'exploitation n'a pas été explicitement considéré et l'impact des mesures de gestion sur les performances économiques des pêcheurs doit donc être considéré avec précaution.

Dans cette etude, nous avons pu mesurer les apports d'une approche de modélisation holistique, qui permet de pleinement intégrer la complexité du fonctionnement des écosystèmes dans l'AEP. Nous nous sommes principalement intéressés à l'écosystème entourant certains poissons plats et aux pêcheries les exploitants, et nous avons simulé l'impact de deux types de mesures de gestion. De plus amples développements du modèle devront être considérés pour améliorer nos connaissances sur le fonctionnement de l'écosystème de la Manche Orientale et pour répondre à d'autres questions sur sa gestion. Des développements futurs pourraient inclure des analyses de sensibilité et de propagation de l'incertitude. Une comparaison du modèle Atlantis EEC avec d'autres modèles écosystèmiques existants (EwE), ou en cours de développement (OSMOSE, ISISfish) sur la zone, pourrait également être réalisée afin de comparer les avantages et inconvénients respectifs de chaque modèle. Le modèle Atlantis EEC pourrait également être adapté pour analyser les conséquences potentielles d'une interdiction des rejets mise en œuvre dans le cadre de la nouvelle Politique Commune des Pêches (PCP), ainsi que l'impact du changement climatique, des phénomènes d'eutrophisation et de la dégradation des habitats. A plus longue échéance, le modèle Atlantic EEC pourrait devenir plus opérationnel et être utilisé en appui aux politiques publiques, et finalement être intégré dans un cadre formel de « Management Strategy Evaluation » (MSE).

Extended Abstract

The implementation of the ecosystem approach to fisheries management (EAFM) requires an enhancement of our knowledge of ecosystem complexity and of its interactions with fishing activities. Understanding both ecosystem and fishers' reaction to management regulation is a key to achieve conservation objectives. Since the past few decades, the development of ecosystem modelling contributed significantly to the improvement of our knowledge on ecosystem functioning in interaction with human activities, and it is now widely used to evaluate management strategies.

Anticipating fisher's behaviour is key to a successful implementation of EAFM. A major challenge for fisheries managers is to be able to anticipate how fishing effort is re-allocated following any permanent or seasonal closure of fishing grounds, given the competition for space and resources with other active maritime sectors. Firstly, a Random Utility Model (RUM) was applied to determine how fishing effort is allocated spatially and temporally by the French demersal mixed fisheries operating in the Eastern English Channel (EEC). The explanatory variables chosen were past effort to mimic experience or habit, previous catch and value per unit effort to represent economic opportunities, fishing effort from other fleets, and proportion of fishing area restricted to fishing activities due to management or maritime traffic. Fishers tended to adhere to past annual fishing practices, except the fleet targeting molluscs, mostly scallops, which was mainly influenced by seasonal patterns. Furthermore, results indicated French and English scallop fishers share the same fishing grounds, and maritime traffic may impact on fishing decision. Finally, the model was successfully validated, using two different approaches, by comparing predicted re-allocation of effort against observed effort.

Secondly, we conducted a meta-analysis of the outcomes of RUMs applications found in the fisheries science literature during the last three decades in various places around the globe. A methodology has been developed to standardize information across the different studies and compare the results they obtained. Six fishers' behaviour drivers have been considered: the concentration of other vessels, tradition, expected revenue, species targeting, cost, and risk-taking. We performed three separate linear models to analyse the extent to which these different drivers impact fisher's behaviour. First, a binary analysis showed that fishers are attracted by their expected revenue, tradition, species targeting and concentration of other users, and avoid choices involving large costs. Second we evidenced that active demersal fleets are generally more driven by seasonal patterns than by short-term information. Finally, it was evidenced that demersal vessels are generally risk–averse, and also that tradition and species targeting influence more fishers' decisions than expected revenue. Cost and concentration of other users have a similar impact on fishers' decision-making than revenue. Pelagic fleets appear to consider all drivers as important as expected revenue but due to the lack of information on this group, results have to be considered with caution.

To fully represent the interaction and impacts of fishers' behaviour on the EEC ecosystem a better understanding of ecosystem structure and functioning was required. Since the last decades we saw an increase of interest in ecosystem modelling and the development of many applications to better understand marine ecosystems functioning, which is crucial for improving future management plans of marine resources. We calibrated the ecosystem model Atlantis in our study to investigate the dynamics of the key Eastern English Channel ecosystem processes, with a particular focus on two commercial flatfish species, sole (*Solea solea*) and plaice (*Pleuronectes platessa*). This complex model was parameterized with data collected from diverse sources (literature review, surveys, logbooks, stock assessment and other model outputs) and tuned so the simulated catch and biomass fit the 2002-2011 averaged figures. Here, we mainly present the outputs for the two focus species and for some others vertebrates found to be important in the trophic network. The calibration process revealed the importance of coastal areas in the Eastern English Channel and of nutrient inputs from estuaries. It emerged that a lack of river nutrients decreases the productivity of nursery grounds and adversely affects the production of sole and plaice. The role of discards has been highlighted in the trophic network as well. Even if sole and plaice did not have a strong influence on the vertebrates' trophic network, they are important predators for benthic invertebrates and compete for food with crustaceans, whiting (*Merlangius merlangus*) and other demersal fish. In addition, two key species emerged from the ecosystem representation, cod (*Gadus morhua*) and whiting. Using an end to end model substantially improved our understanding of the Eastern English Channel ecosystem dynamics, and of the role of sole and plaice within that ecosystem.

In the last part of our study, we evaluated the consequences of management measures on both the EEC ecosystem and related fishing activities. We first coupled the Atlantis EEC ecosystem model with various fishers' behaviour models, to identify the fleet dynamics model that allowed the best fit between forecast and observed fishing effort. The model combining sole targeting and tradition as fishers' behaviour drivers was found to be the most suitable model. We then evaluated the consequences of implementing area closures and/or fishing effort reduction on the EEC ecosystem and on the French netters fleet targeting sole. We analysed both the modification of fishers' behaviour and ecosystem functioning after 50 years of management constraint. We observed a noticeable benefit of combining area closures and effort reduction on the biomass of most commercial species, including sole and plaice, and on the total value landed per unit effort. A decrease of the main prey' biomass was shown, however, this decrease was limited by a change in the predator-prey relationship. Indeed, with the decrease of the availability of prey functional groups, most of the large predators foraged on the juveniles of the main predator groups. The response of the ecosystem varied across the métiers and species considered. In our study, the cost of fishing was not explicitly considered, so the impacts of management measures on fishing performance should be considered carefully.

In this study we showed the crucial importance of applying a holistic modelling platform to fully encompass the complexity of ecosystem functioning in the EAFM. We focused mainly on the ecosystem and fishing activities related to some flatfish species and we simulated the impact of two management measures. Further developments should be considered to improve our knowledge on the EEC ecosystem functioning and to address management issues in a broader sense. Future developments should include analyses of sensitivity and of the propagation of uncertainty through the different model processes. A comparison of Atlantis EEC with other EEC ecosystem models, existing (EwE) or currently in development (OSMOSE, ISISfish), could also be conducted to test the respective strengths and weaknesses of those models. The Atlantis EEC model could also be adapted to investigate the potential consequences of the discard ban that will be gradually implemented in the context of the recent EU Common Fisheries Policy, and also the impact of climate change, eutrophication and habitat degradation scenarios. In the longer term the Atlantis EEC model could be made more operational, and used to inform future management, and eventually be formally included in a comprehensive MSE framework.

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Abbreviation

ABM: Agent-Based Model

CFP: Common Fisheries Policy

CGFS: Channel Ground Fisheries Survey

CPUE: Catch Per Unit of Effort

DCF: Data Collection Framework

DPMA: Directorate for Marine Fisheries and Aquaculture

DSVM: Dynamic State Variable Modeling

EAF: Ecosystem Approach to Fisheries

EAFM: Ecosystem Approach to Fisheries Management

EBM: Ecosystem-Based Management

EEC: Eastern English Channel

EMS: European Marine Sites

EUNIS: EUropean Nature Information System

EwE: Ecopath with Ecosim

FAO: Food and Agriculture Organization of the United Nations

GLM: Generalized Linear Model

IBM: Individual-Based Modelling

ICES: International Council for the Exploration of the Sea

IFDIdeal Free Distribution

IIA: Independence of Irrelevant Alternatives

IMP: Integrated Marine Policy

LRI: Likelihood Ratio Index

MCZ: Marine Conservations Zones

MPA: Marine Protected Area

MSE: Management Strategy Evaluations

MSFD: Marina Strategy Framework Directive

RUM: Random Utility Model

SACs: Special Area of Conservations

SMS: Stochastic Multi-Species model

SPAs: Special Protected Areas

SR: Stock Recruitment

SSB: Stock Spawning Biomass

TAC: Total Allowable Catches

VB: Von Bertalanffy

VMS: Vessel Monitoring System

VPUE: Value Per Unit of Effort

VSS: Vessel Separation System

Chapters

1. Chapter 1: Introduction

The overarching objective of this research thesis is to improve knowledge on the reciprocal interactions between fishing activities, other non-fishing human activities and marine ecosystem biological components, and also to explore the conservation and utilization consequences of various management measures aiming at mitigating the fisheries pressure. The concepts and methods developed in this thesis, e.g., in relation to ecosystem and fleet dynamics modelling, will be applied to the Eastern English Channel (EEC) maritime domain, with a particular attention paid to the ecosystem around, and the commercial exploitation of, flatfishes.

1.1. Interactions between fisheries resources, fishers and the wider ecosystem: moving from a single-species towards an ecosystem-based approach to management

Most fisheries resources worldwide are already fully exploited or over-exploited (FAO 2012). In 2009, 57 percent of fish stocks were considered as fully exploited, 30 percent were considered as over-exploited, and only 13 percent were characterised as under-exploited (FAO, 2012). The exploitation of marine living resources generates substantial value added at national and world levels. Indeed, in 2010, worldwide fisheries and aquaculture supplied about 148 million tonnes of fish representing a total value of US$217.5 billion. Of these 148 million tons, 131 million tonnes were directed to human consumption. The primary sector of fish production was estimated to provide income for 54.8 million people in 2010 (FAO, 2012). In the meantime, food security has become a growing concern, with demand and production increasing consistently along with the increase in human global population (Figure 1.1), with attendant increases in human pressure on marine ecosystems and in turn a growing list of consequences (e.g. eutrophication, littering and more broad scale effects of climate change and ocean acidification).

The complexities of fisheries management have been widely investigated (Crutchfield and Zellner, 1963; Daskalov and Mamedov, 2007; Gordon, 1954; Hilborn et al., 2001; Radovich, 1982), and some of these are highlighted by the tragedy of the commons (Hardin, 1968; Ostrom, 1990). Indeed, fisheries resources, similar to water, atmosphere and some lands resources, are common resources, and as such are characterised by a competition between multiple economic actors and a difficulty in regulating their access. In the absence of any form of management, the common resource inevitably becomes depleted, although some authors argue that the issue is more its free access rather than the common property of the resource by itself (Berkes et al., 1989; Ostrom, 1999). As a result, scientific advice to fisheries management should not only build on stock assessments and predictions in support of, e.g., TAC (Total Allowable Catches) setting, but also provide for a wider understanding of the dynamics of the human agents harvesting the common resources: the fishers.

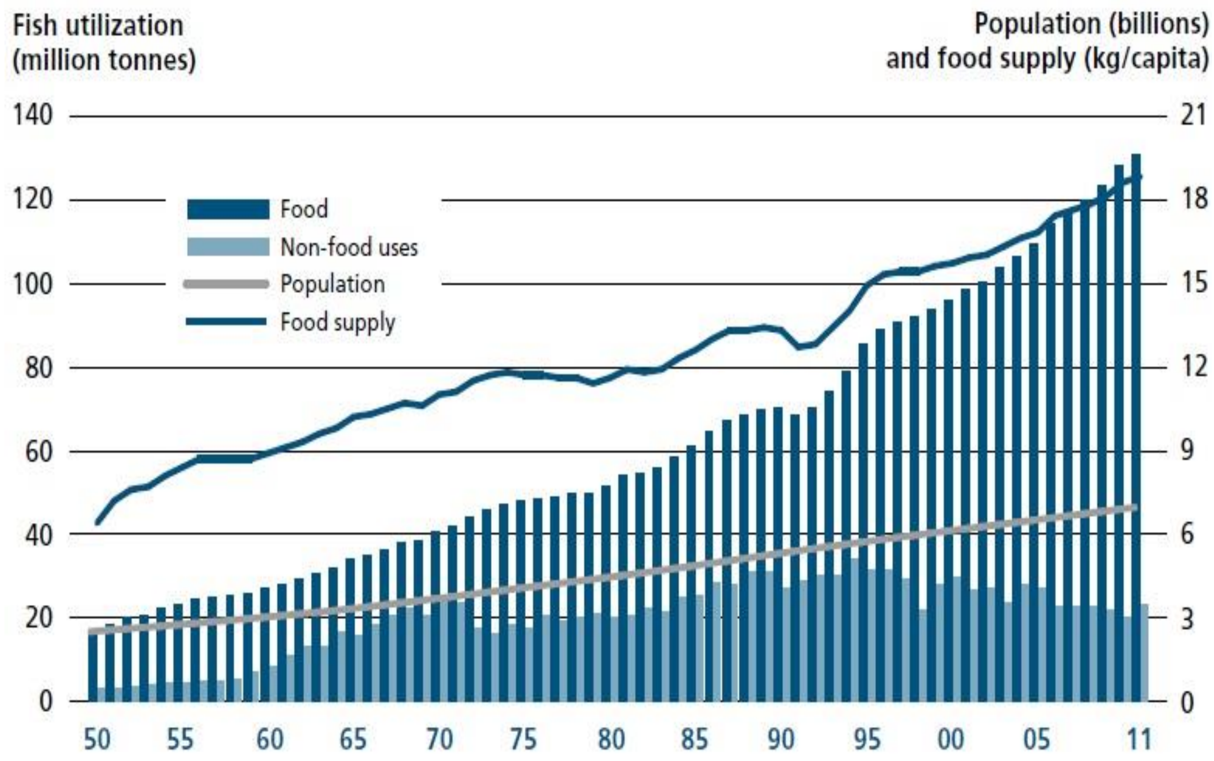


Figure 1.1 World fish utilisation and supply from 1950 to 2011 (source: FAO, 2012)

Plasticity more broadly – whether plasticity of fishers towards regulations, or fish distributions, markets, or competition with other fisheries or human activities – create problems for management and neglect of this plasticity has been repeatedly shown to contribute to fisheries management failures (Daw and Gray, 2005; Hardin, 1968); for example the stock collapses of Caspian sea anchovy (Daskalov and Mamedov, 2007), Californian sardine (Radovich, 1982), North Sea herring (Dickey-Collas et al., 2010) and a number of North Atlantic cod stocks (Poulsen et al., 2006; Walters and Maguire, 1996). While recruitment failures, competition with other species, or exceptional environmental conditions have been highlighted, studies of the stock collapses listed above all identified misunderstanding of fishers' reactivity combined with fisheries management complexity as one of the key drivers of the conservation outcomes (Allen and McGlade, 1987; Degnbol et al., 2006; Hilborn, 2007; Peterson, 2000).

Fishing activities have often been reported as the main human pressure exerted to date on marine ecosystems (Jackson et al., 2001). Indeed, fishing activity had been showed to have adverse effects on the structure and functioning of marine ecosystems (Buchen, 2009; FAO, 2012), first from their direct impact on fish stock biomass and natural habitat quality but also their indirect impact on trophic network functioning and on the population genetic drift. Learning from past failures and successes (Daw and Gray, 2005; Hilborn, 2004; Hilborn et al., 2001), management has gradually been moving from traditional single-species consideration (Garcia, 1994; Ludwig, 2002; McAllister and Kirchner, 2002; Rosenberg, 2002) towards a comprehensive ecosystem-based management (EBM) approach building in the full complexity of ecosystem interactions (Botsford et al., 1997; Browman and Stergiou, 2004; Garcia et al., 2003; Pikitch et al., 2004). In the past two decades, there has been an increasing societal and public demand from the governance, industries and non-governmental organizations to provide a sound and integrated scientific support to EBM (Arkema et al., 2006; Browman and Stergiou, 2004; Fulton et al., 2014; Garcia and Cochrane, 2005).This scheme has been applied to fisheries management, and it is referred to as the Ecosystem Approach to Fisheries Management (EAFM) (Browman and Stergiou, 2004). The EAFM aims at maintaining or restoring fisheries resources to sustainable levels, while mitigating the adverse ecological impacts of fishing (Pauly et al., 2002).

In the EU, this move towards a more holistic and inclusive approach to the management of marine resources is reflected by the inception, in 2008, of the cross-sectorial Marine Strategy Framework Directive (MSFD) (EC, 2008). The aim of the MSFD is to encourage sustainable use of marine resources, in accordance with current policies including the holistic EU Integrated Marine Policy (IMP) (EC, 2011) and, when it comes to fisheries management, the Common Fisheries Policy (CFP) (Roth and O'Higgins, 2011). A prerequisite to the effective application of the EAFM is to better understand the different components of ecosystems, the processes driving the dynamics of marine ecosystems and of the fishing fleets that impact them (Degnbol et al., 2006; Fulton et al., 2011a, 2014; van Putten et al., 2012; Wilen et al., 2002). Moreover, understanding each phase of the fishery system, from the assessment of the ecosystem health and the impact of fisheries to the application of management rules, is necessary to fully evaluate the performance of different management strategies (Figure 1.2) (Fulton, 2010).

1.2. Objectives and plan of the thesis

The first objective of this thesis is to identify and quantify the factors determining fishers' behaviour in the Eastern English Channel, including spatial interactions with other fleets (French or international), other sectors of activity (maritime traffic) and area-based management. This first approach provides insights into the fishers' decision-making process, which is required to enhance the efficiency of marine ecosystems management. The relative weighting of the drivers of EEC fisheries is then contrasted to that of other fisheries around the world.

To fully understand the knock-on effects of EEC flatfish fisheries on sole and plaice and on the marine ecosystem embedding these species, I have developed a modelling platform that mimics the ecosystem functioning of the EEC including fishing behaviour (objective 2). Finally, to simulate the ecological and utilization impacts of various conservation and access restrictions to the ECC maritime domain (objective 3), I have coupled a fleet dynamics model developed in relation to the first objective and the ecosystem model developed in relation to the second objective.

This thesis is composed of four scientific papers, each forming one chapter of this thesis (one published - chapter 2 - one under review - chapter 4 - and two others in preparation).

Chapter 2 is dedicated to the analysis of the main factors driving fishers' behaviour in the Eastern English Channel. We applied a Random Utility approach (RUM) to estimate the main drivers of fleet dynamics in the French demersal mixed fishery. Several other human activities have been considered: (i) other French fleets, (ii) United Kingdom fleets, (iii) shipping, and (iv) fishing restriction in the 12 miles coastal area.

In Chapter 3, I perform a review of the fleet dynamics and fishers' behaviour studies around the globe over the last three decades. We focused on approaches using RUM to highlight drivers of fishers' behaviour. Multiple analyses have been performed to extract and summarize the main components of fishers' behaviour.

In Chapter 4, we described the development of a whole ecosystem end-to-end model, Atlantis, to capture the salient features of the EEC ecosystem functioning (and more particularly focusing on the ecosystem embedding of sole and plaice). We gathered all the information available to calibrate each part of the ecosystem from the biogeochemical cycle to the fishing activities in the EEC. The model is mainly calibrated using observed catch data over the period 2002-2010, and its goodness of fit is evaluated.

In Chapter 5, I first tested the relevance of coupling two alternative fleet dynamics model to the Atlantis model developed previously. The first model tested is the RUM developed in Chapter 2 and the second one is a gravity model, weighted based on information gathered, described in Chapter 3. The fleet dynamics models have been applied to the netters fleet mainly targeting sole.

Finally, Chapter 6 summarizes the main conclusions of this thesis and analysed the limits and assumptions made in this study. An account of the future development of this work is then presented.

1.3. Fishers' behaviour: a key process for the management of marine resources.

1.3.1. Description of fishers' behaviour

Fishers are key components of marine ecosystems, and understanding their behaviour is critical if we are to anticipate their likely responses to management measures (in terms of, e.g., spatial allocation of fishing effort, discarding practices), and the knock-on effects on impacted ecosystem components (Fulton et al., 2011a; Hilborn, 2007; Leslie and McLeod, 2007; Wilson and McCay, 2001).

The mechanisms of change in the behaviour of human agents have been widely studied using a variety of approaches. Van Putten et al. (2012) present an overview of the different models and theories proposed and applied over the past three decades to explain and predict fishers' behaviour. In particular, (van Putten et al., 2012), characterised fishers' decision-making into short-term and long-term choices also termed as tactics and strategies, respectively (Laloë and Samba, 1991), using five behavioural types. The short-term processes were: (i) effort location choice and (ii) discarding; and the long-term processes were: (iii) exit/entry; (iv) compliance with management rules and (v) investment (Figure 1.3).

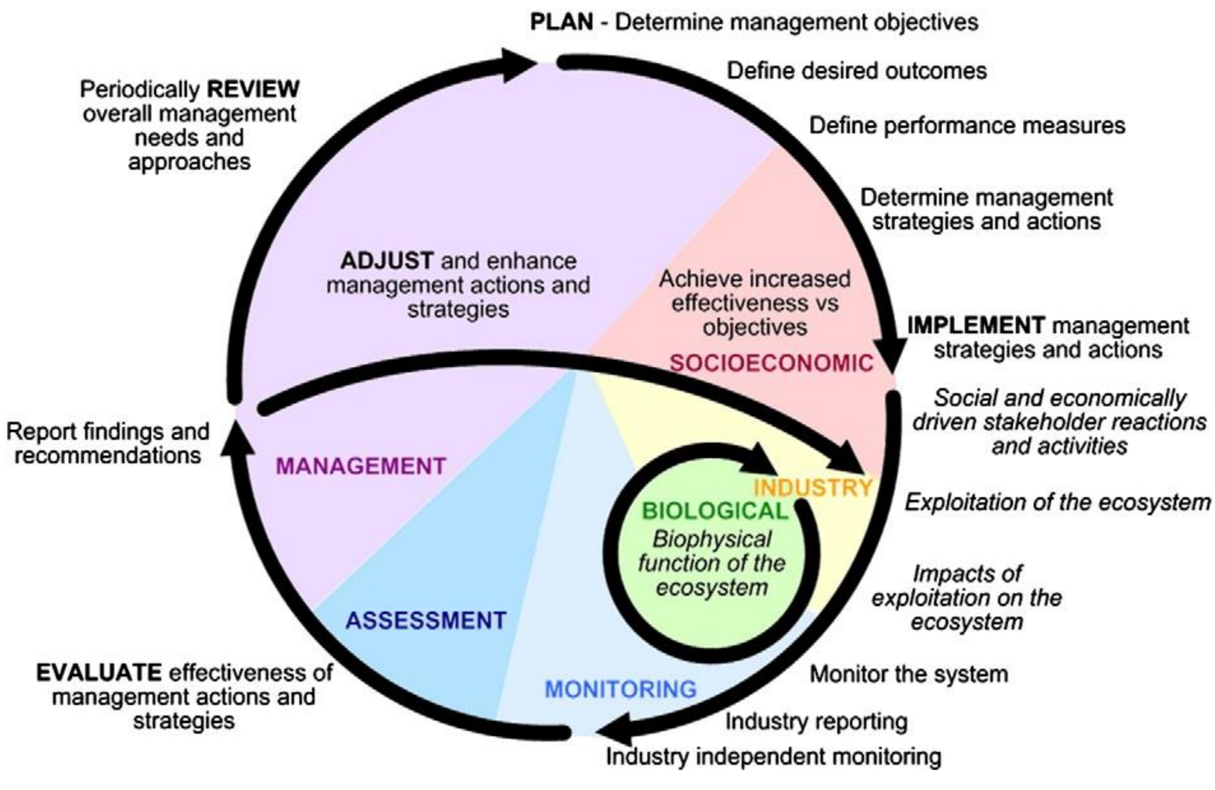


Figure 1.2 Adaptive management cycle (source: Fulton, 2010)

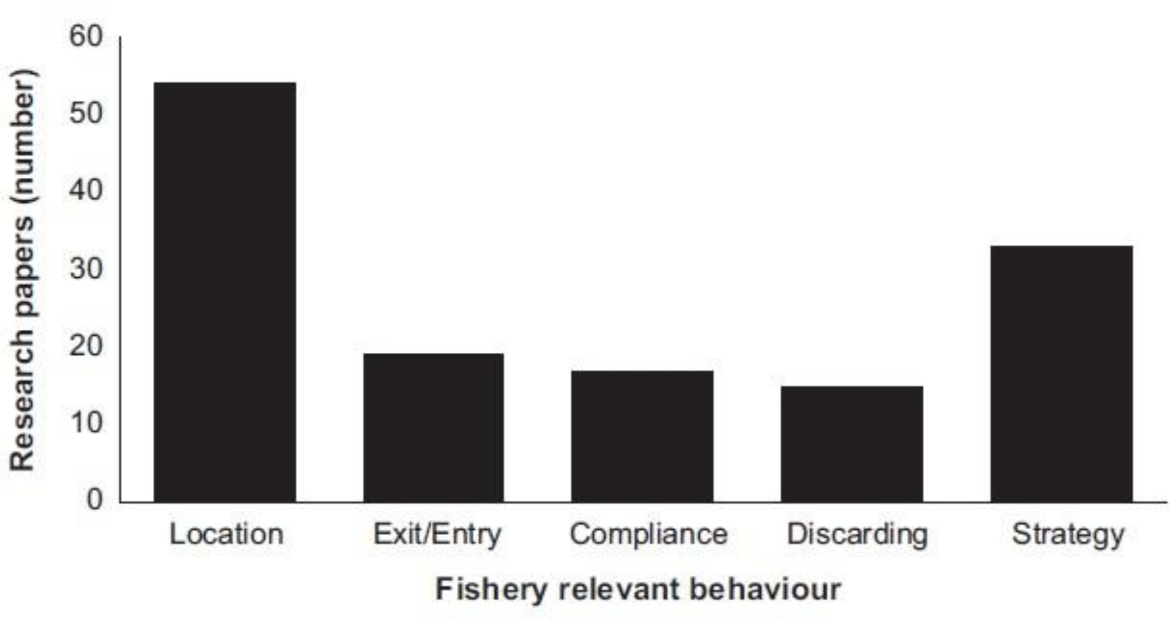


Figure 1.3 Categorization of 141 reviewed publications by van Putten et al., 2012 into five types of fishery-relevant behaviour.

In this thesis, I have investigated fishers' short-term behaviour through location choices. To provide context for this work I present below a summary of the different modelling and conceptual approaches that have been developed in the past three decades. Some consider the shape of vessels' trajectories (Section 1.3.2.1), others the distribution of fishing effort (Section 1.3.2.2). I characterise a particular focus on Random Utility Models, which I have relied on extensively to provide fisher behaviour in the models presented here (Section 1.3.2.3).

1.3.2. Modelling fishers' location

Dorn (1998) characterised the behaviour relative to tactics decision as a two class set of choices: (i) the choice of fishing area in medium term and (ii) the choice to fish or not (and resulting vessel trajectories) in the short term.

1.3.2.1. Vessel trajectories and fishing states modelling

Since 2000, the use of land based satellite tracking and monitoring devices (VMS for Vessel Monitoring System) has made it possible to analyse the vessel trajectories and to estimate fishing states at a fine spatial and temporal resolution, with the objective of improving the precision of fishing effort metrics. The advance made possible by this data was the separation of steaming (plain travelling) from fishing operations, or the distinguishing of different types of fishing activities (or métiers) that fishers may execute while at sea. The analysis of vessel trajectory is mainly performed using random walks or the Lévy motion (Bertrand et al., 2005, 2007), while fishing state investigations typically applied bayesian approaches such as hierarchic models and hidden Markov chains (Vermard et al., 2010). These modelling approaches are certainly promising approaches to fleet dynamics investigations. However, due to the non-availability of VMS data at the beginning of this thesis, and considering the complexities of forecasting vessels trajectories based on current approaches, we focused here on more traditional fishing effort allocation models, as detailed below.

1.3.2.2. Fishing effort allocation modelling

In this study we concentrated on short-term behaviour, and in particular the factors that determined fishing effort allocation both spatially and across métiers (Andersen et al., 2012; Hilborn, 1985; Hutton et al., 2004). An increasing number of studies have investigated and modelled short-term fishers' behaviour using both conceptual and data-driven approaches. Conceptual approaches include applications of the Ideal Free Distribution (IFD) theory (Gillis, 2003; Rijnsdorp et al., 2000), optimal foraging theory (Dorn, 2001), Individual-Based Modelling (IBM) (Millischer and Gascuel, 2006; Soulié and Thébaud, 2006) and gravity models (Caddy, 1975). Many data-driven approaches to fishers' behaviour modelling have built in Random Utility Models (RUMs)(Andersen et al., 2012; Holland and Sutinen, 1999; Hutton et al., 2004; Marchal et al., 2009; Pradhan and Leung, 2004; Vermard et al., 2008; Wilen et al., 2002).

Foraging theories were first applied in animal behavioural ecology, and then extended to fisheries ecology. Fishing vessels are considered as individual foragers that aim at optimizing their choices (e.g., spatial allocation of fishing effort) to maximize their revenue with a minimum cost (Gordon, 1954; Hilborn and Kennedy, 1992). One of the main foraging theories is the IFD. The IFD is based on several strong hypotheses: (i) the quality of the resources is not impacted by the fishing activity, (ii) fishers have a total knowledge of the spatial and temporal distribution of the resources and (iii) they are able to choose any location without any constraint (Abernethy et al., 2007; Gillis, 2003; Gillis et al., 1993; Rijnsdorp et al., 2000). Provided these assumptions are valid, the number of vessels allocated in each fishing area is then proportional to the relative resource density (e.g., as reflected by survey indices of the catch per unit of effort - CPUE) available in that zone. However, even if IFD has been already applied in case of management constraint, the hypothesis of full stock knowledge is generaly unrealistic in the case of versatile and uncertain fishery resources.

An alternative to a forage based approach is the IBM approach, also called agent-based models (ABM), which explicitly recognises that individual fishers do not respond in the same way to their environmental, ecological, economic or management drivers. Fishers' behavior is then assumed to be driven by a set of rules that may differ across individuals, depending on their own intrinsic characteristics. The evolution of the characteristics, rules and decisions are tracked through time and space for each individual. Individuals are able to interact with each other, either through information-sharing or competition in each area. Several theoretical fishers' behavior studies building in IBMs have been carried out (Maury and Gascuel, 2001; Millischer and Gascuel, 2006; Soulié and Thébaud, 2006). However, due to the data-intensive and computationally-demanding character of those model only few studies have been applied to real case studies (Helu et al., 1999; Jules Dreyfus-León, 1999; Little et al., 2009).

Another model type is the gravity model (Caddy, 1975), which was originally applied to predict the effort allocation between different métiers and areas. The distribution of the total effort is considered to be proportional to the relative attractiveness of each métier and area. The attractiveness is measured as a proportion of the expected profit of a métier in a given function to resource availability and cost of fishing. It can integrate the notion of communication between vessels, the distance to the harbour or also the price of target species (Allen and McGlade, 1986; Walters and Bonfil, 1999; Walters et al., 1993). Gravity models are used extensively in the "Ecospace" modelling framework to redistribute fishing effort.

Other types of fleet dynamics models have been developed such as the dynamic state variable modeling (DSVM) approach (Babcock and Pikitch, 2000; Gillis et al., 1995; Poos et al., 2010a, 2010b), game theory as applied to fisheries (Bailey et al., 2010; Lindroos et al., 2007), or network theory based approaches (Ramirez-Sanchez and Pinkerton, 2009). More details on limits, weakness and robustness of these approaches may be found in van Putten et al. (2012).

Amongst the most popular models of effort allocation that have been developed in the past three decades are Random Utility Models (RUMs), and these have been used extensively in this thesis. In Chapter 2, we applied a RUM to understand and highlight the main drivers of the Eastern English Channel fleets dynamics. In Chapter 3, we conducted a meta-analysis of RUM applications around the globe. In Chapter 5, we evaluated the prediction performance of various RUMs and gravity models when coupled to an ecosystem model. We present in Section 1.3.2.3 the theory underpinning RUMs as well as their main application domain and case studies.

1.3.2.3. Discrete choices modeling and utility maximization theory applied to fishers' behaviour: the random utility model

The mechanisms of change in the behaviour of human agents have been widely studied using a variety of approaches. One of the most dominant approaches found in the economic literature is discrete-choice modelling (McFadden 1974; Greene, 2003; Train, 2003). A founding principle of discrete-choice models is that an agent facing multiple choices allocates a utility to each alternative, and then chooses that with the greatest utility. Discrete-choice models in the form of a random utility function, also known as Random Utility Models (RUMs), have been applied in various disciplines including, e.g., households' and consumers' preferences (Bougherara et al., 2009; Gracia and de Magistris, 2008; Zhang et al., 2009), school choice (Cohen-Zada and Sander, 2008; Glick and Sahn, 2006), or travelling options (Ettema et al., 2007; McFadden, 1974).

RUMs provide an appropriate and functional approach to describe how fishers make a choice among a panel of finite alternatives (Wilen et al., 2002) and allow incorporation of economics and non-economic variables. Such a discrete-choice modelling approach has been applied to analyse how fishers choose their fishing grounds (Hutton et al., 2004; Tidd et al., 2012; Wilen et al., 2002), their target species (Marchal et al., 2014a; Pradhan and Leung, 2004; Vermard et al., 2008), their fishing gear (Andersen et al., 2012; Holland and Sutinen, 1999) or a combination of these (Marchal et al., 2009).

In the case of fishery studies, the utility function can depend on various variables such as the past experience or tradition, the expected profit or revenue, the vessel characteristics and the target species. In most fleet dynamics studies, skippers have been assumed to choose their fishing ground, gear and/or target species, based on their own experience (e.g. their past choices/activity) and on their economic expectations for a given choice (e.g. past profit achieved). For example, fishers' behaviour can be influenced by fish price fluctuations, which are often seasonal and are an important factor to take into account when evaluating the expected profitability of alternative potential choices (Dupont, 1993; Loannides and Whitmarsh, 1987). Anecdotal evidence suggests that other factors which have seldom been considered in past empirical studies could determine fishers' behaviour. These factors include communication between fishers, or radar-screening of concurrent vessels which may indicate the presence of target species in a specific area. By contrast, skippers compete for space and resources, not only with other fishers, but importantly also with other sectors of activity operating in the same maritime area. Exploitation of marine resources, for example aggregate extraction, offshore wind farms and maritime traffic can impact the choice of fishing grounds by restricting access or decreasing the availability of fish resources. In Chapter 2, we explore the relative importance of those factors on fishers' decision making, in the case of the Eastern English Channel fishing fleets.

The RUM is fitted on individual observed data and the coefficients estimated for each driver can be used to predict fishers' decision and then used as a basis for testing various scenarios of management (Holland and Sutinen, 1999; Hutton et al., 2004). Prediction performances have been tested and compared in Chapter 2, using a standard and a novel method of allocating fishing effort based on the model outputs.

However, to evaluate the long-term conservation and utilization performances of different scenarios of management, it is necessary to model not only the dynamics of the fishing fleets, but also the dynamics of fish population and of the entire marine ecosystem. In Chapter 5, we have thus coupled a variety of fleet dynamics model with a comprehensive end-to-end ecosystem model. We provide in Section 1.4 a review of existing marine ecosystem models (Sections 1.4.1 and 1.4.2), and a deeper description of the Atlantis end-to-end model that has been developed and used during this thesis (Section 1.4.3).

1.4. Understanding ecosystem functioning with the help of ecosystem models.

Over the last few decades, interest in ecosystem modelling has grown substantially (Arkema et al., 2006; Brodziak and Link, 2002; Browman and Stergiou, 2004; FAO, 2003; Fulton, 2010; Garcia et al., 2003; Sanchirico et al., 2006), and as a consequence, understanding of marine ecosystems has improved, and advice-givers have increasingly recognized the importance of accounting for ecosystem dynamics.

1.4.1. End to end model to represent the complexity of marine ecosystems

Different end-to-end models have been developed to emulate the dynamics of marine ecosystems. In some of these models, human activities are considered as a full component of the ecosystem (Leslie and McLeod, 2007), rather than a forcing driver. Plagányi (2007) performed a review of some of the models available to implement the ecosystem approach to fisheries. Hereafter, we only consider models that were characterized as whole ecosystem models or dynamic system models by Plagányi (2007). Plagányi (2007) presented a modified categorization of ecosystem models building on Hollowed et al. (2000). This framework classified each model in accordance to its complexity and the type of ecosystem processes being represented (Figure 1.4 andFigure 1.5).

Ecosystem dynamics are driven by different types of processes, including hydrodynamics, biogeochemistry, habitat characteristics, life cycles, trophic relationships, as well as the interaction with human activities. Coupling these different ecosystem components in holistic models is necessary to mimic the effects they exert on each other, and the extent to which such interactions could explain the fluctuations observed in key ecological or exploitation variables, e.g., biomass, catches, fishing effort (Fulton, 2010; Travers et al., 2007). One of the first ecosystem models that included a large set of species groups is the Ecopath with Ecosim (EwE) model. EwE represents the trophic connections in ecosystems (Christensen and Walters, 2004). EwE applications are now widely spread across the world and often include interaction with fishing activities. Lately, other models have emerged, focusing on mixed fisheries dynamics, e.g., ISIS-Fish (Mahévas and Pelletier, 2004), or taking into account other ecosystem processes such as nutrient cycling , environmental variability and habitats: e.g., OSMOSE (Shin and Cury, 2001, 2004; Travers et al., 2009; Travers-Trolet et al., 2014), APECOSM (Maury et al., 2007), NEMURO.FISH (Kishi et al., 2011), SEAPODYM (Lehodey et al., 2008) and Atlantis (Ainsworth et al., 2012; Fulton et al., 2005, 2007; Kaplan et al., 2012; Savina et al., 2013).

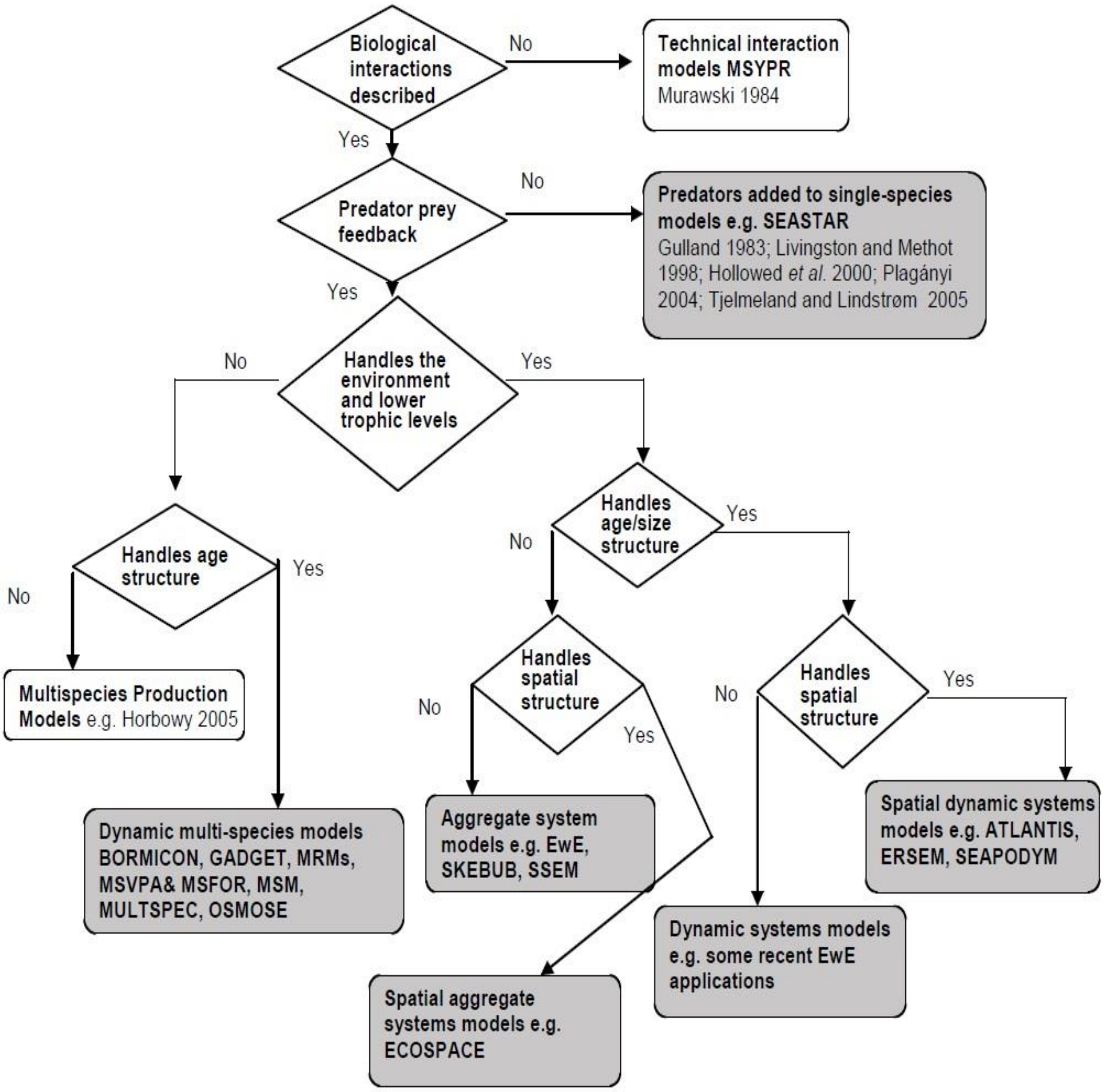


Figure 1.4 Flowchart representing the classification of models reviewed in Plagányi (2007) and modified from Hollowed et al. (2000).

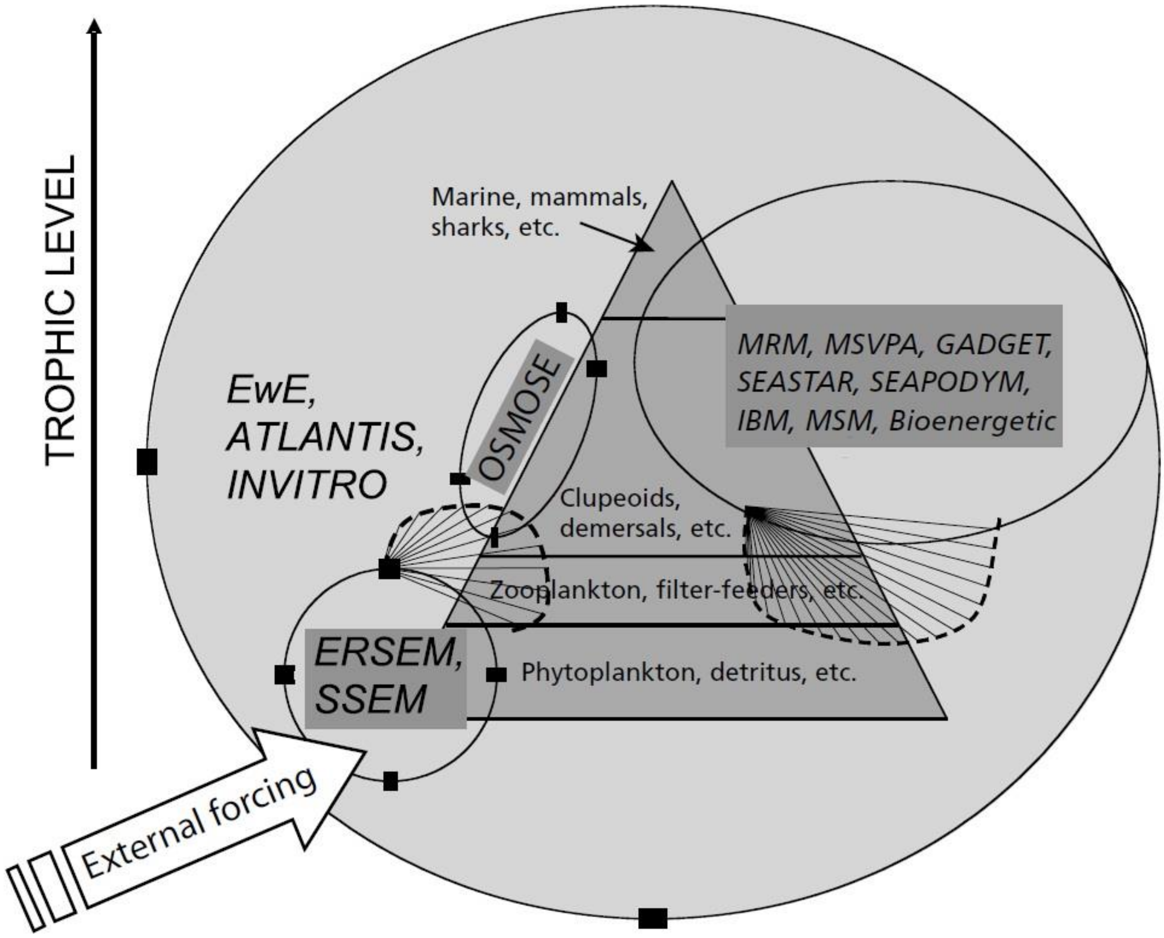


Figure 1.5 Trophic level focus of different multi-species models (Plagányi, 2007)

1.4.2. Description of a selection of ecosystem models

EwE is an entire trophic network model that represents the relation between the functional groups and between the functional groups and the fisheries. ECOPATH is a mass balance model and reflects the state of the ecosystem at a given time. The development of ECOSIM allowed representation of time varying dynamics within the ecosystem. Further developments of the model included the possibility of applying age-structured functional groups (Christensen and Walters, 2004), or spatially-explicit models (ECOSPACE) (Daskalov and Mamedov, 2007) that includes a gravity model to redistribute fisheries effort (Walters et al., 1999).

SEAPODYM (Spatial Environmental Population Dynamics) has been developed to understand the interactions that occur within tuna fisheries of the Pacific Ocean (Lehodey et al., 2008). This is a two dimensional model, coupling physical-biological interactions. SEAPODYM is also age-structured and models both fish migrations and the movement of nutrients based on advection-diffusion equations. SEAPODYM has been recently enhanced with the implementation of spawning temperature dependence, finer scale spatial and temporal resolution, integration of management scenarios (Lehodey et al., 2011).

OSMOSE is an IBM that follows the traits of individuals through their life cycle (Shin and Cury, 2001, 2004). This model is two dimensionally resolved and uses simple rules to model prey-predator relationships: trophic relations are size-based and also depend on spatial overlap. OSMOSE focuses mainly on fish species, both commercial and non-commercial, gathered in schools moving in 2 dimensions. In the lastest developments, OSMOSE has been forced by plankton distribution data (Marzloff et al., 2009) and fully coupled to biogeochemical models (Travers et al., 2009) to allow the consideration of climate effects on fish communities dynamics (Travers-Trolet et al., 2014).

ISIS-fish focuses on the dynamics of fishing fleets and of commercial species populations more than on ecosystem dynamics. ISIS-Fish is particularly well-suited to modeling mixed fisheries dynamics and is divided into three sub models, the fishery, the biology and the management, all spatially resolved in two dimensions. Trophic interactions, however, are not explicitly considered in this model (Mahévas and Pelletier, 2004).

In Chapter 4, we applied Atlantis, one of the most comprehensive marine ecosystem model developed in the world. Compared to the other ecosystem models presented above, Atlantis integrates all ecosystem processes, ranging from nutrients all the way through fisheries and management. Atlantis was originally developed to evaluate the performance of management strategies (Fulton et al., 2005, 2007) and the structure of the Atlantis model is detailed in Section 1.4.3.

1.4.3. Atlantis: a comprehensive ecosystem end-to-end model.

Atlantis is an end-to-end modeling framework built in the context of management strategy evaluations (MSE) (Fulton et al., 2005). It represents explicitly each important component of the management process (Jones, 2009), including the biophysical system, trophic interactions, the human users of marine resources, the three major components of an adaptive management strategy (monitoring, assessment and management decision processes) and socioeconomic drivers of human behavior. Atlantis includes dynamic, two-way coupling of all these system components. The model is coarsely 3D spatialized and it emcompasses of explicit physics and biogeochemical dynamics. The use of a biogeochemical framework allows the representation of both bottom-up and top-down controls (Fulton et al., 2011b). There are currently >30 Atlantis model applications worldwide, and there are several others currently under development (Baltic Sea, North Sea, Strait of Sicily).Ecosystem and fisheries modules are fully detailed in Chapter 4 and 5, which also present the application of this model to the Eastern English Channel. The monitoring, assessment and management submodels are not detailed as they were not used in this thesis, but detailed information on these packages can be found in Fulton et al. (2011b).

Each Atlantis sub-model is a deterministic, spatially-resolved model and discretized into irregular spatial polygons (allowing a match with the salient ecosystem spatial features). This structure facilitates the tracking of nitrogen flows through the trophic network. Relations in this network are implemented with differential equations with a time step of 12 or 24h. The main processes in the ecosystem are represented in the model: consumption, production, waste production, movement and migration, predation, recruitment, habitat-dependency and mortality. Functional groups can be either biomass pools (mainly used for invertebrates) or age-class structured (often used for vertebrates). The fishing sub-model allows for multiple fleets, each with its own characteristics (including gear selectivity, habitat association, targeting, and effort allocation). Atlantis also makes provision for fleet dynamics, economics and dynamic management rules, which were not considered in the context of this study (Fulton et al., 2011b).

Plagányi (2007) defined Atlantis as the best model currently available to assess the functioning of entire marine ecosystems – from the environmental variables through to the economics and management cycle. This is because:

1) Atlantis encompasses the whole trophic network in a single modelling framework;
2) Atlantis represents more efficiently physiological processes compared to other biogeochemical models;
3) Vertebrates are age-structured, which allows for gear selectivity to be represented;
4) Invertebrates can be separated by stock and class of maturity (adult or juveniles) and the upper trophic level is better represented than in other biogeochemical models;
5) Atlantis is 3D spatially-resolved;
6) Atlantis has a modular structure allowing for the application of a variety of process representations.
7) The nutrient pool formulation allows the analysis of increased nutrient inputs from river and eutrophication scenarios.
8) The coupling between physical and biological processes is well detailed.
9) The complexity of Atlantis makes the pametrization, the validation and the achievement of long-term stability very difficult to obtain.

1.5. The Eastern English Channel ecosystem and dependent human activities

1.5.1. Commercial fish and fisheries

Several countries operate in the EEC, such as, France, the United Kingdom, the Netherlands, Belgium Germany, Ireland and Denmark. The main fisheries present in the EEC over the period 2002-2010 were the French fishery with 56% of the landings, followed by the Dutch with 23% and the UK with 15%. Here, we focused on the French fleets targeting sole (*Solea solea*) and plaice (*Pleuronectes platessa*) in the EEC. This fishery presents several specificities, which need to be accounted for in subsequent analyses. First, it is a mixed demersal fishery, consisting of several fleets (netters, bottom trawlers, dredgers or passive demersal) using different métiers (trammel net, dredge, pots, otter bottom trawl, mid-water otter trawl) and targeting different species assemblages during the year. Moreover, even if these fleets target sole, one of the most valuable commercial species in the EEC, or plaice, they may also catch other species such as cod (*Gadus morhua*), whiting (*Merlangus merlangus*), seabass (*Dicentrarchus labrax*), herring (*Clupea harengus*), scallops (*Pecten maximus*), squid (*Loligo forbesii* and *Loligo vulgaris*), or cuttlefish (*Sepia officinallis*). This implies an important number of alternative choices available (area, target species, métier choice) to fishers and complicates the analysis of human behaviour in such fisheries. Second, plaice and sole are two benthic flatfish species, which strongly depend on benthic environment and on the quality of nursery grounds (Arbach Leloup et al., 2008; Cugier et al., 2005; Dauvin and Desroy, 2005; Loizeau et al., 2001; Moore et al., 2004; Riou et al., 2001; Rochette et al., 2010). Finally, this fishery operates in one of the busiest maritime areas in the world. Fishing fleets compete for space and resources with other fleets (French or international) and other human activities (recreational fisheries, gravel extraction, shipping, MPAs, etc.), which we may potentially affect fishers' behaviour.

1.5.2. The Eastern English Channel ecosystem

The EEC is a shallow epicontinental sea covering a total area of approximately 35,000 km². It is delimited by the United Kingdom in the North and France in the South. The ECC is connected to the North Sea by the Dover strait in the East and is separated from the Western English Channel by the Cotentin peninsula (Figure 1.6).

The EEC is a shallow sea with a maximum depth of approximately 100 m in the western part, off the Cotentin peninsula, and a maximum depth of 40m in the eastern part (from 0° to 2°E), the Dover strait (Figure 1.7). As a result the EEC is strongly influenced by tidal current, with residual currents mainly coming from the West and going Eastwards(Salomon and Breton, 1991). These currents result in structuring the distribution of sediments in the EEC (Larsonneur et al., 1982) (Figure 1.8). Gravels and pebbles are found in the middle of the EEC with the strongest currents, while mud and sands are mainly distributed in coastal area, nearby bays and estuaries (Dauvin and Lozachmeur, 2006). In the

Dover strait, the residual tidal current circulation goes up to 120 $m^3.s^{-1}$ and represents nearly 30% of the flow rate entering the North Sea. The formation of gyres near the coastal area creates local retention areas that impact the circulation of nutrients in the EEC and influence the distribution of both sediments and primary production (Figure 1.9).

Figure 1.6 The Eastern English Channel, delimitated by English and French coasts, the Dover Strait, and to the West by a line from the Cotentin Peninsula to Bournemouth (Carpentier et al., 2009)

Salinity and temperature in the EEC are strongly influenced by river inputs. In autumn, an inversion of gradient occurs with coastal water becoming colder than waters from the middle of the ECC during winter, while in summer, coastal waters are warmer than the waters from the middle of the ECC. In addition, due to river inputs the salinity is lower close to the coast than offshore for all seasons (Carpentier et al., 2009).

The shallow depth and the strong currents that characterize the EEC induce a constant mixing of the water column. This hydro-dynamism leads to a constant resuspension of nutrients and detrital matter, especially in the coastal area. This increases the turbidity and decreases the light penetration in the EEC (Figure 1.10). The nutrient inputs from rivers favours primary production; although in the years with strong currents and turbidity, light penetration is decreased, which may adversely affect production. As in most temperate regions, the main bloom of phytoplankton occurs during spring. A second phytoplankton event, less intense than the spring one, may be observed in autumn. The EEC is not subject to strong upwelling, so the phytoplankton blooms are highly correlated with the river nutrient inputs and hence are mainly concentrated in estuaries (Figure 1.10).

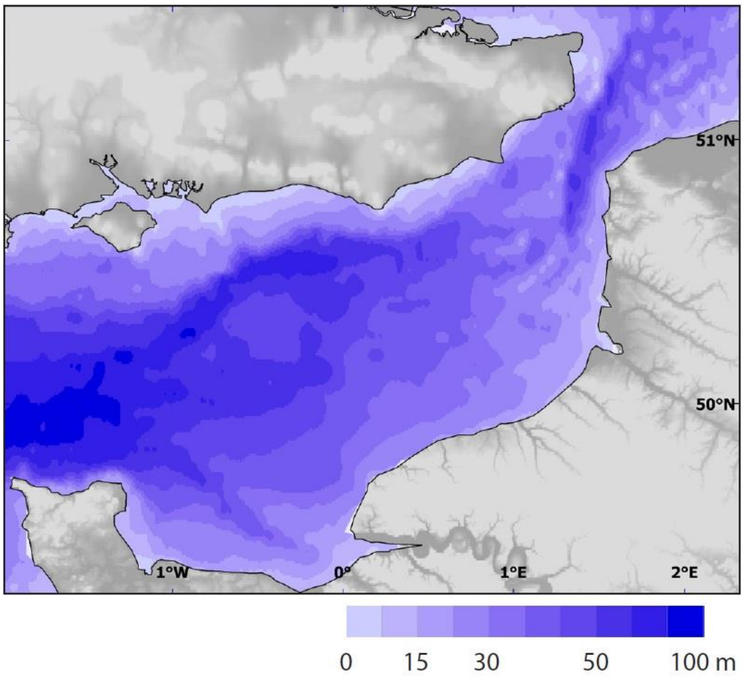


Figure 1.7 Bathymetry in the Eastern English Channel (Carpentier et al., 2009)

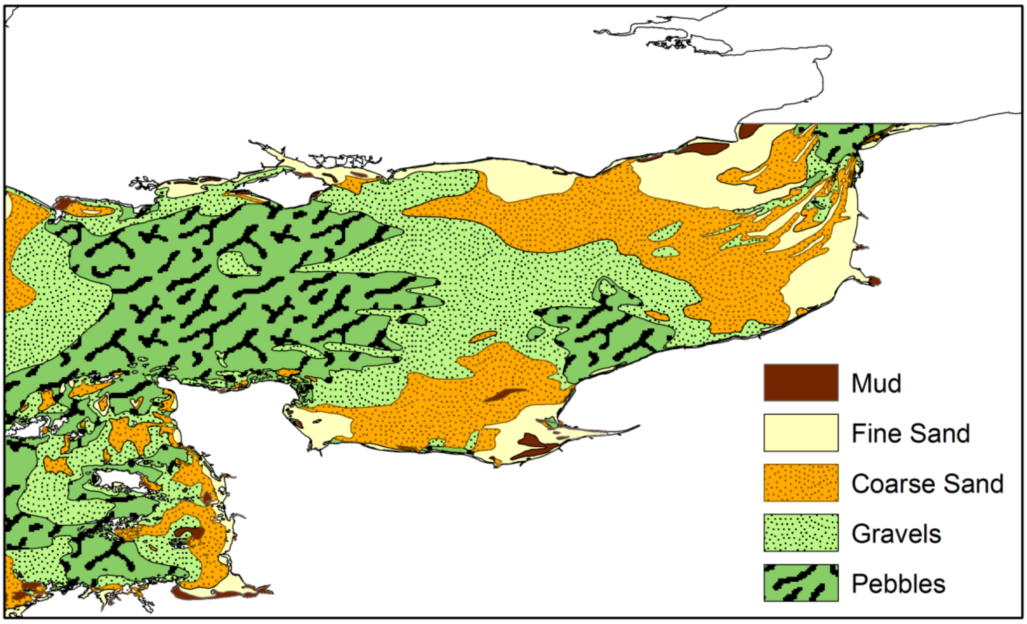


Figure 1.8 Sediment types in the Eastern English Channel, as derived from Larsonneur et al. (1982)

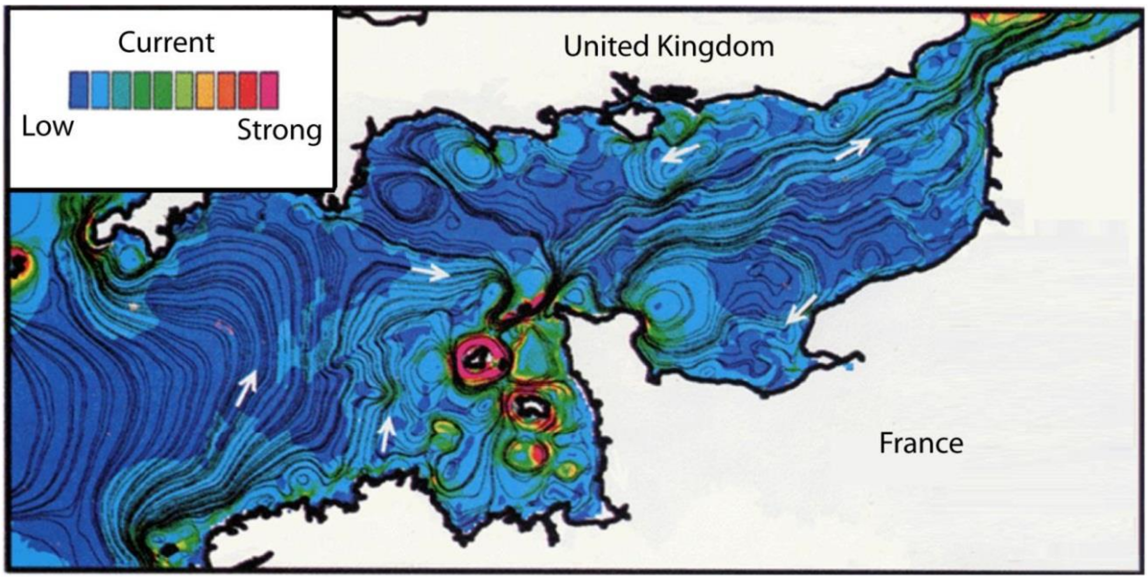


Figure 1.9 Residual tidal currents in the English Channel (Salomon and Breton, 1991)

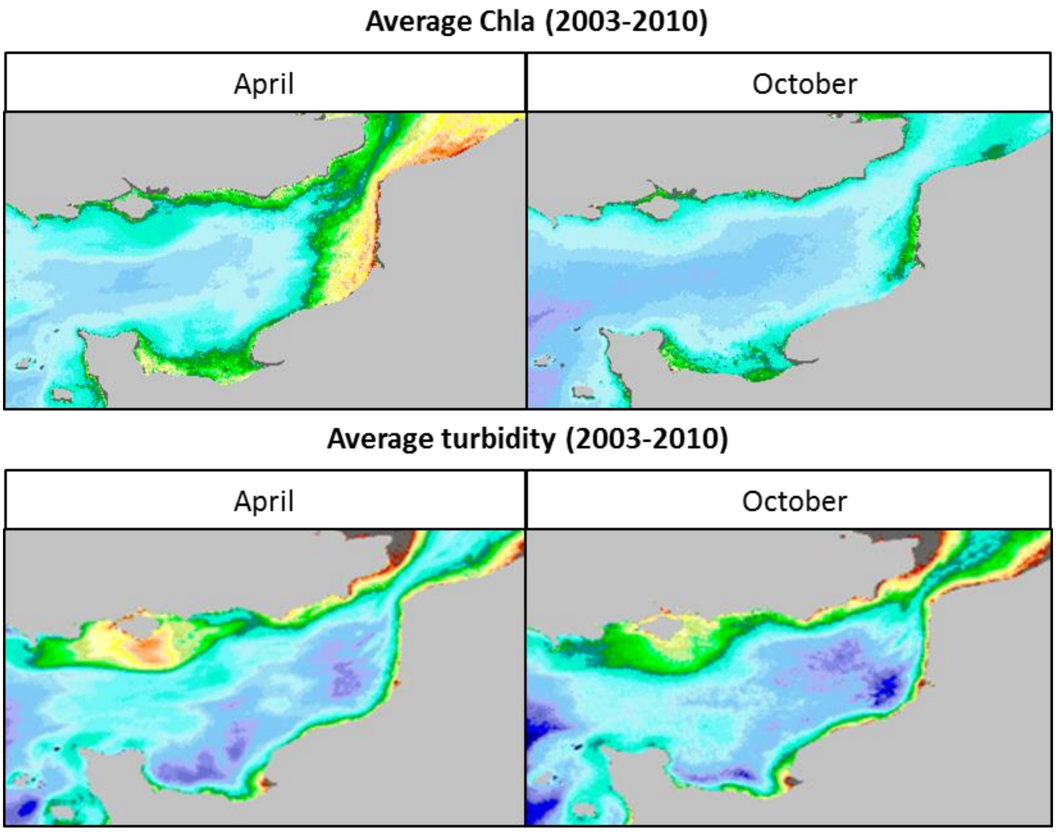


Figure 1.10 Average chlorophyll-a and turbidity in April and October between 2003 and 2011. Estimated by Ifremer from combined satellite and in-situ observations.

A classification of the different benthic habitats has been performed by (Cabioch et al., 1978) (Figure 1.11). This distribution of benthic habitats is based on the common European framework EUNIS

(EUropean Nature Information System), which includes both abiotic environmental information and the main biological benthic taxa. The benthic community in the EEC is a key component of this shallow ecosystem. It is a source of food for many vertebrates, and sole and plaice are particularly dependent on the abundance of benthic fauna and also on its species composition (Arbach Leloup et al., 2008; Cachera, 2013; Carpentier et al., 2009; Dauvin and Desroy, 2005; Loizeau et al., 2001; Moore et al., 2004).

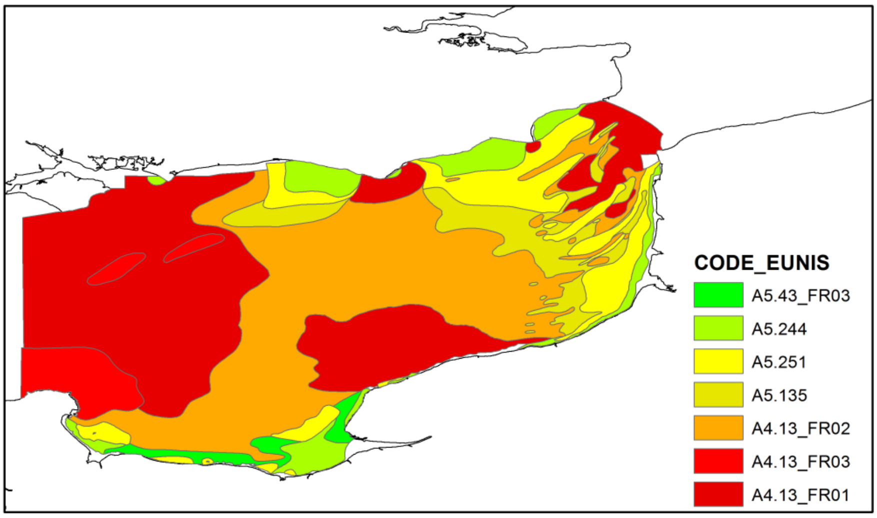


Figure 1.11 Classification EUNIS of benthic habitats performed by Cabioch et al. (1978); green: mud and sand, yellow: coarse sand, orange: gravels and red: pebbles.

The distribution and abundance of demersal fish is significantly related to the benthic community in the EEC. Vaz et al. (2007) performed a classification of communities in the EEC based on data collected in October during the Channel Ground Fisheries Survey (CGFS) since 1988.

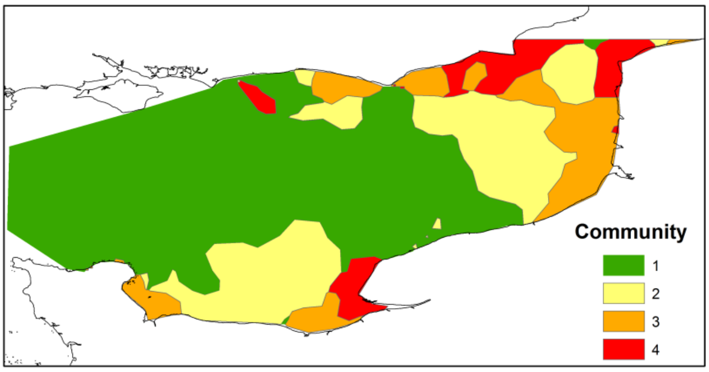


Figure 1.12 Marine fish communities' classification (Vaz et al. 2007)

They showed that fish populations are structured in response to both the sediment type and environmental conditions. Fish assemblages were separated into four classes (Figure 1.12). Class 1 represents an offshore community mainly dominated by rays, dogfishes and poor cod. Classes 2 and 3 are dominated by both pelagic and demersal species but with a difference of abundance. Class 2 is an intermediate community between coastal and offshore areas, while class 3 is a coastal community. Finally, class 4 is a coastal heterogeneous community with high densities of flatfish species (Vaz et al., 2007). The first two classes have a lower diversity level than classes 3 and 4.

Delavenne (2012) characterized the structure of pelagic habitats in the EEC based on physical, environmental and biological information. The main taxa tested were zooplankton, phytoplankton and 11 groups of fishes and invertebrates. Seven main season-dependent water groups were discriminated (Figure 1.13). Plankton groups were demonstrated to be strongly related to water types. However, the high variability of the distribution of fishes and invertebrates blurred the perception of the links between water types and fishes.

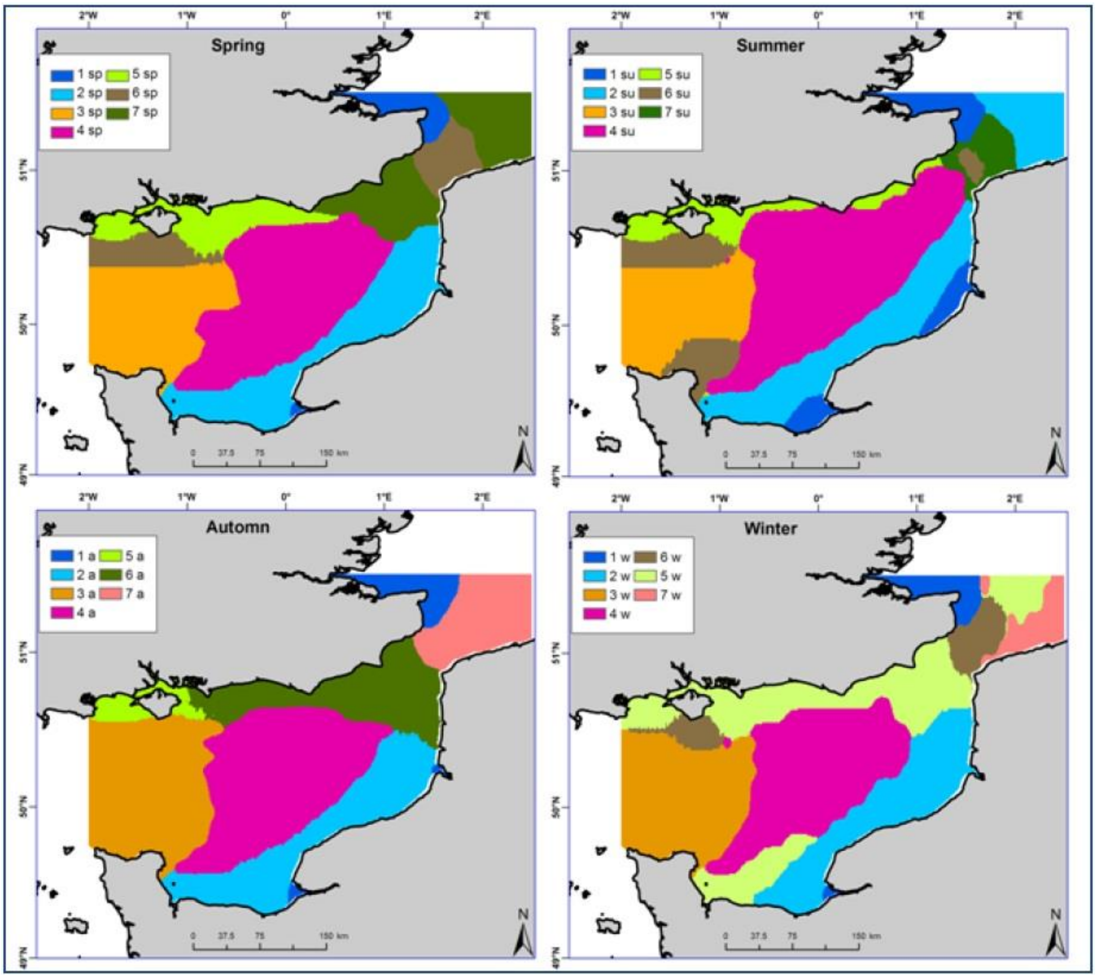


Figure 1.13 Typology of pelagic water column: seven water types were defined for each season (Delavenne, 2012).

The various EEC ecological compartments have been widely investigated in past studies aiming at understanding the functioning and structure of EEC communities. Hydrodynamics and sediments composition have been explored, communities have been characterized and numerous ecological processes have been investigated for most of the commercial species. In addition, stock assessment and surveys provided insight into population dynamics and biomass status. This information was gathered in order to implement the Atlantis Eastern English Channel application (hereby referred to as Atlantis EEC).

1.5.3. Human activities in the Eastern English Channel

1.5.3.1. The French mixed demersal fishery of the Eastern English Channel

The fishing area of the EEC corresponds to the ICES (International Council for the Exploration of the Sea) Sub-Division VIId, and it is divided into 15 statistical rectangles (Figure 1.14). The most important fishery operated in the EEC, both in number of vessels and in landings, is the French fishery. The main other countries fishing in the EEC are UK, Belgium and The Netherlands, mainly operating beam-trawls (targeting sole and plaice) and Danish seines (targeting non-quota species). Seven maritime districts are distributed along the French coast, from the North east to the South west, Dunkerque, Boulogne-sur-mer, Dieppe, Fécamp, Le Havre, Caen and Cherbourg.

To better understand fishing patterns of the French demersal mixed fleets, how they interact with the ecosystem and how they respond to management measures, we will investigate in chapter 2 the behaviour of the fleets presented in Table 1.1, which will then be compared to the dynamics of other fleets worldwide in chapter 3. Finally we will focus on the netters fleets targeting sole to test different scenarios of management measures in chapter 5.

We will present here the distribution of vessels and landings (in both weight and value) of the French demersal mixed fishery in 2007-2008. The data we used for this description have been collected by the French Directorate for Marine Fisheries and Aquaculture (DPMA) from mandatory fishers' logbooks combined with sales slips information. These data have been extracted from the "Harmonie" database of the Fisheries Information System managed by IFREMER.

In this study, the vessels are categorized by fleets and métiers (Table 1.1), based on the standard typology defined by the Data Collection Framework (DCF) of the European Union (EC, 2008). A fleet represents a group of fishing vessels sharing similar attributes in terms of technical characteristics (length class, horse power, capacity) and/or major activity (e.g., main gear used, main species targeted) during a particular year. Vessels belonging to a fleet group may still operate different fishing activities, or métiers, during the year. A métier is then defined as a group of fishing trips targeting a similar (assemblage of) species, using similar gear, during the same period of the year and/or within the same area, and which are characterised by a similar exploitation pattern (Marchal 2008).

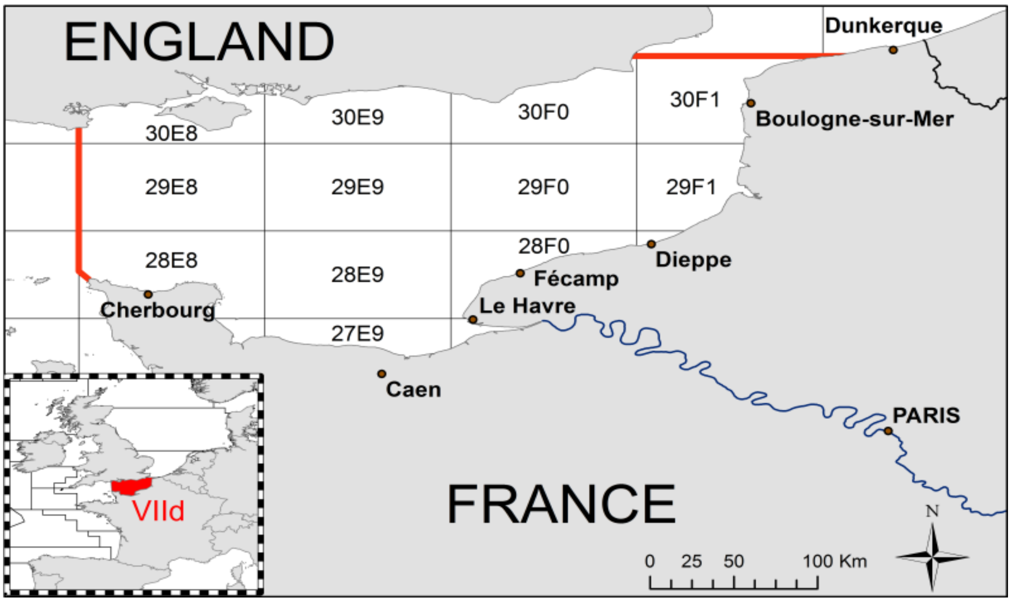


Figure 1.14 Statistical rectangles and main fishing harbours in the Eastern English Channel (ICES Sub-Divisions VIId)

Here, we focused on those fleets impacting significantly on the mortality of sole and plaice. We hence selected those fleets catching more than 2% of the total landings in weight of sole and plaice in this area. Nine fleets using towed gears are investigated: five demersal trawlers of length classes less than 10m, 10-12m, 12-18m, 18-24m and 24-40m, two dredgers of length classes 10-12m and 12-18m, and finally two polyvalent vessel groups of length classes 10-12m and 12-18 m. Three fleets using passive gears are analysed, two composed of less than 10m and 10-12 m polyvalent vessels, and the last one comprising 12-18 m vessels rigged with fixed nets. Six specific métiers are selected, and a seventh group is made by aggregating all other métiers.

During the year 2008, 694 vessels have been registered in the EEC maritime quarter in French logbooks. These vessels landed 109.2 thousand tonnes of sea products, for a turnover of 179.1 million euros, which represents approximately 23.6% of the total French landing in weight, and 16.3% in value. During the same period, 6 thousand tonnes of flatfish (sole, plaice, dab, brill and turbot) were landed, for a turnover of 35.8 million euros.

Table 1.1 Description of the fleets (a) and métiers (b) investigated in this study, as defined in the Data Collection Framework (DCF) of the European Union (EC, 2008b). The fleets and métiers coding are specific to this study.

a)

Gear type	Main gear	Vessels length (m)	Fleet code
Active gears	**Demersal Trawlers**	<10	**FL07**
		10-11.99	**FL08**
		12-17.99	**FL09**
		18-23.99	**FL10**
		24-39.99	**FL11**
	Dredgers	10-11.99	**FL26**
		12-17.99	**FL27**
	Vessels using Polyvalent 'active' gears only	10-11.99	**FL38**
		12-17.99	**FL39**
Passive gears	**ALL**	<10	**FL43**
		10-11.99	**FL44**
	Fixed nets	12-17.99	**FL49**
Other fleet		ALL	**FLZZ**

b)

Gear	Fishing activity	Métier code
Boat dredge	Molluscs	**NOS01**
Bottom otter Trawl	Demersal fish	**NOS05**
	Mixed cephalopods and demersal fish	**NOS07**
Beam trawl	Demersal fish	**NOS22**
Mid water otter Trawl	Small pelagic fish	**NOS24**
Trammel net	Demersal fish	**NOS34**
Others		**NOSZZ**

The main fishing harbour, in terms of landing (in weight and value), is Boulogne sur mer, with more than 17 000 tons landed representing 83.5 million euros, followed by Caen (11 000 tons and 42.1 million euros of landing) (Figure 1.15).

Scallop (*Pecten maximus*) is the top species landed in terms of weight and value (Figure 1.16). Sole and plaice represent a smaller part of the total landings (in weight) of the French fishery, except for the 10-12m passive demersal fleet. However, sole represents the second ranked species landed in terms of value and accounts for an important part of the revenue generated by the fleets (Figure 1.16).

The polyvalent passive gear fleets and the 12-18 m dredgers were the most active fleets in 2008, representing more than 50% of all fishing trips and half of the vessels in the EEC (Table 1.2a). The three main métiers used by these fleets were dredging for molluscs (mainly scallops); bottom otter-trawling for demersal fish, and trammel-netting for demersal fish (Table 1.2b), which represented 57% of all fishing trips operated in 2008.

Table 1.2 Importance of each fleet (a) and métier (b) in the Eastern Channel during the years 2007 and 2008, in terms of activity and landing (Data used derived from the French logbooks).

a)

Fleet code	%Number of trip	%Number of Vessels	% Flatfish landing	% Flatfish turnover
FL07	3.7	4.1	2.4	2.4
FL08	4.5	4.1	6.1	4.8
FL09	3.2	2.9	3.1	2.4
FL10	5.7	6.0	13.1	4.1
FL11	2.6	2.6	5.1	1.6
FL26	4.1	4.1	5.2	4.6
FL27	12.7	12.6	6.3	7.1
FL38	4.1	2.6	2.0	2.0
FL39	5.4	3.7	3.3	3.7
FL43	28.5	23.7	3.5	4.5
FL44	10.0	15.0	28.5	38.4
FL49	3.4	3.0	8.4	11.3
FLZZ	12.1	15.6	13.0	13.1

b)

Métier code	%Number of trip	% Flatfish landing	% Flatfish turnover
NOS01	18.3	2.3	2.2
NOS05	21.7	34.1	21.5
NOS07	1.9	3.1	1.3
NOS22	2.8	8.4	8
NOS24	2.9	1.4	1.7
NOS34	17.2	41.5	55.9
NOSZZ	35.2	9.2	9.4

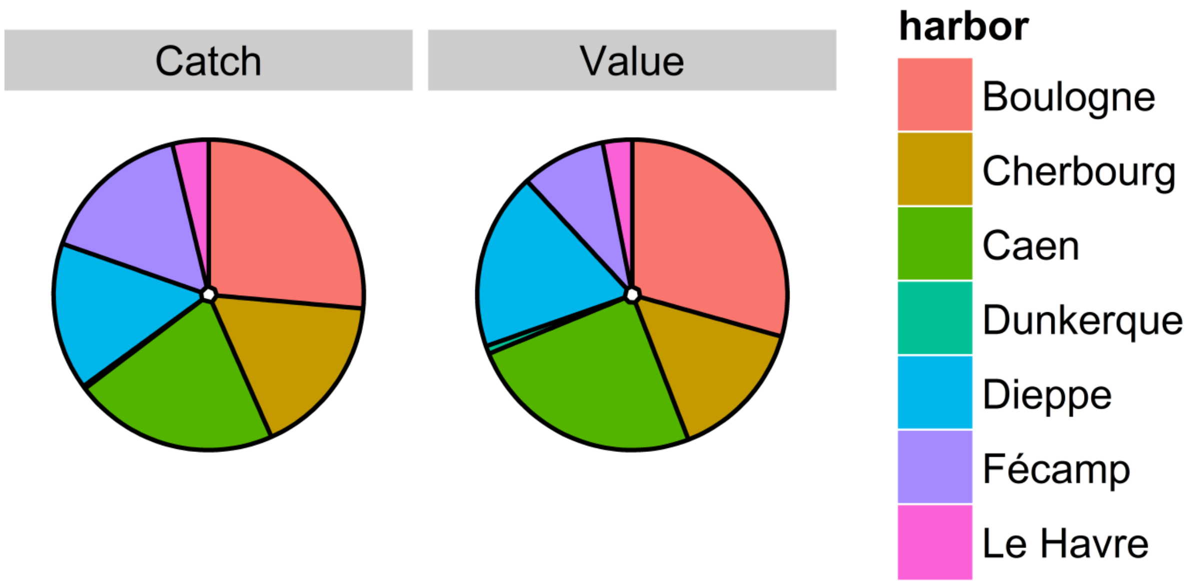
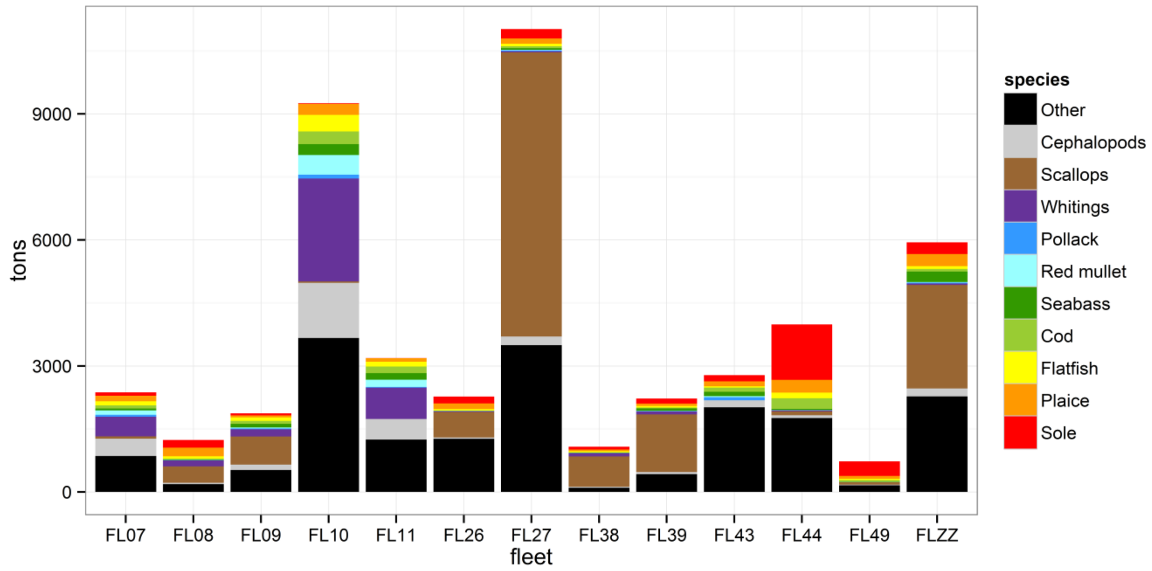


Figure 1.15 Proportion in weight and value of landing by French vessels in each maritime district in 2008.



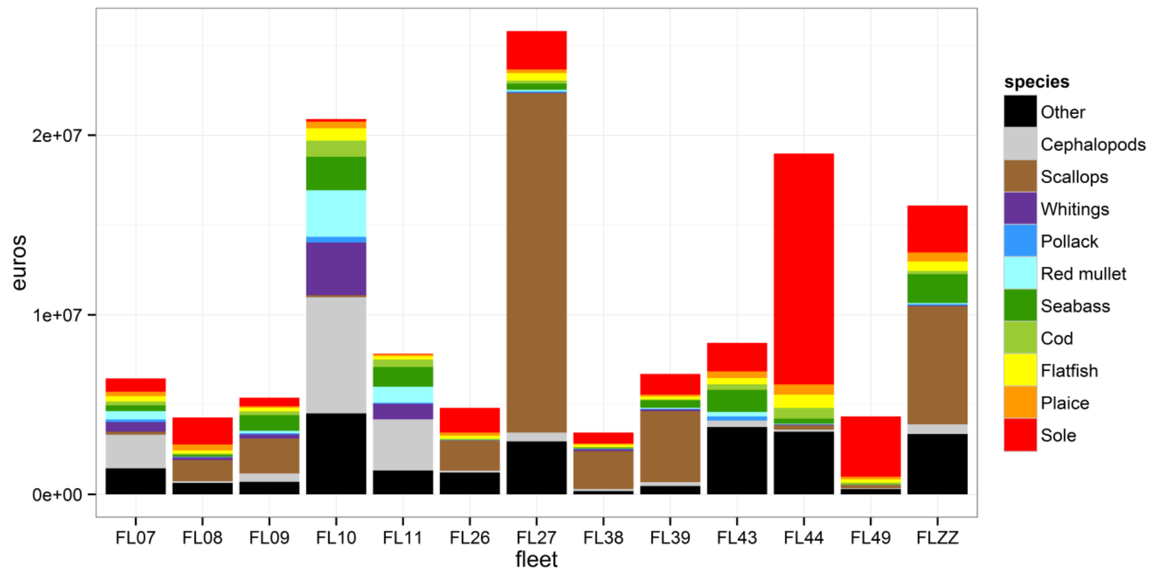


Figure 1.16 Landing in weight and value of the main species caught in the EEC for each French Fleets

Table 1.2a shows that three fleets landed more than 50% of flatfish landings by weight. These fleets were the fixed-net fleet (FL49), the 10-12 m polyvalent passive gear fleet (FL44), and the 18-24 m demersal trawlers (FL10) (Figure 1.17). However, the contribution of the 18-24 m demersal trawlers to the total flatfish turnover (4%) is lower than that of the 12-18 m dredgers (7%) (FL27). The three main fleets accounted for 57% of the total flatfish landings in value. Two métiers deserve particular attention: trammel net and bottom otter–trawl, both targeting demersal fish (respectively NOS34 and NOS05). These two métiers accounted for 76% of the landings (by weight) and 76% of the overall flatfish turnover.

Figure 1.17 landing of sole and plaice observed in 2008 in weight and value for each French fleet in the EEC. See Table 1.1 for description of fishing fleet codes.

Both Eastern English Channel sole and plaice stocks are currently supposed to be confined within ICES Division VIId. Sole and plaice are the only EEC-delimited stocks for which an age-structured stock assessment is conducted annually by ICES (ICES, 2013). In Figure 1.18 we present the results of these evaluations since 1982. From 1982 to 2012, the total biomass of sole seems to increase while plaice stock oscillates around 10 000 tons. Two periods can be highlighted, the late 90's with a higher biomass of plaice and the period 2000-2010 with a particular low biomass of this stock. Even if, both biomass of plaice and sole seemed to increase, both species are still considered as overexploited in the EEC (Figure 1.19). The only performance measure available against which to assess the exploitation of plaice is the fishing mortality at maximum sustainable yield (Fmsy), estimated by ICES;

the exploitation level of plaice was close to, but still higher, than Fmsy. The status of sole is better, despite it being several years since the stock spawning biomass (SSB) of sole was higher than the sustainable biomass estimated (Bmsy), the fishing pressure is still between the fishing mortality limit of the precautionary approach (Fpa) and the fishing mortality limit (Flim) over which the stock is likely to collapse. In response to the declining biomass state since 2005 the sole ECC TAC for 2015 (EC, 2014) has been decreased significantly, even if the total biomass of sole seemed to increase substantially overall across the last decades. It is still too early to determine the cause of the recent population decrease in sole, but both fishing pressure and extreme environmental conditions may explain a part of this change.

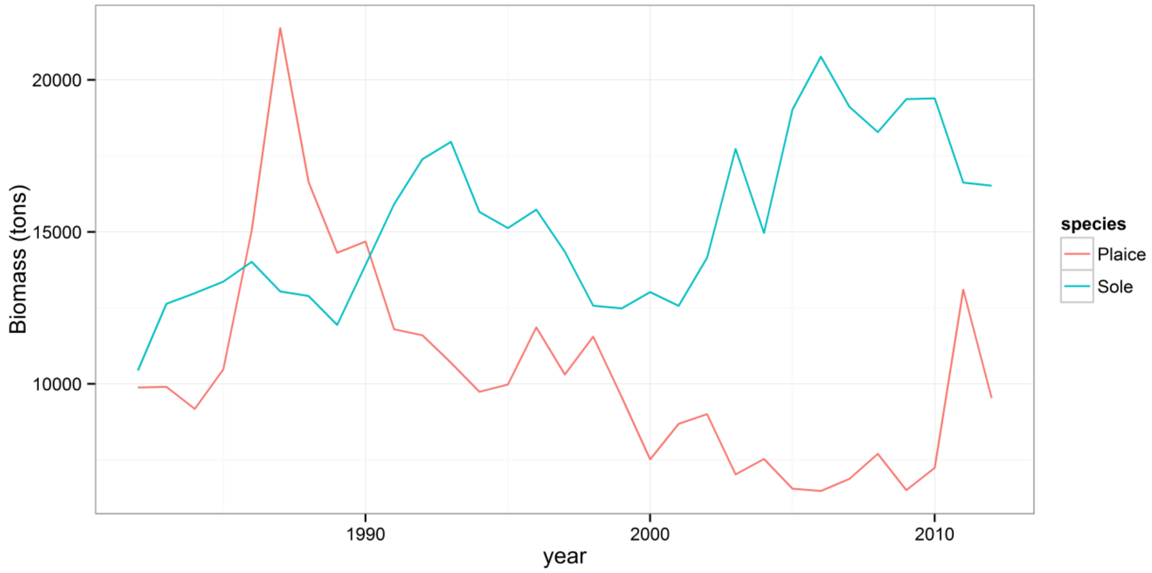


Figure 1.18 Evolution of the total biomass of plaice and sole in the VIId ICES sub-area, estimated in stock assessment (ICES, 2013)

1.5.3.2. Other Human Activities

Fishers interact (through collaboration or competition for space and resources) with each other. Fishers also compete with other maritime activities, although this aspect has rarely been considered in fisheries management and in the fisheries literature. The EEC is one of the most congested seas in the world and concentrates numerous human activities (shipping, fishing, recreational fishing, sailing) (Halpern et al., 2008). The EEC is the main shipping lane that connects many EU harbours to the rest of the world (Figure 1.20). In addition, leisure crafts and ferries connect France and the United Kingdom. The EEC bottom ground also contains coarse sands, which are exploited by aggregate extraction companies (Lozach and Dauvin, 2012). Even if only a few dredging areas are currently in operation, licenses have been granted to extract aggregates in large parts of the EEC. Aggregate extraction may impact the benthic organisms and may also interact with fisheries as well (Desprez et al., 2014; Marchal et al., 2014b). Finally many windfarm plans have been approved in the EEC, which could potentially increase the competitive pressure on fishing activities (Figure 1.20).

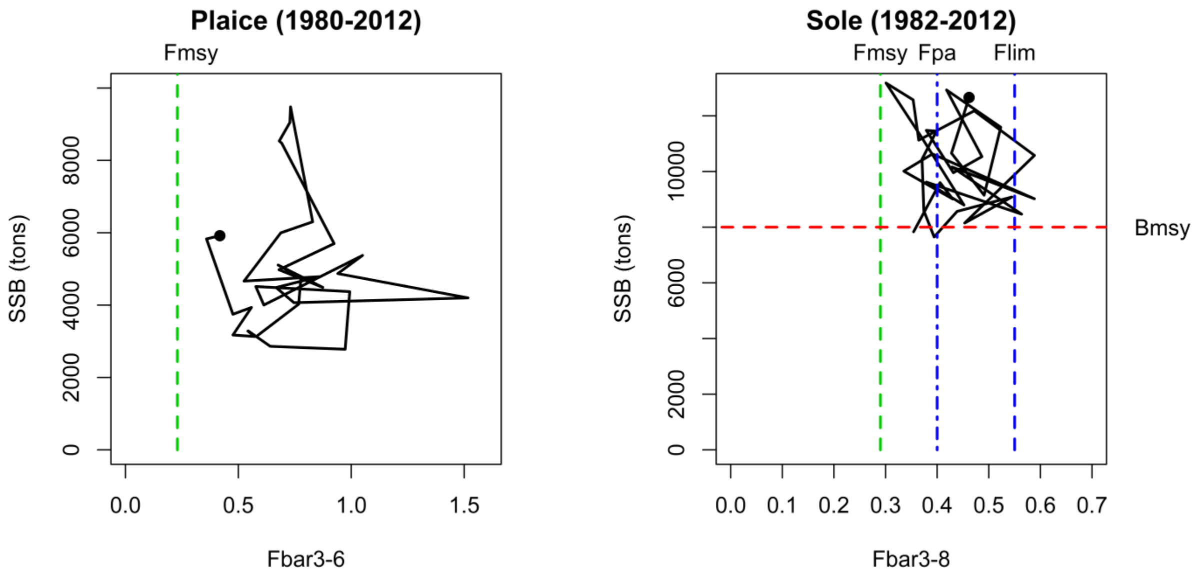


Figure 1.19 Evolution of plaice and sole spawning stock biomass (SSB) and fishing mortality (Fbar, averaged from ages 3 to 6 for plaice and 3 to 8 for sole) compared to their reference points. The black point represents the year 2012 last value of the time series. Fmsy (dashed green line) is the estimated fishing mortality at the maximum sustainable yield, Fpa is the fishing mortality limit advice by the precautionary approach and Flim is the limit over which the stock has a high risk to collapse. Bmsy trigger is the target biomass estimated to obtain the maximum sustainable yield (ICES, 2013)

As a first attempt at considering cumulative impacts on the ECC, in Chapter 2, the impact of the shipping lane on fishers' behaviour has been investigated. The shipping lane is by far the single largest spatial user of the ECC outside of fishing. The impact of other sectors of activity on fishers' behaviour has not been considered in this thesis because these currently represent only a limited spatial coverage (only a very small area of aggregate extractions zones is truly exploited each year) or because they have not yet been implemented (windfarms), but they have benn considered in the recent paper by Tidd et al. (2015).

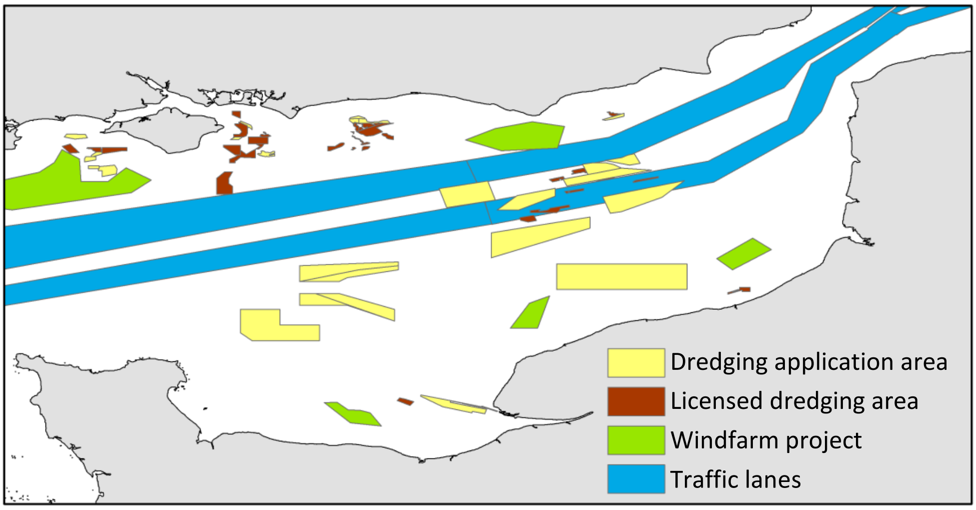


Figure 1.20 Distribution of current and possible future areas impacted by other human activities.

1.5.3.3. Management rules in the Channel

Carpentier et al. (2009) have made a summary of the different rules and legislation applied in the EEC and counted up to 216 legal instruments. In this review, rules are categorized in four groups: (i) conservation of marine habitats and species, (ii) fisheries, (iii) marine pollution and maritime security, (iv) and marine works. Marine Protect Areas are included in the first group and Figure 1.21 we present the actual MPA network implemented in the EEC. Delavenne (2012) reviewed the different types of MPAs in the ECC. The Natura 2000 protected areas are a consequence EU directives aiming at protecting biodiversity. Natura 2000 sites are chosen for their habitats importance and the preservation of sensitive species. The selected sites in the EEC are named European Marine Sites (EMS) and separated in two groups, Special Area of Conservation (SACs) under the Habitat Directive and Special Protection Areas (SPAs) under the Birds Directive. The French "Natural Marine Park" named "parc naturel marin des Estuaires Picards et de la mer d'Opale" has recently been created on the French coast of the Eastern part of the EEC. It aims at protecting and developing knowledge about the EEC marine ecosystem and at promoting sustainable marine activities. In a similar way, Marine Conservations Zones (MCZ) aim at protecting nationally important marine ecosystem features of the UK coastal area (Delavenne, 2012).

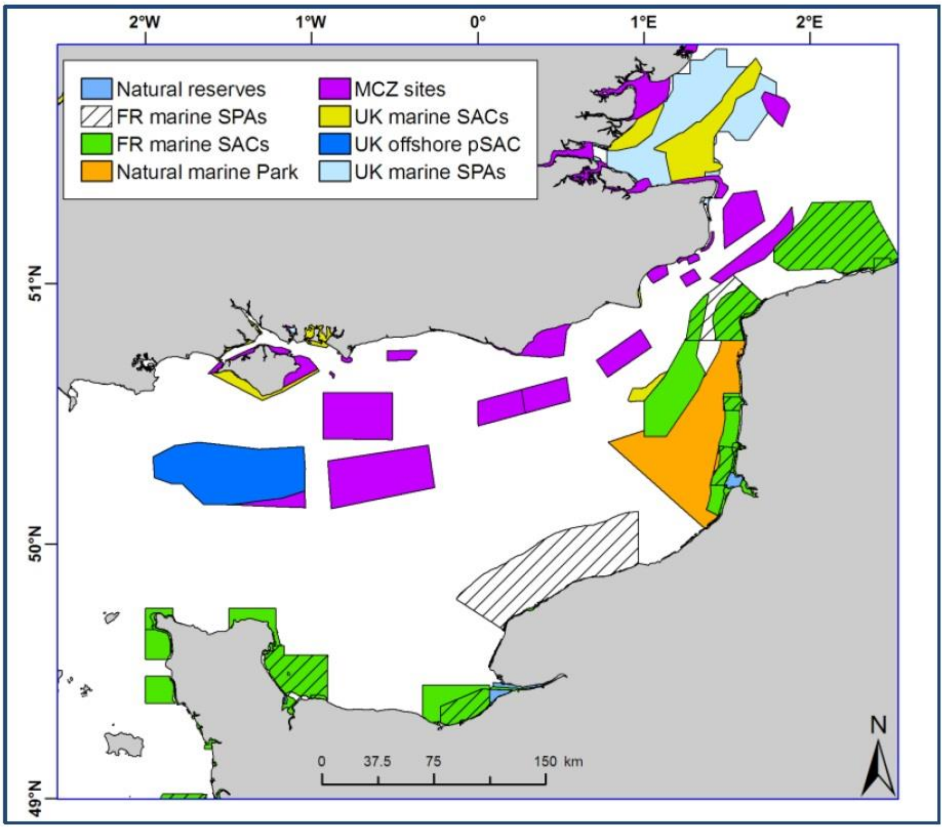


Figure 1.21 Actual MPA and potential MPA in the Eastern English Channel

While MPAs generally concern several human activities, they have currently a very limited impact on fishing activities, mainly as no management regimes have yet been set in some areas. However, these MPAs could potentially become more legally-binding to fisheries in the future. Therefore, we used this distribution of MPAs in Chapter 4 to simulate and evaluate the impact of total no-take zones as an extreme scenario of access restriction.

A number of EU and national management measures specifically target fishing activities in the EEC. These include catch limits, through TACs, for a number of commercial species (including sole and plaice), direct fishing effort limitations and a set of technical measures (gear and/or mesh size restrictions, closed areas and/or seasons). Spatially and seasonally resolved fisheries closures have been paid particular attention in this thesis (especially in Chapters 2 and 4). These include the seasonal closure of scallop fisheries (from October to May), and the restricted access for large vessels operating towed gears within3 and the 12 miles area from the shore.

1.6. References

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2. Chapter 2: Predicting fisher response to competition for space and resources in a mixed demersal fishery.

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In review, Ocean and Costal Management

2.1. Abstract

Understanding and modelling fleet dynamics and their response to spatial constraints is a prerequisite to anticipating the performance of marine ecosystem management plans. A major challenge for fisheries managers is to be able to anticipate how fishing effort is re-allocated following any permanent or seasonal closure of fishing grounds, given the competition for space with other active maritime sectors. In this study, a Random Utility Model (RUM) was applied to determine how fishing effort is allocated spatially and temporally by the French demersal mixed fleet fishing in the Eastern English Channel. The explanatory variables chosen were past effort i.e. experience or habit, previous catch to represent previous success, % of area occupied by spatial regulation, and by other competing maritime sectors. Results showed that fishers tended to adhere to past annual fishing practices, except the fleet targeting scallops which exhibited within year behaviour influenced by seasonality. Furthermore, results indicated French and English scallop fishers share the same fishing grounds, and maritime traffic may impact on fishing decision. Finally, the model was validated by comparing predicted re-allocation of effort against observed effort, for which there was a close correlation.

Keywords: Effort allocation; Random Utility Model; spatial competition; demersal mixed fishery; Eastern English Channel; spatial management

2.2. Introduction

According to the FAO (2012) most fisheries resources are already fully exploited or over-exploited due in part to excess fishing capacity and fishing power. Fishing activities can also have adverse effects on the structure and functioning of marine ecosystems (Buchen, 2009; FAO, 2012). To address that challenge, many fisheries management agencies have adopted an Ecosystem Approach to Fisheries Management (EAFM) (Browman and Stergiou, 2004), by implementing long-term management plans. This approach aims at maintaining or restoring fisheries resources to sustainable levels, while mitigating the adverse ecological impacts of fishing (Pauly et al., 2002). To accurately assess and evaluate fisheries management performance, it is essential to better understand the processes driving the dynamics of the marine ecosystems and the fishing fleets that impact them (Fulton et al., 2011; van Putten et al., 2012; Wilen et al., 2002).

Understanding and predicting the complex interactions between resource users and ecosystem dynamics is essential to reduce the risk of management failure (Hilborn, 2004). A founding principle of ecosystem-based management is that humans are fully part of ecosystems (Leslie and McLeod, 2007), and one of the main challenges for decision-makers is to better understand the factors that influence human behaviour (Wilson and McCay, 2001). This is of particular importance to fisheries managers who need to better understand the mechanisms of fishing effort allocation, so to better anticipate fishers' reactions to management.

Fishers' decision-making can be cast in terms of short- versus long-term choices (van Putten et al., 2012). For example long-term choices include decisions about capital investment, or about whether to enter or exit a particular fishery (Nostbakken et al., 2011). Conversely short-term decisions may consist of immediate actions, such as choosing a fishing area and/or a type of fishing activity (sometimes referred to as a "métier") at the beginning of, or during a fishing trip, and also includes actions, such as discarding fish (Andersen et al., 2012; Hilborn, 1985; Hutton et al., 2004). In this study we concentrated on short-term behaviour, and in particular the factors that determined fishing effort allocation both spatially and across métiers. An increasing number of studies have investigated and modelled short-term fishers' behaviour using both conceptual and data driven approaches. Conceptual approaches include applications of the Ideal Free Distribution (IFD) theory (Gillis, 2003; Rijnsdorp et al., 2000), optimal foraging theory (Dorn, 2001), Individual-Based Modelling (IBM) (Millischer and Gascuel, 2006; Soulié and Thébaud, 2006) or vessel trajectory analysis (Bertrand et al., 2005; Vermard et al., 2010). Many data-driven approaches to fishers' behaviour modelling have built in Random Utility Models (RUMs). RUMs provide an appropriate and functional approach to describe how fishers make a choice among a panel of finite alternatives (Wilen et al., 2002). Such a discrete-choice modelling approach has been applied to analyse fishers' choice of fishing ground

(Hutton et al., 2004; Wilen et al., 2002), target species (Pradhan and Leung, 2004a; Vermard et al., 2008), and gear type (Andersen et al., 2012; Holland and Sutinen, 1999; Marchal et al., 2009).

Fishers do not necessarily know all of the surrounding environmental factors and so may only have partial information about the precise position and availability of their target species. In most fleet dynamics studies, skippers have been assumed to choose their fishing ground, gear and/or target species, based on their own experience (e.g. their past choices/activity) and on their economic expectations for a given choice (e.g. past profit achieved). For example, fishers' behaviour can be influenced by fish price fluctuations, which are often seasonal and are an important factor to take into account when evaluating the expected profitability of alternative potential choices (Dupont, 1993; Loannides and Whitmarsh, 1987). Anecdotal evidence suggests that other factors which have seldom been considered in past empirical studies could determine fishers' behaviour. These factors include communication between fishers, or radar-screening of concurrent vessels which may indicate the presence of target species in a specific area. By contrast, skippers compete for space and resources, not only with other fishers, but importantly also with other sectors of activity operating in the same maritime areas. Exploitation of marine resources, for example aggregate extraction, offshore wind farms and maritime traffic can impact the choice of fishing grounds by restricting access or decreasing the availability of fish resources. In EU waters, the Marine Strategy Framework Directive (MFSD) of the European Union (EC, 2008a) requires that the different sectors of activity operating on the same maritime domain be managed jointly rather than in isolation. A key issue for fisheries managers then becomes to understand how fishers operate their activities and adjust their tactics in area-constrained environments.

To assess spatial constraint impact, this paper aimed to identify and quantify the determinants of fishing fleet dynamics in one of the most congested maritime areas in the world, the Eastern English Channel (ICES Division VIId)(Figure 1.14).

The analysis focused on French fleets catching flatfish species, sole (*Solea solea*) and plaice (*Pleuronectes platessa*). The flatfish species represent an important source of revenue for fishers in this area, however this fishery has important impacts on the marine ecosystem (Riou et al., 2001). Random utility modelling is used to gain insights into how fishers choose a métier and/or an area, at the scale of a trip, whilst interacting with other fishing fleets, maritime activities and spatial management (regulations). Maritime traffic in the Channel is thought to interact substantially with fishing activities due to one of the world's busiest shipping lanes, encompassing a large proportion of the Channel (Figure 1.14 andFigure 2.1). The main form of spatial regulation for commercial fishing activities in the Channel is the coastal area within twelve nautical miles from the coastline (hereafter called the "12-mile zone") where trawling is prohibited to vessels with an engine power exceeding

221 kW or an overall length exceeding 24 meters. Finally we tested the predictive capability of the model to forecast effort re-allocation one year ahead using two different predictors, and then predicted re-allocation of effort was compared against realised/observed re-allocation of effort.

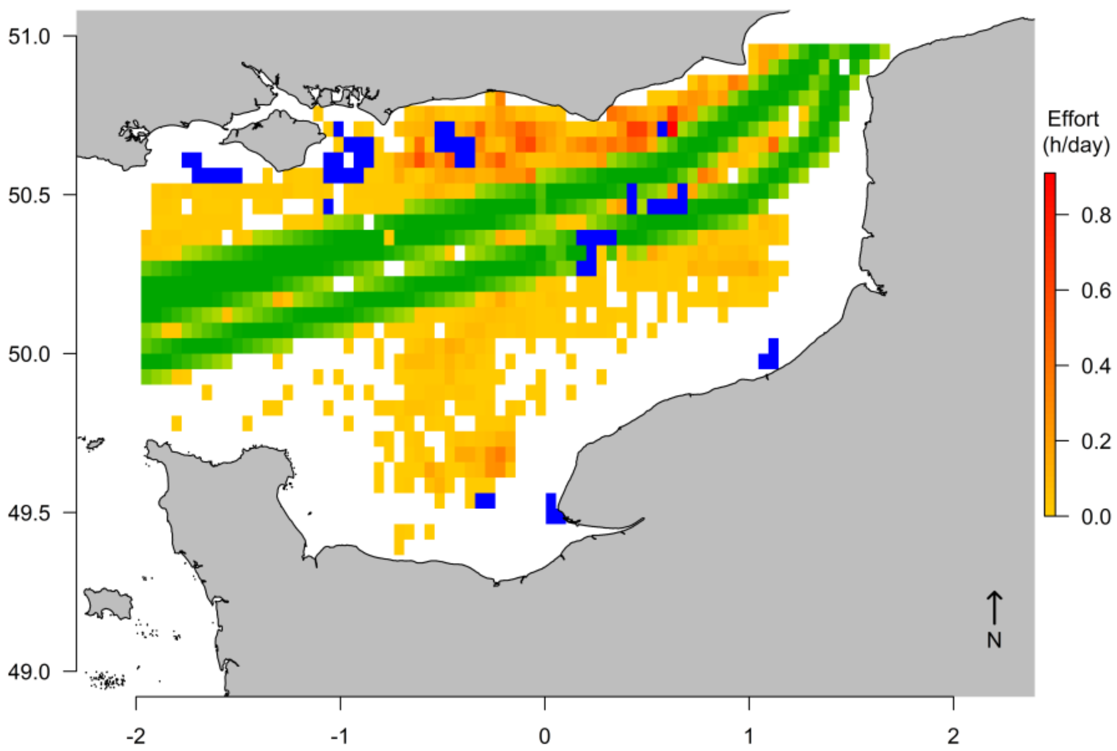

Figure 2.1 Intensity of the other uses of the maritime area in the Eastern English Channel per ICES pixel (0.05°. of longitude x 0.05°. of latitude) in 2008. The maritime traffic is represented in green. The aggregate extraction is in blue and the daily average cumulated effort of the English fishery is represented in shade of red/yellow (data derived from VMS data in 2008).

2.3. Materials and methods

2.3.1. Data

2.3.1.1. French fishing fleets

French landings (in both weight and value) and fishing effort data are collected by the French Directorate for Marine Fisheries and Aquaculture (DPMA) from mandatory fishers' logbooks combined with sales slips information. They are available on the "Harmonie" database of the Fisheries Information System managed by IFREMER. Landings in weight and value as well as fishing effort (in hours fished) are available by vessel, fishing trip, gear type and statistical rectangle (ICES rectangle with a surface of 1° longitude × 0.5° latitude, Figure 1.14). Price per species and per month was derived from the average monthly value of landings. Fishing vessels were categorised into Data Collection Framework (EC, 2008b, 2010; DCF) fleets based on the IFREMER national fleet register and trips were categorised into métiers based on monthly activity calendars (Marchal, 2008). Consistent with EC (2008a), a fleet represents hereby a group of fishing vessels sharing similar attributes in terms of technical characteristics (length class, horse power, capacity) and/or major activity (e.g., main gear used, main species targeted) during a particular year. Vessels belonging to a fleet group may still operate different fishing activities (hereby referred to as métiers) during the year. A métier is defined as a group of fishing trips targeting a similar (assemblage of) species, using similar gear, during the same period of the year and/or within the same area, and are characterised by a similar exploitation pattern. The different fleet and métier groups considered in this study are shown in Table 1.1.

We analysed fisheries data per vessel and fishing trip for the period 2007-2009. During a trip, vessels can operate in multiple ICES (International Council for the Exploration of the Sea) rectangles (Figure 1.14). Where a vessel visited several ICES rectangles in the same trip, the rectangle wherein it spent most of its fishing effort was attributed to the trip under consideration. The French vessels selected were those registered in the main Channel maritime districts (ICES Division VIId), i.e., Boulogne sur Mer, Cherbourg, Caen, Dieppe, Fécamp, Le Havre and Dunkerque (Figure 1.14). Those vessels, which had never fished in VIId during the period 2007-2009, were excluded from the analyses. Furthermore, analysis of the landing profiles of each fleet allowed us to determine the flatfish fishery by selecting flatfish landings which represented more than 2% of the total flatfish landings by weight in this area.

Allocation of the fleets' effort across métiers varied intra-annually. Figure 2.2 illustrates for all demersal trawlers smaller than 18 m (FL07, FL08 and FL09), polyvalent active gear fleets (FL38 and FL39), and for the dredger fleets (FL26 and FL27), the seasonal shift between dredging for scallops (mainly performed in the winter) and bottom otter-trawling for demersal fish (mainly performed in the summer), or also midwater otter trawl for fleets polyvalent active gear fleets. In addition, an important part of the activity of the smallest trawlers and smallest dredgers (respectively <10 m, FL07, 10-12m, FL26) was composed of the "other métiers" (NOSZZ). In contrast, demersal trawlers larger than 18 m (FL10 and FL11) almost exclusively used bottom otter-trawl for demersal fish (NOS05) throughout the year. Polyvalent passive gear fleets (FL43 and FL44) showed a more constant pattern of activity throughout the year, which was mainly dominated by trammel-netting (NOS34) for the larger vessels, and by the "other métier" group (NOSZZ) for the smallest vessels. A seasonal shift to dredging was observed for the 12-18 m fixed nets fleet (FL49), similar to that observed for the towed gear fleets (Figure 2.2).

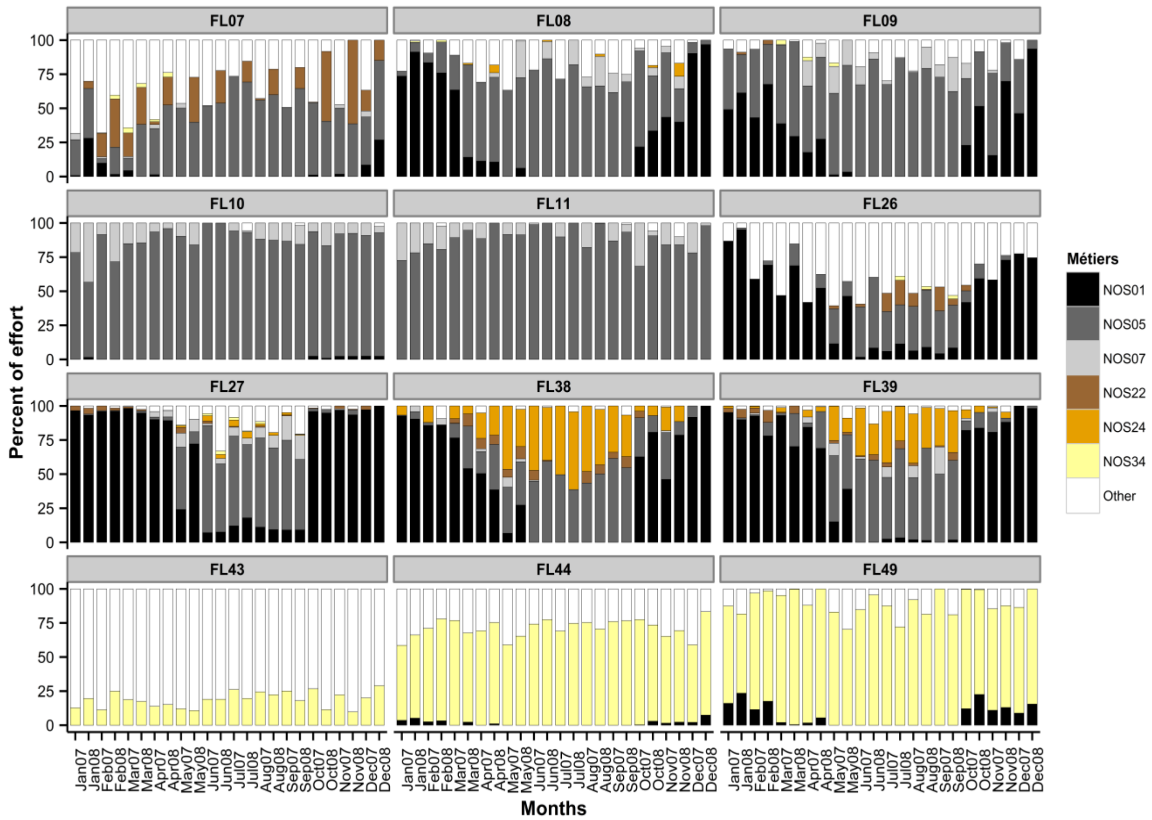

Figure 2.2 Proportion of métiers used by each selected fleet per month in 2007 and 2008, in percentage of trip chosen (Data used derived from French logbooks and monthly activity calendars).

2.3.1.2. <u>Other sectors of activity and spatial restrictions</u>

The interactions between each of the French fleets presented in Section 2.3.1.1 were examined in relation to (i) other French and English fishing fleets, (ii) maritime traffic, and (iii) spatial management. The fishing activity of English vessels (mainly beam trawlers) was represented by their aggregate effort (in hours fished) per day and per ICES statistical rectangle. Most of the large-scale maritime traffic in the Channel occurs through a corridor referred to as the extended Vessel Separation System (VSS; Figure 2.1). For the purpose of this study, we assumed that the pressure exerted by maritime traffic on fishing activities to be represented by the percentage of an overlap of VSS on the ICES statistical rectangle. The 12-mile management zone was represented by the percentage of spatial overlap between this zone and each statistical rectangle. The spatial overlaps described above were calculated using a Geographic Information System (GIS) and then normalized with the surface of each statistical rectangle using R statistical software (R Core Team, 2012).

2.3.2. Fleet dynamics modelling

In order to understand and forecast fishing behaviour, we developed a discrete-choice model using a random utility function. Such models have been widely applied to analyse and model human behaviour and activities (Earnhart, 2002; Holland and Sutinen, 1999; McFadden, 1974; Sammer and Wüstenhagen, 2006). The main assumption of RUM is that individuals seek to maximize their utility (Pascoe and Robinson, 1998; Robinson and Pascoe, 1997; Wilen et al., 2002). Different explanatory variables were tested in order to identify the best model by running the RUM with different sets of explanatory variable (Table 2.1). A model was parameterized for each of the fleets shown in Table 1.1.

Several RUM types building on different probability distributions have been used to model fishing choice behaviour. In the present case, two distributions have been considered. First, a conditional logit model (McFadden, 1974 ; Vermard et al., 2008) was used. This is the simplest sort of distributions to be considered, and also the one for which model outcomes are the easiest to interpret. A potentially critical aspect of this distribution model is that it should accommodate the property of independence of irrelevant alternatives (IIA). This IIA requires that for any choice alternatives, the relative odds of choosing one alternative rather than another are the same, irrespective of the availability of the other alternatives or consideration of their attributes (Train, 2003). A nested logit model was then also tested. Nested logit models impose a more complex hierarchical structure that could both alleviate the risk of failing the IIA property by limiting its application to alternatives of the same nest, and better mimic, at least conceptually, the fishers' decision-making process (Holland and Sutinen, 1999; Marchal et al., 2009).

For the conditional logit model, within a fleet, we assumed that at the start of a fishing trip, each individual vessel (*v*) may choose among several alternatives (*i*). Each alternative was defined by combining a métier and a statistical rectangle (Figure 1.14 and Table 1.1). This allocation process is divided in two steps in the nested logit model, with at first, métier's choice corresponding to the nest and then within each nest, the area selection. All areas visited by fishing vessels outside Sub-Division VIId were merged in a unique area (named ZZZZ in this study). Each alternative was associated to a utility function.

2.3.3. RUM explanatory variables

The deterministic part of the Utility function (U_i) was composed of 7 explanatory variables. We assumed that fishers choose their métier and fishing ground with the aim to maximize their returns based on their own experience and also on information gleaned from the other vessels in the same fleet operating the same métier, such as the profit realised by the fleet in the past (Holland and Sutinen, 1999; Marchal et al., 2009). We also assumed that fishers interact spatially with other

French and English fleets and that they may be constrained by both Channel maritime traffic and the 12-mile zone.

Table 2.1 Description of the explanatory variables used in the Random Utility Model

RUM explanatory variables	Lag	Description
VPUE_MONTH_1	1 month	Revenue expected by choosing a given métier, based on value per unit effort experienced in the past with this métier
VPUE_MONTH_12	12 months	
EFF_MONTH_1	1 month	Habit of a vessel, reflected by past effort allocation by métier
EFF_MONTH_12	12 months	
EFF_OTH	No lag	Pressure exerted by other French fleets in a given statistical rectangle
EFF_GB	No lag	Pressure exerted by English fleets in a given statistical rectangle
∑POURC_CPUE	1 month	Proportion of each main species in the landing of a vessel one month before the current trip
SURF_AREA_OCCUP	No lag	Spatial constraint exerted by maritime traffic, estimated by the proportion of each statistical rectangle overlapped by the extended vessel separation system
SURF_12NM	No lag	Spatial constraint exerted by the 12-mile coastal zone, estimated by the proportion of each statistical rectangle overlapped by the management area.

The main economic variable driving effort allocation decisions was assumed to be $VPUE_i$ defined as the expected returns from choosing métier in a given area. To take into account the potential effects of price differences between species, $VPUE_i$ was derived by weighting past $CPUE_{i,s}$ aggregated per species group s, month and métier, with current monthly average price (€/kg) per species $Price_s$ (Equation 1).

$$VPUE_i = \sum_s (CPUE_{i,s} * Price_s) \qquad (1)$$

Most studies of fishing decisions to date have shown that the decisions by fishers are also often based on their own past fishing patterns (i.e., there is a degree of adherence to traditional fishing grounds and/or métiers)(Holland and Sutinen, 1999, 2000; van Putten et al., 2012, 2013; Vermard et al., 2008). For this reason we included a variable $EFF_{i,v}$ which represents the past monthly average effort allocated for each alternative by each vessel. $EFF_{i,v}$ can be considered to represent the habits of fishers but also their knowledge of fishing grounds. Two different time lags (1 month and 12 month) were applied for each of the variables above (the suffixes *_MONTH_1* and *_MONTH_12* were added to distinguish between these two categories of lagged variables). The monthly average proportion of catch of a species s per unit of effort of a vessel *v*, $POURC\_CPUE_{i,v,s}$, was introduced in the model to represent the degree of targeting of specific species or groups of species by fishers. This was calculated for the top six species in terms of commercial value for the fleets under investigation. These included two flatfish species, sole (SOL) and plaice (PLE), seabass, *Dicentrarchus labrax* (BSS), cephalopods, *Sepia officinalis* and *Loligo forbesi* (CEPH) and cod, *Gadus morhua* (COD). Scallops (SCE), were also included as the main target species for the dredging fleets. Other species were aggregated in a seventh species group (OTHFF). Only one month lag was applied for those variables. This is because when the $POURC\_CPUE_{i,v,s}$ with 1- and 12-month lags were used in the same model, none of the other explanatory variables were significant, likely due to a problem of multiple correlations between explanatory variables which was not observed when only one of the two lagged variables was used. The two different time lags for the variables $VPUE_i$ and $EFF_{i,v}$ were kept, to explicitly represent the effect of seasonality in fishing some of the target species, and the influence on decisions of the most recent exploitation cues, hereby observed in the previous month.

To capture the impact of other fishing activities on fisher choices, three choice-specific variables were introduced in the model. The first, EFF_oth_i, represents the spatial interaction with the other French fleets, and it is derived from the sum of monthly average current effort allocated by other fleets to a particular area. The second, EFF_GB_i is the mean cumulative effort allocated by English vessels to a particular area, and represents the spatial interaction between the French fleet under consideration and English vessels fishing at the same time.The two remaining explanatory variables that were calculated represent the spatial constraint exerted by maritime traffic and area-based management on the French fleets. $SURF\_AREA\_OCCUP_i$ is the monthly average overlap between the extended VSS and the fishing grounds, and provides an estimate of the pressure exerted by maritime traffic per ICES rectangle. The variable $SURF\_12NM_i$ represents the 12-mile zone, and was calculated as the percentage of each statistical rectangle that overlapped with this restricted fishing zone. Finally, correlations have been tested between each couple of variables.

In summary, the deterministic part of the utility function was written as follows (equation 2):

$$Ui \sim VPUE_i + EFF_{i,v} + EFF\_oth_i + EFF\_GB_i + \sum_s POURC\_CPUE_{i,v,s} + SURF\_AREA\_OCCUP_i + SURF\_12NM_i \quad (2)$$

2.3.4. Model selection and probability

The two different models, nested and conditional logit, were tested on each fleet. Both models were tested against the IIA hypothesis. The test consists of comparing the estimation of the model with the set of all alternatives *C,* with the same model using only a subset of alternatives *A*. Hausman and McFadden (1984) provide a description of this test which leads to the formulation of a test statistic *S* (equation 3):

$$S = (\vartheta_A - \vartheta_C)' * [\mathrm{cov}\,(\vartheta_A) - \mathrm{cov}\,(\vartheta_C)]^t * (\vartheta_A - \vartheta_C) \tag{3}$$

where ϑ_A and ϑ_C are respectively the maximum likelihood estimators of the conditional logit model with the subset of alternatives *A* and the one with the set of alternatives *C*. This test statistic *S* follows a χ^2 distribution. The test is performed by comparing the full-alternative model with the model with one alternative missing, for each alternative.

Selection of the best model is based on the McFadden's likelihood ratio index (LRI) (McFadden, 1974), which is similar to a R^2. The model was fitted to 2007-2008 data. The model retained can then be used to calculate the probability of each possible choice i by maximizing U_i. The calculus of this probability is detailed in equation 4 for the conditional logit model with N as the total number of alternative choices for a given fleet.

$$\mathrm{P}\,(i) = \exp\,(U_i) \,/\, \textstyle\sum_{\mathrm{i=1:N}} \exp\,(U_i) \tag{4}$$

Concerning the two-level nested logit model, this probability may be described as (equation 5; Train, 2003):

$$\mathrm{P}(i) = \mathrm{P}(m) * \mathrm{P}(i|m) \tag{5}$$

where $\mathrm{P}(i/m)$ (equation 6) is the conditional probability that the skipper will choose the alternative i after having selected the métier m. $\mathrm{P}(m)$ (equation 8) is the unconditional probability that the skipper will choose the métier m before each trip. The deterministic component of U_i can be derived on factors apply to the selection of a nest (a métier) hereafter called Z and others use in the second decision step (ICES area) hereafter called Y. $\mathrm{P}(i|m)$ can be expressed as

$$\mathrm{P}(i|m) = \exp(\beta' \mathrm{Y}_{i|m}) \,/\, \exp(\mathrm{IV}_m) \tag{6}$$

where β is the parameter vector to be estimated, and

$$\mathrm{IV}_m = \log\,\{\, \Sigma_{i\epsilon Cm} \exp(\beta' \mathrm{Y}_{i|m})\} \tag{7}$$

is the inclusive value for métier *m*. The unconditional probability of selecting à métier *m* is

$$P(m) = \exp(\gamma' Z_m + \sigma_m IV_m) / \Sigma_{m \in C} \exp(\gamma' Z_m \sigma_m IV_m) \quad (8)$$

where σ_m is the inclusive parameter value for métier *m* and γ is the parameter to be estimated. The consistency of using a nested logit model is assessed by testing the null hypothesis σ_m=1 with a Wald χ^2 test.

2.3.5. Forecast

We used the models previously calibrated over 2007-2008 to forecast trip choices in 2009. For each fleet, a set of explanatory variables was considered, and only the coefficients associated to the variables that best explained the model's variability ($p < 0.05$) were used to predict choices in 2009. The input data were derived from the same source of information that was used to describe the fleet choices over the 2007-2008 period, and these were processed in the same way. In many fisheries applications of discrete-choice models, the forecasted choice is taken to be that with the highest probability (see equations 4 and 5) (Marchal et al., 2009; Vermard et al., 2008). However, this approach appears to be rather *ad hoc*, and the prediction performances of the maximum probability estimator have to our best knowledge never been contrasted with those of alternative predictors, such as the median of the distribution.

In the present case, two methods of prediction were used. With the first method, the choice actually made is assumed to be as in previous studies, the alternative with the highest probability. The second method requires performing 200 simulations. Within each simulation, the choice is randomly selected from a multinomial distribution parameterized by the probability distribution derived from the model calibration. The frequency of each of the alternative choices actually made is then calculated for both methods for each month. For the second method, the median of the 200 frequencies obtained with the random iterations is defined as the frequency of forecasted choices.

To assess the capacity of each method to forecast the trip choice made in 2009, the frequencies of forecasted choices per month are compared to the observed frequencies. χ^2 tests are usually performed to compare observed and theoretical proportions. However, in our case, some choices will not be selected given the information provided by the explanatory variables. Because theoretical frequencies are used as denominators in the χ^2 equation, null values will by construction compromise the utilization of that test. For that reason, another indicator has been calculated in order to evaluate the respective performances of the two prediction methods. This is the mean absolute error (*MAE*) weighted by the total number of trips per month obtained with each method, for each fleet (equation 9)(Willmott and Matsuura, 2005).

$$MAE = [1 / (M * N)] * \sum_{i=1:M} \sum_{j=1:N} |F_{i,j} – Fpred_{i,j}| / F_i \quad (9)$$

Where $F_{i,j}$ is the frequency of observed choice j during month i; $Fpred_{i,j}$ is the frequency of the forecasted choices; F_i is the total number of trips performed by a fleet during month I; N is the number of alternative choices for a given fleet; and M is the number of months during which trips are operated. The method with the smallest *MAE* is considered to be the one which best predicted the global behaviour of the fleets. The package mlogit of the R 2.14.1 software was used to estimate the model and perform the forecasts (Croissant, 2011; R Core Team, 2012).

2.4. Results

The correlation between explanatory variable is most of time less than 0.2, except for some fleets for which it was around 0.5 (especially for variable VPUE or EFF with two different time lags), so all the variables previously described have been tested. The goodness of fit tests for the two models for each fleet using 2007-2008 data are presented in Table 2.2. For all fleets, the McFadden R^2 was slightly higher when the nested logit model was used and the same result was observed with the likelihood ratio test.

2.4.1. Model goodness of fit

The IIA was tested for each fleet; however the property was never fully satisfied for the demersal trawlers of length below 10 m (Table 2.3). The statistic S was often negative, which does not necessarily contradict with the IIA assumption (Hausman and McFadden 1984). Nevertheless, the S statistic is higher than the critical value for some alternatives (e.g., NOS22 29F1 for FL07, NOS34 outside area VIId and NOSZZ 27E9 for FL43 in Table 2.3), which contradicts the IIA property. Even if the model was further tested using the nested RUM (this approach relaxes IIA and assumes correlation across alternative choices e.g. (see, Ben-Akiva and Lerman, 1985), the IIA property within nests was still not fully satisfied. Moreover, the goodness of fit of both the nested and the conditional logit models, as given by the LRI index, were very similar, and there were overall little difference between model estimates (Figure 2.3).

In addition, considering the result of the Wald χ^2 test, nested models for 24-40m demersal trawlers, 12-18m vessel using polyvalent active gears and 12-18m netters are considered similar to conditional logit models ($p > 0.05$; Table 2.2). So, further analyses were performed using the most parsimonious model, the conditional logit model. Overall the model provided a good fit for all fleets in 2007-2008 and on average resulted in a McFadden LRI of 50% and a maximum value of 68% for the fleet of polyvalent passive gears of vessel < 10 m. The other fleets resulted in a McFadden LRI of 30% which is still reasonable for a mixed fishery while the poorest fit observed was for the 'large demersal trawler' fleet, with a McFadden LRI of 17% (Table 2.2).

Table 2.2 Comparison, for each fleet, of the model's goodness of fit to the 2007-2008 data , for the conditional logit model (MNL) and the nested logit model (NMNL) and test of the nested structure with a Wald χ² test. An alternative correspond to a métier and area choice combined.

		Number of alternatives	McFadden R^2		LRTEST		Wald χ² test (p-wald)
			MNL	NMNL	MNL	NMNL	NMNL
Demersal trawlers	<10m	**16**	**0.68**	**0.69**	**4309.5**	**4358.8**	**90.18** (<2.2e-16)
	10-12m	**29**	**0.55**	**0.56**	**5796.2**	**5876**	**134.32** (<2.2e-16)
	12-18m	**38**	**0.41**	**0.42**	**3272.2**	**3364.9**	**81.71** (<2.2e-16)
	18-24m	**31**	**0.26**	**0.26**	**3208.5**	**3221.5**	**7.59** (0.022)
	24-40m	**24**	**0.17**	**0.18**	**816.59**	**819.54**	**4.34** (0.114)
Dredgers	10-12m	**25**	**0.54**	**0.55**	**4874.4**	**4967.3**	**93.43** (<2.2e-16)
	12-18m	**56**	**0.38**	**0.39**	**12222**	**12272**	**60.69** (6.64e-14)
Polyvalent active gear	10-12m	**33**	**0.33**	**0.34**	**3435.1**	**3548.3**	**60.92** (3.74e-13)
	12-18m	**38**	**0.31**	**0.31**	**3789.4**	**3790.1**	**0.59** (0.75)
All passive gear	<10m	**21**	**0.68**	**0.68**	**28941**	**29298**	**197.11** (<2.2e-16)
	10-12m	**28**	**0.58**	**0.59**	**9838.5**	**9870.5**	**17.31** (0.00017)
Fixed nets	12-18m	**24**	**0.64**	**0.64**	**3252**	**3234.4**	**2.29** (0.32)

Table 2.3 Tests for the IIA property, based on the S statistic, performed on demersal trawlers composed of vessels of less than 10 m and passive gear fleet composed of vessels of less than 10m.

FL07			FL43		
Deleted choice	**S statistic**	**P-value**	**Deleted choice**	**S statistic**	**P-value**
NOS01 27E9	Negative	-	NOS34 27E8	34.22	0.27
NOS01 out of VIId	16.12	0.93	NOS34 27E9	13.90	0.99
NOS05 27E9	15.01	0.96	NOS34 27F0	1.27	1
NOS05 27F0	9.09	0.99	NOS34 28E8	Negative	-
NOS05 28E9	Negative	-	NOS34 28E9	17.80	0.96
NOS05 28F0	Negative	-	NOS34 28F0	Negative	-
NOS05 28F1	6.53	0.99	NOS34 28F1	Negative	-
NOS05 29F1	31.42	0.21	NOS34 29F1	41.84	0.07
NOS05 out of VIId	Negative	-	NOS34 30F1	Negative	-
NOS22 28F1	1.16	1	NOS34 out of VIId	399.90	<0.0001
NOS22 29F1	241.84	<0.0001	NOSZZ 27E8	Negative	-
NOS34 out of VIId	Negative	-	NOSZZ 27E9	338.32	<0.0001
NOSZZ 27E9	Negative	-	NOSZZ 27F0	Negative	-
NOSZZ 27F0	Negative	-	NOSZZ 28E8	Negative	-
NOSZZ 28F0	Negative	-	NOSZZ 28E9	Negative	-
NOSZZ 29F1	Negative	-	NOSZZ 28F0	Negative	-
			NOSZZ 28F1	Negative	-
			NOSZZ 29F0	Negative	-
			NOSZZ 30F0	4.58	1
			NOZZZ 31F1	1.70	1
			NOSZZ out of VIId	Negative	-
Degree of freedom	26			30	
Critical chi-squared[df]	38.89			43.77	

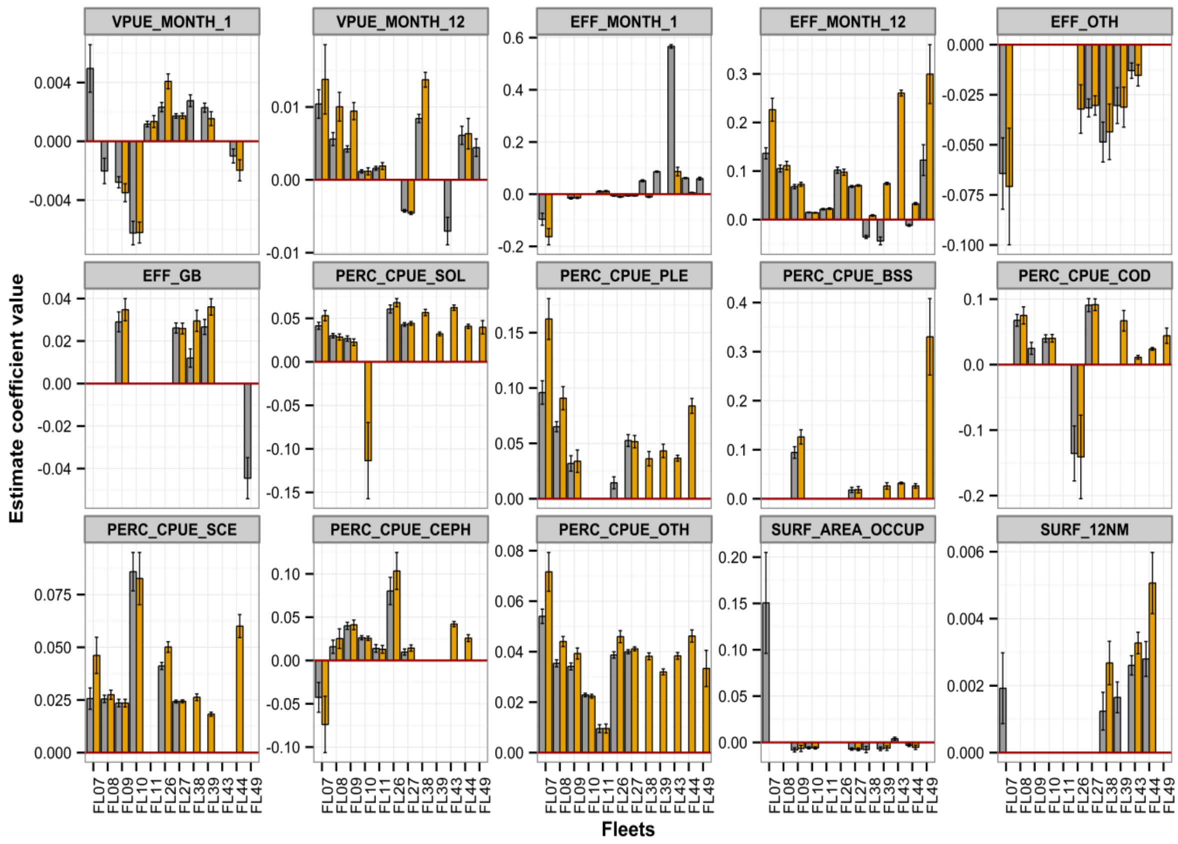

Figure 2.3 Comparison of each estimate per selected fleet. Grey bars represent the conditional logit model and orange one the nested logit model with the choice of a métier for the first level and an ICES area for the second level. Only significant estimates are presented.

2.4.2. Parameter estimation

2.4.2.1. Expected economic opportunities

The effort allocation of all demersal trawlers fleets (from FL07 to FL11) and all of the passive gear fleets >10 m (FL44 and FL49) were always positively influenced by the variable *VPUE_MONTH_12*$_i$, while the effect of variable *VPUE_MONTH_1*$_i$ was dependent on vessel length and gave a negative coefficient for all demersal trawlers of length range 10-24 m (FL08, FL09 and FL10). By contrast, the effort allocation of all dredgers and polyvalent active gear fleets respectively (FL26, FL27, FL38 and FL39) was positively affected by the variable *VPUE_MONTH_1*$_i$, while the impact of variable *VPUE_MONTH_12*$_i$ was dependent on vessel length (Table 2.4).

2.4.2.2. Traditional fishing

The current effort allocation of all fleets rigged with active gears (demersal trawlers, dredgers and polyvalent active gears) was negatively (or not) affected by their past short-term effort allocation, except for the fleet of demersal trawlers 24-40 m (FL11) and all of the passive gears, which were positively influenced by past effort in the same month in the previous year.

2.4.2.3. Influence of other uses of maritime space

Three different variables represent the potential spatial interactions, which may potentially interact with the French fishing fleet. These include (i) other fleets from France or England, (ii) maritime traffic, and (iii) the 12-mile zone where trawling is prohibited to large trawlers. The presence of English vessels reflected by the variable EFF_GB_i was positively correlated with several of the French fleets: 12-18 m demersal trawlers (FL09), 12-18 m dredgers (FL27), and all polyvalent vessels using active gears (FL38 and FL39). That presence has no effect on the other fleets.

However, most of the French fleets tend to avoid areas with an overlap with other French fishing fleets, as represented by EFF_oth_i which always has a negative influence on the choice of a statistical rectangle.

The proxy representing maritime traffic, $SURF\_AREA\_OCCUP_i$, had a negative influence on the choice of activities by fleets of larger active gear vessels (FL09, FL10, FL27 and FL39), and also the 10-12 m passive gear fleet (FL44). However, the smallest demersal active gear fleets (<12 m, FL07, FL08, FL26 and FL38) displays a positive or null coefficient. Choices by the fixed nets fleet (FL49) are also positively impacted by the maritime traffic overlap variable.

The proxy representing the overlap with the 12-mile coastal management area, $SURF\_12NM_i$, has a positive coefficient for fleets consisting of small vessels: demersal trawlers under 10 m (FL07), 10-12 m polyvalent active gears fleet (FL38), under 10 m and 10-20 m passive gears fleet (FL43 and FL44).

Table 2.4 Parameters estimates from RUM on trip choice behaviour for each fleet. Only the significant parameters are shown and used to forecast the choice of a métier and an ICES area during the year 2009. The positive coefficients are shown in bold characters (P-value: 0 '*', 0.001 '**', 0.01 '*', 0.05< '-')**

Variables	Demersal Trawlers					Dredgers		Polyvalent active gears		All Passive gears		Fixed nets
	<10m	10-12m	12-18m	18-24m	24-40m	10-12m	12-18m	10-12m	12-18m	<10m	10-12m	12-18m
VPUE_MONTH_1	0.0050**	-0.002*	-0.003***	-0.006***	0.0012***	0.0023***	0.0017***	0.0009*	0.0015**	-	-0.0015**	-
VPUE_MONTH_12	0.0104***	0.0056***	0.0042***	0.0012***	0.0016***	-	-0.0043***	0.0104***	-	-	0.0059***	0.0028*
EFF_OTH	-0.064***	-	-	-	-	-	-0.0316***	-0.0434***	-0.0298**	-0.0166***	-	-
EFF_GB	-	-	0.0290***	-	-	-	0.0262***	0.0291***	0.0371***	-	-	-
EFF_MONTH_1	-0.096***	-	-0.015***	-	0.0106***	-0.0056*	-0.0051***	-0.0102***	-	0.0875***	0.0046***	-
EFF_MONTH_12	0.1364***	0.1052***	0.0678***	0.0147***	0.0214***	0.1018***	0.0681***	0.0105***	0.0741***	0.2370***	0.0290***	0.2591***
POURC_CPUE_SOL	0.0414***	0.0300***	0.0268***	-	-	0.0607***	0.0430***	0.0462***	0.0320***	0.0509***	0.0321***	0.0340***
POURC_CPUE_PLE	0.0959***	0.0651***	0.0318***	-	-	0.0144*	0.0526***	0.0313***	0.0427***	0.0309***	0.0707***	-
POURC_CPUE_BSS	-	-	0.0943***	-	-	-	0.0180***	-	0.0263**	0.0286***	0.0207***	0.2920***
POURC_CPUE_COD	-	0.0677***	0.0248*	0.0400***	-	-0.1356**	0.0909***	0.0237*	0.0664***	0.0102***	0.0216***	0.0382***
POURC_CPUE_SCE	0.0257***	0.0255***	0.0235***	0.0859***	-	0.0411***	0.0242***	0.0237***	0.0183***	-	0.0569***	0.0165***
POURC_CPUE_CEPH	-0.0426*	0.0158*	0.0400***	0.0262***	0.0140**	0.0803***	0.0098***	-	-	0.0359***	0.0241***	-
POURC_CPUE_OTH	0.0540***	0.0354***	0.0341***	0.0228***	0.0095***	0.0386***	0.0400***	0.0320***	0.0325***	0.0340***	0.0427***	0.0302***
SURF_AREA_OCCUP	0.1506**	-	-0.008***	-0.006***	-	-	-0.0071***	-	-0.0060**	-	-0.0037*	0.0060*
SURF_12NM	0.0019*	-	-	-	-	-	-	0.023***	-	0.0028***	0.0043***	-

2.4.3. Forecasted fishing effort allocation (2009)

Table 2.5 Comparison of two methods to forecast the trip choice in 2009 based on the parameter estimates from discrete choices models previously analysed. The MAE (Mean absolute error) of each method is shown for each fleet.

Forecast method	FL07	FL08	FL09	FL10	FL11	FL26	FL27	FL38	FL39	FL43	FL44	FL49
Maximum of probability	8.32	2.06	1.89	1.62	3.02	2.65	1.24	3.49	2.48	5.5	1.40	2.31
200 random iterations	6.64	2.01	1.83	1.60	2.50	2.71	1.12	2.65	2.324	3.51	1.07	2.14

The test of the two ways to forecast area and métier choice (based on either the maximum probability or the simulated median method) for 2009, indicated that the median value derived from a random sampling of 200 alternative within the multinomial probability distributions estimated by the RUM best matched the observations. As shown in Table 2.5, the *MAE* was always lower with the random sampling method than with the method using the maximum of probability as a choice. Only the small dredger fleet had a better forecast with the maximum of probability method. On average, the percentage of error in the prediction (*MAE*) is low, in most cases less than 5%, and always less than 10%, which is confirmed by visual inspection (see examples in Figure 2.4 and Appendix I).

2.5. Discussion

In this study different drivers of fishers' behaviour were quantified using a random utility modelling approach. A novel dimension of our investigation is that, in addition to the explanatory variables usually considered in this type of exercise (e.g., expected revenue, tradition), we also considered the impact on the effort dynamics of selected French fleets, in terms of spatial interactions between fleets, the overlap with a spatially competing sector of activity (maritime traffic), and the area based management constraint (12-mile zone). Our results showed the existence of different behavioural dynamics, depending on the main gear used by the fleets and the size of the vessels in these fleets.

2.5.1. Models' selection

All of the models provided a reasonable fit to the 2007-2008 data, even though the IIA property was not satisfied. For spatial analysis, Wilen et al. (2002) have shown that the use of a conditional logit model often causes the IIA property to be at fault. An alternative used in many studies is the nested logit model (Holland and Sutinen, 1999; Marchal et al., 2009; Wilen et al., 2002). However, by considering the nested model, the IIA property is still not satisfied within each nest and the information provided is similar to that obtained with the conditional logit model. Train (2003) suggested using the mixed logit model, for which the IIA property is relaxed. Although the mixed logit model can also include choices and individual characteristics, it is also more difficult to interpret, and so was not tested in this study. While the IIA property was not respected, the conditional logit RUM fitted the 2007-2008 fishing effort data well, providing satisfactory predictions compared to the actual 2009 data (average prediction error always lower than 10%).

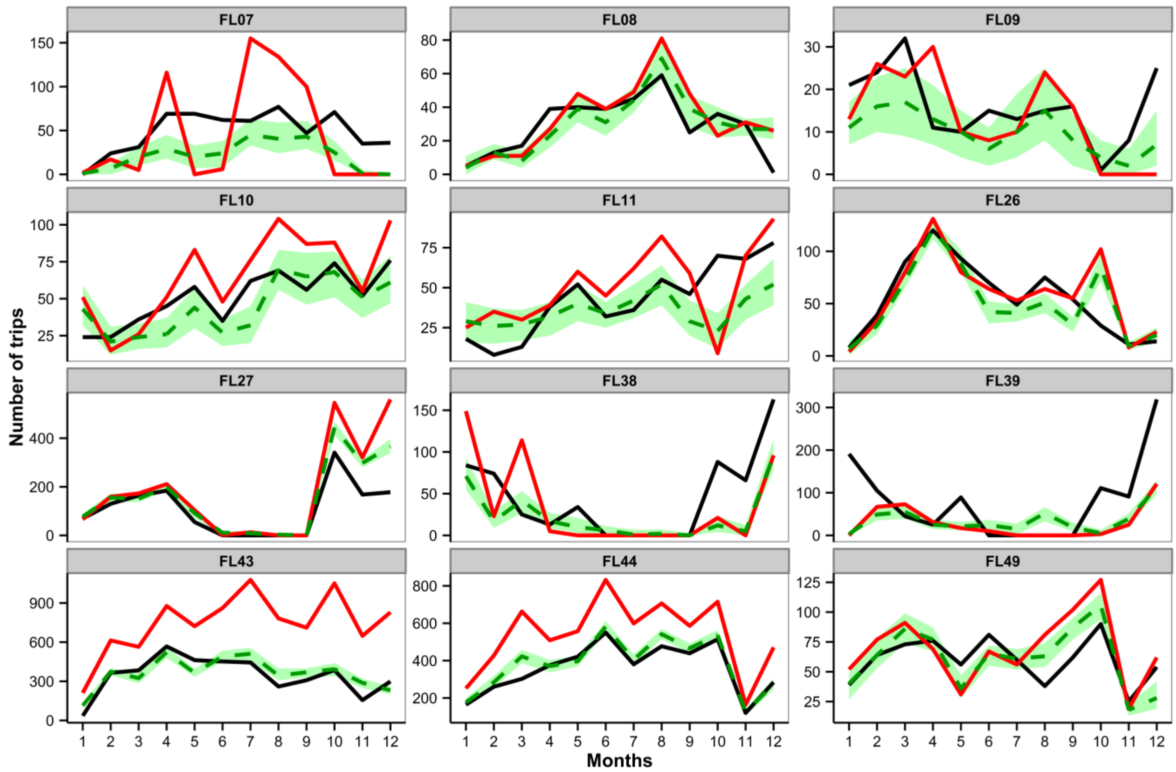

Figure 2.4 Example of forecast of data in 2009 in number of trips per month for most frequent alternative choice of each fleet: the fleet FL07 when other métiers are chosen in the area 27F0; the fleet FL08 when bottom otter trawling for demersal fish in area 27E9; the fleets FL09, FL10, FL11 when bottom otter trawling for demersal fish in the outside of area VIId; the fleet FL26 when other métiers are chosen in the area 29F1; the fleet FL27 when dredging for molluscs in the area 29F0; the fleets FL38 and FL39 when dredging for molluscs in the area 27E9; the fleet FL43 when other métiers are chosen outside of area VIId; the fleets FL44 and FL49 when trammel netting for demersal fish in the area 30F1. The dark line represents the observed choice in 2009, the red line represents the forecast based on the maximum of probability predictor, the green dotted line represents the median predictor derived from the 200 random iterations and the green area represents the range of predictors obtained with the 200 random iterations.

Another important finding of this study were the limitations associated with the maximum of probability method (e.g. amplification of model outliers) often used to simulate fisher's decision based on random utility models (e.g., Holland and Sutinen, 1999; Vermard et al., 2008; Marchal et al. 2009). We proposed here a method where an alternative is randomly sampled within the probability distribution derived from the RUM. This technique smoothens the predictions, and it also takes into account of the variability of the fitted model. However, this method is more computer-intensive due to the increased number of simulations that are needed to reduce prediction error.

2.5.2. Fishers' behaviour driven by past activities

The decisions made by the different fleets in our models are strongly determined by the past activity of each fleet and more precisely by their activity in the previous year. However, the analysis of active demersal fleets also highlights the importance of scallop dredging in the Eastern Channel, which to a large extent determines the short term behaviour of these fleets. Scallop dredging is prohibited to French vessels between the 15th of May and the 1st of October, by ministerial order. Given the economic importance of this activity in the overall pattern of fishing of the fleets, any change in the regulation of this métier can be expected to induce important modifications in fisher behaviour. This regulation implies a seasonal switch in the métier choice of demersal active fleets (Figure 2.2), which is reflected in the estimated coefficients. Hence, fishers' métier choices are negatively impacted by their past short-term effort allocation, which confirms the strong seasonal variations in fishing effort observed for these fleets. The fleets maintain a similar pattern of choice from one year to another that is shown by the positive value of the variable associated to long-term habits.

The influence of expected returns differs between the demersal trawlers and the other active gear fleets. The positive impact of the *VPUE_MONTH_1* on the small demersal trawlers, dredgers and polyvalent active gears (respectively FL07, FL26, FL27, FL38 and FL39) may be due to their ability to change métier relatively more easily compared to the larger demersal trawlers. Indeed, operators of these small trawlers are used to working across a greater diversity of fishing activities than those of larger trawlers. The large demersal trawlers (from FL08 to FL11) appear to be less reactive to changes in the relative profitability of alternative activities. Based on the model results, it appears that operators of these vessels tend to plan their fishing strategy based on the returns per métier in the previous year, when scheduling a change in the gear used and (or) in the area fished. The largest class of demersal trawlers (FL11) targeting fish (NOS05) as its main activity responds positively to variation in expected returns in the previous month, which could be explained by the fact that most of the activity of this fleet occurs outside of the Channel. The same hypothesis could be invoked to explain the behaviour of the passive gear fleet of vessels 10-12m in length (FL44), the activity of which is mainly focused on the use of trammel net (more than half of the fleet's effort is allocated to this métier)(NOS34). The only fleet with a negative response to relative expected revenue in the previous year is the dredger fleets of 12-18m vessels (FL27). This could be explained by two different hypotheses. Firstly, the effort allocation of this fleet could be explained by an increase of scallop biomass in 2008 compare to 2007. The impact of Scallop availability is shown in Figure 2.2 where the proportion of effort allocated to the dredge métier (NOS01) in May 2008 (more than 60%) is much higher than it was in May 2007 (20%). Secondly, fishers could have reached their scallop catch quota earlier than expected in the 2007 season, which could also explain the previous observation. However, the results obtained with respect to this 12-month lagged variable must be interpreted with great caution, since only two years of data have been used in this study.

2.5.3. Is there an impact of spatial management and other maritime activities on fisher's behaviour?

Large vessels fishing with active gears are spatially excluded from the fishing activity regulation within the 12-miles zone, inducing an allocation of their effort in the middle of the Channel. Their activity then competes with maritime traffic which is highly concentrated in this part of the Channel. Fishers seem to change their effort allocation during the period of the year with the most important shipping intensity, as shown by the negative coefficient for the variable $SURF\_AREA\_OCCUP_i$ for demersal trawlers (FL09 and FL10), dredgers and polyvalent vessels using active gears (respectively FL27 and FL39). By contrast, the small demersal trawlers fleet (FL07) and the fixed nets fleet (FL49) choose fishing areas where traffic is intense. Fleets of small vessels using active gears (FL07 and FL39) focus their activity in the inshore area, as shown by the positive correlation with the variable $SURF\_12NM_i$ where (except in the Dover Strait) they are not impacted by shipping. This fleet spends most of its fishing activity near the Dover Strait where the maritime traffic is the most intense due to the narrowing of the strait in that part of the Channel but this fact is not captured by the model. Unlike larger boats, smaller vessels using active gears are also limited in terms of the distance to fishing grounds (these vessels operate daily trips, have limited fish storage capacity and limited engine power). For the fleet fishing with fixed nets (FL49), the Strait of Dover corresponds to the presence of their target species and more particularly sole, which could explain the positive correlation of their area choices with the variable $SURF\_AREA\_OCCUP_i$ (Carpentier et al., 2009). This fleet thus allocates its effort in the statistical rectangle close to the Strait where shipping is the most intense. Moreover this fleet sets its nets on each side of the maritime traffic lines (Carpentier et al., 2009), while the demersal trawlers, dredgers and polyvalent vessels using active gears need to be able to travel across the VSS whilst fishing, which could explain the behaviour difference.

When the interaction between fishing fleets is significant, vessels seem to avoid areas occupied by other French fleets. Small vessels generally fish inshore, while larger vessels using active gears are not meant to be fishing within the 12 miles area, which could partially explain the spatial separation between these fleets. Another hypothesis could be that smaller vessels are able to profitably fish in areas with lower fish density than larger vessels. If this is the case, if localised depletion of fish or congestion of fishing capacity is observed in an area, smaller boats might be able to reallocate their effort to an area with lower fish density but with less competition. Moreover each target species get its own spatial distribution that could explain the difference of effort allocation observed between each French fleet. The model also, rather counter-intuitively, predicts that French 12-18 m demersal trawlers and dredgers, as well as both polyvalent active gear fleets (respectively FL09, FL27, FL38 and FL39) seem to prefer fishing in areas where UK vessels also allocate their effort. The English fleet is mainly composed of beam trawlers and dredgers both targeting the same species as the fleet segments in France. In particular, both the French and English vessels operate the métier targeting scallops (NOS01), a poorly mobile species, which probably explains why English and French fleets targeting scallops coexist on the same fishing grounds.

2.5.4. Forecast 2009 data

The forecasting model fitted the 2009 data reasonably well. This indicates that our model may be used to predict effort allocation one year ahead with a small level of error. By using the methods of forecast with several iterations, we take into account model variability and increase the accuracy of the prediction, even for the fleets with the weakest model fit. However RUMs are strongly data-driven and they need to be re-evaluated in case of a stepwise regime shift such as the introduction of a brand new spatial constraint (e.g., a marine protected area, or a wind farm). The model could be improved using finer scale data for fishing effort allocation (e.g. satellite based information). Such high resolution data could also be used to assess the impacts of aggregate extraction on fishing effort allocation. The use of detailed indicators of shipping intensity could also add information to our study.

2.5.5. Perspectives

To simulate the ecosystem conservation performances of different management regimes, this model needs to be integrated in a holistic modelling framework which can also predict the responses of the key target species to alternative harvesting patterns. Changes in spatial effort distribution and/or species targeting will change the dynamics of the underlying populations of these species, which might in turn lead to new changes in fishing effort allocation. Such a holistic model should in principle also take into account the process of entering and exiting the fishery. Some studies have already investigated this complex process (Le Floc'h et al., 2011; Pradhan and Leung, 2004b; Thébaud et al., 2006; Tidd et al., 2011), exploring the processes driving structural changes in fishing fleets. In the present paper the RUM can be used as the basis for a fleet dynamics sub-model in an existing holistic model such as ISIS-Fish (Pelletier and Mahévas, 2005), that simulates all the dynamics of the fishery from the biology of the target species to the response to management strategies, or Atlantis (Fulton et al., 2007) that takes into account all parts of the marine ecosystem in interaction with human activities and their management. Such coupled models can be used to test different management strategies and the effect of spatial interactions between different uses of the marine ecosystem.

2.6. Conclusion

In this study, RUMs have been used to understand fishers' behaviour interacting with other maritime activities in one of the busiest sea of the world, the Eastern English Channel. Several explanatory variables have been used in accordance with literature. To assess the impact of others maritime activities, the overlap between fishing activities, maritime traffic area and the 12 miles restricted management area has been built in our model. Finally, the between-fleets interactions are also represented in those models. Two different models have been tested, the conditional logit and the nested logit models. None of them fully satisfied the IIA property, and both fitted the 2007-2008 data similarly, so we selected the more parsimonious logit model in subsequent analyses. We showed that all of the fleets considered in this study were strongly influenced by their past activities with specific responses depending on the fleet considered. However, we also showed the importance of the maritime traffic which negatively impacted large vessels using active gears. To simulate the ecosystem conservation performances of different management, considering all of the interactions that occurred between the different maritime activities, this model needs to be integrated in a holistic modelling framework.

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3. Chapter 3: Thirty years of fleet dynamics modelling: what did we learn?

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3.1. Abstract

Anticipating fisher's behaviour is key to a successful implementation of marine ecosystem-based management. Of the different concepts that have been developed to mimic fisher's decision-making, Random Utility Models (RUMs) have attracted considerable attention by the scientific community in the past three decades. In our study, we aim at summarizing and analysing the information gathered from RUM used in the fisheries science literature during the last three decades in various places around the globe. A methodology has been developed to standardize information across the different studies and compare the results they obtained. Six fishers' behaviour drivers have been considered: the concentration of other vessels, tradition, expected revenue, species targeting, cost, and risk taking. We performed three separate linear modelling analyses to analyse the extent to which these different drivers impact fisher's behaviour. First, a binary analysis showed that fishers are attracted by their expected revenue, tradition, species targeting and concentration of other users, but avoid choices involving large costs. Second we evidenced that active demersal fleets are generally more driven by seasonal patterns than by short-term information. Finally, the comparison of other drivers compared to expected revenue highlighted that demersal vessels are risk–averse, and also that tradition and species targeting influence fishers; decisions more than expected revenue. Cost and concentration of other users have the same impact on fishers' decision-making as revenue. Pelagic fleets appear to consider all drivers as important as expected revenue but due to the lack of information on this group, results have to be considered with caution.

Keywords: Random Utility Model, fisher's behaviour, meta-analysis

3.2. Introduction

The exploitation of marine living resources generates substantial valued added at national and European levels. As a result, there has been an increasing societal and public demand from governance, industries and non-governmental organizations to provide sound and integrated scientific support to ecosystem-based management (EBM) (Arkema et al., 2006; Browman and Stergiou, 2004; Fulton et al., 2014; Garcia and Cochrane, 2005). In Europe, this move towards a more holistic and inclusive approach to the management of marine resources is reflected by the inception, in 2008, of the Marine Strategy Framework Directive (MSFD) (EC, 2008). The aim of the MSFD is to encourage sustainable use of marine resources, in accordance with current policies including the holistic EU Integrated Marine Policy (IMP) (EC, 2011) and, when it comes to fisheries management, the Common Fisheries Policy (CFP) (Roth and O'Higgins, 2011). A prerequisite to the effective application of the Ecosystem Approach to Fisheries Management (EAFM) is to better understand the different components of the ecosystem (Degnbol et al., 2006; Fulton et al., 2014). Fishers are key components of marine ecosystems, and understanding their behaviour is particularly critical to anticipate their likely responses to management measures (in terms of, e.g., spatial allocation of fishing effort, discarding practices), and the knock-on effects on impacted ecosystem components (Fulton et al., 2011; Hilborn, 2007; Leslie and McLeod, 2007).

Yet, the adaptability of fishers towards regulations and environmental variability has been often disregarded, leading to fisheries management failures (Daw and Gray, 2005; Hardin, 1968). In the past, different examples of stock collapses have been investigated, e.g., Caspian sea anchovy (Daskalov and Mamedov, 2007), Californian sardine (Radovich, 1982), North Sea herring (Dickey-Collas et al., 2010) or different North Atlantic cod stocks (Poulsen et al., 2006; Walters and Maguire, 1996). While recruitment failures, competition with other species, or exceptional environmental conditions have been highlighted, all these studies also identified misunderstanding of fishers' reactivity combined with fisheries management complexity as key drivers of these adverse conservation events (Allen and McGlade, 1987; Degnbol et al., 2006; Hilborn, 2007; Peterson, 2000).

The mechanisms of change in the behaviour of human agents have been widely studied using a variety of approaches, one of the most dominant found in the economic literature being discrete-choice modelling (Mc Fadden 1974; Greene, 2003; Train, 2003). A founding principle of discrete-choice models is that an agent facing multiple choices allocates a utility to each alternative, and then chooses that with the greatest utility. Discrete-choice models building in a random utility function, also known as Random Utility Models (RUMs), have been applied in various disciplines including, e.g., households' and consumers' preferences (Bougherara et al., 2009; Gracia and de Magistris, 2008; Zhang et al., 2009), school choice (Cohen-Zada and Sander, 2008; Glick and Sahn, 2006), or travelling options (Ettema et al., 2007; McFadden, 1974). RUMs have also increasingly been applied to fisheries, to analyse how fishers choose their fishing grounds (Hutton et al., 2004; Tidd et al., 2012; Wilen et al., 2002), their target species (Marchal et al., 2014; Pradhan and Leung, 2004; Vermard et al., 2008), their fishing gear (Andersen et al., 2012; Holland and Sutinen, 1999) or a combination of these (Marchal et al., 2009). Many other fleet dynamics studies have been conducted using RUMs, see van Putten et al. (2012) for a qualitative review. These studies have investigated the relative weights of different drivers on fishers' behaviour for a variety of countries, fishing fleets, fishing periods and underlying model structures. To our best knowledge, there has not been any attempt to gather and compare the outcomes of these numerous discrete-choice modelling studies. The objective of our study is therefore to review and compare, in a standardized fashion, the evidence drawn from fleet dynamics investigations building on RUMs which have been conducted in the past three decades. In doing so, we highlight the main key drivers of fishers' behaviour and investigate whether any common patterns could be detected across those case studies. Particular attention will be paid to how expected revenue drives fishers' behaviour compared to other possible drivers (e.g., traditions, target species), and also to whether fishers are more incentivized to make decisions based on short-term rather than longer-term (seasonal) information.

3.3. Materials and Methods

3.3.1. Materials

Most of our references have been collected from a previous qualitative fleet dynamics review (van Putten et al., 2012). Van Putten et al. (2012) present an overview of the different models and theories proposed and applied over the past three decades to explain and predict fishing behaviour. This review analysed also the different drivers tested and the nature of the fleet dynamics process being investigated (entry/exit, location choice, discarding…). Van Putten et al. (2012), however, did not attempt to compare the outcomes of the different fleet dynamics investigations, which is precisely the scope of our study. To complete our material, we browsed the fisheries and economic literature, to include scientific achievements derived from fleet dynamics studies that were conducted since 2010, and which were not considered by van Putten et al. (2012). Our literature review was based on several criteria. First of all, we focused on papers highlighting the factors driving fishers' decision-making. We further constrained our research to analyses based on utility maximisation theory and discrete-choice models, mainly RUMs (Greene, 2003). Also, the publications selected focused on fishing effort allocation, in terms of fishing ground, fishing gear, target species, or a combination of these (also called métier). Finally, only papers where the entire model output was presented have been retained.

Overall, 26 papers from several authors and case studies worldwide have been considered following our literature review, and these have been summarized in Table 3.1. Those studies have been mainly applied to fishing fleets operating in the EU, North America and Oceania, using data collected between 1976 and 2010. A total of 61 models were fitted to various fishing fleets. The most commonly used RUM are the conditional (Hutton et al., 2004; Marchal et al., 2014, 2014; Vermard et al., 2008) and multinomial logit models (Berman, 2007; Dupont, 1993; Maravelias et al., 2014; Mistiaen and Strand, 2000; Prellezo et al., 2009), nested logit model (Andersen et al., 2012; Bucaram et al., 2013; Campbell and Hand, 1999; Curtis and McConnell, 2004; Eales and Wilen, 1986; Holland and Sutinen, 1999, 2000; Smith, 2002; Smith and Wilen, 2003; Wilen et al., 2002) and the Mixed logit model (Eggert and Tveteras, 2004; Marchal et al., 2014; Pradhan and Leung, 2004; Tidd et al., 2012). Note that both the nested logit and the mixed logit models were often used to relax somehow the independence of irrelevant alternative choices property (IIA) (Greene, 2003; Train, 2003). For the purpose of this study, we grouped the different fleets examined in the 26 fleet dynamics papers into three main categories: active demersal, passive demersal and pelagic (Table 3.1).

In this study we compared the explanatory variables driving fishers' behaviour overall, but also across the different fleet categories being investigated. To our knowledge, no such analysis has ever been performed before. Although some authors have compared the outputs derived from different RUMs (Greene, 2003; Greene and Hensher, 2003; Koppelman and Wen, 1998; McFadden, 1974; Swait and Louviere, 1993; Train, 2003; Wen and Koppelman, 2001), these comparisons were performed, either with the same model structure (i.e. the same set of explanatory variables, to compare results across different fleets), or with the same input data (to compare model differences).

Table 3.1 References included in the analysis with a descriptive of their case study. Fleets selections and explanatory variable groups used in studies are displayed.

Reference	Ref. #	Area	Period	Method	Fleet	Fleet categories	Explanatory Variable	Model
(Andersen et al., 2012)	1	Danish North Sea	1997-2005	Nested logit	Gill netters >12m	Passive dem.	Revenue, Information, Risk, Tradition	1
(Berman, 2007)	2	Bering Sea USA	1998	Multinomial logit, Poisson	Demersal trawlers	Active dem.	Revenue, Cost	4
(Bucaram et al., 2013)	3	Galapagos	2002-2008	Nested logit	Lobster fishery	Passive dem.	Revenue, Cost	3
(Campbell and Hand, 1999)	4	Western Pacific USA	1988-1995	Nested logit	Tuna Purse seiners	Pelagic	Revenue, Tradition, Cost	1
(Curtis and McConnell, 2004)	5	Hawaii USA	1997	Nested logit	Pelagic longliners	Pelagic	Revenue, Information, Tradition	1
(Dupont, 1993)	6	Canadian British Columbia	1982	Multinomial logit	Seiners, gillnetters, longliners	Passive dem; Pelagic	Revenue, Risk	4
(Eales and Wilen, 1986)	7	California USA	1976	Nested logit, Logit	Shrimp trawlers	Active dem.	Revenue, Cost	2
(Eggert and Tveteras, 2004)	8	Swedish west coast	1995	Mixed logit	Demersal trawlers	Active dem.	Revenue, Risk, Tradition	1
(Holland and Sutinen, 1999, 2000)	9; 10	New England USA	1990-1993	Nested logit	Otter trawlers	Active dem.	Revenue, Information, Risk, Tradition, Cost	2
(Hutton et al., 2004)	11	English North Sea	1999-2000	Conditional logit	Beam trawlers	Active dem.	Revenue, Tradition	1
(Larson et al., 1999)	12	Bering Sea, Aleutian Islands USA	1991-1992	Logit	Demersal trawlers	Active dem.	Revenue, Information, Risk	1
(Maravelias et al., 2014)	13	Eastern Mediterranean	2000-2004	Multinomial logit	Purse seiners	Pelagic	Revenue, Information, Tradition	1
(Mistiaen and Strand, 2000)	14	Gulf and East Coast USA	1996	Multinomial logit	Pelagic longliners	Pelagic	Revenue, Risk	1
(Pradhan and Leung, 2004a)	15	Hawaii	1991-1998	Mixed logit	Pelagic longliners	Pelagic	Revenue, Risk, Tradition	1
(Prellezo et al., 2009)	16	VI,VII,VIII ICES areas Spain	1996-2002	Multinomial logit	Demersal trawlers	Active dem.	Revenue, Risk, Tradition, Cost	1
(Smith and Wilen, 2003)	17	Northern California USA	1988-1997	Nested logit	Urchin fishery	Passive dem.	Revenue, Cost	1
(Smith, 2002)	18	Northern California USA	1988-1997	Nested logit	Urchin fishery	Passive dem.	Revenue, Risk, Cost	1
(Tidd et al., 2012)	19	English North Sea	1997-2007	Mixed logit	Beam trawlers	Active dem.	Revenue, Tradition, Cost	9
(Valcic, 2009)	20	Oregon USA	1999-2002	Heteroscedastic Extreme Value	Demersal trawlers	Active dem.	Revenue, Tradition, Cost	1
(Marchal et al., 2014a)	21	Eastern English Channel France	2007-2008	Conditional logit	Demersal trawlers, dredgers, netters	Passive dem.; Active dem.	Revenue, Information, Tradition, Targeting	12
(Marchal et al., 2014c)	22	Eastern English Channel UK	2005-2010	Mixed logit	Dredgers	Active dem.	Revenue, Information, Tradition, Cost	1
(Marchal et al., 2014d)	23	Eastern English Channel Netherland	2002-2010	Mixed logit	Demersal seiners	Active dem.	Revenue, Information, Tradition, Cost	1
(Marchal et al., 2014b)	24	German Bight, North Sea Netherland	2008-2010	Conditional logit	Beam trawlers	Active dem.	Revenue, Information, Tradition, Targeting	8
(Vermard et al., 2008)	25	Bay of Biscay France	2000-2004	Conditional logit	Pelagic trawlers	Pelagic	Revenue, Tradition, Targeting	1
(Wilen et al., 2002)	26	Northern California USA	1988-1997	Nested logit	Urchin fishery	Passive dem.	Revenue, Cost	1

3.3.2. Standardizing fishers' behaviour drivers

Considering the variety of fishers' behaviour explanatory variables being used in the models reviewed, we first classified them into a few main categories: revenue, cost, risk-taking, traditions, targeting and the concentration of other users (Table 3.1). The first category, which strongly influences fishers' behaviour, is its experience, otherwise termed as habits or traditions. Traditions are usually included in the utility function as past effort (Holland and Sutinen, 1999; Tidd et al., 2012; Vermard et al., 2008). It is also commonly accepted that expected revenue, which represents the positive part of the overall fisheries profit, is a key economical behaviour component. So we gathered into the revenue category variables such as past gross revenue or value per unit effort, which are often used as proxy for economic opportunism. The fishing costs represent the negative part of the overall profit, and these were introduced in previous studies through proxies including fuel price, time spent at sea or distance from harbour (Berman, 2007; Bucaram et al., 2013). Fishers' attitude towards risk-taking has also been considered as a driver to their decision-making. Fishers have often been categorized in two categories: risk-seekers and risk-adverse (Andersen, 1982; Branch et al., 2006; Dupont, 1993; Hilborn and Ledbetter, 1979; Mistiaen and Strand, 2000). Risk-adverse fishers would be expected to choose alternatives with low revenue variability, while risk-seekers may take the chance of selecting more variable options. Risk-seeking behaviour, however, appears to be rare within fisheries and it may additionally be confounded with poor decisions based on a lack of information on fishing grounds' profitability (Branch et al., 2006). By contrast, risk-aversion is much more widely spread across fisheries, as fishers seem to seek areas likely to generate a stable revenue (Dupont, 1993; Hilborn and Ledbetter, 1979). The fishers' perception of risk has usually been represented by the variance of past revenues, when fishing in a given area or by using a given gear (Holland and Sutinen, 1999, 2000; Pradhan and Leung, 2004). In addition to their own experience, another way for fishers to gain information is to scrutinize the activity of other fishers, and then to move into areas where fishing vessels are most concentrated (Vignaux, 1996). On the other hand, the presence of too many vessels or other activities (maritime traffic, aggregate extraction, wind farms) could result in congestion or local stock depletion (Curtis and Hicks, 2000; Marchal et al., 2014). The presence of other agents in fishing areas is usually approximated by a metric representing their activity (e.g., total fishing effort or number of vessels in the case of fisheries). Finally, the last drivers' group considered is species targeting, which gathers variables referring to the price, the catch, or the Catch Per Unit of Effort (CPUE) of a particular species. Indeed, fishers can be constrained to target (or avoid) specific species assemblages as a result of management plans or quota availability.

In addition to the categorization of fishers' behaviour drivers into the six groups summarized above, we also discriminated some of these groups based on whether fishers use long-term (or seasonal) information made available during the previous year, or short-term knowledge from the previous month, day or fishing trip (Holland and Sutinen, 2000; Tidd et al., 2012). In our review, this time-scale differentiation has been applied to revenue and tradition variable groups.

3.3.3. Standardizing model outputs

Comparing the outcomes of 26 independent fleet dynamics studies, using different data inputs, model structures and explanatory variables, provides several challenges.

First, it is necessary to find a common standard score that could be used to compare the respective effects of the different factors potentially influencing fishers' behaviour across all models. We selected for that purpose the value of the test (t value or z value), used to assess the significance of the RUMs' estimated coefficients. This is calculated as the estimated mean value (μ) divided by standard deviation (σ). In the models we reviewed, more than one explanatory variable is generally associated to the same group, as defined in Section 3.3.2. Only the explanatory variables with a significant effect on fishers' behaviour were considered. To avoid a bias linked to the difference of variables' number per group between each models, we decided for each model to select only one explanatory variable per group. We chose to assign for each group (*g*) in each model (*m*) the variable with the highest absolute value of the ratio between estimated mean and standard deviation. The score used for subsequent analyses may be formulated as (equation 1)

$$\text{Score}_{m,g} = \text{Max}(|\mu_{m,g} / \sigma_{m,g}|) \quad (1)$$

In addition, due to varying model complexities and structures, the scores of the variables belonging to the same group could not be compared directly across the different case studies. To make drivers more comparable across case studies, we calculated the ratios of the scores between variables belonging to two different groups within the same model, instead of considering the absolute score values.

3.3.4. Analysis design

We performed three analyses to address three questions concerning fishers' behaviour. The first question is whether fleet dynamics drivers consistently have the same positive (attracting) or negative (avoiding) effects on fishers' choosing a given alternative, or whether these effects are case dependent. Second, we investigated whether fishers' decisions were more influenced by short-term or long-term drivers. Finally, we investigated the importance of the different fleet dynamics drivers relative to expected revenue, across the different RUM studies being reviewed.

3.3.4.1. Attraction or avoidance?

We analysed the sign of the estimated coefficient of each explanatory variable selected after calculating the score (Equation 1). In that purpose, we compared, for each variable group, the relative percentage of negative and positive values, with a Chi-square test. This analysis has been realised on all the data sets (Table 3.1), but also for each fleet (active demersal, passive demersal and pelagic fleets) separately if there were enough observations.

3.3.4.2. Short-term or long-term decisions?

As a result of data availability, only two groups were considered in the analysis, the revenue and the tradition, to compare the respective influence of long-term and short-term information. We investigated the inclination of each fleet type (*f*) for long- or short-term information by analysing, using a Generalized Linear Model (GLM), the logarithm of the ratio between the short-term and the long-term of explanatory variables belonging to the revenue and/or tradition groups (equations 2 and 3)

$$\text{Ratio}_{m,g,f} = \text{Score}_{m,g,f}\,(\text{short term}) \,/\, \text{Score}_{m,g,f}(\text{long term}) \tag{2}$$

$$\text{Log}(\text{Ratio}_{m,g,f}) \sim \text{Fleet}_f + \text{Method}_m + \varepsilon_{m,g,f} \text{ with } \varepsilon_{m,g,f} \sim N(0,1) \tag{3}$$

The Method_m factors represent the type of RUM or regression model used in the study *m*. It is added to the GLM to separate the response of the fleet type which we focus on, and the impact of the method (and/or of the particular geographical area) which is more difficult to interpret. The normality hypothesis is tested with the Shapiro Wilk test. Firstly, we assessed the overall influence of long-term versus short-term information on fishers' decisions by calculating the ratio between the highest long-term and short-term scores, irrespective of the driver to which these were associated. Second, we conducted the same calculations for each driver group separately. Due to data availability, we applied this latter approach, using equations (2) and (3), to revenue and tradition response values.

3.3.4.3. How do economics drive fishers' behaviour?

We compared the relative importance of expected revenue compared to that of the other key drivers selected in Section 3.3.2. It may be assumed that fishers are economic agents, and make decisions susceptible to maximize their expected revenue. We tested here the importance of the other behaviour drivers (fishing costs, attitude towards risk, habits, targeting, and concentration of other users) relative to expected revenue. To do so, we applied a similar methodology to the one used to test the relative importance of long-term versus short-term drivers in fishers' decision-making (Section 3.3.4.2). For each model, we calculated the ratio between each other drivers and the revenue. We focused on two different questions, on the one hand, how important are the different drivers, overall, compared to expected revenue and, on the other hand, how the attractiveness of the different drivers could differ across fishing fleets. To do so, two GLM analyses have been realised, one combining all fleet categories together (equation 4), and another one integrating a fleet effect (equation 5):

$$Log(Score_{m,g}/Score_{m,revenue}) \sim Groups_{m,g} + Method_m + \varepsilon_{m,g}\text{, for all } g \neq \text{revenue, with } \varepsilon_{m,g,f} \sim N(0,1) \quad (4)$$

$Groups_{m,g}$ is an explanatory factors that represent the sort of other drivers.

$$Log(Score_{m,g,f}/Score_{m,revenue,f}) \sim Fleet_f + Method_m + \varepsilon_{m,g,f}\text{, for all } g \neq \text{revenue, with } \varepsilon_{m,g,f} \sim N(0,1) \quad (5)$$

The residuals from both models were tested for normality.

All of the variable types where not present simultaneously in each paper reviewed. Due to this overall lack of consistency in information available, we only performed the analysis of ratio between one driver and the revenue when they were both investigated in the same paper. As a result, the set of case studies considered could be different depending on the driver being analysed (Table 3.2).

Table 3.2 Cases studies considered per analyses. The reference numbers are the one from Table 3.1.

Analyses	Models	Reference number
Ratio Short / Long term	Both variables	1; 9; 10; 19; 21 ;22 ;23; 24
	Revenue	9; 10; 21; 24
	Tradition	1; 9; 10; 19; 21 ;22 ;23; 24
Ratio Score $_g$ / Score revenue	Entire fleet	1-12; 15; 17-26
	Information	1; 5; 9; 10; 21; 23; 24
	Cost	1-4; 7; 9; 10; 17-19; 22; 23; 26
	Targeting	21; 24; 25
	Tradition	1; 4; 8-11; 15; 19,21-25
	Risk	1; 6; 8; 9; 10; 15

3.4. Results

3.4.1. Attraction or avoidance?

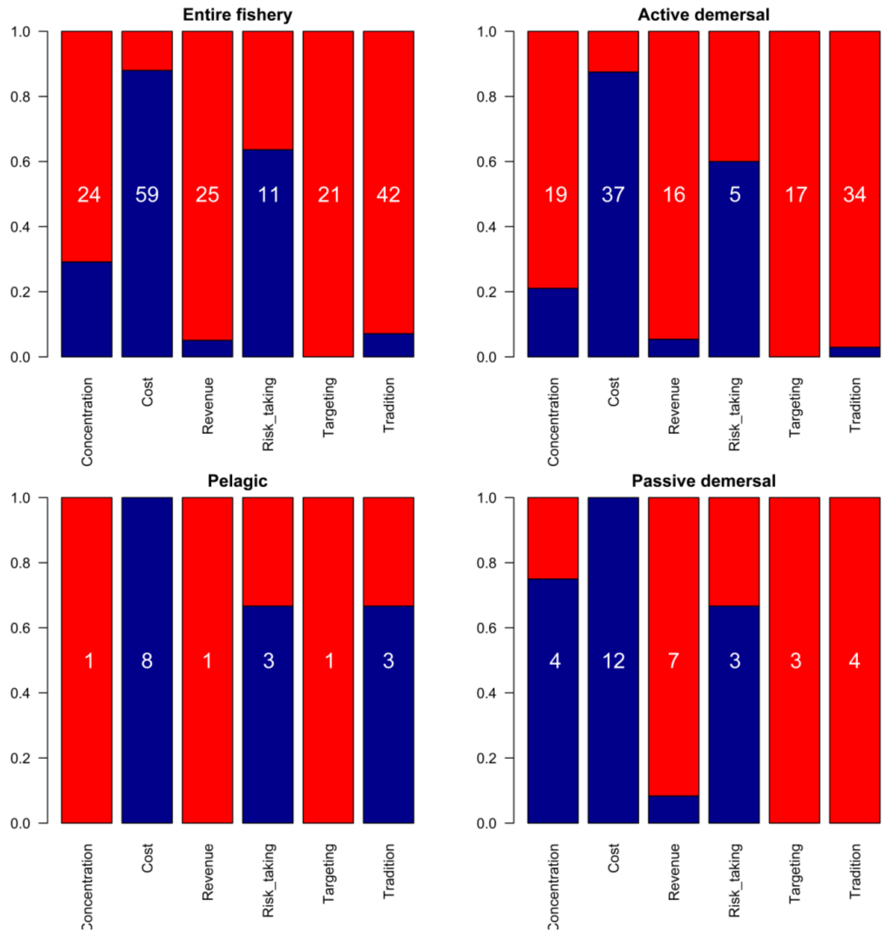

Figure 3.1 Observed proportion of positive coefficients relative to the different RUM explanatory variables (in red) compared to the negative ones (in blue), for each variable group. Those proportions are shown for the entire fishery and each fleet group separately. The total number of observations is indicated with a white colour.

The observed proportions of "plus" and "minus" signs for each fleet and fishers' behaviour drivers groups are shown in Figure 3.1. As a result of data availability, the sign analysis of the coefficients estimated for all of the different explanatory variables could be performed only when all fleets were combined. For the active demersal fleet alone, the analysis could be performed for all fishers' behaviour drivers, except for the risk group. With the passive demersal fleet alone, the analysis could only be carried out with expected revenues. There were no sufficient data to conduct any sign analysis with the pelagic fleet category separately. All of the proportions tested are significantly different form 50% except for the risk group (Table 3.3).

Table 3.3 Analysis of the explanatory variable negative versus positive proportions for each fleets and variable group with chi square test (p value: . <0.1; * <0.05; ** <0.01; * <0.001).**

Variable	Concentration	revenue	cost	Risk	Targeting	Tradition
Entire fleet	**4.1667 ***	**47.6102 *****	**14.44 *****	0.8182	**21 *****	**30.8571 *****
Active dem.	**6.3684 ****	**29.4324 *****	**9 *****	-	**17 *****	**30.1176 *****
Passive dem.	-	**8.3333 *****	-	-	-	-

For the entire fleet and the active demersal fleet specifically, concentration, revenue, species targeting and tradition are mainly positive, while cost has overall a negative effect. Expected revenue has a positive effect for passive demersal fleets, while risk-taking has a negative effect when all fleets are grouped together. Overall, it appears that two groups of explanatory variable can be separated, an "attraction" group including concentration, revenue, targeting, tradition, and an "avoidance" group including cost and risk-taking.

3.4.2. Short-term or long-term decisions?

We analyse the fishers' response to previous year drivers, compared to short-term drivers, for passive and active demersal fleets only. Short-term and long-term drivers were not tested simultaneously in the studies investigating the dynamics of pelagic fleets, and are therefore not considered here. We can observe that active demersal fleets are more reactive to past year drivers (mean value negative), compared to passive demersal fleets (Figure 3.2) that are more responsive to short-term drivers. More than 40% of the variability is explained by each model, more than half of which is explained by the fleet type, and the remaining part by the RUM Method applied and/or the geographical fishing area (Table 3.4). The residuals were normally distributed. Active demersal fleets' estimates are always significant and negative for the three models used.

Table 3.4 GLM analysis results for the comparison of long-term and short-term drivers scores (p value: . <0.1; * <0.05; ** <0.01; * <0.001).**

Models		Short/Long	revenue (Short/Long)	Tradition (Short/Long)
nb. Obs		46	16	30
Active_dem		**-0.7108 *** (0.1254)**	**-0.5996 ** (0.1399)**	**-0.7947 *** (0.1760)**
Passive_dem		0.2107 (0.3072)	-0.8437 . (0.4423)	0.6857 . (0.3836)
Mixed logit		**0.7057 ** (0.2076)**	0.4292 (0.2912)	**0.8391 ** (0.2640)**
Nested logit		**09101 ** (0.2934)**	**1.201 ** (0.3426)**	0.5396 (0.4033)
R^2 adjusted		**0.4234**	**0.5807**	**0.4519**
Explained variability	Fleet	0.24	0.35	0.32
	Method	0.23	0.34	0.21
Shapiro-Wilk test	W (p)	**0.9656 (0.2217)**	**0.9051 (0.0969)**	**0.9538 (0.2643)**

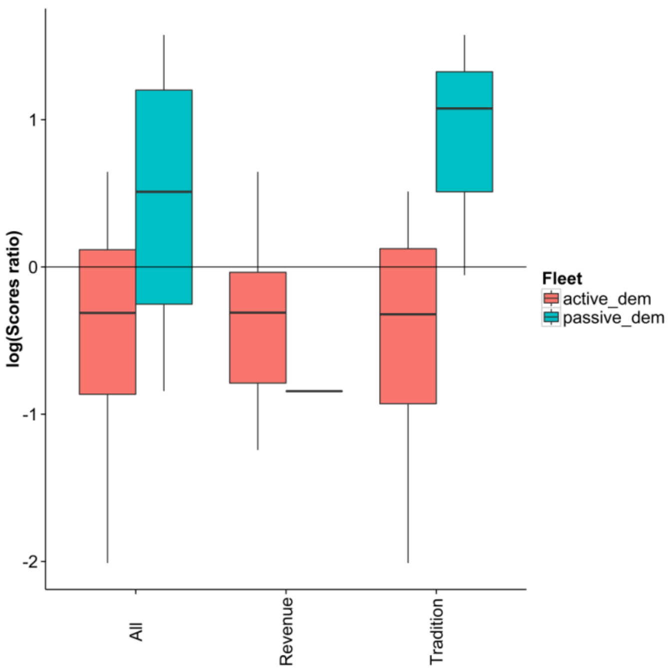

Figure 3.2 Logarithm of the ratio between short-term and seasonal scores observed for each fleet, with the revenue and tradition divers groups taken together or separately. Negative value represent a preference for past year driver contrary to positive one that reflect a greater importance of short-term information (Active fleet are in red and passive fleet in blue).

These fleets seem to be more driven by previous year knowledge and seasonal cycles than by more recent information. In contrast, passive demersal fleets seem to be more influenced by most recent evidence.

3.4.3. How do economics drive fishers' behaviour?

3.4.3.1. Relative importance of revenue as a driver of fishers' behaviour.

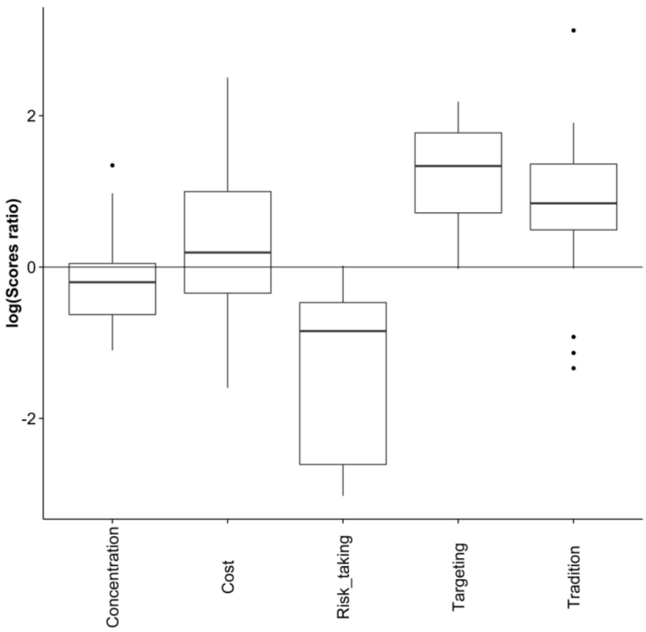

Figure 3.3 Logarithm of the ratio of the score of each drivers group (except revenue) and of the revenue score. Negative values reflect preference for the revenue explanatory variable.

With all fleets combined, revenue is more influential than risk-taking, but it is less important than fishing costs, species targeting and tradition (Figure 3.3 and Table 3.5). Revenue and the concentration of other users are given a similar weighting. 50% of the variability is captured by the model, most of which being explained by the drivers group (and 1% by the RUM method used). The residuals were normally distributed.

Table 3.5 Relative importance of revenue compared to other drivers of fishers' behaviour. Results of GLM analysis of score ratios as a function of variable group types and RUM methods (p value: . <0.1; * <0.05; ** <0.01; * <0.001).**

Variables	Factor levels	Estimates	Standard Deviation	Explained variability
Score ratio	Concentration/Revenue	-0.1078	0.3613	0.48
	Profit cost/ Revenue	**0.4990 .**	0.29977	
	Risk taking/ Revenue	**-1.1359 ****	0.3936	
	Targeting/ Revenue	**1.5418 *****	0.4110	
	Tradition/ Revenue	**1.1691 *****	0.3233	
Methods	Conditional logit	-0.3292	0.3529	0.01
	Mixed Logit	-0.3536	0.3432	
	Multinomial Logit	-0.4297	0.4866	
	Nested Logit	-0.0616	0.3312	
R^2 adjusted		**0.50**		
Shapiro-Wilk test: W (p-value)		**0.986 (0.31)**		

3.4.3.2. Do fishers from different fleets consider expected revenue in the same way?

Table 3.6 Relative importance of revenue compared to other drivers of fishers' behaviour, for each fleet type. One model has been performed for each score ratio type. Results of GLM analysis of score ratios as a function of fleet types and RUM methods (p value: . <0.1; * <0.05; ** <0.01; * <0.001).**

Models		Concentration/ Revenue	Cost / Revenue	Risk/ Revenue	Targeting/ Revenue	Tradition/ Revenue
nb. Obs.		20	23	9	18	36
Active_dem		**-0.3808 * (0.1305)**	**1.1859 . (0.6435)**	**-2.4372 ** (0.3065)**	**1.212 *** (0.1668)**	**0.8662 *** (0.1923)**
Passive_dem		-0.4007 (0.2329)	1.3049 (1.0028)	**-2.1108 ** (0.4335)**	**1.5688 *** (0.3604)**	**1.8008 *** (0.4005)**
Pelagic		**0.9739 . (0.5267)**	-0.4122 (1.4388)	0.0774 (0.4454)	0.1511 (0.6242)	-0.1444 (0.4844)
Logit		**1.7243 ** (0.4624)**	-0.7172 (1.2869)	-	-	-
Mixed logit		**0.8503 * (0.3398)**	-1.3498 (0.7881)	-0.3432 (0.4087)	-	-0.1508 (0.2854)
Nested logit		-0.0001 (0.2839)	-0.7676 (0.91)	**2.0587 ** (0.3684)**	-	-0.0979 (0.4374)
Multinomial logit		-	-0.6631 (1.0174)	-0.6981 (0.4454)	-	-
R^2 adjusted		**0.52**	**0.31**	**0.9881**	**0.79**	**0.63**
Explained variability	Fleet	0.23	0.18	0.66	-	0.63
	Method	0.42	0.13	0.32	-	0.08
Shapiro-Wilk test	W (proba)	**0.9577 (0.4718)**	**0.9802 (0.9102)**	**0.9561 (0.7408)**	**0.9782 (0.9299)**	**0.974 (0.5445)**

We generally observe a similar response to other drivers compared to expected revenue for all demersal fleets even if some slight differences remain between active and passive fleets (Figure 3.4 and Table 3.6). The dynamics of demersal fleets are more influenced by expected revenue than by concentration of other users and risk-taking, but fishing cost, species targeting and tradition remain their key drivers. Passive demersal fleets are more influenced by species targeting and tradition than active demersal fleets. By contrast, the behaviour of active demersal fleets is more guided by concentration of other users and risk-taking than that of passive demersal fleets. Despite less abundant data, we could see that pelagic fleets seemed more driven by expected revenue than by risk-taking, tradition and cost. Pelagic fleets are also more influenced by the concentration of other users than other fleets.

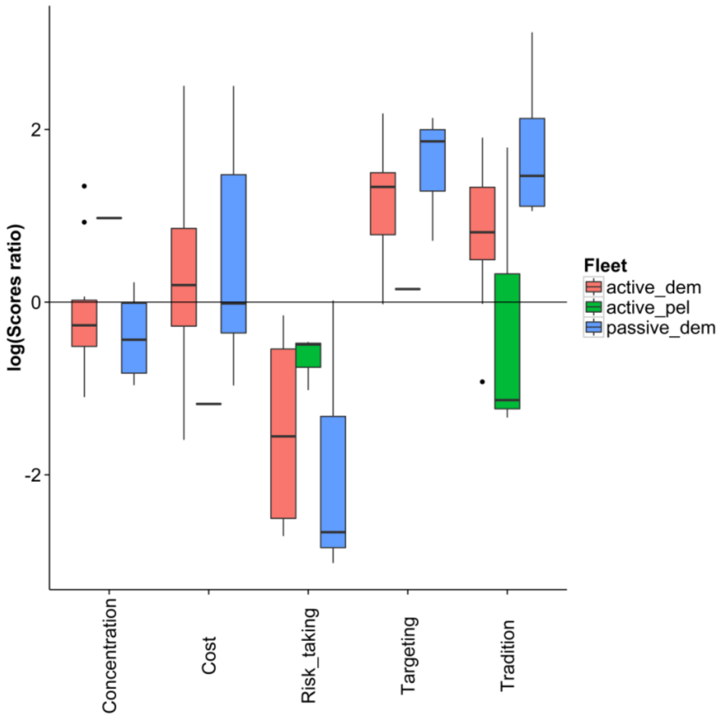

Figure 3.4 Logarithm of the ratio between the score of each drivers group (except revenue) and of the revenue score, for each fleet. Negative values reflect preference for the revenue explanatory variable (Active fleets are in red, pelagic fleets in green and passive demersal fleets in blue).

We obtain an adjusted R^2 above 50% for four out of the five models tested; the one performed on score ratio cost versus expected revenue got an adjusted R^2 adjusted of 31% (Table 3.6). The main part of the variability is explained by the fleet type except for the concentration versus revenue model, and the residuals were normally distributed. The Method factor has not been considered in the targeting versus revenue model, because those two variables are investigated simultaneously only in conditional logit approaches. The estimates associated with active and passive demersal fleets are significant (p = 0.05) for the scores of risk-taking, species targeting and tradition scaled to expected revenue. Concentration of other users is only significant for active demersal fleets. The estimates associated to pelagic fleets were not significant (Table 3.6). To help the understanding of models outcomes, we show the estimated coefficients drawn from each model and fleet type in a radar plot (Figure 3.5). Demersal fleets are more influenced by tradition and targeting than by revenue. However, the revenue expected by demersal fleets is more influential than the concentration of other users and risk-taking. No differentiation could be observed between cost and revenue.

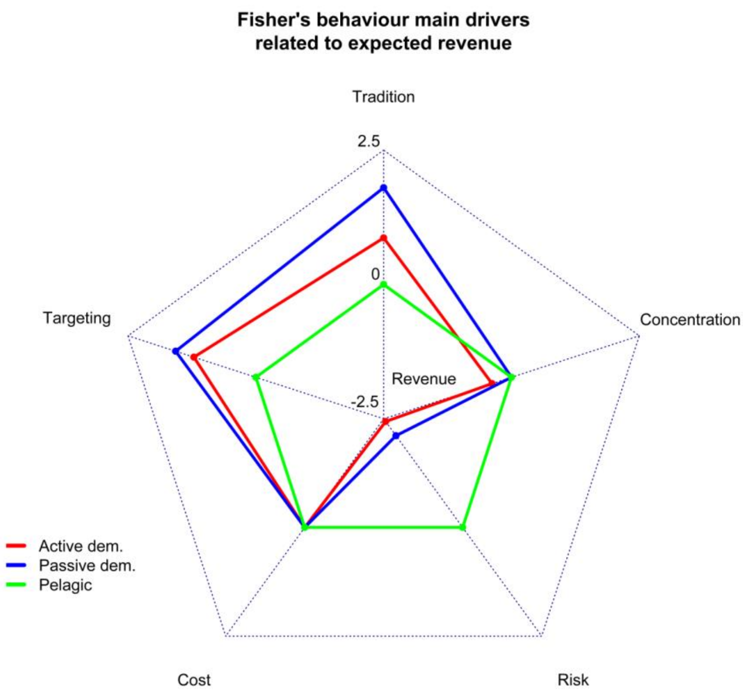

Figure 3.5 Relative importance of expected revenue, as a fishers' behaviour driver, compared to all other drivers. Each axis represents estimates of one driver compared to revenue. Any point inside the black dotted pentagon line indicates a preference for revenue compared to the other driver examined. Active demersal fleets are shown in red, passive demersal fleets in blue and pelagic pelagic fleets in green. Significant values with p<0.05 are represented others are set to zero.

3.5. Discussion

The detailed literature review of fleet dynamics studies building on discrete-choice models, conducted in the past three decades, led us to consider six main groups of fishers' behaviour drivers. Although these studies have investigated different types of alternatives (e.g., choice of fishing grounds, targeting species or gear types) (Andersen et al., 2012; Holland and Sutinen, 1999; Vermard et al., 2008), and considered a great variety of fishing fleets and RUM structures, our study highlighted some general patterns of fishers' responses in relation to each group of drivers. To perform our analysis, it was necessary to standardize both the model inputs and outputs across the different case studies being reviewed.

First of all, we gathered a variety of explanatory variables that often differed across case studies into six common groups of fishers' behaviour drivers. In addition, to deal with the complexity of each model and the presence of sometimes more than one explanatory variable per main driver group, we retained only one explanatory variable per main driver in each model. Although we systematically chose the variable with the highest score, this procedure may have decreased, in a few cases, the level of information provided by the models. Also, the influence of each main driver group in our analysis might be impacted by the number of variables belonging to that group included in the initial model. Finally, to alleviate the impact of RUM structure and parameterization in comparing drivers' effects across case studies, we analysed score ratios rather than plain scores (e.g., between short-term and long-term drivers, between revenue and other drivers), and the model effect was separated from the drivers and/or fleet effects using GLMs.

Overall, although our results should be interpreted cautiously due to the assumptions made to standardize the various RUMs' inputs and outputs and enable a comparison of their results, different main pattern responses have been found for drivers with differences between each fleet type that could be qualitatively explained by general fisheries knowledge, as elaborated below.

3.5.1. Attraction or avoidance?

As a result of data availability, the analysis of the sign of the drivers' effects focused only on the entire all-fleets fishery and on the active demersal fleet taken separately. The drivers could be categorized into "attraction" and "avoidance" groups.

As shown in several studies (Holland and Sutinen, 1999; Marchal et al., 2009; Pradhan and Leung, 2004; Vermard et al., 2008), fishers are motivated by past experience and expected revenue and they tend to make decisions that are in accordance with their habits and from which they expect a greater income. Our results bear out these conclusions.

Fishers were risk-adverse in most of the case studies we investigated, a result already anticipated by Dupont (1993) and Hilborn and Ledbetter (1979). In the fisheries science literature, risk-taking is always approximated by the variability of past revenues (Curtis and McConnell, 2004; Larson et al., 1999). We showed here that fishers generally preferred to minimize risk by looking for alternative decisions with a more stable expected revenue (Andersen, 1982; Bockstael and Opaluch, 1983).

However, few of them would still take the chance of visiting/operating uncertain areas/métiers, with the hope of achieving very high revenue (Mistiaen and Strand, 2000). Dupont (1993) pointed out that fishers are mainly risk-adverse. However, when harvested stocks are abundant and in good condition, some skippers could be inclined to select more risky options with the hope of earning outstanding returns. Although risk-seeking attitudes could not be fully evidenced in this study due to data limitations, there is evidence that fishers operating in pelagic fleets might be more risk-takers than in other demersal fleets (Mistiaen and Strand, 2000), possibly due to the large natural variability of the resources they harvest.

In the papers we reviewed, the concentration of other users was generally seen by fishers as a source of information rather than a congestion issue (e.g., Vignaux 1996). Campbell and Hand (1999) also suggested that it is common for vessels to share information with others or to track other vessels with the AIS (Automatic Identification System) on-board. The few cases where concentration had an avoidance (congestion or competition) effect occurred when other human users were included: maritime traffic, aggregate extraction (Marchal et al., 2014) or other fleet types (Hilborn and Ledbetter, 1979; Marchal et al., 2014). If we consider general behavioural patterns in ecology, competition amongst predators or parasites occur when multiple agents forage a common prey (Sutherland, 1983; Tregenza et al., 1996). Considering fishery dynamics, both resource and spatial competitions have been evidenced (Gillis, 2003; Samples, 1989). Fishers compete locally for resource when, in the context of stock depletion, the harvest of one fleet or boat affects the amount of fish left for others (Gillis and Peterman, 1998; Rijnsdorp et al., 2000). Spatial competition (or congestion) occurs when vessels crowding reduces their fishing efficiency (Gillis, 2003; Pet-Soede et al., 2001; Samples, 1989).

3.5.2. Is fishers' behaviour more influenced by seasonal or immediate knowledge?

In general, fishers have a tendency to follow their past exploitation pattern (Bockstael and Opaluch 1983, Holland and Sutinen 1999). The active demersal fleets seemed to favour seasonal over immediate information. By contrast, the passive demersal fleets adhered to their most recent, rather than their previous year's, fishing effort distribution. This might reflect that, compared to active demersal fleets, passive demersal fleets are often composed of small polyvalent multi-gear vessels, with a variable year-to-year fishing activity.

3.5.3. Which drivers for which fleets?

The relative importance of fishers' behaviour drivers differed substantially between demersal and pelagic fleets, even if the conclusions drawn from our analyses should be treated with caution due to the limited amount of fleet dynamics studies having investigated pelagic fisheries.

Demersal fleets are mainly driven by traditions and targeting, which all are more important than revenue. Pelagic fleets by contrast, seemed to give a similar importance to revenue, costs, tradition, targeting, risk and concentration of other users.

It came as no surprise that fishing costs had highest weighting for demersal active fleets, because towed gears are more energy-intensive and fuel-consuming than fixed nets. However, we anticipated costs to have more influence on pelagic fleets' behaviour. Fishing costs are often estimated by the distance to port, which does not consider the amount of time spent at exploring fishing areas. Due to the patchiness of their target species, the costs of searching for schools are likely important in the case of pelagic fleets, and should be considered as a potential driver when information becomes available. The main fishing costs for demersal passive fleets are induced by their travelling to fishing grounds. Because these fleets generally consist of small vessels, fishing grounds are often close to the coast, which might explain the lower influence of costs relative to expected revenue.

As expected (Wilson, 1990), tradition appeared to be the main driver of fishers' behaviour. Previous experience has been widely used as proxy of habits (Holland and Sutinen, 2000; Marchal et al., 2014; Valcic, 2009). However, as highlighted in the review of van Putten et al. (2012), the large explanatory power of tradition could be linked to a substantial overlap with economic opportunism. Fishers are hardly discovering new fishing grounds and fish on fishing grounds which theirs and others' experience have revealed as profitable a long time ago. However active fleets seem to be more opportunistic than passive ones. This observation might be due to vessels rigged with passive gears fishing on a more limited spatial extent (closer to the coast) compared to active fleets also they may be deployed at a particular time of the year or tide, and in a particular locality to target migrating species (e.g. gill nets). Pelagic fleets are more economically opportunistic than active demersal fleets. This might be because pelagic fleets tend to target patchy and migratory fish, which requires exploring greater areas than those covered by active demersal fleets.

The importance of species targeting as a major driver of fishers' behaviour driver, compared to overall revenue, bears out evidence from earlier studies (Marchal et al. 2009). Indeed, some fishers are subject to single-species landing restrictions (e.g., Total Allowable Catches) and also need to land species for which there is a market demand. Those two constraints may explain why demersal fleets target species for which they have quota and a market channel, rather than fish assemblages of a possibly greater value but which they would not be able to sell or even retain on-board. Pelagic fleets are selective vessels that usually target only one or two species. Therefore, the species targeting effect is strongly correlated with that of expected revenue and habits, and so might have a similar influence on their decision-making.

Fishers operating in demersal fleets were more concerned by expected revenue than by the uncertainty around these revenues (risk-taking) and the information drawn from other vessels' concentration. Risk-taking and the concentration of other vessels had a greater importance for the pelagic fleets compared to the demersal ones. Pelagic fleets target fish subject to large spatial fluctuations and spatial patchiness (shoaling). So, the greater consideration of risk-taking in pelagic fleet is an important component of their harvesting success. Moreover, the information gathered from others allow them to reduce their searching area and make fishing operations more profitable (Vignaux, 1996). In contrast, the species targeted by demersal fleets are generally less variable and distributed in a less patchy fashion, so they might be able to rely to a greater extent on habits and expected revenue.

3.6. Conclusion

In this review, we proposed a methodology to summarise, standardize and compare information collected in the past three decades on fishers' behaviour. We focused mainly on studies applying discrete-choice modelling and more particularly RUM to highlight fishers' main drivers. As expected, the main behaviour driver is the past experience of fishers with a particular influence of seasonal patterns in the case of active demersal fleets. However, species targeting may be as influential as tradition in the decision-making process. Cost and risk-taking are both disincentives for fishers, with costs having a greater influence on demersal active fleets. Finally, including a larger number of pelagic and passive fleets' case studies would be necessary to provide a more complete picture of the decision-making drivers for both those fleets.

3.7. Acknowledgements

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4. Chapter 4: Identification of the main processes underlying ecosystem functioning in the Eastern English Channel, with a focus on flatfish species, as revealed through the application of the Atlantis end-to-end model

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4.1. Abstract

The need for ecosystem based management is now widely accepted by the scientific and fishery management communities. In the meantime, we have also seen an increase of interest in ecosystem modeling and the development of many applications to better understand marine ecosystem functioning, which is crucial to support the development of future management plans of marine resources. We used the ecosystem model Atlantis to investigate the key dynamics and processes that structure in the Eastern English Channel ecosystem, with a particular focus on two commercial flatfish species, sole (*Solea solea*) and plaice (*Pleuronectes platessa*). This complex model was parameterized with data collected from diverse sources (a literature review, survey data, as well as landings and stock assessment information available from ICES working groups) and tuned so that the simulated biomass and catchs fitted 2002-2011 observations. Here, we mainly present the outputs for the two focus species and for some others vertebrates found to be important in the trophic network. The calibration process revealed the importance of coastal areas in the Eastern English Channel and of nutrient inputs from estuaries: a lack of river nutrients decreases the productivity of nursery grounds and adversely affects the production of sole and plaice. The role of discards in the trophic network is also highlighted. While sole and plaice did not have a strong influence on the vertebrates' trophic network, they are important predators for benthic invertebrates and compete for food with crustaceans, whiting (*Merlangius merlangus*) and other demersal fish. In addition, the model identified two key species, cod (*Gadus morhua*) and whiting. Using an end-to-end model substantially improved our understanding of the Eastern English Channel ecosystem structure and dynamics, and of the role of sole and plaice within this ecosystem.

Keywords: Ecosystem modelling, Eastern English Channel, Atlantis

4.2. Introduction

Over the last few decades, interest in ecosystem modelling has grown within both the scientific and fishery management communities (Arkema et al., 2006; Brodziak and Link, 2002; Browman and Stergiou, 2004; FAO, 2003; Fulton, 2010; Garcia et al., 2003; Sanchirico et al., 2006). In the meantime, understanding of marine ecosystems has also improved, and stakeholders have increasingly recognized the importance of accounting for ecosystem dynamics. Drawing lessons from past failures and successes (Daw and Gray, 2005; Hilborn, 2004; Hilborn et al., 2001), management has gradually been moving from traditional single-species considerations (Garcia, 1994; Ludwig, 2002; McAllister and Kirchner, 2002; Rosenberg, 2002) towards a comprehensive ecosystem-based management approach (EBM) building in the full complexity of ecosystem interactions (Botsford et al., 1997; Browman and Stergiou, 2004; Garcia et al., 2003; Pikitch et al., 2004). Although some multispecies models such as the Stochastic Multi-Species model SMS are already applied to assess fish stocks (Lewy and Vinther, 2004) , these only focus on commercial fish species and top predators interacting with these species (seabirds, seals and porpoises for the model developed in the North Sea)(ICES, 2014), but do not build in bottom-up processes (e.g., impact of prey abundance on predators' growth and survival).The Ecosystem Approach to Fisheries (EAF) has been adopted by many management institutions worldwide (Brodziak and Link, 2002; Sinclair and Valdimarsson, 2003) and in the EU, the holistic approach to ecosystem and resources management is a requirement of the Marine Strategy Framework Directive (MSFD) (EC, 2008a). EBM implementation requires a better understanding of the complexity of marine ecosystem interactions and of the impacts of multiple human activities; ecosystem models can help achieve that understanding (Browman and Stergiou, 2004; Fulton et al., 2011a; van Putten et al., 2012; Wilen et al., 2002).

A range of end-to-end models have been developed to emulate the dynamics of marine ecosystems. In some of these models, human activities are considered as a full component of the ecosystem (Leslie and McLeod, 2007), rather than a forcing driver. Ecosystem dynamics are driven by different types of processes, including hydrodynamics, biogeochemistry, habitat characteristics, population life cycles, trophic relationships, as well as the interaction with human activities. Coupling these different ecosystem components in holistic models is necessary to mimic the effects they exert on each other, and the extent to which such interactions could explain the fluctuations observed in key ecological or exploitation variables, e.g., biomass, catches, fishing effort (Fulton, 2010; Travers et al., 2007). One of the first ecosystem models that included a large set of species groups is the Ecopath with Ecosim (EwE) model. EwE represents the trophic connections in ecosystems (Christensen and Walters, 2004). EwE applications are now widely spread across the world and often include interaction with human activities. Lately, other models have emerged focussing on the spatial dimensions of fisheries dynamics and fish stocks, such as ISIS-Fish (Mahévas and Pelletier, 2004), or taking into account nutrient cycling, food web dynamics, and environmental variability, e.g., OSMOSE (Shin and Cury, 2001, 2004), APECOSM (Maury et al., 2007), NEMURO.FISH (Kishi et al., 2011), SEAPODYM (Lehodey et al., 2008) and Atlantis (e.g: Ainsworth et al., 2012; Fulton et al., 2005, 2007; Kaplan et al., 2012; Savina et al., 2013).

In our case study, we applied the Atlantis end-to-end model to mimic the dynamics of the Eastern English Channel ecosystem, the key components of which have already been well described (Carpentier et al., 2009). Past studies of this ecosystem have focused on hydrodynamics (Bailly du Bois and Dumas, 2005; Korotenko et al., 2013) and biogeochemistry (Beaugrand et al., 2000; Vanhoutte-Brunier et al., 2008), benthic fauna and substrate (Cabioch and Glaçon, 1975; Ellis and Rogers, 2000; Garcia et al., 2011; Holme, 1961, 1966; Larsonneur et al., 1982; Savina and Ménesguen, 2008), larval dispersal (Ellien et al., 2000), nursery grounds (Cugier et al., 2005; Riou et al., 2001; Rochette et al., 2010), fish assemblages (Vaz et al., 2007)(Vaz et al., 2007), fleet dynamics and fishery management (Batsleer et al., 2013; Marchal et al., in press), and spatial planning (Delavenne et al., 2012; Ulrich et al., 1999).

We focused more specifically on the trophic network involving two commercially important flatfish species: sole (*Solea solea*) and plaice (*Pleuronectes platessa*). Both species are nursery-dependent and mainly feed on benthic invertebrates and detritus that have been shown to be a key component of the Eastern English Channel ecosystem (Arbach Leloup et al., 2008; Dauvin and Desroy, 2005; Loizeau et al., 2001; Moore et al., 2004). Moreover, both species (and more particularly sole) are important for the French, UK, Dutch and Belgian fishing industries which target them to varying degrees. According to ICES, both sole and plaice are considered "true" resident Eastern English Channel stocks (unlike other species such as cod, whiting or herring which are more widely distributed outside the area), and their biomasses are assessed annually (ICES, 2004, 2005, 2011, 2013a, 2013b).

By using the Atlantis platform, we were able to provide a context for the dynamics of sole and plaice within an integrated picture of the system. The Eastern English Channel is strongly influenced by river inputs, mainly from the Seine River on the French side (Carpentier et al., 2009) and this was combined in Atlantis with other ecosystem and environment drivers (e.g., biogeochemistry cycle, benthic invertebrates' dynamics) to provide a comprehensive consideration of the influences on the two focus species. This represents a step forward in the understanding and representation of the Eastern English Channel ecosystem, which would not have been possible using other ecosystem-based modeling approaches (Plagányi, 2007).

This work builds on previous ecosystem modelling experience in the region. Two EwE models have been developed in this area, one for the entire English Channel (Stanford and Pitcher, 2004) and another for the Eastern English Channel specifically (Carpentier et al., 2009; Daskalov et al., 2011). The application developed by Daskalov et al. (2011) focusing on sand and gravel extraction in the English Channel has revealed the sensitivity of the model outputs to the mortality of benthic functional groupsand suggested that the EwE foodweb should be coupled with a biogeochemical model to more fully mimic the ecosystem dynamics; this is effectively what has been achieved with this new Atlantis model.

Atlantis applications have previously been used to consider the impact of spatial management (eg. in the Gulf of California (Ainsworth et al., 2012), along the California coast (Kaplan et al., 2012) and in South East Australia (Savina et al., 2013b) as well as other forms of fisheries and ecosystem based management approaches (Fulton et al., 2014). It has been used to examine the ecological-economic implications of alternative assumptions regarding the response of fishing fleets to changes in the economic circumstances of their activities (van Putten et al., 2013).

Atlantis is a “end-to-end” framework built for use in management strategy evaluation (MSE) (Fulton et al., 2005). It represents each important component of the management process (Jones, 2009), including the biophysical system, the human users of the marine resources, the three major components of an adaptive management strategy (monitoring, assessment and management decision processes) and socioeconomic drivers of human behaviour. Atlantis includes dynamic, two-way coupling of all these system components. The model is 3D spatialized and includes explicit physics and biogeochemical dynamics. The use of a biogeochemical framework allows the representation of bottom-up and top-down controls (Fulton et al., 2011b). There are currently 18 Atlantis models in use and more than 30 others under development across a range of scales and ecosystem types (Baltic Sea, North Sea, Strait of Sicily, Iceland, Great Lakes, Lake Victoria, Great Barrier Reef, Juan Fernandez Archipelago, Antarctica etc.).

The objective of this study is to build on the data and knowledge accumulated on the Eastern English Channel ecosystem and its exploitation to develop a decision enhance tool that could support ecosystem-based management and spatial planning of this maritime domain, with specific focus on sole (*Solea solea*) and plaice (*Pleuronectes platessa*) and their related fisheries. The main challenge in such an undertaking is to integrate available information on all ecosystem compartments from hydrodynamics to human activities in a single modelling framework, and to use this to successfully replicate recent observations of the Eastern English Channel for simulating correctly the recent observations. A full in-depth exposition of such a model is beyond the scope of a single paper and hereafter we focus on the ecological dynamics - presenting and discussing the results of the calibrated Atlantis Eastern English Channel (Atlantis-EEC) model. Although a wide range of outputs are available from the model, we focus here mainly on those functional groups directly related to plaice and sole in the trophic network of the Eastern English Channel. First we present the impact on vertebrate predators and prey, before presenting an analysis of biomass and catches of functional groups in the trophic network including plaice and sole. Finally the mortality rates of assessed species are presented.

4.3. Material and Methods

4.3.1. Atlantis

Atlantis is a modular modeling framework. Here we focus on the biophysical and fisheries modules, as the monitoring, assessment and management sub-models have not yet been used in this work. Interested readers can find further information on all aspects of the modelling framework in Fulton et al. (2011b).

As implemented in Atlantis-EEC each sub-model is deterministic and spatially resolved, discretized into irregular spatial polygons (allowing a match with the ecosystem spatial features). This structure facilitates the tracking of nitrogen flows through the trophic networks. Processes are implemented via differential equations, in this case using a time step of 24 hours. The main biophysical processes that are represented in the model include: consumption, production, waste production, movement and migration, predation, recruitment, habitat dependency and mortality. Functional groups can be either biomass pools (mainly used for invertebrates) or age class structured (for vertebrates). The fishing sub-model allows for multiple fleets, each with its own characteristics (including gear selectivity, habitat association, targeting, effort allocation and management structures). Atlantis also makes provision for fleet dynamics, economics and management rules, however these features have not yet been utilized in the context of this study (Fulton et al., 2011b).

4.3.2. MODEL STRUCTURE/ implementation in the Eastern English Channel

The Atlantis-EEC model is implemented for the ICES Division VIId area, which covers approximately 35,000 km². The model grid uses 35 polygons with three water column depth layers on the vertical axis (0-15m, 15-30m, and 30m to the bottom which attains 60m maximum in the Eastern English Channel), and a single sediment layer (Figure 4.1). Two of the polygons were boundary boxes representing the Western English Channel and the Southern North Sea, where conditions (nutrients concentrations, plankton abundance, salinity and temperature) were specified as forcing time-series rather than being calculated dynamically.

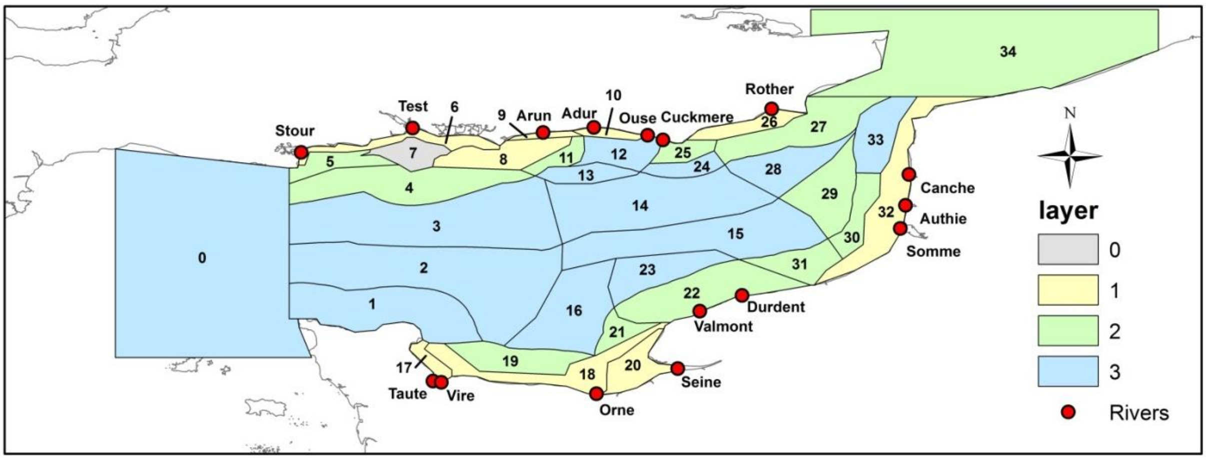


Figure 4.1 Spatial structure of the Atlantis application in the Eastern English Channel. The number of layers are shown with different colours, yellow for one layer (<15m), green for two (<30m) and blue for three (<60m). Red dots represent the position of river estuaries. The river names are indicated for reference.

The design of the spatial structure of the model was based on: (i) the biogeography of the Channel, i.e. the sediment partitioning between soft (mud and fine sand), flat (coarse sand and gravels) and hard (pebble) (Larsonneur et al., 1982); (ii) the EUNIS classification of the benthic habitat (Cabioch et al., 1978); (iii) the distribution of the demerso-benthic community (Vaz et al., 2007); (iv) the bathymetry; and (v) the main flatfish nursery grounds (Riou et al., 2001; Rochette et al., 2010) (Figure 4.2). In addition, we also explicitly marked out the administrative boundaries represented by the territorial waters of France and of the UK, as well as the coastal areas of these countries (12 nautical miles from the shore), where different regulations apply (access and gear restrictions, vessel size or horse power limitations) (Figure 4.2d).

Physical exchanges between boxes (advection and diffusion) were computed from outputs of the MARS3D hydrodynamic model, developed and tested by IFREMER and used as part of French coastal ocean forecasting program known as “Previmer” (https://www.previmer.org) (Bailly du Bois and Dumas, 2005;). Flows across each face of all the polygons were interpolated using R, by allocating each cell of MARS3D to a particular Atlantis polygon and integrating the flows of all MARS3D cells located at the boundaries of Atlantis polygons. The vertical flows were not available from the MARS3D model outputs, so these were calculated these taking into account both conservation of matter within each cell and the average sea level variation (derived from MARS3D output) in each polygon. The MARS3D outputs used corresponded to the simulation of the year 2006 (this was then continuously looped to allow the model to run indefinitely).

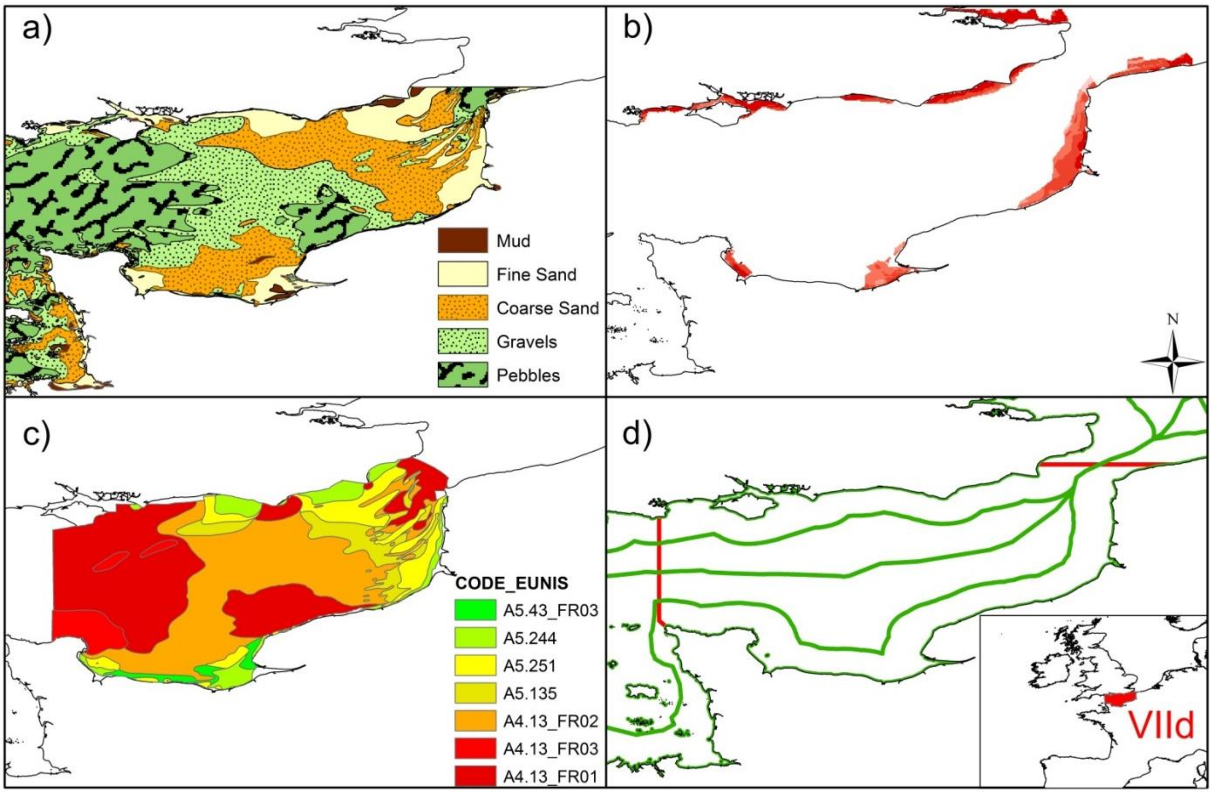


Figure 4.2 Biogeography and administrative boundary in the Eastern English Channel. The maps represent a) the distribution of sediment in the Channel separate in five sediment types defined by Larsonneur et al. (1982); b) the main nursery ground of Sole, but also most of the commercial species produced in the CHARM project 3 (Carpentier et al., 2009); c) the distribution of the bio-sediment EUNIS typology; and d) the boundaries of the VIId ICES area, the French and UK Exclusive Economic Zones and the 12 miles coastal waters zone.

As the Eastern English Channel is largely influenced by river nutrient flows, notably the Seine on the French side, 16 rivers (nine from France and seven from the UK) were represented in the model (Figure 4.1). Daily freshwater flows (from 2006) were extrapolated from available river gauging station datasets to obtain the flow at the river outlet. Time series of nutrient flows were then calculated from nutrient/flow relationships existing in the main rivers (Guillaud, 2008). Inter-polygons fluxes in Atlantis were then corrected to account for the amount of water coming from the rivers. We finally standardized these fluxes to account for hyperdiffusion (due to the assumption of uniform water column make up across the breadth of an individual polygon), by dividing the fluxes by the distance between each polygon's centroids. A tracer was used to check for any remaining hyperdiffusion (dummy concentration in R and nitrate in Atlantis).

For consistency with the fluxes, temperature and salinity were forced using the outputs of the MARS3D model (2006) averaged over the Atlantis polygons. Nutrient and organic matter concentrations were initialised using the outputs of the MARS3D model (2006) averaged over the Atlantis polygons.

Functional groups were defined on the basis of their habitat, preys and predators, growth characteristics (mainly maximum size and longevity) and migration patterns. Considering the variety of information available, and its lack of homogeneity, the groups were defined using expert judgment rather than using a clustering method. 40 functional groups were thus defined (Table 4.1). Vertebrates alone accounted for 21 of those groups, including a seabird and two mammal groups, seven groups of fish species of commercial interest and eleven other functional groups. We also included 16 groups of invertebrates, among which four planktonic groups and seven groups of commercial interest. Finally, detritus were binned into three separate functional groups as required by Atlantis (labile, refractory, carrion). Each vertebrate group was further subdivided into age classes; in this case each age class represents 1/10th of the total life span of the group (

Table 4.2). Amongst the invertebrates, cephalopods were represented as age-structured biomass pools (juveniles and adults), while each of the other functional groups were considered as single biomass pools per box.

The initial biomasses of the different groups were derived from a variety of approaches. Stock assessment outputs resulting from ICES analyses (International Council for the Exploration of the Sea) were available for six commercial species: sole, plaice, cod, whiting, herring and mackerel (ICES, 2004, 2005, 2011, 2013a, 2013b) (

Table 4.2). The stock distribution of sole and plaice is currently assumed to be restricted to ICES Division VIId, so the ICES estimated biomass could be used directly as input into our model. For cod, whiting, herring and mackerel, stock assessments consider larger areas than the Eastern English Channel, so biomass estimates were scaled down using the ratio between the landings over the entire areas versus those from ICES Division VIId. In doing so, we assumed that fishing activity and fish biomass overlapped spatially in the stock areas and that landings were proportional to biomass for each sector.

Table 4.1 Description of functional groups species composition in Atlantis Eastern English Channel application. Focus functional groups and key functional groups are underlined.

Code	Group	Species
SB	Seabirds	fulmar (*Fulmarus glacialis*), Manx shearwater (*Puffinus puffinus*), storm petrel (*Hydrobates pelagicus*), gannet (*Sula bassana*), cormorant (*Phalacrocorax carbo*), gulls (*Larus melanocephalus*, *Larus ridibundus*, *Larus canus*, *Larus fuscus*, *Larus argentatus*, and *Larus marinus*), kittiwake (*Rissa tridactyla*), terns (*Sterna sandvicensis*, *Sterna dougalli*, *Sterna hirundo*, *Sterna paradisaea*, and *Sterna albifrons*), guillemot (*Uria aalge*), razor bill (*Alca torda*) and puffin (*Fratercula arctica*)
CET	Toothed cetaceans	Harbour porpoise (*Phocoena phocoena*), common dolphin (*Delphinus delphis*) and longfinned pilot whale (*Globicephala melas*).
SXX	Seals	Grey seals (*Halichoerus grypus*) and harbour seals (*Phoca vitulina*)
<u>COD</u>	<u>Cod</u>	<u>North Atlantic cod (*Gadus morhua*)</u>
RAY	Rays and Dogfishes	cuckoo ray (*Raja naevus*),Common stingray (*Dasyatis pastinacus*), spurdog (*Squalus acanthias*), lesserspotted dogfish (*Scyliorhinus canicula*), smalleyed ray (*Raja microocellata*), greater-spotted dogfish (*Scyliorhinus stellaris*), blonde ray (*Raja brachyura*), longnosed skate (*Dipturis oxyrinchus*), blue skate (*Dipturis batis*) spotted ray (*Raja montagui*) and thornback ray (*Raja clavata*).
SHK	Sharks	tope (*Galeorhinus galeus*), porbeagle (*Lamna nasus*), blue shark (*Prionace glauca*), starry smooth-hound (*Mustelus asterias*), smooth-hound (*Mustelus mustelus*) and thintail thresher (*Alopias vulpinus*)
CEP	Cephalopods	veined squid (*Loligo forbesii*), European squid (*Loligo vulgaris*) and common cuttlefish (*Sepia officinalis*)
<u>WHG</u>	<u>Whiting</u>	<u>Whiting (*Merlangius merlangius*)</u>
POL	Pollack	Atlantis pollock (*Pollachius pollachius*)
LBT	Large bottom fish	Hake (*Merluccius merluccius*), anglerfish (*Lophius piscatorius*), ling (*Molva molva*) and conger eel (*Conger conger*)
BSS	Seabass	European seabass (*Dicentrarchus labrax*)
<u>SOL</u>	<u>Sole</u>	<u>common sole (*Solea solea*)</u>
<u>PLE</u>	<u>Plaice</u>	<u>European plaice (*Pleuronectes platessa*)</u>
DAB	Dab	common dab (*Limanda Limanda*)
OFF	Other flatfishes	lemon sole (*Microstomus kitt*), megrim (*Lepidorhombus whiffiagonis*),topknot (*Zeugopterus punctatus*), turbot (*Psetta maxima*), brill (*Scophthalmus rhombus*), European flounder (*Platichthys flesus*), Sand sole (*Pegusa lascaris*), Thickback sole (*Microchirus variegatus*), Solenette (*Buglossidium luteum*), scaldfish (*Arnoglossus sp.*)
MAC	Mackerels	North-east Atlantic mackerel (*Scomber scombrus*) and horse mackerel (*Trachurus trachurus*)
CLU	Clupeidae	Atlantic herring (*Clupea harengus*), European sprat (*Sprattus sprattus*) and European pilchard (*Sardina pilchardus*)
SPA	Sparidae	blackspot seabream (*Pagellus bogaraveo*),Common pandora (*Pagellus erythrinius*), gilthead seabream (*Sparus auratus*) and black bream (*Spondyliosoma cantharus*).

Table 4.1 (continued)

Code	Group	Species
GUX	Gurnards	red gurnard (*Chelidonichthys cuculus*), tub gurnard (*Chelidonichthys lucerna*) and grey gurnard (*Chelidonichthys gurnardus*)
MUL	Mugilidae	thinlip mullet (*Liza ramada*), golden grey mullet (*Liza aurata*), thicklip grey mullet (*Chelon labrosus*) and striped red mullet (*Mullus surmuletus*).
GAD	Other Gadoids	pouting (Trisopterus luscus), and poor cod (Trisopterus minutus)
SMD	Small demersal fishes	sand goby (*Pomatoschistus minutus*), hooknose (*Agonus cataphractus*), dragonet (*Callionymus maculates*),rokling fish, Greater weever (*Trachinus draco*), lesser weever (*Enchiichthys vipera*), Blenniidae and sandeels (*Ammodytes tobianus*, *Ammodytes marinus*) .
LBE	Lobsters	European lobster (*Homarus gammarus*)and spiny lobster (*Palinurus elephas*)
CRA	Crabs	shore crab (*Carcinus maenas*), common hermit crab (*Pagurus bernhardus*), hairy crab (*Pilumnus hirtellus*) ,velvet swimming crab (*Necora puber*), edible crab (*Cancer pagurus*), and spider crab (*Maja squinado*)
SHP	Shrimps	common prawn (*Palaemon serratus*), brown shrimp (*Crangon crangon*), Norway lobster (*Nephrops norvegicus*)
WHE	Whelks	common whelk (*Buccinum undatum*)
SUS	Suspension feeder	Benthic cnidarians, sponges, bryozoans and ascidians
DEP	Deposit feeder	polychaeta and amphipods
SCE	Scallops	common scallops (*Pecten maximus*), and queen scallops (*Chlamys opercularis*)
BIV	Bivalves	cockles (*Cerastoderma edule*), softshelled clam (*Mya arenaria*), blue mussels (*Mytilus edulis*), and oysters (*Ostrea edulis* and *Crassostrea gigas*)
ECH	Echinoderm	*Asterias rubens*, *Astropecten irregularis*, *Spatangus purpureus*, *Psammechinus miliaris*, *Echinus esculentus*, *Solaster endeca*, *Ophiura ophiura*, *Crossaster papposus*, *Echinocarsium cordatum* and *Ophiothrix fragilis*
ZOO	Zooplankton	
ZOC	Carnivorous zooplankton	
ZOG	Gelatinous zooplankton	
PP	Phytoplankton	
BB	Benthic bacteria	
PB	Pelagic bacteria	
DL	Labile detrital	
DR	Refractory detrital	
DET	Carrion/Discard	

Table 4.2 Initial biomass, age class structure and growth rate per class and length-weight relationship for vertebrates in the Atlantis Eastern English Channel application, after calibration. Biomass † from EwE (Carpentier et al., 2009), * from CGFS survey, # from assessment (ICES, 2004, 2013a, 2013b)

specie	Initial condition		Growth per class (mg N. d-1)										Length/weight	
	Biomass (tons)	age per class	1	2	3	4	5	6	7	8	9	10	a	b
SB	70.6†	4	1000	800	800	800	800	800	800	800	800	800	0.02	3
CET	125.5†	8	$9\ 10^{6}$	$2\ 10^{5}$	$5\ 10^{4}$	$5\ 10^{4}$	$5\ 10^{4}$	$5\ 10^{4}$	$5\ 10^{4}$	$5\ 10^{4}$	$5\ 10^{4}$	$5\ 10^{4}$	0.01	3
SXX	6.6†	5	$1\ 10^{6}$	$5\ 10^{4}$	5000	5000	5000	5000	5000	5000	5000	5000	0.035	2.9
COD	15 041#	2	34.52	54.52	250	250	250	250	250	250	250	250	0.00835	3.0532
RAY	13 128*	2	26.61	66.61	82.92	84.81	90.81	90.81	90.81	90.81	90.81	90.81	0.003048	3.1783
SHK	2 281*	5	1892.8	1646.1	1646.1	1646.1	1646.1	1646.1	1646.1	1646.1	1646.1	1646.1	0.00273	3.1533
WHG	123 499#	2	48.45	57.13	55.13	65.13	65.13	55.13	45.13	45.13	45.13	45.13	0.00621	3.1028
POL	8 292†	2	33.80	120.93	150	150	150	200	200	200	200	200	0.00613	3.1153
LBT	12 603†	3	180.51	483.39	802.93	1213.9	1219.2	1320.2	1320.2	1320.2	1320.2	1320.2	0.03328	2.7659
BSS	4 842†	2	5.49	10.21	20.86	51.37	81.37	81.37	101.37	101.37	101.37	101.37	0.01244	2.9526
SOL	14 099#	2	6.41	8.17	8.17	10.17	10.17	10.17	10.17	10.17	10.17	10.17	0.00391	3.2639
PLE	9 054#	2	30.57	80.44	100.44	121.44	121.44	101.44	81.44	81.44	81.44	81.44	0.0103	3.0169
DAB	20 563†	1	0.88	4.32	4.90	4.90	4.90	4.90	4.90	4.90	4.90	4.90	0.00547	3.2211
OFF	18 241†	2	5.05	6.62	7.49	7.49	7.49	7.49	8.49	8.49	8.49	8.49	0.01018	3.0514
MAC	87 530#	2	8.90	8.25	8.25	7.25	7.25	7.25	7.25	7.25	7.25	7.25	0.00338	3.1085
CLU	516 240#†	1	3	1.3	1.3	1.3	1.3	1.3	1.3	1.3	1.3	1.3	0.00564	3.0576
SPA	6 965†	2	6.07	12.45	30.29	30.29	30.29	30.29	30.29	30.29	30.29	30.29	0.0982	3.1414
GUX	15 588†	2	2.16	7.17	7.17	7.17	10.17	10.17	10.17	10.17	10.17	10.17	0.00528	3.1407
MUL	82 915†	1	4.35	6.06	8.81	10.26	12.44	12.42	12.42	12.42	12.42	12.42	0.00756	3.0574
GAD	126 030†	1	6.13	8.61	10.61	10.61	10.61	10.61	10.61	10.61	10.61	10.61	0.00728	3.1333
SMD	199 360†	1	0.15	0.15	0.15	0.15	0.15	0.15	0.15	0.15	0.15	0.15	0.0123	2.8092

Other sources of information included the CGFS bottom trawl survey (Carpentier et al., 2009) , the COMOR Bay of Seine scallops survey (Foucher, 2012), previous Ecopath with Ecosim (EwE) models (Carpentier et al., 2009; Daskalov et al., 2011; Mackinson and Daskalov, 2007) and ECOMARS3D outputs for the planktonic groups and nutrients (Table 4.3). The availability matrix (the prey's availability for each predator) was derived from the stomach contents database DAPSTOM (http://data.gov.uk/dataset/dapstom) (Pinnegar, 2014) and from the EwE models for all the functional groups (Table 4.4). Growth rates (Von Bertalanffy VB curve) and size-weight relationships were collected from Fishbase (http://www.fishbase.org/search.php) and from the Eastern English Channel atlas (Carpentier et al., 2009) (

Table 4.2). To allocate energy to reproduction after maturity in Atlantis, the initial growth rates of mature fish were considered constant and equivalent to the growth rate at first maturity. Using both, VB curves and size weight relationships we determined the initial weight of each age class per vertebrates functional groups. We assumed a Beverton and Holt stock-recruitment (SR) relationship for all the vertebrates except for mammals and birds, for which the amount of offspring per mature individual was assumed constant. SR curves were either drawn directly from ICES information (http://standardgraphs.ices.dk/stockList.aspx), or fitted using available stock assessment data (Table 4.5).

Table 4.3 Initial Biomass and biological parameters of invertebrates in the Atlantis Eastern English Channel application, after calibration. Biomass † from EwE (Carpentier et al., 2009), * from COMOR survey (Foucher, 2012) and # from ECOMARS3D.

species	Initial condition	Growth (mg N d-1)	Mortality (d-1)		
	Biomass (tons)		linear	quadratic	fishing
CEP	3130†	0.007	0	0	0.002
LBE	439†	0.0014	$1.6681\ 10^{-4}$	0.005	6.0910^{-5}
CRA	425 014†	0.006	0	$2\ 10^{-7}$	$4.67\ 10^{-4}$
SHP	391 101†	0.002	$8.1071\ 10^{-5}$	$5\ 10^{-4}$	$1.19\ 10^{-6}$
WHE	8 338†	0.016	0	$4\ 10^{-4}$	$1.23\ 10^{-4}$
SUS	171 154†	0.08	0	$1\ 10^{-6}$	0
DEP	829 146†	0.08	0	$1\ 10^{-7}$	0
SCE	379 854*	0.08	$7.4593\ 10^{-4}$	$7\ 10^{-7}$	$4.51\ 10^{-5}$
BIV	675 162†	0.08	0	$1\ 10^{-6}$	$3.37\ 10^{-6}$
ECH	296 396†	0.016	0.0001	$3\ 10^{-5}$	0
ZOO	19802#	0.2	0.05	$2\ 10^{-4}$	0
ZOC	15132#	0.04	$5\ 10^{-4}$	$1\ 10^{-5}$	0
ZOG	840†	0.02	0.0005	0.005	0
PP	297 015#	1	0.1	0	0
BB	90#	1.5	0.1	0	0
PB	13 823#	1.2	0.8	0	0

Table 4.4 Availability matrix in the Atlantis Eastern English Channel, after calibration. The information used to initialize those parameters is derived from EwE diet matrix (Carpentier et al., 2009; Daskalov et al., 2011; Mackinson and Daskalov, 2007) and DAPSTOM stomach content database. For vertebrates, Adults (adt) and juveniles (juv) were separated if their parameters were different.

		pred	**1**		**2**	**3**	**4**		**5**		**6**		**7**	**8**	
	prey	**stage**	**adt**	**juv**			**adt**	**juv**	**adt**	**juv**	**adt**	**juv**		**adt**	**juv**
1	**SB**														
2	**CET**														
3	**SXX**														
4	**COD**	**adt**			0.57	0.26	0.01								
		juv	0.03	0.03	0.2	0.26	0.05	0.005	0.027	0.002				0.002	0.004
5	**RAY**	**adt**									0.12	0.02			
		juv							0.01		0.01	0.02			
6	**SHK**	**adt**													
		juv									0.06				
7	**CEP**				0.424	0.08	0.002	0.004	0.037	0.001	0.203	0.12	0.025	0.008	0.021
8	**WHG**	**adt**			0.057	0.1	0.182		0.117			0.019		0.002	
		juv	0.09	0.09	0.196	0.1	0.06	0.046	0.093	0.022		0.005		0.02	0.007
9	**POL**				0.57	0.15	0.04		0.002	0.002					
10	**LBT**	**adt**				0.28					0.01				
		juv				0.28	0.04		0.04		0.04				
11	**BSS**														
12	**SOL**					0.3	0.017	0.004	0.029				0.004	0.003	0.002
13	**PLE**	**adt**				0.3	0.01		0.054		0.061				
		juv				0.3	0.153		0.054		0.061				
14	**DAB**					0.3	0.17	0.006	0.037			0.007		0.007	
15	**OFF**					0.3	0.055	0.022	0.022					0.006	0.004
16	**MAC**		0.108	0.108	0.18	0.1	0.015	0.001	0.024		0.2	0.37	0.0094	0.01	0.002
17	**CLU**	**adt**	0.02	0.02	0.0212	0.0012	0.0056	0.0015	0.0075	0.0075	0.0126	0.0106	0.005	0.0096	0.015
		juv	0.021	0.02	0.0212	0.0012	0.0056	0.00175	0.0075	0.0075	0.0013	0.0036	0.00	0.0111	0.015

Table 4.4 (continued)

		pred	1		2	3	4		5		6		7	8	
	prey	**stage**	**adt**	**juv**			**adt**	**juv**	**adt**	**juv**	**adt**	**juv**		**adt**	**juv**
18	**SPA**				0.057								0.003		
19	**GUX**						0.006	0.002	0.002		0.048				
20	**MUL**								0.06	0.06			0.01		
21	**GAD**		0.002	0.002	0.048	0.1	0.132		0.258	0.015	0.01	0.003	0.084	0.1195	0.046
22	**SMD**		0.075	0.075	0.0537	0.025	0.04	0.092	0.1375	0.0992	0.01625	0.00625	0.0555	0.0412	0.0412
23	**LBE**														
24	**CRA**		0.00016	0.00016			0.00081		0.00175	0.00081	0.000035	0.00014	0.00276	0.000255	0.00005
25	**SHP**						0.00162	0.003	0.0005	0.00053		0.00084		0.0005	0.003
26	**WHE**								0.00007	0.00007					
27	**SUS**								0.0001	0.0001					
28	**DEP**		8 10-5	8 10-5			2.8 10-5	0.001	0.0003	0.0003			7.5 10-6	0.000175	0.00005
29	**SCE**														
30	**BIV**				0.00001				0.00004	0.00004				0.00015	
31	**ECH**						0.00009		0.00007	0.00007				0.00003	
32	**ZOO**							0.0001		0.0001					0.0009
33	**ZOC**			0.0005				0.0005		0.00017					0.0009
34	**ZOG**								0.00036			0.00027			
35	**PP**												1 10-9		
36	**BB**														
37	**PB**														
38	**DL**														
39	**DR**														
40	**DET**		0.1	0.1	0.1	0.1					0.1	0.1			

Table 4.4 (continued)

		pred	**9**		**10**		**11**		**12**		**13**		**14**	
	prey	**stage**	**adt**	**juv**	**adt**	**juv**	**adt**	**juv**	**adt**	**juv**	**adt**	**juv**	**adt**	**juv**
1	**SB**													
2	**CET**													
3	**SXX**													
4	**COD**	**adt**			0.09	0.09								
		juv			0.09	0.09								
5	**RAY**	**adt**												
		juv			0.006	0.006								
6	**SHK**	**adt**												
		juv												
7	**CEP**				0.071	0.071	0.002							
8	**WHG**	**adt**			0.092	0.092								
		juv			0.092	0.092								
9	**POL**													
10	**LBT**	**adt**												
		juv			0.003									
11	**BSS**													
12	**SOL**										0.003			
13	**PLE**	**adt**			0.003	0.003								
		juv			0.003	0.003								
14	**DAB**				0.024	0.024								
15	**OFF**				0.1	0.1								
16	**MAC**				0.053	0.053								
17	**CLU**	**adt**			0.012	0.012					0.000375			
		juv			0.012	0.012					0.000375			

Table 4.4 (continued)

		pred	9		10		11		12		13		14	
	prey	stage	adt	juv	adt	juv	adt	juv	adt	juv	adt	juv	adt	juv
18	SPA													
19	GUX													
20	MUL													
21	GAD		0.41	0.41										
22	SMD		0.118	0.118	0.037	0.037	0.1705				0.05		0.0165	0.0165
23	LBE													
24	CRA				3.5 10-5	3.5 10-5	0.00271	0.00025			0.00008		0.00005	0.00005
25	SHP				0.00066	0.00066	0.00117	0.0045	0.0013	0.0013	0.00015	0.0047	0.00045	0.00045
26	WHE													
27	SUS		0.003	0.003										
28	DEP		0.00125	0.00125	0.00015	0.00015	0.000015	0.001125	0.00125	0.00125	0.0015	0.000282	0.000945	0.000945
29	SCE								0.00037	0.00037				
30	BIV				0.00016	0.00016						0.00013		
31	ECH													
32	ZOO			0.00002				0.0005		0.00001		1 10-5		0.00013
33	ZOC			0.00002				0.0005		0.00001		1 10-5		0.00013
34	ZOG										0.00003			
35	PP													
36	BB													
37	PB													
38	DL													
39	DR													
40	DET				0.1	0.1								

Table 4.4 (continued)

		pred	15		16		17		18	19		20		21	
	prey		adt	juv	adt	juv	adt	juv	adt	adt	juv	adt	juv	adt	juv
1	SB														
2	CET														
3	SXX														
4	COD	adt													
		juv	0.051	0.051	0.006	0.006				0.026	0.026				
5	RAY	adt													
		juv													
6	SHK	adt													
		juv													
7	CEP		0.003	0.003	0.017	0.017			0.09	0.015	0.015			0.001	0.001
8	WHG	adt													
		juv	0.4	0.4						0.087	0.087			0.004	0.004
9	POL														
10	LBT	adt													
		juv													
11	BSS														
12	SOL														
13	PLE	adt													
		juv													
14	DAB		0.16	0.16						0.009	0.009			0.005	0.005
15	OFF		0.088	0.088						0.015	0.015				
16	MAC				0.006	0.006				0.001	0.001				
17	CLU	adt	0.0375	0.0375	0.00375	0.00375				0.001875	0.001875			0.0005	0.0005
		juv	0.0375	0.0375	0.0075	0.0075				0.001875	0.001875			0.0005	0.0005

Table 4.4 (continued)

		pred	15		16		17		18	19		20		21	
	prey	stage	adt	juv	adt	juv	adt	juv	adt	adt	juv	adt	juv	adt	juv
18	SPA														
19	GUX				0.005	0.005									
20	MUL														
21	GAD		0.25	0.25	0.046	0.046				0.069	0.069			0.017	0.017
22	SMD		0.0775	0.0775	0.025	0.025			0.0005	0.025	0.025			0.01875	0.01875
23	LBE														
24	CRA		0.000595	0.000595	8.5 10-5	8.5 10-5				0.00031	0.00031	0.00014	0.00014	0.00139	0.00139
25	SHP		0.0004	0.0004	5 10-5	5 10-5				0.0019	0.0019			0.00001	0.00001
26	WHE														
27	SUS				0.00069	0.00069			0.00038					0.00456	0.00456
28	DEP		0.00125	0.00125	0.00125	0.00125			0.002	0.00075	0.00075	0.002	0.002	0.00075	0.00075
29	SCE														
30	BIV		0.00128	0.00128	7 10-5	7 10-5			1 10-5					2 10-5	2 10-5
31	ECH		0.00032	0.00032					0.00053						
32	ZOO			0.00003		0.008		0.01			0.003		0.0014		0.00014
33	ZOC			0.00003	0.008	0.008	0.01	0.01		0.0003	0.003	0.00014	0.0014	0.000014	0.00014
34	ZOG		0.00071	0.00071											
35	PP						1 10-9	1 10-9							
36	BB														
37	PB														
38	DL				0.002	0.002	0.002	0.002				0.002	0.002		
39	DR											0.0001	0.0001		
40	DET				0.02	0.02	0.002	0.002				0.02	0.02		

Table 4.4 (continued)

		pred	**22**		**23**	**24**	**25**	**26**	**27**	**28**	**29**	**30**	**31**	**32**	**33**	**34**
	prey	**stage**	**adt**	**juv**												
18	**SPA**															
19	**GUX**															
20	**MUL**															
21	**GAD**															0.03
22	**SMD**															
23	**LBE**															
24	**CRA**		0.00139	0.00139	0.0015	0.00015		0.0005								
25	**SHP**		0.00041	0.00041		0.002		0.0005								0.00064
26	**WHE**				0.00032											
27	**SUS**				0.00048								0.00005			
28	**DEP**		0.002	0.002	0.0000625	0.0005	0.000075	0.00175					0.000275			
29	**SCE**															
30	**BIV**		0.0006	0.0006	0.000107	0.0002		0.0001					0.0005			
31	**ECH**		0.00014	0.00014	0.00168			0.0005					0.0006			
32	**ZOO**			0.00003			0.0036		0.0045					0.0003	0.01	0.003
33	**ZOC**		0.00003	0.00003			0.0036								0.0002	0.003
34	**ZOG**															
35	**PP**		1 10-09	1 10-09	0.0001		0.001		0.005		0.005	0.005	1 10-09	0.1		0.001
36	**BB**															
37	**PB**		0.0005	0.0005												
38	**DL**				0.001	0.008	0.005	0.005	0.005	0.01	0.005	0.005	0.00725	0.0007		
39	**DR**				0.001	0.008	0.005	0.005	0.005	0.01	0.005	0.005	0.00725	0.0007		
40	**DET**				0.1	0.5	0.5	0.5		0.7			0.0725			

Vertebrates, cephalopods and scallops spatial distributions were initialised based on existing Eastern English Channel habitat models (Carpentier et al., 2009) or surveys. For five functional groups (mackerel, whiting, sea bass, herring within the clupeidae and cephalopods), annual migrations and spawning were implemented by setting periods of migration and spawning based on Pawson (1995) (Appendix II and Appendix III). Plankton and invertebrates groups were initialised homogeneously throughout the model domain, except for the initial scallop spatial distribution, which was based on COMOR survey abundance indices and commercial catches.

Fisheries were explicitly built in our model through a selection of fishing fleets operating a variety of métiers. Both fleets and métiers were defined using the EU DCF (Data Collection Framework) terminology (EC, 2008b). A métier is characterized by the type of gear used and the species or group of species targeted during a fishing operation. A fleet is a group of fishing vessels of similar characteristics (size, power, capacity, etc.) and operating the same main métier during the year. 62 fleets could be identified using this typology. In this study, we focused on the French fishing fleets targeting sole, which has traditionally been one of the main commercial species in the Eastern English Channel. This reduced the number of DCF fleets to 20 (essentially netters and dredgers), to which we added one group to include all the other French and foreign vessels operating in the Eastern English Channel (hereby referred to as the international fleet), making it 21 fleets overall (Table 4.6).

The main target species of each fleet were assigned according to catch compositions observed over the period 2002-2011. Those data are collected by the French Directorate for Marine Fisheries and Aquaculture (DPMA) from mandatory fishers' logbooks, and stored in the IFREMER Harmonie database. We also used the data collected during the French at-sea observers program OBSMER, to quantify the catch and discard per species and size class in the French fishery. Discard per species were implemented as a proportion of catch per functional groups for each fleet.

Table 4.5 Reproduction and mortality parameters for vertebrates in the Atlantis Eastern English Channel application, after calibration. Two reproduction relationships are used: a Beverton and Holt stock recruitment curve (BH), or a number of recruits per adult. Only the fishing mortality induced by the international fleet is shown.

species	Reproduction				Mortalities (d^{-1})(10^{-6})		
	class at first maturity	BH α (10^9)	BH β (10^8)	# per adult	natural linear		fishing
					juvenile	adult	
SB	2			0.38	500.0	100.0	0
CET	2			0.5	1100.0	0.1	0
SXX	2			0.5	800.0	50.0	0
COD	2	0.05	5.00		160.0	263.36	414.778
RAY	2	0.17	6.00		200.0	22.703	586.306
SHK	2	0.00392	20.0		0.1	0	752.5
WHG	2	18.0	16.7		0	0	867.605
POL	2	0.012	1.16		200.0	548.11	80.6
LBT	2	0.008	1.39		100.0	228.77	63.7
BSS	3	0.07	0.676		8.0	0	695.239
SOL	2	0.41	3.72		0	0	193.887
PLE	2	0.98	1.49		0	0	2904.508
DAB	3	12.5	14.9		0	0.02154	1791.801
OFF	2	0.14	5.50		4.0	2.5674	108.316
MAC	2	4.6	10.0		0	0	263.036
CLU	2	30 000	230.0		0.02	0	456.967
SPA	2	0.7	1.00		40.0	442.2	674.114
GUX	2	0.42	1.00		20.0	240.76	49.2636
MUL	2	2.8	10.0		300.0	840.48	49.7
GAD	2	100.0	32.4		0	0	143.0
SMD	2	58 000	48.7		0	0	10.5

Table 4.6 Description of Atlantis Eastern English Channel combinations of DCF fleets and métiers, with their implementation, selectivity curves type and parameters' values.

Index	Code	DCF fleets	DCF métiers	Implementation	Selectivity	α(cm)	β	Main species
FC1	fl26dredSCE	dredgers 10-12m	dredge on scallops	Spatial effort	Logistic	10.0	2.3	SCE
FC2	fl26otbMUL	dredgers 10-12m	bottom trawl on demersal fish	Spatial effort	Logistic	16.2	0.48	PLE, CEP
FC3	fl26tbbSOL	dredgers 10-12m	beam trawl on demersal fish	Spatial effort	Logistic	16.5	0.9	SOL, PLE, OFF
FC4	fl26netSOL	dredgers 10-12m	trammel nets	Spatial effort	Normal	25.5	5.6	SOL
FC5	fl26othLBT	dredgers 10-12m	others	Spatial effort	Logistic	45.0	0.26	PLE, MAC
FC6	fl27dredSCE	dredgers 12-18m	dredge on scallops	Spatial effort	Logistic	9.0	2.3	SCE
FC7	fl27otbCOD	dredgers 12-18m	bottom trawl on demersal fish	Spatial effort	Logistic	17.5	0.3	CEP, MAC, RAY
FC8	fl27otbCEP	dredgers 12-18m	bottom trawl on cephalopods	Spatial effort	Logistic	10.0	0.4	CEP, RAY
FC9	fl27tbbRAY	dredgers 12-18m	beam trawl on demersal fish	Spatial effort	Logistic	8.5	1.2	RAY, SOL, PLE
FC10	fl27midwcCLU	dredgers 12-18m	mid water otter trawl on pelagic fish	Spatial effort	Logistic	15.5	0.3	MAC, CLU
FC11	fl27netSOL	dredgers 12-18m	trammel nets	Spatial effort	Normal	25.5	5.6	SOL, PLE, OFF
FC12	fl27othWHE	dredgers 12-18m	others	Spatial effort	Logistic	16.5	0.5	PLE, MAC, CRA
FC13	fl43netSOL	Passive gears <10m	trammel nets	Spatial effort	Normal	25.5	5.6	SOL, PLE, RAY
FC14	fl43othWHE	Passive gears <10m	others	Spatial effort	Logistic	23.5	2.3	CEP, BSS, WHE
FC15	fl44dredSCE	Passive gears 10-12m	dredge on scallops	Spatial effort	Logistic	8.5	2.3	SCE
FC16	fl44netSOL	Passive gears 10-12m	trammel nets	Spatial effort	Normal	28.0	3.5	SOL, PLE, RAY
FC17	fl44othWHE	Passive gears 10-12m	others	Spatial effort	Logistic	35.0	0.12	RAY, MAC, WHE
FC18	fl49dredSCE	Trammel netters 12-18m	dredge on scallops	Spatial effort	Logistic	8.8	2.0	SCE
FC19	fl49netSOL	Trammel netters 12-18m	trammel nets	Spatial effort	Normal	30.0	5.6	SOL, PLE, RAY
FC20	fl49othCEP	Trammel netters 12-18m	others	Spatial effort	Logistic	35.0	0.12	SOL,PLE, COD
FC21	IntOTH	Internationnal fleets		Fishing mortality	none			

4.3.3. MODEL CALIBRATION

After setting the parameter values based on available literature and data, we calibrated the model to the 2002-2011 ecosystem state through four different steps, corresponding with sub-models of increasing complexity.

4.3.3.1. Calibration of a simplified NPZD (Nutrient, Phytoplankton, Zooplankton and Detrital matter) model

During the first stage of the calibration, only the nutrients, organic matter, and planktonic groups were active. We aimed at reproducing the nutrients and plankton dynamics as well as the recycling of organic matter such as they were modelled in ECOMARS3D. Light penetration, sedimentation of nitrogen, hydrodynamic fluxes in the model and parameter related to plankton and bacteria were tuned at this step (Table 4.3).

4.3.3.2. Calibration of the ecosystem model

Once the NPZD calibration was complete, we ran the full ecosystem model (i.e. all the functional groups), without running the fishing module. Fishing mortalities of the exploited groups were accounted for through the natural mortality coefficient. Non-commercial functional groups natural mortality was set to 0.3 per year. This process saw the calibration of growth, natural mortality, diet and recruitment parameters (Table 4.4 and Table 4.5), which were adjusted to achieve outputs with the ecological, physical and chemical processes within an acceptable range and to avoid any erratic behaviour of the model and was only achieved after 100 years of run. The state after 100 years will be referred hereafter as stabilized (no major divergence and biological cycle reproduced) even if the equilibrium is not fully achieved. The final parameter set selected from this tuning process satisfied the following criteria:

(i) individual weights at age of vertebrates were maintained within 20% of their initial values throughout the duration of the run;

(ii) the average biomass of each functional groups were between the minimum and the maximum of biomass observed in literature, assessment or surveys between 2002 and 2011

(iii) the natural (non-predation) mortality was maintained at low levels as far as possible.

4.3.3.3. Calibration with explicit forcing fishing pressure

Once the biophysical model reflected the average observed biomass, we removed the fishing mortality component of the natural mortality coefficient and instead included fishing activities explicitly through the 21 French fleets targeting sole and the international fleet. Partial fishing mortalities per métier and functional groups were derived from combined 2002 logbook catches, ICES stock abundance numbers and survey indices, and they were applied to represent the impact of the international fishing activities on the ecosystem. Daily spatialized time series of catch per functional groups and polygons were used to force catches of the focused fleets; these were calculated from logbook data averaged over 2002-2011.

Biological parameters and the international fishing mortality rate (Table 4.3 and Table 4.5) were tuned to adjust vertebrate biomasses in the range of assessment or survey estimation over 2002-2011 and catches of the international fleet similar to the observed catches (http://www.ices.dk/marine-data/dataset-collections/Pages/Fish-catch-and-stock-assessment.aspx) over the same period. Only the international fleet's fishing mortality was adjusted in this way as it was the most uncertain.

4.3.3.4. Calibration with spatially resolved fishing effort

In this last step, series of fishing effort in days at sea were applied for each of the French sole fleets per polygon and quarter instead of the catch time series (Appendix IV-Appendix VII). Fishing effort time series were computed from observed values in logbooks, and averaged over 2002-2011. The initial fleet catchabilities (*q*) (Appendix VIII) were derived from 2002 CPUE (catch per unit of effort) per fleets (*f*) and functional groups (*s*) relatively to the 2002 biomass of each functional group (equation 1):

$$q_{f,s} = \mathrm{CPUE}_{f,s} / \mathrm{Biomass}_s \quad (1)$$

Two different selectivity curves were applied: a normal distribution (equation 2) for netters and a logistic (equation 3) one for the other French sole fleets (Huse et al., 1999, 2000; Madsen et al., 1999; Millar and Fryer, 1999) with $S_{f,sp,i}$ the selectivity of fleet *f* on species *s* in class *i*, $L_{sp,i}$ the average length of species *s* in class *I* and the curve parameters α_f and β_f for fleet *f* (Table 4.6).

$$S_{f,sp,i} = -(L_{sp,i} - \alpha_f)^2 / (2 \times \beta_f^2) \quad (2)$$

$$S_{f,sp,i} = 1 / (1 + \exp(-\beta_f \times (L_{sp,i} - \alpha_f))) \quad (3)$$

Equations (2) and (3) were fitted using the OBSMER data collected between 2002 and 2011 for each sole fleet. Catchability values were adjusted from their initial values such that the catches modelled during the first year of run matched the average catches per species realised for each sole fleet for the period 2002-2011. Then growth rates and recruitment were adjusted to keep functional groups biomass in the range of 2002-2011 observed data (

Table 4.2 and Table 4.5). Finally, catchability and fishing mortality were adjusted for the catch of the fleets remaining out of the acceptable range defined previously.

During this four-step calibration of increasing complexity, we avoided as much as possible changing parameters tuned during the previous step of the calibration. However, it was still inevitable that some adjustements were necessary in order to take into account new interactions and indirect effects. We focused here on calibrating a base scenario that captured the dynamics of the Channel ecosystem between 2002 and 2011. We aimed at:

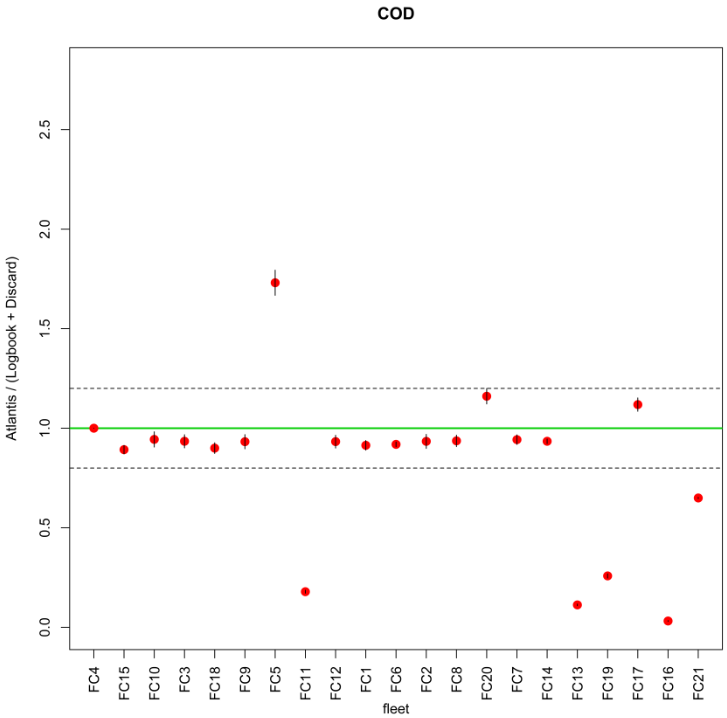
(i) keeping the biomass of vertebrate functional groups biomass between the maximum and the minimum biomass observed in assessment and surveys and also,

(ii) maintaining the catches by fleet within +/- 20% of the observed average for each functional group during that period. We had to run the model for 100 years in each simulation in order to stabilize species size and biomass with a time step of 24h for all groups except for the nutrients, detritus and plankton, where an adaptive time step (which could be as fine as an 1h or less depending on flux dynamics).

4.4. Results

4.4.1. Evaluation of the calibration on catch data

To evaluate the goodness of fit of our model, we compared the catch outputs from Atlantis (landings and discards) with total observed catches (i.e., landings and discards when available) recorded over the period 2002-2011. We only present here, the catches per fleet of the focus groups, shown as the ratio between modeled and observed catch. The standard error around the average Atlantis catch is small, with most of the fleets within the +/- 20% of average observed data that was considered as acceptable (Figure 4.3). The main exceptions to this are that cod caught by trammel-netters (FC11, FC13, FC16 and FC19) and by the international fleet (FC21) are poorly represented while these fleets have an important contribution to the cod catch. By contrast, cod catches are overestimated for the dredgers of length 10-12m operating other métiers. Whiting catches are mainly within the +/- 20% acceptability range. The only fleet that the model failed to represent in relation to whiting is the 10-12 m dredgers using trammel nets. Sole catches are best represented with most of the fleets within the +/- 20% acceptability boundary. Only one fleet had catches 30% above observations: the 12-18 m trammel netters using other métiers. On average, all the fleets consistently catch a slight excess of sole compared to observations. Finally, in the case of plaice, modeled catches of most fleets are in the +/- 20% acceptability range compared to observed average, despite a tendency for underestimation. Fleets using trammel net and the international fleet are out of range. For these fleets which importantly contributed to the plaice catch, the model overestimated observed values. Details on the goodness of fit for other functional groups are provided in Appendix IX.



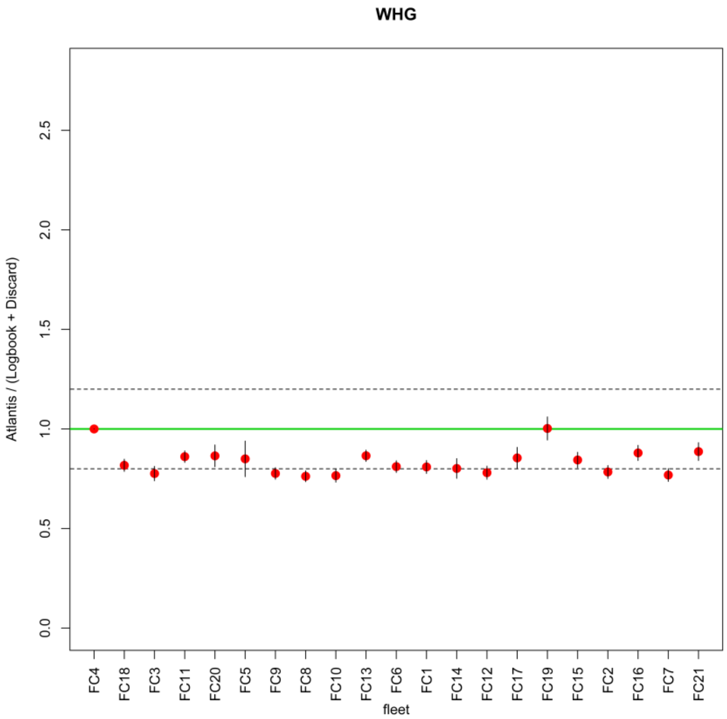


Figure 4.3 Estimation of the performance of the model for sole, plaice, cod and whiting catches output from Atlantis over years 80 to 120. Catches of each combination of fleet and métier forecast by Atlantis are compared to logbook and discard data over 2002-2011. The Y-axis represents the ratio between output and observed data. The X-axis represents the codes of the different combinations of fleets and métiers order incraesingly according to their landing in weigth of the species considered. The green line represents an exact match, and the black dotted lines represent the range of acceptance (i.e. 20% around that level). The red dots represent the ratios between forecast and observations, and the black segment their standard error.

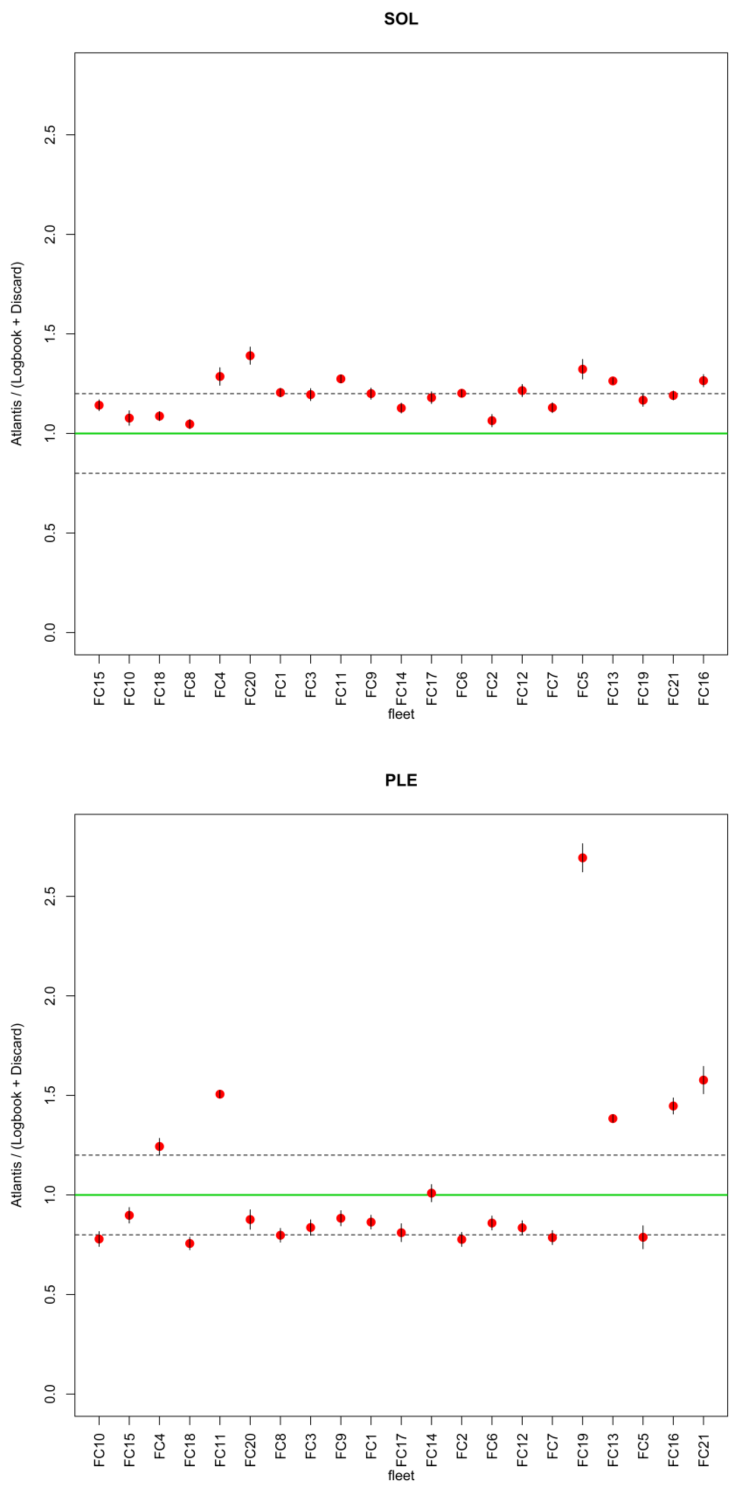


Figure 4.3 (continued)

4.4.2. Average spatial distribution of focus groups species.

Even if, the seasonal distribution of functional groups is forced in Atlantis EEC based on the distribution observed in 2002, the sediment dependency of each groups influence their final distribution. We compared the sole, plaice, cod and whiting spatial distributions of derived from the Atlantis application and from the survey-inferred atlas produced by Carpentier et al. (2009) over the period 1988-2006 in October(Figure 4.4). We show abundance distribution with all ages aggregated in the case of cod and sole. For whiting and plaice, we show the spatial distribution of juveniles (age 0 group) and adults (1+ group) separately.

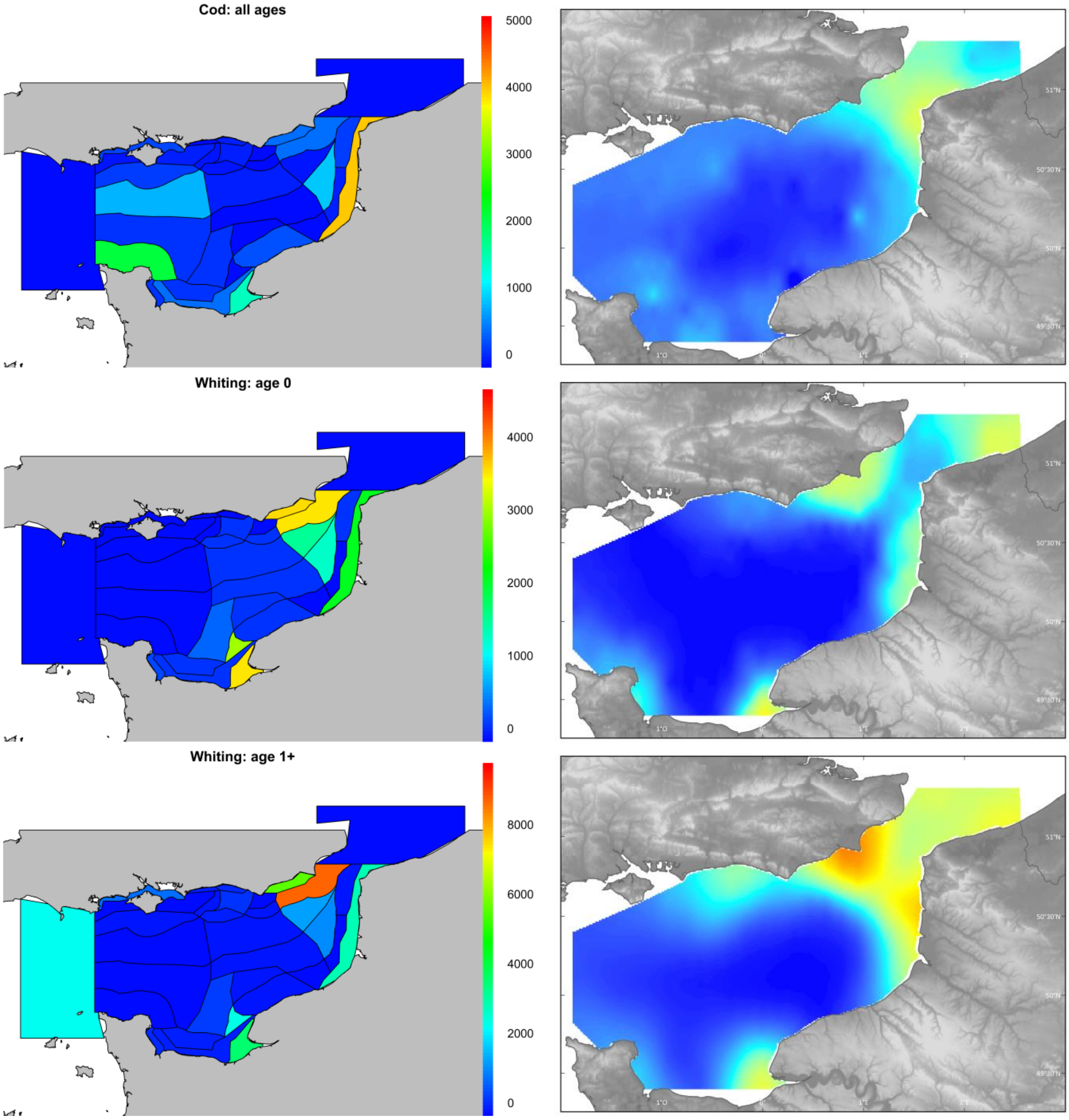


Figure 4.4 Spatial distribution of sole, plaice, cod and whiting biomasses, (left) forecast by Atlantis in the last quarter of the year and, (right) compared to observed distribution during CGFS October survey, over the period 1989 to 2006.

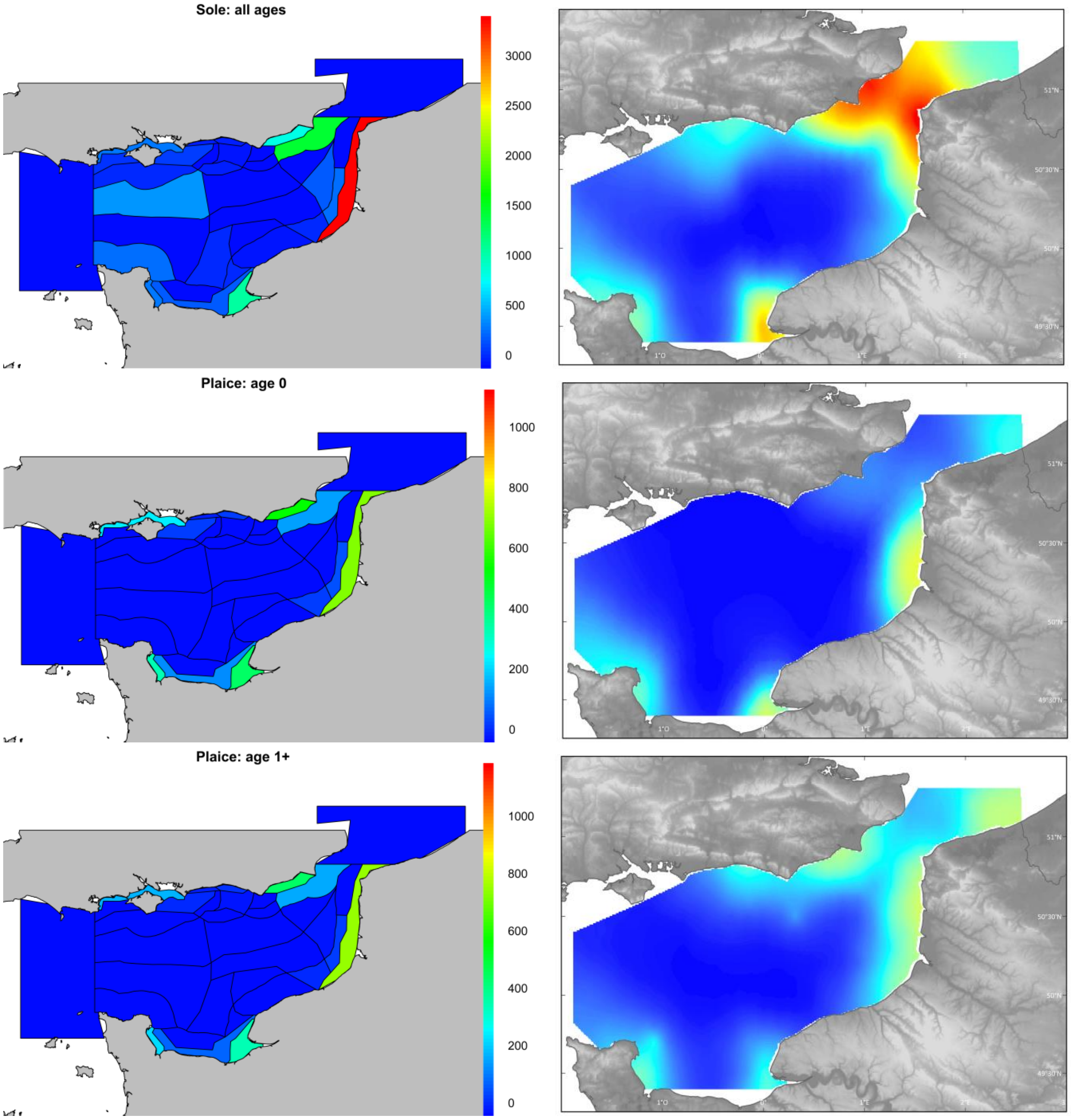


Figure 4.4 (continued)

Cod is distributed in accordance with observations, with a main concentration around the Strait of Dover in the Northeastern part of the Eastern English Channel. Cod, however, is over-represented by the model in the Southwestern part of the Eastern English Channel, which corresponds to Box 1 in Figure 4.1. Consistent with Carpentier et al. (2009), whiting and plaice juveniles are predicted to be distributed along the shore with, for whiting an extension of distribution towards the middle of the Strait of Dover. In the case of the sole, the overall distribution in the Eastern English Channel matches that produced in the atlas. However, the model underestimates the sole offshore distribution in the Strait of Dover.

4.4.3. Comparison of total biomass estimated by Atlantis (calibration run), single-species stock assessment and surveys

We analyzed the average biomass output between the 80th and the 120th years of the calibration run. The calibration period was chosen to allow the system to stabilize, including the biomass but also the structural and reserve weight of vertebrate groups. We then compared the overall Atlantis biomass of any surveyed or assessed vertebrates with the range of biomasses from assessments and CGFS surveys between 2002 and 2011 (Figure 4.5).

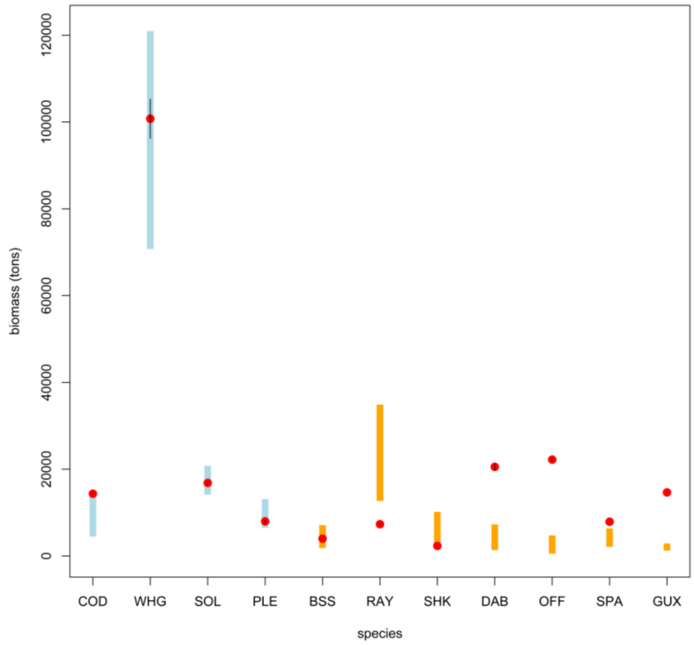


Figure 4.5 Comparison of Atlantis biomasses between years 80 and 120 of the calibration and biomasses assessed or observed during the CGFS survey. The Y-axis represents the biomass in tons and the X-axis represents the different species considered. Red dots represent the average biomasses of each species during the period considered in Atlantis with their standard error (black segment). The blue bars represent the maximum and minimum biomasses assessed by the ICES over 2002-2011 and the orange bars represent the same limits applied to CGFS survey data over the same time period.

The biomass of each functional group is relatively stable, although slightly more variable in the case of whiting. The biomasses of sole and whiting from our model are consistent with stock assessment figures, with an average biomass of 100000 tons for whiting and 17000 tons for sole. The biomass of plaice and cod is comprised between the maximum and minimum of the stock assessment over the period 2002-2011. Cod with an average of 14000 tons estimated by the model, is in the upper bound of the stock assessment estimation and plaice at the opposite is in the lower bound with an average biomass of 8000 tons. Of the other groups that are not subject to an ICES stock assessment, only seabass and shark biomasses are in the range of CGFS observation between 2002 and 2011. Dab, sparidae, gurnards and other flatfishes biomasses are over-estimated in the Atlantis application and rays biomass is underestimated (Figure 4.5). We show the simulated biomass trends of each functional group in Appendix X and Appendix XII.

4.4.4. Analysis of interactions within the trophic network

One of the key components of ecosystem models is the representation of the trophic network structure. We considered both the functional groups' predation mortality and biomass after 100 years of the calibration run to analyse the complex interactions between each of them. We present here only interactions between vertebrates. We show two complementary facets of the Eastern English Channel trophic network architecture in Figure 4.6. The first feature of the trophic network is the proportional contribution per predators to the predation mortality of each prey (Figure 4.6a), which highlights the main predator functional groups and their relative importance in the trophic network. We can observe that whiting, cod, rays, sharks and large bottom fishes are the main contributors to vertebrate mortality. Whiting and cod are the most opportunistic species in our model with more than eight functional groups impacted by those predators; cod is the most important predator of sole and plaice, the two species we focus on in this study, while whiting represents more than 25% of sole predation mortality. In contrast, neither plaice nor sole feed on vertebrate groups. The second feature of the trophic network is the relative importance of prey functional groups in predators' diet (Figure 4.6b). Three main forage functional groups emerge: Clupeidae, mackerel and horse mackerel group, and small demersal fishes. Whiting can be considered as a fourth source of food in the model. Whiting is important both as predator and as prey in the Eastern English Channel ecosystem. Sole and plaice contribute relatively little to the diet of other vertebrate groups and are only noteworthy in the diets of cod, rays and seals. Figure 4.6 also highlights the importance of cannibalism for some functional groups - including cod, whiting, sharks, rays, other flatfishes and large bottom fishes.

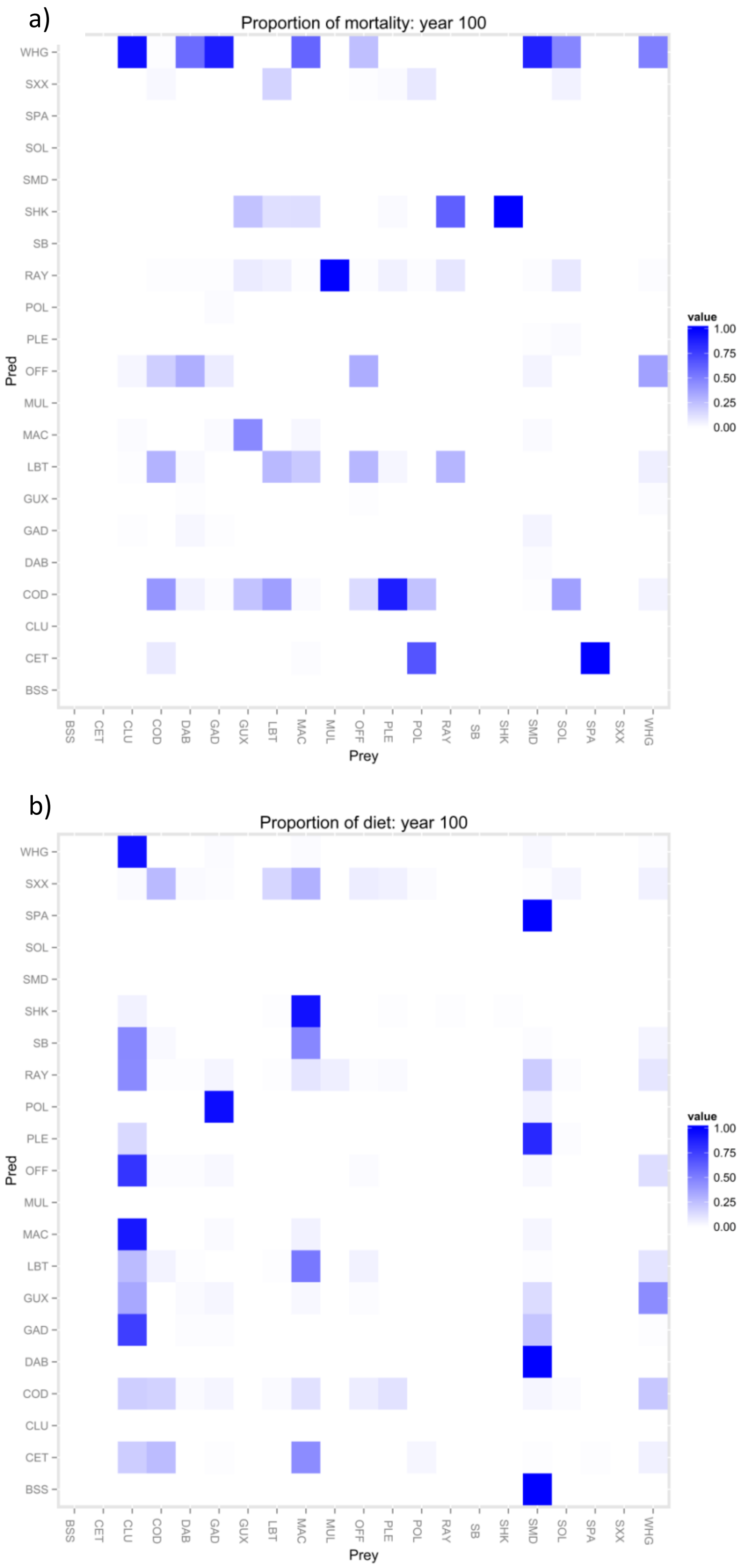


Figure 4.6 Overall structure of the modelled trophic relationships. The rows of this matrix represent the predators and the columns the prey: for a) the proportion of mortality per predation for each prey; and b) the proportion of each prey in the predators' diet. A shade of blue is used to characterize the intensity of each proportion.

Having analysed the global structure of the Eastern English Channel vertebrate trophic network, we now consider the full trophic network around sole and plaice (Figure 4.7), with a representation of predator impact and predation mortality (Figure 4.7a) and the proportion of preys in predators' diet (Figure 4.7b). We ranked the different functional groups by deriving their trophic level from the model calibration outputs. Plaice and sole mainly forage on lower level prey such as deposit feeders, scallops for sole and small demersal fishes and crabs for plaice. Plaice and sole mainly compete for food with crabs. Plaice also competes with small demersal fishes and whiting, but to a lesser extent. Crabs are also the main source of food for whiting and cephalopods. Sole is the only predator of scallops. As to be expected from the general patterns reported above, sole and plaice represent only a small proportion of their predators' diet and are only significant in the diet of cod, rays and large bottom fishes (Figure 4.7a). Three other groups represent a large proportion of the top predators' diet: whiting for rays, large bottom fishes for cod; cod and large bottom fishes for seals. However, when we consider the importance of predators to the predation mortality of sole and plaice, it is clear that cod is the main predator of both species. Rays, whiting and seals also contribute somewhat to sole predation mortality (Figure 4.7b). Regarding the other groups in this sub-network; whiting considerably impacts Clupeidae and small demersal fishes; cod contributes significantly to the mortality of large bottom fishes and whiting; and seals, represent the top predator of this trophic network and impacted mainly cod, large bottom fishes, sole and plaice.

Two important facts emerge from this trophic network analysis focused on sole and plaice. First, cod and whiting represent the main predators in the system, with whiting being both a food competitor for sole and plaice and also a prey for several vertebrate groups. Conversely, sole and plaice mainly impact invertebrates and are poorly represented in vertebrate diets, with cod and whiting being their main source of predation mortality. Therefore, particular attention will be paid to subsequent results obtained for sole and plaice, the main species of interest for this study, but also cod and whiting.

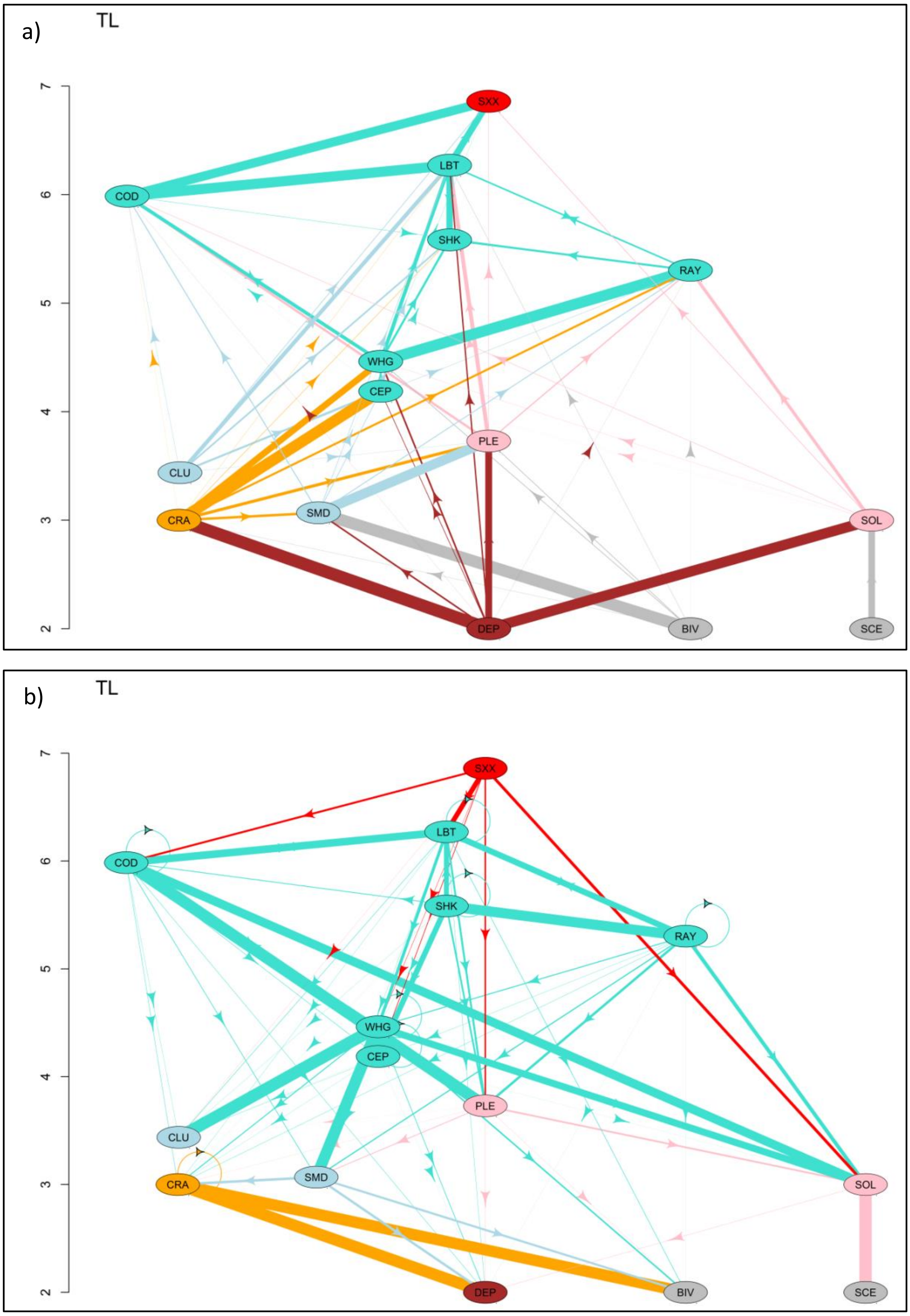


Figure 4.7 Trophic networks around sole and plaice, for a) the proportion of the prey in the predator's diet; and b) the proportion of predation mortality due to a predator. The recalculated trophic level is represented on the Y-axis. The thickness of arrows represents the intensity of the trophic relation between two species.

4.4.5. Importance of fishing mortality on the target groups.

We compared the three types of mortality assumed in Atlantis: the predation mortality, the natural mortality other than predation, and the fishing mortality (Figure 4.8). For plaice, sole and cod the mortality per predation does not exceed 0.03 and represents a small part of the total mortality compared to the fishing mortality, which is around 0.2 for sole and cod, and 0.7 for plaice. Whiting is the only group that is more impacted by predation mortality than by fishing mortality with less than 0.2 for fishing mortality and 0.27 for predation mortality. We then compared the Atlantis fishing mortalities with those derived from ICES single- and multi-species stock assessments. Our model estimates a smaller cod, whiting and sole fishing mortality and a larger plaice fishing mortality compared to single-species stock assessments. We observed, however, that our outputs are closest to the fishing mortality resulting from the multispecies assessment for cod and whiting. Mortalities of vertebrates as estimated by Atlantis EEC are detailed in Appendix XI.

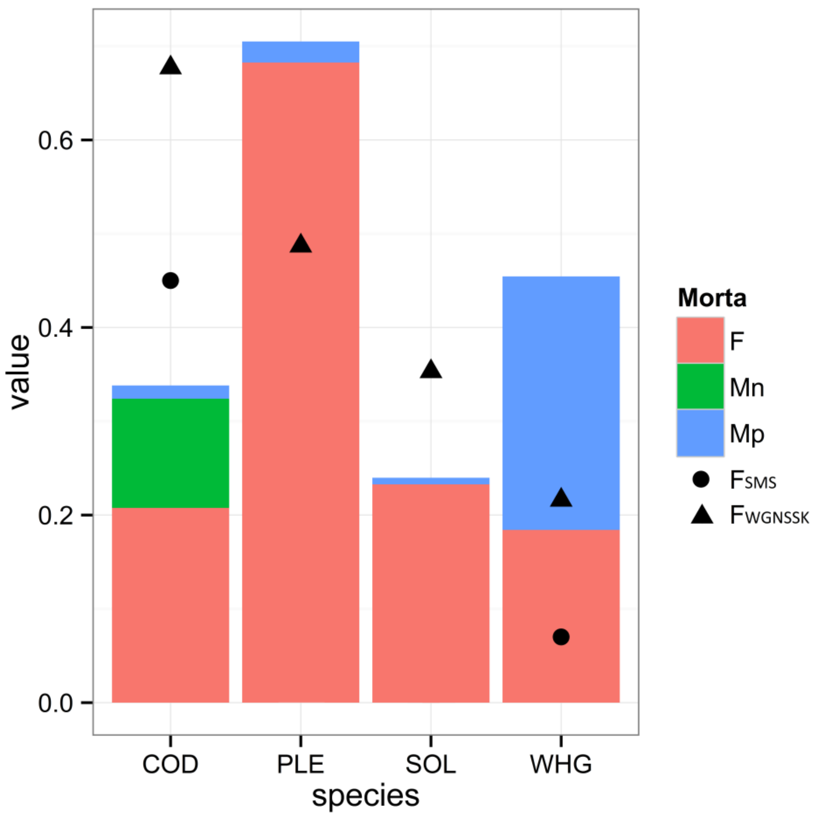


Figure 4.8 Sole, plaice, cod and whiting mortalities output from Atlantis between the year 80 and 120 of calibration: the red bar represents fishing mortality (F), the blue bar the predation mortality (Mp), the green bar the natural (other than predation or fishing) mortality (Mn). Fishing mortalities derived from ICES single-species and multispecies stock assessments are represented as black triangles and black dots, respectively.

4.5. Discussion

Interest in ecosystem-based management has increased during the past four decades. This process has been supported by considerable improvement of the scientific knowledge around marine ecosystem functioning and increased computing performance, which have favored the development of several end-to-end models building in comprehensive ecosystem dynamics (Fulton et al., 2011b; Plagányi, 2007). Each of these models is based on different assumptions that need to be flagged when interpreting their outcomes. Indeed, the structure of these models and what they aim to represent can have significant impact on ecosystem forecasts (Fulton and Smith, 2004).

4.5.1. Model performance

The model was calibrated by using catches per fleet and per métier. We reproduced the catches of the focused species for most of the fleets. We only failed to reproduce cod and plaice catches for fleets that do not usually target those species. Indeed, for these fleets, fitting the catches proved more challenging due to the paucity of data available to estimate catchability and/or selectivity. This is particularly true for netters, which use trammel nets when targeting sole. In the absence of better information on that gear in the Eastern English Channel, the selectivity of trammel nets was modeled using a normal distribution, based on an analysis of existing at-sea observers catch and length frequency data. In our model, netters mainly targeted fish of the first and second age classes. When we first tried to fit netters cod catches, the increase of catchability led to a sharp decrease of cod juveniles, and eventually a collapse of the cod population after a few years. So we reduced the netters cod catchability and increased cod natural mortality (other than predation) to counterbalance the low adult fishing mortality from netters, the representation of missing predation on juvenile and the likely migration to the North Sea.

The spatial distribution of cod, whiting, sole and plaice was generally consistent with CGFS survey data. However, we showed some difference for sole with an underestimation in the middle of the Eastern English Channel. This can be explained by the poor selectivity for sole of the CGFS gear (large opening bottom trawl) (Carpentier et al., 2009). This lack of spatial overlap might also be explained by the absence of spatial density-dependence in our model, so that spatial distributions were not related to total biomass trends. The calibrated model represents the biomass of assessed species well, with the biomass simulated by the model in the range of the 2002-2011 average stock assessment (ICES, 2013a). We may have observed a difference between Atlantis biomass and CGFS survey (Figure 4.7) because the input biomasses were derived from a previous EwE model (Carpentier et al., 2009). We did not directly use the total biomass derived from CGFS due to the need of applying a correction linked to the selectivity of the gear used (Table 4.6).

Finally, we estimated the mortality resulting from both fishing and predation in our model (Figure 4.8). The comparison with single-species stock assessment results was only indicative, since only fishing mortality is calculated in single-species evaluations, while natural mortality is kept constant over time (ICES, 2013a). It is still instructive that the main source of mortality for cod, sole and plaice is fishing activities, as observed in the assessment (ICES, 2013a). For whiting, however, predation and fishing have an equivalent contribution to the total mortality. In the whiting assessment, fishing mortality reference points are poorly estimated, which could be due to an underestimation of fishing mortality that could be alleviated by applying a multispecifies stock assessment model for this stock (ICES, 2013a; Lewy and Vinther, 2004). Figure 4.8 suggests that cod and whiting fishing mortalities output from both Atlantis and multispecies stock assessment are lower than the fishing mortality derived from single-species stock assessments.

4.5.2. Assumptions made in the model construction

Atlantis is one of the most complete marine ecosystem modelling frameworks available (Plagányi, 2007), due to the wide range of processes that can be included (from bacteria to fisheries). It allows for the simulation of the ecological and economic effects of a variety of environmental change and management scenarios. The downside of this multiple process integration is that Atlantis is exceptionally data intensive to fully implement and typically several components of the models will be poorly constrained due to a lack of data. Consequently, a number of assumptions are required (Link et al., 2010, 2011). This is true of Atlantis-EEC. First, we considered that our functional groups were independent of temperature or salinity, due to the lack of information about temperature and salinity optimum in the area studied and also because evaluating the effects of environmental changes was not part of the scope of this study. Second, we neglected density - dependence and habitat quality when dictating the spatial distribution of species and their movements, again this was due a lack of knowledge on these processes, and also because we did not aim at evaluating the effects of habitat degradation scenarios. Finally for most of our vertebrate functional groups, we applied a Beverton and Holt stock-recruitment relation, which was best documented in the fisheries literature.

4.5.3. The calibration process: what we learned about the ecosystem and the model’s behavior?

The development of this Atlantis application improved our knowledge of the Eastern English Channel ecosystem functioning, and provided further experience of the model’s behavior that could help future applications, consistent with the conclusions of Link et al. (2010).

Atlantis can be implemented with any spatial structure. The definition of the polygons is the first and one of the most important steps in the model development. In our study, we initially began with 38 polygons, which we had to reduce to correct some issues with the biogeochemical dynamics in the model. The nutrients in some of the smallest polygons were being vented too quickly, while other polygons acted as sinks for nutrients due to local eddies occurring near the estuary of the Seine (polygon 20) and in the Bay of Veys (polygon 17) (Figure 4.1). To deal with these issues the geometry was simplified and the over-accumulation of nutrients was corrected by including the river plume during the creation of the hydrodynamics file.

The second step is to choose the structure of the functional groups. Vertebrates can be split into several age classes, with each age class representing a part of the group's life cycle. However, there is a tradeoff between the computational cost of running additional age classes and the fact that the calibration is much harder the longer (in term of years) the age class represents (i.e. its harder to calibrate the biological parameters of groups with long age classes). In our application, the easiest groups to calibrate were those with only one year per age class. Moreover, when we implemented fishing effort and gear selectivity in our model, the assumption of homogeneity within each age class became an issue for some groups. These two difficulties complicated the calibration of biomass, length and catch per class of sole and plaice. For instance, we started with four years per age class for plaice and three for sole but we had to reduce these to two years per age class for both species. This is due to the average size of fish in the first age class being too high otherwise, so all the juveniles were available to fishing too quickly leading to steep biomass decreases.

The choice of the reproduction model also proved decisive during the calibration. We built in reproduction through two processes: (i) a constant number of offspring for mammals, seabirds, rays and sharks and (ii) a stock recruitment relationship for the other vertebrates. In the quasi-absence of very large natural predators (or explicit density dependent controls on the highest trophic levels), the calibration of the top predators' natural mortality was highly uncertain. This proved particularly problematic for rays and sharks which ultimately had to be modelled with reproduction represented as a stock recruitment relationship to stabilize the biomass of that population.

To stabilize catch and biomass to recent average levels for each functional group, we had to modify several parameters simultaneously. The calibration thus allowed us to gain a good understanding of the key parameters driving the modeled Eastern English Channel ecosystem dynamics (Fulton, 2001), focusing first on the growth to achieve a sensible vertebrates' length size, then adjusting the natural mortality if necessary, and finally the reproduction parameters.

4.5.4. Knowledge on the Eastern English Channel ecosystem dynamics

We summarize here the salient features. At the beginning of the calibration we ran the model without explicit river inputs. We then noticed that in each coastal polygon, the nutrients and the detrital matter decreased quickly, which caused the main benthic invertebrates to collapse and impaired the recruitment of the nursery-dependent functional groups, especially flatfish. These results bear out the crucial importance of estuaries on the nurseries productivity (Kostecki et al., 2010; Le Pape et al., 2013; Riou et al., 2001; Rochette et al., 2010).

It also emerged that the adjustment of available detrital matter and deposit feeders groups was essential to achieve sufficient growth for predators across the entire model. For instance, most of the demersal vertebrates did not grow properly when the benthic groups' availability was too low. This feature had also been revealed in the English Channel EwE application (Carpentier et al., 2009; Daskalov et al., 2011), which had a particular sensitivity of the model output to the mortality of benthic groups. In the Eastern English Channel, most of the species are highly linked to the benthos (Carpentier et al., 2009; Dauvin and Desroy, 2005). The diet of demersal functional groups is mainly composed of benthic invertebrates (Cachera, 2013) and the deposit feeder functional groups represent a key component of our model.

Two key vertebrates groups emerged during the calibration: cod and whiting. Any modification of the biological parameters of these groups impacted the entire ecosystem dynamics. For instance, increasing the biomass of those groups decreased considerably the biomass of the other prey or predator groups. Similar patterns have been evidenced during the calibration of the OSMOSE model Eastern English Channel application (Travers, pers. com.). In contrast, plaice and sole had a limited impact on other vertebrates groups, but were strongly dependent on the productivity of nursery ground and on the availability of benthic invertebrates, which bears out the outcomes of previous studies (Dauvin and Desroy, 2005; Kostecki et al., 2010; Le Pape et al., 2013; Riou et al., 2001; Rochette et al., 2010).

During the calibration process, we first included fishing activities without discards. When discards were introduced in the model, we noted an increase of the benthic biomass, resulting in an increase in the growth and biomass of vertebrates (which then had to be recalibrated to maintain both biomass and average length of these groups within observation range). It highlights the relative importance of discards in the trophic network. As had already been observed for dogfish – where the discarded blue whiting (*Micromesistius poutassou*) impacts the diet of lesser spotted dogfish (*Scyliorhinus canicula*). There is then a dual effect of fishing activities on the ecosystem: a pressure exerted on blue whiting abundance, and a source of food resulting from discards for other ecosystem components, e.g., lesser spotted dogfish (Olaso et al., 1998). In the EU, a discard ban will gradually be implemented as part of the Common Fisheries Policy (EC, 2013). Our results confirm that all aspects of the CFP discard ban need to be considered by decision-makers, the direct impacts on fishing fleets and fisheries, but also the more indirect ecosystem effects.

4.5.5. Perspectives

In addition to the EwE of Daskalov et al. (2011) and our Atlantis application, two other spatially-explicit models are currently in development for the Eastern English Channel. An application of the OSMOSE ecosystem model (Travers-Trolet, 2012) has also recently been developed and integrates the spatial and temporal dynamics of a large range of age- and/or length-structured vertebrate and cephalopod groups, as well as their trophic interactions. However, at present this OSMOSE model does not build in the dynamics of lower trophic level functional groups or fishing activities (Plagányi, 2007; Shin and Cury, 2001, 2004). The other multi-species model of the area is ISIS-fish (Lehuta et al., in press), which focuses on the spatial dynamics of fish species, and makes explicit provision for mixed fisheries dynamics. ISIS-fish is divided into three sub-models, the fishery, the biology and management. Trophic interactions, however, are not explicitly considered in this model (Mahévas and Pelletier, 2004).

A logical future step would then be to compare the outcomes of all the ecosystem models of the region, particularly the spatially explicit Atlantis, OSMOSE, ISIS-Fish models. Such a comparison would provide insights into how robust our understanding of ecosystem dynamics are (i.e. where do the models agree or differ), and would identify the strengths and weaknesses of the various models in relation to the scientific questions they address.

Even before such a comparison, our Eastern English Channel Atlantis application could be used to evaluate the impact of different scenarios including the interactions between fishing activities and other uses (e.g., aggregate extractions, maritime traffic, or wind farms) as well as area-based management. It might also be used to investigate consequences of future climate change and/or changes in riverine runoff.

4.6. Conclusions

The development of the Atlantis Eastern English Channel application improved our knowledge of the functioning of this ecosystem. Even if some of the information was not available, we successfully managed after several steps of calibration to reproduce most of the dynamics of the ecosystem. Two main species, cod and whiting, were highlighted as key components of that ecosystem as well as the benthic invertebrates groups, which provide food for most of the upper trophic levels groups. While sole and plaice were found to be less important in the upper trophic network they are highly dependent on benthos and are important predators for this group.

From the implementation of the fishery in the model, four key observations emerged: (i) the relative prominence of the fishing mortality (compared to other sources of mortality) for cod, plaice and sole,

(ii) the necessity of applying a multispecies approach to assessing the impact of fishing (with the example of the whiting), (iii) the lack of knowledge of gear selectivity and catchability and (iv) the importance of discarding in the trophic network.

This model application, which includes some representation of the entire marine ecosystem, represents our best (current) understanding of the Eastern English Channel and as such may be useful as a decision support tool for future marine management plans, including spatial planning. However to get an appreciation of the robustness and reliability of the model results, the outcomes of the Atlantis application should be compared with OSMOSE and ISIS-Fish (Fulton et al., 2003); ensemble approaches by far being the best means of capturing uncertainty associated with the complexity of ecosystems like the Eastern English Channel.

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5. Chapter 5: Consequences of management measures on fishers' behaviour and marine ecosystems: the example of the Eastern English Channel flatfish fisheries.

Chapter 5: Consequences of management measures on fishers' behaviour and marine ecosystems: the example of the Eastern English Channel flatfish fisheries.

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5.1. Abstract

Understanding the ecosystem reaction to management regulation is key to achieving conservation and sustainability objectives. The implementation of such an ecosystem approach to fisheries (EAF) requires improved knowledge of ecosystem complexity. Over the past few decades, the development of ecosystem modelling has contributed significantly to the improvement of our knowledge on ecosystem functioning, and interactions with human activities. It is now widely recognized used to evaluate management strategies. In our study, we coupled the ecosystem model Atlantis with various fishers' behaviour models, and applied the coupled models to the Eastern English Channel (EEC) ecosystem and fisheries. We evaluated the consequences of implementing a combination of area closures and effort reduction on the EEC ecosystem and on the French netters fleet targeting sole (*Solea solea*). We analysed both the modification of fishers' behaviour and ecosystem functioning after 50 years of management constraint. We observed a noticeable benefit of the application of the combination of MPAs and effort reduction on the biomass of most commercial species, including sole and plaice (*Pleuronectes platessa*), and on the total value landed per unit effort. The response of the ecosystem varied across the metier and the species considered.

Keywords: Ecosystem modelling, Fishers' behaviour, EAFM, management scenarios

5.2. Introduction

Neglecting the plasticity of fishers towards regulations, but also fish distribution, markets, and/or competition with other fisheries or human activities in past management strategies has led to several management failures (Daw and Gray, 2005; Hardin, 1968; Hilborn, 2004; Hilborn et al., 2001). Building on past experiences, fisheries management is gradually moving from a single-species approach towards more holistic ecosystem based management (EBM) (Garcia, 1994; Ludwig, 2002; McAllister and Kirchner, 2002; Rosenberg, 2002). In the EU, this move towards more integrated management is reflected by the inception of the Marine Strategy Framework Directive (MSFD) (EC, 2008). However, a better understanding of marine ecosystem functioning and interaction with human activities is required to implement efficient management strategy plans (Browman and Stergiou, 2004; Fulton et al., 2011a; van Putten et al., 2011; Wilen et al., 2002). Indeed, the reaction of both ecosystems and fishers to management is complex to predict, due to the plasticity of the distribution of resources, environmental fluctuations, socio-economic changes, and the complexity of fishers' behaviour (Burrows et al., 2014; Glaser et al., 2014; Kraak, 2011; Nielsen et al., 2014). Enhancing our knowledge of how fishers react to different management rules is crucial for improving the management decisions process (An and López-Carr, 2012; Prigent et al., 2008; van Putten et al., 2012, 2013; Salas and Gaertner, 2004; Wilen et al., 2002). Over the past few decades, several ecosystem models, explicitly including fishing activities in various forms, have been developed to address this challenge (Fulton, 2010; Fulton et al., 2007; Rose et al., 2010). The integration of both ecological processes and fisheries dynamics in end to end models could enable more effective long-term decision making (investment, disinvestment, multiannual resources exploitation plan, etc.) and integrated marine resource management as mandate by international legislation (Jennings and Lee, 2012; Plagányi et al., 2014).

Within the EBM approach, several management tools have been developed to ensure the conservation of ecosystem health and sustainability of fishery objectives, such as total allowable catch (TAC), fishing effort limitations, technical measures and, more recently, spatial management. Over the last three decades, particular attention has been given to spatial planning through the design and implementation of Marine Protected Areas (MPAs) (Delavenne, 2012). MPAs have been considered as effective tools for EBM (Gewin, 2004; Hall, 1998; Roberts et al., 2002), however, even if efficacy within the protected area has generally been demonstrated (Colléter et al., 2012; Francour et al., 2001; Gell and Roberts, 2003; Halpern and Warner, 2002; Lester et al., 2009; Mosquera et al., 2000; Valls et al., 2012), the benefit for adjacent fisheries and the ecosystem outside the protected area have been more complex to evaluate (Bastardie et al., 2014; Fock et al., 2011; Greenstreet et al., 2009; Hiddink et al., 2006; Mosquera et al., 2000; Murawski et al., 2005; Suuronen et al., 2010; Willis et al., 2003).

Ecosystem modelling has been widely applied to assess the impact of spatial management. Ecopath with Ecosim (EwE) (Christensen and Pauly, 1992; Christensen and Walters, 2004), Ecospace (Walters et al., 1999), and also lately the ecosystem modelling platform Atlantis (Fulton et al., 2005a, 2007) have been applied to investigate the effects of area closures in different marine regions around the world. These models have investigated essentially the impacts on biomass (Valls et al., 2012), trophic networks functioning (Albouy et al., 2010; Colléter et al., 2012; Libralato et al., 2010; Martell et al., 2005; Walters et al., 1999), the impact on the area outside the MPA (Ainsworth et al., 2012; Colléter et al., 2012; Guénette et al., 2014; Kaplan et al., 2012; Martell et al., 2005; Morzaria-Luna et al., 2012; Salomon et al., 2002; Savina et al., 2013; Walters et al., 1999) and also their regional impact on fishers' economic performance, i.e. beyond the MPA boundaries (Ainsworth et al., 2012; Kaplan et al., 2012; Morzaria-Luna et al., 2012). In most of these studies, the application of MPAs alone was not sufficient to reach conservation objectives and, in some cases, could even have a negative effect on both ecosystems and fish stocks overall. For instance, the misplacement of MPAs (Weeratunge et al., 2014), and/or the aggregation of fishing vessels along the edges of MPAs (Bastardie et al., 2014; Fock et al., 2011; Greenstreet et al., 2009; Hiddink et al., 2006; Suuronen et al., 2010) can lead to an increase of fishing impact on sensitive seabed and potentially even to the depletion of specific fish stocks. Most of these studies suggested that implementing MPAs could only deliver conservation benefits if accompanied by more direct constraints on fishing pressure, e.g., by reducing total fishing effort (Ainsworth et al., 2012; Bastardie et al., 2014; Dinmore et al., 2003; Hiddink et al., 2006; Kaplan et al., 2012; Nøstbakken, 2008; Salomon et al., 2002).

Here, we investigated the impact of several management restrictions, combining total effort reductions and area closures, on the ecosystem, fishers' behaviour and their performance in terms of stated ecological and economic objectives. Our case study was the Eastern English Channel (EEC) and we focused more particularly on the French flatfish fishery targeting sole (*Solea solea*), one of the most important species in weight and value landed in the EEC. The complexity of the interactions between ecosystem functioning and fishing activities was captured by coupling an Atlantis ecosystem model with three fleet dynamics models.

Atlantis is a "end to end" modelling framework, that was initially designed as a platform for management strategy evaluation (MSE) (Fulton et al., 2005a). Atlantis represents each important component of the management process (Jones, 2009), including the biophysical system, the human users of the marine resources, the three major components of an adaptive management strategy (monitoring, assessment and management decision processes) and socioeconomic drivers of human behaviour. Atlantis includes dynamic, two-way coupling of all these system components. The model is 3D spatialized and includes explicit physics and biogeochemical dynamics. The use of a biogeochemical framework allows the representation of bottom-up and top-down controls (Fulton et al., 2011b). Here, we applied the EEC Atlantis model that was developed and calibrated in Chapter 4 (Girardin et al., submitted).

Some classic fleet dynamics modelling is already implemented in Atlantis. However, most of these effort allocation options rely on fleet dynamics drivers that are mainly restricted to economics (costs and expected profits) or CPUE distributions. The outcomes of Chapters 2 and 3 suggested that drivers other than economics influence fishers' behaviour, including tradition (Holland and Sutinen, 1999; Tidd et al., 2012; Vermard et al., 2008) and species targeting (Quirijns et al., 2008). While an effort allocation model incorporating these influences has been developed for Atlantis previously (Fulton et al., 2007), it was specifically tailored to the specifics of south-eastern Australia and does not incorporate standardised approaches, such as the well-respected random utility approach. Therefore, the approach pursued here has been to derive two fleet dynamics models from the outcomes of Chapters 2 and 3, which were then coupled with the existing Atlantis EEC model.

To determine the allocation of effort across spatial units and fishing activities (or métiers), we first applied a model derived from the random utility model (RUM) developed in Chapter 2 (Girardin et al., 2015). The RUMs have increasingly been used to mimic fishing effort allocation (Andersen et al., 2012; Holland and Sutinen, 1999; Hutton et al., 2004; Marchal et al., 2009, 2014; Pradhan and Leung, 2004; Tidd et al., 2012; Vermard et al., 2008; Wilen et al., 2002). RUMs are a discrete choice model, based on utility maximization theory (McFadden, 1974) and are well suited to describe choice of finite alternatives such as location, métier or target species. The second model used was a gravity model (Caddy, 1975; Walters and Bonfil, 1999; Walters et al., 1993). Gravity models assume that effort allocation is proportional to the relative attractiveness of the alternatives choices. Gravity models have been included in end-to-end model such as Ecospace (Walters et al., 1999). In this study, we focused on the French flatfish fishery targeting sole, the second most important species landed in value in the EEC. The main métier targeting sole in the EEC is trammel netting. So we focused on the DCF fleet of 12-18m netters that mainly operated this métier.

The objectives of this study are, (i) to implement the coupling between Atlantis and several fleet dynamics models, (ii) to select the fleet dynamics model that is best suited based on fit to past observations and, (iii) to simulate the effects of management scenarios combining effort reductions and area closures on biomasses, ecosystem functioning, and on the behaviour and performance of French netters targeting sole in the EEC.

5.3. Materials and methods

To better understand the interactions between fishers, the ecosystem and their responses to management, a fully integrated modelling approach has been applied (Fulton, 2010; Garcia et al., 2003; Leslie and McLeod, 2007; Pikitch et al., 2004; Plagányi, 2007; Sanchirico et al., 2006). In this study, we focussed on the French DCF fleets (EC, 2008) of 12-18m netters operating in the Eastern English Channel (Table 5.1), i.e. the ICES Division VIId (Figure 1.14). To represent this system we coupled the end-to-end ecosystem model Atlantis (Fulton et al., 2005b, 2007) with several fleet dynamics models. The three fishing classes in Atlantis EEC corresponding to the 12-18m netters fleet operating, trammel netting (FC19), dredging (FC18) and "other métiers" (FC20) (Table 5.1), were implemented dynamically. The remaining fishing classes were implemented with a spatially resolved effort time series and a fishing mortality per species for the international group (Girardin et al., submitted). We present below (i) the different models used, (ii) the data used to couple both ecosystem and fleet dynamics models, (iii) the structure of the resulting models, (iv) how we selected the fleet dynamics model that, when coupled with Atlantis, provided the best fit with available data and finally, (v) the management scenarios being simulated.

Table 5.1 Description of the fishing classes (combinations of DCF fleets and métiers) integrated dynamically in Atlantis EEC

Index	Code	DCF fleets	DCF métiers
FC18	fl49dredSCE	Trammel netters 12-18m	dredge on scallops
FC19	fl49netSOL	Trammel netters 12-18m	trammel nets
FC20	fl49othCEP	Trammel netters 12-18m	others

5.3.1. Description of the models

5.3.1.1. Atlantis EEC: a representation of the Eastern English Channel ecosystem.

The ecosystem model used in this study is the model Atlantis EEC (Figure 5.1), which we developed and calibrated in Chapter 4 (Girardin et al., submitted). We summarize here the main features of this model. Atlantis EEC is structured in 35 spatial polygons. The biogeochemical sub-model is divided into 40 functional groups with 21 vertebrates groups including seven groups of fish species of commercial interest and run with a 24h time step. In the fishery sub-model, 21 fishing classes are made explicit, with each class corresponding to a combination of a DCF fleet and a métier (Table 4.6). We calibrated the model based on French monthly averaged catches per polygon, fishing class and species, for the 20 French fishing classes, and on yearly averaged catches in the EEC for the international fishing class, over the period 2002-2011. The fishery sub-model is implemented as a monthly effort time series for 20 French fishing classes and as fishing mortality per species for the international fishing class. We used catches and effort collected by the French Directorate for Marine Fisheries and Aquaculture (DPMA) from mandatory fishers' logbooks, and stored in the IFREMER Harmonie database. Although the Atlantis platform includes several fleet dynamics models, these did not provide the flexibility to build in the desired fishers' behaviour drivers such as traditions and species targeting, which were highlighted as influential in Chapters 2 and 3. In this study, we considered three fleet dynamics that considered economic drivers, but also traditions, species targeting and competition with other users, and these are described in Section 5.3.1.2.

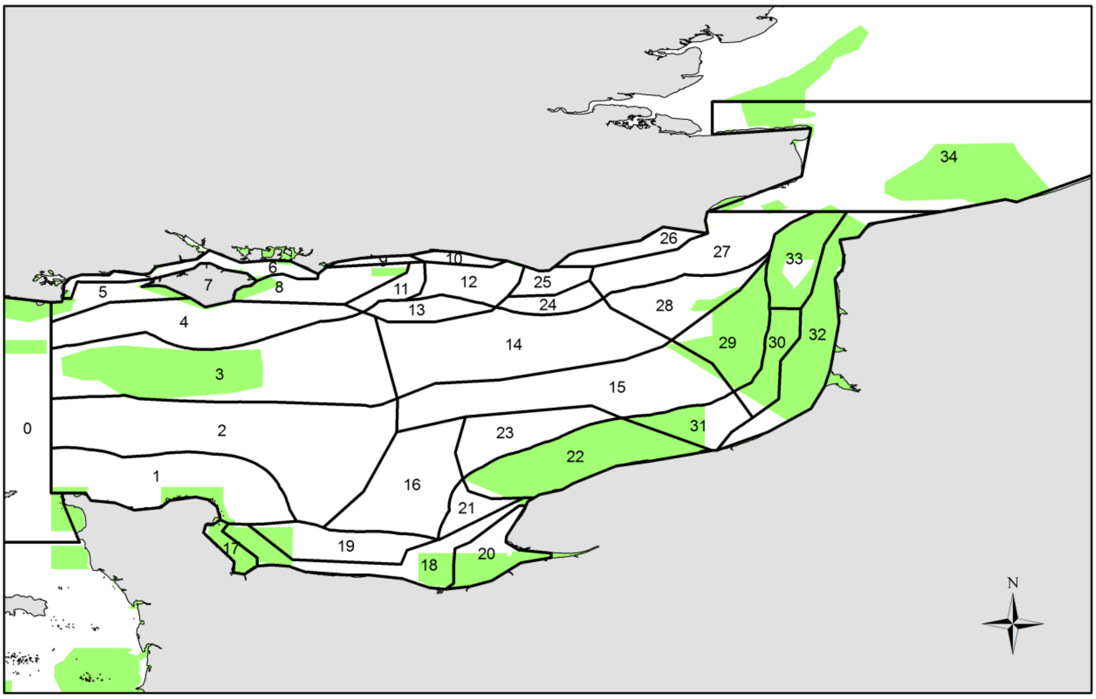

Figure 5.1 MPA distribution in the EEC and Atlantis EEC polygons

5.3.1.2. Fishers' behaviour models

The first fleet dynamics model we applied is derived from the RUM analysis conducted in Chapter 2 (Girardin et al., 2015). Each choice alternative in this model is discretised as a combination of métier f and statistical rectangle *i* (Figure 5.1), and the model is run monthly. As in Atlantis EEC, the netters fleets operated three métiers. Eight factors had a significant effect ($p < 0.05$) on fishers' behaviour (Table 5.2).

The Value Per Unit Effort per métier and location $VPUE_{i,f}$ is defined as the expected returns from choosing the alternative *i,f* is approximated by the previous year's VPUE realised during the same month. Fishers' tradition is mimicked by $EFF_{i,f}$ which represents the past monthly average effort allocated to that alternative in the previous year. Targeting was included through the monthly average proportion of CPUE per species *s* realised during the previous month ($PROP\_CPUE_{s,i,f}$). The species considered were sole, seabass (*Dicentrarchus labrax*), Atlantic cod (*Gadus morhua*), scallop (*Pecten maximus*) and other species gathered. Most of the large-scale maritime traffic in the EEC occurs through a corridor referred to as the extended Vessel Separation System (VSS). The pressure of shipping on fishers' behaviour $SURF\_AREA\_OCCUP_i$ was considered and represented by the monthly average overlap between the extended VSS and the fishing ground. Each alternative choice was represented by a utility function $U_{i,f}$ (equation 1)):

$$U_{i,f} \sim VPUE_{i,f} + EFF_{i,f} + \Sigma_s PROP\_CPUE_{s,i,f} + SURF\_AREA\_OCCUP_i \quad (1)$$

Table 5.2 Description of fishers' behavior indicators used to describe 12-18m netters French fleet.

Fishers' behavior indicators	Description
VPUE	Return expected by choosing a given métier, based on value per unit effort experienced the last year in the same month with this métier
Effort	Habit of vessel, reflected by last year in the same month effort allocation by métiers
POURC_CPUE_SOL	Proportion of Sole in the landing of Atlantis fishing groups the month before
POURC_CPUE_BSS	Proportion of Seabass in the landing of Atlantis fishing groups the month before
POURC_CPUE_COD	Proportion of Cod in the landing of Atlantis fishing groups the month before
POURC_CPUE_SCE	Proportion of Scallops in the landing of Atlantis fishing groups the month before
POURC_CPUE_OTH	Proportion of other species in the landing of Atlantis fishing groups the month before
Shipping	Spatial constraint exerted by maritime traffic, estimated by the proportion of each statistical rectangle overlapped by the extended vessel separation system

The coefficients estimated for the utility function (Table 5.3) can be used to predict fishers' decision and test various scenarios of management (Holland and Sutinen, 1999; Hutton et al., 2004). The probability of choosing the alternative *i,f* is then computed as (equation 2):

$$P(i,f) = \exp(U_{i,f}) / \Sigma_{i,f}\exp(U_{i,f}) \text{ as } \lambda\ i,f\ P(i,f)\ \pi\ [0;1] \quad (2)$$

The second model applied was a gravity model, that has already been used in past studies to predict effort allocation by areas and metiers (Caddy, 1975; Walters and Bonfil, 1999; Walters et al., 1993). This model considered effort allocation as proportional to the attractiveness of each alternative. In our study, we tested two different models: (i) a model in which effort allocation is proportional to the expected revenue and tradition, called GravVPUE (equation 3), and (ii) another one in which effort allocation is proportional to targeting and tradition, called GravSOL (equation 4). These drivers were approximated using the same proxies as in the RUM application. The target species considered for the 12-18m netters fleet was sole.

$$P(i,f) = \alpha\ VPUE_{i,f} / \Sigma_{i,f} VPUE_{i,f} + \beta\ EFF_{i,f} / \Sigma_{i,f} EFF_{i,f} \quad (3)$$

$$P(i,f) = \alpha\ PROP\_CPUE_{sole,i,f} / \Sigma_{i,f}\ PROP\_CPUE_{sole,i,f} + \beta\ EFF_{i,f} / \Sigma_{i,f} EFF_{i,f} \quad (4)$$

as λ i,f, $P(i,f)$ π [0;1] and $\alpha + \beta = 1$

where α an β are weighting coefficients that represent how influential the drivers are on fishers' decision-making (Table 5.3). To determine those coefficients we used the outcomes of a worldwide review of fleet dynamics (Chapter 3), and more particularly the relative contribution of traditions and species targeting relative to VPUE in the case of passive demersal fleets (Table 5.3) (Girardin and Marchal, in prep).

Table 5.3 Coefficient of each fishers' behavior indicator in each fleets dynamics model and goodness of fit of each models. RUM is the random utility model, GravVPUE is the gravity model applied with past VPUE and effort, and GravSOL is the gravity model applied with proportion of Sole in past catches and past effort.

Fishers' behavior indicators	RUM	GravVPUE	GravSOL
VPUE	0.0028	0.357	
Effort	0.2591	0.643	0.5344
POURC_CPUE_SOL	0.0340		0.4656
POURC_CPUE_BSS	0.2920		
POURC_CPUE_COD	0.0382		
POURC_CPUE_SCE	0.0165		
POURC_CPUE_OTH	0.0302		
Shipping	0.0060		
Goodness of fit	**7.8180**	**4.4906**	**1.3631**

5.3.2. Forcing data and assumptions needed for the coupling

First, Atlantis EEC and fleets dynamics models are spatially resolved using different structures, irregular polygons and statistical rectangles, respectively (Figure 1.14 andFigure 5.1). To couple both models, transfer matrices were calculated, one to map from polygon to statistical rectangle (where each cell was the proportion of overlap of each polygon for each statistical rectangle) and the other mapping statistical rectangles to polygons. The assumption made was that catches and effort were evenly distributed within a polygon and within a statistical rectangle. Second, Atlantis and the fleet dynamics models used daily and monthly time steps, respectively. We then assumed that, within each month, the daily effort applied in Atlantis was constant for each fishing group in each polygon, and corresponded to a fraction of the effort allocated by the fleet dynamics models. Finally, fleet dynamics models provided the proportion of fishing effort allocated to the different metiers and statistical rectangle. We assumed that the total monthly effort in the Eastern English Channel remained constant from one year to another, and that was calculated as the daily averaged effort per month allocated in VIId by the 12-18m French netters fleet during the 2002-2011 period.

Atlantis EEC stores the values of catch and effort of each fishing class, in Division VIId. However, information on catch and effort outside Division VIId is also needed to input fleet dynamics models. Catch and effort allocated by the 12-18m netters outside Division VII were calculated as the daily observed values averaged per month over the period 2002-2011. Species' prices were not explicitly modelled in our application. They were provided to the model as time series for the 12-18m netters. To that purpose we used landing data by value and weight, from combined sales slips and logbooks, over the period 2002-2011. Species prices were calculated as monthly averaged values over this period. Although prices were assumed constant from one year to another, seasonal patterns were made explicit using this procedure. Data on shipping pressure $SURF\_AREA\_OCCUP_i$ were also forced, considered constant in the model, and derived from the averaged per month overlap between statistical rectangle and extended VSS (Girardin et al., 2015). Finally, to prevent VPUE (the ratio between landing value and fishing effort) from reaching high values exceeding computational limits, we applied a minimum meaningful effort threshold of 30min spent per month and per alternative *i,f*. Any effort lower than 30min was considered non-significant and set to zero. We also assumed that if less than 30min were allocated to an alternative, that alternative was unattractive, so we set the VPUE and the species-specific CPUEs to zero as well.

5.3.3. Implementation of the coupling

The first challenge in implementing the coupling was to interface the Atlantis code in C++ with the fleets dynamics models implemented in R (R Core Team, 2012). The implementation was performed by calling the R freeware directly within the Atlantis C++ code. Ecological and catch processes were performed in Atlantis, while the spatial transfer matrix converting polygons to statistical rectangles and vice versa, as well as the forecast effort allocation from fleet dynamics models were performed in R. Atlantis EEC required 100 years in each simulation to stabilize species size and biomass for each functional groups, c.f. Chapter 4 (Girardin et al., submitted). After 100 years, effort allocation for the Atlantis fishing groups corresponding to the 12-18m French netters fleet was performed dynamically by starting the coupling between Atlantis and fleet dynamics models (Figure 5.2).

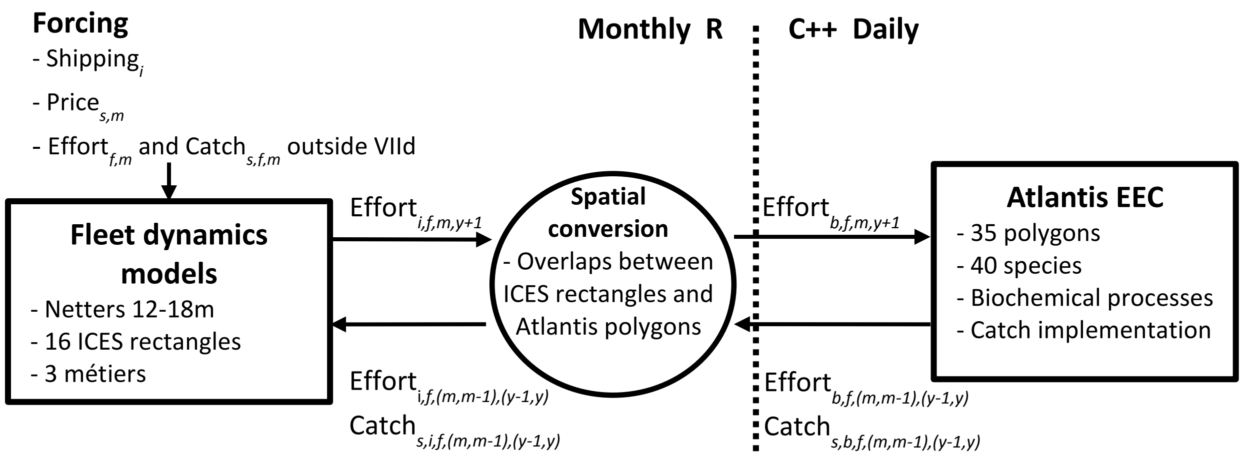

Figure 5.2 Simplified description of the coupling between Atlantis EEC and fleet dynamics model, with species *s*, polygons *b*, statistical rectangles *i*, fishing groups *f*, month *m* and year *y*.

Atlantis stores the spatially resolved cumulative effort and catches per species and fishing classes for each month during a year. At the beginning of each month *m*, before running the Atlantis fishery sub model, fleet dynamics models provide the spatial distribution of fishing effort per métier for month *m+1*. The effort and catch per polygon output from Atlantis were converted and transferred into statistical rectangles and the fishers' behaviour drivers proxies were computed (R code), also using the forcing data presented previously. Once the probability of choosing each alternative *i,f* were calculated (R code), these were multiplied with the netters' total fishing effort , resulting in daily averaged effort values per fishing class and per statistical rectangle. The predicted effort allocation was then converted from statistical rectangles into Atlantis polygons using the transfer matrix and input in Atlantis EEC. The Atlantis fishery and ecological sub-models were executed. The implementation of the coupling is detailed below (Figure 5.3).

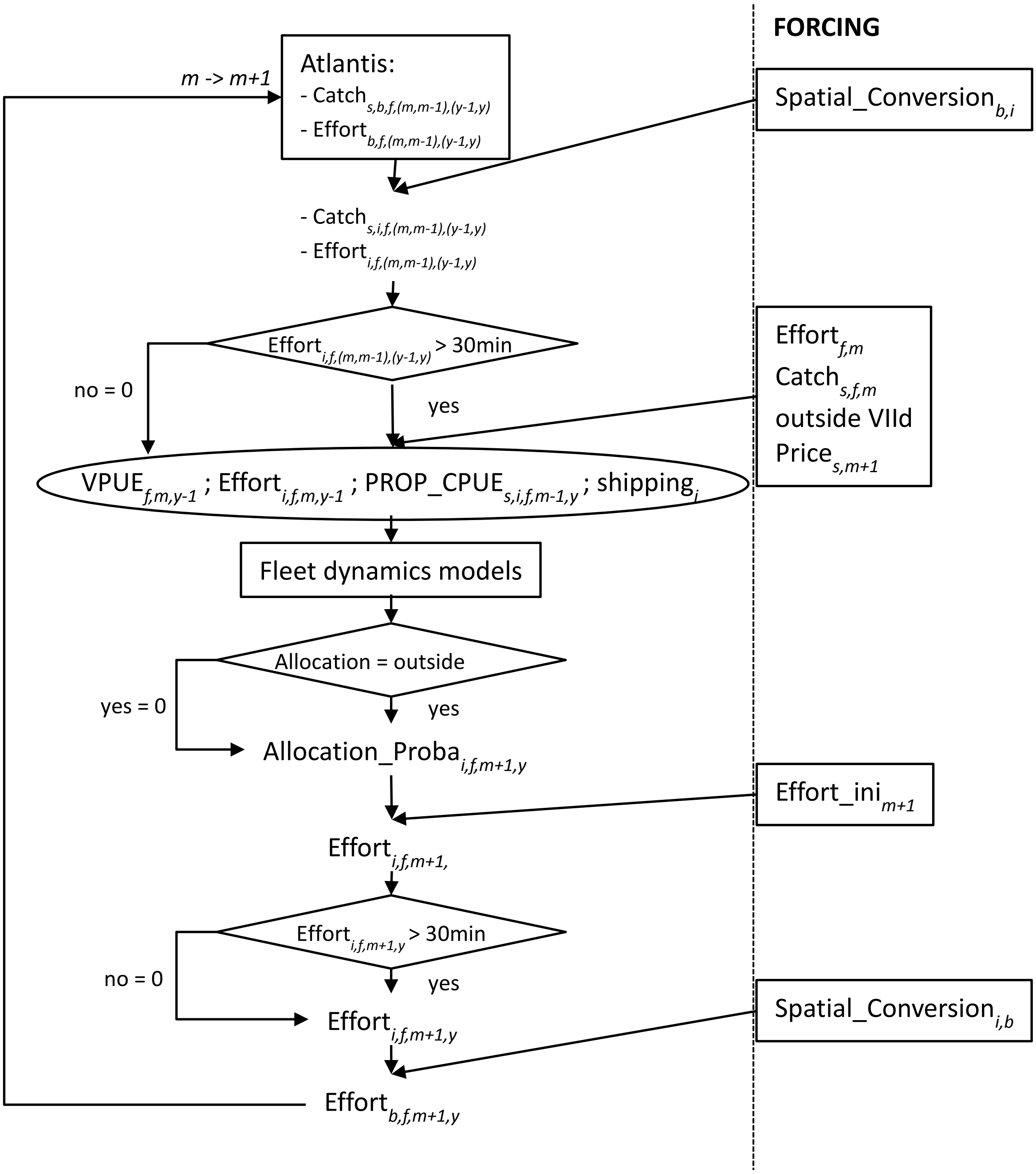

Figure 5.3 Detailed description of the coupling between Atlantis EEC and fleet dynamics models, with species s, polygons b, statistical rectangles i, fishing groups f, month m and year y.

Every month (*m*) catches per species (*s*), polygons (*b*), and fishing classes (*f*) ($Catch_{s,b,f,m,y-1}$), as well as fishing effort per polygons and fishing classes ($Effort_{b,f,m,y-1}$) from the previous year (*y-1*) and the same month, were reallocated from polygons to statistical rectangles (*i*) ($Catch_{s,i,f,m,y-1}$; $Effort_{i,f,m,y-1}$) with a spatial conversion matrix ($Spatial\_Conversion_{b,i}$). Before computing the fishers' behaviour proxies, $Effort_{i,f,m,y-1}$ were set to zero if under 30min. Then, if the fishing effort used to calculate these metrics was not null, VPUE per fishing class during the same month of the previous year ($VPUE_{f,m,y-1}$), and the CPUE proportion of species of interest in the previous month (PROP_CPUE $_{s,i,f,m-1,y}$) were calculated. When fishing effort was null, it was assumed that alternative was unattractive, and both VPUE$_{f,m,y-1}$ and PROP_CPUE$_{s,i,f,m-1,y}$ were set to zero. As in Girardin et al. (2015), the expected revenue was approximated by VPUE$_{f,m,y-1}$ and calculated using equations (5) and (6):

$$\text{CPUE}_{s,i,f,m,y-1} = \text{Catch}_{s,i,f,m,y-1} / \text{Effort}_{i,f,m,y-1} \qquad (5)$$

$$\text{VPUE}_{f,m,y-1} = \Sigma_s\ \text{CPUE}_{s,i,f,m,y-1} * \text{Price}_{s,m,y} \qquad (6)$$

where Price$_{s,m,y}$ is the current fish price input as a forced time series. CPUE and VPUE proxies outside Division VIId were necessary to run the fleet dynamics models, and these were estimated from forced time series of effort and catches (Catch_out$_{s,f,m}$; Effort_out$_{f,m}$). Shipping spatial pressures were derived from forced time series as well (SURF_AREA_OCCUP$_{i,m}$).

The probabilities of choosing each alternative *i,f* were then forecast using the fleets dynamics models (equations 1-4). We constrained the distribution of effort inside Division VIId by removing the alternative choice to go outside and by rescaling the probabilities (Allocation_Proba$_{i,f,m}$). These probabilities were then multiplied by the forced time series of daily averaged total netters' fishing effort per month to obtain the daily effort allocated to each alternative in month *m* and in year *y* (Effort$_{i,f,m}$). The same threshold of 30min was applied to the forecast effort and then the resulting effort distribution was normalised to sum to one. Finally, the forecast effort was reallocated to Atlantis EEC polygons by using a transfer matrix (SpatialConversion$_{i,b}$). Then Effort$_{b,f,m}$ was input in Atlantis EEC and processed during the following month to estimate catches.

In Atlantis, the catches (Catch$_{s,a,f,d,b}$) per species, age classes, fishing classes, polygons and depth layer (*d*) were then implemented as follows:

$$\text{Catch}_{s,a,f,d,b} = \text{FCpressure}_{s,a,f,d,b} * \text{Biom}_{s,a,d,b} \qquad (7)$$

With Biom$_{s,a,d,b}$ the biomass of species *s*, in age class *a*, in polygon *b*, and in depth layer *d*. The fishing pressure FCpressure$_{s,a,f,d,b}$ was defined as:

$$\text{FCpressure}_{s,a,f,d,b} = \text{Effort}_{b,f} * \text{Vertdistrib}_{d,f} * q_{s,f} * (\text{Swept\_area}_f / \text{FCvol}_{f,d,b}) * \text{Sel}_{s,a,f} \qquad (8)$$

where Effort$_{b,f}$ is the fishing effort forecast by the fleet dynamics models in the case of the netters' fleet. Vertdistrib$_{d,f}$ is the proportional vertical fishing effort distribution for the fishing class in each depth layer; $q_{s,f}$ is the catchability per fishing class and per species. Swept_area$_f$ is the swept area per unit time of the fishing gear used. FCvol$_{f,d,b}$ is the volume of water accessible to a fishing class for each depth layer and polygon (this parameter allows for the implementation of partial polygon spatial closures). Finally, $Sel_{s,a,f}$ is the selectivity of each fishing class fishing each vertebrate age class. During the calibration of Atlantis EEC (Girardin et al., in prep), $Vertdistrib_{d,f}$, $Swept\_area_f$, were defined and $q_{s,f}$ was tuned to mimic the fishing classes' catches observed over the period 2002-2011. Considering the 12-18m netters fleet, two selectivity curves were used: (i) a normal distribution for those fishing classes operating trammel nets (equation 9), and (ii) logistic distributions for fishing classes operating dredges and other métiers (equation 10).

$$Sel_{s,a,f} = -(L_{s,a} - \alpha_f)^2 / (2\beta_f^2) \quad (9)$$

$$Sel_{s,a,f} = 1 / (1 + \exp(-\beta_f * (L_{s,a} - \alpha_f))) \quad (10)$$

Where $L_{s,a}$ is the average length size of vertebrates s in the age class a. α_f and β_f are selectivity coefficients that were fitted based on observed data (Girardin et al., submitted).

5.3.4. Evaluation of the three fleet dynamics models.

Three preliminary scenarios were tested to select the fleet dynamics model that best fitted recent observations. After the 100 years of simulation required by Atlantis ECC to stabilize biological processes, the total effort of the 12-18m netters fleet was spatially allocated to the three fishing classes in Atlantis corresponding to that fleet. The coupling was then simulated over 50 years. We compared the total effort allocation per métiers through time for each fleet dynamics models with the observed allocation between 2002-2011 periods. Then, we compared the temporal, spatial and métier effort allocation during the final year of simulation obtained with the three fleet dynamics models ($Effort\_prop\_predict_{f,b,t}$) to the averaged effort allocation observed over the period 2002-2011 for the 12-18m netters ($Effort\_prop\_obs_{f,b,t}$). The relatives errors across spatial units, $Error_{f,b}$ (equation 11), and quarters t, $Error_{f,t}$ (equation 12), were defined as follows:

$$Error_{f,b} = (\Sigma_t\, Effort\_prop\_predict_{f,b,t} - \Sigma_t\, Effort\_prop\_obs_{f,b,t}) / \Sigma_t\, Effort\_prop\_obs_{f,b,t} \quad (11)$$

$$Error_{f,t} = (\Sigma_b\, Effort\_prop\_predict_{f,b,t} - \Sigma_b\, Effort\_prop\_obs_{f,b,t}) / \Sigma_b\, Effort\_prop\_obs_{f,b,t} \quad (12)$$

where $\Sigma_t\, Effort\_prop\_predict_{f,b,t}$ or when $\Sigma_b\, Effort\_prop\_predict_{f,b,t}$ were no null.

Finally, we estimated the goodness of fit (G) obtained with each fleet dynamics model as:

$$G = \Sigma_{b,t}\, (Effort\_prop\_predict_{f,b,t} - Effort\_prop\_obs_{f,b,t})^2 \quad (13)$$

5.3.5. Evaluation of fisheries restriction management scenarios

Having selected the fleet dynamics model that best fitted available data, we used the coupled model to evaluate four management scenarios: (M1) a status-quo scenario with no constraint, (M2) a network of no-take areas, (M3) a 20% reduction in the current total EEC fishing effort and, (M4) a combination of (M2) and (M4) (Table 5.4).

Table 5.4 Constrained management scenarios

Codes	Scenarios
M1	Status-quo with no management measure
M2	Areas of no-take
M3	20% effort and fishing mortality reduction
M4	Areas of no-take + 20% effort and fishing mortality reduction

To implement the area closure, we used the map of existing and planned MPAs in the EEC (Figure 5.1). These MPAs are not currently constraining the fishing effort allocation in the EEC and, for most of them, they focus on the protection of specific maritime habitats and/or top predators such as mammals and seabirds (Delavenne, 2012). In our case, we applied an extreme scenario where fishing activities were totally banned from MPAs. This constraint was directly implemented in Atlantis EEC for all fishing classes through the parameter $FCvol_{f,d,b}$ (equation (8)). $FCvol_{f,d,b}$ was considered as the proportion of overlap between MPAs area and each Atlantis ECC polygon.

In scenario M3 the 20% effort reduction was applied to all the fishing classes. Non-dynamic fishing classes were constrained by decreasing effort spatial distribution by 20% and for the international fishing class by decreasing the fishing mortality per species by 20%. For the three dynamic fishing classes (Table 5.1), we reduced the forced total effort time series by 20% for each month.

We implemented both the fleet dynamics model selected and the management scenarios after 100 years of simulation, and we applied scenarios for 50 years. We analysed the consequences of each management scenario by comparing the outputs of the three constrained scenarios with those of the status-quo scenario in the last year of simulation.

First, we analysed the change in the total effort allocation through time per métier for each scenario, and the relative difference of effort allocation between constrained scenarios and status-quo scenario over space (equation 11) and over quarters (equation 12) for each métier of the 12-18m netters fleet (corresponding to three fishing classes in Atlantis).

Then we investigated the impact of the four scenarios on the EEC ecosystem. As previously, we compared the relative difference of biomass of each functional group between the three constrained scenarios and the status-quo scenario. An analysis of structural changes in the trophic network was carried out by evaluating, for each prey, the relative change in the proportion of the predation due to each predator and also the relative change of prey as a proportion of the diet of each predator (for vertebrate functional groups only). We explored as well the relative change of total fishing mortality for each functional group across the different scenarios.

Finally, we investigated the consequences of the different scenarios in term of the fishing performance of the 12-18m netters fleet per metiers, by comparing the resulting main species' CPUE and also the total VPUE.

5.4. Results

5.4.1. Evaluation of three fleet dynamics models.

First, we compared the fishing effort allocated to the three metiers operated by the 12-18m netters fleet derived from the three fleet dynamics model simulations to observed metiers allocation over the period 2002-2011 (Figure 5.4). When coupled with Atlantis ECC, the RUM allocated nearly the entire fleet's effort to the trammel nets métier. The GravVPUE model favoured the dredgers métier and reduced the effort allocated to the trammel nets (by nearly 40%) and to the other métiers (by nearly 70%). Finally, the GravSOL decreased dramatically the effort allocated to dredgers and reduced slightly the effort allocated to trammel netters and increased the allocation of effort to the other metiers. The change of allocation occurs more quickly in the RUM taking place in two years, compared to nearly 5 years for the GravSOL model and 20 years for the GravVPUE model.

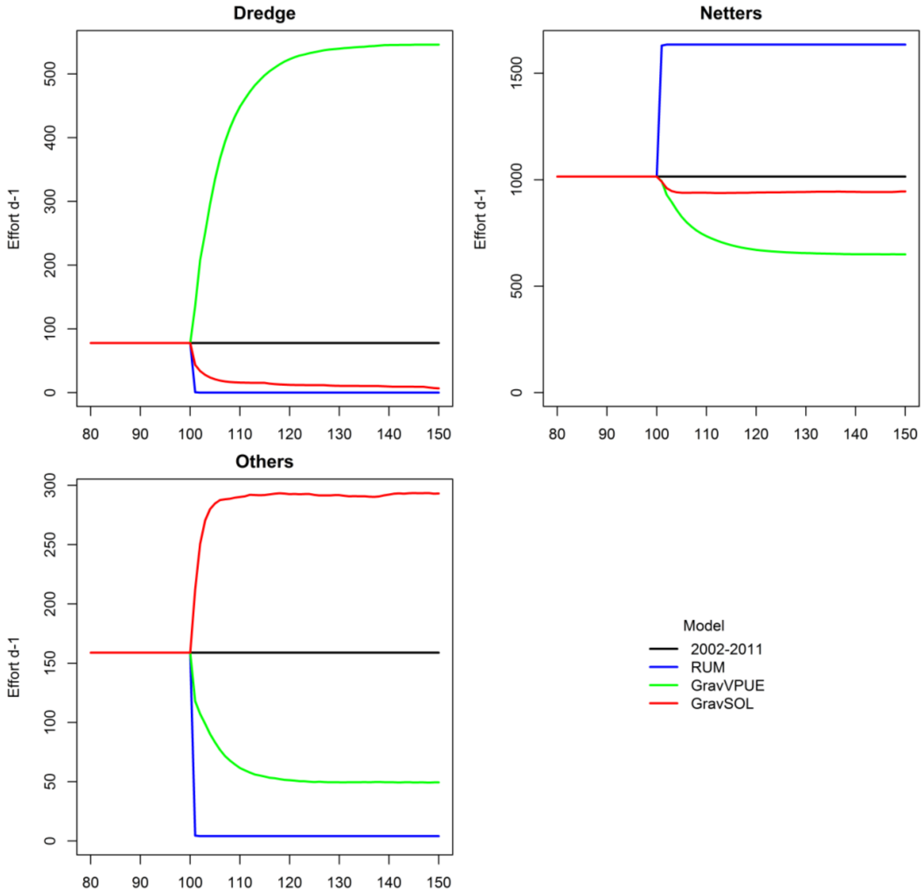

Figure 5.4 Total effort allocation per métier with three fleet dynamics models coupled to Atlantis EEC. The fleet dynamics models are active after 100 years of Atlantis running with constant monthly and spatially-resolved fishing effort. In black, the average effort realized between 2002-2011 for each métier; in blue, effort allocation derived from the RUM-based model; in green, effort allocation derived from the gravity model using past VPUE and effort; in red, effort allocation derived from the gravity model using past proportion of sole in catches and past effort.

Figure 5.5 Simulated effort allocation in space and per métier after 50 years of running the ecosystem-fleet model compared to observed 2002-2011 effort allocation. We represent the standard error between each fleet dynamics output and the effort allocation realized over the period considered per métier and Atlantis polygons. + corresponds to an overestimation by the fleet dynamics model when no effort was allocated during the period considered. RUM are the output from the RUM-based model, Gsol the output from the gravity model with the proportion of sole, and Gvpue the output from the gravity model with the total VPUE.

In Figure 5.5, we observe the same métier shift as in Figure 5.4, as well as variations in the spatial allocation of fishing effort. After 50 years of running the fleet dynamics models coupled with Atlantis EEC, we observe that the dredgers effort decreases evenly through space for the RUM and the GravSOL models, while the GravVPUE reallocated the dredgers métier to the middle of the EEC (Figure 5.1 andFigure 5.5). The decrease of trammel nets allocation in the GravSOL model was evenly distributed with a slightly more important decrease in the western part of the ECC and in the French side of the Dover Strait. In contrast, RUM and GravVPUE models reallocated a large fraction of the trammel nets effort to the western part of the EEC and, for the RUM, also to the Bay of Seine and the middle of the EEC. Finally, shifts in the effort allocated to the other métiers is evenly distributed with the RUM model, while GravVPUE increased the allocation in the western part of the EEC and the GravSOL model increased evenly the effort allocation to other métiers with a slight preference for the Bay of Seine, the middle of the EEC and the French side of the Dover Strait.

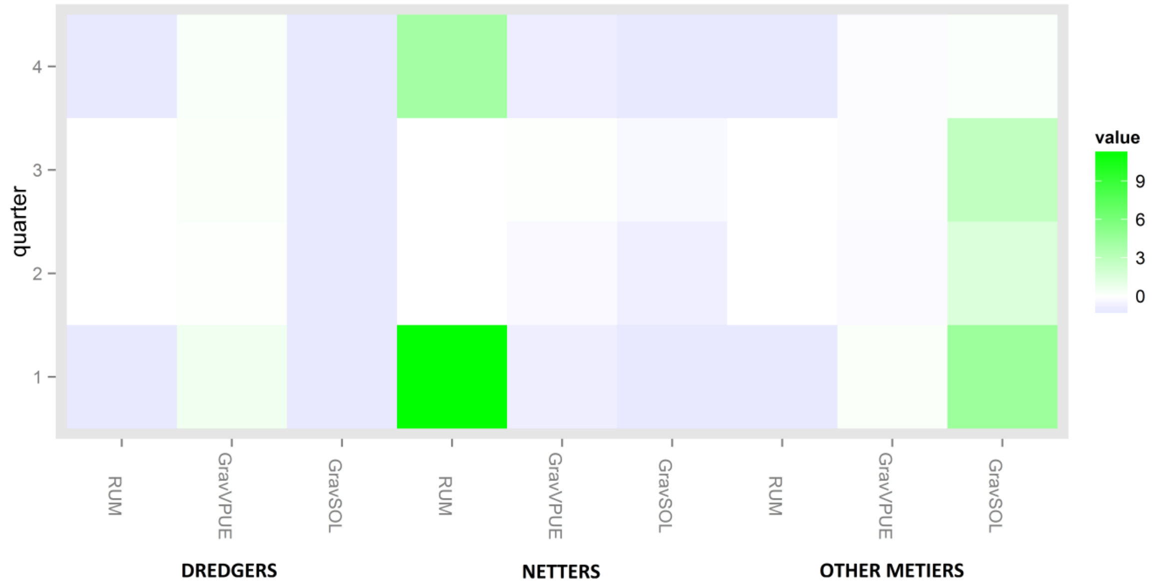

Figure 5.6 Simulated effort allocation over time, per quarter and per métier, after 50 years of running the ecosystem-fleet model compared to effort allocation between 2002 and 2011. The codes are those used in Figure 5.5.

The analysis of the effort reallocation over quarters (Figure 5.6) suggested that the RUM model mainly reduced the allocation to dredgers métier during the first and last quarters, GravVPUE model increased the allocation during the first quarter and GravSOL model reduced evenly the allocation to dredgers. In contrast, for the netters métier, the RUM increased effort allocation during the first and last quarters, GravVPUE and GravSOL mainly decreased the allocation during the first and last quarters. At last, the effort allocated to other metiers remained unchanged through time with the GravVPUE model; the RUM model decreased the allocation during the first and last quarters and the GravSOL model increased the allocation to this métier during the first three quarters.

We compared the goodness of fit obtained by coupling Atlantis EEC with the three different fleet dynamics models (Table 5.3). The best fit for the GravSOL model (G = 1.36), followed by the GravVPUE model (G = 4.49) and the RUM-based model (G = 7.82).

Based on these preliminary results, we then applied the GravSOL fleet dynamics model in subsequent management scenarios evaluations.

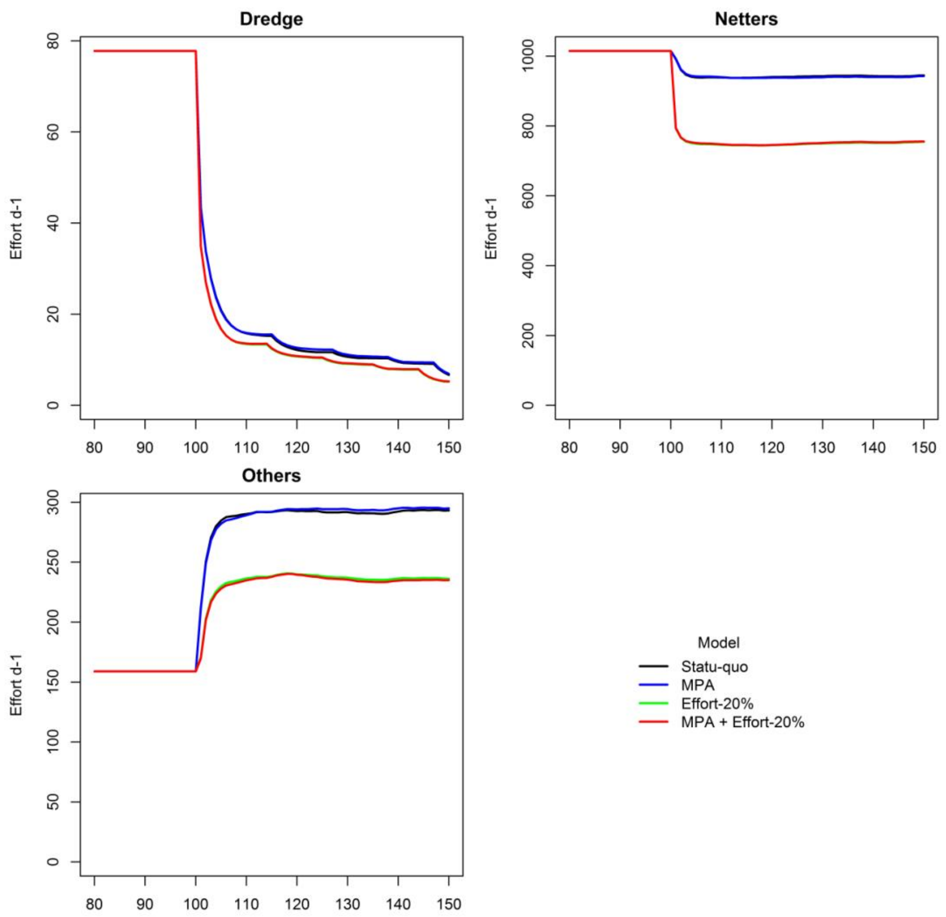

Figure 5.7 Total effort allocation per métier for the three management scenarios tested. In black, the status-quo scenario; in blue, the scenario with areas closed, in green the 20% effort reduction scenario; in red, the scenario combining areas closure and a 20% effort decrease.

5.4.2. Influence of three management scenarios on the EEC ecosystem and related flatfish fisheries.

5.4.2.1. Fishers' response to management constraints.

The effort allocated to the three métiers of the netters' fleet did not seem to be affected by the three constraining management scenarios (Figure 5.7).

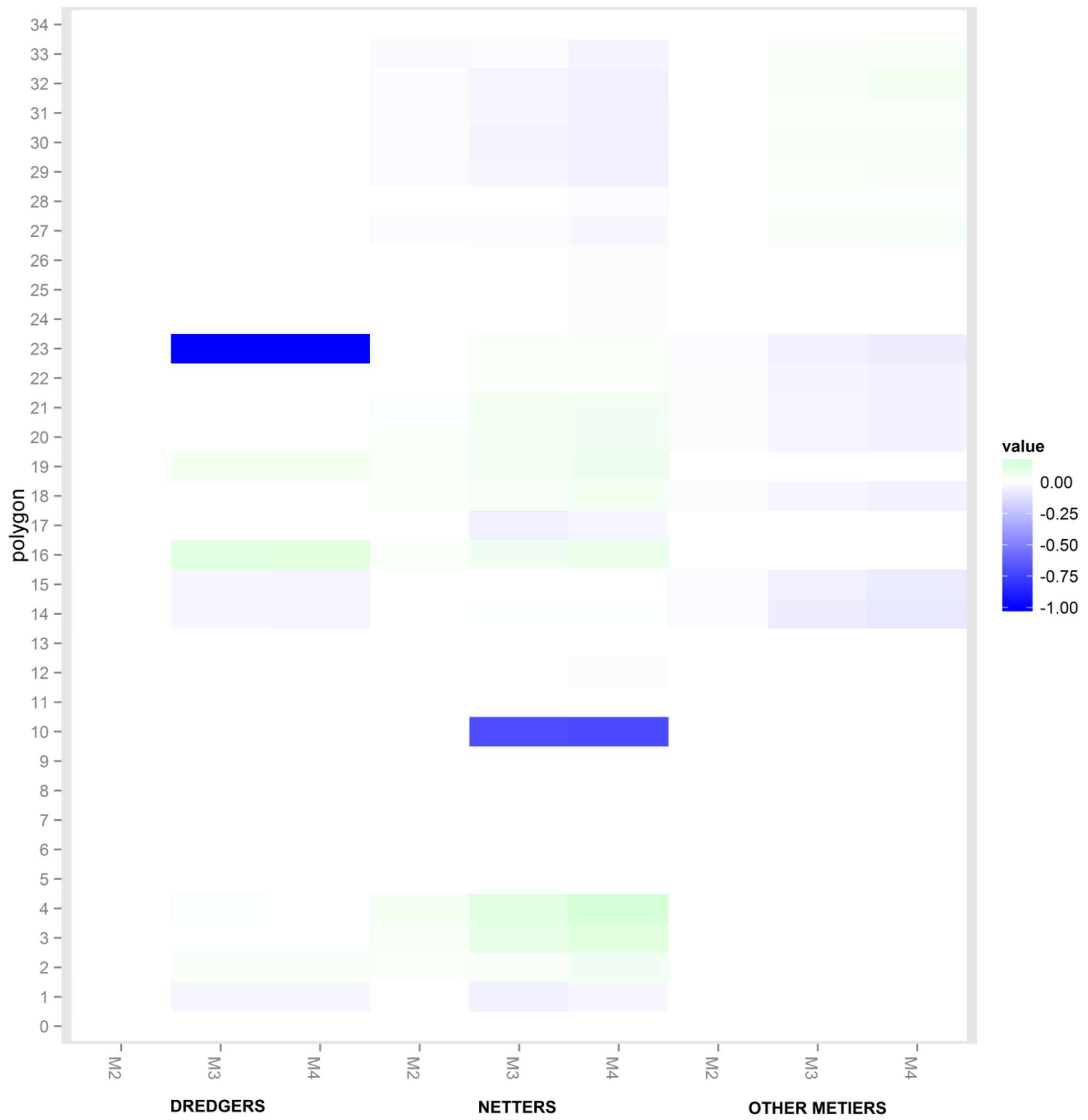

Figure 5.8 Variation of effort allocation in space and per métier compared to the status-quo for each management scenario (standardized difference). EFF20 refers to 20% effort decrease scenario, MPA refers to the area closures and MPAEFF20 referes to the management scenario combining effort reduction and area closures.

However, noticeable changes of effort allocation were observed across space and time (Figure 5.1 and Figure 5.8). First, the area closure scenario M2 slightly affected effort allocation over time, with relatively more effort allocated by the netters' métier to the western part of the EEC and less to the French side of the Dover Strait. For the two other constrained scenarios (M3 and M4) involving a 20% reduction in total fishing effort, the spatial allocation of dredgers was the most impacted with a decrease of effort in the middle and the western part of the EEC. We also highlighted a concentration of effort off the Bay of Seine (polygons 16, 19) for both scenarios. In M3 and M4, we observed for netters a concentration of effort in the Bay of Seine and in the western side of the EEC and a decrease of effort in the Bay of Veys (polygon 17), in the English coastal area (polygon 10) and in the French side of the Dover Strait. These areas, except polygon 10, are mainly covered by closed areas (Figure 5.1). In contrast, more effort was allocated to the other métiers in the Dover Strait, and less in the middle of the EEC and the Bay of Seine. The only seasonal changes were observed for the dredgers' métier, with a switch of allocation between the first and the second quarter (Figure 5.9).

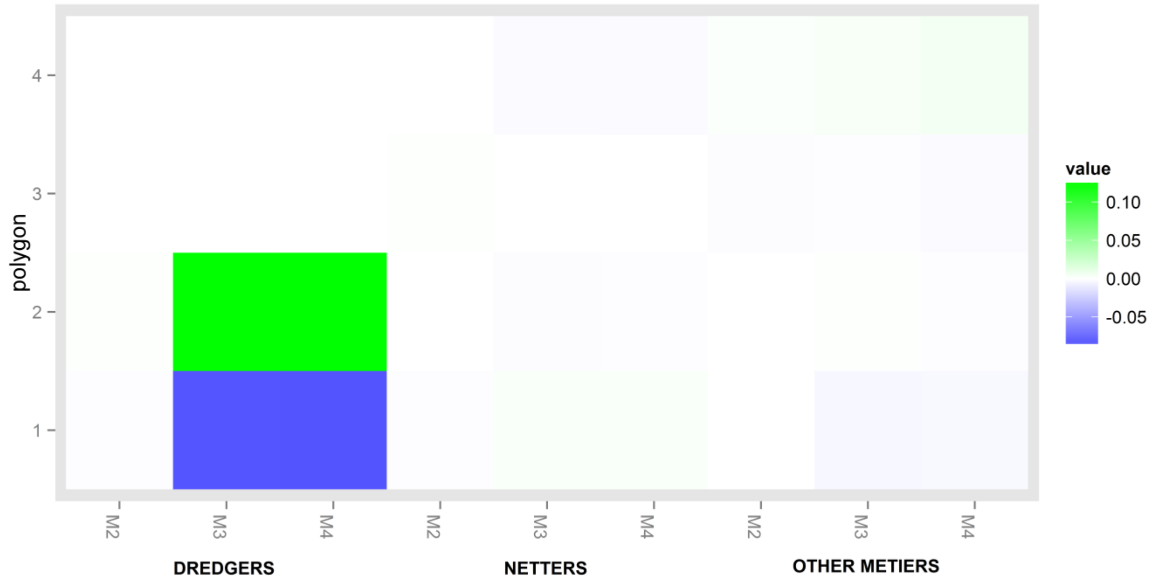

Figure 5.9 Variation of effort allocation per quarter and per métier compared to the status-quo for each management scenario. The codes are those used in Figure 5.8.

5.4.2.2. Impact of management constraint on the EEC ecosystem.

We focused first on the biomass variation of each upper trophic functional group in the Atlantis EEC (Figure 5.10). Each of these functional groups reacted to the three management constraints consistently, although with a difference in the amplitude of the response. M2 had the lowest effect on biomass, however, the biomasses of sole, shark, ray and plaice was slightly increased. M4 had a larger impact than the effort reduction alone. In short, the biomass of predator fishes (rays, sharks, plaice, sole, seabass, dab, gurnard and cod) increased, while the biomass of their main prey (small demersal fish, benthic deposivor and small gadoid) decreased.

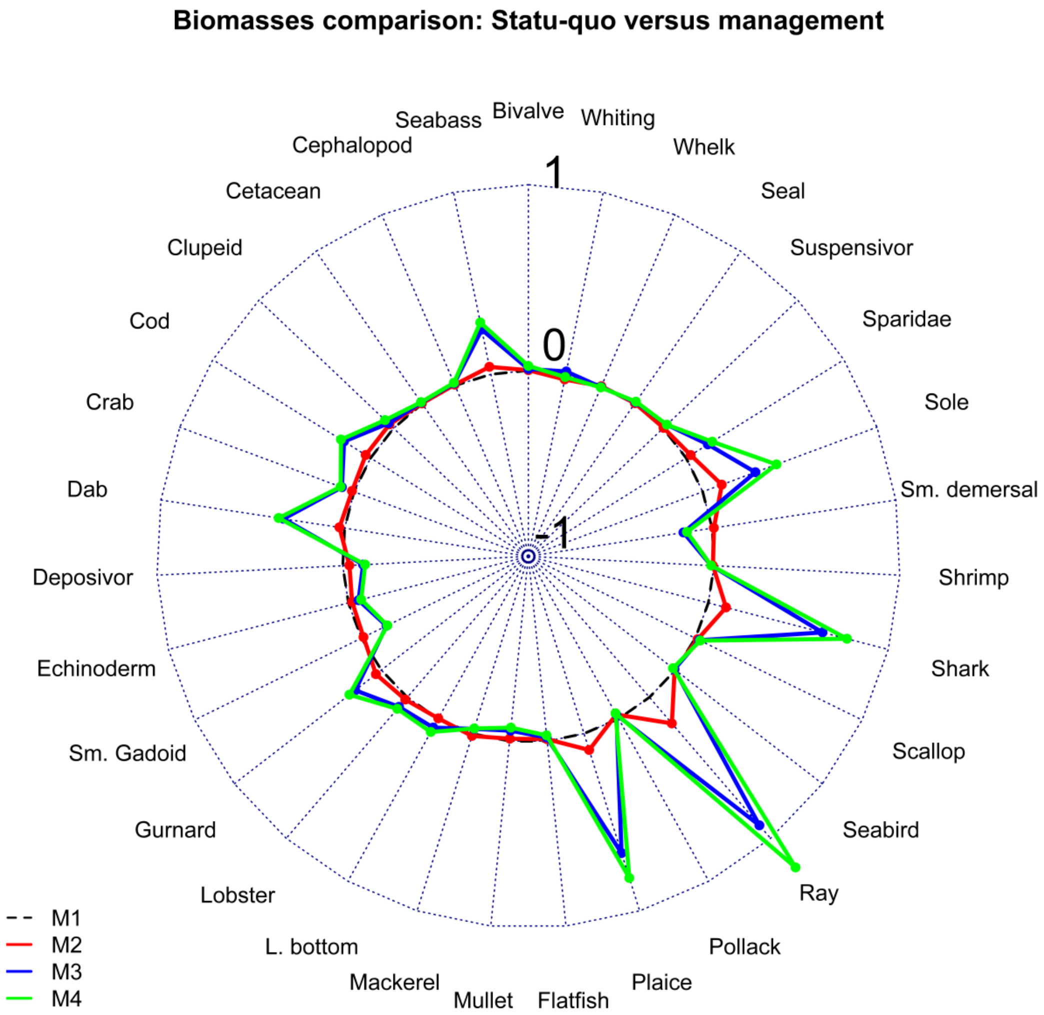

Figure 5.10 Relative variation of simulated biomasses after 50 years of simulating each management scenario. The black dashed line is the status-quo, the red line represents the area closure scenario, the blue line the 20% effort decrease scenario and the green line a combination of area closure and effort reduction.

The trophic relationships were similarly impacted. As observed previously, the least impacting scenario was M2 (Appendix XIII-Appendix XVI), and the most impacting was that combining both management measures (M4). The predation importance of rays, sharks, plaice, gurnard and cod increased after 50 years of management constraints (Figure 5.11) and the importance of mammals and seabirds decreased in the meantime. Predators' diet also changed after 50 years (Figure 5.12), with an increase of plaice, ray, sole, dab, cod, and gurnards in the diet. In contrast, whiting, small demersal fish and gadoids, and mackerels decreased in predator' diets.

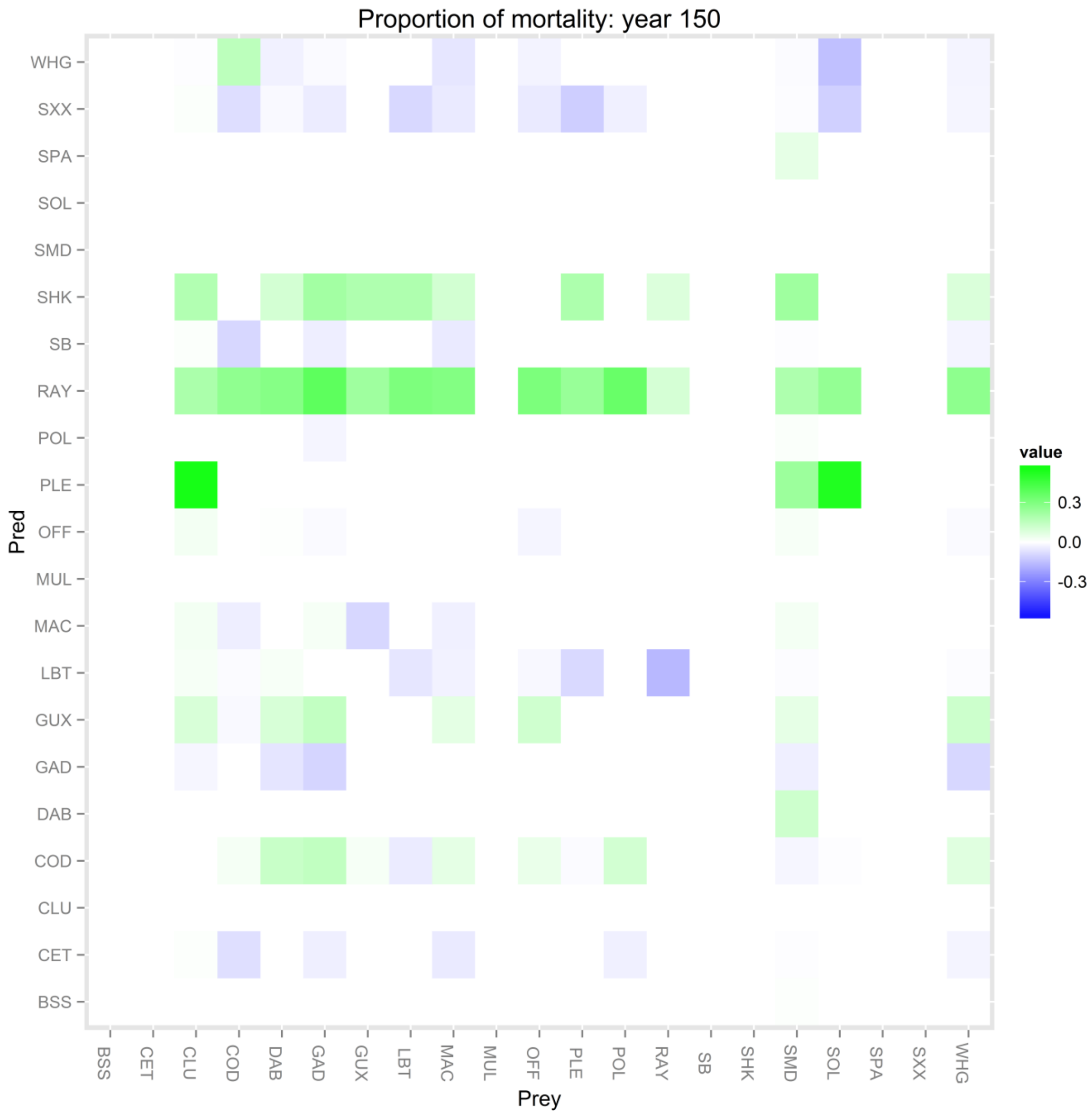

Figure 5.11 Predator-prey relationship modification after 50 years of simulating the 20% effort decrease and area closure management scenario compared to the status-quo: variation of the proportion of mortality per predation explained per predator.

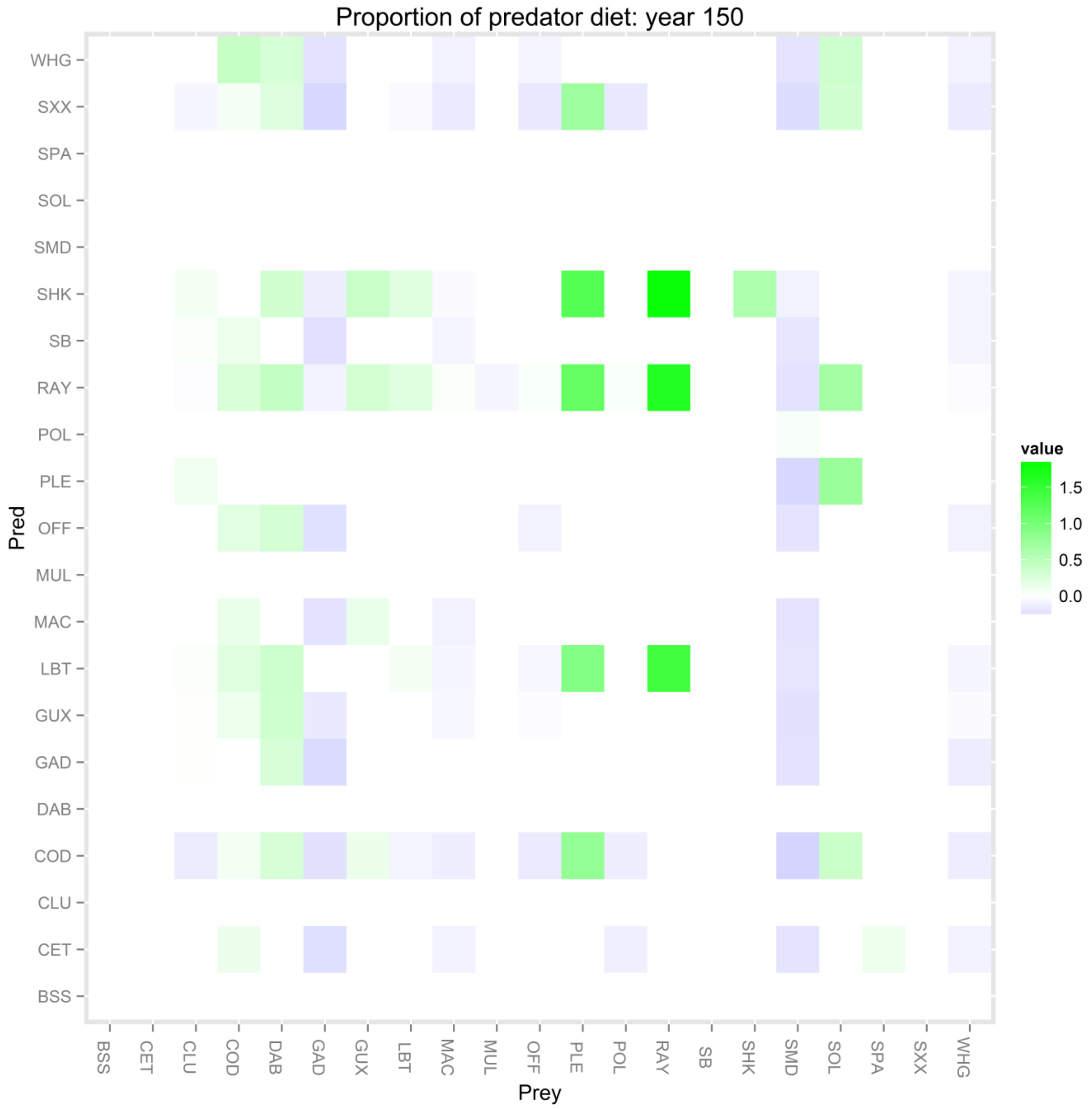

Figure 5.12 Predator-prey relationship modification after 50 years of simulating the 20% effort decrease and area closure management scenario compared to the status-quo: variation of the proportion of preys in predators' diet.

Finally, the total fishing mortality decreased for the functional groups in the Atlantis ECC, with a wider impact of the area closure when combined with the effort decrease (M4) and a less noticeable change with the area closure alone (M2). The decrease of fishing mortality was around 5% for each functional group in M2, between 20% and 24% for M3 scenario and between 23% and 26% in M4 scenario. However, Clupeid fishing mortality appeared to be the same in each scenario with a decrease of total fishing mortality of around 5% (Figure 5.13).

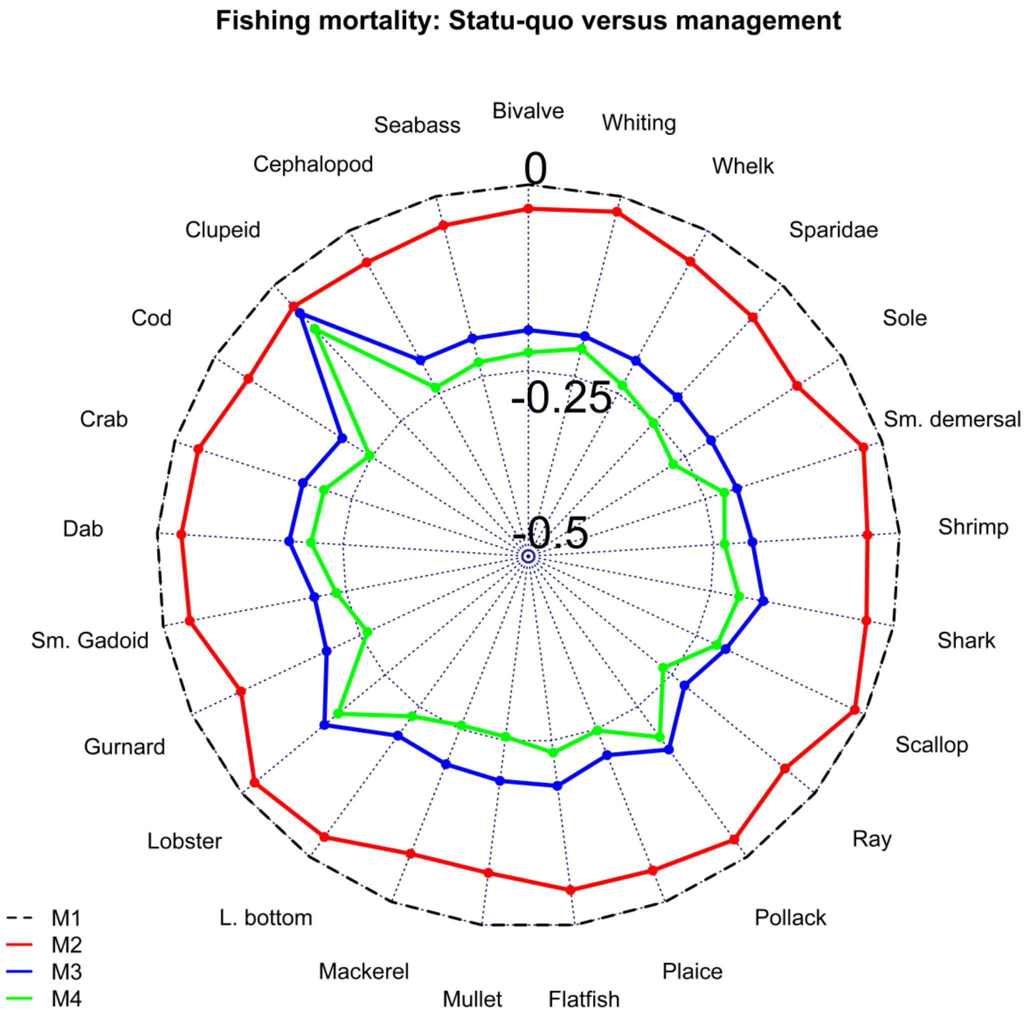

Figure 5.13 Total fishing mortality after 50 years of simulations. Black dashed line, status-quo scenario; red line, area closure scenario; blue line, 20% of effort reduction scenario; green line, scenario combining area closures and 20% effort decrease.

5.4.2.3. Evaluation of 12-18m netters fleet fishing performance.

We analysed both short term impact (2 years after the beginning of scenarios) and long term impact (after 50 years of scenario). However, no noticeable changes were observed in the netters' fishing performance in the short term (Appendix XVII), so we only present here the long term results (Figure 5.14). Again, we highlighted a larger impact of scenarios involving effort decreases (M3, M4) compared to those involving area closures alone (M2). We noticed an increase of sole, plaice and dab CPUEs for the three métiers, with the most important increase for the other métiers. We also observed an increase of the total VPUE of netters métier and "other métiers" in the three constrained management scenarios.

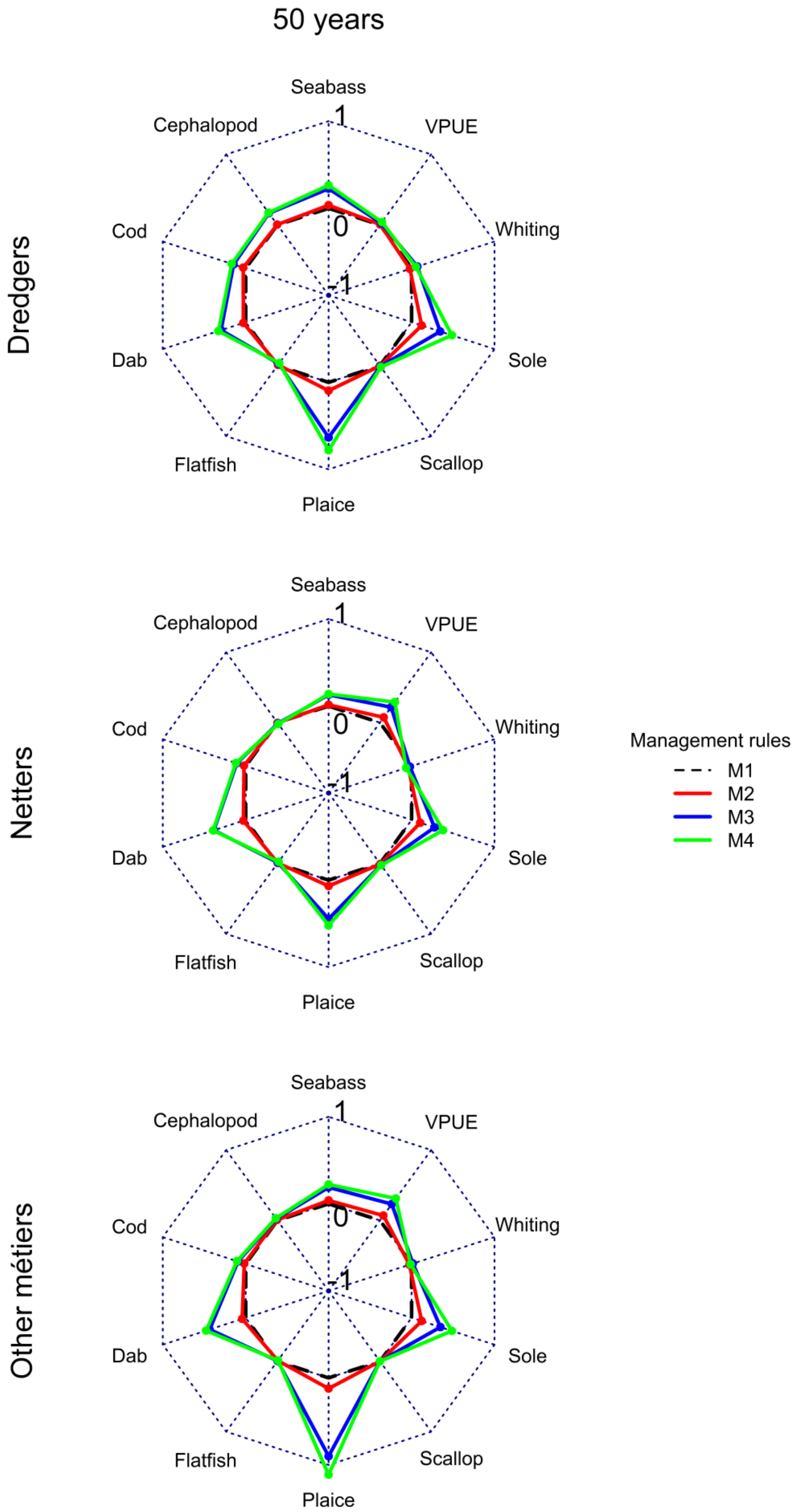

Figure 5.14 CPUE of the main species and total VPUE for each scenario and each métier after 50 years of simulation. Black dashed line, status-quo; red line, area closure scenario; blue line, 20% effort decrease scenario; green line, scenario combining area closures and 20% effort decrease.

5.5. Discussion

5.5.1. Evaluating the coupling of three different fleet dynamics models with Atlantis EEC

In the first part of this study, we coupled Atlantis with three fleet dynamics models, a RUM-based model, and two gravity models. We showed that the RUM-based model provided the poorest representation of French netters' fishing effort allocation, as observed over the period 2002-2011, although this model included most of the fishers' behaviour drivers identified in Chapter 2. RUMs are statistical, data-driven, models. We used here the RUM parameterized in Chapter 2, and which was fitted over three years only (Girardin et al., 2015). As a result, even if the RUM was well suited to mimic fishers' behaviour over the short period 2007- 2009 (Appendix I), it was not appropriate to provide forecasts 50 years ahead. Once coupled with Atlantis, the netters fleet allocated its fishing effort to the alternative choice with the most important utility after a few years. This resulted in some alternatives becoming not selected at all. Because the model uses information (e.g., VPUE, CPUE) collected during the previous year when operating the different alternatives, that information becomes gradually unavailable to the fleet for these alternatives that are not selected anymore after a few years ("loss of memory"), even where these alternatives have a great utility value.

In the case, of the gravity model building on total VPUE (GravVPUE), fishing effort was increasingly allocated to the dredger métier. This model is only driven by two drivers: total expected revenue and tradition, through information on VPUE and effort the previous year for each alternative choice. The increased effort allocated to the dredgers métier then results from vessels operating dredges in the EEC mainly targeting scallop, the most important species landed in value (and in terms of VPUE). Even if the tradition factor limits the inter-annual métier changes, the weight of the VPUE factor gradually inflates over the 50 simulated years, resulting in the netters' fleet increasingly operating the most profitable choice, e.g., the dredgers métier. This highlights a limit of our fleet typology as well, since within the netters' fleet the activity of which is dominated by dredging should be converted into a dredgers' fleet. This shift across fleets, however, could not be easily implemented in the absence of reliable economic data.

Finally, the model of gravity with the targeting of sole and the tradition (GravSOL) best mimicked the spatiotemporal allocation of effort and métiers choices. This is because, for both trammel netting and "other métiers" alternatives, sole is a very important component of the catch (first for trammel netters and the second for "other métier"). These results of the coupling between Atlantis EEC and the GravSOL model also bears out the outcomes of Chapter 3 (Girardin and Marchal, in prep), i.e., that both targeting and tradition are relatively more important than expected revenue in in terms of driving the behaviour of passive demersal fleets.

5.5.2. Impact of several management scenarios on both the EEC ecosystem and the French netters' fleet

5.5.2.1. Main response to management scenarios

We explored extreme scenarios in this study to provide a first pass assessment of ecosystem and fishers' behaviour response to management. Such extremes were used because if no response was found here there is little expectation of more subtle management shifts leading to observable management outcomes.

The scenario including only the area closures (M2) had the most limited impact on each component of the ecosystem. Previous work on area closures showed similar results, suggesting MPAs alone could not deliver substantial conservation benefits (Ainsworth et al., 2012; Bastardie et al., 2014; Dinmore et al., 2003; Hiddink et al., 2006; Kaplan et al., 2012; Nøstbakken, 2008; Salomon et al., 2002). The reallocation of effort outside closed areas could also explain that the biomass increase due to the area closure is limited by increased fishing effort and catches outside these areas (Bastardie et al., 2014; Goñi et al., 2008; Hiddink et al., 2006; Murawski et al., 2005; Russ et al., 2003; Valls et al., 2012). However, in our case study, a small benefit of MPA was still observed, with some increase of the main top predators' biomass and a decrease of the total fishing mortality (Colléter et al., 2012; Gell and Roberts, 2003; Halpern and Warner, 2002; Mosquera et al., 2000). In this study, area closures are mostly distributed in coastal area where most of the EEC nursery grounds are located, so the implementation of no-take areas increases the survival of juveniles in our model and hence recruitment to the overall population (Libralato et al., 2010), which could explain the conservation benefit observed.

We also showed that the 20% total effort reduction management scenario has an important beneficial impact on the entire ecosystem, which confirms previous results by Salomon et al. (2002). The management scenario leading to the best conservation performance is the one combining area closure and effort reduction (M4), which bears out results from previous studies which indicate that a mix of management options must be used to meet objectives across habitats, predators, prey and socioeconomic objectives (Ainsworth et al., 2012; Bastardie et al., 2014; Dinmore et al., 2003; Hiddink et al., 2006; Kaplan et al., 2012; Nøstbakken, 2008; Salomon et al., 2002).

5.5.2.2. Changes in fishers' behavior

Even if we didn't observe major changes in effort allocation per métiers, the spatial distribution of fishing effort was altered after 50 years of simulation (Kaplan et al., 2012) in both scenarios with effort reduction (M3 and M4). As above, the application of management measures outside area closure had a greater impact on the outcomes (Bastardie et al., 2014; Hiddink et al., 2006; Kraak, 2011; Salas and Gaertner, 2004). The most impacted métier was the dredging with an increased allocation of effort towards the scallops fishing grounds off the Bay of Seine. Netters seemed to allocate their effort offshore and in the Western part of the EEC in both scenarios involving effort reduction, while the other métiers concentrated their effort in the Dover Strait. This might be explained by an increase of the proportion of sole in those areas.

5.5.2.3. Ecosystem functioning

As shown in some previous studies (Colléter et al., 2012; Kaplan et al., 2012; Libralato et al., 2010; Mosquera et al., 2000; Murawski et al., 2005; Salomon et al., 2002), an increase of top predators' biomass occurred in all the scenarios with effective management and the associated top-down effect was observed with a decrease in the biomass of the main prey species. However, compared to the magnitude of the increase of predator' biomass, the decrease of prey' biomass was relatively low. This differential change may be due to a diet shift also the relative biomass of different groups, very small for top predators and large for prey groups. We saw a change in diet, with an increase of consumption of upper trophic levels species compared to the status-quo scenario and less predation on forage species, which could explain the low impact of the different management scenarios on lower trophic level species.

The two scenarios involving effort decreases showed an important conservation benefit for the main target species (Roberts et al., 2001), especially for sole. Fishing pressure seemed to decrease evenly across functional groups in each scenario except for the Cupleoid, which was the most important migratory group in our model. The effect of effort reduction in this case was not sufficient during the time spent inside the EEC to increase the biomass of this group. Moreover, some predators (sharks, plaice and seabirds) slightly increased the proportion of Clupeid in their diet, which may have also contributed to the reduction in biomass of this group.

5.5.2.4. Fishing performance

We noticed no significant change of fishing performance (measured in terms of VPUE and CPUE) after 2 years of management, irrespective of the constraint being imposed, which bears out the outcomes of Hamon et al. (2013), who analysed the consequences of climate change perturbation on fishers' behaviour. Hamon et al. (2013) suggested that the plasticity of fishers' behaviour might balance the effects of management measures. In case of various constraints on fishing activities, fishers tend to optimize their fishing ground choices to minimize revenue losses. The lack of short-term effects on fishers' economic performance could result from that neither fuel costs, nor the increase of time spent steaming to reach fishing grounds due to the implementation of area closures (McManus, 1997; Nøstbakken, 2008) were considered. Accounting for these aspects could have led to adverse economic results when implementing area closures. After 50 years, only slight changes were noticed when implementing area closures without effort reductions (M2). Again here, this could result from the lack of consideration of fishing costs in our model.

The impact of management scenarios implying effort reductions were métier-dependent (Ainsworth et al., 2012; Kaplan et al., 2012; Morzaria-Luna et al., 2012) – an increase of flatfish species CPUEs was observed and induced an increase of the trammel nets' and other metiers' VPUE. This could result from the biomass increase of the main target species. The dredgers' VPUE, however, was unchanged. This is because neither the biomass nor the CPUE of the dredgers' main target species, scallops, was impacted by any management regulation. Effort reduction management scenarios (M3 and M4) increased the bycatch of dredgers, which could constrain their activity if the catches of such "choke" species are limited by TACs.

5.5.2.5. Limits and perspectives of our study.

We made several assumptions to achieve the calibration of the Atlantis EEC model and the coupling with the fleet dynamics models, which could influence the outcomes of the simulated management scenarios. In Atlantis EEC, the spatial dynamics of functional groups was influenced only by forced seasonal patterns. Indeed, due to the lack of information on density dependence relationship for each species we assumed that the distribution of fish was not affected by their density. This hypothesis might over-estimate the spill-over effect of area closures by reallocating the species' biomass increase inside closed areas accordingly to their seasonal distribution, as implemented in the model. Moreover, the distribution of juveniles might not reflect exactly the variability that can be observed in reality and could impact the evaluation of the impact of area closures. We assumed also that the stock recruitment relationship for the fish implemented in Atlantis EEC follows a Beverton and Holt formulation. Using a Ricker stock-recruitment relationship would have limited the importance of top predators biomass increases and changed the variation observed in the ecosystem functioning. We did not consider the impact of habitat quality on fish distribution and so the effect of decreasing the impact of fishing on the seabed was not explored.

To perform the coupling between Atlantis EEC and the fleet dynamics models we assumed that both catches and effort were evenly distributed within each statistical rectangle and each Atlantis polygon. By using fine-scale vessel monitoring system (VMS) data, which were not available when we fitted the RUM, we could have applied the same spatial structuration in both the fleet dynamics models and Atlantis EEC. We also assumed that fishers were not searching for new fishing grounds, as in other cases (Dinmore et al., 2003; Rijnsdorp et al., 2001). Moreover, prices and total effort were not implemented dynamically as these were provided to the model as observed time series. As highlighted previously the fishing cost was not considered, so the full profit achieved by netters could not be calculated. All of these factors could have influenced the dynamic nature of the resulting model.

Further development of the model could be considered. We could apply the fleet dynamics model to several fleets concomitantly and not only the netters. More management scenarios could be performed, including TAC and harvest control rules building on the evaluation of the fishing mortality at the maximum sustainable yield (Fmsy). The wide range of processes implemented in Atlantis and its modularity allows many possible analyses. In addition, two ecosystem models of the EEC are currently under development, OSMOSE (Travers-Trolet, 2012) and ISISfish (Lehuta et al., in press). A comparison of the performances of these models, when testing similar scenarios, could allow a better understanding of their limits and robustness.

5.6. Conclusion

In this study, we showed that integrating fishers' behaviour into an ecosystem model is not straightforward. Data-driven RUM-based models were not appropriate to perform long-term projections. We demonstrated, however, that information on tradition and species targeting included in a fleet dynamics gravity model fitted the observed netters' effort allocation well and this is the approach used in the end-to-end model Ecospace (Walters et al., 1999). The evaluation of the different management scenarios suggested some benefits for all the compartments of the ecosystem and in terms of fishing performance. The application of area closures combined with a 20% reduction of total effort was the most efficient scenario, especially for top predator species and fishing performance. However, we demonstrated the complexity of understanding the response of the coupled ecosystem-fishers model after the introduction of management measures. Thus, the ecosystem response was varied across the different functional groups, across the different fishing classes and in the spatial sub-regions (polygons). Most of our results are qualitatively in accordance with those achieved in previous studies. Our results provide a tool to implement ecosystem-based management and maritime spatial planning in the Eastern English Channel.

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6. Chapter 6: General discussion and perspectives

6.1. Synthesis

Stakeholders and scientists widely recognize the need of holistic approaches to achieve conservation and utilisation objectives for complex marine ecosystems. The improvement of our knowledge from past failures and successes (Daw and Gray, 2005; Hilborn, 2004; Hilborn et al., 2001), has led stakeholders and scientists to move gradually from traditional single-species management (Garcia, 1994; Ludwig, 2002; McAllister and Kirchner, 2002; Rosenberg, 2002) towards an ecosystem-based management approach (EBM), building in the full complexity and dynamics of ecosystems of their interactions with human activities (Botsford et al., 1997; Browman and Stergiou, 2004; Garcia et al., 2003; Leslie and McLeod, 2007; Pikitch et al., 2004). The Ecosystem Approach to Fisheries Management (EAFM) (Browman and Stergiou, 2004) aims at maintaining and restoring fisheries resources to sustainable levels, while minimizing the adverse effects on marine ecosystems (Pauly et al., 2002). In the EU, this approach has been endorsed in 2008 with the introduction of the cross sectorial Marine Strategy Framework Directive (MSFD) (EC, 2008).

6.1.1. Thesis objectives

The objective of this thesis was to assess the ecological and utilization impacts of various conservation and access restrictions to the EEC maritime domain. An ecosystem modelling approach was used to mimic most of the processes governing marine ecosystem dynamics in the region. We focused our study of the ecosystem around sole and plaice and on the French fleets targeting those species in particular. The evaluation of management impacts on marine ecosystem and fishery was performed in three steps. First we focused on a key component of the ecosystem, the fishers' behaviour. By using fleet dynamics models building in a random utility function, we characterized the main factors driving fishers' decisions as well as their interactions with other human activities (maritime traffic). We completed this study with a thirty years review of fishers' behaviour literature worldwide. Second, to understand the knock-on effects of EEC flatfish fisheries on sole and plaice, we developed an ecosystem model which represented the main processes governing the EEC marine domain including fishing activities. Finally, to evaluate the consequences of various management scenarios on ecosystem functioning and on fishers' behaviour, we coupled fleet dynamics models derived from Chapters 2 and 3 with the Atlantis EEC calibrated in Chapter 4. We evaluated the impact of area closures and effort restrictions on the functioning of the ecosystem-fleet coupled system.

6.1.2. Contribution to the improvement of our knowledge in ecosystem functioning: the case studies of the EEC flatfish fisheries.

In this study, we applied recent methodological approaches to analyse the functioning of the EEC marine ecosystem and its interactions with fishing activities, and to evaluate the performance of different management scenarios. In Chapters 2 and 3, we focused on the understanding of fishers' behaviour. In the past, the response of fishers to regulation and environmental changes was often disregarded by fisheries advisers and decision-makers, leading to fisheries management failures (Daskalov and Mamedov, 2007; Daw and Gray, 2005; Dickey-Collas et al., 2014; Hardin, 1968; Poulsen et al., 2006; Radovich, 1982; Walters and Maguire, 1996). Fishers are a key component of the ecosystem, and the understanding of their behaviour is critical to anticipate their reaction to management rules and the effect on the impacted ecosystem (Fulton et al., 2011a; Hilborn, 2007; Leslie and McLeod, 2007; Wilson and McCay, 2001).

In Chapter 2, we investigated the processes underlying short-term decisions (van Putten et al., 2012) of spatial and métier allocations, with particular attention paid to the EEC flatfish fisheries. We explored the factors driving fishers' behaviour, including their knowledge of stock availability, their past experiences, and we also considered spatial interactions with other human activities (other fishing fleets, maritime traffic) and management. To do so, we applied a Random Utility Model (RUM), which was based on the maximization of a utility function associated to each alternative choices (in our case: a combination of métier and of a fishing area). RUMs have already been applied in several occasions to model métier choices (Andersen et al., 2012; Holland and Sutinen, 1999; Marchal et al., 2009), area choices (Hutton et al., 2004; Wilen et al., 2002) or choices of target species (Pradhan and Leung, 2004a; Vermard et al., 2008). However, past fleet dynamics studies mainly focused on two drivers: past experience and economics. One innovative aspect of our study is the inclusion of the spatial interaction with other human activities and management. The mixed demersal fleets we investigated were strongly influenced by past effort (tradition) and past VPUE (expected revenue) as already shown in other fishers' behaviour studies (Holland and Sutinen, 2000). Our study also showed different responses across fleets. For instance, large trawlers relied more on seasonal information compared to smaller trawlers. Management proved to be a substantial driver of fishers' behaviour. We then showed that the scallop dredging opening season induced a seasonal switch of métier choices for dredgers and polyvalent active fleets, which was reflected in our model by a negative effect of past short term effort allocation. Other maritime activities also influenced fishers' behaviour. Larger vessels tended to avoid areas where marine traffic was intensive, or where spatial competition with other French fleets was important. However, French dredgers generally favoured fishing areas with large UK vessels concentration, which could be due to the presence of common target species (e.g., scallops).

Finally, the RUM parameters estimated over 2007-2008 were successfully used to forecast the 2009 effort allocation. By developing a forecast method based on several random iterations, we took into account model variability and increased the accuracy of the prediction, even for the fleets with the weakest model fit. It should, however, be emphasized that RUMs are statistical, data-driven, and therefore their parameterization may need to be re-evaluated following major changes concerning management and/or the economic, environmental and ecological context of fisheries.

In Chapter 3, we placed the outcomes of Chapter 2 in a broader perspective by reviewing evidence from past fishers' behaviour studies worldwide, focusing on discrete choice models., We hence performed a meta-analysis of the past three decades studies using RUM, building on the literature review conducted by van Putten et al. (2012). We identified six groups of fishers' behaviour drivers: tradition, expected revenue, species targeting, cost, risk-taking and concentration of other human activities. The main challenge of our meta-analysis was that the information provided by the studies being reviewed differed in terms of the drivers under consideration and in terms of the RUM structures being used. That challenge was overcome to the extent possible, by standardizing the outcomes of the different studies in a stepwise fashion. As shown in several studies (Holland and Sutinen, 1999, 2000; Marchal et al., 2009; Pradhan and Leung, 2004a; Vermard et al., 2008), fishers used information gained during their past activities to make decisions in accordance with their habits and from which they expect a substantial revenue. We also showed that fishers are mainly risk-adverse and prefer to choose alternatives with low revenue variability, a result already anticipated by Dupont (1993) and Hilborn and Ledbetter (1979). In this review, concentration of other users mainly appeared to be a source of information rather than a congestion issue for fishers (Campbell and Hand, 1999; Vignaux, 1996). The congestion issue mainly occurred when drivers were derived from other humans activities such as maritime traffic, aggregate extraction (Marchal et al., 2014a; Tidd et al., 2015), or other fleet types (Hilborn and Ledbetter, 1979; Marchal et al., 2014a). We also highlighted that passive and active demersal fleets are not necessarily influenced by the same drivers. While active demersal fleets relied more on seasonal (long-term) drivers to make their decision, passive demersal fleets favoured immediate information. Although fishers are often assumed to be profit-maximizing economic agents, we showed in our meta-analysis that they may value tradition and species targeting more than expected revenue (Holland and Sutinen, 2000; Marchal et al., 2009, 2014a; Valcic, 2009; Vermard et al., 2008; Wilson, 1990). Expected revenue and fishing costs had a similar influence on demersal fleets' behaviour. Concentration of other human-activities and risk-taking were less influential than expected revenue.

As shown in Chapters 2 and 3, understanding of fishers' behaviour is critical to implement of efficient management. However, the evaluating ecosystem-based management strategies also requires an enhanced understanding of the ecosystem response to versatile fleet dynamics (Browman and Stergiou, 2004; Fulton et al., 2011b, 2014; van Putten et al., 2012; Wilen et al., 2002). Atlantis has been described as the most suitable ecosystem model available to support the EAFM process (Plagányi, 2007). In Chapter 4, we implemented and calibrated the Atlantis end-to-end model to mimic the functioning of the EEC ecosystem. The Atlantis EEC model reproduced reasonably the observed catches for the most important fishing groups. A number of valuable insights regarding the EEC ecosystem functioning were gained during the calibration process. We first showed the importance of detritus in the shallow EEC, an ecosystem mainly influenced by river inputs and by benthic fauna (Arbach Leloup et al., 2008; Cachera, 2013; Carpentier et al., 2009; Daskalov et al., 2011; Dauvin and Desroy, 2005; Loizeau et al., 2001; Moore et al., 2004). This was particularly true in the coastal flatfish nursery grounds (Kostecki et al., 2011; Le Pape et al., 2013; Riou et al., 2001; Rochette et al., 2010). We also highlighted the importance of two key vertebrates groups in the EEC ecosystem: cod and whiting. Both cod and whiting are opportunistic predator species, which impacted most of the functional groups represented in our model. Whiting was at the same time an important source of food for top predator species. In contrast, due to their ecology, the two focus species of this study, sole and plaice, had a limited influence on the vertebrates groups. Sole and plaice were, however, strongly dependent of the productivity of nursery grounds and on the availability of benthic invertebrates, and they played a crucial roles in the benthic functioning of the ecosystem, which bears out the outcomes of previous studies (Dauvin and Desroy, 2005; Kostecki et al., 2011; Le Pape et al., 2013; Riou et al., 2001; Rochette et al., 2010; Trimoreau et al., 2013). Finally, the explicit implementation of fishing activities in the Atlantis ECC model demonstrated the complexity of assessing the catchability and selectivity of various fishing groups, and it also revealed the influence of discards on the trophic networks. Discards could be a significant source of organic matter for the benthic community and for the predators depending on it.

In Chapter 5, we explored the consequences of several management scenarios on ecosystem functioning and fishers' behaviour. We gathered to that purposes information gained in Chapters 2-3 (fleet dynamics) and 4 (EEC ecosystem functioning). We focused exclusively on one flatfish fleet, consisting of the 12-18m French netters, and we implemented management rules building on area closures and effort reduction in isolation or in combination. First, we coupled three fleet dynamics model with the Atlantis EEC, compared the outcomes of the resulting combined model with observations, and identified the fleet dynamics model that provided the best fit. One fleet dynamics model was derived from the RUM parameterized in Chapter 2, and the two others were gravity models (Allen and McGlade, 1986; Caddy, 1975; Walters and Bonfil, 1999; Walters et al., 1993) based on the outcomes of Chapter 3. The most suitable fleet dynamics model was found to be the gravity model including tradition and sole targeting drivers, which was parameterized based on the results of Chapter 3.

We then evaluated four different scenarios: (i) no management restrictions, (ii) area closures, considering the current spatial distribution of MPA in the EEC, (iii) a 20% fishing effort reduction for the entire fishery and, (iv) a combination of (ii) and (iii). In accordance with previous studies, applying simultaneously area closures and effort reduction outside the protected domain delivered the most efficient conservation benefits, while closing areas without reducing effort was generally inefficient (Ainsworth et al., 2012; Bastardie et al., 2014; Dinmore et al., 2003; Hiddink et al., 2006; Kaplan et al., 2012; Nøstbakken, 2008; Salomon et al., 2002). We showed that the area closure scenario led to a 5% reduction in fishing mortality (averaged across vertebrates species), which contrasts with 20% fishing mortality decrease brought about by the effort reduction scheme. This confirms that the implementation of area closures and fish stocks should be accompanied by additional conservation measures to reduce the pressure on non-protected areas.

Overall, the EEC ecosystem, as we perceived it, reacted similarly to each management scenario, with only a difference in the intensity of the response. Considering fishers' behaviour, no change of métier was observed. However, the spatial allocation of effort varied across métiers with netters and dredgers concentrating their effort in the middle of the EEC and other métiers focussing their effort in the Dover Strait. The most impacted metier was dredging. As observed in previous studies, the management measures were the most beneficial for top predator fish species with a noticeable increase of their biomasses (Colléter et al., 2012; Kaplan et al., 2012; Libralato et al., 2010; Mosquera et al., 2000; Murawski et al., 2005; Salomon et al., 2002). In contrast, the impact on prey species was less apparent than observed in several studies. Part of the predation was reallocated towards the juveniles of the impacted predators, which could explain the more limited impact on prey functional groups. Area closures without effort reduction had a limited, but positive, impact on the ecosystem functioning overall. These small conservation benefits result from a decreased fishing pressure on coastal nursery grounds areas, where these area closures are mainly located. Fishing performances were also enhanced to varying degrees by the three management scenarios, with some differences across métiers. However, this result should be considered with caution, because fishing costs were not accounted for in our analysis.

6.2. Ecosystem modelling platform a requirement to implement management strategy.

There has been a growing interest in ecosystem modelling in past decade (Arkema et al., 2006; Brodziak and Link, 2002; Browman and Stergiou, 2004; FAO, 2003; Fulton, 2010; Garcia et al., 2003; Sanchirico et al., 2006). Progress have been made in the understanding of ecosystem functioning, which has contributed to the development and the application of end to end ecosystem models to inform decision-makers of the long-term effects of management strategies on natural living resources (Fulton, 2010; Fulton et al., 2011b; Plagányi, 2007; Sainsbury et al., 2000; Travers et al., 2007). Our study, building on a holistic modelling approach, contributed to enhance our understanding of the complexity of ecosystem dynamics and of the dual interactions between ecosystems and fishing activities. Based on these results, Operational Management Procedures (OMP) (Butterworth and Punt, 1999; de Oliveira et al., 1998) or Management Strategy Evaluation (MSE) (Smith et al., 1999) could then provide scientific recommendations for management measures (Plagányi, 2007). Several levels of complexity will have to be considered to implement such models, and also to apply EAFM to achieve conservation and utilization objectives. Atlantis was originally created for the prupose of MSE and so this would be a logical next step.

6.2.1. The EAFM approach: a multidisciplinary analysis of the ecosystem

A wide range of processes were considered to mimic the complexity of interacting ecosystem and fishing fleets dynamics. It was thus necessary to improve our understanding of most of the ecosystem compartments, starting from environmental/physical processes all the way through bio-economics. In Chapter 4, we first processed outputs from the MARS3D model (Bailly du Bois and Dumas, 2005). We had to deal with issues such as the integration of river flows hyper-diffusion, or the influence of eddies in coastal area. The second step was the implementation of biogeochemical cycles. At this stage, the mineralisation of nutrients by bacteria, the growth of phytoplankton through photosynthesis and the dynamics of zooplankton required to understand the mechanisms of light penetration, the efficiency of assimilation and also mortality rates of the lowest trophic groups. Similarly, the dynamics of upper trophic layers, such as benthic invertebrates and vertebrates functional groups were modelled and parameterised using a wide range of information. In addition, predator-prey relationships were integrated to implement top-down and bottom-up effects in the trophic network. To mimic vertebrates' population dynamics, knowledge on reproduction seasonality, stock-recruitment, but also migratory patterns were also required. The representation of fishing pressure necessitated the description of gear selectivity, catchability, discarding and targeting processes and also species availability. Finally, we coupled in Chapter 5 the EEC ecosystem model developed in Chapter 4 with fishers' behaviour models building on the outcomes of Chapters 2 and 3, which added another layer of complexity. Such a multidisciplinary approach was made possible by collaboration with scientists from many backgrounds, ranging from biogeochemical to economics. To render this approach fully operational in the context, e.g., of the EAFM, the domain of expertise required should be even broadened so to involve stakeholders directly, and foster a dialogue on the management measures that could be tested using the modelling platform developed in this study.

6.2.2. Various spatio-temporal scales

To investigate the dynamics of the whole EEC ecosystem, fishing activities included, we had to consider a variety of spatio-temporal scales, due to the processes distribution and/or the data available to investigate these processes. In Chapter 4, we thus linked various spatial layers to represent the EEC ecosystem. Hydrodynamics were derived from the model MARS3D, which is discretised at a fine spatial scale of regular 3km x 3km cells to mimic both water flows and biogeochemical cycles. Then, to implement the structure of Atlantis EEC into polygons, we combined the spatial distribution of benthic habitats, sediments, depth but also administrative boundaries. Having defined 35 polygons, we input a spatial distribution for each functional group, including nursery grounds, mainly from recurrent bottom-trawling survey datasets, each with their own spatial scale (limited to the bay of Seine at a resolution of 3'x4.6' for the dredge survey COMOR, all EEC at a resolution of 0.25°x0.25° for the bottom-trawling survey CGFS). Moreover, the implementation of fleet dynamics in Atlantis EEC required the introduction of the spatial distribution of fishing effort at the scale of the ICES rectangle (0.5x 1°) i.e. the finest spatial scale available through logbook data. To understand the spatial complexity of the ecosystem we had to standardize the information available on each compartment and processes of the EEC ecosystem into the 3D spatialised Atlantis EEC structures. . Coupling fleet dynamics models and Atlantis EEC proved particularly challenging, due to the relatively coarse spatial resolution of fishing effort data. We assumed that both fishing effort allocation and catches where evenly distributed within a polygon and a statistical rectangle.

Finally, to introduce the distribution of MPAs in the EEC we standardized the information to the scale of Atlantis EEC polygons to compare the MPAs distribution with the effort allocation within statistical rectangles. Increasing spatial resolution could facilitate the integration of local processes such as larval dispersal or nutrient flow, as used in OSMOSE and MARS3D. However, increasing spatial resolution will dramatically inflate computation time to allow a representation of the entire ecosystem. This is why in holistic models such as Atlantis, the spatial resolution has to be reduced, at the cost of missing some information for highly localised ecosystem processes.

Considering temporal resolution, we used a 24h time step in Atlantis EEC to avoid the consideration of tidal effect. However, various time scales were considered and combined for the different processes being implemented. Biogeochemical and physiological mechanisms are high frequency processes. A typical order of magnitude to investigate such processes would be the minute, possibly the second. In Atlantis, these processes were discretised using a 1 hour time step. Fishers' behaviour involves decisions made in the short-term (less than a day), medium–term (from a day to a month) and long-term (several months or years) (van Putten et al., 2012). In our application, we considered a monthly time step to mimic fishers' decision-making (Chapter 2 and 5). In Atlantis, however, catch processes were implemented with a 24h time step. To input fishing effort in the catch equation, we assumed that the daily effort in Atlantis was constant within a month for each fishing group and represented a proportion of the monthly effort predicted by the fleet dynamics models. Several other processes were implemented with a seasonal scale, such as migration and reproduction. Vertebrates' functional groups were discretised into age classes of one or several years. Finally, the evolution of both trophic networks and fishers' behaviour was analysed after several decades of simulations in the Chapter 5.

The variability of spatio-temporal scale increased the difficulty in understanding and interpreting the functioning of the whole ecosystem, fishers included. However, after discretization of space and time in the modelling platform based on homogeneity assumptions, the interaction of the various scales processes could be analysed and the overall framework used to evaluate the performance of various management measures. This issue is common in ecosystem modelling and it is almost inevitable when integrating a large variety of ecosystem processes. Several options can be used to address the issue of scale differences. First, all the information available could be standardized at a common scale. Alternatively, the different processes may be integrated into modules involving different resolution scales. In our case, both approaches have been used, particularly when coupling Atlantis EEC with fleet dynamics models. Many assumptions had to be made to couple these two modelling approaches, such as the homogeneity of catches and effort within polygons and statistical rectangles. These assumptions almost certainly increase the model uncertainty and influence the outcome of our analyses to an extent that could not be evaluated during this thesis project.

6.2.3. End to end modelling as a tool to mimic ecosystem functioning and to inform management

It has been demonstrated in earlier studies that ecosystem models could be useful tools to evaluate the interactions between environmental, ecological and human interaction, which is crucial to implement efficient management rules (Fulton, 2010; Fulton et al., 2011b; Plagányi, 2007; Sainsbury et al., 2000; Travers et al., 2007). Ecosystem modelling can address a number of issues and questions, and we present below those we focused on in this study (see Plagányi (2007) for a much more complete list):

(i) Improving our understanding of the EEC ecosystem structure and functioning;

(ii) Investigating the importance of sole, plaice, cod and whiting in the EEC trophic network;

(iii) Investigating the effect of the distribution of fishing effort across spatial units and métiers on the EEC ecosystem sustainability; and

(iv) Investigating the conservation performances of management measures building on area closures and/or fishing effort reductions.

6.3. Limits of our study

We made several assumptions to implement dynamically ecosystem functioning and structures, including fishing activities in our model (Chapter 4), and also to evaluate the performance of management scenarios (Chapter 5). We discuss here the main limitations that result from these assumptions.

The analysis of fishers' behaviour in Chapter 2 was conducted using logbook and sales slips information, available at the coarse ICES rectangle scale. Fine-scale VMS data were not available at the time of this study, and concern only vessels larger than 12m. To couple Atlantis EEC and the fleet dynamics models (Chapter 5) we assumed that both effort and catch distribution within ICES rectangle were homogeneous, a hypothesis that is likely to be at fault. Moreover, with VMS data we could have investigated the impact of more local scale activities e.g., sand and aggregate extraction or planned offshore windfarms, on fishers' behaviour and the ecosystem (Marchal et al., 2014b; Tidd et al., 2015). Another limit lies on the short time series (three years of data only) used to fit the RUMs. Using longer time series might have improved the efficiency of the coupling (Chapter 5). We could also have taken in consideration long-term behaviour, e.g., fisheries entry-exit processes (Pradhan and Leung, 2004b; Tidd et al., 2011; Ward and Sutinen, 1994). However, preliminary investigations suggested that it was hardly possible, in the case of the EEC fleets under investigation, to relate the number of vessels and/or total effort to external drivers with available data. In Chapter 3, we had to standardize the outputs of the various fishers' behaviour studies to be able to compare the relative drivers' importance. This process was indispensable, but decreased the initial information provided by these studies.

The calibration of the Atlantis EEC model (Chapter 4) was particularly data-, time- and computationally intensive. Several assumptions were made to simplify the model to the extent possible, and also due to the lack of information on some components of the ecosystem, such as plankton and benthos dynamics, physiological mechanisms per species, vessels' catchability and gear selectivity. Further investigations would be needed to enhance our knowledge on particular poorly known species and on their interactions with the rest of the ecosystem, at different scales: habitat, stock and community. At the basis of the food web structure, the availability matrix in Atlantis was derived from diet matrices output from EwE English Channel model application (Stanford and Pitcher, 2004), or from the stomach content database DAPSTOM. The matrix was poorly informed for juvenile stages and also for non-commercial species. Prey-predators relationships are crucial to understand ecosystem functioning, and a greater effort should be dedicated to the collection of diet data. Further investigation on fish diets have been and are currently being analysed (Cachera, 2013). The use of more up-to-date and complete data in our model could refine our representation of the EEC ecosystem. Most of the unknown parameters were tuned during the calibration, such as the natural (non-predation) mortality. We assumed a Beverton and Holt stock recruitment relationship for each vertebrate functional group. A possible negative influence of large stock biomass on recruitment, e.g., via a Ricker stock-recruitment relationship, was not considered. The application of the Ricker formulation might have decreased the biomass of top predators.The effect of density on the population dynamics have been in part considered through cannibalism and forced quadratic mortalities for some functional groups. However, we did not consider density-dependence for vertebrates and so their spatial distributions were forced (with seasonal variations). We were then unable to analyse in Chapter 5 the potential source-sink effects of area closures (Hansen, 2011; Ludford et al., 2012). Only, 20 fishing groups were spatially resolved in our application. For the other groups, we applied a fishing mortality derived from ICES stock assessments or survey analyses. Some of the stocks are distributed over a wider area than the EEC. For those stocks, we assumed that the partial fishing mortality was proportional to the ratio between catch inside the EEC compared to the catch observed over the entire stock distribution. Finally, discarding was not implemented dynamically in the model, and discarding patterns were considered constant for each fishing group and each functional group. This assumption might have artificially increased the influence of discards on the ecosystem.

In Chapter 5, we forced several processes such as the species price, total effort per fleet and per month, as well as the effort and catches outside the model domain. These assumptions could have been avoided by implementing exit-entry and priced elasticity models. We also showed the limits of using a statistical fleet dynamics model (RUM), fitted over few years, to provide forecast several decades ahead. Using more data to fit the RUM and/or implementing long-term behaviour explicitly might have improved the fit. Finally we did not have sufficient data to implement fishing costs. Considering the cost of fishing in our study could have changed the relative benefits of management scenario on the profit realised by the focused fleet.

However, despite the several assumptions needed to calibrate the ecosystem and fleet dynamics models, we did capture many insights into the functioning of the EEC ecosystem, improved knowledge on its interactions with fishing activities and on the benefits that could be expected from a combination of spatial and conservation management measures.

6.4. Perspectives

During this thesis project, we developed an ecosystem model, and coupled it with a fleet dynamics model, mainly focusing on the ecosystem around flatfish and the related fisheries. Several compartments of the ecosystem were not fully investigated, such as other human activities, long-term fishing behaviour, adaptive management, which would improve the current model version.

Importantly also, it was not possible, during the time allocated to this thesis, to perform an analysis of uncertainty and/or sensitivity of the model. Such analysis would have highlighted the key uncertain parameters for which a refined tuning would be necessary. An important step forward would hence be to identify and analyse the main sources of uncertainty in our model. Due to the numerous parameters considered in Atlantis, the application of sensitivity/uncertainty propagation analyses would require using, (i) meta-models (Grace et al., 2010), (ii) experimental plans to reduce the number of simulations based on, e.g. the Morris methods (Lehuta et al., 2013; Morris, 1991), Latin Hypercube Sampling (LHS, Gasche et al., 2013; Helton and Davis, 2003; McKay et al., 1979), (iii) sobol indices (Sobol', 2001) to explore the uncertainty in our application or, (iv) other technics such as adaptative screening already tested on Atlantis (Pantus, 2007). Adaptative screening allows modification of the experimental plans during the simulation process rather than the use of a static experimental plan established before the analysis. Another way to analyse the robustness and weakness of our application would be to compare the outcomes of our model with those from other EEC ecosystem models, when evaluating the same scenarios. An EwE EEC application has already been calibrated (Carpentier et al., 2009; Daskalov et al., 2011), and two other spatial ecosystem models are currently under development in this area, OSMOSE (Travers-Trolet, 2012) and ISISfish (Lehuta et al., in press).

In this study, only the 12-18m netters fleet was dynamically represented. A next step would be to apply a fleet dynamics model to each fishing group in Atlantis. This would allow the representation of interactions between fleets that were not considered in this study and a more complete understanding of fleet dynamics complexity.

The parameterization of density-dependence for some functional groups could also provide new insights on the potential effects of area closures. Similarly, the application of dynamic price and fisheries' entry-exit models would improve our understanding of fishers' behaviour under management constraints.

Atlantis could also be applied to analyse global change scenarios by, e.g., implementing temperature–dependence, and also habitat quality dependence that would relate species' distribution to the environmental status of the fishing grounds. Scenarios of climate change, increase of nutrient inputs from rivers or degradation of habitats due to human activities (aggregate extraction or fishing pressure) could be therefore be tested. In addition, by implementing dynamically the fleets impacted by maritime traffic (Chapter 2), we could investigate effects of maritime traffic increase on both fishing activities and the exploited ecosystem. Several Atlantis models were calibrated during the EU project VECTORS in addition to the Atlantis EEC, in the North Sea, the Baltic Sea and the Strait of Sicily. It is also planned to extend our application to the Western

English Channel. By combining some of these applications, larger scale nested analyses would be feasible, with the consideration of migration patterns for example across the existing interfaces. Atlantis EEC is already planned to be used in the next EU H2020 project DiscardLess. DiscardLess will develop practical, achievable, acceptable and cost-effective Discard Mitigation Strategies (DMS1) to either avoid or utilise unwanted catches, in order to reduce discards while maintaining viable fisheries. Discards are already implemented in the current version of the application and could be refined to evaluate the effects of a landing obligation (EC, 2014) on the ecosystem functioning and the resulting outcomes for the fishery. Finally, in the longer term, this model could be used to inform future management and be included in a formal MSE process for the EEC.

6.5. References

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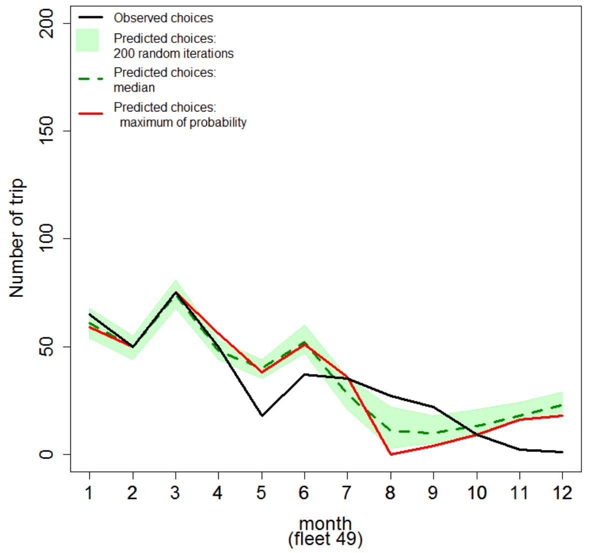
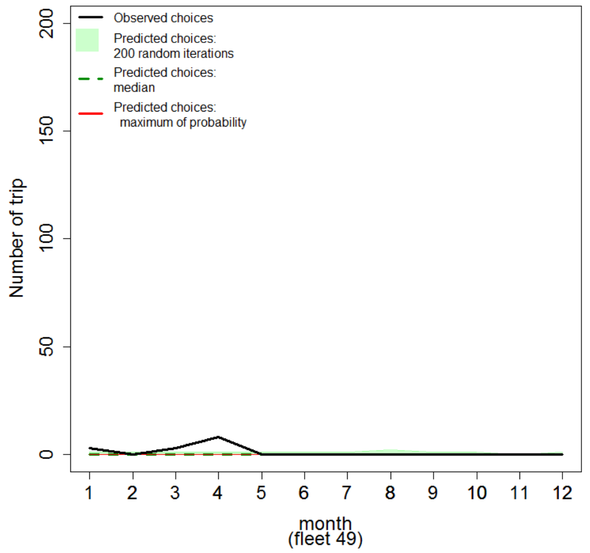
7. Appendix

Appendix I : Forecast of data in 2009 in number of trips per month for the focused fleet in Chapter 5, the 12-18m netters (FL49). Each graphic represent one alternative, the combinaition of statistical rectangle and métier. The dark line represents the observed choice in 2009, the red line represents the forecast based on the maximum of probability predictor, the green dotted line represents the median predictor derived from the 200 random iterations and the green area represents the range of predictors obtained with the 200 random iterations.

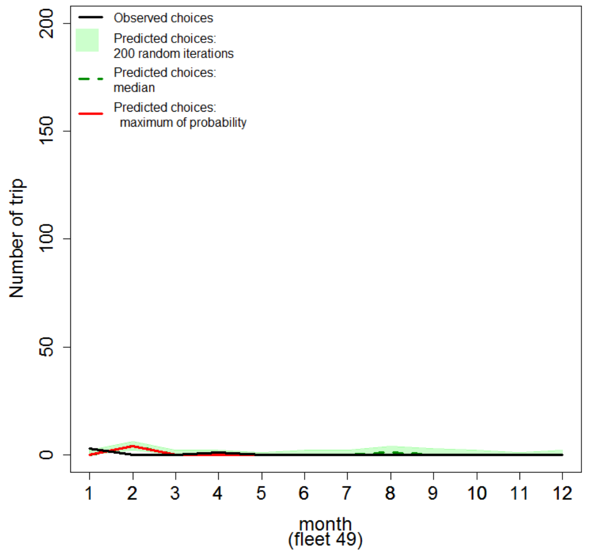
Number of trip for métier NOS34 in area ZZZZ : prediction vs observation in 2009



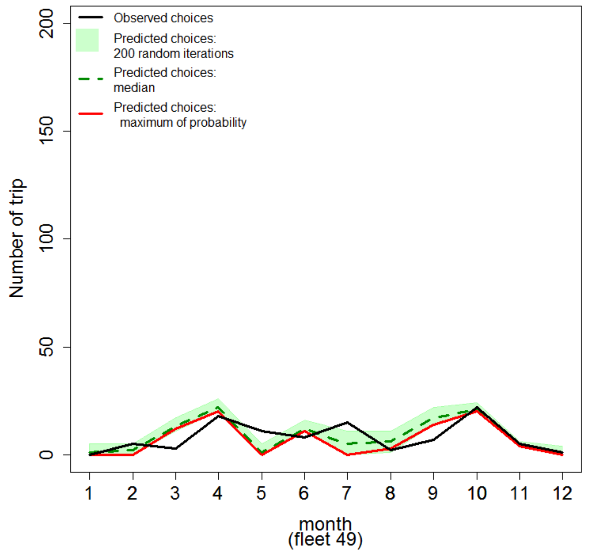
Number of trip for métier NOS34 in area 29E8 : prediction vs observation in 2009



Number of trip for métier NOS34 in area 29E9 : prediction vs observation in 2009

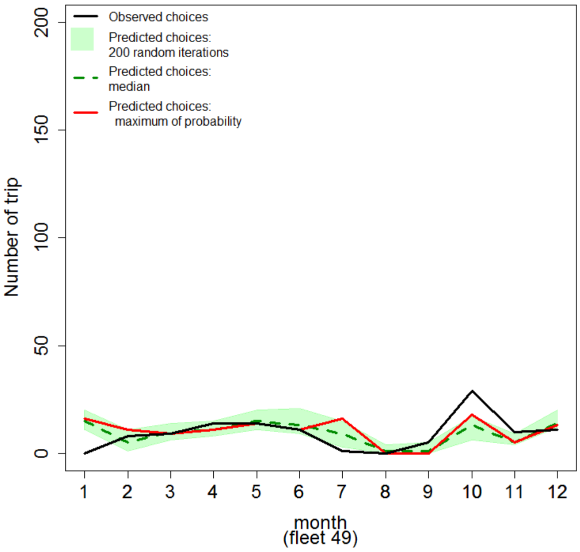


Number of trip for métier NOS34 in area 29F0 : prediction vs observation in 2009

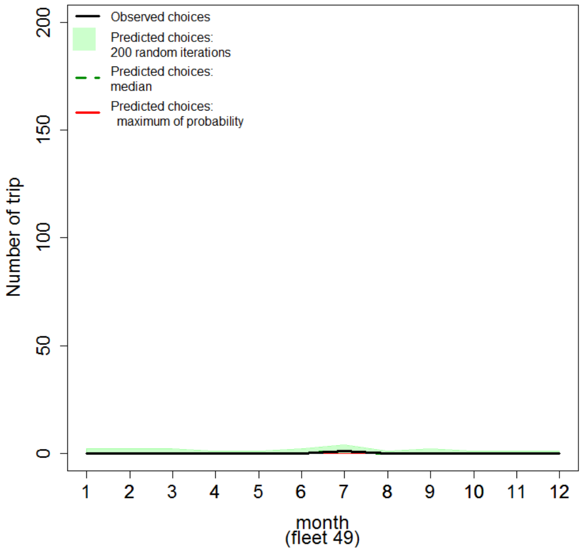


Appendix I (continued)

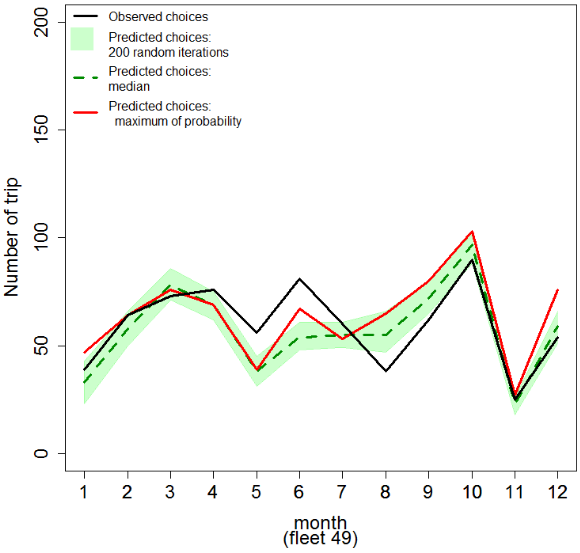
Number of trip for métier NOS34 in area 29F1 : prediction vs observation in 2009



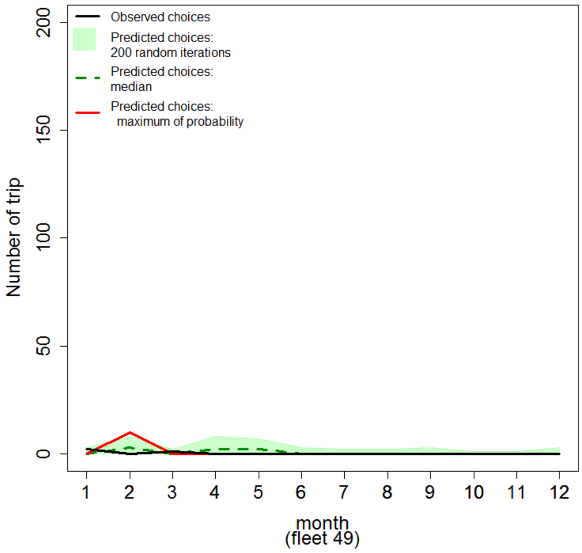
Number of trip for métier NOS34 in area 30F0 : prediction vs observation in 2009

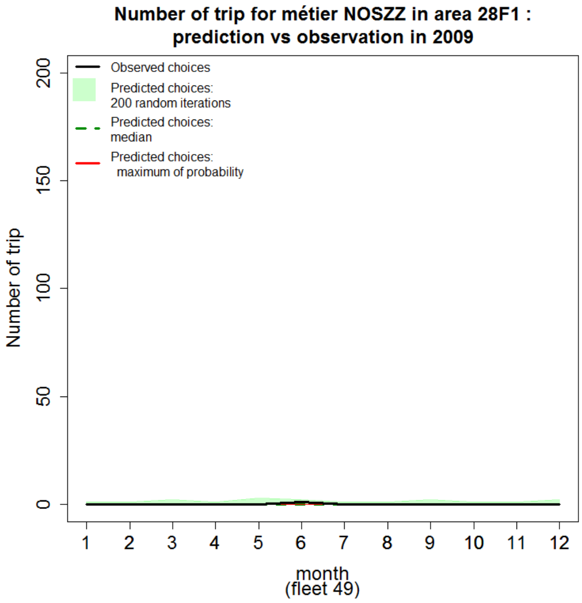


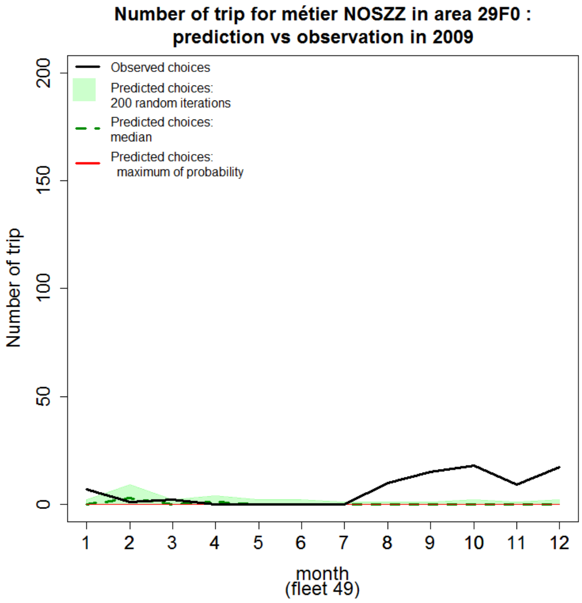
Number of trip for métier NOS34 in area 30F1 : prediction vs observation in 2009



Number of trip for métier NOSZZ in area 28F0 : prediction vs observation in 2009




Number of trip for métier NOSZZ in area 28F1 :
prediction vs observation in 2009
Observed choices
Predicted choices: 200 random iterations
Predicted choices: median
Predicted choices: maximum of probability
Number of trip
month
(fleet 49)


Number of trip for métier NOSZZ in area 29F0 :
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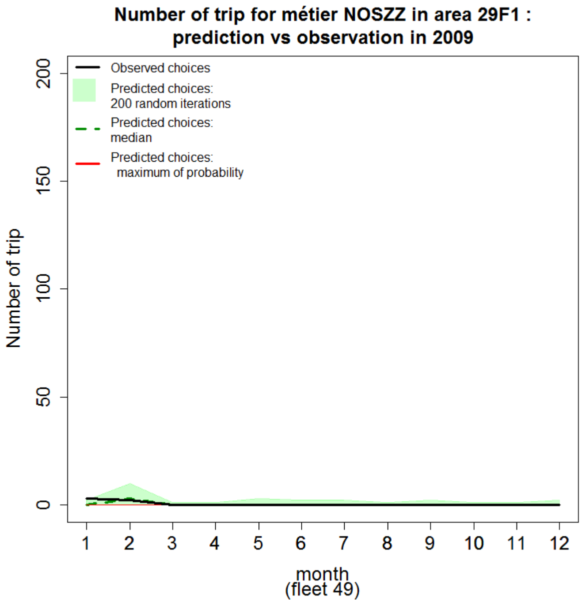
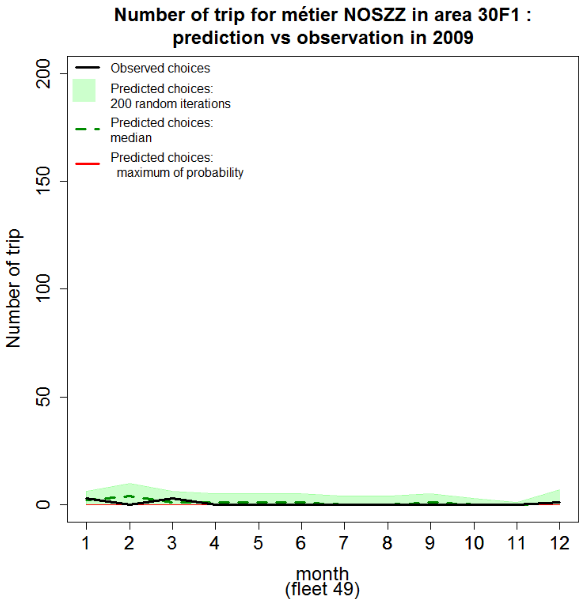
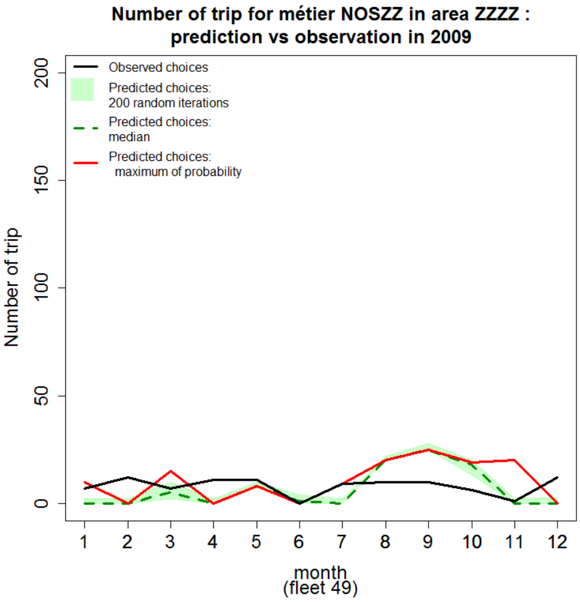
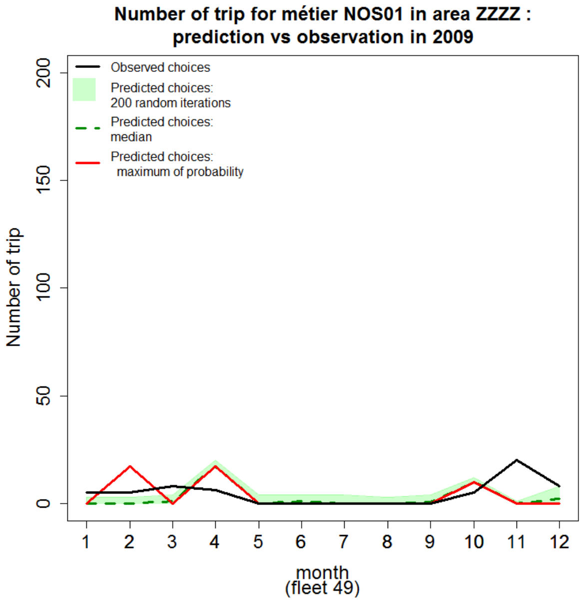
Observed choices
Predicted choices: 200 random iterations
Predicted choices: median
Predicted choices: maximum of probability
Number of trip
month
(fleet 49)

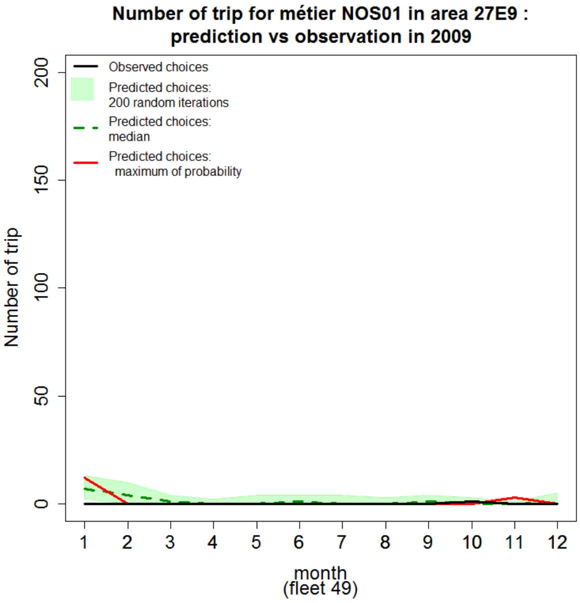

Number of trip for métier NOSZZ in area 29F1 :
prediction vs observation in 2009
Observed choices
Predicted choices: 200 random iterations
Predicted choices: median
Predicted choices: maximum of probability
Number of trip
month
(fleet 49)

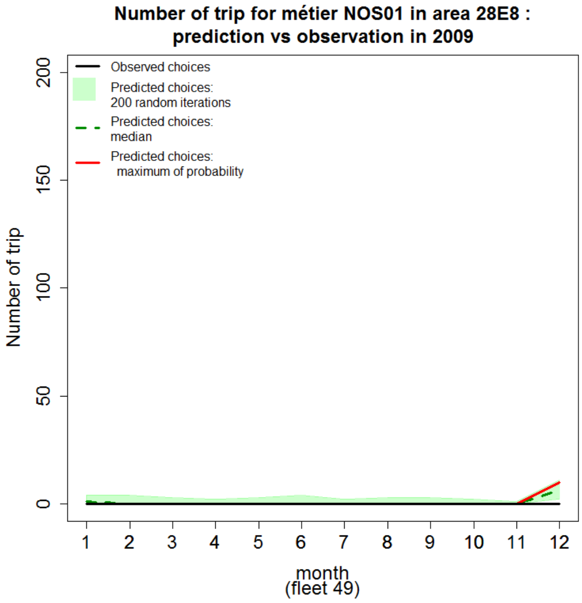

Number of trip for métier NOSZZ in area 30F1 :
prediction vs observation in 2009
Observed choices
Predicted choices: 200 random iterations
Predicted choices: median
Predicted choices: maximum of probability
Number of trip
month
(fleet 49)

Appendix I (continued)

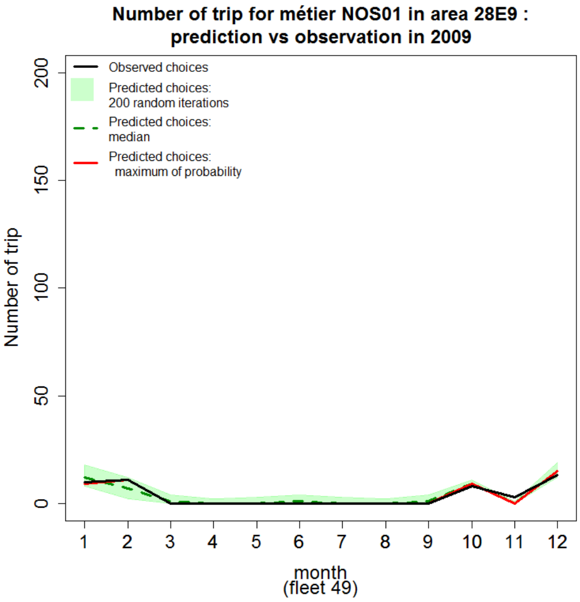


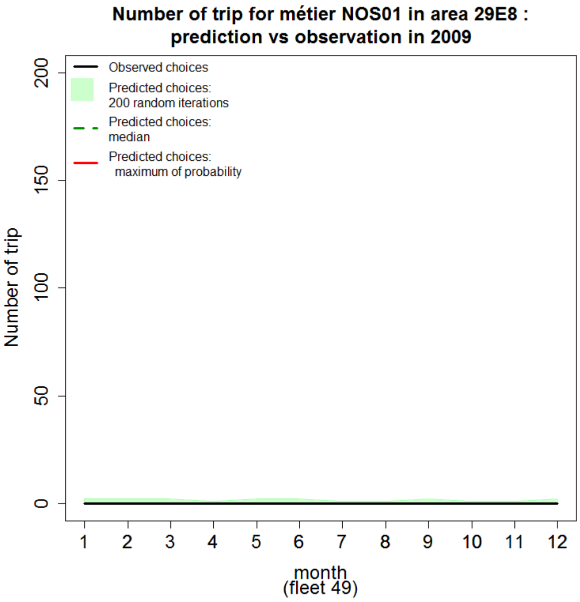


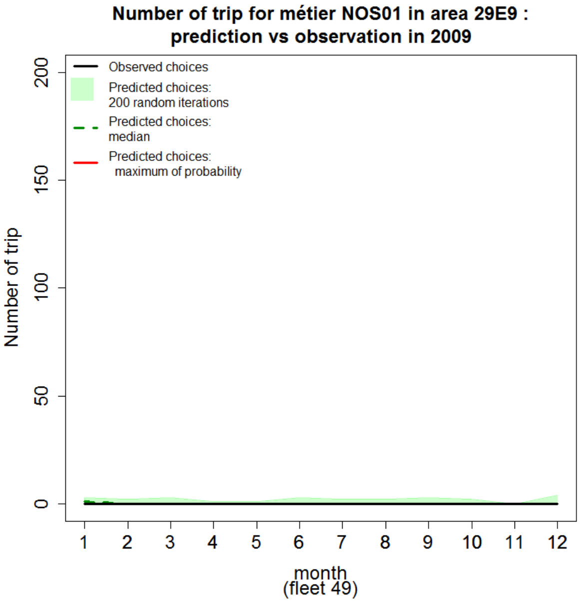


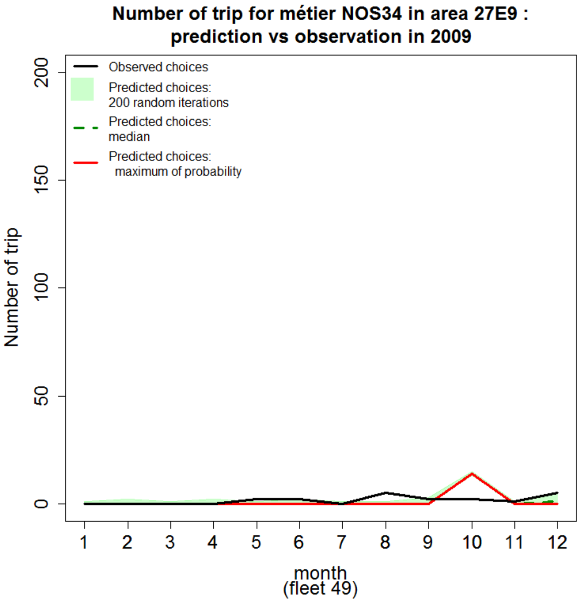


Appendix I (continued)

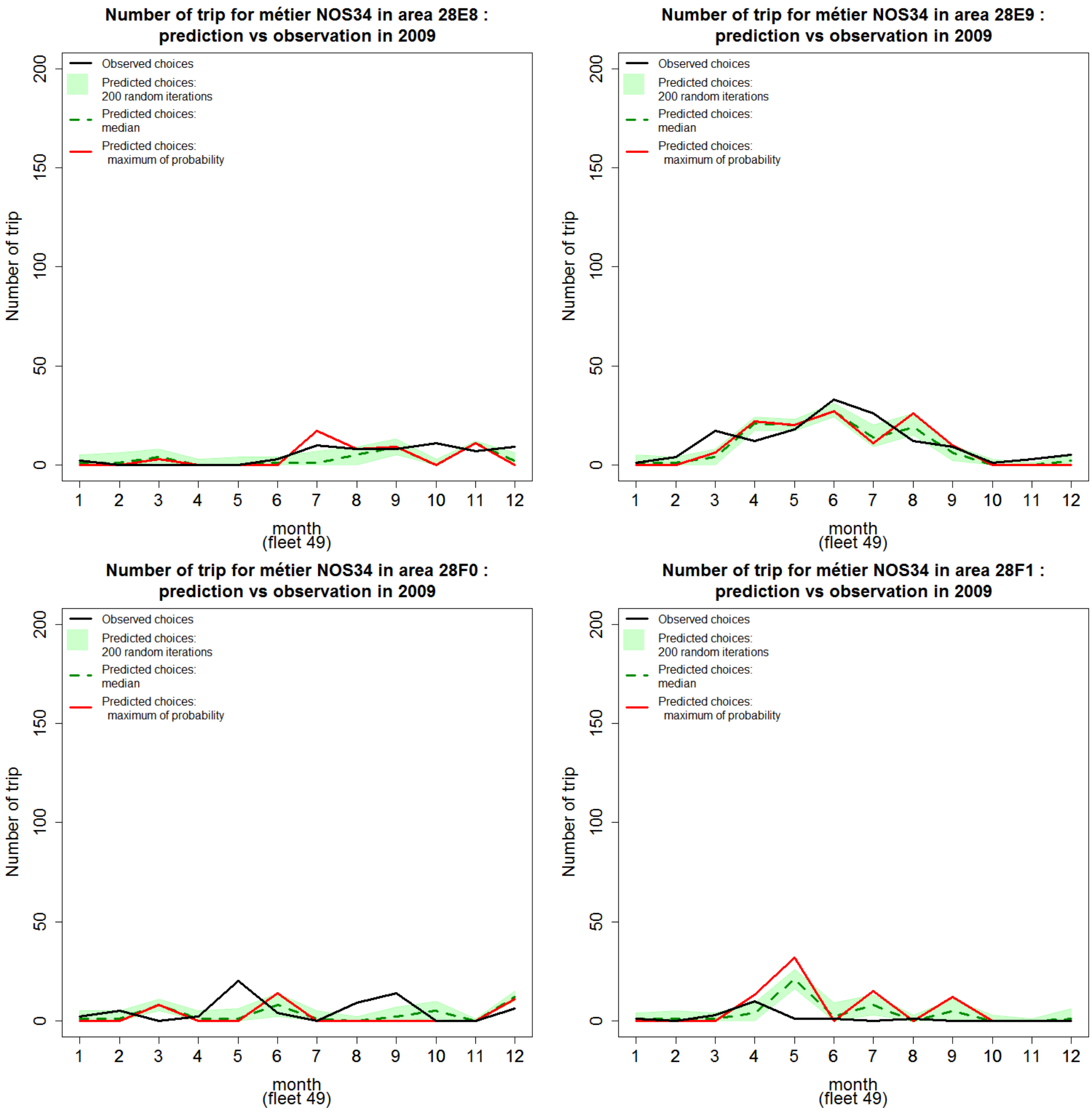








Appendix I (continued)



Appendix II : Vertebrates' biological parameters in Atlantis EEC of adults: spawning and migration periods in day of the year, recruitment in days after spawning and sediment types preferences.

Species	Spawning		Recruitment	Migration		Sediment types preference		
	start	Period (days)	Days after spawning	leave	return	pebble	gravel	sand
SB	90	100	200			x	x	x
CET	90	60	330			x	x	x
SXX	240	90	330			x	x	x
COD	330	180	100			x	x	
RAY	60	180	150			x	x	x
SHK	30	60	330			x	x	
WHG	0	180	45	120	304		x	x
POL	0	180	45				x	x
LBT	60	150	100			x	x	
BSS	120	50	46	273	90		x	x
SOL	40	160	35				x	x
PLE	330	120	100				x	x
DAB	30	90	35				x	x
OFF	30	150	35				x	x
MAC	90	150	40				x	x
CLU	320	90	100	62	274		x	x
SPA	150	110	30	304	120	x	x	
GUX	90	150	30				x	x
MUL	120	90	30				x	
GAD	31	120	30			x	x	x
SMD	0	365	30				x	x

Appendix III : Vertebrates' biological parameters in Atlantis EEC of juveniles: migration periods in day of the year and sediment types preferences.

Species	Migration		Sediment types preference		
	leave	return	pebble	gravel	sand
SB			x	x	x
CET			x	x	
SXX				x	x
COD				x	
RAY			x	x	x
SHK			x	x	
WHG				x	x
POL				x	x
LBT			x	x	
BSS					x
SOL					x
PLE					x
DAB					x
OFF					x
MAC				x	x
CLU	62	274		x	x
SPA					x
GUX			x	x	x
MUL				x	x
GAD				x	x
SMD				x	x

Appendix IV : Effort allocation in Atlantis EEC per polygons for each fishing class and the averaged daily effort per fishing class during the 1rst quarter. Data derived from logbooks effort over the periods 2002-2011. Fishing classes' description can be found in Table 4.6.

Fishing class	Daily effort (days)	Spatial allocation (proportion per polygons)																	
		0	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
FC1	2.64		4.7E-02	3.5E-02	1.3E-03	3.4E-04						1.5E-05		2.4E-04	2.1E-06	1.9E-02	2.5E-02	2.1E-02	6.7E-03
FC2	3.75		2.3E-02	9.9E-03													4.7E-04		1.6E-03
FC3	1.33		2.4E-02	2.5E-02												1.3E-04	3.8E-03	2.5E-02	1.2E-03
FC4	0.1																		
FC5	5.75			1.1E-04	2.7E-04	1.0E-05						6.0E-05		9.8E-04	8.6E-06	3.0E-03	6.4E-03	1.5E-05	
FC6	20.42		1.4E-02	4.6E-02	5.4E-03	5.3E-04		2.2E-05		1.4E-04	2.2E-05	4.3E-05	5.9E-05	4.0E-04	7.1E-05	4.5E-02	6.0E-02	4.3E-02	2.3E-03
FC7	21.67		2.0E-03	1.4E-03	1.4E-04	5.3E-06										6.8E-04	9.0E-04	8.2E-04	7.5E-04
FC8	5.38		2.8E-04	3.5E-03														3.8E-03	
FC9	1.58		1.5E-01	1.0E-01	1.5E-03	1.2E-04										9.4E-04	4.7E-04	4.6E-02	5.4E-03
FC10	0.27		1.9E-03	2.3E-02														2.6E-02	
FC11	0.64																		
FC12	3.95		3.2E-02	1.5E-02	2.6E-03	9.4E-04						2.3E-06		3.8E-05	3.3E-07	1.8E-03	2.9E-03	5.8E-03	1.5E-03
FC13	14.35		7.3E-03	4.9E-03								2.0E-05		3.3E-04	2.9E-06	1.6E-03	1.4E-03	1.9E-03	5.4E-03
FC14	16.84		3.9E-02	1.7E-02	1.8E-05	1.0E-04		4.0E-05		2.6E-04	4.0E-05	1.2E-05	1.1E-04	1.0E-04	2.1E-05	5.6E-04	8.3E-04	8.0E-04	3.3E-02
FC15	0.53		2.8E-01	1.2E-01	2.0E-02	2.5E-03						6.0E-05		9.8E-04	8.6E-06	6.2E-03	4.5E-03	7.4E-03	1.1E-02
FC16	31.71		7.2E-04	4.4E-03	3.0E-04	6.5E-05						1.1E-05		1.9E-04	1.6E-06	5.2E-03	7.2E-03	5.3E-03	3.9E-04
FC17	5.88		2.4E-02	1.1E-02	2.2E-03	7.6E-04						1.5E-05		2.4E-04	2.1E-06	4.4E-03	6.2E-03	5.3E-04	2.8E-02
FC18	0.86		3.7E-03	3.8E-02	6.3E-02	2.7E-03										4.6E-02	1.3E-02	3.7E-02	1.8E-04
FC19	11.14		1.1E-03	6.5E-03	2.3E-03	3.2E-04						7.5E-06		1.2E-04	1.1E-06	7.8E-03	1.0E-02	6.2E-03	8.8E-05
FC20	1.74		9.7E-05	1.2E-03												9.7E-03	1.4E-02	1.4E-03	

Appendix IV (continued)

Fishing class	Spatial allocation (proportion per polygons)																
	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	34
FC1	2.8E-02	1.8E-02	1.2E-02	4.1E-03	1.6E-02	1.1E-02	4.7E-04	4.3E-04	5.7E-04	5.0E-03	2.7E-03	2.6E-02	2.2E-02	1.2E-02	4.2E-02	5.6E-03	
FC2	2.3E-03	1.1E-03	8.5E-04	5.1E-05	1.7E-03	2.6E-04				3.4E-03	9.4E-04	2.8E-03	2.0E-03	8.5E-04	4.5E-03	4.1E-03	
FC3	1.8E-03	4.3E-03	3.6E-04	2.9E-03	1.7E-03	2.4E-03					9.5E-06	2.7E-02	4.6E-02	9.9E-03	8.3E-02		
FC4																	
FC5	1.3E-04	1.7E-04	6.5E-05	9.1E-06	3.2E-04	8.3E-04	1.9E-03	1.8E-03	2.3E-03	1.9E-03	1.1E-03	3.7E-02	3.1E-02	1.4E-02	5.8E-02	1.9E-03	
FC6	3.2E-02	2.8E-02	1.5E-02	7.9E-03	2.3E-02	2.2E-02	6.4E-04	5.8E-04	7.7E-04	2.2E-03	4.3E-03	8.2E-03	1.9E-03	1.1E-02	5.1E-03	1.4E-03	
FC7	5.8E-04	2.9E-04	8.6E-05	1.3E-04	4.0E-04	2.6E-04				7.2E-05	5.0E-05	1.1E-04	3.2E-05	2.6E-04	5.2E-04	8.8E-05	
FC8	3.5E-04	8.8E-04	2.0E-04	7.3E-04	2.0E-03	8.1E-04								1.1E-04			
FC9	1.8E-02	1.7E-02	4.5E-03	8.1E-03	3.5E-03	6.6E-03											
FC10	1.9E-03	5.9E-03	5.2E-04	4.5E-03	1.9E-03	3.7E-03											
FC11																	
FC12	2.4E-03	1.5E-03	8.1E-04	1.0E-03	4.3E-04	1.1E-03	7.5E-05	6.8E-05	9.0E-05	4.5E-04	2.8E-04	1.1E-02	9.2E-03	4.2E-03	1.7E-02	4.9E-04	
FC13	1.3E-02	6.7E-03	4.6E-03	8.6E-04	1.8E-02	3.2E-03	6.5E-04	5.9E-04	7.8E-04	1.9E-02	5.3E-03	1.2E-02	4.0E-03	2.9E-03	2.1E-02	2.3E-02	
FC14	2.9E-02	4.7E-03	4.5E-03	5.7E-04	1.3E-02	2.2E-03	1.8E-04	1.7E-04	2.2E-04	2.0E-03	7.2E-04	1.0E-03	3.5E-04	1.0E-03	1.8E-03	2.0E-03	
FC15	1.4E-02	6.4E-03	1.2E-03	1.3E-03	4.3E-04	1.1E-03	1.9E-03	1.8E-03	2.3E-03	1.5E-02	4.0E-03	5.6E-03	1.5E-03	4.1E-04	1.2E-02	1.7E-02	
FC16	1.2E-03	1.3E-03	8.7E-04	7.5E-04	6.1E-03	2.8E-03	3.7E-04	3.4E-04	4.4E-04	3.9E-02	1.1E-02	2.3E-02	6.0E-03	2.2E-03	3.7E-02	4.6E-02	
FC17	2.0E-02	7.2E-04	8.8E-04	3.6E-04	1.0E-02	2.9E-03	4.8E-04	4.4E-04	5.7E-04	1.1E-02	3.3E-03	6.9E-03	1.9E-03	1.9E-03	1.1E-02	1.4E-02	
FC18	2.7E-02	2.5E-02	1.1E-02	7.0E-03	3.0E-03	5.8E-03						1.3E-02	1.1E-02	4.6E-03	2.0E-02		
FC19	1.8E-03	2.6E-03	8.4E-04	9.5E-04	3.9E-03	3.2E-03	2.4E-04	2.2E-04	2.9E-04	2.9E-02	8.7E-03	2.2E-02	8.5E-03	4.0E-03	3.7E-02	3.6E-02	
FC20	6.2E-04	8.1E-04	9.5E-04	4.1E-04	1.2E-02	3.4E-03				9.4E-03	2.7E-03	1.9E-02	1.5E-02	7.2E-03	3.4E-02	1.1E-02	

Appendix V : Effort allocation in Atlantis EEC per polygons for each fishing class and the averaged daily effort per fishing class during the 2nd quarter. Data derived from logbooks effort over the periods 2002-2011. Fishing classes' description can be found in Table 4.6.

Fishing class	Spatial allocation (proportion per polygons)																	
	0	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
FC1		6.6E-03	7.1E-03	2.1E-04	8.0E-06						1.9E-05		3.2E-04	2.8E-06	2.0E-02	2.0E-02	5.9E-03	8.0E-03
FC2		2.0E-02	2.2E-02								2.2E-06		3.6E-05	3.1E-07	1.8E-02	1.8E-02	1.6E-02	5.4E-03
FC3		8.6E-04	4.1E-03												4.0E-03	9.2E-03	4.3E-03	4.0E-05
FC4																		
FC5		4.2E-03	7.2E-03	1.1E-04	4.3E-06						4.4E-05		7.3E-04	6.3E-06	5.1E-03	1.2E-02	8.5E-03	4.8E-03
FC6		4.2E-03	2.5E-02	5.1E-03	3.6E-04	1.2E-04	5.2E-05		1.7E-04	2.9E-05	4.1E-05	7.9E-05	3.0E-04	8.4E-05	2.9E-02	3.7E-02	2.5E-02	6.7E-04
FC7		1.6E-02	3.3E-02	7.1E-03	1.7E-03	6.8E-05	4.3E-04		2.5E-03	3.9E-04	5.9E-04	1.1E-03	2.5E-03	1.6E-03	2.6E-02	3.2E-02	2.6E-02	2.9E-03
FC8		7.2E-03	1.6E-02	9.6E-03	4.3E-04		8.0E-05		5.1E-04	8.1E-05	1.1E-04	2.2E-04	3.4E-04	3.2E-04	1.3E-02	1.6E-02	1.5E-02	7.4E-03
FC9		3.3E-02	2.3E-02	6.6E-04	2.5E-05										2.2E-02	2.7E-02	9.4E-03	1.8E-02
FC10		1.3E-02	4.9E-02	8.0E-03	2.9E-03										6.2E-03	8.8E-03	4.9E-02	9.4E-03
FC11															2.5E-02	2.5E-02		8.9E-02
FC12		6.0E-03	1.3E-02	2.8E-03	1.8E-03	5.4E-03	2.4E-03		1.1E-03	1.2E-04	8.7E-05	3.2E-04	3.2E-04	2.4E-04	9.5E-03	1.7E-02	1.1E-02	2.8E-02
FC13		7.0E-03	8.0E-03								3.2E-05		5.2E-04	4.6E-06	1.5E-03	2.1E-03	6.5E-03	9.3E-03
FC14		1.0E-01	4.5E-02	2.2E-04	8.2E-05						1.9E-06		3.2E-05	2.8E-07	2.2E-04	4.1E-04	3.4E-03	8.2E-02
FC15		7.1E-02	3.1E-02												5.3E-04	7.4E-04		2.6E-03
FC16		8.6E-04	7.6E-03	3.8E-04	1.6E-05						2.2E-05		3.6E-04	3.2E-06	6.5E-03	9.1E-03	8.2E-03	1.1E-03
FC17		4.8E-02	2.5E-02	3.3E-03	1.2E-03						4.4E-07		7.2E-06	6.3E-08	1.6E-02	2.4E-02	4.1E-03	3.5E-02
FC18		5.1E-04	6.4E-03														7.1E-03	
FC19		2.0E-03	1.4E-02	4.6E-03	3.6E-04						3.5E-06		5.8E-05	5.0E-07	1.1E-02	1.5E-02	1.3E-02	3.6E-04
FC20		5.5E-03	7.0E-03												5.0E-03	8.2E-03	5.3E-03	3.2E-03

Appendix V (continued)

Fishing class	Spatial allocation (proportion per polygons)																
	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	34
FC1	1.4E-02	6.2E-03	4.7E-03	1.2E-03	7.5E-03	6.1E-03	6.3E-04	5.7E-04	7.5E-04	3.6E-03	2.1E-03	1.9E-02	1.5E-02	9.4E-03	2.8E-02	3.8E-03	
FC2	4.4E-02	3.1E-02	1.8E-02	3.0E-03	8.4E-03	6.5E-03	7.1E-05	6.5E-05	8.6E-05	1.4E-02	3.8E-03	3.1E-02	2.3E-02	1.1E-02	4.7E-02	1.7E-02	
FC3	3.7E-03	3.0E-03	1.5E-03	7.5E-04	4.4E-04	1.3E-03				2.0E-02	3.0E-03	5.0E-02	6.0E-02	1.7E-02	1.2E-01	2.4E-02	
FC4																	
FC5	1.0E-02	6.2E-03	5.0E-03	1.8E-03	8.9E-04	1.9E-03	1.4E-03	1.3E-03	1.7E-03	9.4E-03	3.5E-03	7.2E-02	6.0E-02	2.6E-02	1.1E-01	1.2E-02	
FC6	8.0E-03	9.3E-03	3.7E-03	4.5E-03	1.2E-02	1.3E-02	4.2E-04	3.9E-04	5.1E-04	1.5E-03	2.7E-03	5.2E-03	1.4E-03	6.8E-03	3.5E-03	9.4E-04	
FC7	4.7E-02	3.5E-02	2.3E-02	4.9E-03	1.6E-02	1.2E-02	1.7E-03	1.5E-03	2.0E-03	8.6E-03	5.8E-03	1.1E-02	5.1E-03	8.5E-03	1.5E-02	7.2E-03	
FC8	2.3E-02	1.7E-02	1.1E-02	3.1E-03	2.7E-02	7.9E-03					8.5E-04	5.1E-03	3.0E-03	9.5E-03	1.3E-02		
FC9	1.4E-02	3.1E-03	9.7E-04	1.7E-03	1.8E-03	4.6E-03				1.4E-03	1.6E-03	9.9E-03	1.1E-02	6.7E-03	2.1E-02	1.7E-03	
FC10	3.1E-02	2.7E-02	1.2E-02	8.8E-03	1.1E-02	8.3E-03				2.0E-02	3.1E-03	2.5E-02	2.8E-02	4.7E-03	5.8E-02	2.5E-02	
FC11	4.0E-02	6.4E-03	8.1E-03		2.4E-03	8.0E-03				1.9E-03	1.4E-03	3.0E-02	3.9E-02	1.3E-02	7.2E-02	2.3E-03	
FC12	4.4E-02	2.0E-02	1.3E-02	2.3E-03	9.4E-03	5.2E-03	1.5E-04	1.3E-04	1.8E-04	3.3E-03	1.3E-03	4.0E-02	3.3E-02	1.5E-02	6.2E-02	3.9E-03	
FC13	4.0E-02	2.3E-02	2.1E-02	2.0E-03	3.4E-02	6.2E-03	1.0E-03	9.4E-04	1.2E-03	2.6E-02	7.3E-03	1.8E-02	6.3E-03	5.8E-03	3.1E-02	3.1E-02	
FC14	7.4E-02	1.9E-02	1.3E-02	1.2E-03	2.6E-02	4.2E-03	6.2E-05	5.7E-05	7.5E-05	2.5E-03	6.6E-04	2.5E-03	1.5E-03	2.3E-03	4.9E-03	3.0E-03	
FC15	3.5E-03	1.4E-03	4.5E-05	1.9E-05	6.9E-04	2.6E-04					3.8E-05	7.0E-05		1.5E-04			
FC16	4.8E-03	4.2E-03	2.7E-03	1.8E-03	1.6E-02	5.2E-03	7.1E-04	6.5E-04	8.6E-04	5.6E-02	1.7E-02	3.3E-02	9.4E-03	3.9E-03	5.5E-02	6.7E-02	
FC17	2.7E-02	2.2E-03	2.7E-03	1.6E-03	3.5E-02	1.0E-02	1.4E-05	1.3E-05	1.7E-05	1.0E-02	4.0E-03	1.6E-02	8.5E-03	9.5E-03	2.4E-02	1.3E-02	
FC18	5.1E-04	1.6E-03	1.4E-04	1.2E-03	5.4E-04	1.0E-03											
FC19	8.8E-03	7.7E-03	4.6E-03	2.9E-03	2.2E-02	7.9E-03	1.1E-04	1.0E-04	1.4E-04	4.2E-02	1.2E-02	3.3E-02	1.3E-02	8.5E-03	5.5E-02	5.1E-02	
FC20	1.9E-03	1.3E-03	2.5E-03	1.3E-03	3.1E-02	5.8E-03				2.8E-02	4.4E-03	2.6E-02	2.5E-02	1.0E-02	4.7E-02	3.4E-02	

Appendix VI : Effort allocation in Atlantis EEC per polygons for each fishing class and the averaged daily effort per fishing class during the 3rd quarter. Data derived from logbooks effort over the periods 2002-2011. Fishing classes' description can be found in Table 4.6.

Fishing class	Spatial allocation (proportion per polygons)																	
	0	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**
FC1		1.4E-02	8.3E-03													1.0E-04	5.6E-03	5.2E-03
FC2		4.6E-03	3.8E-02								4.9E-06		8.1E-05	7.1E-07	2.8E-02	3.0E-02	4.7E-02	1.2E-03
FC3															6.7E-03	8.3E-03		1.5E-01
FC4																		
FC5		1.1E-02	6.7E-03								4.1E-05		6.8E-04	5.9E-06	5.0E-03	1.1E-02	6.6E-03	3.1E-03
FC6		7.3E-03	3.9E-03	3.9E-04	1.3E-04						7.3E-07		1.2E-05	1.0E-07	3.2E-04	4.4E-04	8.1E-04	1.1E-03
FC7		2.1E-02	6.0E-02	1.5E-02	3.3E-03	3.7E-05	6.6E-04		4.1E-03	6.5E-04	9.1E-04	1.8E-03	4.3E-03	2.3E-03	4.6E-02	5.5E-02	5.1E-02	3.7E-03
FC8		8.8E-03	4.9E-02	1.8E-02	1.7E-03	1.9E-04	2.3E-04		1.3E-03	2.0E-04	4.2E-04	5.3E-04	3.2E-03	8.1E-04	8.5E-02	1.1E-01	4.7E-02	5.4E-03
FC9		5.0E-03	6.8E-03	5.5E-05	2.1E-06										3.5E-02	3.1E-02	5.3E-03	2.5E-02
FC10		3.7E-02	5.6E-02	3.5E-03	1.3E-03										2.7E-02	3.0E-02	4.5E-02	3.9E-03
FC11		1.5E-03	1.9E-02												6.4E-02	3.9E-02	2.1E-02	
FC12		9.8E-03	2.7E-02	3.2E-03	1.8E-03	3.9E-03	2.0E-03		2.6E-03	6.6E-04	9.2E-04	1.8E-03	3.0E-03	2.6E-03	1.0E-02	1.7E-02	2.6E-02	1.0E-02
FC13		1.5E-02	1.2E-02								1.6E-06		2.6E-05	2.3E-07	9.3E-04	1.4E-03	1.2E-02	1.5E-02
FC14		9.4E-02	4.1E-02	2.5E-04	6.7E-05										4.0E-04	2.3E-04	1.2E-03	4.5E-02
FC15															5.3E-04	7.4E-04		
FC16		9.3E-04	6.2E-03	5.8E-04	2.4E-05		4.0E-06		2.6E-05	4.0E-06	1.0E-05	1.1E-05	1.6E-04	3.7E-06	6.6E-03	9.4E-03	6.5E-03	9.1E-04
FC17		5.1E-02	2.7E-02	3.2E-03	1.2E-03						3.5E-06		5.8E-05	5.0E-07	7.6E-03	9.9E-03	8.3E-03	3.1E-02
FC18																		
FC19		3.7E-03	1.3E-02	6.6E-03	3.4E-04						1.4E-06		2.3E-05	2.0E-07	1.1E-02	1.4E-02	1.2E-02	7.9E-04
FC20		3.8E-04	6.9E-03	5.3E-03	2.0E-04										4.9E-03	6.1E-03	5.6E-03	

Appendix VI (continued)

Fishing class	Spatial allocation (proportion per polygons)																
	18	**19**	**20**	**21**	**22**	**23**	**24**	**25**	**26**	**27**	**28**	**29**	**30**	**31**	**32**	**33**	**34**
FC1	6.4E-03	2.3E-03	1.3E-03	9.2E-04	4.7E-03	1.3E-03				9.2E-05	2.5E-05	4.9E-04	4.1E-04	4.6E-04	7.6E-04	1.1E-04	
FC2	2.9E-02	2.5E-02	1.3E-02	7.8E-03	2.0E-02	1.4E-02	1.6E-04	1.5E-04	1.9E-04	1.8E-02	4.9E-03	4.7E-02	3.5E-02	1.9E-02	7.0E-02	2.8E-02	
FC3	5.4E-02	1.5E-03	1.8E-03		6.3E-04	2.1E-03					4.8E-04	3.3E-02	2.7E-02	1.2E-02	5.0E-02		
FC4										2.5E-01	6.9E-02	1.3E-01	2.7E-02		2.2E-01	3.0E-01	
FC5	5.0E-03	2.6E-03	4.3E-03	1.3E-03	1.8E-03	1.5E-03	1.3E-03	1.2E-03	1.6E-03	8.3E-03	2.5E-03	6.3E-02	5.2E-02	2.3E-02	9.8E-02	9.6E-03	
FC6	1.6E-03	7.2E-04	3.9E-04	1.6E-04	5.9E-04	2.6E-04	2.4E-05	2.2E-05	2.8E-05	4.9E-04	9.1E-05	3.4E-04	4.6E-04	1.6E-04	6.0E-04	5.7E-04	
FC7	6.0E-02	4.7E-02	3.1E-02	9.7E-03	3.6E-02	2.3E-02	3.6E-03	3.3E-03	4.4E-03	1.4E-02	1.1E-02	1.6E-02	6.6E-03	1.4E-02	2.0E-02	1.1E-02	
FC8	3.3E-02	2.8E-02	1.4E-02	7.9E-03	2.9E-02	3.3E-02	4.7E-03	4.3E-03	5.7E-03	1.0E-02	1.0E-02	2.4E-02	1.1E-02	2.4E-02	2.7E-02	7.9E-03	
FC9	2.8E-02	9.5E-03	6.3E-03	9.4E-04	1.9E-03	6.1E-03				2.2E-03	3.6E-03	2.7E-02	2.4E-02	1.4E-02	4.4E-02	2.6E-03	
FC10	4.2E-02	3.5E-02	1.7E-02	8.2E-03	1.2E-02	9.7E-03				1.1E-03	2.1E-03	3.6E-02	4.4E-02	1.1E-02	8.0E-02	1.3E-03	
FC11	7.2E-02	5.0E-02	3.0E-02	3.7E-03	6.8E-03	2.2E-02				8.8E-03	3.5E-03	3.0E-02	3.5E-02	1.5E-02	6.9E-02	1.1E-02	
FC12	5.7E-02	3.8E-02	2.4E-02	4.6E-03	2.1E-02	8.6E-03	5.2E-04	4.7E-04	6.2E-04	2.9E-03	1.1E-03	3.3E-02	2.9E-02	1.6E-02	5.7E-02	3.3E-03	
FC13	5.1E-02	3.1E-02	2.2E-02	2.0E-03	3.7E-02	6.4E-03	5.1E-05	4.7E-05	6.2E-05	2.1E-02	5.8E-03	1.7E-02	7.4E-03	7.1E-03	3.2E-02	2.6E-02	
FC14	4.1E-02	1.0E-02	6.9E-03	5.3E-04	1.2E-02	1.9E-03				2.3E-03	5.8E-04	1.2E-03	3.4E-04	8.7E-04	2.4E-03	2.8E-03	
FC15					5.0E-05	1.7E-04					3.8E-05	7.0E-05		1.2E-04			
FC16	6.1E-03	4.8E-03	4.3E-03	1.2E-03	1.5E-02	4.8E-03	3.1E-04	2.8E-04	3.7E-04	4.1E-02	1.2E-02	2.6E-02	7.6E-03	3.9E-03	4.2E-02	4.9E-02	
FC17	2.3E-02	2.2E-03	9.0E-04	1.1E-03	1.1E-02	4.1E-03	1.1E-04	1.0E-04	1.4E-04	7.5E-03	2.3E-03	7.2E-03	3.2E-03	3.0E-03	1.0E-02	9.1E-03	
FC18																	
FC19	7.0E-03	6.0E-03	3.5E-03	2.4E-03	1.5E-02	6.2E-03	4.6E-05	4.2E-05	5.6E-05	2.8E-02	8.2E-03	2.4E-02	1.1E-02	6.5E-03	3.9E-02	3.4E-02	
FC20	5.5E-03	5.6E-03	4.4E-03	1.9E-03	4.0E-02	7.0E-03				1.5E-02	3.4E-03	1.4E-02	1.0E-02	6.3E-03	2.7E-02	1.8E-02	

Appendix VII : Effort allocation in Atlantis EEC per polygons for each fishing class and the averaged daily effort per fishing class during the 4th quarter. Data derived from logbooks effort over the periods 2002-2011. Fishing classes' description can be found in Table 4.6.

Fishing class	Spatial allocation (proportion per polygons)																	
	0	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**
FC1		3.6E-02	3.7E-02	1.1E-03	9.1E-05						2.1E-05		3.4E-04	3.0E-06	2.8E-02	3.0E-02	2.7E-02	1.1E-02
FC2		2.2E-02	1.1E-02	9.5E-04	3.5E-04										3.4E-04	1.6E-03	3.1E-03	3.8E-03
FC3		1.8E-02	7.8E-03													1.4E-03		6.6E-04
FC4																		
FC5		3.9E-03	1.7E-03												8.1E-04	5.0E-03	6.9E-05	6.2E-04
FC6		1.5E-02	5.9E-02	5.5E-03	4.4E-04		4.3E-05		2.8E-04	4.4E-05	4.2E-05	1.2E-04	3.2E-04	8.2E-05	3.6E-02	4.6E-02	5.8E-02	2.9E-03
FC7		6.9E-03	7.4E-03												7.8E-04	1.3E-03	5.0E-03	1.7E-03
FC8		1.1E-02	4.1E-02												3.2E-03	4.5E-03	4.1E-02	1.2E-03
FC9		4.4E-02	3.7E-02														2.9E-02	6.1E-03
FC10		2.7E-04	3.4E-03														3.7E-03	
FC11																		
FC12		1.3E-02	8.6E-03		4.1E-04	2.7E-04	2.3E-04		7.2E-05						1.4E-03	3.5E-03	3.2E-03	7.7E-03
FC13		1.1E-02	6.9E-03								1.9E-05		3.2E-04	2.7E-06	7.5E-04	1.1E-03	3.9E-03	7.8E-03
FC14		6.5E-02	2.9E-02								1.9E-06		3.1E-05	2.7E-07	2.1E-04	2.1E-04	1.5E-03	2.5E-02
FC15		1.1E-01	5.4E-02	2.2E-03	4.5E-04										2.0E-02	2.8E-02	1.4E-02	8.8E-03
FC16		9.9E-04	3.8E-03	8.8E-04	6.2E-05	4.8E-05	4.1E-05		1.3E-05		2.0E-05		3.3E-04	2.9E-06	7.0E-03	9.5E-03	3.7E-03	8.2E-04
FC17		3.7E-02	1.7E-02	2.6E-03	9.6E-04						1.2E-05		1.9E-04	1.6E-06	1.2E-02	1.6E-02	2.4E-03	3.1E-02
FC18		1.1E-02	7.7E-02	7.1E-02	2.7E-03						1.4E-06		2.3E-05	2.0E-07	5.8E-02	2.2E-02	7.7E-02	4.7E-03
FC19		2.1E-03	4.5E-03	1.5E-04	5.9E-06						1.0E-06		1.7E-05	1.5E-07	8.1E-03	1.1E-02	4.1E-03	1.6E-03
FC20		1.6E-03	3.9E-02	8.3E-02	3.2E-03										6.0E-02	4.4E-02	2.0E-02	

Appendix VII (continued)

Fishing class	Spatial allocation (proportion per polygons)																
	18	**19**	**20**	**21**	**22**	**23**	**24**	**25**	**26**	**27**	**28**	**29**	**30**	**31**	**32**	**33**	**34**
FC1	4.9E-02	3.2E-02	1.9E-02	4.9E-03	9.9E-03	1.2E-02	6.7E-04	6.1E-04	8.1E-04	5.2E-03	3.2E-03	2.2E-02	1.5E-02	1.1E-02	3.1E-02	5.9E-03	
FC2	9.8E-03	5.4E-03	4.1E-03	5.4E-04	1.9E-04	3.7E-04				1.6E-03	2.4E-04	8.6E-03	7.2E-03	3.9E-03	1.3E-02	2.0E-03	
FC3	8.9E-04	3.7E-04										9.9E-03	8.5E-03	3.7E-03	1.5E-02		
FC4																	
FC5	1.8E-03	9.3E-04	2.1E-03	1.2E-05	7.7E-05	2.6E-04				1.7E-03	3.3E-04	3.3E-02	2.8E-02	1.2E-02	5.2E-02	2.1E-03	
FC6	3.8E-02	3.5E-02	1.7E-02	1.1E-02	2.1E-02	2.0E-02	4.8E-04	4.3E-04	5.7E-04	2.1E-03	3.5E-03	7.2E-03	2.2E-03	9.2E-03	5.6E-03	1.4E-03	
FC7	6.3E-03	4.1E-03	2.9E-03	1.0E-03	3.5E-03	1.2E-03					5.6E-05	1.5E-03	3.7E-03	6.5E-04	6.7E-03		
FC8	2.3E-02	2.2E-02	8.8E-03	7.3E-03	2.6E-03	5.1E-03				1.9E-04	2.6E-04	4.7E-04	2.0E-05	7.0E-04	1.6E-04	2.3E-04	
FC9	7.5E-03	7.0E-03	1.5E-03	5.1E-03	2.2E-03	4.1E-03											
FC10	4.4E-03	3.2E-03	1.8E-03	6.6E-04	2.8E-04	5.3E-04											
FC11	6.5E-02	4.4E-02	2.8E-02														
FC12	6.5E-03	2.0E-03	1.1E-03	6.0E-04	8.6E-04	7.6E-04				3.7E-04	1.3E-04	1.7E-02	1.5E-02	6.5E-03	2.7E-02	4.5E-04	
FC13	3.2E-02	1.8E-02	1.3E-02	1.2E-03	2.4E-02	4.0E-03	6.2E-04	5.7E-04	7.5E-04	2.1E-02	6.0E-03	1.5E-02	5.9E-03	4.6E-03	2.7E-02	2.5E-02	
FC14	2.5E-02	7.5E-03	5.9E-03	5.9E-04	1.4E-02	2.2E-03	6.2E-05	5.6E-05	7.4E-05	3.4E-03	9.6E-04	2.4E-03	9.2E-04	1.3E-03	4.5E-03	4.1E-03	
FC15	1.3E-02	6.2E-03	4.9E-03	2.6E-03	6.5E-03	6.8E-03				1.1E-02	3.3E-03	9.8E-03	6.6E-03	6.0E-03	2.0E-02	1.4E-02	
FC16	5.5E-03	4.0E-03	3.4E-03	6.0E-04	9.2E-03	3.6E-03	6.5E-04	6.0E-04	7.9E-04	4.3E-02	1.3E-02	2.5E-02	6.3E-03	2.9E-03	4.0E-02	5.1E-02	
FC17	2.7E-02	3.7E-03	2.6E-03	5.6E-04	1.4E-02	5.9E-03	3.7E-04	3.4E-04	4.5E-04	2.3E-02	7.4E-03	1.8E-02	7.1E-03	5.3E-03	2.7E-02	2.8E-02	
FC18	6.0E-02	5.5E-02	2.4E-02	1.5E-02	6.5E-03	1.2E-02	4.6E-05	4.2E-05	5.5E-05	9.3E-05	7.4E-04	4.1E-02	3.5E-02	1.5E-02	6.3E-02		
FC19	3.0E-03	2.2E-03	1.7E-03	9.5E-04	1.3E-02	4.6E-03	3.3E-05	3.0E-05	4.0E-05	3.3E-02	9.7E-03	2.6E-02	1.0E-02	5.6E-03	4.2E-02	4.0E-02	
FC20	1.1E-02	1.3E-02	5.1E-03	2.8E-03	1.8E-02	1.2E-02				1.8E-02	5.3E-03	1.6E-02	8.2E-03	8.1E-03	2.7E-02	2.2E-02	

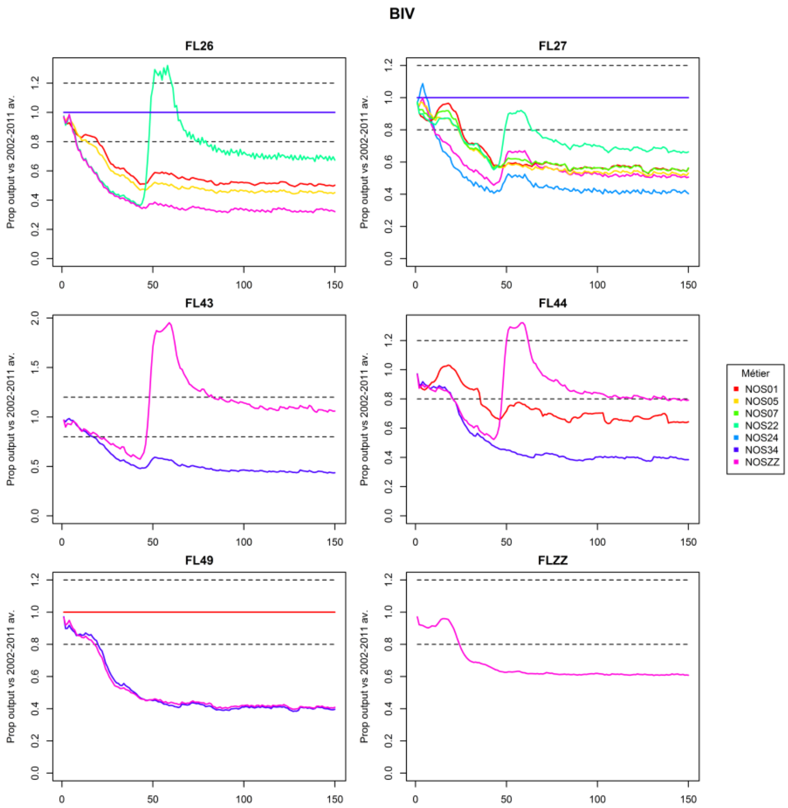
Appendix VIII : Catchability used in Atlantis EEC per functional groups and fishing classes. Description of functional groups is provided in Table 4.1 and of fishing classes in Table 4.6.

Fishing class	Functional group												
	COD	RAY	SHK	WHG	POL	LBT	BSS	SOL	PLE	DAB	OFF	MAC	CLU
FC1	5.6E-01	6.9E+01	3.3E+02	1.0E+01	3.8E+00	2.5E+01	1.1E+01	4.8E+01	1.3E+02	3.8E+01	2.5E+01	1.6E+01	2.5E+01
FC2	1.5E+00	1.5E+02	1.3E+03	2.5E+01	3.7E+01	4.7E+00	2.3E+01	5.8E+01	1.7E+02	2.3E+02	3.3E+01	5.9E+01	9.5E+00
FC3	1.5E-02	4.1E+01	7.3E+02	5.3E-02	4.1E+00		3.0E-01	4.4E+01	4.3E+01	3.4E+01	1.1E+01	1.4E-01	
FC4			1.8E+07					8.0E+02	1.1E+03	3.1E+01	9.5E+01	6.8E+00	
FC5	1.0E-01	3.4E+01	1.9E+03	1.0E+02	2.2E+01	2.5E+02	1.4E+01	1.2E+03	7.6E+02	1.5E+04	1.4E+02	4.2E+01	1.5E+03
FC6	1.1E+00	1.7E+01	1.3E+02	1.6E+00	1.6E+01	8.6E+00	1.9E+01	2.8E+01	8.7E+01	7.7E+00	2.0E+01	4.4E+00	5.4E+00
FC7	3.8E+00	1.6E+02	1.5E+03	1.4E+01	4.8E+01	1.1E+01	5.9E+01	6.3E+01	6.3E+01	1.1E+02	1.3E+01	3.9E+01	2.3E+00
FC8	8.6E+00	1.4E+02	9.5E+02	3.1E+00	2.0E+01	1.6E+01	2.6E+01	7.2E+00	3.3E+01	9.2E+01	1.3E+01	4.8E+00	7.2E-01
FC9	1.9E-01	1.9E+02	5.3E+02	1.0E+01	9.7E+00	9.2E+00	2.0E+00	1.4E+02	1.3E+02	2.8E+01	1.6E+01	3.0E+01	2.1E+01
FC10	6.6E-02	1.8E+02	4.3E+02	7.9E+01	2.7E+01	4.6E+02	3.7E+01	3.1E+01	2.0E+01	4.1E+01	6.7E+00	1.8E+03	5.5E+03
FC11	6.3E+00	1.8E+00	7.5E+12	1.2E+00	7.8E+01	3.5E-02	2.0E+01	3.8E+02	2.4E+02	5.5E+01	1.2E+02	3.9E-01	
FC12	1.9E-01	7.5E+01	1.0E+03	9.0E+00	7.1E+00	1.4E+02	7.6E+00	1.3E+02	1.2E+02	5.5E+01	2.9E+01	8.6E+01	7.0E+01
FC13	2.6E+01	3.3E+01	3.1E+12	1.5E+00	8.7E+03	7.1E-01	3.3E+02	4.9E+02	9.2E+02	5.5E+01	1.0E+02	2.1E+00	1.3E+01
FC14	9.5E+00	3.9E+02	1.7E+03	2.7E+00	2.7E+02	1.2E+02	2.0E+02	9.9E+00	1.2E+01	2.0E+01	3.6E+00	3.4E+00	2.6E+01
FC15	5.7E-02	1.1E+01	3.1E+00	2.4E+02		7.6E+00	7.8E-01	3.7E+00	2.6E+01	1.7E+01	1.3E+01		4.0E+00
FC16	3.1E+01	2.7E+01	3.6E+12	4.1E+00	1.0E+05	1.3E+00	4.9E+02	1.6E+03	1.9E+03	2.0E+02	2.9E+02	1.9E+00	4.5E+01
FC17	3.7E+01	6.3E+02	4.4E+03	2.2E+01	8.9E+01	1.3E+03	7.1E+01	6.8E+01	4.2E+01	2.6E+02	1.2E+01	2.3E+00	1.1E+02
FC18	2.4E-01	1.9E+01	1.3E+02	5.5E-02	9.9E+00	1.6E+00	1.6E+00	1.1E+01	4.5E+01	4.5E+00	1.7E+01		
FC19	2.0E+01	1.4E+01	8.8E+12	5.1E+00	1.8E+03	8.7E-01	1.3E+02	5.8E+02	1.5E+03	2.0E+02	1.2E+02	6.6E-01	2.7E+01
FC20	1.7E+01	1.4E+02	1.1E+03	1.3E+01	9.9E+01	1.3E+01	6.5E+01	1.5E+02	1.5E+02	2.4E+02	2.6E+01	2.9E-01	6.0E+01

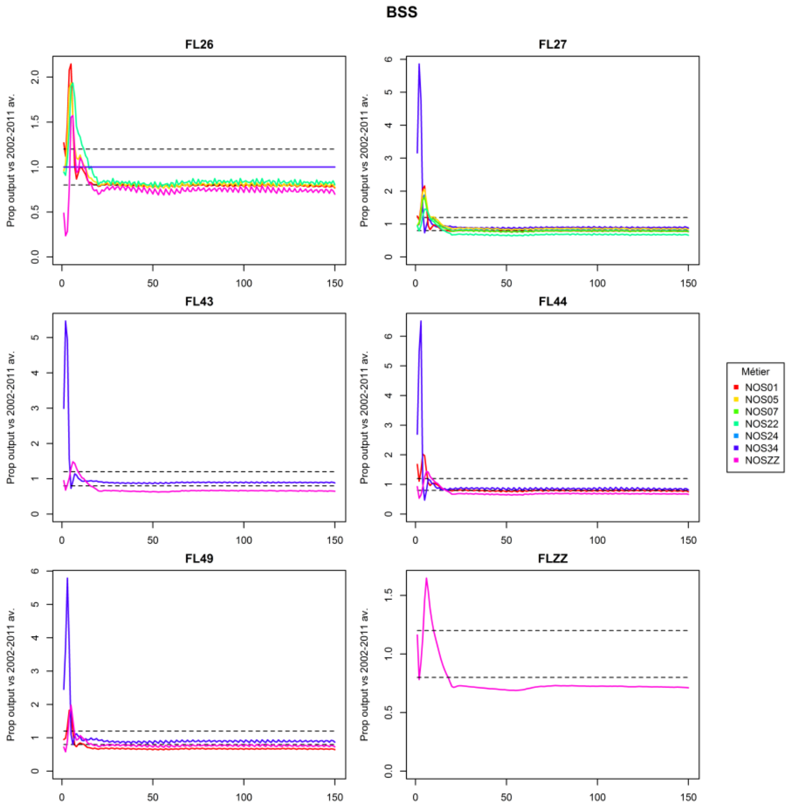
Appendix VIII (continued)

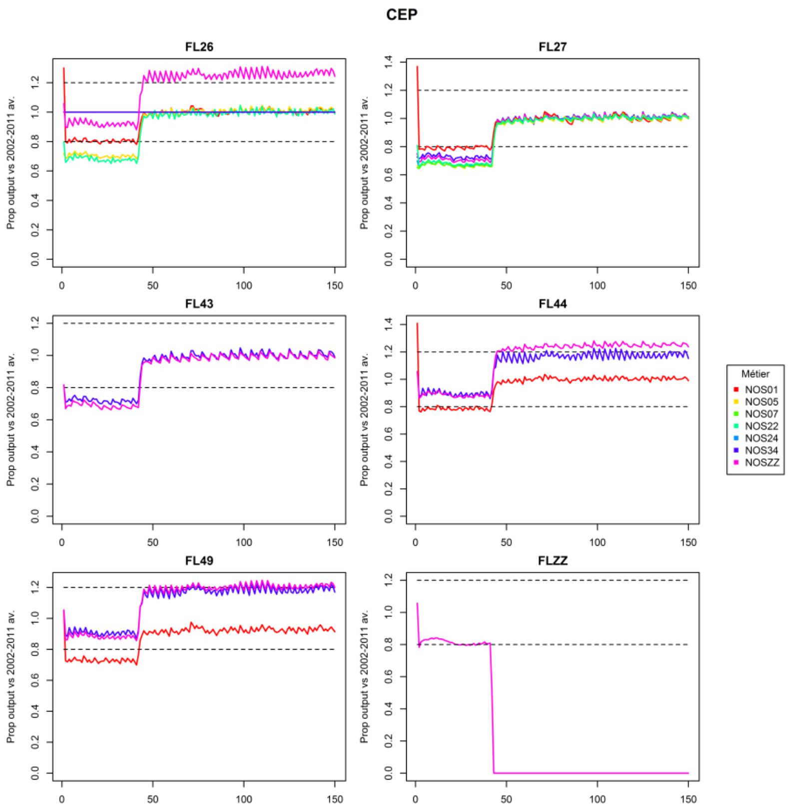
Fishing class	Functional group											
	SPA	GUX	MUL	GAD	SMD	CEP	CRA	LBE	SHP	WHE	SCE	BIV
FC1	5.0E+00	5.9E+00	2.7E-02	3.6E+00	3.0E+01	4.2E+11	1.3E+10		1.6E+09	1.6E+10	1.5E+11	6.7E+10
FC2	4.6E+01	4.1E+01	3.8E-01	1.6E+01	2.4E+01	3.9E+05	4.5E+04	3.7E+04	7.9E+02	3.0E+04	1.2E+03	1.2E+04
FC3	6.2E-01	1.8E+01	2.6E-03	5.1E+00	4.7E+00	1.6E+07	1.2E+07		3.9E+04			1.9E+06
FC4				8.2E-02								
FC5	3.4E+01	1.8E+02	4.3E-01	9.3E+01	2.2E+03	7.7E+06	4.2E+05		1.4E+05	1.3E+06	1.9E+05	4.0E+04
FC6	2.1E+01	3.3E+00	1.4E-02	2.7E+00	1.3E+00	5.0E+10	4.6E+08	7.3E+07	5.0E+04	6.1E+07	2.8E+10	2.1E+09
FC7	2.3E+02	4.3E+01	1.9E-01	1.7E+01	1.5E+01	2.9E+04	1.1E+03	1.5E+02	9.5E+00	1.3E+02	5.6E+01	5.1E+02
FC8	3.9E+02	1.5E+02	1.9E-01	1.5E+01	4.1E+00	3.5E+04	4.7E+02	2.3E+01			1.3E+01	8.6E+01
FC9	6.1E+00	8.3E+01	3.4E-01	1.9E+01	1.5E-01	8.4E+05	1.0E+04				1.4E+04	1.4E+05
FC10	1.6E+02	2.3E+01	4.7E-01	5.3E+01	5.5E+00	4.4E+03	7.1E+02	2.6E+04			1.6E+03	2.9E+03
FC11	3.2E-01	5.0E-02	2.3E-02	6.9E-03	3.7E+00	4.4E+03		2.4E+05			1.0E+03	
FC12	5.7E+01	1.8E+01	5.0E-02	2.9E+00	9.7E+00	3.6E+05	3.1E+04	4.0E+04		7.8E+05	9.7E+03	2.4E+04
FC13	1.2E+00	1.1E-01	7.4E-02	1.2E-01	3.0E+00	1.2E+04	8.0E+03	6.8E+05	1.4E+03	2.3E+03	5.8E+02	1.2E+03
FC14	3.2E+01	1.3E+00	4.5E-01	2.4E+00	2.4E+09	2.5E+25	9.6E+25	6.8E+25	3.1E+22	2.1E+26	6.4E+21	4.9E+22
FC15		2.0E-01	3.5E-03	6.5E-01		1.1E+10	4.6E+08			6.2E+09	3.6E+09	6.5E+07
FC16	3.6E-01	2.7E-01	2.1E-02	3.1E-01	7.9E+03	2.6E+13	2.4E+12	5.3E+14	6.3E+10	8.7E+10	8.3E+11	9.6E+09
FC17	6.6E+00	1.8E+01	4.5E-02	4.2E+01	1.8E+01	3.0E+04	1.3E+05	2.5E+05	1.2E+01	5.0E+05	1.7E+01	2.4E+00
FC18	3.6E-01	9.6E-01	2.8E-04	8.9E-01	2.1E-02	1.2E+09	1.0E+07				7.0E+08	
FC19	2.0E-01	2.1E-01	1.1E-02	2.8E-01	2.2E+02	1.5E+06	3.5E+05	2.5E+07	1.3E+04	2.3E+03	1.2E+06	2.0E+03
FC20	6.6E+00	8.7E+00	1.1E-02	3.4E+01	2.7E+01	1.1E+04	1.3E+10	3.2E+03	1.6E+09	8.0E+04		6.8E-01

Appendix IX : Estimation of the performance of the model on catch output from Atlantis EEC over 150 years of simulation for each functional group. Catches of each combination of fleet and métier forecast by Atlantis are compared to logbook and discard data over 2002-2011. The black dotted lines represent the range of acceptance (i.e. 20% around that level). Details on functional groups' codes can be found in Table 4.1 and on fleets and métiers codes in Table 1.1.

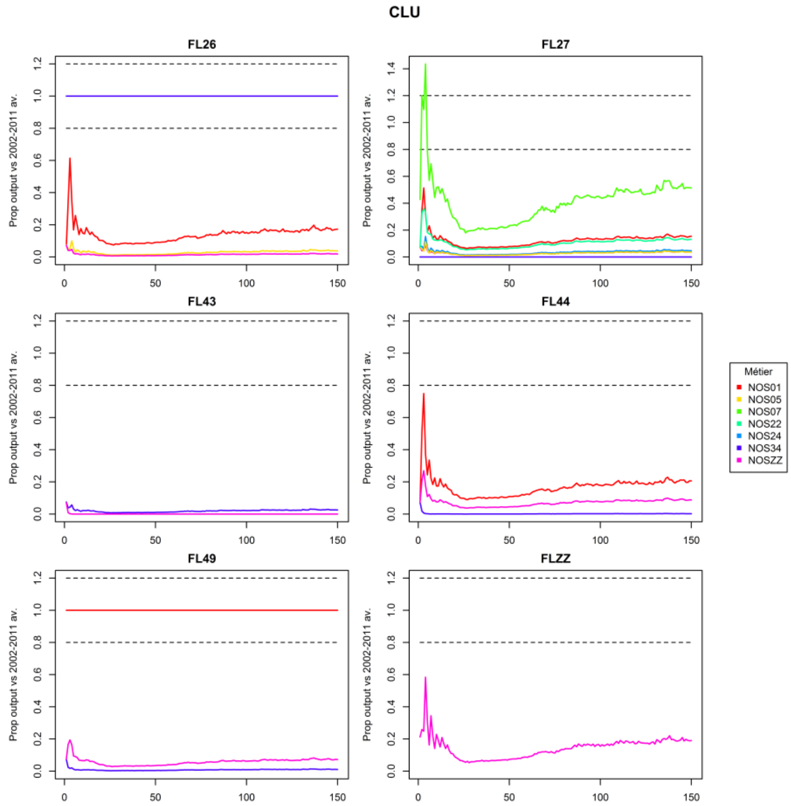


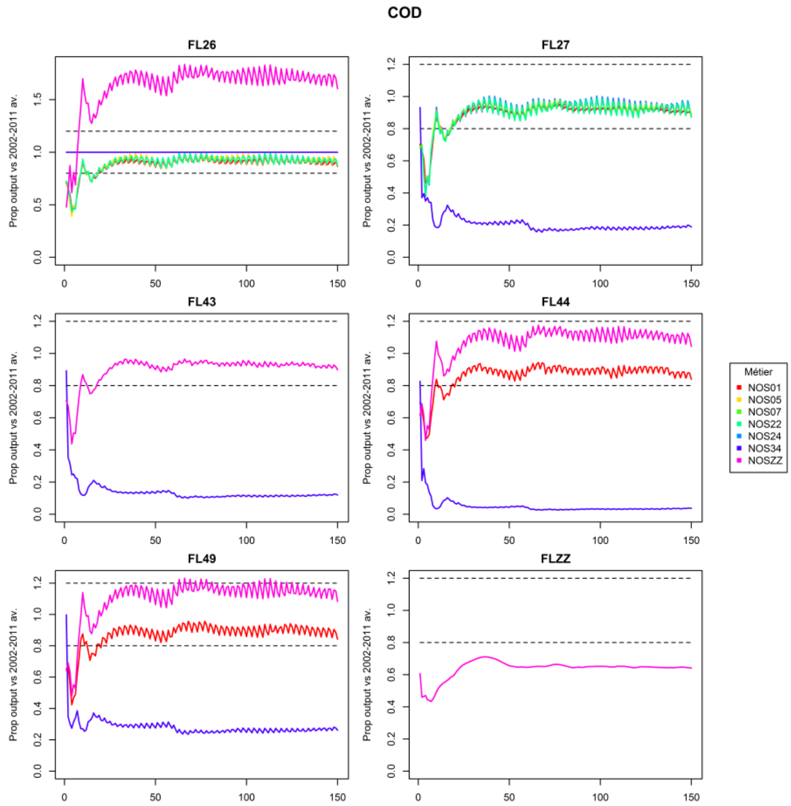
Appendix IX (continued)



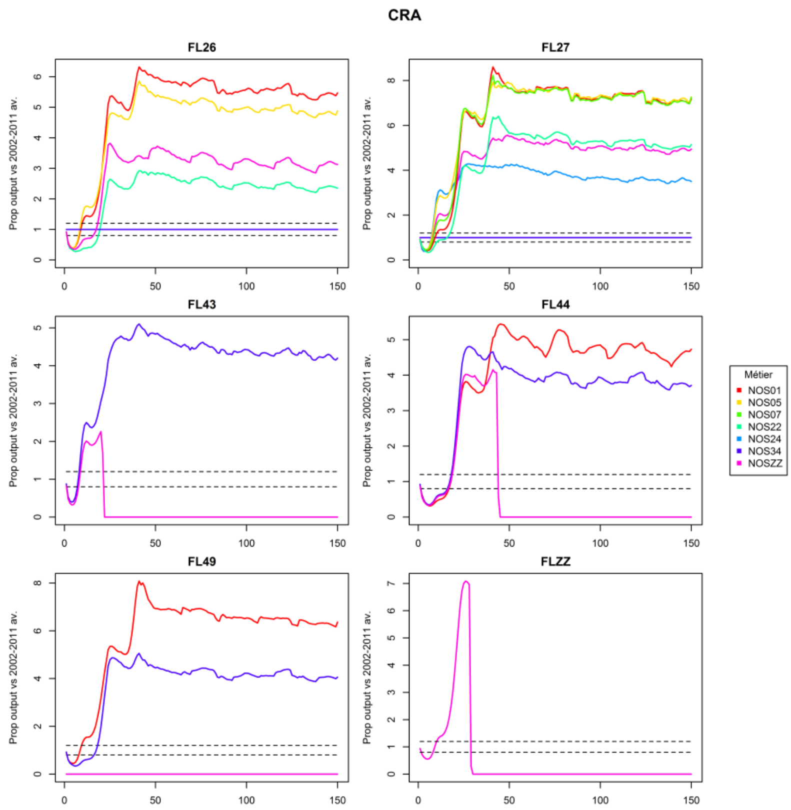


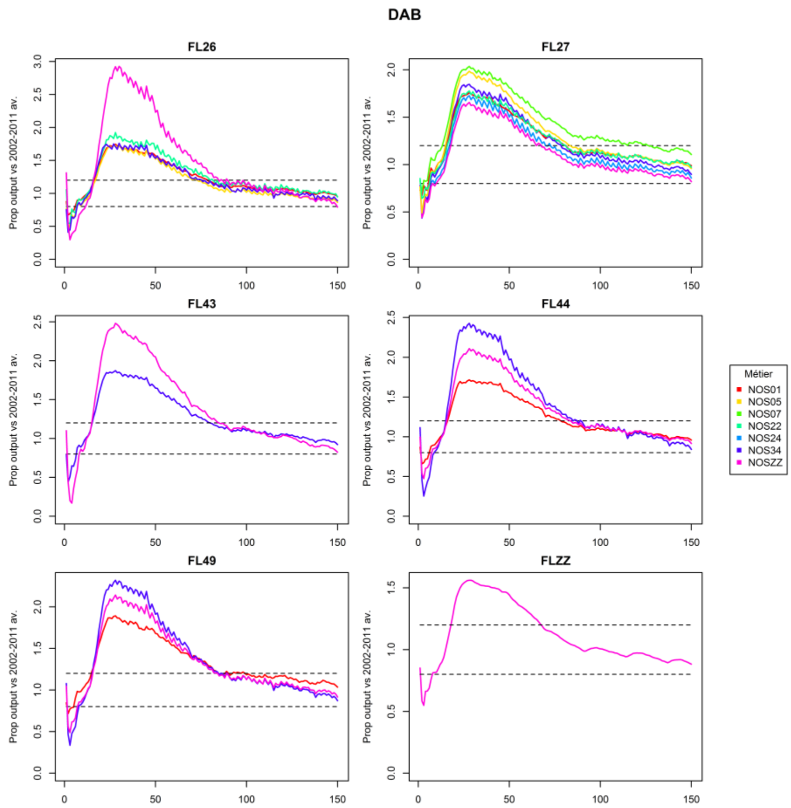
Appendix IX (continued)



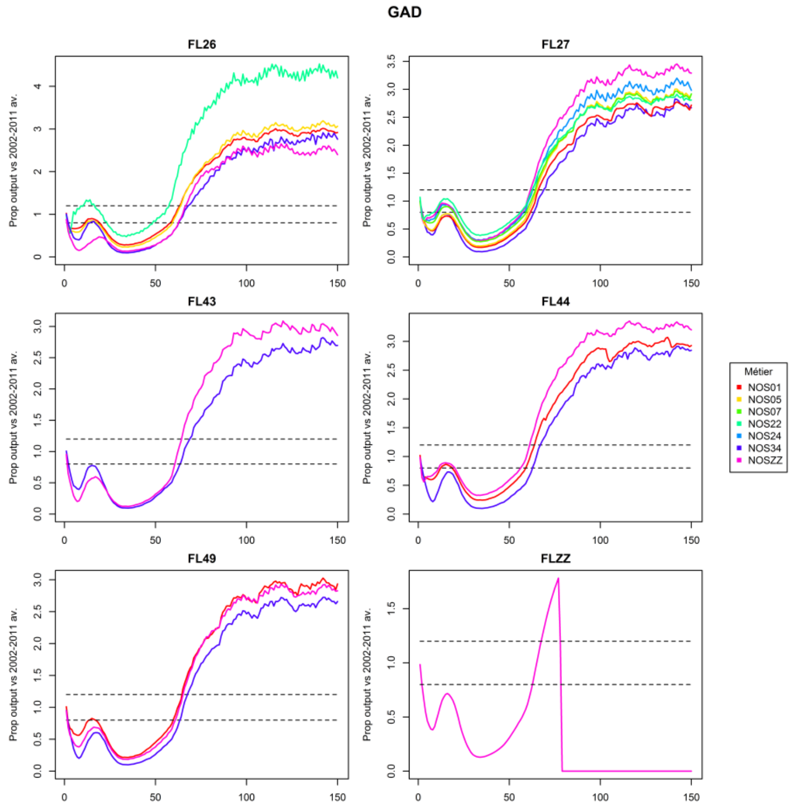


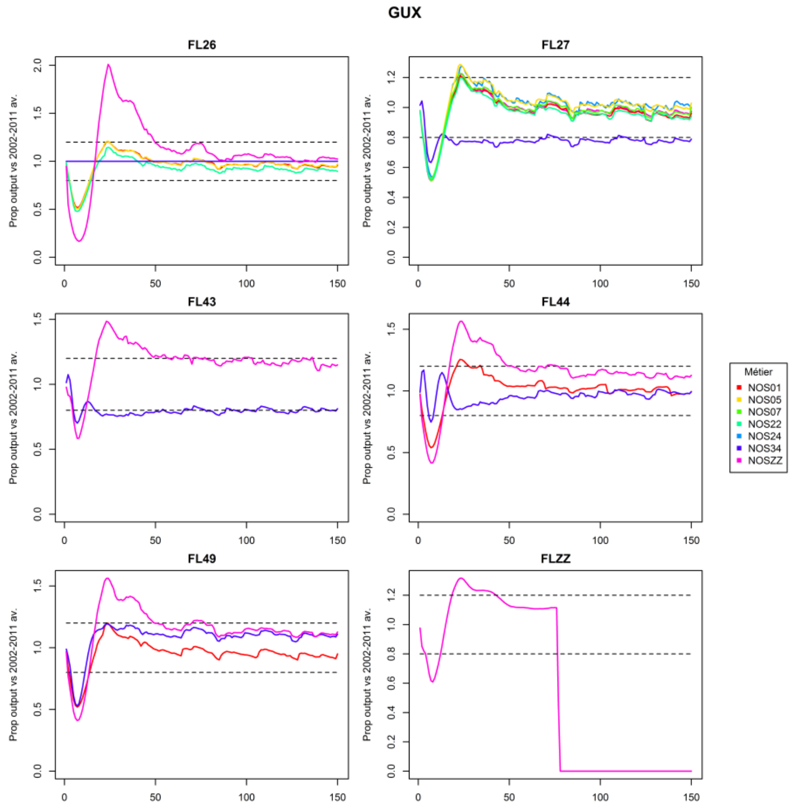
Appendix IX (continued)



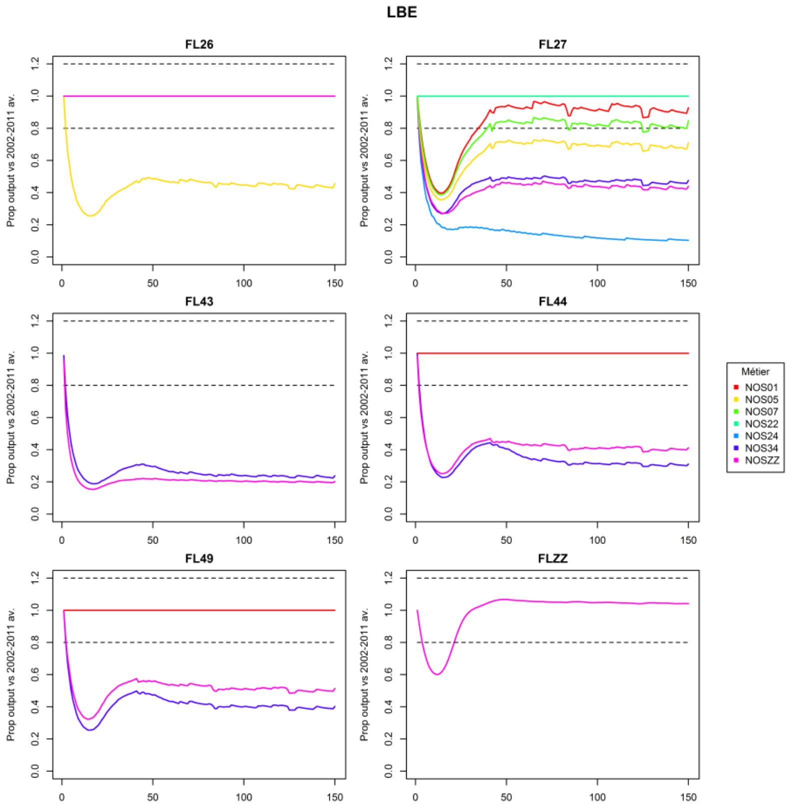


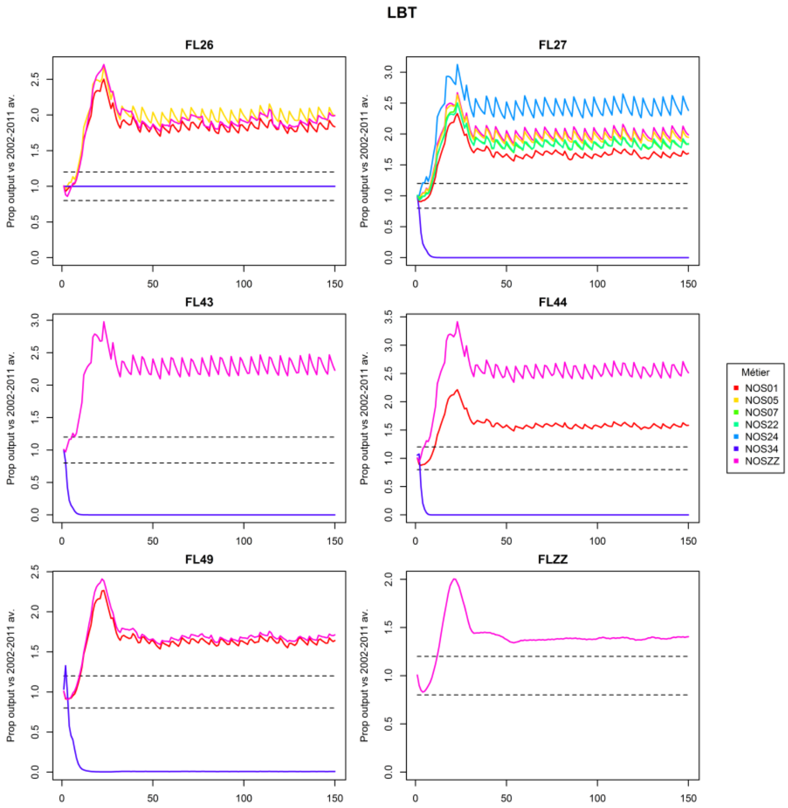
Appendix IX (continued)





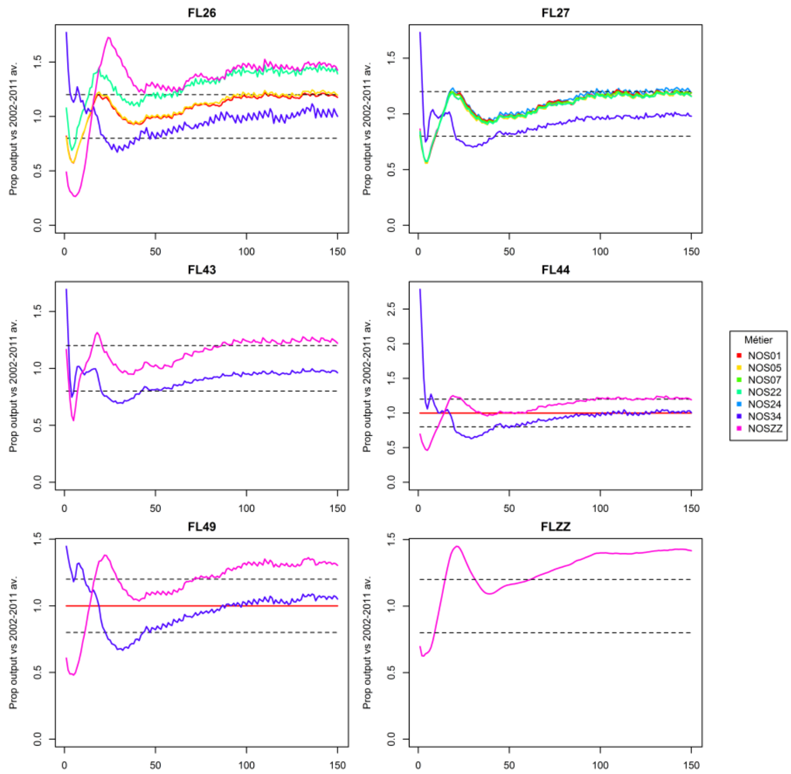
Appendix IX (continued)



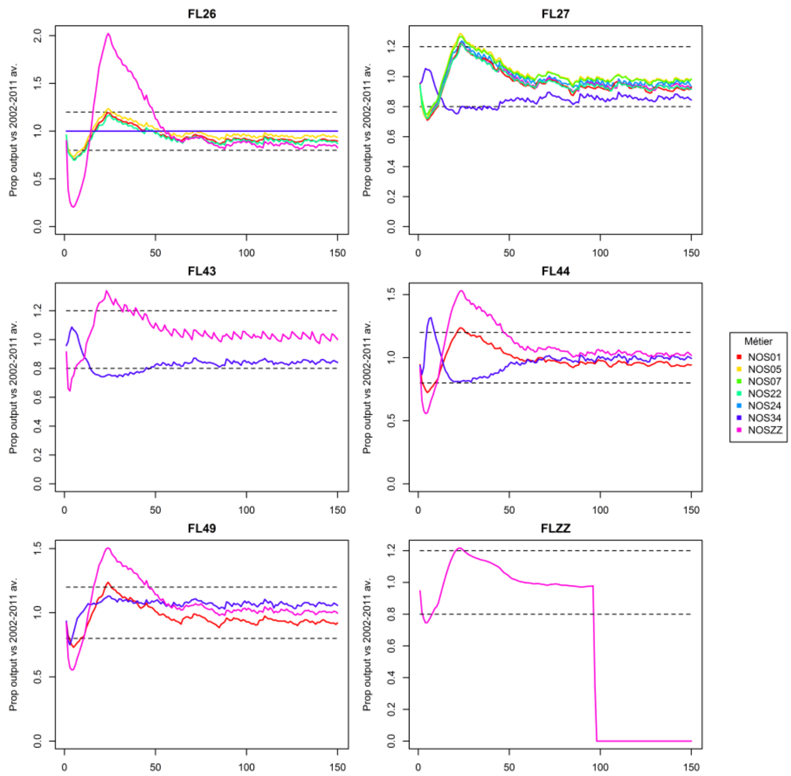


Appendix IX (continued)

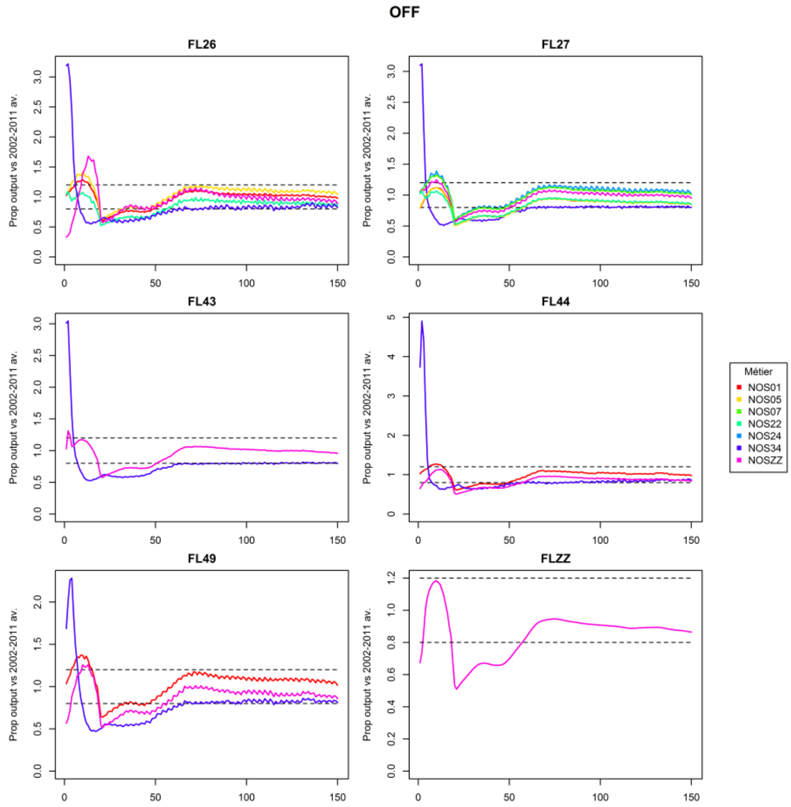
MAC

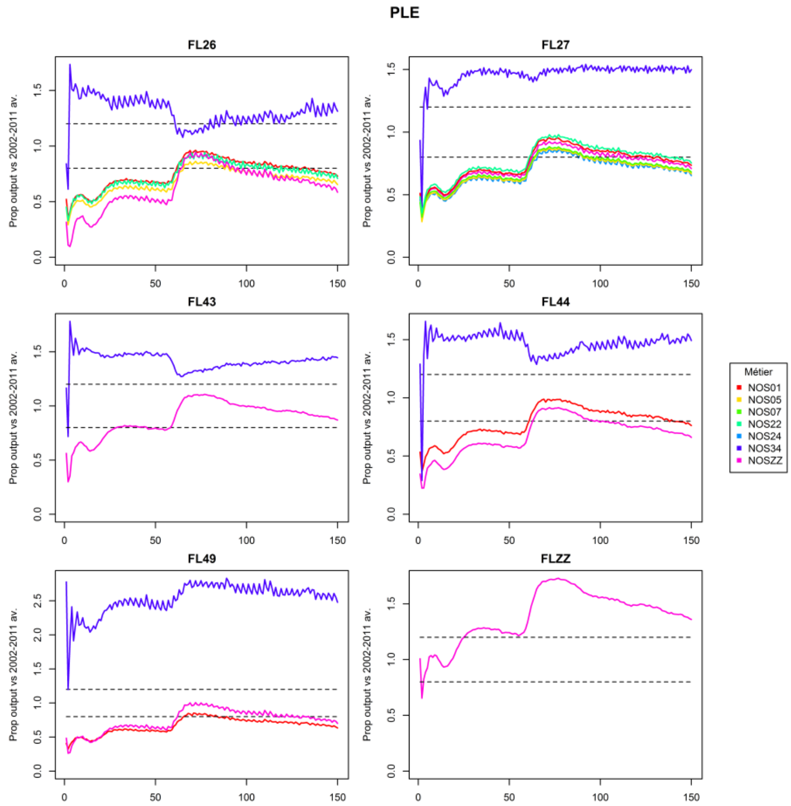


MUL



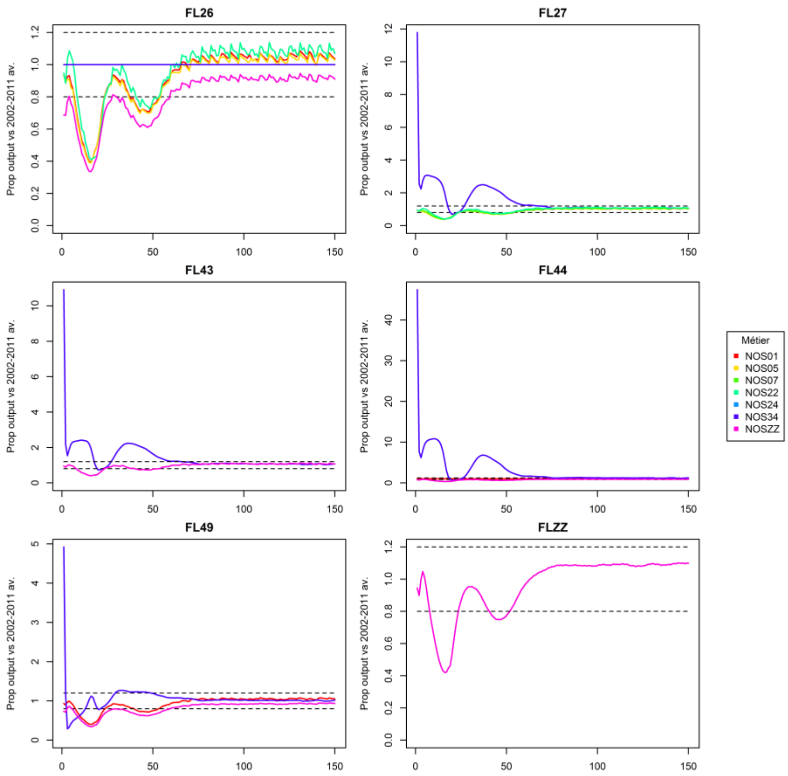
Appendix IX (continued)



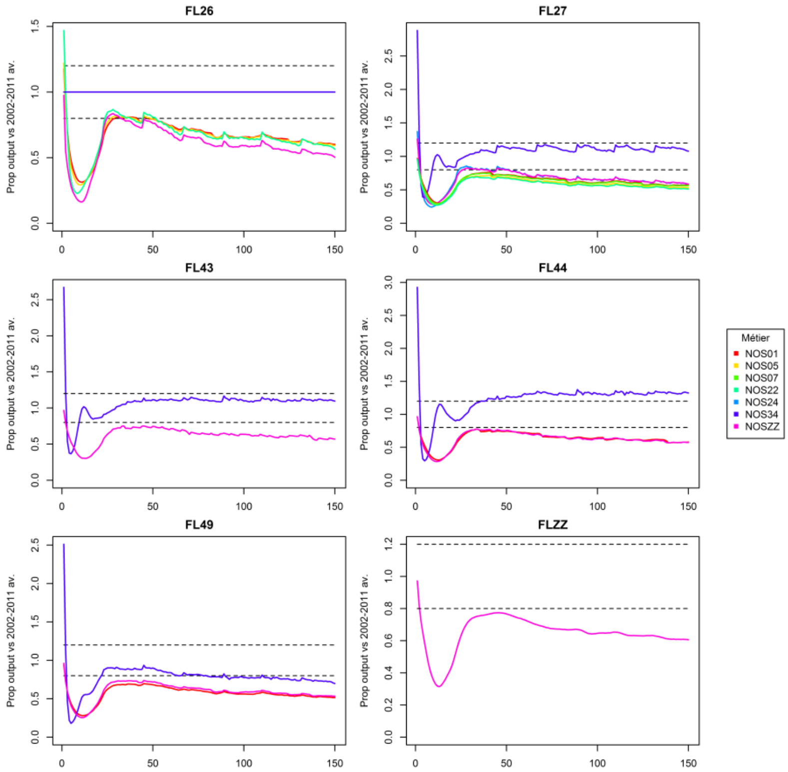


Appendix IX (continued)

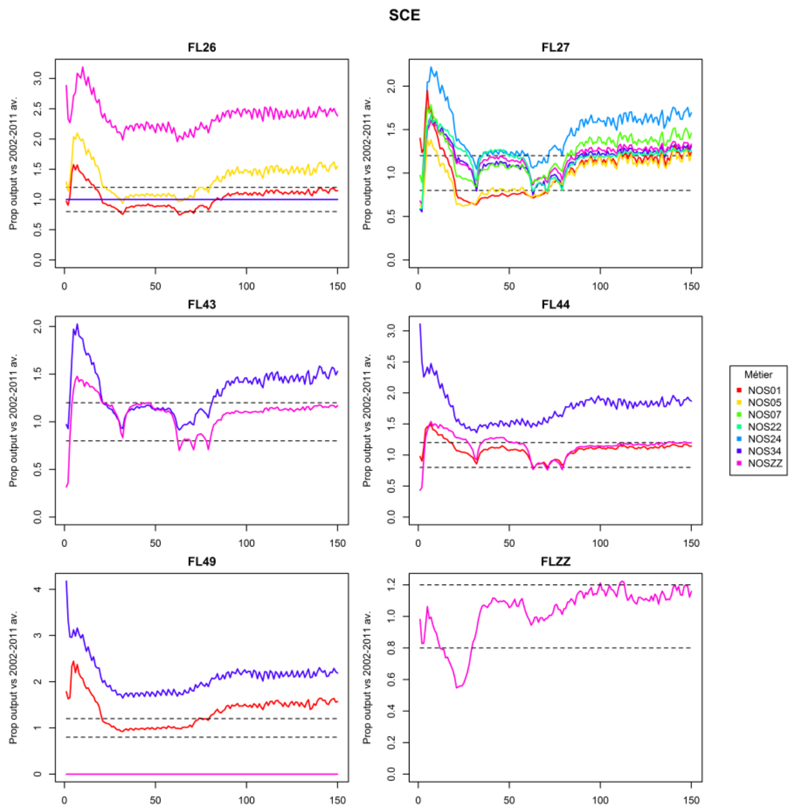
POL

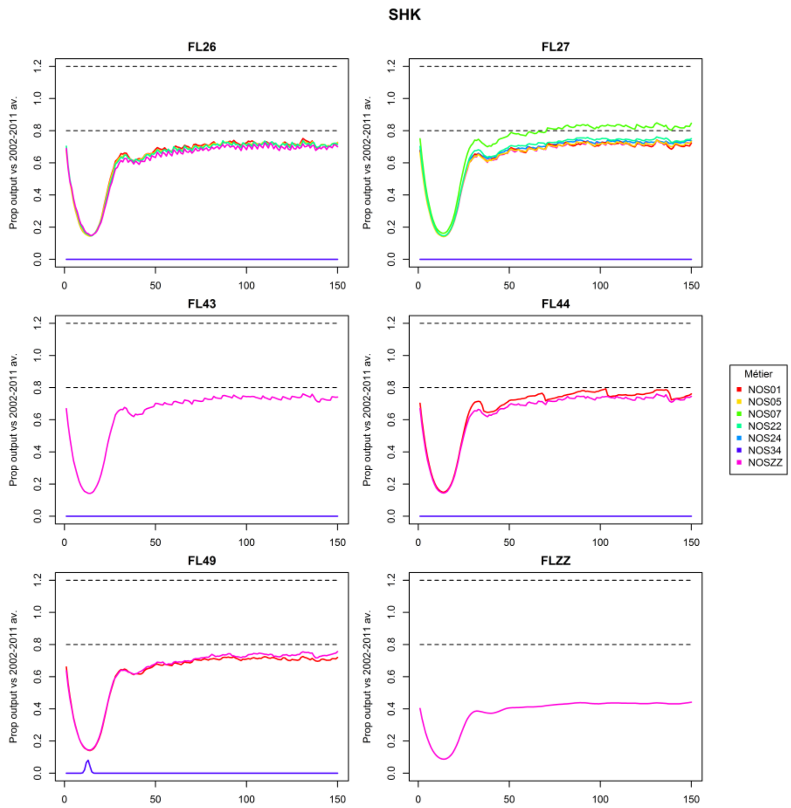


RAY

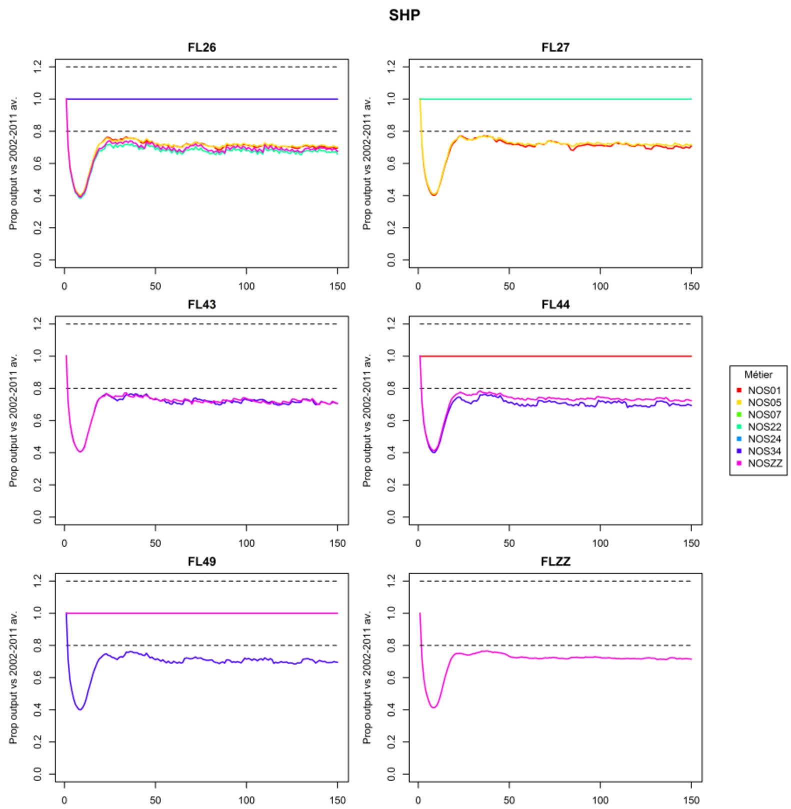


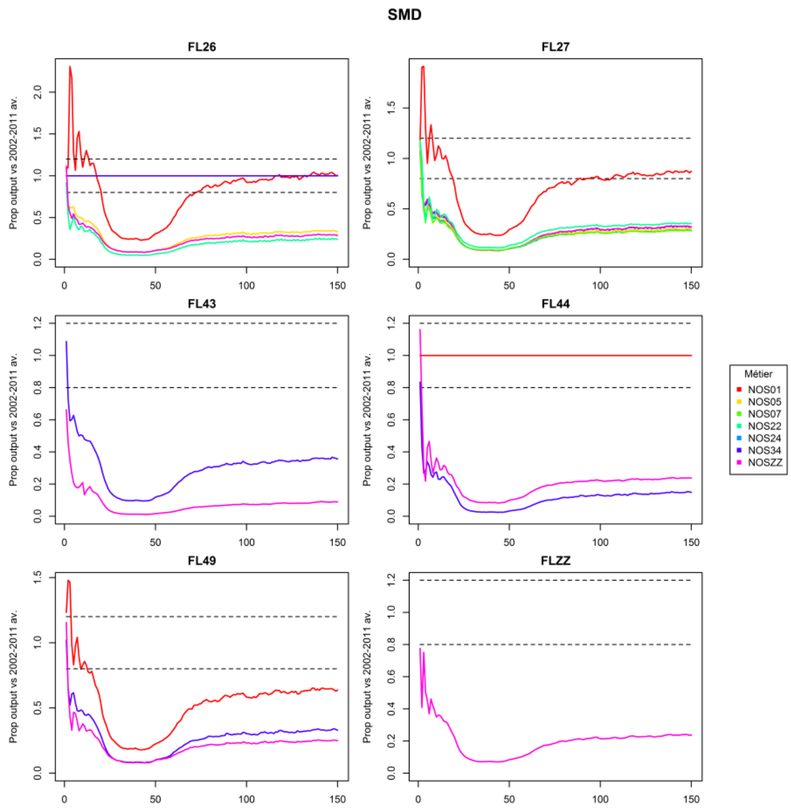
Appendix IX (continued)



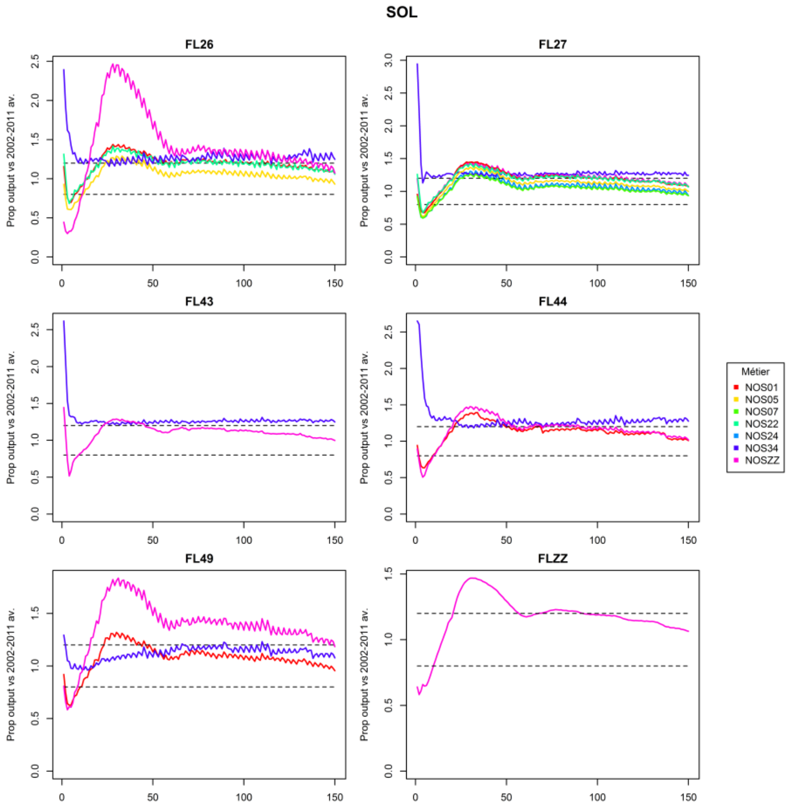


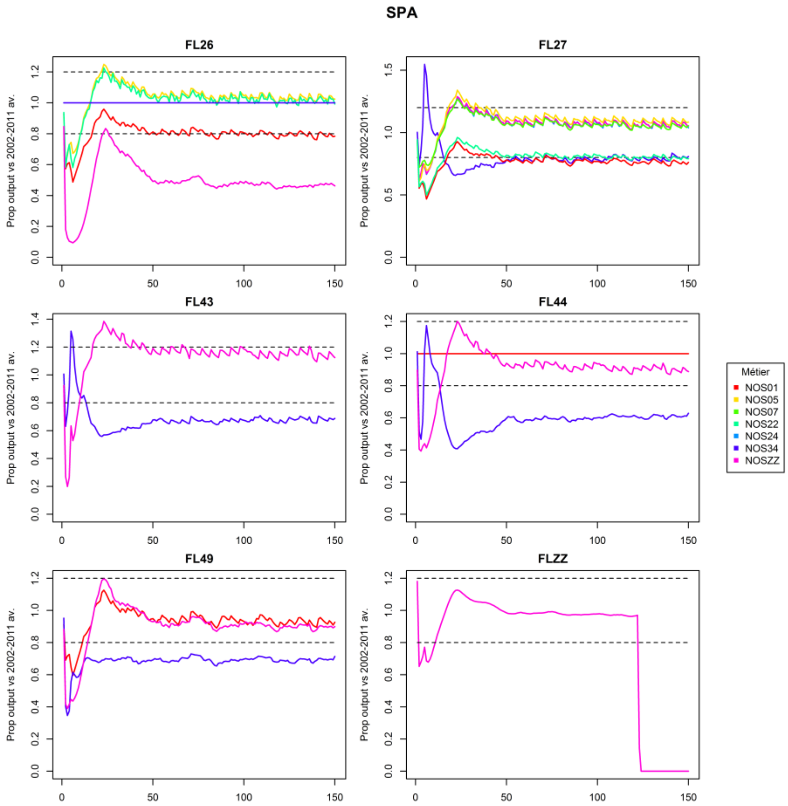
Appendix IX (continued)



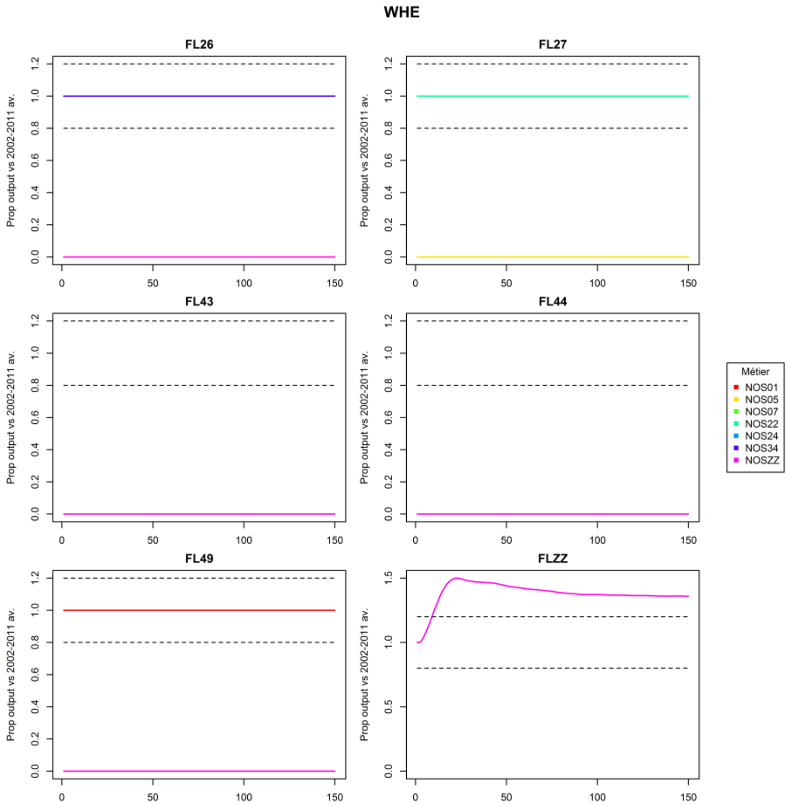


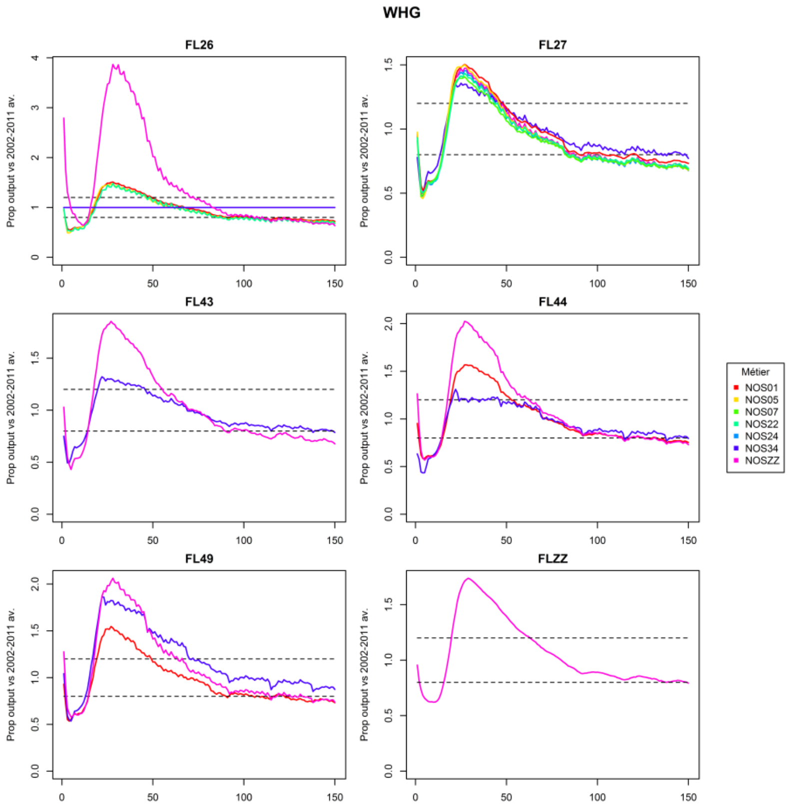
Appendix IX (continued)





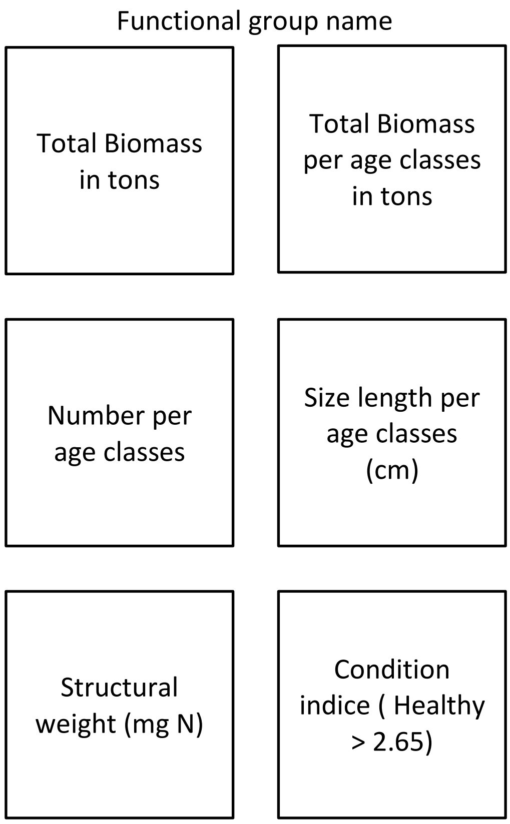
Appendix IX (continued)





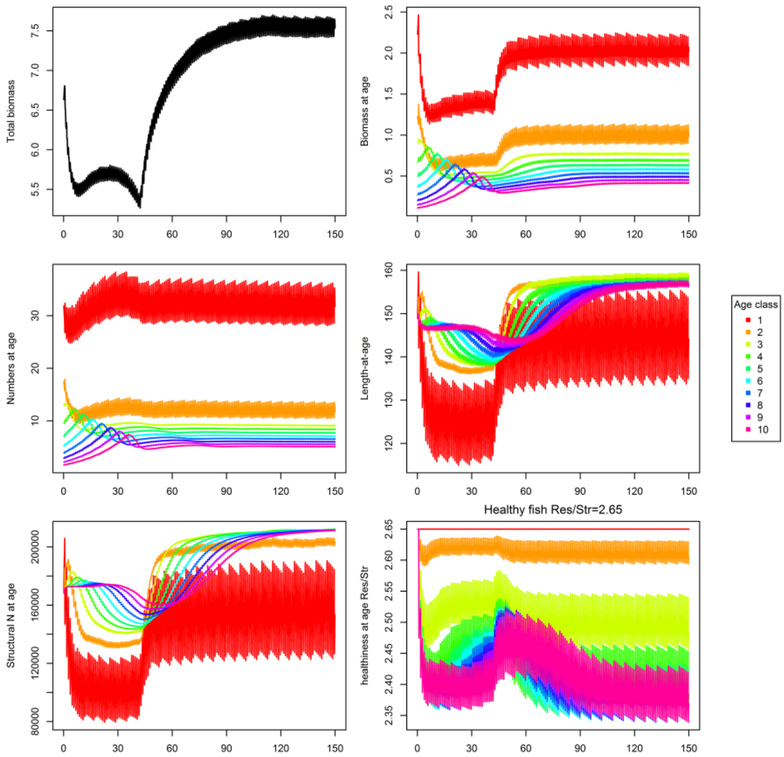
Appendix X : Vertebrates' biological outputs from Atlantis EEC calibrated after 150 years of simulation.

We present here, the total biomass in tonnes, the total biomass per age classes in tonnes, the number per age classes, the size length per age classes (cm), the structural weight (mg N) and the indice of condition used in Atlantis to assess the fertility of fish (reserve weight / structural weight). Each figure is structure as follows :

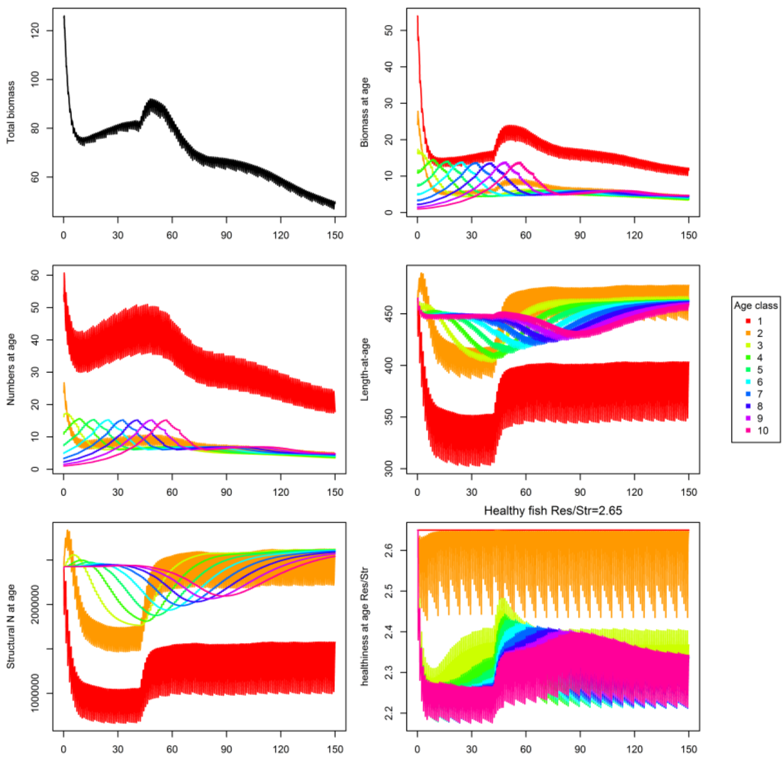


Appendix X (continued)

Seals: Entire model

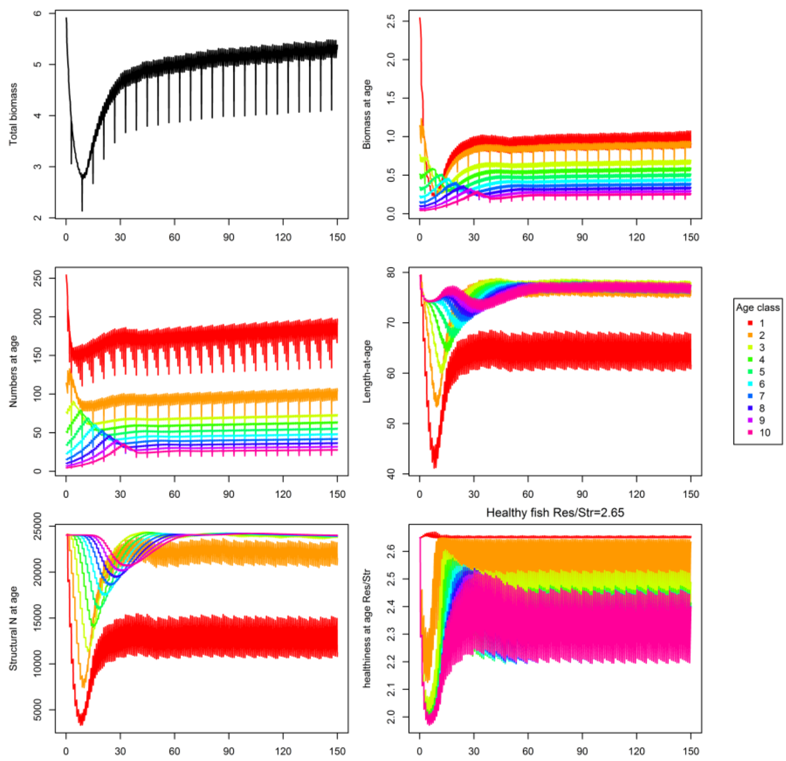


Toothed_cet: Entire model

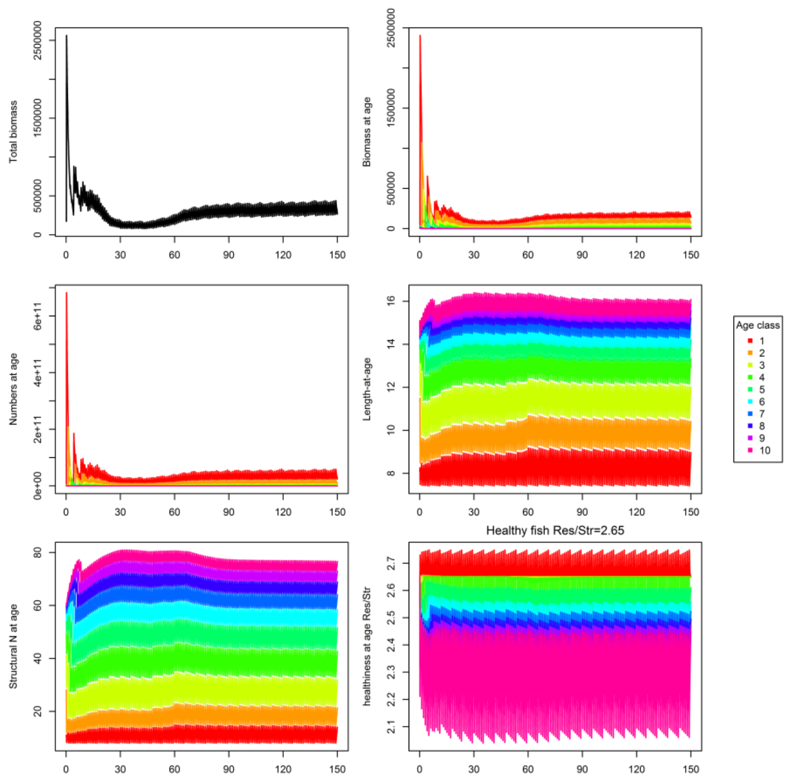


Appendix X (continued)

Seabird: Entire model

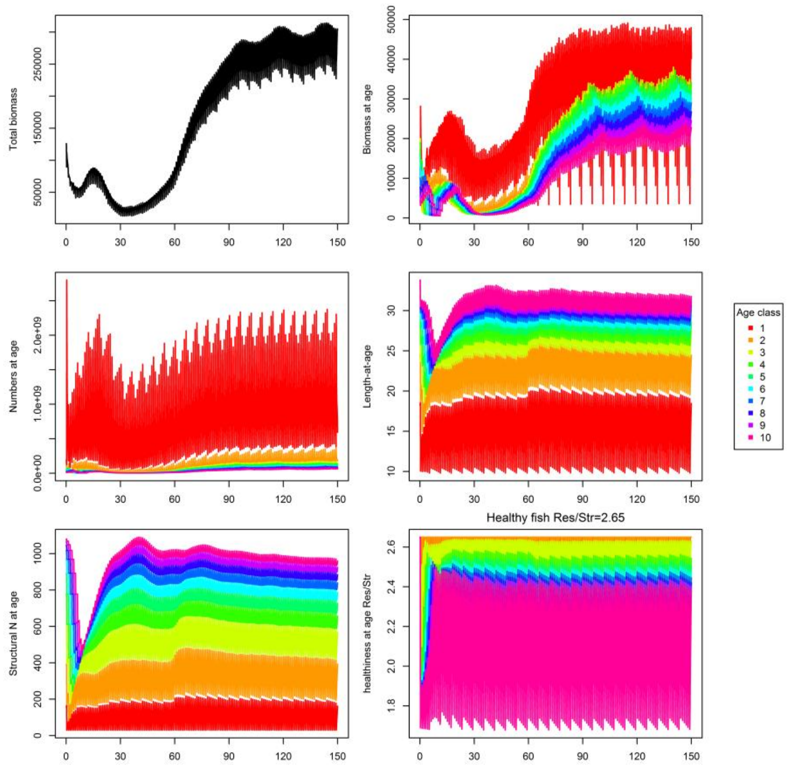


Sm_demersal: Entire model

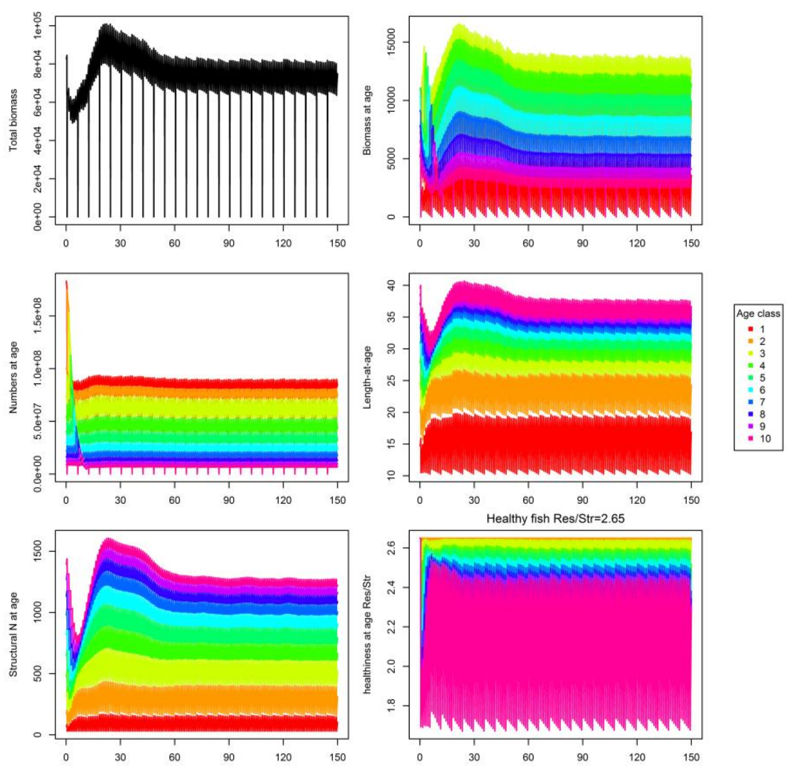


Appendix X (continued)

O_Gadoids: Entire model

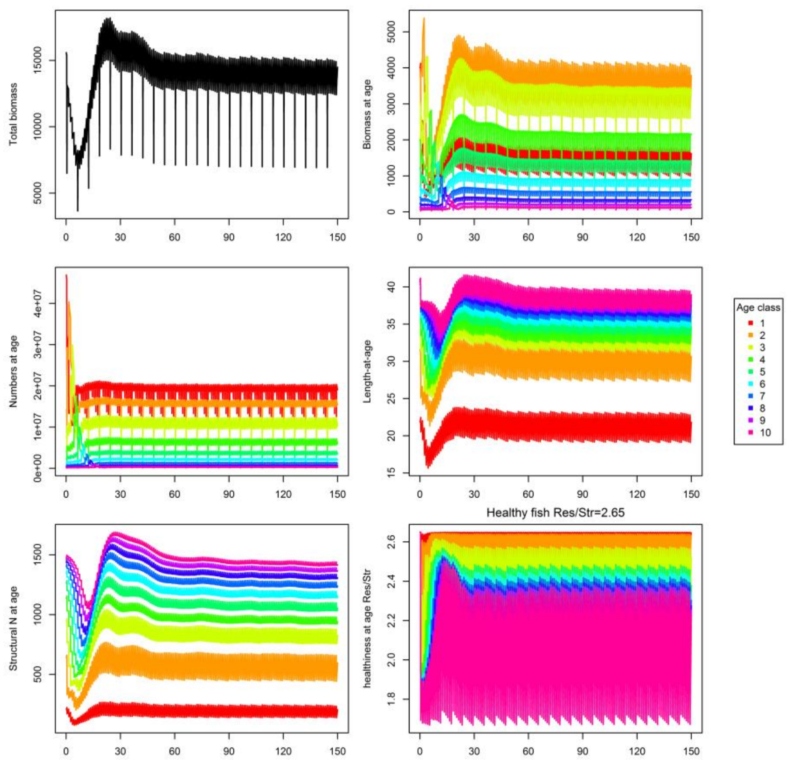


Mugilidae: Entire model

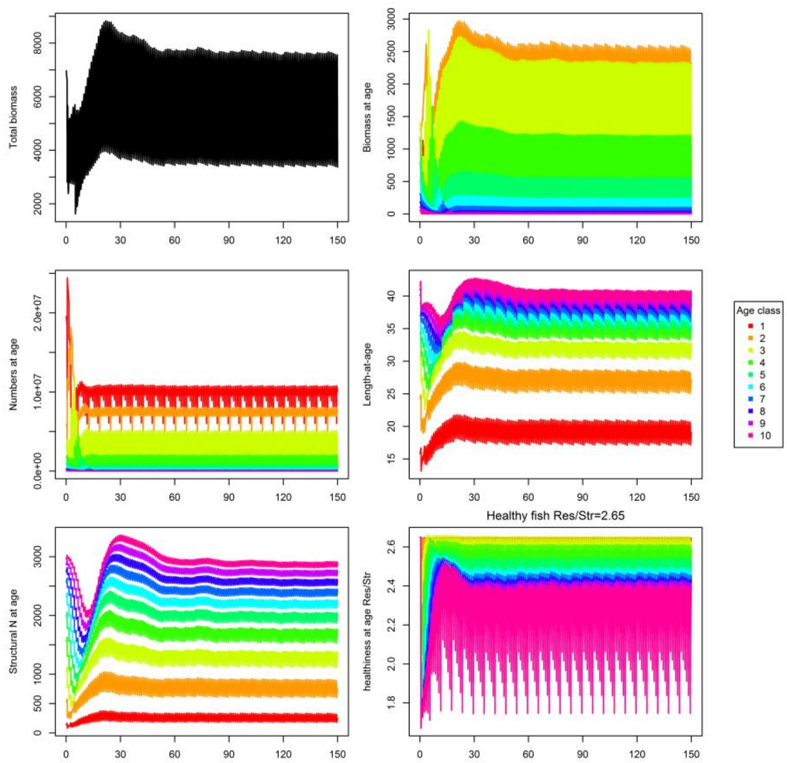


Appendix X (continued)

Gurnards: Entire model

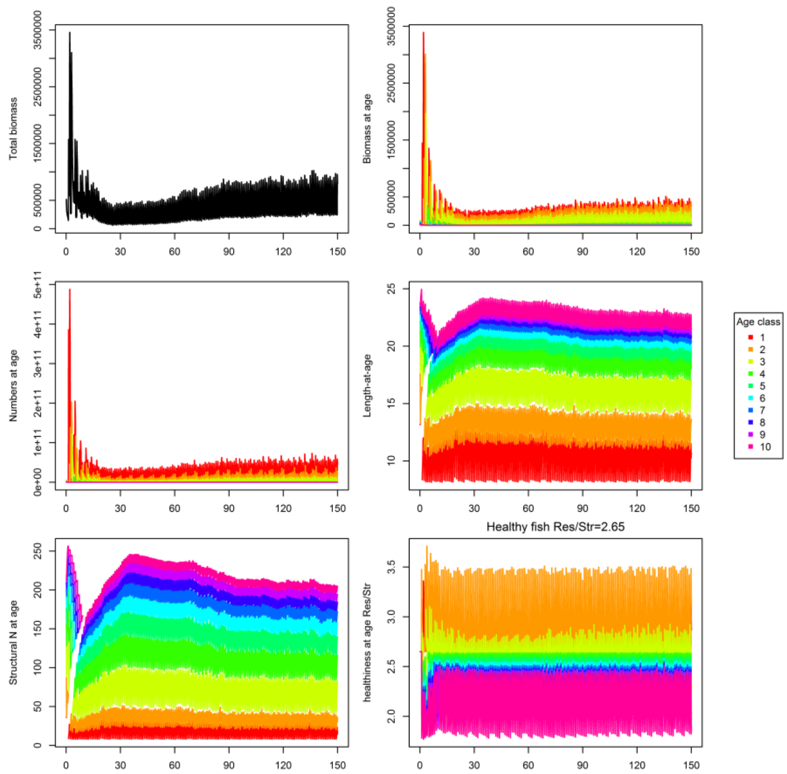


Sparidae: Entire model

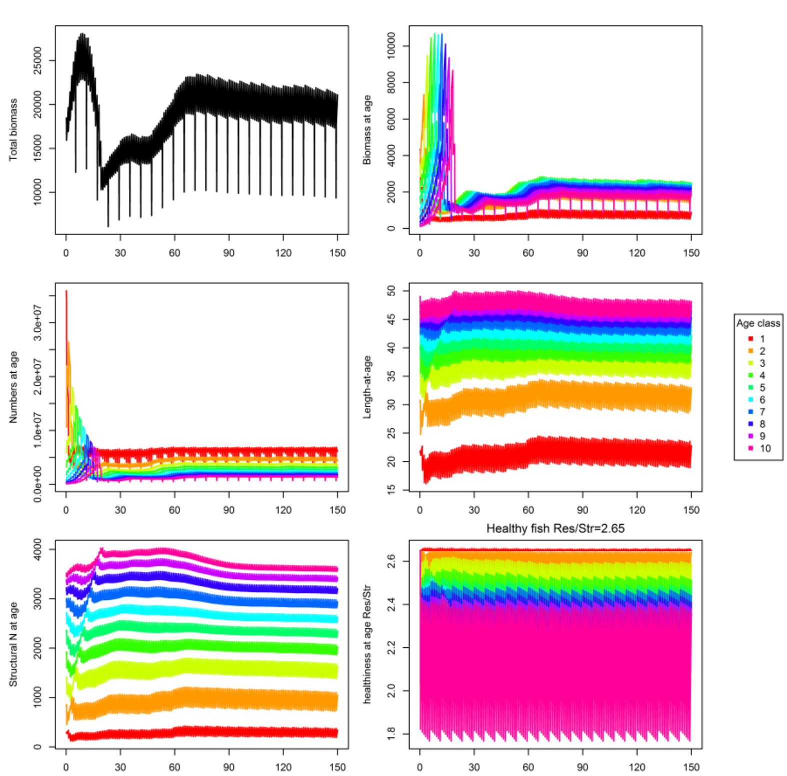


Appendix X (continued)

Clupeidae: Entire model

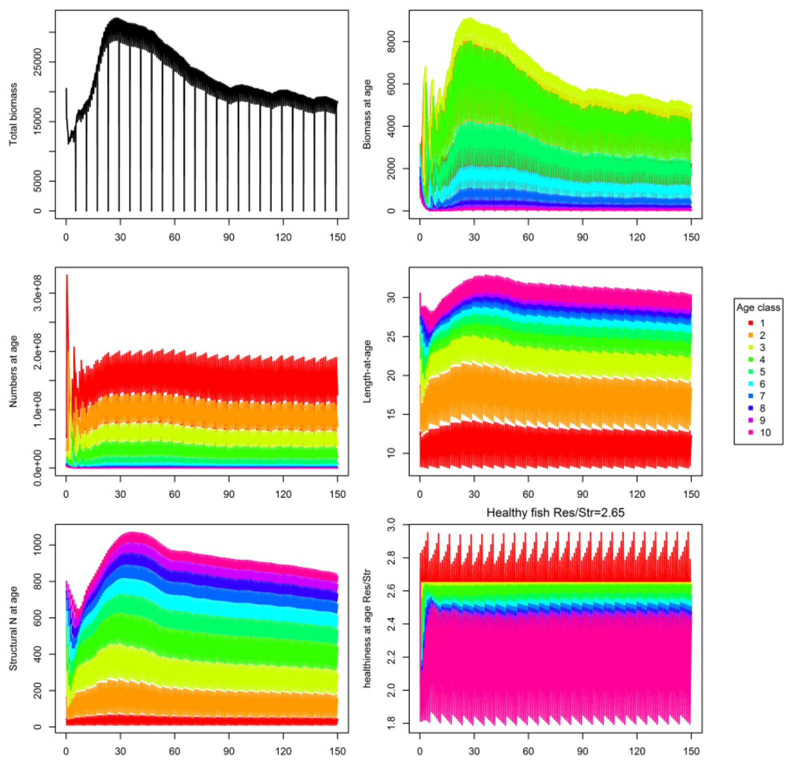


O_flatfish: Entire model

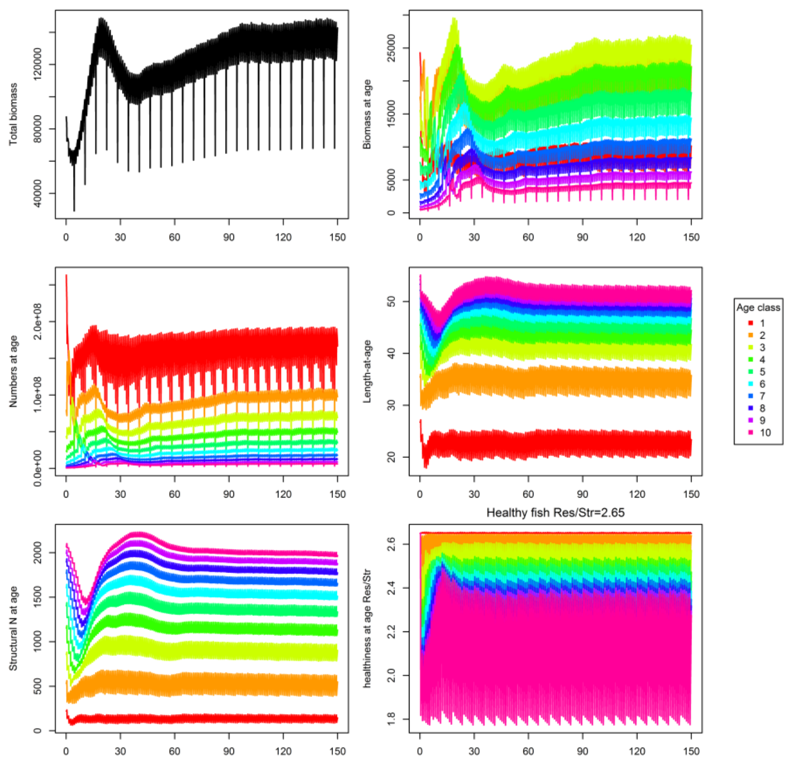


Appendix X (continued)

Dab: Entire model

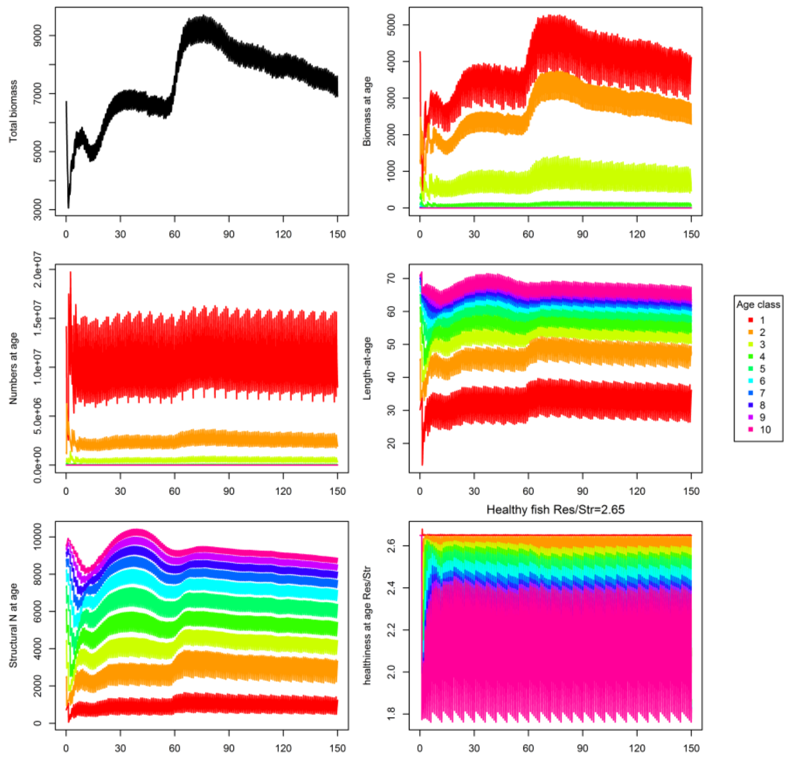


Mackerels: Entire model

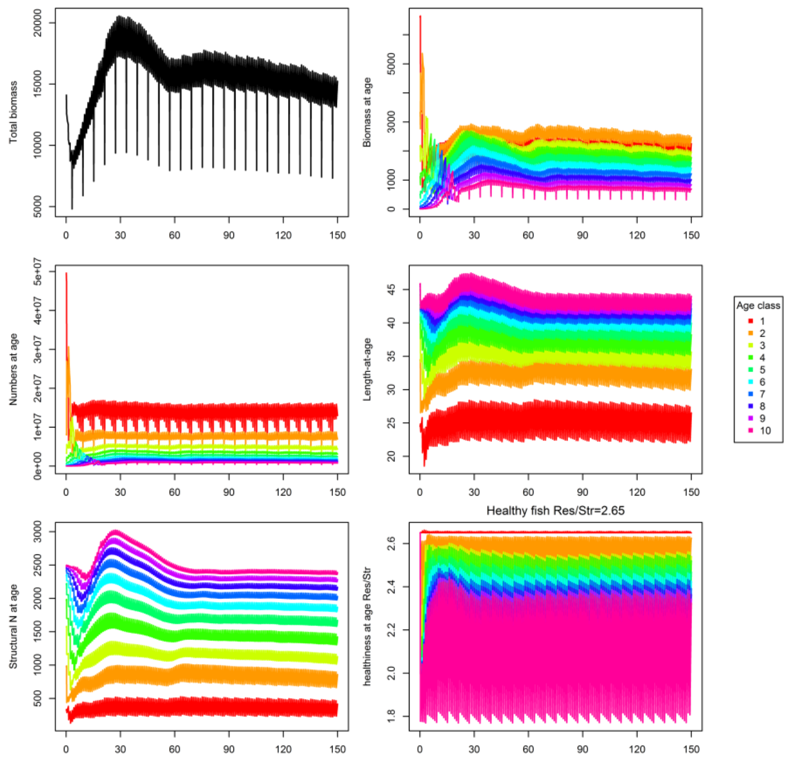


Appendix X (continued)

Plaice: Entire model

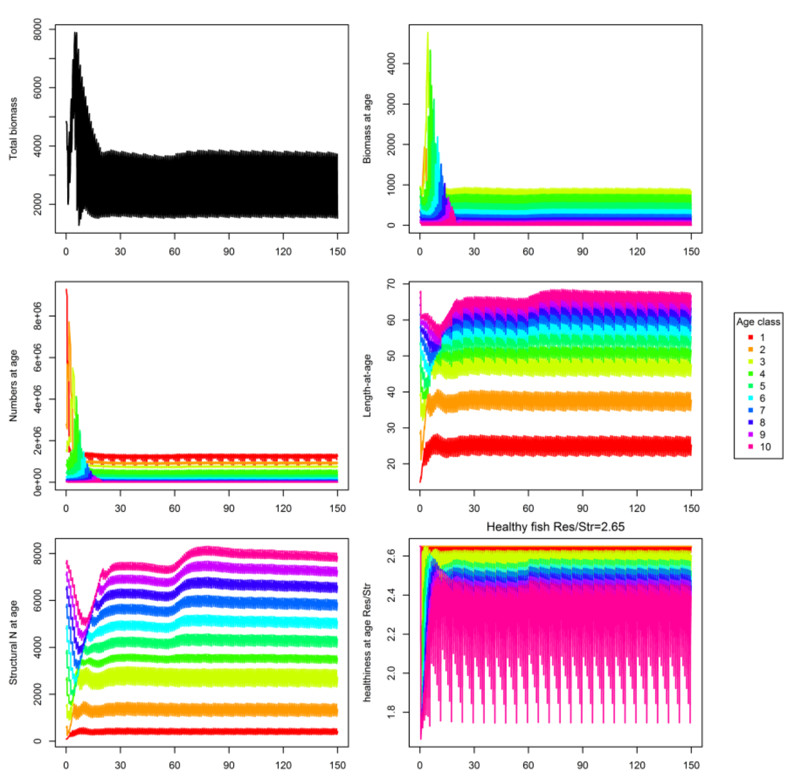


Sole: Entire model

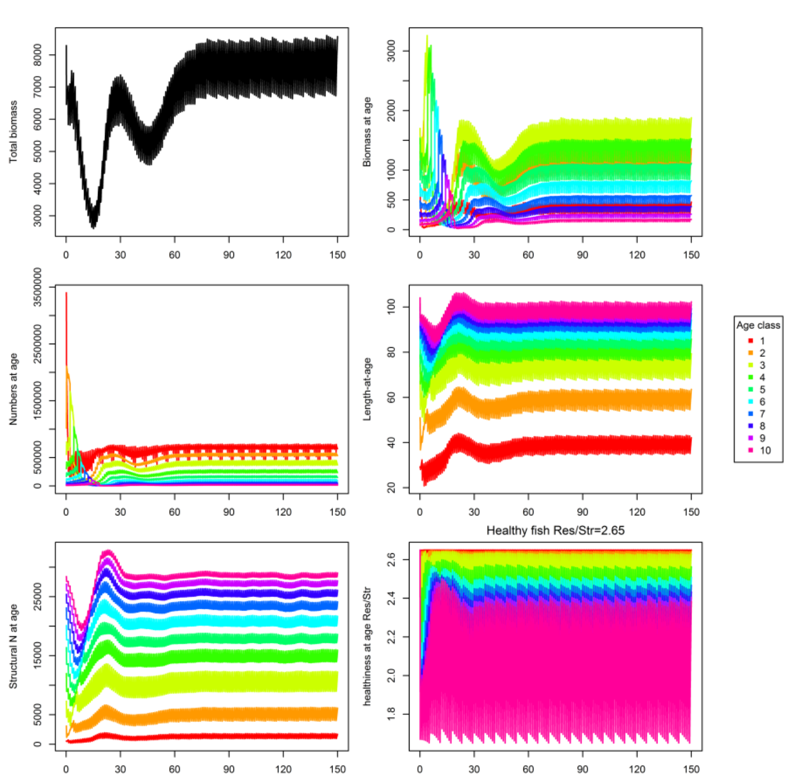


Appendix X (continued)

Seabass: Entire model

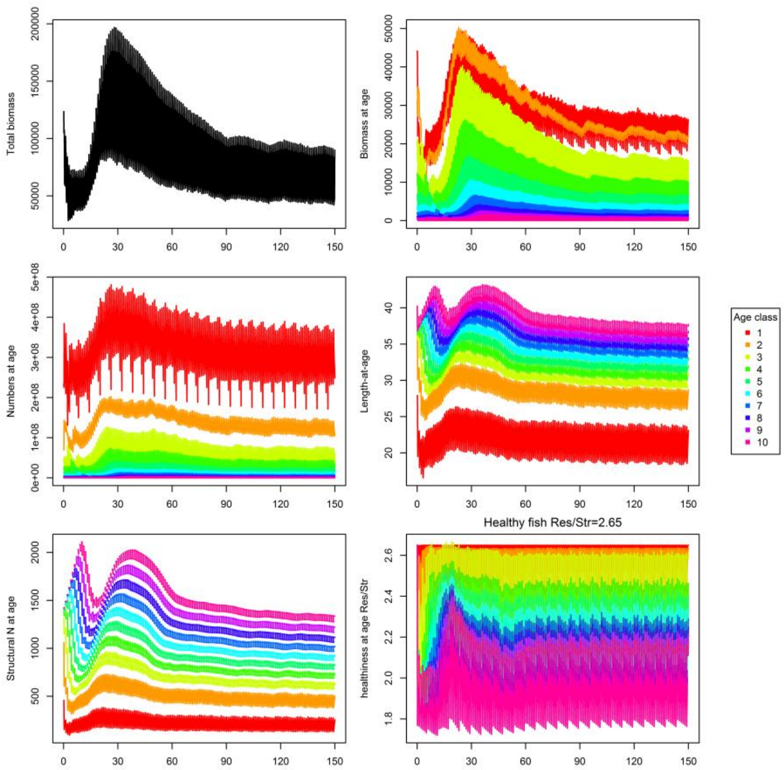


Pollack: Entire model

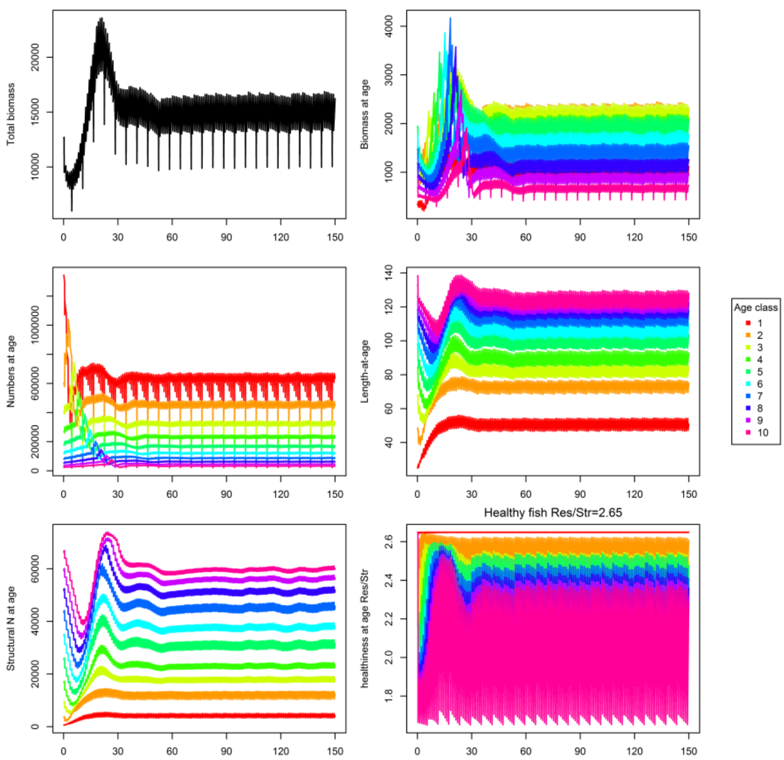


Appendix X (continued)

Whiting: Entire model

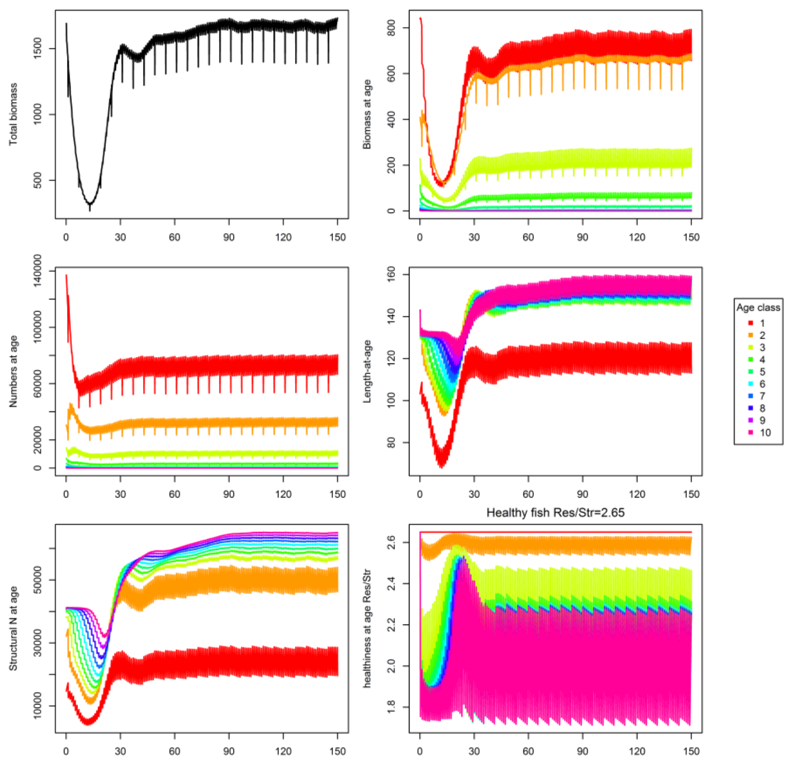


L_Bottom: Entire model

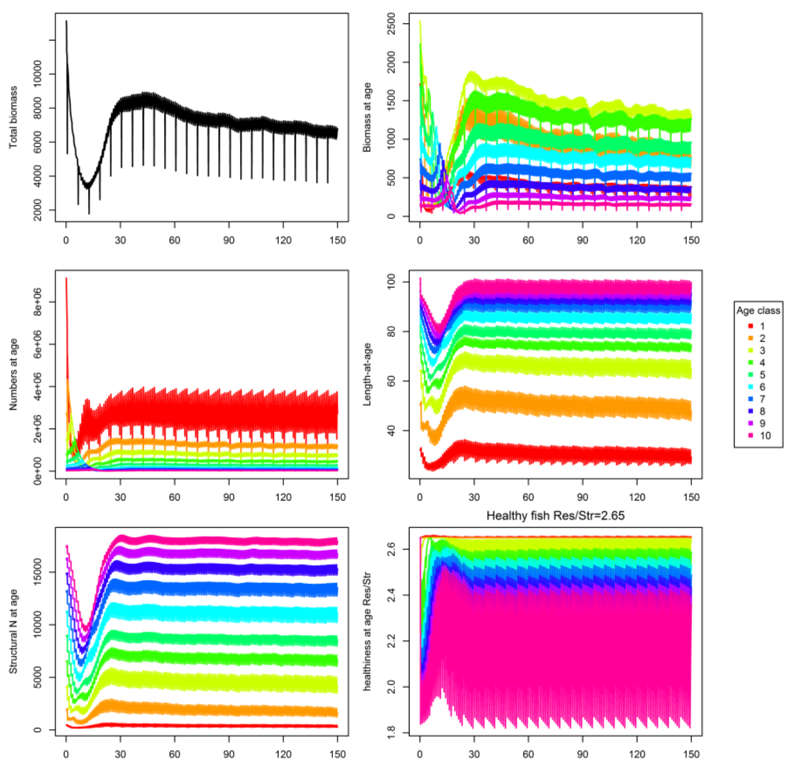


Appendix X (continued)

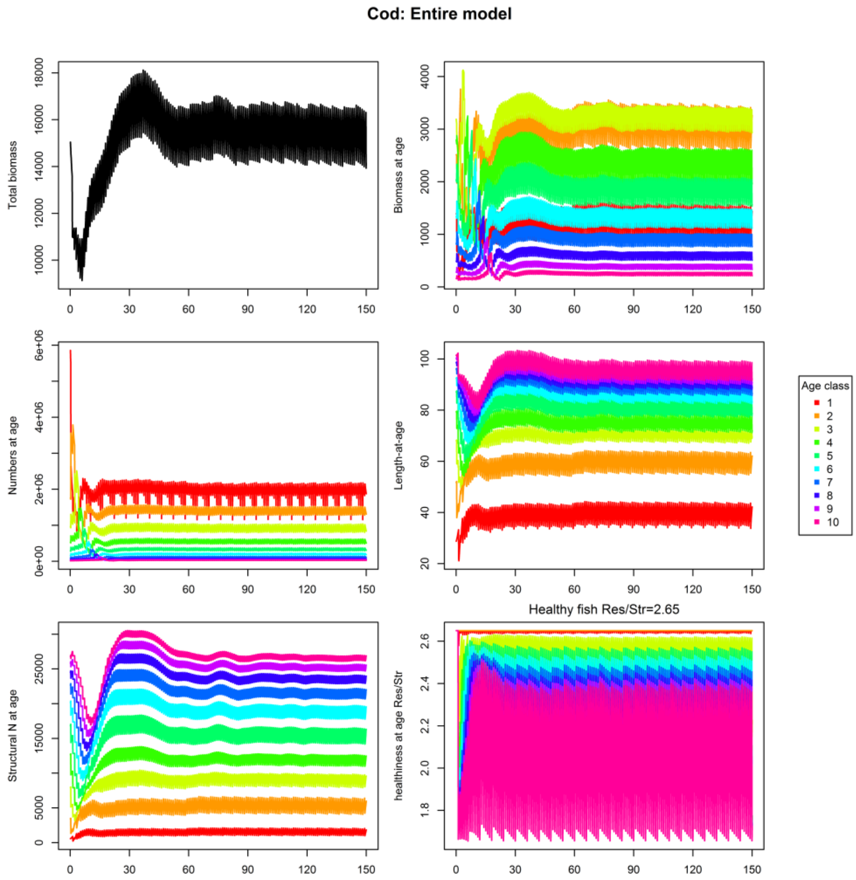
Sharks: Entire model



Rays_Dogs: Entire model



Appendix X (continued)



Appendix XI : Mortalites, biomasses and total catch outputs from Atlantis EEC calibrated after 150 years of simulation.

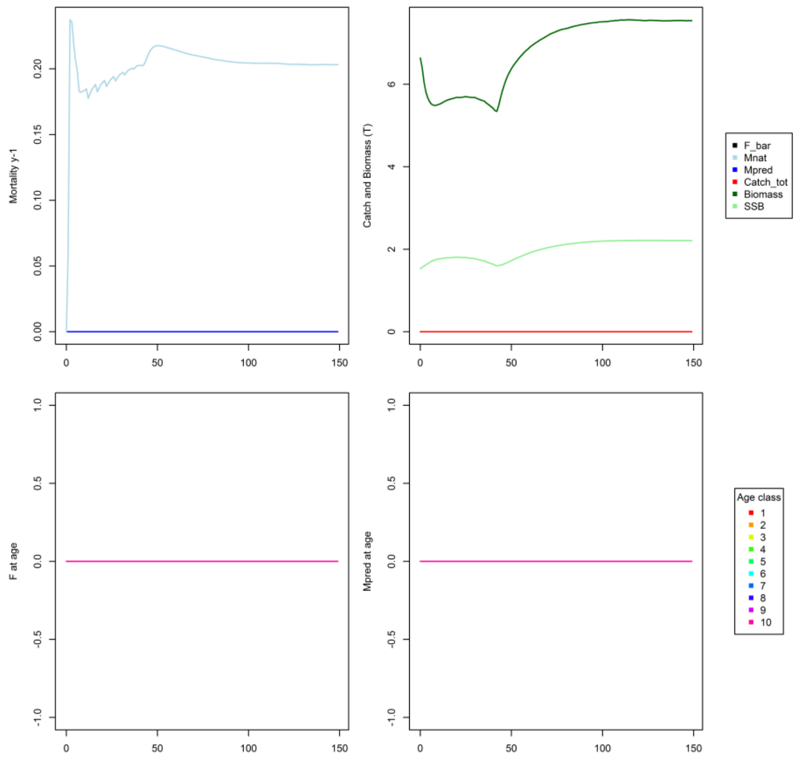
We present the total natural (Mnat), predation (Mpred) and fishing mortality (F), as well as the total biomass, spawning stock biomass (SSB) and total catch, and finally the fishing mortality and the predation mortality per age classes for each functionnal group. Each figure is structured as follows :

Functional group name

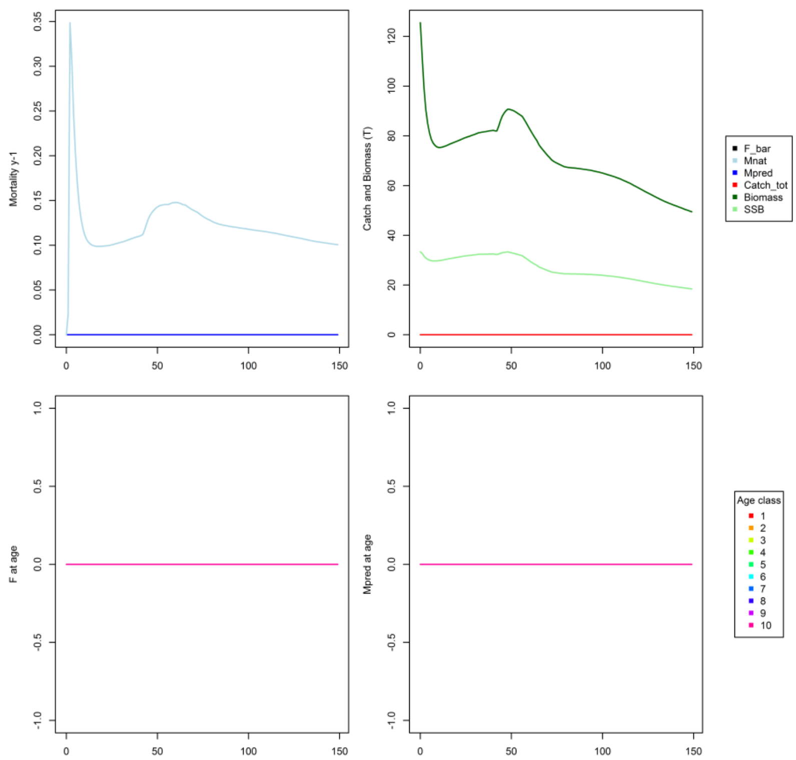
Total Mortalities (Mnat, Mpred, F)	Total Biomass, SSB, and catch in tons
F per age classes	Mpred per age classes

Appendix XI (continued)

Seals: Entire model Fishing

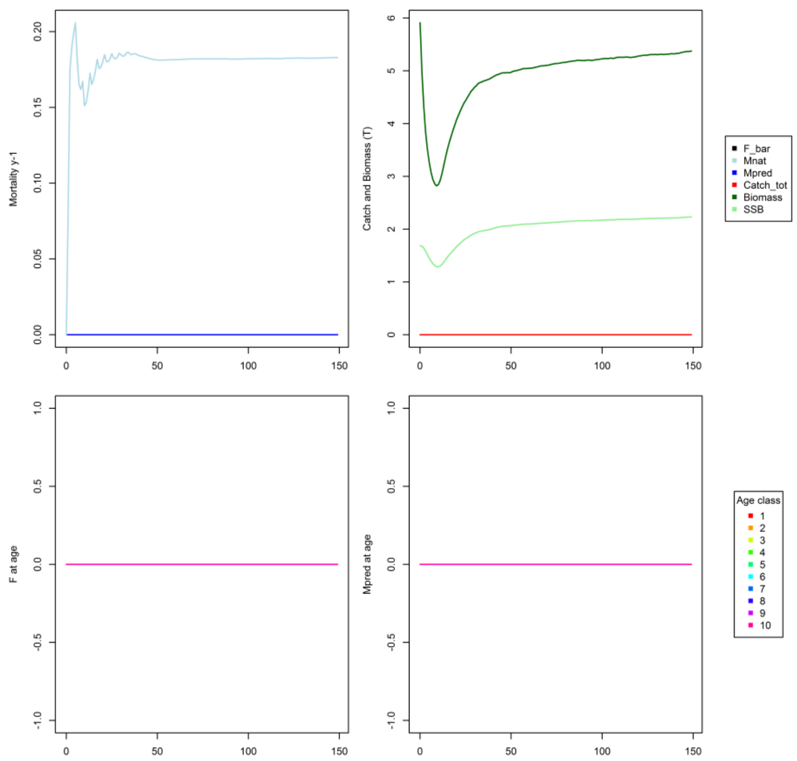


Toothed_cet: Entire model Fishing

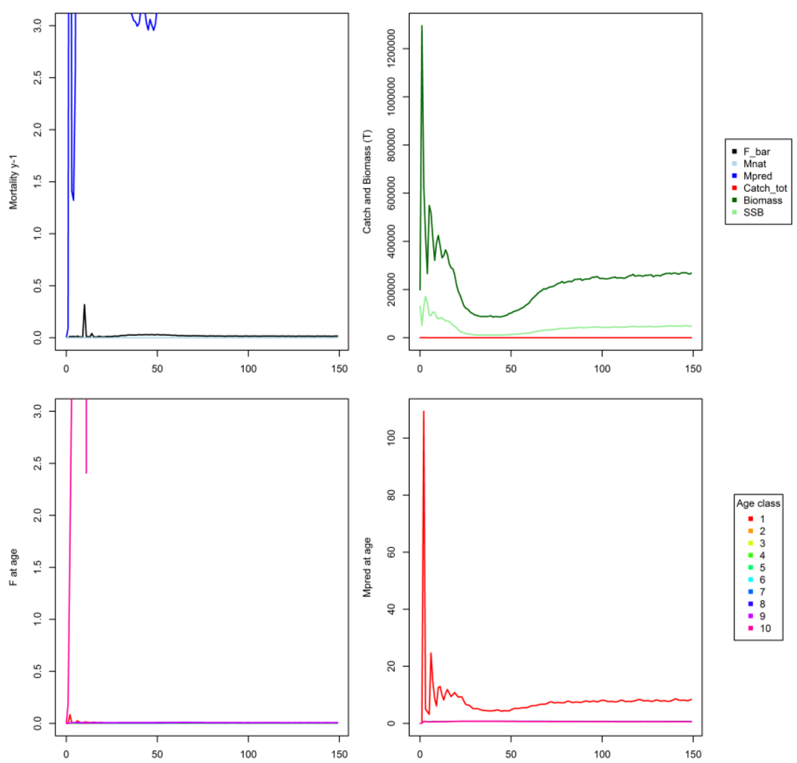


Appendix XI (continued)

Seabird: Entire model Fishing

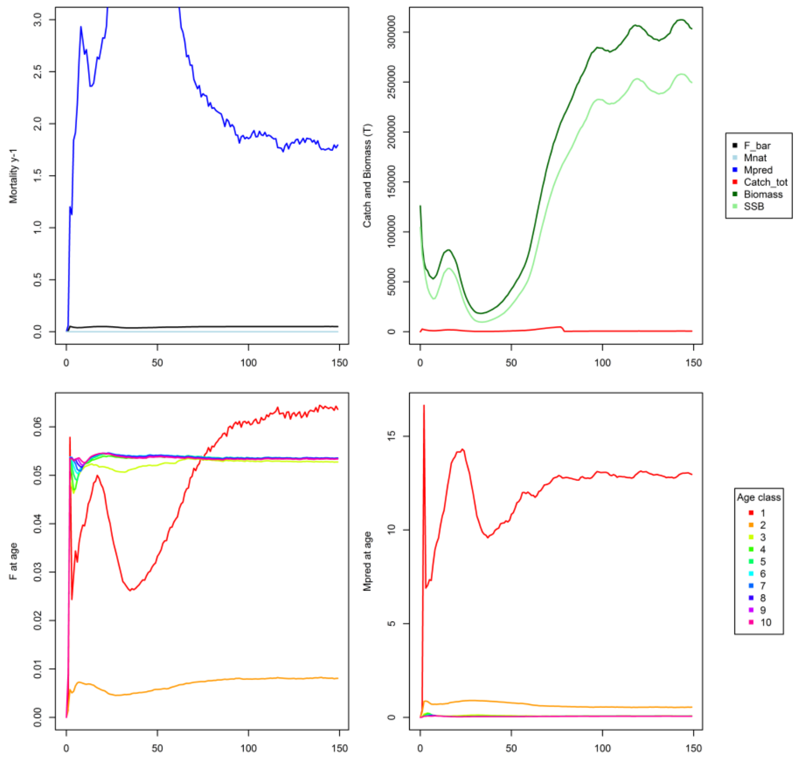


Sm_demersal: Entire model Fishing

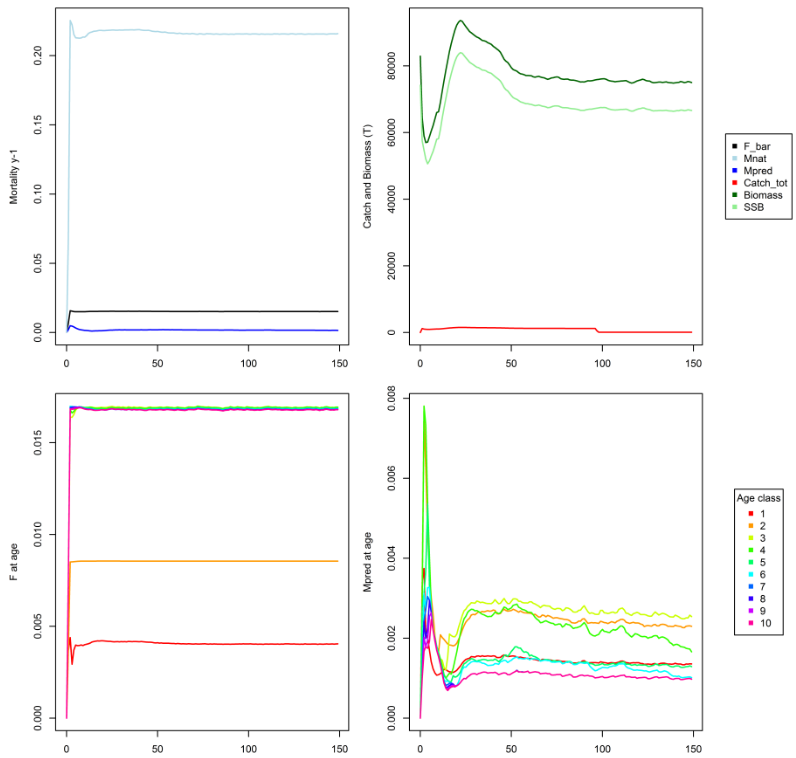


Appendix XI (continued)

O_Gadoids: Entire model Fishing

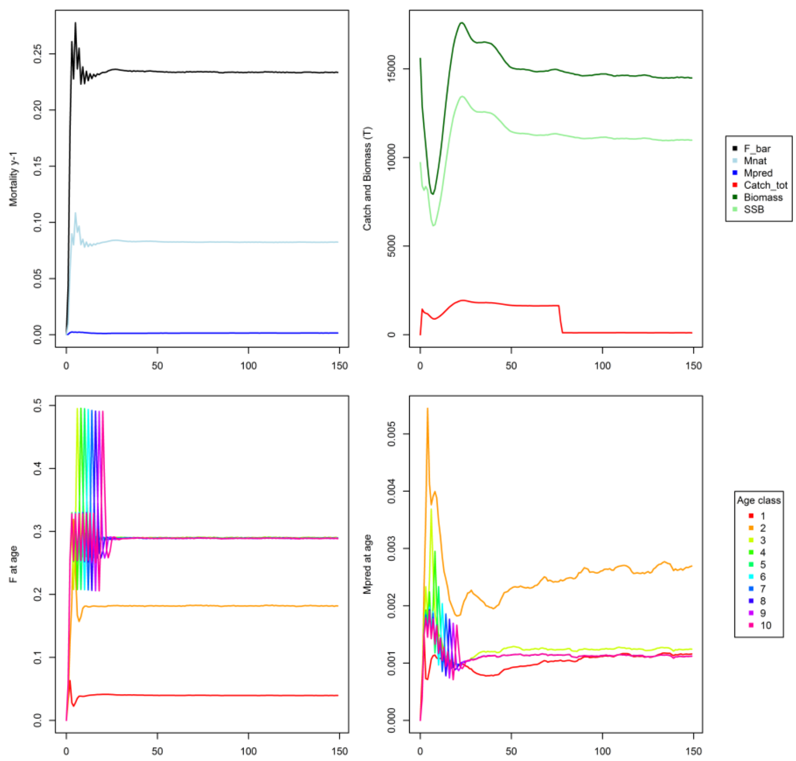


Mugilidae: Entire model Fishing

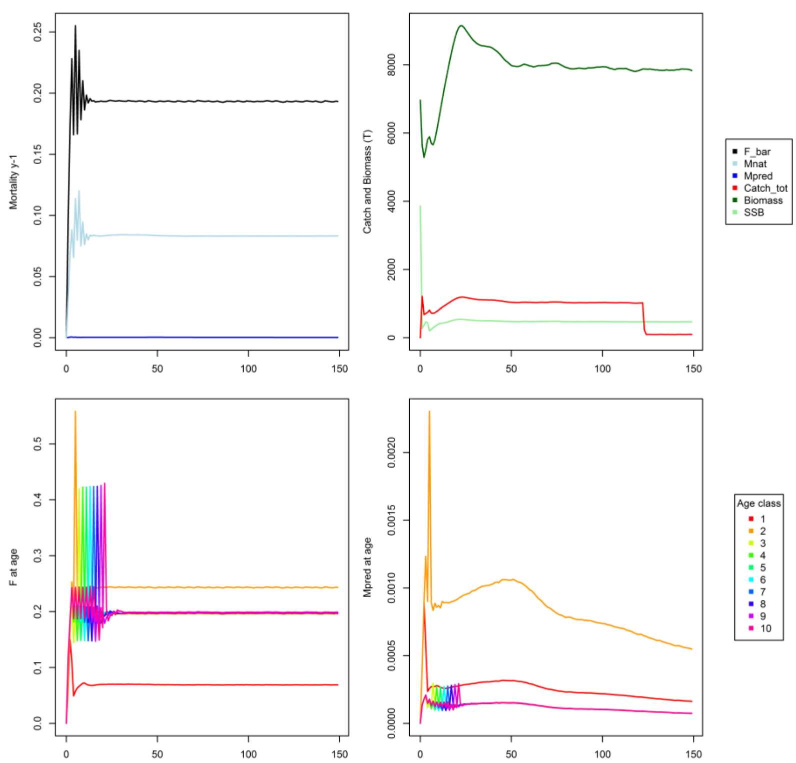


Appendix XI (continued)

Gurnards: Entire model Fishing

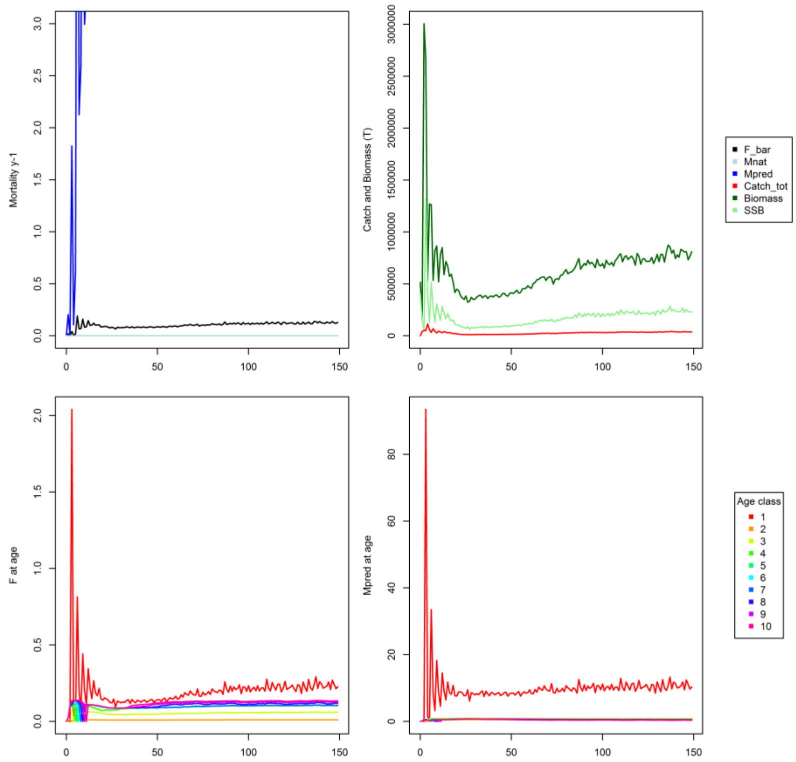


Sparidae: Entire model Fishing

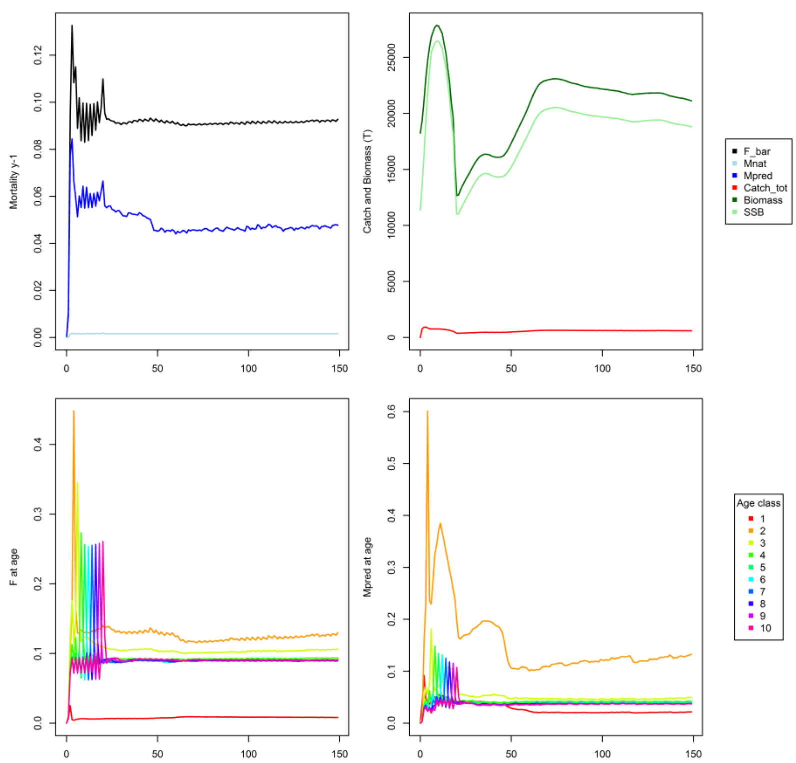


Appendix XI (continued)

Clupeidae: Entire model Fishing

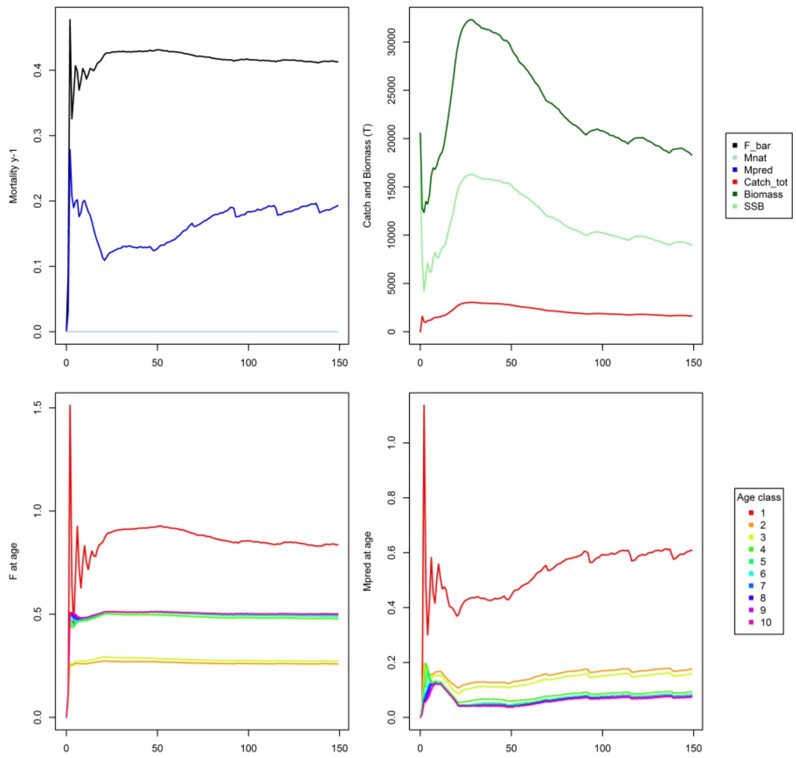


O_flatfish: Entire model Fishing

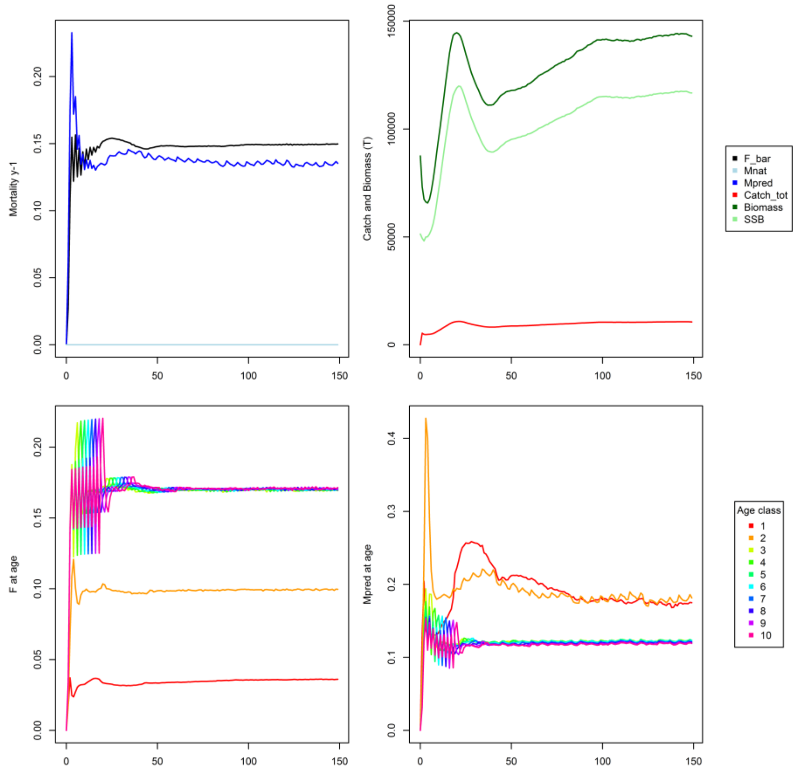


Appendix XI (continued)

Dab: Entire model Fishing

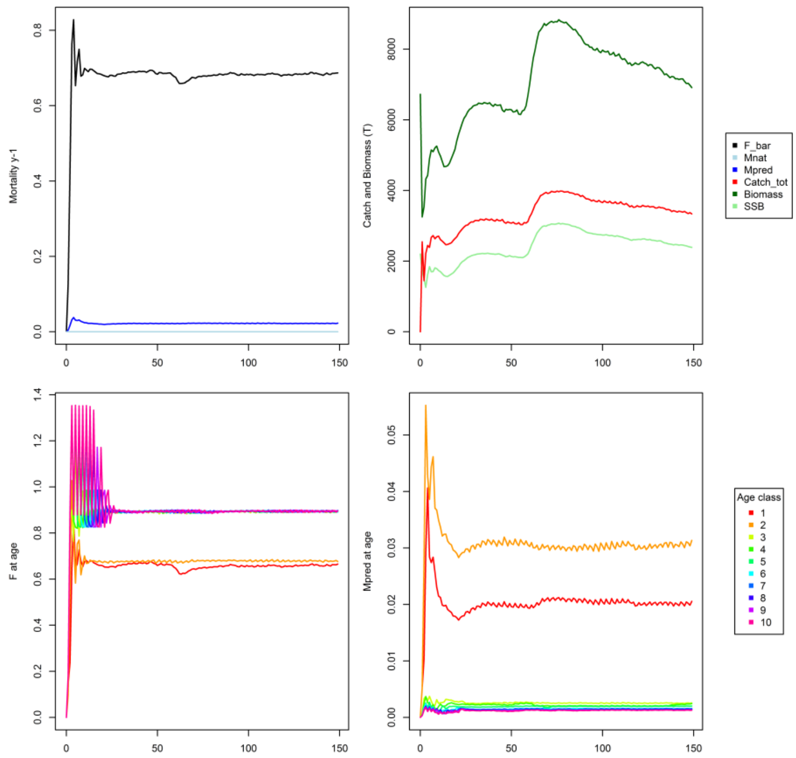


Mackerels: Entire model Fishing

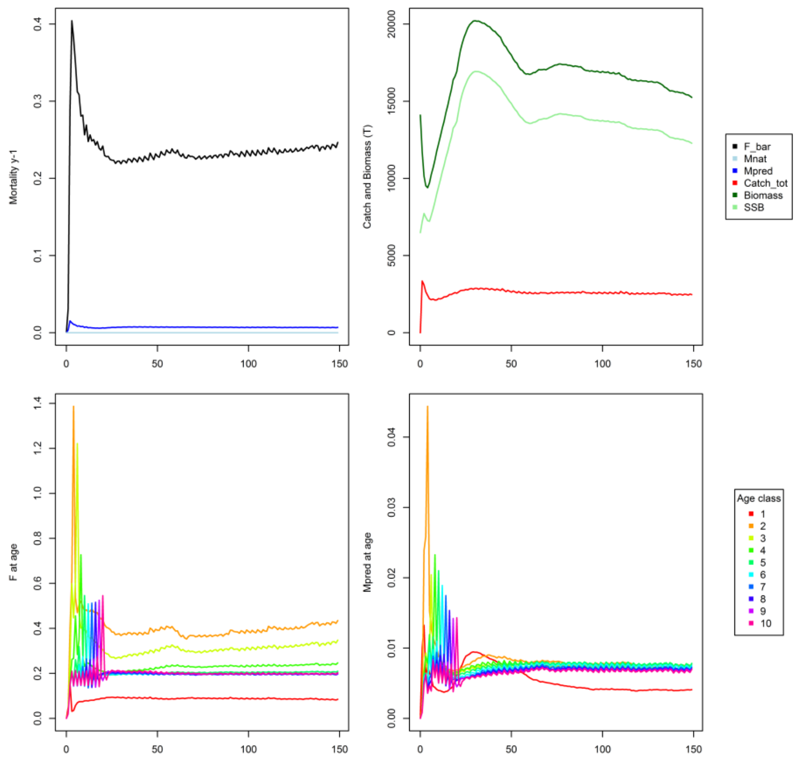


Appendix XI (continued)

Plaice: Entire model Fishing

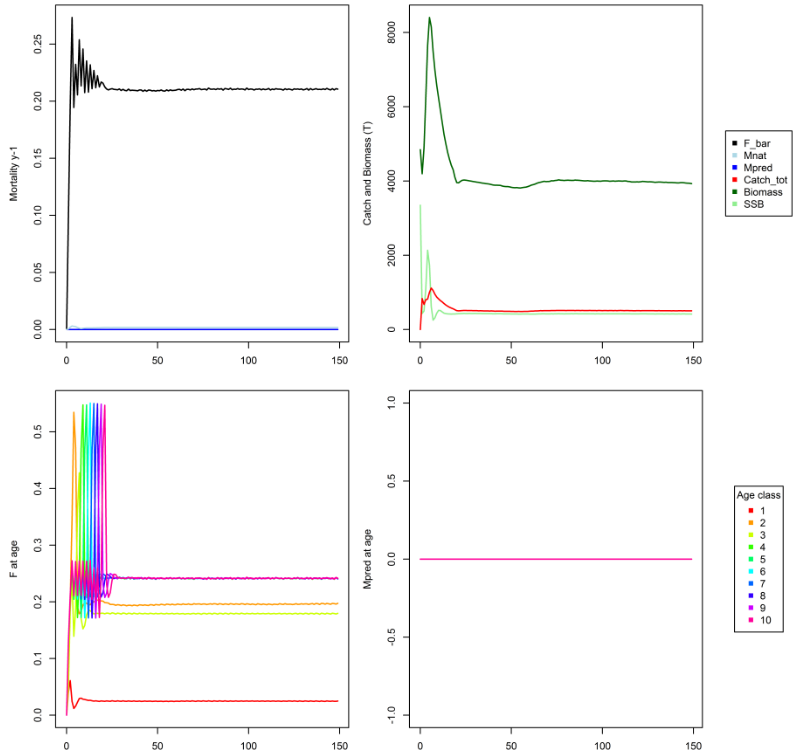


Sole: Entire model Fishing

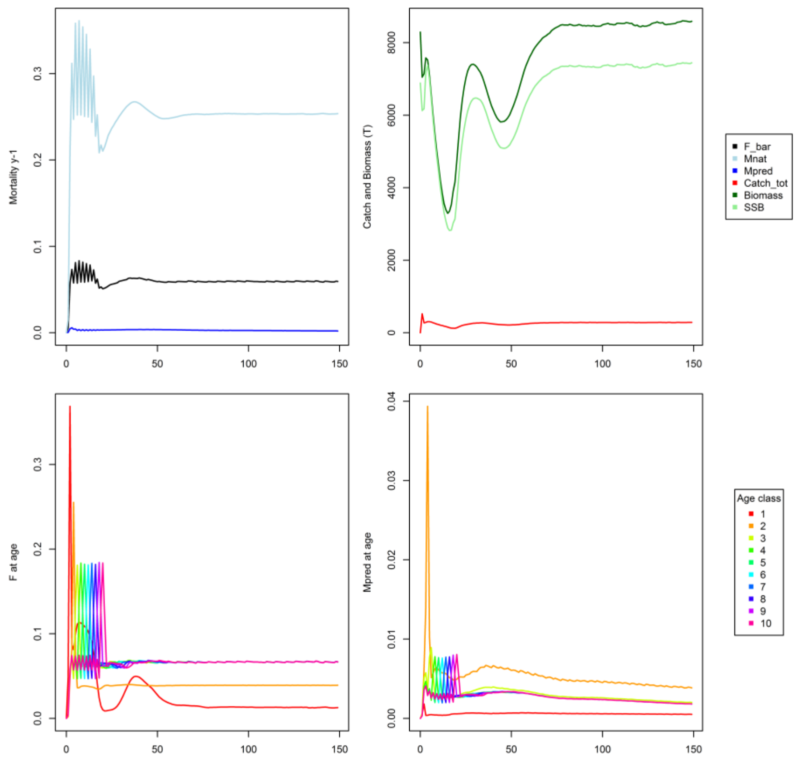


Appendix XI (continued)

Seabass: Entire model Fishing

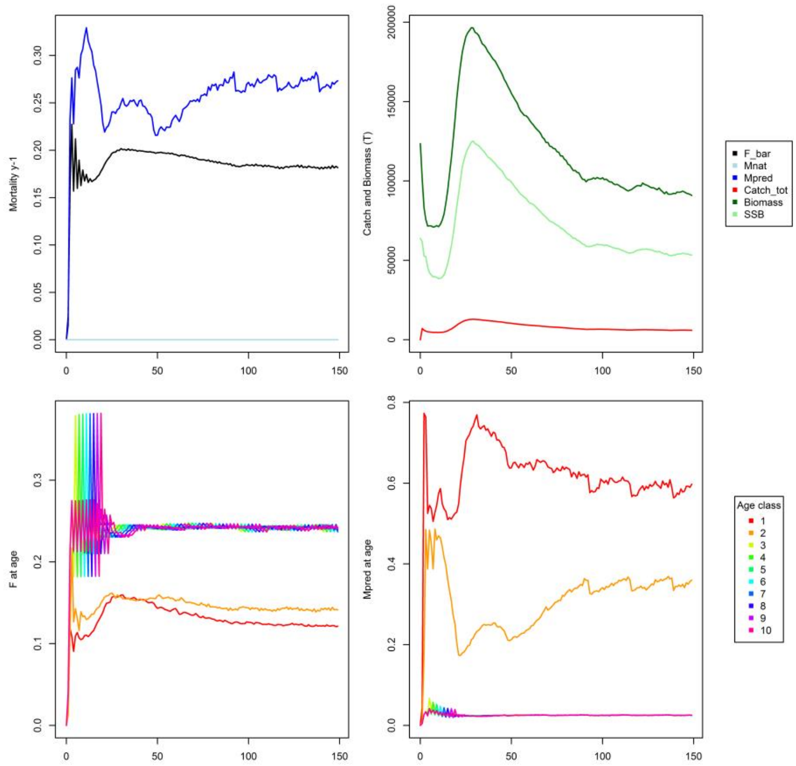


Pollack: Entire model Fishing

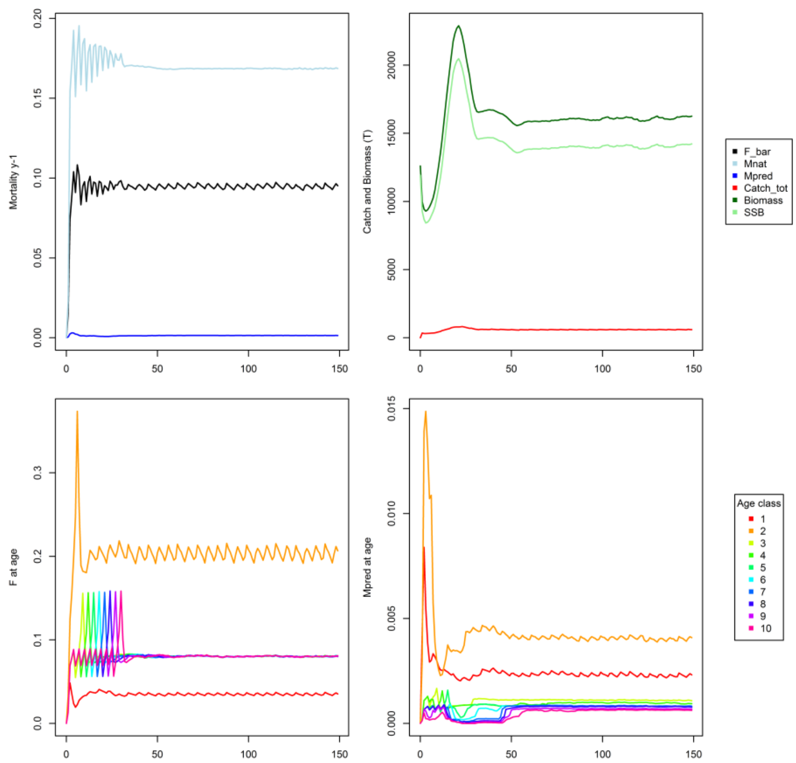


Appendix XI (continued)

Whiting: Entire model Fishing

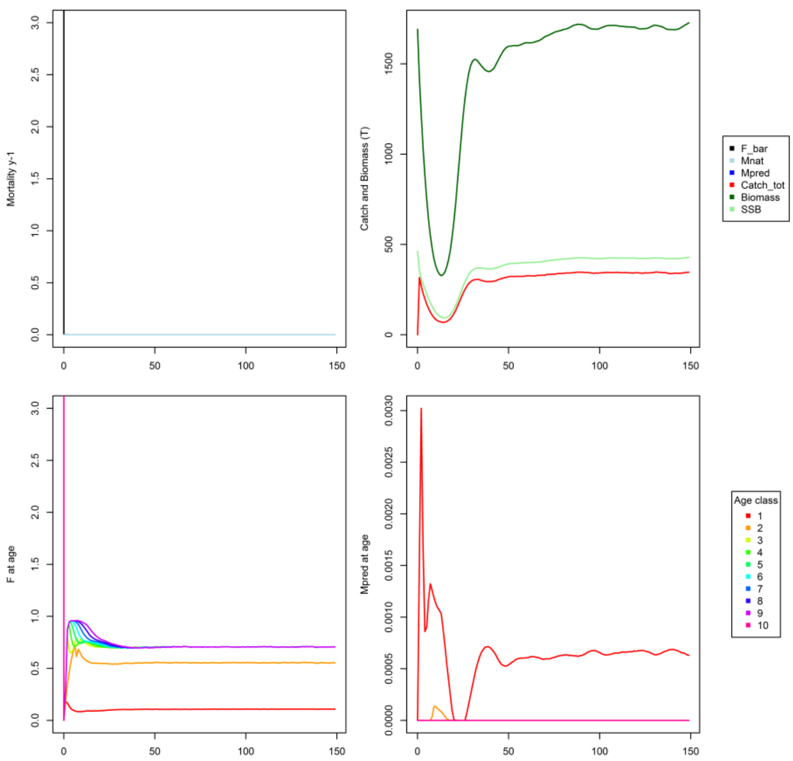


L_Bottom: Entire model Fishing

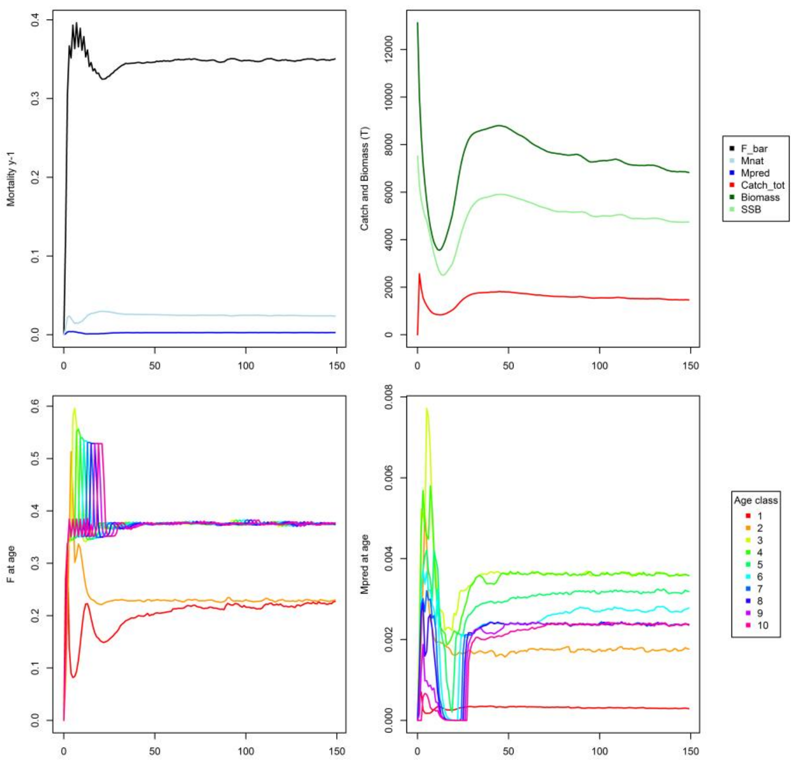


Appendix XI (continued)

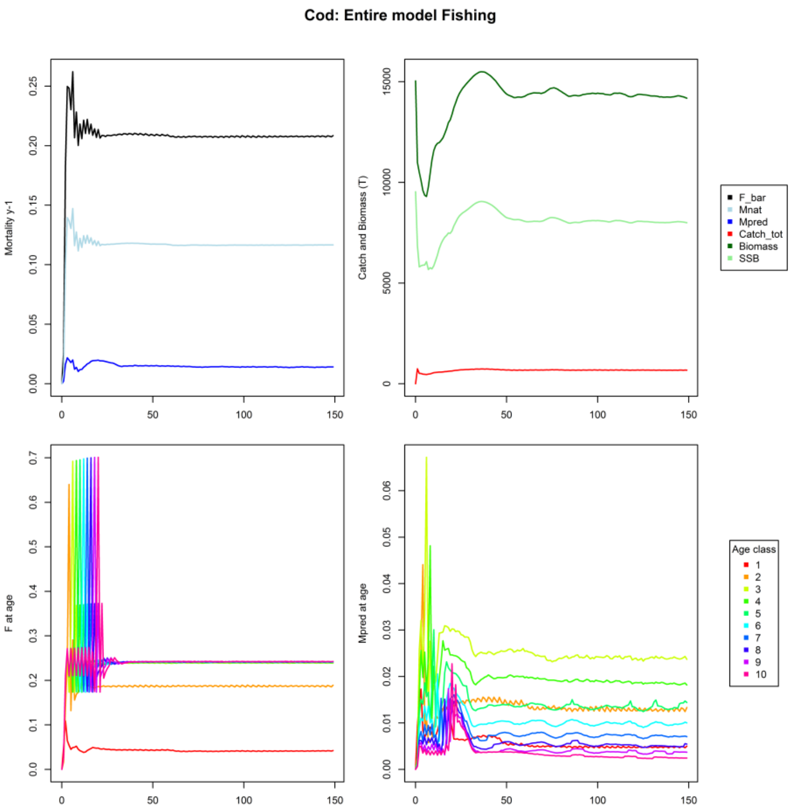
Sharks: Entire model Fishing



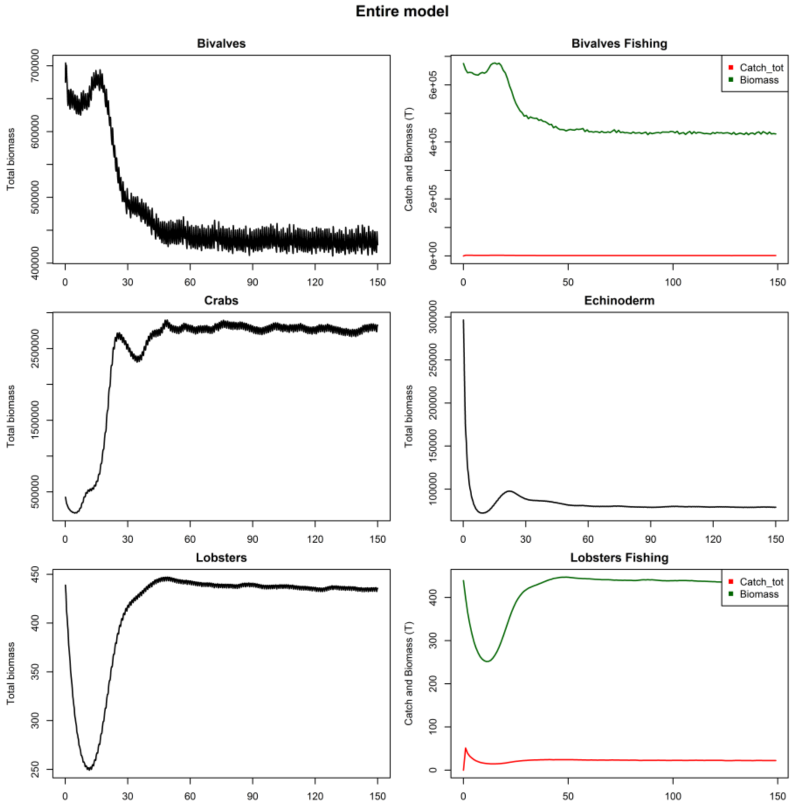
Rays_Dogs: Entire model Fishing

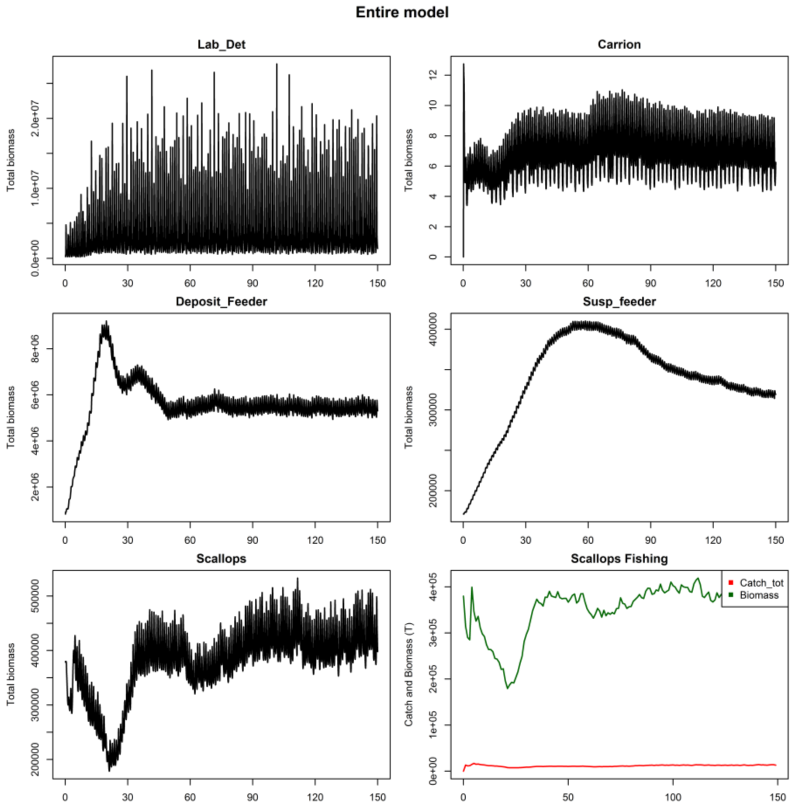


Appendix XI (continued)

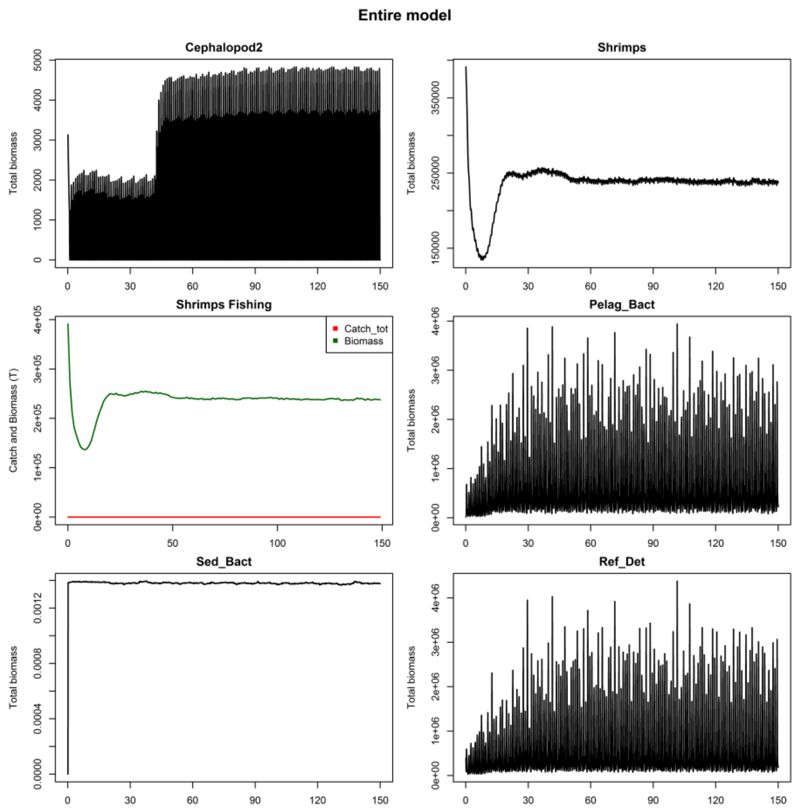


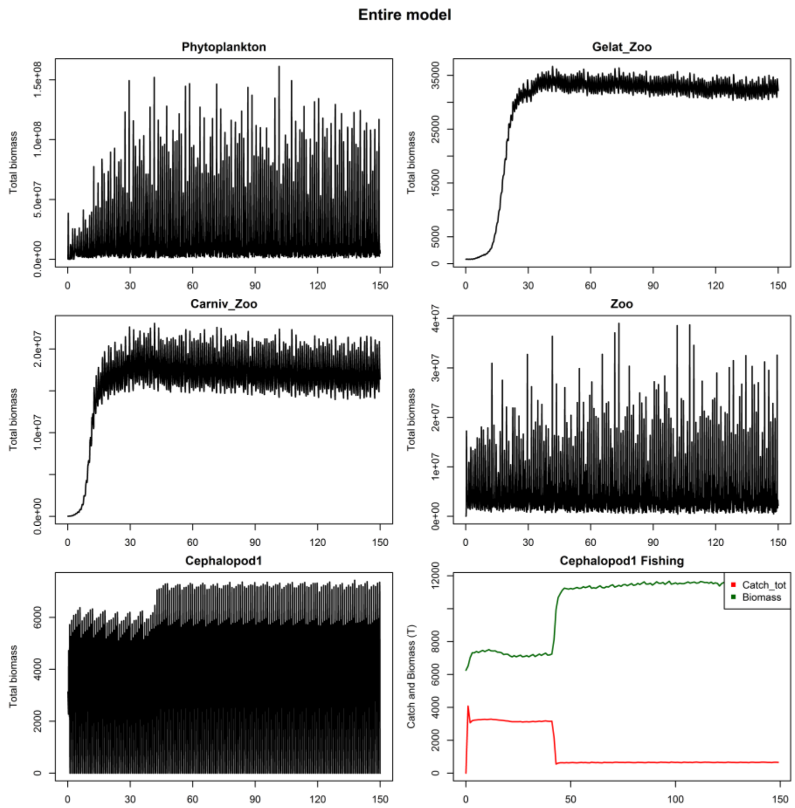
Appendix XII : Total biomass of invertebrates' functional groups from calibrated Atlantis EEC. When functional groups are fished we present in red the total catch compared to the biomass in green.



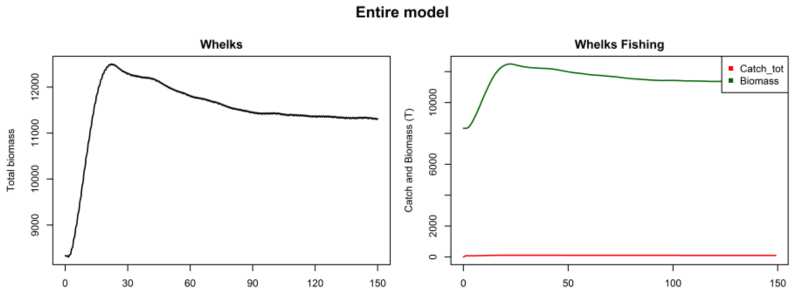


Appendix XII (continued)

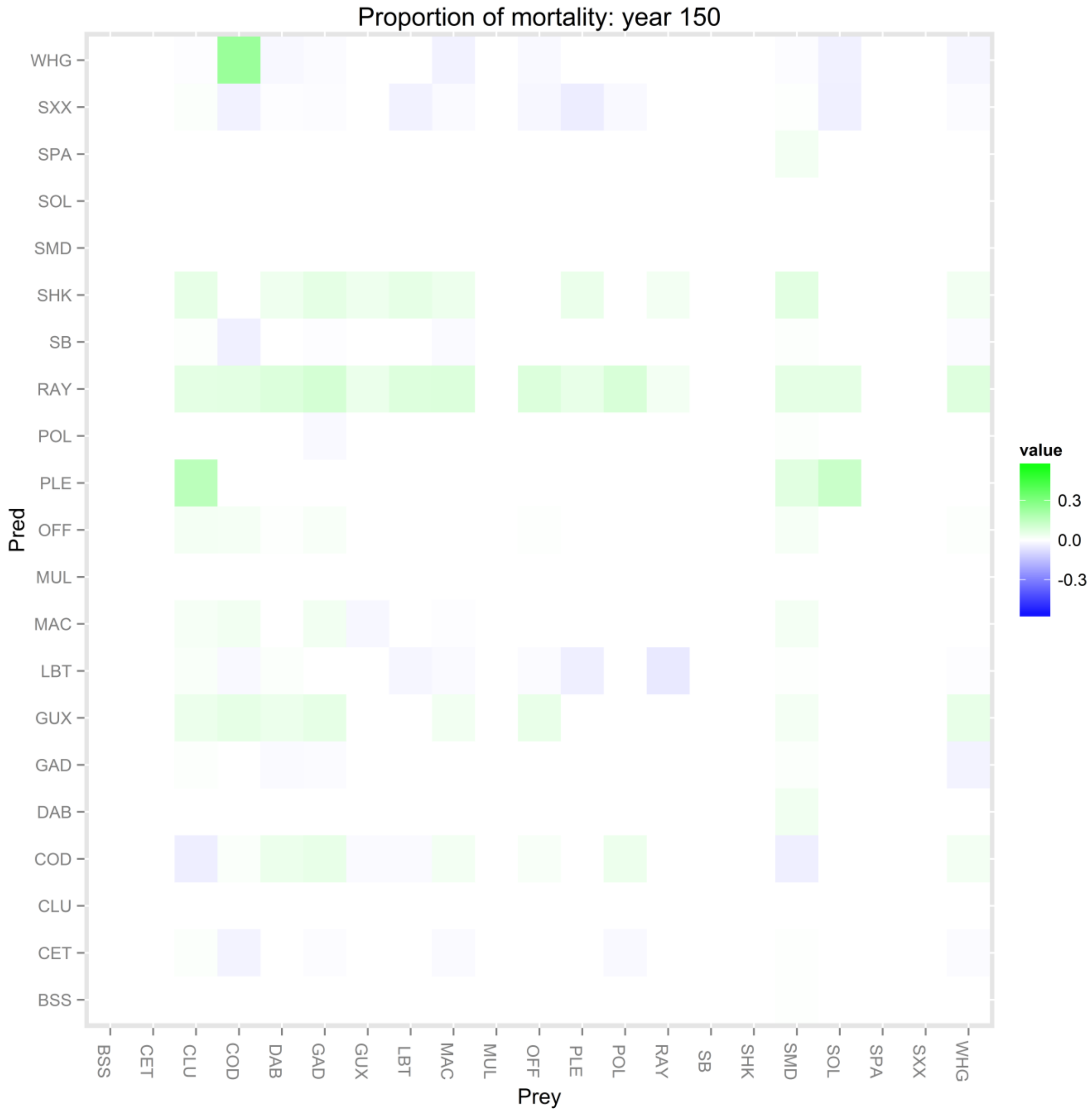




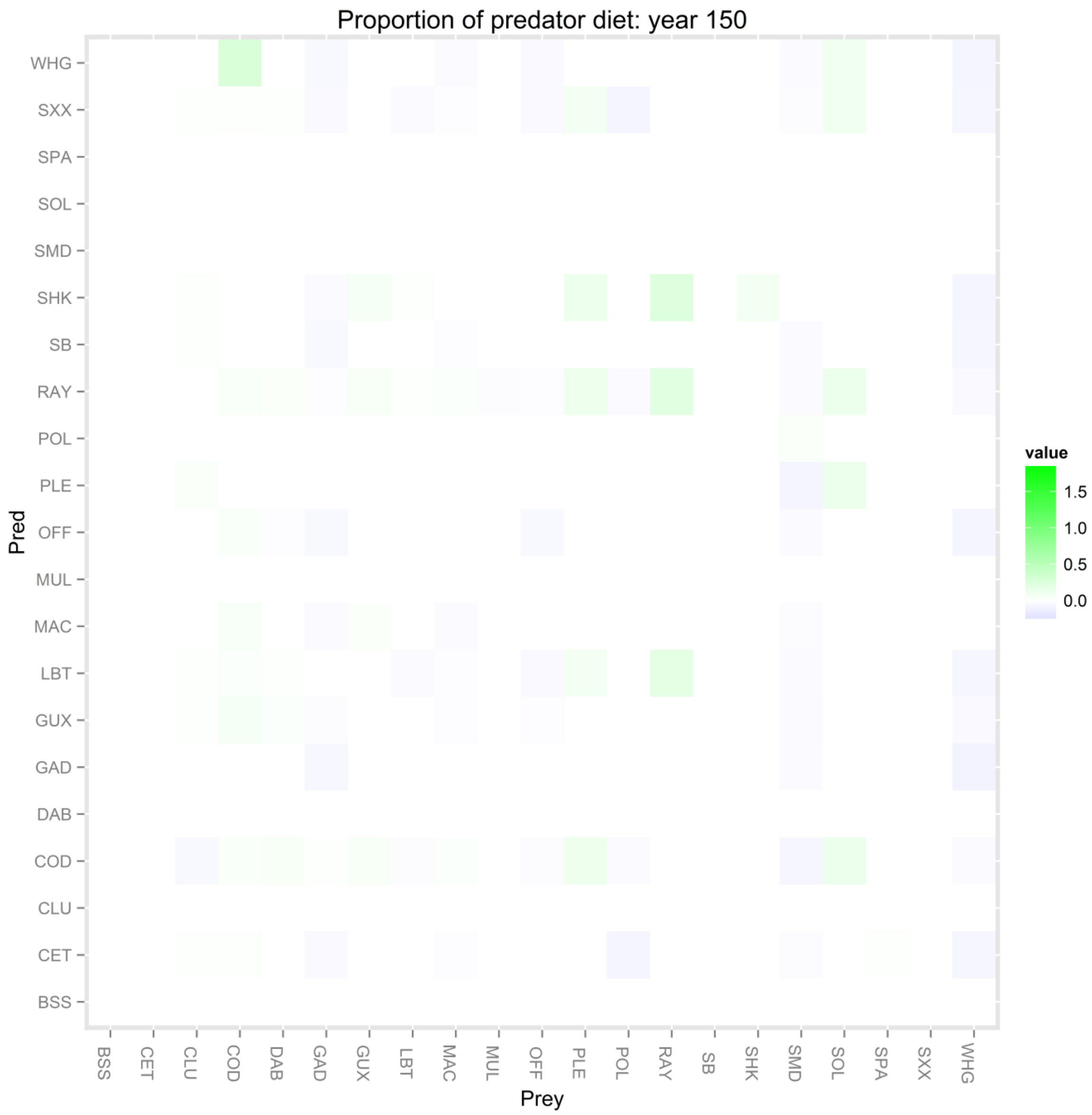
Appendix XII (continued)



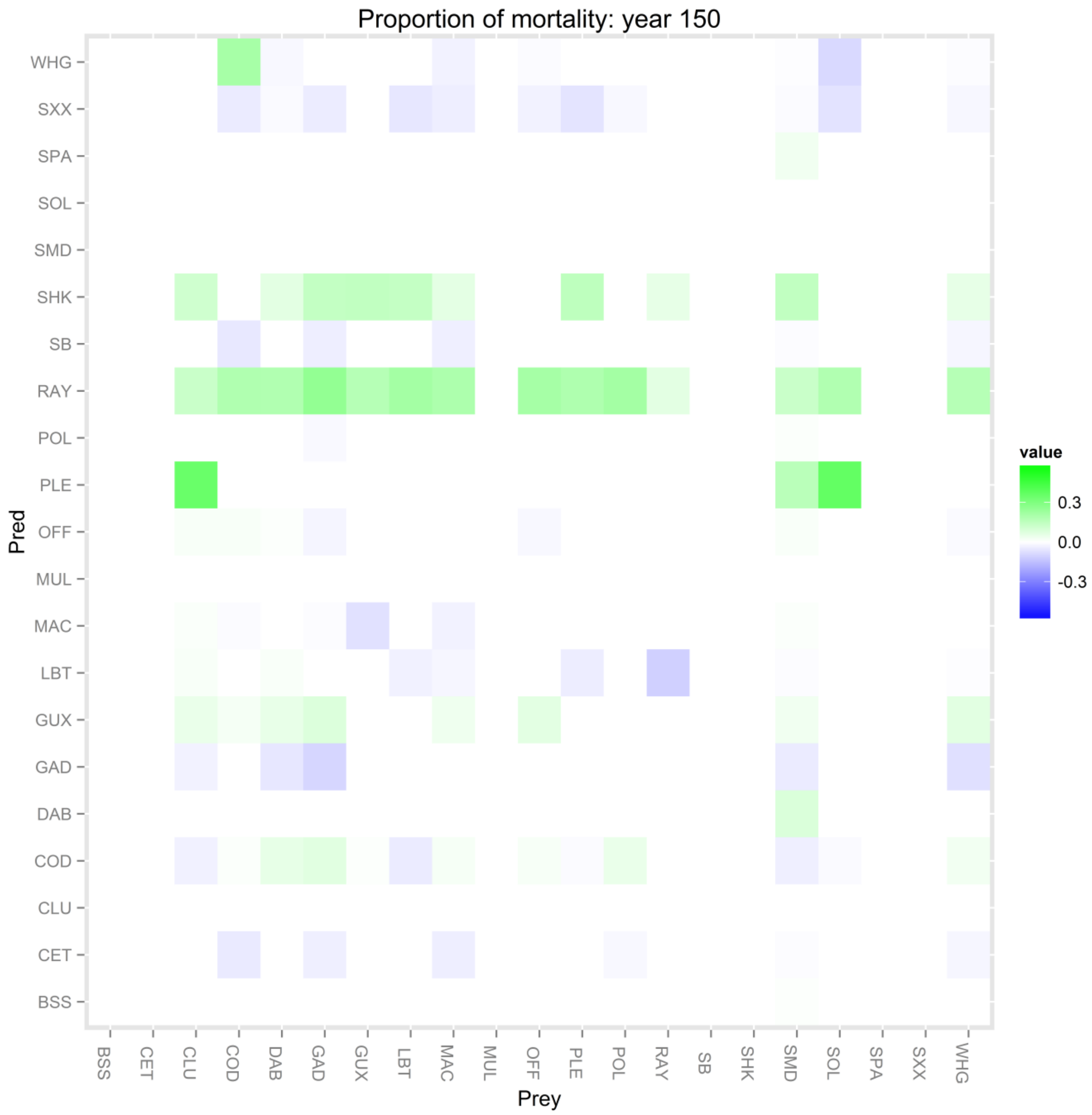
Appendix XIII : Predator-prey relationship modification after 50 years of simulating the area closure management scenario compared to the status-quo: variation of the proportion of mortality per predation explained per predator.



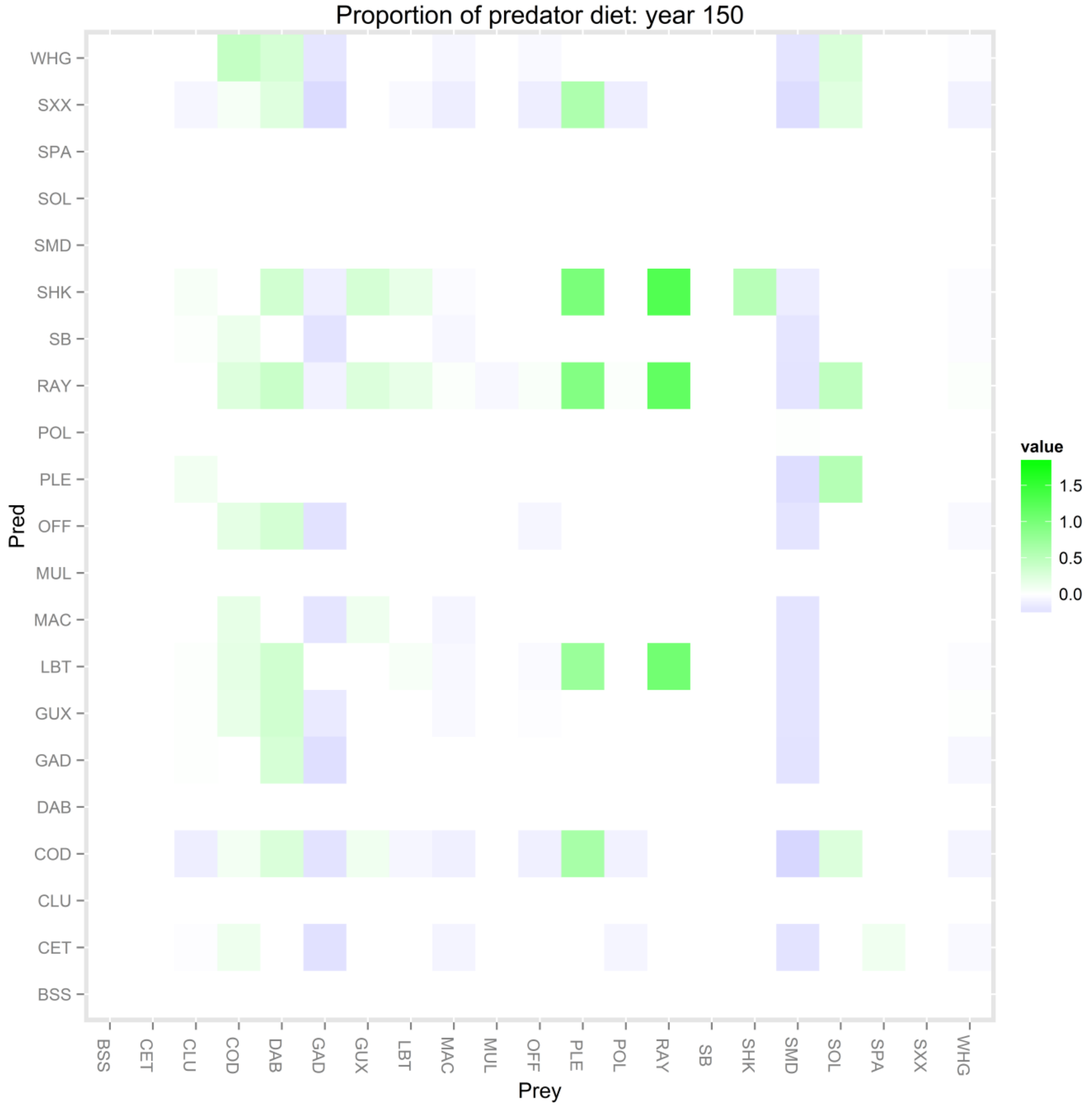
Appendix XIV : Predator-prey relationship modification after 50 years of simulating the area closure management scenario compared to the status-quo: variation of the proportion of preys in predators' diet.



Appendix XV : Predator-prey relationship modification after 50 years of simulating the 20% effort decrease management scenario compared to the status-quo: variation of the proportion of mortality per predation explained per predator.



Appendix XVI : Predator-prey relationship modification after 50 years of simulating the 20% effort decrease management scenario compared to the status-quo: variation of the proportion of preys in predators' diet.



Appendix XVII : CPUE of the main species and total VPUE for each scenario and each métier after 2 years of simulation. Black dashed line, status-quo; red line, area closure scenario; blue line, 20% effort decrease scenario; green line, scenario combining area closures and 20% effort decrease.