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Impact of a landing obligation on coupled dynamics ecosystem-fishers: individual-based modelling approach applied to Eastern English Channel

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AIC: Akaike Information Criterion

BaU: Business as Usual

BTS: Beam Trawl Survey

CCR: Constant Catch rate Ratio model

CFP: Common Fisheries Policy

CG: Centre of Gravity

CGFS: Channel Ground Fish Survey

CTC: cuttlefish

COD: cod

COMOR: Coquilles Manche Orientale / Scallops Eastern English Channel

CPUE: Catch Per Unit of Effort

DPMA: Direction des Pêches Maritimes et de l'Aquaculture / Marine Fisheries and Aquaculture Administration

DRB: Dredge

DSVM: Dynamic-State Variable Model

EAF: Ecosystem Approach to Fisheries

EBFM: Ecosystem-Based Fishery Management

EC: European Commission

Eco-eco: Ecological-economic

EEC: Eastern English Channel

EU: European Union

FAO: Food and Agriculture Organization

Fcube: Fleet and Fisheries Forecast

fe: effective fishing effort

fn: nominal fishing effort

FPO: Pots and traps

GLM: Generalized Linear Model

GOV: Grande Ouverture Verticale / Very High Vertical Opening (VHVO)

GTR: Trammel net

HCR: Harvest Control Rules

IBM: Individual-Based Model

IBTS: International Bottom Trawl Survey

ICES: International Council for the Exploration of the Sea

IFD: Ideal Free Distribution

ISIS-Fish: Integration of Spatial Information for Simulation of FISHeries

Isodar: Iso-Darwin

JK: Jackknife

L50: 50% retention length

LIC: Local Index of Collocation

L_s: common length threshold above which a species is considered to be correctly selected by different gears

L-BFGS-B: Limited-memory quasi-Newton Broyden-Fletcher-Goldfarb-Shanno bound-constrained optimization

LFI: Large Fish Indicator

LO: Landing Obligation

LTL: Lower Trophic Levels

MAC: mackerel

MEV: Moran’s EigenVectors

MLS: Minimum Landing Size

MML: Mean Maximum Length

MoHuB: Modelling Human Behavior

MSY: Maximum Sustainable Yield

MTI: Marine Trophic Index

MUR: red mullet

NEA: NorthEast Atlantic

OBSMER: OBServations des captures en MER / Observation of catches at sea

OSMOSE: Object-oriented Simulator of Marine Ecosystems Exploitation

OTB: Bottom Otter Trawl

OTM: Midwater Otter Trawl

POM: Pattern-Oriented Modelling approach

PTM: Pelagic Pair Trawl

Q: Quartile

RSSQ: Residual Sum of Squares

RUM: Random Utility Model

SAC: Spatial AutoCorrelation

SIH: Système d'Information Halieutique / Fisheries Information System

SSB: Spawning Stock Biomass

SQZ: squids

STECF: Scientific, Technical and Economic Committee for Fisheries

TAC: Total Allowable Catches

VMS: Vessel Monitoring System

VPUE: Value Per Unit of Effort

WHG: whiting

Chapter 1

Introduction

1.1 Fisheries management overview

Human beings have contributed to animal extinctions since at least late Pleistocene, through hunting activities in particular (Miller *et al.*, 1999; Roberts *et al.*, 2001). Humans became aware of the need to adapt their behaviour to environmental fluctuations since millenaries (Diamond, 2006). Such awareness has led societies to manage their natural resources, e.g., the forest management in Japan during Tokugawa period (1600-1868). It is now evidenced that humans have tried to adapt to the marine coastal environment more than 125,000 years ago (Rick and Erlandson, 2008), and that fish has been a source of protein for them since millenaries (Hu *et al*., 2009). Nowadays, marine species are an important source of proteins for human consumption, representing 17% of animal proteins consumed worldwide (FAO, 2016). However the growth potential of wild fish and shellfish fisheries, in terms of protein supplies, is considered limited as global catch has reached a plateau since mid-1990s (FAO, 2016). Since overexploitation has proved to be one of the most impacting anthropic pressures on fisheries resources and their embedding marine ecosystems (Dayton *et al.*, 1995; Jackson *et al*., 2011; Worm *et al*., 2009), societies progressively implemented management measures to mitigate the conservation impacts of fishing activities, even at an early stage (e.g., in the late-1860s in Northwest Atlantic: Anderson, 1998). Management measures implemented worldwide involve a large variety of tools, and can be classified in two categories: input and output controls, even if other classifications exist (e.g. active and passive regulations: Sissenwine and Kirkley, 1980).

1.1.1 Classification of fisheries management measures in Europe

Morison (2004) proposed definition of *input* and *output* controls in fisheries management, which reconciles previous proposals. Output controls focus on "*what* are fishers allowed to fish", while input controls concern "*who*, *where*, *when* and *how* they are allowed to fish". Output controls directly constrain the catch in a fishery, either quantitatively with Total Allowable Catches (TACs; Karagiannakos, 1996) being the most widespread instrument, taxes on the landings or full discard ban (e.g., Canada and New-Zealand; Hall and Mainprize, 2005), or qualitatively, e.g. minimum landing size, limits established based on the sex and/or maturity of the harvest (e.g. the prohibition to harvest egg-bearing female lobsters in the Maine during the 1870s; Ostrom, 2007). Input controls constrain directly the fishing effort, by constraining *who* may fish using access controls (e.g., licences), *when* to fish (e.g., closed seasons), *where* to fish (e.g., closed areas), and *how* to fish (limitations imposed to vessels/gears catching capacity). Each of these management measures can also be further decomposed into quantitative and qualitative instruments (Table 1.1).

Table 1.1. Examples of controls of fishery inputs and outputs, classified as either quantitative or qualitative control measures (source, Morison, 2004).

	Qualitative	Quantitative
Controls on fishery inputs	Closed seasons	Number of licences
	Closed areas	Number of pots
	Types of gear	Number of rods and hooks
	Mesh sizes	Length of nets
Controls on fishery outputs	Protected species	Number of fish
	Size limits	Weight of fish
	Sex	Landing obligation
	Maturity stages	Taxes on the landings

1.1.2 Efficiency of management measures

In worldwide fisheries, management is currently based on a set of complementary input and output controls (e.g. licence, TACs, size limits). In the European Union (EU), fishing effort and the resulting exploitation of fisheries resources have increased considerably during the second part of the 20th century. This has resulted on a gradual strengthening of management measures after the severe decline of many commercially valuable fish (Suuronen and Sardà, 2007; Sparholt *et al*., 2007; Froese and Proelß, 2010), especially in the Northeast Atlantic (Christensen *et al*., 2003). Although important key stocks have recovered (or are recovering) since the beginning of the 21th century, a number of them remain exploited above Maximum Sustainable Yield (MSY; Maunder, 2008) level, while increased concerns have been expressed that the functioning of marine ecosystems has been adversely affected (Cardinale *et al.*, 2013; Fernandez and Cook, 2013; Gascuel *et al*., 2016; see also Figure 1.1).

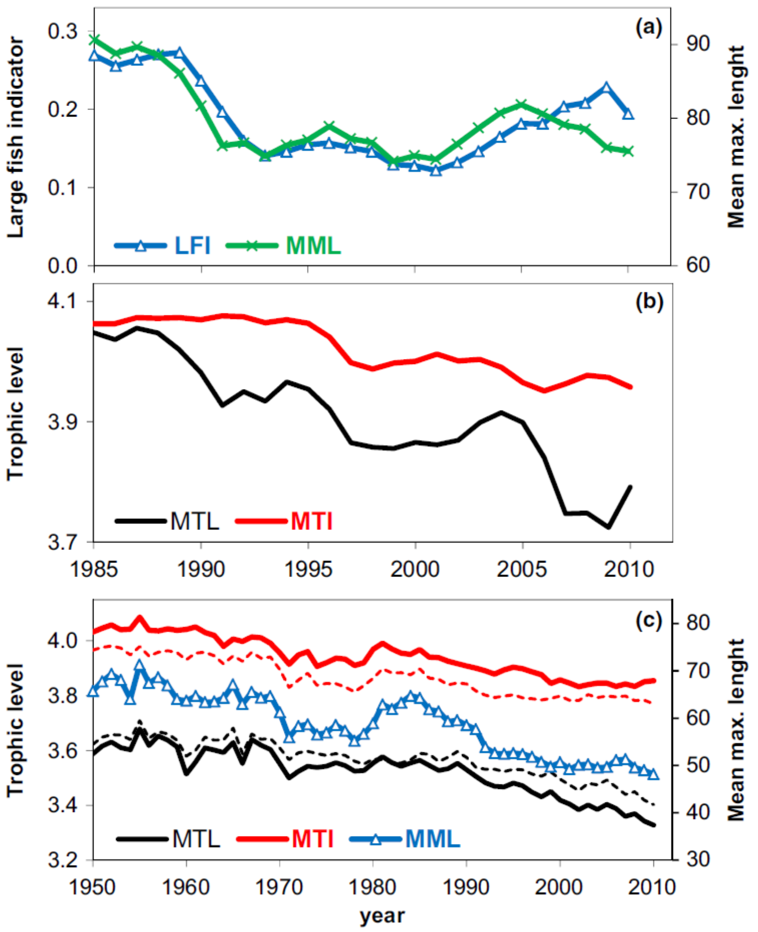

Figure 1.1. Mean trends of ecosystem indicators in European seas: (a) length-based indicators from surveys, (b) trophic level-based indicators from surveys, (c) indicators from commercial fishery landings. LFI, large fish indicator (proportion); MML, mean maximum length (cm); MTL, mean trophic level; MTI, marine trophic index. (Dotted lines in graph c relate to the sensitivity analysis, using trophic levels from local Ecopath models in place of standard values from Fishbase) (source, Gascuel *et al.*, 2016).

EU fisheries management represents a particular challenge, as it concerns many different countries with specificities and interests (Smith and Garcia, 2014; Marchal *et al.*, 2016), which may have delayed decision-making compared to other worldwide jurisdiction (Hyder *et al.*, 2015; Lehuta *et al.*, 2016). The overarching framework of EU fisheries management is the Common Fisheries Policy (CFP), with single-species TACs representing the main implementation tool in the Northeast Atlantic (NEA). TAC-setting is scientifically informed each year by the International Council for the Exploration of the Sea (ICES), and following the objectives set by the European Commission and the CFP for many commercial species. The scientific advice uses population dynamics models, which are commonly fueled by time series of catches (landings when a discard estimate is not available) and abundance indices derived from scientific surveys or commercial fisheries.

TACs are usually calculated on a single-species basis (Vinther *et al.*, 2004), even if multispecies assessment begins to emerge (Lewy and Vinther, 2004; Kempf *et al.*, 2010; Plagányi *et al.*, 2014), by assuming that harvested fish stocks are ecologically isolated and that they are caught independently by the fishing gears, a situation which only occurs in rare mono-specific fisheries (e.g., some pelagics). However in mixed fisheries, which represent the most common case, a great diversity of species is caught simultaneously (ICES, 2017), by an equally large variety of gears (Marchal, 2008; Wilson and Jacobsen, 2009, Ulrich *et al.*, 2011; Prellezo *et al.*, 2012), each of them having its own biological specificities (Pelletier and Ferraris, 2000; Poos *et al.*, 2010). Mixed fisheries can then create situations where fishers may catch fishes over quota, whilst targeting other species for which they still have a catch allowance, thereby inducing discarding (Catchpole *et al.*, 2005; Johnsen and Eliasen, 2011) and/or highgrading practices (i.e. the decision took by a fisher to discard fish of low value to land more valuable fish; e.g. Batsleer *et al.*, 2015). Consequently, in a system where discards and highgrading were hardly estimated and hard to predict without a good assessment of discard practices, catch-based stock assessments have been biased, while single-species TACs have regulated landings rather than catches. As a result, fishing mortality has been poorly monitored and controlled (Ulrich *et al.*, 2012), eventually causing management failures (Daw and Gray, 2005; Penas, 2007; Carpenter *et al.*, 2016).

In an attempt to address the mixed fisheries challenge, a métier-based approach has been progressively implemented to monitor EU mixed fisheries, with a métier describing a group of vessels with similar gear, targeting the same species or assemblage of species during the same period of the year and/or the same area (Deporte *et al.*, 2012; Ulrich *et al.*, 2012): e.g. bottom otter trawlers targeting gadoids in Celtic Sea. This approach has also sometimes been advocated to manage fisheries in combination with an effort control, although it never reached full implementation in the EU.

Overall, the difficulties to account for technical (but also trophic) interactions in TAC-setting have adversely altered the efficiency of mixed fisheries management (Vinther *et al.*, 2004; Ulrich *et al.*, 2012), deterred the credibility of both scientists and managers (Rijnsdorp *et al.*, 2007), and created disincentives for fisher to comply with prevailing regulations (King and Sutinen, 2010). Mixed fisheries thus represent a real challenge requiring extensive understanding of biological and human elements: multiple stocks dynamics and their interactions, operating fleets, fishers' behaviour, gear characteristics, gear-fish interactions, and fishers-regulation interactions.

Fulton *et al*. (2011) thus proposed that fisheries science should shift from a resource state-oriented to a human components-based research to support management more effectively (Figure 1.2), following earlier concerns raised by Hilborn (1985). According to these authors, uncertainty in management due to human components has three main reasons: i) subordination of scientific advice, ii) inadequacy of regulatory control and, iii) unexpected behaviour of resource users.

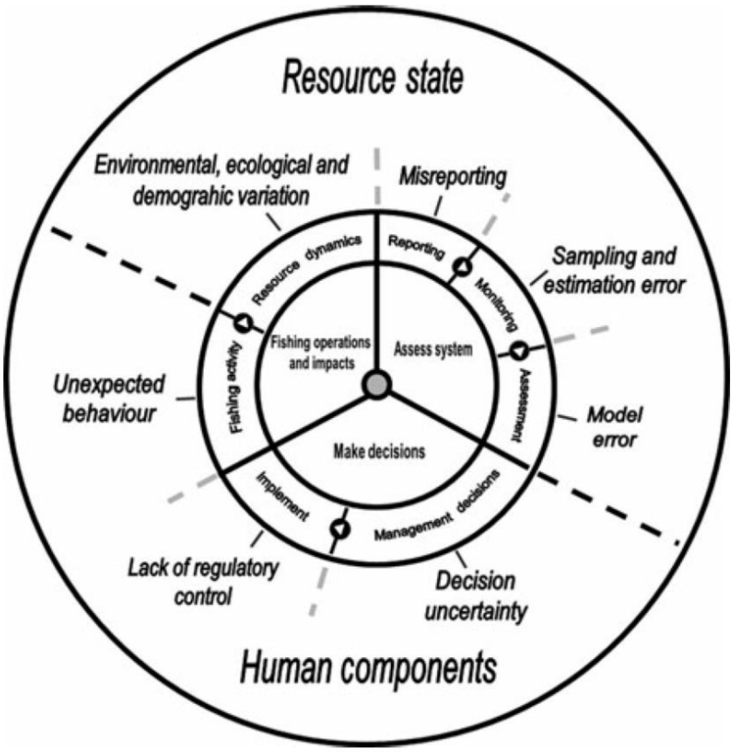

Figure 1.2. Schematic diagram of the management cycle and sources of error or variation that can inhibit the successful execution of fisheries management (source, Fulton *et al*., 2011).

Unexpected behaviour of resources users is a common event (Branch *et al*., 2006), and the overall lack of understanding of fisheries can be an explanation of management failures (Degnbol and McCay, 2007). As presented by the authors, examples of unexpected results from input and output controls are numerous. Concerning input controls, time and area closure restrictions can induce a concentration of fishing effort in specific seasons and/or fishing grounds (Briand *et al*., 2004; Rijnsdorp *et al*., 2001; Dinmore *et al*., 2003; Abbott and Haynie, 2012). Output controls can induce uncontrolled overcapacity (Beddington *et al*., 2007), economic disparities (White and Mace, 1988), misreporting (Kolody *et al*., 2008;

Polacheck and Davies, 2008), but also highgrading and/or discards (Kristofersson and Rickertsen, 2009; Depestele *et al*., 2011; Batsleer *et al.*, 2015; Fernandes *et al*., 2015).

1.2 Landing obligation: a new European fisheries policy to reduce discards

As part of the 2013 revision of the CFP, a binding landing obligation (LO) is being progressively implemented to EU fisheries over the period 2015-2019, with the aim of banning discards so far as possible (European Commission, 2013; Borges, 2015). The fundamental interest of a LO is to encourage fishers to be more selective and thus minimize the discarded part of their catches.

1.2.1 The discards issue

As already discussed in Section 1.1, fishers operating a mixed fishery may be incentivized to discard fish when their catch quota is exceeded. However, discarding may also concern undersized fish, marine organisms with low or no economic value (highgrading) and by-catch species (Alverson *et al.*, 1994; FAO, 1996; Feekings *et al.*, 2015). The amount of discards can vary according to numerous reasons (e.g. fishing area, season, environmental conditions or fishing methods, see Rochet and Trenkel, 2005). Bellido *et al*. (2011) computed the amount and rate of discards in the catch for different fisheries around the world, based on Kelleher (2005), and found that shrimp trawl was by far the most discarding fishery in rate and tonnages. Demersal finfish trawl are the second discarding fishery in terms of tonnage but only fifth in rate with 9.6% in average (with a range between 0.5 and 83%; e.g., 40% of the catches from demersal fisheries in the North Sea; Quirijns and Pastoors, 2014). Generally, small-scale fisheries are assumed to have a lower discard rate than large-scale ones (Kelleher, 2005; Jacquet and Pauly, 2008), even if exceptions exist (e.g. Shester and Micheli, 2011). Discards represented between 7 and 27 millions of tons each year in EU during the 1990's, one quarter of the total catches (European Commission, 2002).

Most fishes die when discarded (Evans *et al.*, 1994; Davis, 2002; Broadhurst *et al.*, 2006), resulting in two impediments. First a high amount of fishes is released dead in the sea, representing a waste as dead fishes will not contribute to stock dynamics while it could have been used for human consumption. Secondly, unreported discarding may lead to biased estimates of fishing mortality, if only the landed part of the catch is considered as input into stock assessment (Borges *et al.*, 2005), resulting in a poor scientific basis for TAC-setting (Crowder and Murawski, 1998; Cotter *et al*., 2004; Rätz *et al*., 2007). This is particularly true when discard practices fluctuate among time (recruitment fluctuation, market impact…) and are not accurately estimated via on-board sampling programs for instance.

Condie *et al*. (2014) reviewed the implementation and impacts of previous LOs operated in several countries/states. They explored the case of Alaska, British Columbia, Faroe Island, Iceland, Norway and New Zealand. Other studies examined the link between discard ban and quotas in the efficiency of management (Condie *et al.*, 2013, 2014b; Hatcher, 2014). All of these studies emphasized the need of additional management measures to obtain a real improvement of selective practices, such as bycatch limits in Alaskan walleye Pollock fishery or area closures in Norway. However the authors also warned that even with more selective fisheries, sustainable fisheries could not be reached without efficient long-term management policies. The good results with Norwegian LO policy encouraged some author on the necessity to use an analogous discard management in Europe (e.g., in North Sea; Diamond and Beukers-Stewart, 2011).

1.2.2 The EU implementation of the landing obligation

In the EU, the LO concerns all the NEA species regulated by TACs, but also Mediterranean species with a minimum landing size. Prohibited species (e.g., angelshark *Squatina squatina,* basking shark *Cetorhinus maximus*) still have to be discarded, while species not regulated by TACs (with the exception of Mediterranean species) can still be discarded. However some species have survivability exemptions: e.g. Norway lobster caught by pots, traps or creels in ICES subareas 27.6 and 27.7 (European Commission, 2015), cod and salmon caught with trap-nets, creels/pots, fyke-nets and pound nets in Baltic Sea (European Commission, 2014). Some *de minimis* exemptions also allow some fisheries to discard a portion of their catches for the first years of the LO application, particularly when selectivity is difficult to achieve for these fisheries (e.g. up to 7% in 2017 and 6% in 2018 for whiting (*Merlangius merlangus*) by bottom trawls and seines with mesh size less than 100mm in ICES 27.7d and 27.7e). Quota swaps between years or even species are also envisaged by the legislation.

The LO is sequentially implementing, and started in 2015 with the pelagic fisheries because they target schooling species, and thus are more selective (Marchal, 2008). It is now progressively applied to demersal fisheries which are typical mixed fisheries (Hall and Mainprize, 2005).

As a consequence of the LO, incentives to promote selectivity in the EU are indirectly based on the quota limitations. Thus, when fishers have reached their quota for a given species (i.e. “choke species”; Schrope, 2010; Baudron and Fernandes, 2014), they have to stop their activity if the gear used is likely to catch this species.

1.2.3 Technical and strategical fishers’ levers

As a consequence of the LO, fishers will have to change their fishing practices if they do not want to stop their activity earlier during the year and be unable to go fishing species with remaining quota as well as other targets and by-catches (Catchpole *et al*., 2017). Fishers have two main solutions to comply with the policy. The first one is to improve their direct selectivity using more selective gears, following French fishers’ slogan: “*trier sur le fond plutôt que sur le pont*” that could be translated by “*to sort on the seabed rather than on the deck*”. To achieve this goal, experimental research was conducted in collaboration with fishers (e.g., Armstrong and Revill, 2010) and documents are proposed for professionals (e.g., O’Neill and Mutch, 2017). Studies have also been performed to evaluate the bioeconomic consequences (Prellezo *et al.*, 2017), perceptions (Villasante *et al.*, 2016) and interests (Batsleer *et al.*, 2016) of a change in gear characteristics for fishers. Lots of work was done and is actually ongoing to improve selectivity by the assessment on ancient and new devices (e.g. O’Neill and Noble, 2017; Vogel *et al*., 2017). The second strategy fishers could adopt would be to change their spatio-temporal allocation of fishing effort, in order to avoid discarding hotspots. Empirical work has been done by Baelde (2001), Branch and Hilborn (2008) and Branch (2009), who studied the reallocation of fishing effort according to unwanted species, namely avoidance behaviour. Eliasen (2014) also examined avoidance behaviour in the context of a LO for cod, and highlighted the importance of information sharing to avoid discarding hotspots. Through a modelling approach where avoidance behaviour was implemented, Poos *et al*. (2010), Batsleer *et al*. (2013) and Simons *et al*. (2015) demonstrated that fishers will adapt more their fishing tactics according to their remaining quota in a LO context than in a business as usual context.

1.2.4 Evaluating the impact of a Landing Obligation

If the final purpose of a LO is to diminish unwanted mortality, incentives must be sufficiently strong for fishers to try to drastically reduce the discarded part of their catches (Borges, 2015). However a fundamental question remains: ‘what is the impact of discarding, or not discarding, unwanted catch on the ecosystem functioning (Sardà *et al*., 2015)?’ Indeed environmental factors impact trophic conditions which affect the fisheries system, but the reverse is true starting from economic and management factors (Prellezo *et al.,* 2012). Such reasoning raised another concern: what is the direct socioeconomic impact of not discarding for fishers (de Vos *et al*., 2016)?

Ecological concerns relate directly to the dead discards that will not feed anymore the diverse scavenging species, including benthic invertebrates, fish, or birds (Hill and Wassenberg, 1990; Garthe *et al.*, 1996; Ramsay *et al.*, 1997; Walter and Becker, 1997; Bozzano and Sardà, 2002; Catchpole *et al.*, 2005; Groenewold and Fonds, 2000), and therefore represent a loss in biomass for the ecosystem cycling. Ecological effects of a LO could also be indirect by changing their fishing habits, fishers' catch profile will evolve, which would modify the abundance and size-structure of different target and by-catch species and *in fine* the predator-preys relationships.

Work has been done to the past to detect discard hotspots and timing (Dunn *et al.*, 2011; Viana *et al.*, 2011), and are currently dedicated to identify them in EU waters (Grazia Pennino *et al.*, 2014; Vilela and Bellido, 2015; Pointin *et al.*, submitted), for example using observations on-board commercial vessels (e.g. OBSMER French programme; Cornou *et al*., 2015), considered as a valuable source of data (Bordalo-Machado, 2006).

All these issues are central to Ecosystem-Based Management (EBFM) and the Ecosystem Approach to Fisheries (EAF) (FAO, 2001; Garcia and Cochrane, 2005; Pikitch *et al*., 2004). Ward *et al*. (2002) defined the EAF as "*an extension of conventional fisheries management recognizing more explicitly the interdependence between human well-being and ecosystem health and the need to maintain ecosystems productivity for present and future generations, e.g. conserving critical habitats, reducing pollution and degradation, minimizing waste, protecting endangered species.*"

Complex ecosystem models, for all their faults (Glaser *et al*., 2014), have the potential to promote an EAF for future management. To achieve such goal, fleets-ecosystem interactions have to be understood in-depth (Bellman *et al*., 2005; Rijnsdorp *et al*., 2012), and considering appropriate spatial and temporal scales to build in the different underlying processes in a dynamic fashion (Caddy and Garcia, 1986; Fulton *et al*., 2004; Bordalo-Machado, 2006; Crowder and Norse, 2008; Ralston and O'Farrell, 2008). Indeed, important feedback loops exist between biological and fishers components: fishers can locally deplete the abundance of a species and rebalanced predators-preys interactions which can *in fine* influence fishers' behaviour.

1.3 Objectives of the thesis and methodology envisioned

The overarching objective of this thesis is to develop knowledge on the reaction of fishers to the implementation of the LO in the EEC, and on the related ecosystem response. To address ecological, economic and social consequences of a LO, there is a real need to understand fishers' behaviour and take it into account in predictive models (Wilen *et al.*, 2002; Smith and Wilen, 2003; Marchal *et al.*, 2013). To understand the fishers' complex response to a new management regime and assess its long term effect, the dynamics of both biological and fisheries compartments must be considered. This thesis aims thereore at developing and using a predictive model representing both fleet and resource dynamics in order to simulate the effects of a LO implementation in an EU ecosystem, the Eastern English Channel.

Understanding fleets-resource interactions at fine scale (e.g. trip or tow: Vignaux, 1996; Potier *et al.*, 1997; Rijnsdorp *et al.*, 2000; Bertrand *et al.*, 2004; Sys *et al.*, 2017) requires primarily to further explore the seasonal regionalisation of fishers and resource interactions, particularly concerning a potential shift in the fishing effort allocation. Indeed the heterogeneous distribution of stocks and fleets was designed as one of the main reasons for the fluctuations of catchability (Paloheimo and Dickie, 1964; Swain *et al*., 1994; Rose and Kulka, 1999; Salthaug and Aanes, 2003; Ellis and Wang, 2007; Wilberg *et al*., 2009; Ye and Dennis, 2009). A particular attention will therefore be given to the consideration of spatio-temporal distributions of fish and fishers.

This thesis is part of the DiscardLess European H2020 project which aims at providing knowledge, tools and technologies to achieve the gradual elimination of discarding, in close collaboration with stakeholders. Withing the tasks tackled by the project, the work conducted in this thesis will contribute to assess at the ecosystem scale the expected effects of the European LO, and will also contribute to the projections of fishing strategies, in reponse to the LO constraints.

1.3.1 Modelling approaches for understanding fleet dynamics

Fishers' decision-making process can be considered at different temporal scales: in the long-term, e.g., to enter or exit a fishery (Ward and Sutinen, 1994; Pradhan and Leung, 2004; Tidd *et al.*, 2011), or in the short-term, e.g., the choice of a fishing ground and/or to discard a portion of the catch (see van Putten *et al.*, 2012 for a review). For this thesis and within the LO context, we will solely focus on the short term decisions fishers make, especially in terms of fishing effort allocation and discarding (avoidance behaviour). Moreover, technical adaptations will not be explored in this thesis.

Fishers are commonly assumed to try to maximize their expected profit (Gordon, 1953). This basic assumption, however, is subject to debates in the view of alternative drivers (Robinson and Pascoe, 1997; Pascoe and Robinson, 1998; Eggert and Martinsson, 2004; Abernethy *et al.*, 2007), particularly for small-scale fisheries (Salas and Gaertner, 2004). The importance of profit maximization as fishers' behaviour driver may depend on the scale of observations: it may not occur at the individual fisher level but rather at the firm or producer organization level (Hilborn, 2007; van Putten *et al.*, 2012). Moreover, fishers may avoid fishing grounds of potentially high short-term revenue for a variety of reasons including quota uptakes (Poos, 2010), but also for avoiding bad weather, continuing with their tradition, trying to reach a certain level of catch (volume instead of value), limiting their physical efforts (e.g., sorting time), *etc.* (Pete-Soede *et al.*, 2001; Branch and Hilborn, 2008; Holland, 2008). To encompass all these drivers, one can stipulate that fishers' goal is rather to maximize their well-being (Branch et al., 2006).

More than 30 years ago, preliminary fleet-dynamic models were developed to reproduce the dynamics of fishing behaviour (e.g., Hilborn and Ledbetter, 1979; 1985; Bockstael and Opaluch, 1983). Van Putten *et al.* (2012) reviewed the different theories underlying fleet dynamics, and distinguished between models focusing on individual behaviours and models focusing on groups of individuals. Three main approaches are used to study fishers as individuals: representing fishers as profit or utility maximizers, as foragers (Gordon, 1954; Hilborn and Kennedy, 1992) or to determine their behaviour by a rule-based approach. A detailed presentation of these three categories is given below. The fleet-dynamics models which consider fishers as part of a group are divided between game theory model (e.g. Munro, 1979) and social networks (e.g. Palmer, 1991; Ramirez-Sanchez and Pinkerton, 2009), but are not detailed here, due to their scarcity and specificities not of interest for the thesis. A focus will be made on four of the main models considering fishers as individuals for fishing effort allocation: gravity models, Random Utility Models (RUMs), Ideal Free Distribution (IFD) models and Dynamic State Variable Models (DSVM).

Gravity modelling (Caddy, 1975; Hilborn and Walters, 1987) is a simple model that predict a fisher' effort allocation and métier considering the attractiveness of an area with resource availability, its price, the cost of fishing along with the distance from the home port (e.g. Walters *et al.*, 1993; Walters and Bonfil, 1999). Gravity models are simple to implement but their simplification of the reality and their lack of consideration of uncertainty do not reinforce their interest to forecast new management regimes such as a LO.

Random Utility Models (RUMs) are discrete choice models, where fishers are assumed to give a multifactorial utility to different choices (see Girardin *et al.*, 2017 for a review). RUMs allow inputting monetary and non-monetary fishers' behaviour drivers, e.g., tradition, risk in the choice of their fishing grounds, specific species-targeting, expected costs, information-sharing, competition with other sea users, and/or management (e.g., Holland and Sutinen, 1999; Hutton *et al.*, 2004; Vermard *et al.*, 2008; Andersen *et al.*, 2012; Tidd *et al.*, 2012; Girardin *et al.*, 2015). In a meta-analysis of worldwide fleet dynamics analyses, Girardin *et al.* (2017) evidenced some fishing fleets specificities. They demonstrated in particular the importance of tradition and species-targeting for active demersal fleets, while pelagic ones where more influenced by risk-taking and vessel density. However, despite their merits in forecasting short-term fishing effort distributions, these statistical models are not designed to forecast fleet dynamics beyond the range of historical data (Girardin, 2015), a situation which is believed to occur when a new management regime, e.g., a O, is implemented.

In the foraging theory, fishers are assimilated to predators, trying to reach the highest short-term revenue. The Ideal Free Distribution (IFD) theory (Fretwell and Lucas, 1970; Fretwell, 1972) is a corollary of optimal foraging. One of the IFD predictions, when applied to fleet dynamics, is that the number of fishers and/or fishing effort will be proportional to the amount of resources and that catch rates (Catch Per Unit Effort, CPUE) will be equalized among resource users (Gillis *et al.*, 1993; Rijnsdorp *et al.*, 2000b; Swain and Wade, 2003). The IFD builds on four main hypotheses: i) fishers have equal competitive abilities, ii) no restrictions exist for effort allocation, iii) fishers have ideal knowledge of fishing grounds' local density and iv) interference competition among vessels exists in proportion to their local density (Gillis, 2003). IFD and its variants (e.g. isodars; Gillis and van der Lee, 2012) are useful to forecast fishing effort distributions one year ahead when little information is available on particular spatio-temporal scales, and may also be implemented in mixed fisheries (Gillis, 2003).

However, when it comes to evaluate the effects of abrupt changes in, e.g., regulations on fishing fleets, average behaviour models fitted with historical data are of limited use, and more conceptual approach that build in the diversity of fishers' behaviour have to be promoted (Salas and Gaertner, 2004), including Individual-Based Models (IBMs) which simulate the decisions and actions of different individual agents having their own characteristics, state and/or set of rules (e.g. Gaertner and Dreyfus-Leon, 2004; Millischer and Gascuel, 2006; Thébaud and Soulié, 2007; Bastardie *et al*., 2010).

Dynamic State Variable Modelling (DSVM) is originally a concept used in general ecology to address biological trade-offs (Houston and McNamara, 1999; Clark and Mangel, 2000). Before dying or overwintering, an animal (e.g. fish, lizard or invertebrate) has the choice to forage or reproduce, depending on its energetic reserves. Each of these choices has a probability, e.g., to die when visiting a particular patch. In recent years, DSVM has been adapted to address various fleet dynamics questions, on different management regimes, e.g., trip limits (Babcock and Pikitch, 2000); individual quotas in mixed fisheries (Poos *et al*., 2010), marine protected areas (Dowling *et al*., 2012) and discard ban (Batsleer *et al*., 2013), but also different fishers' specific behaviours, e.g., highgrading (Gillis *et al*., 1995; Batsleer *et al*., 2015), risk sensitivity (Dowling *et al*., 2015) and gear flexibility (Batsleer *et al*., 2016). In this model, each individual is a utility maximizer, able to take short-term choices, e.g., fishing or not, when/where/how to fish, discarding or not, with medium-term constraints, e.g., catching fish within quota allocation. A set of choices is computed according to the state of a fisher at each moment of the year (e.g., his quota or remaining trips), by starting from the maximum possible utility, and computing the best way to achieve it by backward calculations. When the model is run forward and that fishers experience different profit success, each of them adapts his choices at each time step according to his own condition and the previously computed set of choices. This strength makes DSVM one of the best models to evaluate the effect of new management regimes (van Putten *et al*., 2012).

Fleet dynamic models, such as those presented above, are useful to forecast short-term fishers' behaviour with no feedback from the marine resources. Forecasting impacts of new fisheries management measures on fishers and ecosystem dynamics requires coupling fleet (economic) and biological (ecological) dynamics models ("eco-eco" models). Credible predictions require a good representation of biological compartments, which is often limited in most bioeconomic fisheries models, while fleet-dynamics are usually over-simplified in ecological-oriented model (Thébaud *et al*., 2014). Indeed, bio-economic models mostly rely on a static representation of resource based on historical CPUE, independently to fishers' activities, food webs interaction nor environment (e.g. Batsleer *et al*., 2013; Doyen *et al*., 2012; Gourguet *et al*., 2013). On the other hand, some ecosystem models only use fishing mortality values or fishing effort time series to represent fishers' compartment (e.g. Guénette and Gascuel, 2009). There is a real need to move toward more combined bioeconomic modelling according to the question asked (Figure 1.3), because as Prellezo *et al*. (2012)

depicted: "The fish […] can naturally exist without the fishery, but the fishery cannot exist without the fish".

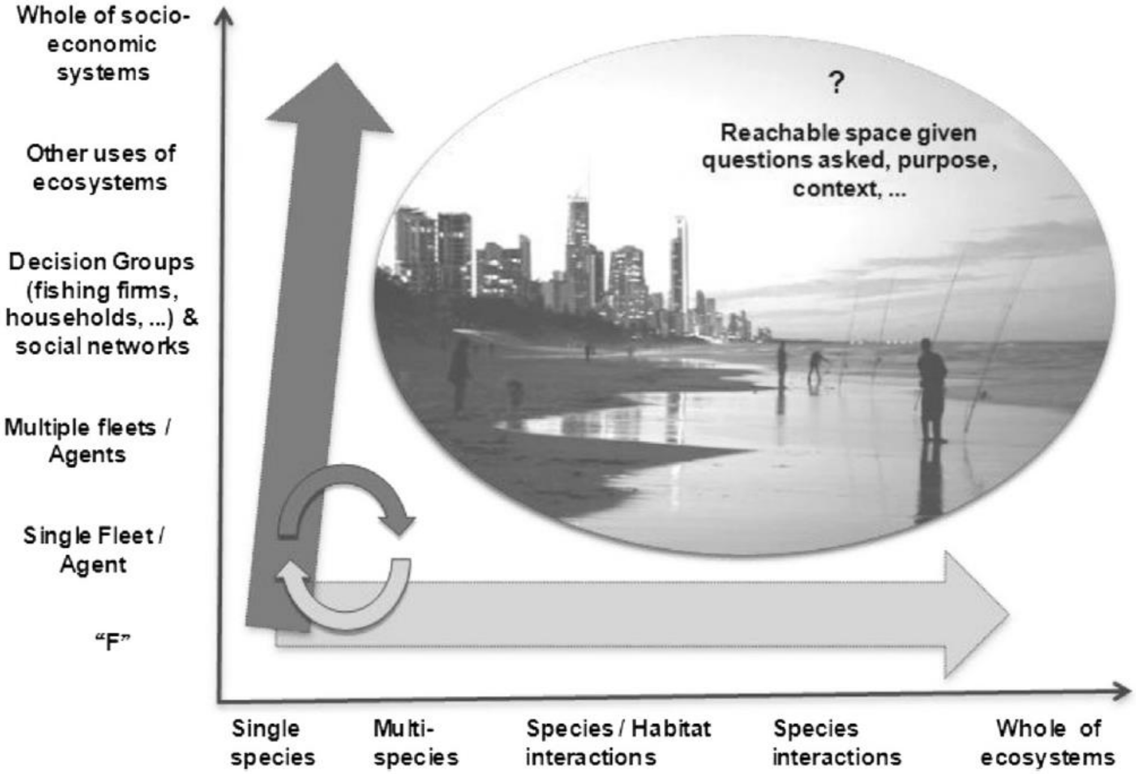

Figure 1.3. Challenges for ecological-economic ("eco-eco") modelling of living marine resource systems (source, Thébaud *et al.*, 2014).

1.3.2 Ecosystem Models

Several reviews were performed on multi-species or ecosystem models useful for an ecosystem approach to fisheries according to different criteria such as complexity, uncertainty, data requirements, spatial representation (Plagányi, 2007; Travers *et al.*, 2007; Rose *et al.*; 2010, Fulton, 2010).

In this introduction, we present the five eco-eco models used in the EU H2020 DiscardLess project: Ecospace, ISIS-Fish, Atlantis, StrathE2E and OSMOSE (Figure S1.1), with a focus on their capacity to evaluate the effects of the LO.

Ecospace (Walters *et al.*, 1999) is an extension of the Ecopath with Ecosim model. Ecopath (Polovina, 1984; Christensen and Pauly, 1992) is a mass-balanced model that gives a representation of the stable state of an ecosystem, with energy- or nutrient-related units' flows between functional groups including fishers. Ecopath is a widespread modelling approach with numerous applications existing all over the world (Colléter *et al.*, 2015). Ecosim (Walters *et al.*, 1997) is a dynamic version of Ecopath used to represent the state of an ecosystem during several years and forecast the evolution of the system under different biological- or fisheries-induced constraints. Ecospace provides spatialization into two dimensions for biological compartments (habitat preferences; Steenbeek *et al.*, 2013; Christensen *et al.*, 2014) and fisheries (profitability and possible costs). Marine species, as well as fishing activity, are able to move at the time scale of the related Ecosim model, i.e. usually the year (Christensen and Walters, 2004; Heymans *et al.*, 2016), even if smaller time scale can be implemented (the day: Orr, 2013; the season: Heymans *et al.*, 2002). In EwE and Ecospace, functional (biological) compartments are aggregated and do not have explicit size- or selectivity-specific representations, except if multi-stanza are defined. Fishers are represented by an annual fishing mortality portioned into fleet groups, and are allotted fixed landings and discard ratios, which are distributed in proportion over space according to a gravity model. Ecospace can offer valuable insights to evaluate the short-term direct impact of stopping discards on scavengers. However due to the overly aggregated temporal and size scaling, and representations, it lacks flexibility to evaluate the effects of the LO concerning fleet dynamics (effort allocation and discarding).

ISIS-Fish (Integration of Spatial Information for Simulation of FISHeries; Mahévas and Pelletier, 2004; Pelletier and Mahévas, 2005; Pelletier *et al.*, 2009) is a deterministic dynamic simulation model focussing on the fishing fleets and their main commercial species in interaction with management. The model has been applied in different contexts: Bay of Biscay hake and pelagic fisheries (Drouineau *et al.*, 2006; Lehuta *et al.*, 2010; 2013a; 2013b), Baltic Sea cod fishery (Kraus *et al.*, 2008), Eastern English Channel fisheries (Marchal *et al.*, 2011; Gasche *et al.*, 2013; Lehuta *et al.*, 2015), Tasmanian coastal mixed fisheries (Ziegler *et al.*, 2013), deep-sea fisheries off the British Isles (Marchal and Vermard, 2013), New Zealand Hoki fishery (Marchal *et al.*, 2009) and Mediterranean fisheries (Hussein *et al.*, 2011a; 2011b). ISIS-Fish is divided into three submodels: fisheries, management, and biology, the latter simulating stock dynamics but neither direct environment impact nor trophic interactions. The time step is the month and the study area is represented in two dimensions following a regular grid. Fishing activity is implemented by a spatio-temporally explicit effort per fleet, and discards can be dynamically implemented in the model. Fish size is explicitly taken into account and impact the size-structure of the catches via the use of selectivity curves. ISIS-Fish allow to consider precise details on the economy, management and catchability. Thus, joined with a fleet-dynamics model, ISIS-Fish could provide good views of the short-term economic impact of a LO, and can be used to tackle new management measures assessment. However, its capacity to simulate the effects of fisheries management on ecosystem dynamics remains limited by the absence of trophic interactions in the model.

Atlantis (Fulton *et al.*, 2007) is an end-to-end model of marine ecosystems, which has originally been designed for management strategy evaluation (Fulton *et al.*, 2005). Many ecosystems around the world were modelled using Atlantis framework (see http://Atlantis.cmar.csiro.au/ for an updated list). Environment (physical and biogeochemical), functional groups with trophic interactions and fishers are explicitly represented in varied details. Spatialization is implemented in three dimensions using an irregular grid of a limited number of boxes. The time step is flexible, e.g., Girardin (2015) used a monthly time step for fleets, an hourly time step for nutrient, detritus and plankton, and a daily time step for all remaining functional groups. Fishing activities are implemented by a spatio-temporal dynamic effort per fleet, and LO be either fixed or dynamically implemented. An explicit average size is given for each age class, onto which gear selectivity applies. When coupled with a fleet dynamics model, Atlantis is comprehensive enough to evaluate the short- and medium-term impacts of the LO on ecosystem and fisheries dynamics. One of its main limits consists in its complexity, and the time requirement to ste up such a model and the difficulty to evaluate the sensitivity of its main outputs to inputs uncertainty (Girardin, 2015).

StrathE2E (Heath, 2012; 2014b) is an end-to-end ecosystem model which explicitely accounts for the nutrient fluxes between detritus, dissolved nutrients, phytoplankton, benthos, zooplankton and groups of higher trophic level. Up to now, this model has only been applied to the North Sea ecosystem. The time step is the month, while spatialization is defined by large aggregated areas, allowing to fit the model to empirical data. The model's primary objective is to explore trophic controls. In StrathE2E the different functional groups have no explicit size, therefore preventing direct use of selectivity curve. Fishers' activities are represented by an annual amount of yield. For their study, Heath *et al.* (2014a) used fixed or biomass-related percentage for pelagic and demersal species, respectively. StrathE2E can give valuable insights on the short-term direct effect of discarding on the ecosystem and biogeochemical cycles. However like Ecospace, the resolution of StrathE2E is too aggregated (in terms of spatio-temporal scaling and fish size composition) and does not provide a frame to build in fleets dynamics easily in the LO context.

OSMOSE (Object-oriented Simulator of Marine ecosystem Exploitation; Shin and Cury, 2001) is a multispecies, size- and spatially-explicit IBM. All trophic interactions are solely based on the spatial-occurrence and size availability of the prey to the predator, an important specificity leading to dynamic trophic interactions independent of any diet matrix. OSMOSE focus on the high trophic levels, fish and cephalopods, while lower trophic levels are considered as forcing biomass prey fields. High trophic levels are grouped into super-individuals, which regroup individuals of the same species with the same characteristics: age, size, weight and geographical location. Every super-individual is subject to life's processes, from eggs to adults: birth, predation, growth, migration, reproduction and death. The model was first used to explore the impacts of fishing on a theoretical fish community, and on its representative size spectra (Shin and Cury, 2004). The model was then applied to a real case study, the Benguela upwelling ecosystem (Shin *et al.*, 2004b; Travers *et al.*, 2006), then coupled to a biogeochemical model (Travers *et al*., 2009). Since then, different applications were developed for different kinds of ecosystems: upwellings (e.g. Marzloff *et al.*, 2009, Travers-Trolet *et al.*, 2014), estuaries (e.g. Brochier *et al.*, 2013), semi-enclosed seas (Fu *et al.*, 2013), shelf seas (e.g. Gruss *et al.*, 2015, Halouani *et al.*, 2016), to evaluate the ecosystem effects of fishing, management, and climate change, sometimes in combination. The time step used for simulations was initially the semester (Shin *et al.*, 2004b), but is now flexible and classically varies between a week and a month. Space is represented in a 2D grid, but a third dimension (depth) is implicitly considered through accessibility coefficients between species of different habitats. Fishing is represented by an amount of catches or a fishing mortality per species, which can be both temporally and spatially discretised, and applied differently according to size and/or age categories. However, OSMOSE in its present form does not build in fleets dynamics (effort allocation and discarding) yet.

Every models, including those presented above, have their pros and cons depending on the scientific questions adressed (Figure 1.4), and multi-model approaches have sometimes been encouraged (Prellezo *et al*., 2012; Steele *et al*., 2013; Thébaud *et al*., 2014; Hyder *et al*., 2015; Nielsen *et al*., 2017; Spence *et al*., 2017). In this thesis, we will choose to use OSMOSE to evaluate the effects of the LO. Due to its overall flexibility, its dynamic trophic interactions and the fine scale at which the model operates (in terms of time, space, fish individuals and demographic structures), OSMOSE is an adequate model to simulate community dynamics for high trophic levels at fine scale. If coupled with fleet dynamics, OSMOSE could also evaluate the medium-term effects of the LO on ecosystem-fisheries dynamics interactions. However OSMOSE could not be used to evaluate the direct impact of LO on scavengers and biogeochemical cycles, because the recycling loop is not modelled in OSMOSE. As we focus on higher trophic levels, neglecting such impacts are reasonable. To explore combined ecosystem-fisheries dynamics, it is mandatory to coupling OSMOSE with a fleet-dynamics model, which constitutes one of the challenges of this thesis. Beneficiating from the individual-based structure of the resulting coupled model, and its consideration of individual variability and flexibility would give innovative insights into the medium-term effects of the implementation of the LO.

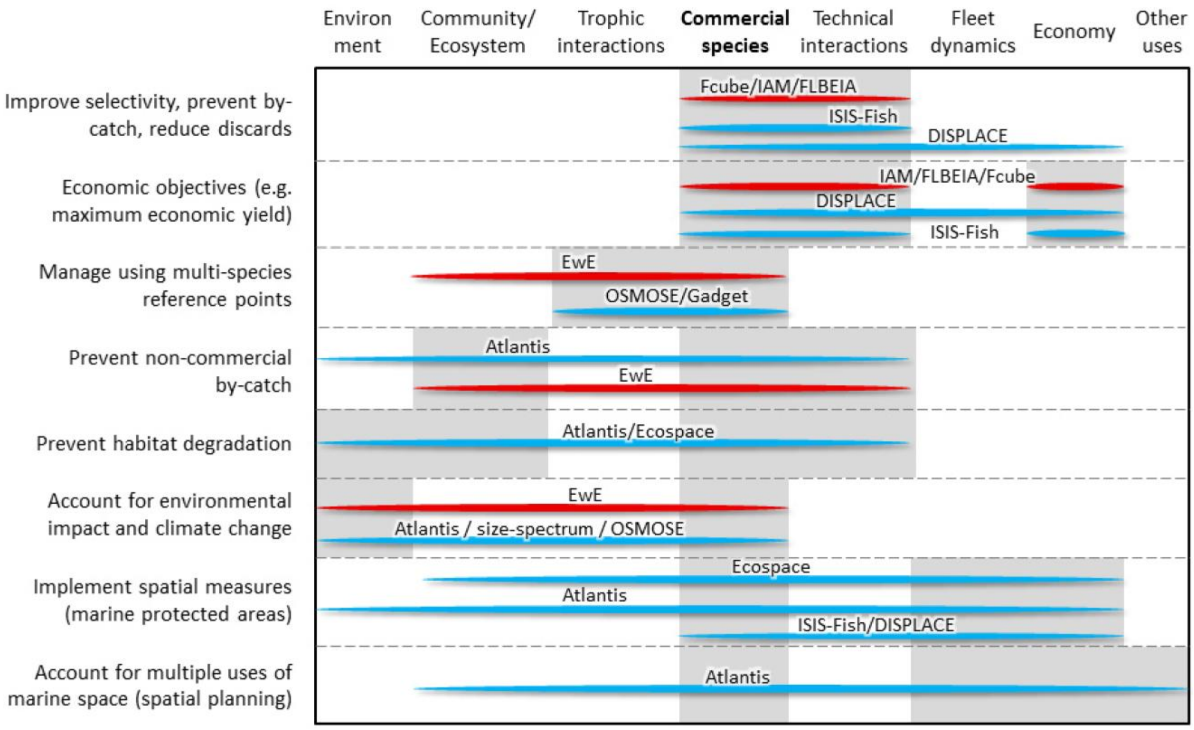

Figure 1.4. Assessment of model complexity and skills against a range of management objectives. Grey boxes represent the minimum components and/or interactions required for a model to be competent to inform a given management objective. Bars represent example models that meet this requirement. This non-exhaustive list includes non-spatial models in red and spatially-explicit ones in blue. As some of these models are modular (i.e. can include or not a particular process or component depending on the application study), only their minimum level of complexity is represented here (source Lehuta *et al.*, 2016).

1.3.3 Spatio-temporal inputs in ecosystem models

Given fishers' plasticity and their ability to switch targets or fishing grounds (Sánchez *et al.*, 2004; Ouréns *et al.*, 2015), particularly when faced with regulation (Christensen and Raakjær, 2006), an efficient model should be able to reduce as much as possible the uncertainty concerning fishers' perception of fish distribution (Johannes *et al.*, 2000; Abernethy *et al.*, 2007; Poos and Rijnsdorp, 2007).

In marine ecosystem models, where mechanisms of fish movement are rarely detailed, species distributions are generally informed by scientific sea surveys, occurring most of the time only once a year, what Mackinson and Nøttestad (1998) named the '*tiny snapshots*'.

Therefore, scientific sea surveys alone generally fail to portray seasonal fluctuations of fish spatial distributions (Petitgas *et al*., 2003; Bordalo-Machado, 2006; Abernethy *et al*., 2007; Lehuta *et al*., 2016). To get around this limitation, different solutions can be considered. First, fishery-dependent CPUEs could be used to reflect abundance trends and species' spatial distributions. However, CPUEs are derived from non-random fishing activities and their adequacy in reflecting species abundance and distributions is adversely affected by technological creeping and tactical adaptations (Maunder and Punt, 2004). Second, fishers' knowledge could be directly used to derive maps of species distribution per season (e.g. Bergmann *et al.*, 2004; Silvano and Valbo-Jørgensen, 2008; Gerhardinger *et al.*, 2009). This approach, however, comes at a monetary cost. Third, some previous studies simulating the seasonal movements of fish (e.g. Bertignac *et al*., 1998; Lehodey *et al*., 2008; Nielsen *et al.*, 2014), could be used as alternative knowledge. However, they focused on specific commercial species subject to large migrations (e.g., tuna) with little scope to inform other particular case studies and associated fisheries. In order to achieve our final modelling objectives, this thesis first needs to improve the knowledge and methodology related to fine-scale fish distributions, and how the combination of several marine resources spatial distributions drives fishing effort allocation.

1.4 Eastern English Channel

1.4.1 Abiotic characteristics and fish community description

The Eastern English Channel (EEC; ICES Division 27.7d) is an epicontinental sea, delineated by latitudes 49.3°N and 51°N and longitudes 2°W and 2°E (Figure 1.5). It constitutes a corridor between the NEA Ocean and the North Sea. This 35 000 km² sea is relatively shallow, with a maximum bathymetry of 70m in the western part of the EEC (Figure 1.6). Sediments are distributed according to the bed shear stress, with pebbles and gravels mainly occurring in the intensely stressed central EEC, while sand and mud accumulate where the stress is weaker on the coasts and estuaries (Carpentier *et al*., 2009; Figure 1.7).

The EEC can be considered as an ecosystem unit, as it displays strong abiotic and biotic differences with the Western English Channel (Dauvin, 2012). Furthermore, the habitat diversity and the overall species richness is greater in the EEC than in the neighbour North Sea (Carpentier *et al*., 2009). EEC fish and invertebrates communities are highly structured and persistent according to autumn survey data analysed by Vaz *et al*. (2007), who found that almost 50% of the community structure variance could be explained by environmental and persistent factors (depth, sediment, bed shear stress). However, EEC is also inhabited by several migrating species: red mullet (*Mullus surmuletus*) (Mahé *et al*., 2005), cuttlefish (*Sepia officinalis*) (Royer *et al*., 2006), mackerel (*Scomber scombrus*) (Eltink *et al*., 1986), herring (*Clupea harengus*) (ICES, 2015), or European seabass (*Dicentrarchus labrax*) (Pawson *et al*., 2007). In contrast, some stocks are considered to reside in the EEC during almost all their life cycles: e.g., sole (*Solea solea*), plaice (*Pleuronectes platessa*) (ICES, 2017). A 25-years time series analysis revealed the occurrence in 1998 of a shift in the fish community structure towards more evenness, which was put in relation with a combination of climate and fishing conditions (Auber *et al*., 2015).

Figure 1.5. 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 (source, Carpentier *et al*., 2009).

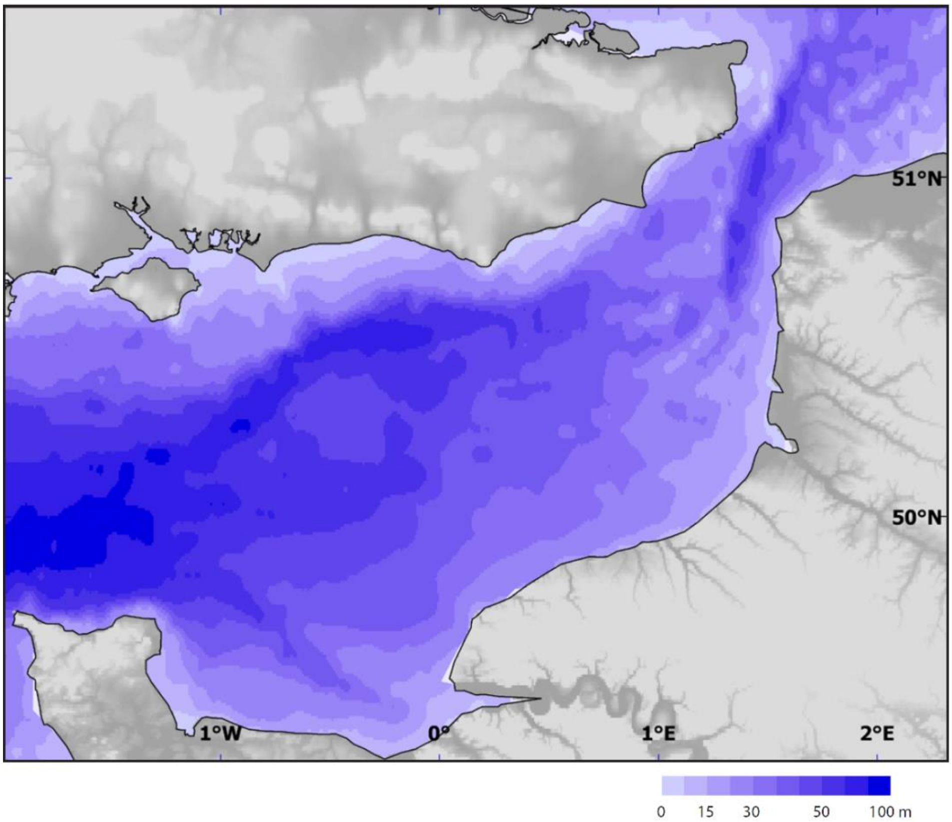

Figure 1.6. Bathymetry in the Eastern English Channel (modified from Carpentier *et al.*, 2009).

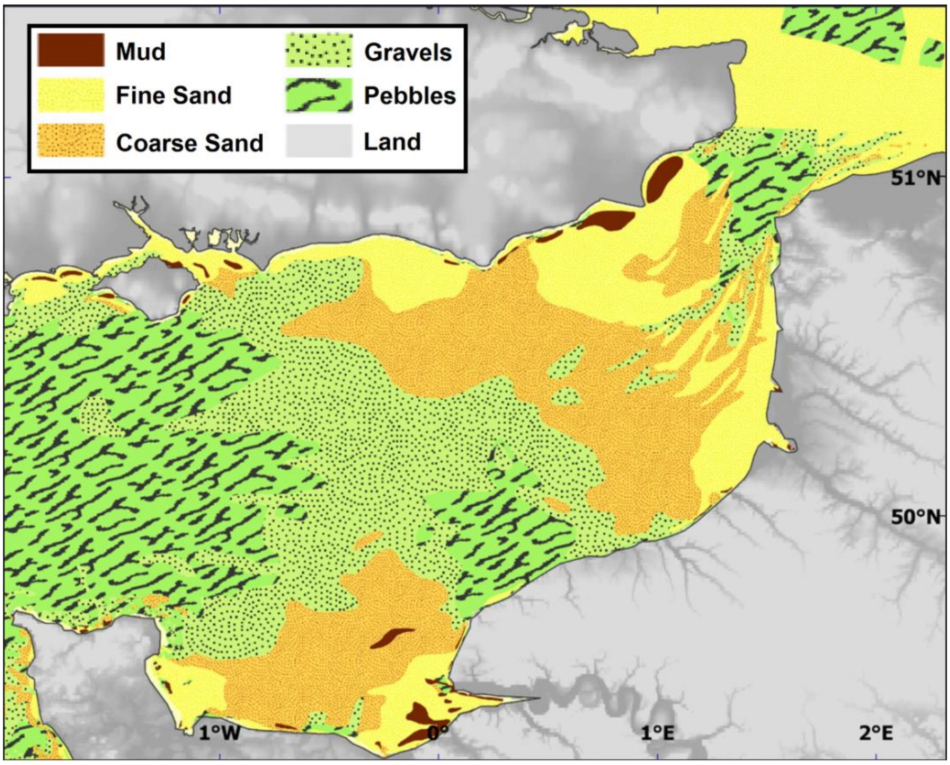

Figure 1.7. Sediment types in the Eastern English Channel, as derived from Larsonneur *et al*. (1982) (modified from Carpentier *et al*., 2009).

1.4.2 Fishing activity and fleets operating in the EEC

EEC fish and shellfish communities have been exploited for centuries (Pfister-Langanay, 1998), and it is noticeable that a high proportion of catches is dedicated to invertebrates since 1970 (Molfese *et al*., 2014). Currently, different countries harvest in the EEC, but most of the activity is performed by France, United Kingdom, Belgium and the Netherlands. Landings are varied, but about ten species contribute to the bulk of the landings value: sole, cod, whiting, European seabass, Scallops (*Pecten Maximus*), plaice, red mullet, herring, mackerel (*Scomber scombrus*), cuttlefish and squids (*Loligo vulgaris* and *Loligo forbesii*). These species are caught by a diversity of fleets involving netters, inshore and mixed trawlers (Carpentier *et al*., 2009). This diversity also implies a diversity of métiers with a part of non-exclusive or polyvalent (i.e. changing the gear according to the season) and exclusive trawlers (Guitton, 2003). For the purpose of this thesis, a focus will be made on French exclusive bottom trawlers, one of the main fleets in terms of landings (volume and value) in the EEC (Table 1.2), but also one of the less selective and thus more affected by the LO (Table 1.3). Large trawlers (~50m) perform their activity outside of English Channel while smaller ones mostly concentrate in the EEC and the southern North Sea. Demersal trawlers are concentrated in two main ports, Boulogne-sur-Mer and Port-en-Bessin, while the rest of the fleet is mainly located in Cherbourg, Fécamp, Dieppe and Le Tréport (Table 1.4; see Figure 1.8 for ports locations). Trawls are usually rigged with a 80mm mesh size.

Table 1.2. Percentage of landings and revenue relative to the total of French fishing activities performed by exclusive bottom otter trawlers in the Eastern English Channel.

	Landings (%)	Revenue (%)
2008	24.4	23.3
2009	30.5	22.1
2010	31.3	24.9
2011	26.6	22.7
2012	20.4	19.3
2013	18.0	16.1
2014	19.8	17.3

Table 1.3. Piélou's evenness index (1975) of landings for French exclusive bottom otter trawlers (Exclusive OTB) and main French gears in the Eastern English Channel during the 2008-2014 period. The index varies between 0 (i.e., only 1 species caught) and 1 (i.e. homogeneous catch of all species). GTR: Trammel net. DRB: Dredge. OTM: Midwater otter trawl. PTM: Pelagic pair trawls. FPO: Pots and traps.

	Exclusive OTB	GTR	DRB	OTM	PTM	FPO
2008	0.53	0.40	0.15	0.23	0.34	0.25
2009	0.53	0.40	0.15	0.28	0.35	0.29
2010	0.54	0.45	0.11	0.29	0.34	0.30
2011	0.54	0.45	0.15	0.24	0.30	0.26
2012	0.54	0.43	0.15	0.16	0.28	0.24
2013	0.55	0.40	0.14	0.23	0.30	0.20
2014	0.55	0.39	0.17	0.22	0.31	0.16
Mean	0.54	0.42	0.15	0.24	0.32	0.24

Table 1.4. Relative contribution of home ports to annual effort of French exclusive bottom otter trawlers in Eastern English Channel during the 2008-2015 period.

Port	Average annual effort (%)
Le Tréport	5.6
Port-en-Bessin	21.6
Boulogne-sur-Mer	51.5
Cherbourg	4.8
Dieppe	11.3
Fécamp	4.4
Others	0.8

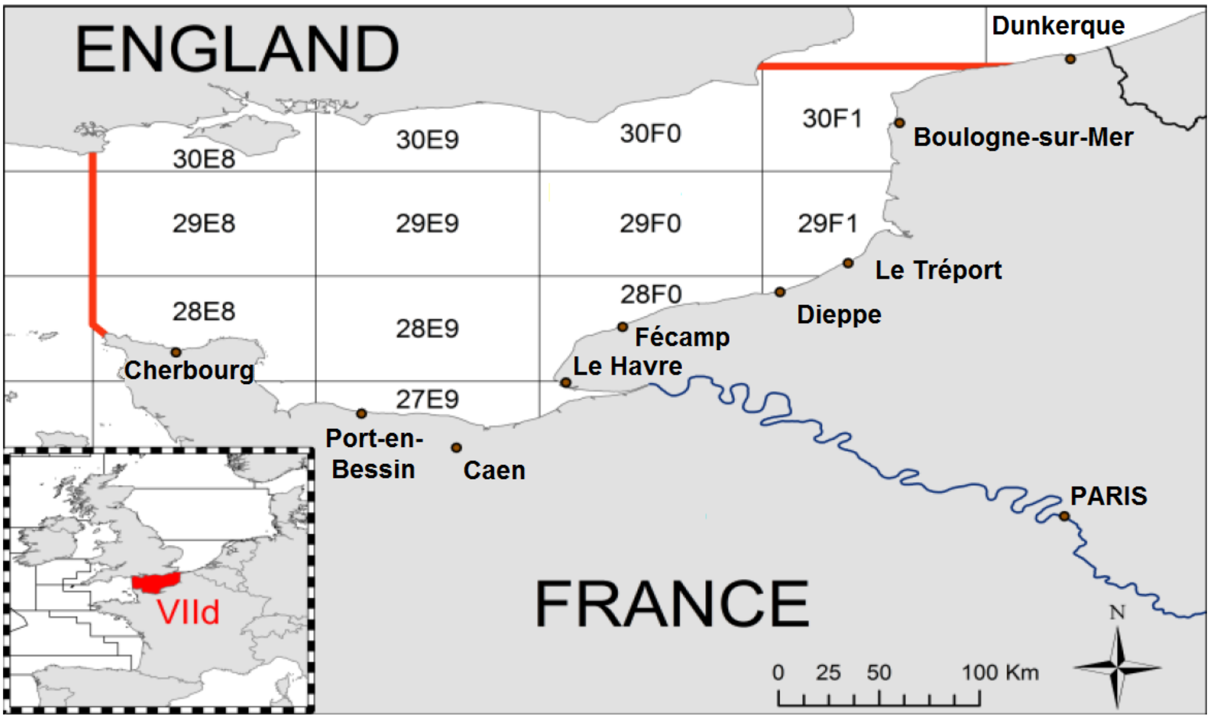

Figure 1.8. Statistical rectangles and main fishing harbours in the Eastern English Channel (modified from Girardin *et al*., 2015).

1.4.3 Management

In the EEC, several species are managed through TACs, set a the EU level and then divided among countries operating in the EEC following historical rules of repartition. French exclusive bottom trawlers rely particularly on TACs for 6 stocks, 2 of them (plaice and sole) being entirely contained in EEC (ICES 27.7d), while the 4 other stocks (cod, herring, mackerel and whiting) are managed on a wider area. Minimum landing sizes exist for a set of species in EEC (Table 9), while mesh sizes and licenses are associated with target species. Spatial restrictions are diverse along French and UK coasts. Complex restrictions apply to specific fishing activities within the 3, 6 and 12-miles areas off these coasts (Tidd *et al.*, 2015; Girardin *et al.*, 2015). French fishers' activity is more restricted around UK coasts, and vice versa. Finally, unilateral solutions such as moratorium (e.g., European bass in France in 2017) or fishing area closures (e.g., scallops in the Baie de Seine) can be pronounced by a country for its fisheries.

Table 1.5. Minimum Landing Sizes (MLS) in Eastern English Channel (ICES 27.7d).

species	MLS (cm)	Common name
Clupea harengus	20	Herring
Dicentrarchus labrax	42	European seabass
Gadus morhua	35	Atlantic cod
Merlangius merlangus	27	Whiting
Pleuronectes platessa	27	European plaice
Sardina pilchardus	11	Sardine
Scomber scombrus	20	Atlantic mackerel
Solea solea	24	Sole
Trachurus trachurus	15	Horse mackerel

1.4.4 An ecosystem supporting many human activities

The EEC hosts intense and diverse human activities other than fisheries. Maritime traffic (tourism and cargo) is indeed one of the most intense in the world, with approximately 500 vessels traveling in the EEC every day (Dauvin, 2012). Additionally, several areas of aggregate extractions and windfarms are already operating and/or planned in the EEC (Figure 1.9). These activities interact with fishing activities. The incorporation of such activities in fisheries analysis can be necessary, according to the scale observed, as multiple studies emphasized more or less pronounced interactions with some métiers (Marchal *et al.*, 2014; Tidd *et al.*, 2015; Girardin *et al.*, 2015).

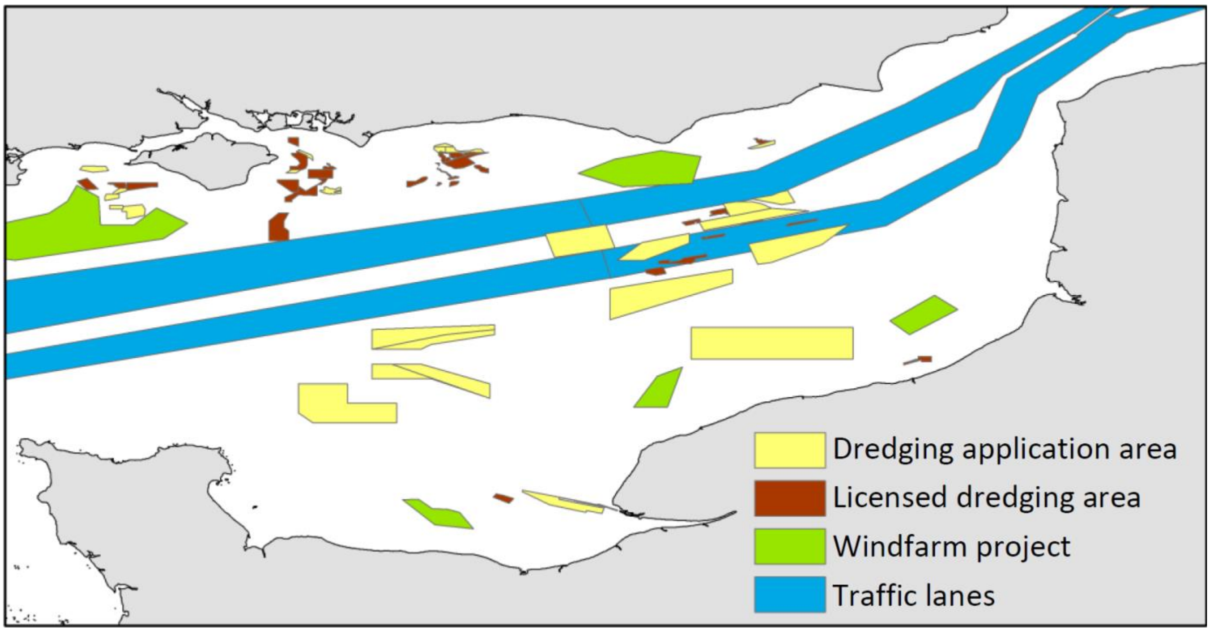

Figure 1.9. Distribution of current and possible future areas impacted by human activities other than fishing in the EEC (source, Girardin, 2015).

1.4.5 Landing obligation in the Eastern English Channel

In ICES area 27.7d, only two species are in December 2017 officially concerned by the LO: sole and whiting. All soles should be landed when caught by beam trawls and nets, while for otter trawlers, sole has to be landed only if the percentage of sole in the total catches of a boat was higher than 5% in 2014 and 2015 (European Commission, 2016). Whiting has to be landed by trawlers and seiners if the percentage of cod, haddock, whiting and saithe combined in the total catches was higher than 20% in 2014 and 2015. Theoretically it should be applied to every species regulated by quotas, except those which already had an exemption. In the EEC, only common sole smaller than its minimum landing size is currently concerned by survivability exemptions, and only when caught by otter trawl gears within the six nautical miles of the coast and outside nursery areas, with a vessel of length and engine power below 10 meters and 180 kW respectively, at a shallower depth than 15 meters and with a tow duration lower than 1:30 hours (European Commission, 2016). The Dutch demersal fishing sector is currently claiming for plaice discarding exemption on the bases of high survivability (de Vos *et al.*, 2016), which recent studies tend to confirm (Methling *et al.*, 2017; Morfin *et al.*, 2017).

1.4.6 Complex models in Eastern English Channel

Here a focus is made on some models in EEC that are or could be used to explore the LO question (Figure S1.2).

Ecospace (Metcalfe *et al.*, 2015) is mostly based on the Ecopath model developed by Carpentier *et al*. (2009) and the Ecosim and Ecospace models of Daskalov *et al*. (2011), and was used for the evaluation of marine protected areas impact on marine ecosystem and fisheries. This original model was composed of 51 functional groups, from phytoplankton and detritus to top predators, including 2 marine mammals, one seabird, 29 fish and 15 invertebrates. The fisheries are decomposed into 8 fleets: beam trawl, demersal otter trawl, dredges, pelagic trawl, hooks and lines, nets, seine, traps and pots. Fishing mortality distribution is based on a gravity model, assuming different ports and related costs according to the distance of fishing grounds from them. Further details are not fully known, but Ecospace configuration *a priori* limits the spatio-temporal discretization of the model. Furthermore, discards should also be represented as a fixed proportion included in the catches of modelled fleets.

ISIS-Fish (Lehuta *et al*., 2015) now includes 6 of the main EEC commercial species: sole, plaice, red mullet, scallops, squids and cuttlefish. The monthly spatial distributions are obtained from three surveys: BTS in July (sole, squids and cuttlefish), CGFS in October (plaice, red mullet, squids and cuttlefish) and COMOR in July (scallops). For species populations, the different polygons represent the main habitats identified by Girardin *et al*. (2016), resulting in 30 polygons. Different fleets are modelled in it: bottom trawlers, beam trawlers, dredgers, gillnetters and trammel netters, each decomposed in 2 or 3 length classes and 2 ports (North and Normandy). Their monthly effort distribution is simulated using a gravity model and dispatched into ICES polygons. Catchability is a complex value influenced by fish accessibility per age, selectivity, gear efficiency, ability to target a particular species, and technical efficiency (mainly vessel length). Finally discards occur in the model when a quota is reached or when a fish' length is under the minimum landing size.

Atlantis (Girardin *et al*., 2016) represents an average state of the EEC ecosystem during the 2002-2011 period. It is composed by 40 functional groups, including 21 vertebrates and 16 invertebrates. Vertebrates are separated into 10 age classes while only cephalopods are separated into 2 size classes within the invertebrates. Species spatial distributions are derived from habitat models built by Carpentier *et al*. (2009) and other existing literature, while invertebrates were spread homogeneously in the EEC (except scallops distributions, which were derived from the COMOR survey). The model's spatial structuration is based on bathymetry, sediment, demerso-benthic community and flatfish nursery grounds. For forecasting, the model was coupled with dynamic maps of fishing effort derived from external RUM or gravity models (Girardin, 2015). Fishing effort is output for 21 fleets at the ICES polygon and monthly scales. Selectivity is based on normal or logistic curves with two parameters estimated for each of the fleets, while catchability is determined per fleet and functional group. In this model discards are implemented as a fixed proportion of catches per fleet (estimated using OBSMER data).

OSMOSE (Travers-Trolet *et al.*, in prep.). The existing model represents an average state of the EEC community during the 2000-2009 period. It is composed of 13 fish species plus one group of cephalopods composed by 2 species (*Loligo vulgaris* and *Loligo fobersii*). A forcing of some lower trophic levels is provided by an ECO-MARS-3D model of the region, while six other ones are homogeneously distributed in EEC and do not build in population dynamics. The time step of the model is of two weeks and the spatial grid is composed by squares of 0.1° x 0.1°. The spatial distributions of high trophic levels are based on literature (Carpentier *et al.*, 2009). Fishing mortality applies, annually for most (12) species or seasonally for squids and horse mackerel, to all fully recruited individuals. Fishing mortality was obtained by calibrating biomasses and landings, which were assumed to equate catches. The ecosystem functioning component of the eco-eco model developed in Chapter 4 will be derived from an expansion of this OSMOSE model.

1.4.7 Data sources in Eastern English Channel

Information related to fishers' activity comes from a variety of national sources. Landings can be computed from combined logbooks and sales slips records (SACROIS). These data are available by vessel, fishing trip, gear used and ICES rectangle. Discards may be estimated using samples collected on-board French commercial vessels through the OBSMER programme (Cornou *et al.*, 2015). The OBSMER programme offers a picture of the total catch (discards included) for a sample of fishers, with precise spatio-temporal information, and at the scale of the fishing operation. Fishing effort distributions are provided by the French Directorate for Sea Fisheries and Aquaculture (DPMA) and derived from satellite-based Vessel Monitoring System (VMS). Information concerning fishers' *a priori* intentions (in terms of species targeting and fishing grounds) have been extracted from activity calendars directly collected through interviews on a regular basis by Ifremer.

Fish community composition and species distributions are available from different surveys: The Channel Ground Fish Survey (CGFS; Coppin and Travers-Trolet, 1989) has sampled the demersal community each year in October since 1988. Since 2007 the French International Bottom Trawl Survey (IBTS; Verin and Lazard, 2015) is sampling the demersal community is the Eastern part of EEC, along with the Southern North Sea. Finally the UK Beam Trawl Survey (BTS) is performed each year in July to sample flatfish, especially sole and plaice. Other surveys are operated in the EEC, but these have a limited spatial extent, focus on specific species assemblages and/or size groups, and hence do not present spatio-temporal scales suitable for this thesis.

1.5 Objectives of the thesis

As exposed in this first introductive chapter, to develop knowledge on the reaction of the ecosystem-fishers couple to the implementation of the LO in the EEC, several objectives have to be achieved.

Prior to the modelling part, it will be necessary to better understand the spatio-temporal relationships between the main EEC commercial marine species and fishers' behaviour at fine scale. The first milestone of this thesis, addressed in Chapter 2, is then to derive, for the first time, reliable estimates of fish abundance and seasonal distribution maps for a maximum of the marine species involved in the ecosystem, using a realistic approach.

We analysed to that purpose spatially- and temporally- resolved CPUE series, which were derived from an on-board observers programme and compared with seasonal knowledge available on these species. Additionally, the relevance of spatial distributions and annual trend reflected by on-board observations was ascertained through a comparison with CGFS survey indices, through a spatial overlap indicator, the Local Index of Collocation, to determine the fine-scale quality of maps obtained from OBSMER. This second chapter has been published in ICES Journal of Marine Science (Bourdaud *et al.*, 2017).

The second milestone of this work was to improve our perception of fishing effort and pressures exerted by mixed fisheries, in relation to management and other external factors. Detailed maps of fishing effort distribution were here derived from satellite-based information (e.g. Bastardie *et al.*, 2010b; Hintzen *et al.*, 2012; Enever *et al.*, 2017). In Chapter 3, we analysed the spatial overlap between key commercial species and fishing fleets to i) improve the relationship between fishing effort and fishing mortality and ii) determine the fishers' inter-annual specific targeting in relation to ecological, economic and regulatory factors, using an innovative optimization approach. Chapter 3 focuses on the added-value of making monthly spatio-temporal representations of fishing effort and pressure more precise, in the case of EEC exclusive bottom otter trawlers and of their main target. First, an effective fishing effort is computed by including the spatial overlap between fishing vessels and their harvested resource in the catchability parameter (i.e., the key coefficient linking fishing effort and fishing mortality). The goodness of fit was assessed by comparing seasonal trends of nominal and effective fishing efforts to available surrogates of fishing pressure, i.e., landings and fishers' intentions obtained from activity calendars. Secondly, a new approach has been developed to measure the relative weight fishers' give to the different species they target, compared to that expected if they were fully driven by expected revenue, by optimizing a spatially-resolved overlap metric. The results from this analysis are then used to quantify the deviations between actual fishers' targeting and those expected from the IFD, and how these are influenced by external factors, including TAC uptake. This third chapter has been submitted for publication in Canadian Journal of Fisheries and Aquatic Sciences.

Finally, in Chapter 4, the effects of the EU LO on the EEC ecosystem and fisheries were evaluated by expanding and subsequently coupling two models with focus on: i) spatio-temporal distributions and dynamic trophic interaction for a set of EEC species (OSMOSE) and ii) a fishers' adaptation (in terms of effort allocation and discarding) to the new management regime and its knock-on effects on exploitable biodiversity (DSVM). After presenting the conceptual coupling between these models and implementing it in the code, OSMOSE-DSVM was used to evaluate the impact of the LO on EEC exclusive bottom otter trawlers (in terms of effort allocation and discarding) and on the ecosystem they impact. A 'business as usual' scenario is compared with a LO scenario applied to whiting and cod, using different indicators representing the states of the main commercial species, the ecosystem and the economic viability for fishers. Links between Chapter 2, 3 and 4 are resumed in the Figure 1.10, with their main productions and the data sources used.

A conclusive section (Chapter 5) provides a synthesis of the main achievements realised in this thesis. First, the main results obtained from the three research chapters are summarised and debated, in the light of existing evidence, and their limits discussed. Secondly, future research perspectives on ecosystem and fleet-dynamics interactions are discussed.

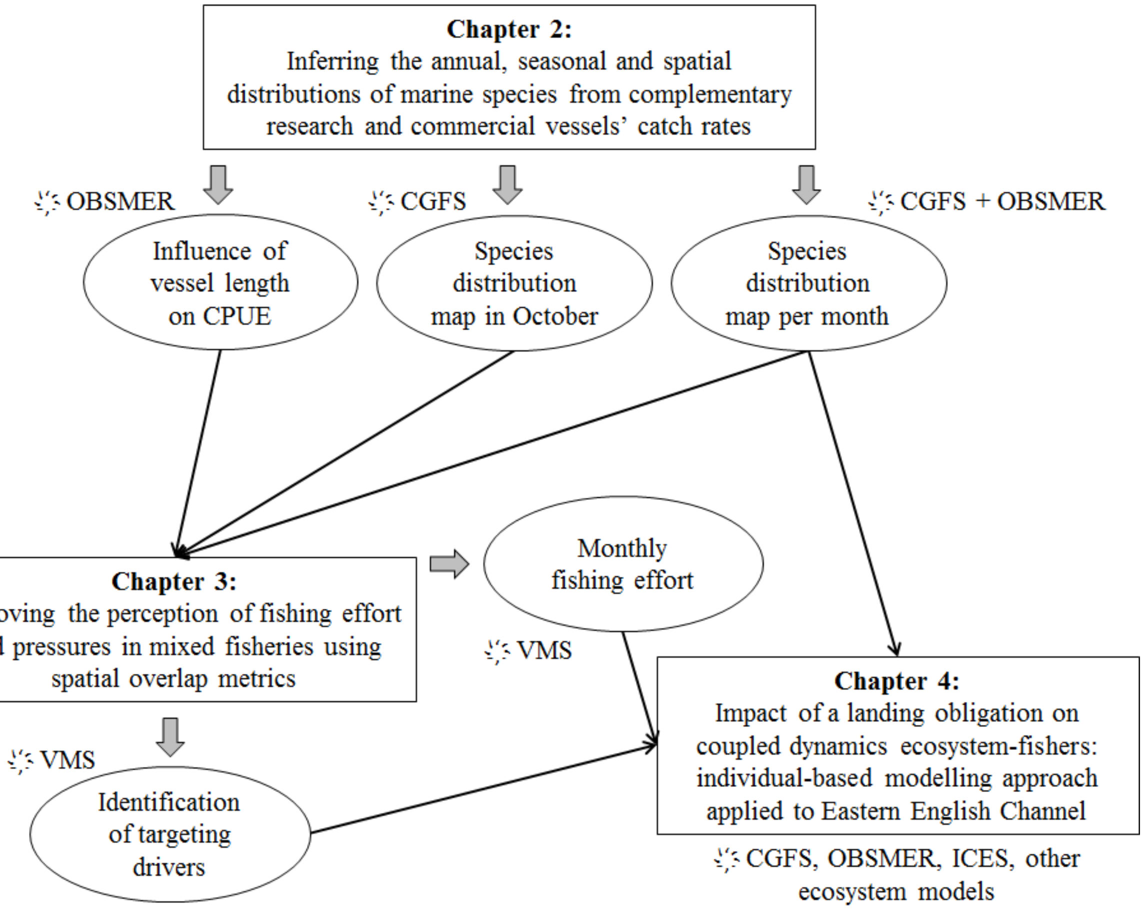

Figure 1.10. Links between the different chapters of the thesis, their main productions and the data sources used.

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Chapter 2

Inferring the annual, seasonal and spatial distributions of marine species from complementary research and commercial vessels' catch rates

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Abstract

The objective of this study is to analyse at fine scale the annual, seasonal and spatial distributions of several species in the Eastern English Channel (EEC). On the one hand, data obtained from scientific surveys are not available all year through, but are considered to provide consistent yearly- and spatially-resolved abundance indices. On the other hand, on-board commercial data do cover the whole year, but generally provide a biased perception of stock abundance. The combination of scientific and commercial catches per unit of effort (CPUEs), standardized using a delta-GLM, allowed to infer spatial and monthly dynamics of fish distributions in the EEC, which could be compared with previous knowledge on their life cycles. Considering the scientific survey as a repository, the degree of reliability of commercial CPUEs was assessed with survey-based distribution using the Local Index of Collocation. Large scale information was in agreement with literature, especially for cuttlefish. Fine scale consistency between survey and commercial data was significant for half of the 19 tested species (e.g. whiting, cod). For the other species (e.g. plaice, thornback ray), the results were inconclusive, mainly owing to poor commercial data coverage and/or to particular aspects of the species biology.

Keywords: spatial distribution, seasonality, commercial data, survey data, Eastern English Channel.

2.1 Introduction

Ecosystem-Based Fisheries Management (EBFM) requires enhancing knowledge of ecosystem functioning, therefore allowing forecasting the impact of fisheries on salient ecosystem components (Long *et al.*, 2015) and to design future management plans and tools including Marine Protected Areas (Meyer *et al.*, 2007) or fishing closures (Hunter *et al.*, 2006). This necessitates a stepwise approach, the first tier of which, and one of the most important, is to gain fine scale knowledge on the seasonal and geographic distribution of marine organisms, in general, and fish stocks in particular (Booth, 2000).

Scientific surveys have been implemented for decades to derive spatially- and yearly-resolved abundance indices of commercial fish and shellfish species (e.g. van Keeken *et al.*, 2007). Surveys provide abundance indices, derived from standardized and controlled protocols, which allow for a wide spatial coverage associated with a weak selectivity (Verdoit *et al.*, 2003). Survey data, however, are costly to obtain and therefore rarely provide for adequate seasonal coverage of the resource distribution. In contrast, information derived from commercial fisheries are generally available all year through. Consequently, the catch per unit of effort (CPUE), the most common and easily collected fishery-dependent index of abundance (Maunder and Punt, 2004), has the potential to reflect fish distributions. However, commercial CPUEs can generally not be used directly as abundance indicators. This is because fishers target rather than sample fish densities, and continuously adapt their activities to prevailing conditions, through technological development and tactical adaptations (Marchal *et al.*, 2006), including discarding practices on which information is often limited (Rijnsdorp *et al.*, 2007).

A major challenge for fisheries scientists is then to reconcile fisheries-independent and -dependent information into abundance indices that consistently mirror the annual, seasonal and spatial dynamics of commercial marine species. Kristensen *et al.* (2014) have reconstructed spatial and seasonal cohorts of cod (*Gadus morhua*) in Skagerrak by kriging, in both time and space, data provided by survey and also by fisheries subject to a survey-like sampling protocol. To our best knowledge, however, no method has yet been developed to estimate spatio-temporal distributions of fish at high resolution, by combining survey and true commercial fisheries data.

The main objective of this paper is to provide detailed annual, seasonal and spatial distributions of major Eastern English Channel (EEC) commercial fisheries resources, using a novel approach combining fisheries-independent and -dependent information. The gain in knowledge on fine scale temporal and spatial fish distribution in the EEC will expand the scope of earlier results (e.g. Vaz *et al*., 2007), and strengthen the science support to an EBFM in this area. To that purpose, we (i) inferred the seasonal and spatial abundance distribution based on survey and commercial abundance data for several species in the EEC, (ii) investigated the degree of similarity of fine scale spatial distributions derived from these two data sources and (iii) investigated abundance indices derived from these data sources.

2.2 Material and methods

2.2.1 Study area

The Eastern English Channel (ICES subdivision VIId) is delimited by latitudes 49.3°N and 51°N and longitudes 2°W and 2°E (Figure 2.1). This shallow area constitutes a corridor between the northeast Atlantic Ocean and the North Sea, and a strategic region in the northeast Atlantic, as it hosts a very intense maritime traffic and human activities such as mixed fisheries, aggregate extraction and wind farms (Dauvin, 2012). This area is also important for several commercially important migratory species, e.g. red mullet (*Mullus surmuletus*) (Mahé *et al*., 2005), cuttlefish (*Sepia officinalis*) (Royer *et al*., 2006), mackerel (*Scomber scombrus*) (Eltink *et al*., 1986), herring (*Clupea harengus*) (ICES, 2015), or European seabass (*Dicentrarchus labrax*) (Pawson *et al*., 2007).

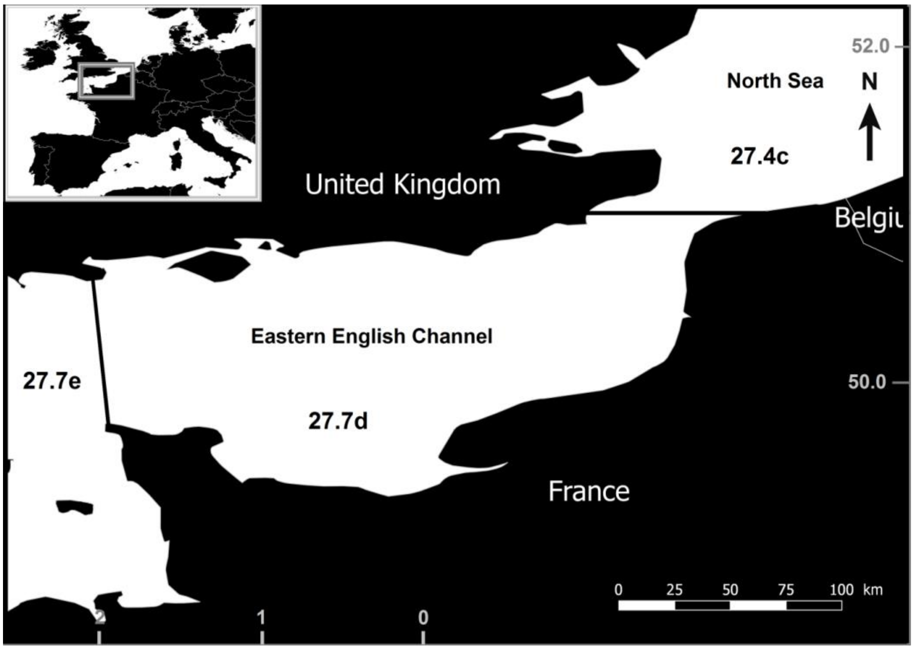

Figure 2.1. Study area of the Eastern English Channel, corresponding to the ICES division VIId.

Fishing is a key socio-economic activity in the region (Carpentier *et al*., 2009), which has also generated a strong pressure on its marine ecosystem (Molfese, 2014).

2.2.2 Data

This study is supported by two main data sources: a scientific survey (the Channel Ground Fish Survey – CGFS; Coppin and Travers-Trolet, 1989) and observations on-board commercial vessels (hereby referred to as the OBSMER French programme; Cornou *et al*., 2015).

The CGFS has sampled the entire EEC demersal community annually since 1988. The survey occurs every year in October, with a systematic fixed sampling design of 88 trawling stations located between 49.3°N and 51.3°N. The sampling gear is a GOV trawl with 3 m vertical opening, 10 m horizontal opening and a 20 mm codend. For each haul, all fish caught are sorted, identified and measured to the nearest inferior centimetre. In case of large catch, random subsampling is performed while ensuring representativeness of species and length distributions. For the current study only survey data from 1998 to 2014 were retained as this period corresponds to a relatively stable state of the community structure with no detected regime shift in species spatial distributions (Auber *et al*., 2015).

The CGFS provides information for a large panel of economically valuable demersal fishes and cephalopods, i.e. European seabass, red mullet, cod, whiting (*Merlangius merlangus*), plaice (*Pleuronectes platessa*), cuttlefish, squids (*Loligo* spp.) and thornback ray (*Raja clavata*). Other commercially important species such as common sole (*Solea solea*), herring or sardine (*Sardina pilchardus*), are poorly sampled by the GOV trawl (Carpentier *et al*., 2009), and thus have not been considered in this study.

On-board observer programmes allow estimating catch and effort for a sample of fishing operations. Unlike other fisheries data collection programmes, e.g. building on port sampling and/or mandatory logbooks, observer’s data are precisely geo-referenced and allow inferring the total catch, including the discarded fraction, and more accurate measurements of effective fishing effort. Although on-board fisheries data can generally not be collected for all the vessels belonging to a given fleet, and although the presence of observers may be perceived as overly intrusive to fishers, they offer an opportunity to derive CPUE-based abundance indicators, at a fine spatial and temporal scale.

The OBSMER programme covers the period 2003-2015. It was developed to better estimate the discards' quantity and assess catch composition. Precise information on ship characteristics (e.g. homeport, length, engine power), fishing activity (time, latitude, longitude, gear, fishing effort, targeted species assemblage) and catch composition (landings and discards of fish and commercial invertebrates) are collected for each fishing operation by scientific observers. For each fishing operation, a subsample of the catch (including both the part to be landed and the part to be discarded) is sorted, identified and measured. This data compilation has already been operated to characterize pressures exerted on communities, discarded fractions of catches, or discarding drivers (Fauconnet *et al*., 2015).

Spatio-temporal species distributions estimated using OBSMER data are primarily expected to corroborate previous knowledge on these species' life cycles. In addition, they could reflect species distributions as observed using scientific surveys (considered as a reference) in converging time lapse. However, because species' spatial distributions are dynamic and vary from one time step to another, and because fishers continuously adapt to prevailing conditions (Eigaard *et al*., 2014), time and spatial variations in CPUE reflect two entangled signals prompted by fisher's plasticity and stock fluctuations. Using CPUEs to reflect time changes in stock abundance therefore requires to preliminarily filter out the skipper effect signal it originally contains (Maunder and Punt, 2004).

2.2.3 Standardizing survey and commercial catch rates

Surveys and commercial fisheries operate at different temporal and spatial scales, with different gears and strategies, thereby targeting dissimilar species assemblages and/or size ranges. The first step of this study was to identify common temporal and spatial scales, then to select a common pool of representative species and size ranges, and finally to standardize survey and commercial catchabilities using a delta- Generalized Linear Model (GLM) approach.

The temporal scale retained is the month, while the spatial scale considered is cells of 0.3° x 0.3° (~ 700 km^2). These seasonal and spatial scales result from a trade-off between having a sufficient amount of data and maintaining a sufficient level of precision, as described further.

Based on these small-scale spatio-temporal units, a mean CPUE index in number of individuals caught per hour is calculated separately from OBSMER data for each month and from CGFS data (only for October) for a set of demersal species (Table 2.1). These species have been selected based on their economic importance, relative abundance and/or catchability by the survey gear being considered. Survey data were only kept from 2005 to 2014 for the cephalopods (i.e. *Sepia officinalis* and *Loligo* spp.), as no length information is available for these species before 2005. To harmonize the survey and commercial gears' selectivities of the species being considered, we used a common length threshold (L_s) above which a species is considered to be correctly selected by the different gears (Table 2.1). L_s was graphically determined from length distribution for each species following the method used by Ravard *et al*. (2014): in commercial data most of the length-frequency were unimodal and L_s was approximately set for each species at the length of the highest mode of the different gears combined. In our study, L_s mainly corresponded to the official minimum landing sizes for the few species concerned. The potential case of a different selectivity of large individuals to particular gears (e.g. Bertignac *et al*., 2012) is not considered in this study.

Table 2.1. List of species considered in this study, with their minimum total length L_s (cm), above which individuals are considered to be equally selected by survey and commercial gears, and Minimum Landing Size (MLS) during the 2003-2014 period in Eastern English Channel when relevant.

species	L_s (cm)	MLS (cm)	Common name
Chelidonichthys cuculus	22	-	Red gurnard
Chelidonichthys lucerna	26	-	Tub gurnard
Dicentrarchus labrax	36	36	European seabass
Gadus morhua	35	35	Atlantic cod
Limanda limanda	21	-	Common dab
Loligo spp.	14 [a]	-	Squids
Merlangius merlangus	24	27	Whiting
Microstomus kitt	25	-	Lemon sole
Mullus surmuletus	20	-	Red mullet
Mustelus asterias	60	-	Starry smooth-hound
Platichthys flesus	29	-	European flounder
Pleuronectes platessa	25	27	European plaice
Raja clavata	49	-	Thornback ray
Scyliorhinus canicula	54	-	Lesser-spotted dogfish
Sepia officinalis	13 [a]	-	Common cuttlefish
Spondyliosoma cantharus	17	-	Black seabream
Trisopterus luscus	25	-	Pouting
Trisopterus minutus	13	-	Poor cod
Zeus faber	21	-	John Dory

[a] mantle length

OBSMER data were filtered to avoid abundance overestimation. Thus, for each species and each size, only hauls with all the subsamples representing at least 5% of the total catch weights each were kept for further calculations. Furthermore, to obtain a clear overview of abundance for each demersal species being studied, only fishing gears sufficiently represented (i.e. > 10 observations for a given species) were kept in the analysis.

Finally, we adjusted the remaining catchability differences by standardizing CPUE values derived from both OBSMER and survey data. This was operated by applying a delta-GLM to the CPUEs of each species under consideration. The delta-GLM first fits the probability of observing a zero catch as a function of the explanatory variables, and then fits another GLM to the non-zero catches (Maunder and Punt, 2004; Meissa *et al*., 2008; among others).

The probability of presence is based on the binomial distribution after a binary recoding (0=absence and 1=presence). For hauls with positive CPUE a logarithmic transformation was first applied on data in order to homogenize variances and to transform the multiplicative effects into additive effects (Meissa *et al*., 2008).

The delta-GLM for OBSMER data contains a maximum of six explanatory variables:

$$\mathrm{logit}(\mathrm{p}_{i,a,m,y}^{>0}) = \beta_a \delta_m + \lambda_y + \rho_g \tau + \upsilon_s \qquad (2.1)$$

$$\log(IA_{i,a,m,y}) = \beta_a \delta_m + \lambda_y + \rho_g \tau + \upsilon_s + \varepsilon_{i,a,m,y} \qquad (2.2)$$

where $p_{i,a,m,y}^{>0}$ is the mean presence probability and $\mathrm{IA}_{\mathrm{i,a,m,y}}$ the CPUE of a species caught by vessel i of length τ rigged with gear g (e.g. bottom otter trawl, trammel net), fishing in (0.3° x 0.3°) area a, year y and month m. βa is the area effect of the fishing operation (treated as factor), δ_m is the month effect of the fishing operation, ρ_g is the gear effect, λ_y is the annual effect, υ_s isthe sediment effect, which accounts for small scale habitat variability and is decomposed into five categories s: mud, fine sand, coarse sand, gravel and pebble, based on a sediment map of EEC from Larsonneur *et al*. (1982), and $\varepsilon_{\mathrm{i,a,m,y}}$ a term of residual error.

Sediments are kept because they proved to have the strongest influence on the distribution of species in the shallow Eastern English Channel, compared with, e.g. depth, temperature and salinity (see Carpentier *et al*., 2009). Engine power information was also available but only vessel length was kept as these two variables are usually highly correlated for bottom otter trawlers (r = 0.94 using OBSMER data), the main size-varied vessels of the available commercial data.

CGFS survey data are always collected in October (i.e. no month effect) with the same research vessel (i.e. no vessel or gear effects), hence the previous formula was reduced to the following, with a maximum of three explanatory variables:

$$\text{logit}(p^{>0}_{a,y}) = \beta_a + \lambda_y + \upsilon_s \tag{2.3}$$

$$\log(IA_{i,a,m,y}) = \beta_a + \lambda_y + \upsilon_s + \varepsilon_{a,y} \tag{2.4}$$

Models' retained explanatory variables were selected for each species based on Akaike information criterion (AIC). Model selection was largely influenced by the previous choice of the spatial resolution for *area* variable.

In none of the models (1-4) an interaction term between area (or area-by-month) and year effects was considered. This requires some clarifications, given such an interaction term could potentially reveal spatial shifts in fish distribution over time.

In the analysis of commercial CPUE indices, spatio-temporal interactions were partly covered by introducing an area-by-month term. It was, however, not possible to explore the effect of introducing the higher-ranked interaction area-by-month-by-year, partly owing to the limited amount of observations available but also to opportunistic fisher's behaviour, which in combination resulted in a variable inter-annual coverage of the OBSMER dataset. In the analysis of survey abundance indices, only area-by-year effects could potentially be considered, since the CGFS is operated in October only. Auber *et al*. (2015) concluded that although October EEC fish communities were subject to a substantial spatial shift in 1997, no significant change was observed during 1998-2014, i.e. the period being considered in this analysis. Still, we did investigate a model including a spatio-annual effect. According to the AIC none of the presence/absence models and only 3 out of the 19 abundance models showed improved goodness of fit performances when an area-by-year interaction term was added (poor cod, starry smooth-hound and thornback ray), without statistically significant differences in the distribution outputs (Table S2.1 and S2.2). Furthermore, 14 out of the 19 presence/absence models did not converge with an area-by-year interaction term.

Final predictions are obtained by the product of presence probabilities and CPUE. Knowing the sediment characteristics of each area, the total abundance in each cell is computed by reallocating the environmental effects in proportions to sediment types coverage.

Finally a limit of 10 observations per cell in both OBSMER and CGFS was determined as the threshold above which the square was kept in the analysis, resulting from a trade-off between a sufficient coverage of the EEC and a consistent number of observations

(Figure 2.2). By applying this limit and our spatial resolution to survey data, 88% of the EEC is covered (for OBSMER data this percentage is variable among month and species). In comparison, using cells of 0.4° x 0.4° instead of 0.3° x 0.3° leads to the representation of 90% of the Eastern English Channel, while using smaller cells of 0.2° x 0.2° only allows representing 68% of the Eastern English Channel. Thus our choice seems to be the best trade-off between precision and coverage.

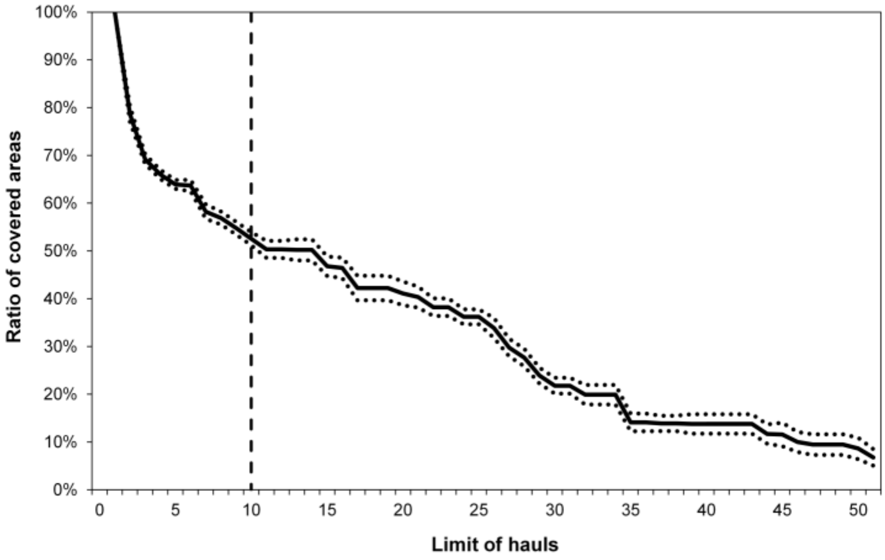

Figure 2.2. Mean percentage of cells kept in the analysis according to the minimal threshold of hauls set per cell. Dotted lines represent the standard deviation along the 19 species. Dashed vertical line represents the chosen limit of 10 observations.

Importantly, the explained variables presented above are likely to include inherent spatial dependence (spatial autocorrelation SAC; Legendre, 1993), owing to the nature of the data at hand. As a result, the values of the dependent variables are unlikely to be conditionally independent as assumed in these models. The SAC inherent to both CGFS and OBSMER data was here accounted for by applying the Moran's Eigenvectors (MEV) mapping method following the protocol described by Cormon *et al.* (2014) with R packages {spdep} (Bivand *et al.*, 2013), {spacemakeR} (Dray, 2013) and {packfor} (Dray *et al.*, 2013). The concept of this method is to allow the translation of the spatial arrangement of the data into a set of explanatory variables through the eigenvector decomposition of data coordinate connectivity matrix previously built (Dormann *et al.*, 2007). For OBSMER data, MEV are computed and selected for each month separately, and then integrated in the whole model set of parameters. Temporal dependencies were not examined in the study.

2.2.4 Assessing the similarity between fisheries- and survey-based spatial abundance

The data treatment described above allows to produce monthly maps of species abundance distribution. While the global seasonal patterns obtained can be compared with disparate knowledge available for some species, the degree of reliability of the fine scale spatial distribution derived from commercial data can be addressed through comparison to survey-based maps.

To quantitatively determine how similar spatial distribution derived from commercial and survey data are at fine scale, we estimated, for October, the local overlap between distributions, using the geostatistical index Local Index of Collocation (LIC, Woillez *et al.*, 2009):

$$\text{LIC} = \frac{\sum z_{obsmer}(i) z_{survey}(i)}{\sqrt{\sum z_{obsmer}^2(i) \times \sum z_{survey}^2(i)}} \tag{2.5}$$

where $z_{obsmer}(i)$ and $z_{survey}(i)$ are the computed abundances in area i, as provided by OBSMER and CGFS data, respectively. LIC was computed using R package {RGeostats} (Renard *et al.*, 2014). This spatial indicator is considered appropriate to assess local overlapping between two densities of population, without taking the mean abundance into account (Woillez *et al.*, 2009).

This index theoretically ranges between 0, showing absolutely no match between the two spatial distributions ($z_{obsmer}(i) = 0$ if $z_{survey}(i) > 0$, $z_{survey}(i) = 0$ if $z_{obsmer}(i) > 0$, $\forall\, i$), and 1, demonstrating a perfect match between them ($z_{obsmer}(i) = z_{survey}(i)$, $\forall\, i$).

The significance of index values was assessed using random permutations of OBSMER abundance values against constant CGFS ones. This procedure is repeated 5000 times, and the spatial distributions derived from commercial data were considered to overlap spatial distributions derived from the CGFS survey when the actual LIC value was above the 95th percentile of the LIC randomly permutated values.

The Horn's index (Horn, 1966) was also tested for the study, but it provides approximately the same results and is less efficient with extreme values of abundance, thus only results based on LIC are presented.

Finally, to assess the sensitivity of our results to the set of areas being considered, a jackknife resampling was operated for all species, by removing sequentially each area, and by evaluating its impact on LIC significance.

2.2.5 Comparing yearly abundance indices

Additionally to the spatial abundance, the model provides a year effect that can be used to derive an inter-annual abundance index in both survey and OBSMER data following the method of Lo *et al*. (1992). The time series ranges from 1998 to 2014 for survey data (2005-2014 for cephalopods series) and from 2003 to 2015 for OBSMER data. It is obtained by varying only the year parameter on the computation of CPUEs, and taking the mean of all areas in natural space to avoid variance disparities. Pearson's correlation index was computed to quantify the correlation between abundance indices from the two data sources.

2.3 Results

2.3.1 Monthly spatial distribution patterns

In the delta-GLM applied to commercial CPUEs, every parameters were kept, with an exception for the sediment parameter in the presence/absence model of cuttlefish (Table S2.3). However, area-by-month was replaced *by* month alone in the presence/absence models of starry smooth-hound, flounder and John Dory. In the delta-GLM applied to survey CPUEs, the parameters selection is more variable (Table S2.4). For example, the year parameter is not kept in both presence/absence and abundance models for tub gurnard, and the sediment one is not kept for three species: cod, pouting and tub gurnard. The area parameter was always significant and kept. The monthly spatial distribution of cuttlefish derived from the delta-GLM models applied to commercial and survey CPUEs is presented in Figure 2.3. This species has been chosen for illustration because it is one of the main species in terms of yields in the EEC (Royer *et al*., 2006). These maps are partial and do not cover the same areas over all months, owing to varying fisheries distributions. The map presented for October results from survey-based information, hence explaining its wider spatial coverage. Some informative spatial patterns can be evidenced for cuttlefish: their quasi-absence in the EEC from January to March, a coastal aggregation along the French coast in May-June, and a more offshore distribution in October-November indicate the existence of a seasonal migration pattern for this species.

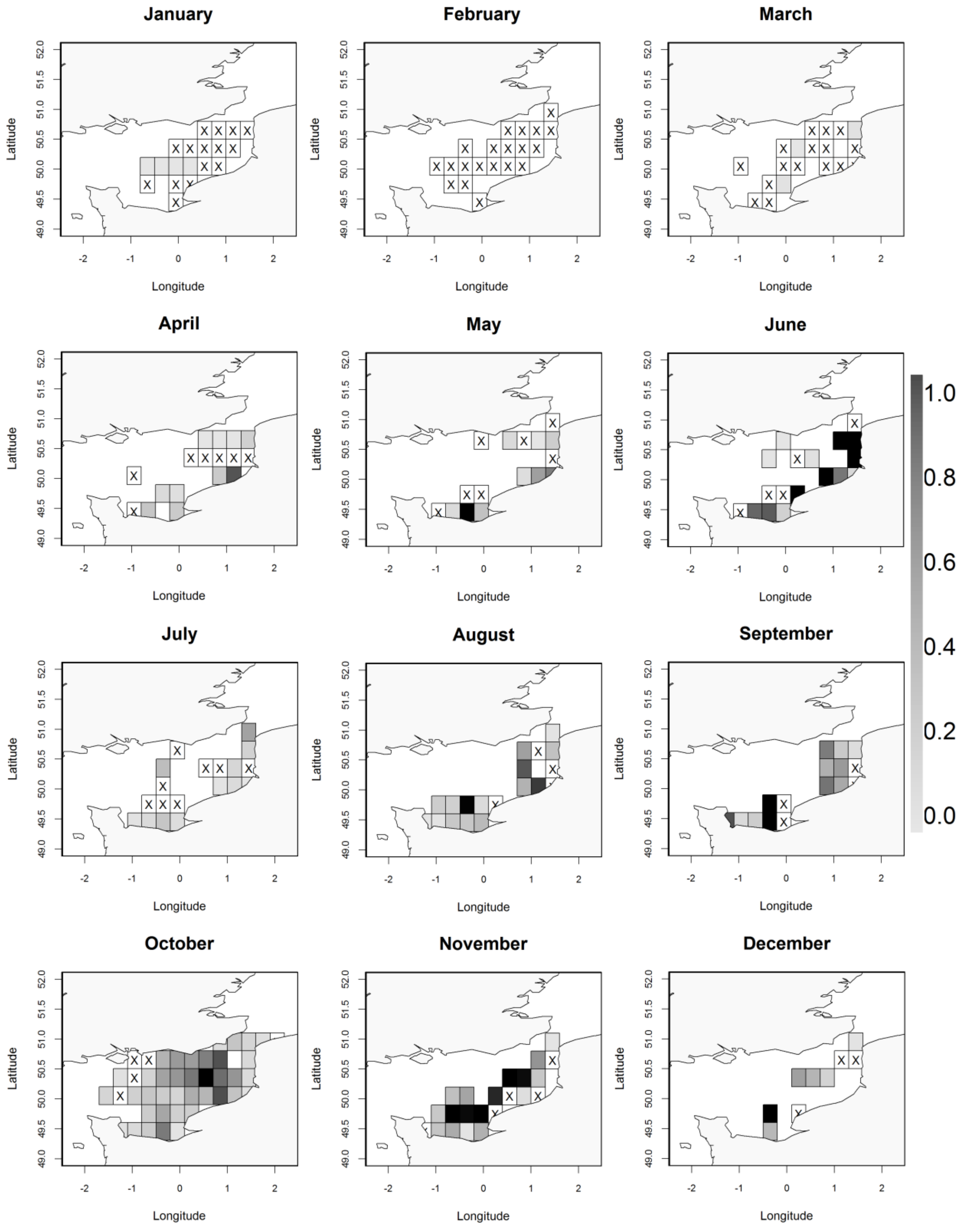

Figure 2.3. Monthly spatial abundance distribution estimated from OBSMER and CGFS for cuttlefish. ‘X’ represents areas where no cuttlefish was ever fished during a month in the database.

2.3.2 Comparison of fine scale spatial distributions from survey data and commercial data

The fine scale match between the spatial abundances estimated from fisheries and survey has been quantified for each species by computing the LIC value, and testing its significance with 5000 random permutations of CPUE abundances. Of the 19 tested species, 9 had a LIC significance above 95%, 6 between 75% and 95%, and only 4 under 75% (Figure 2.4). Considering 95% significance threshold, survey- and fisheries-based spatial distributions were therefore found to overlap for half of the species under investigation. Although the distribution of LIC values resulting from the permutation tests is variable among species, the results highlight that almost all species with a LIC above 0.6 showed high significance (except John Dory for which the LIC value of 0.67 falls just below the third quartile of permutations), while species with a LIC value smaller than 0.6 showed no significant overlap (except cod with a LIC of 0.52). It can also be noted that John Dory, the only species showing no significant overlap despite a LIC above 0.6, shows a very low variability of LIC in the permutation test.

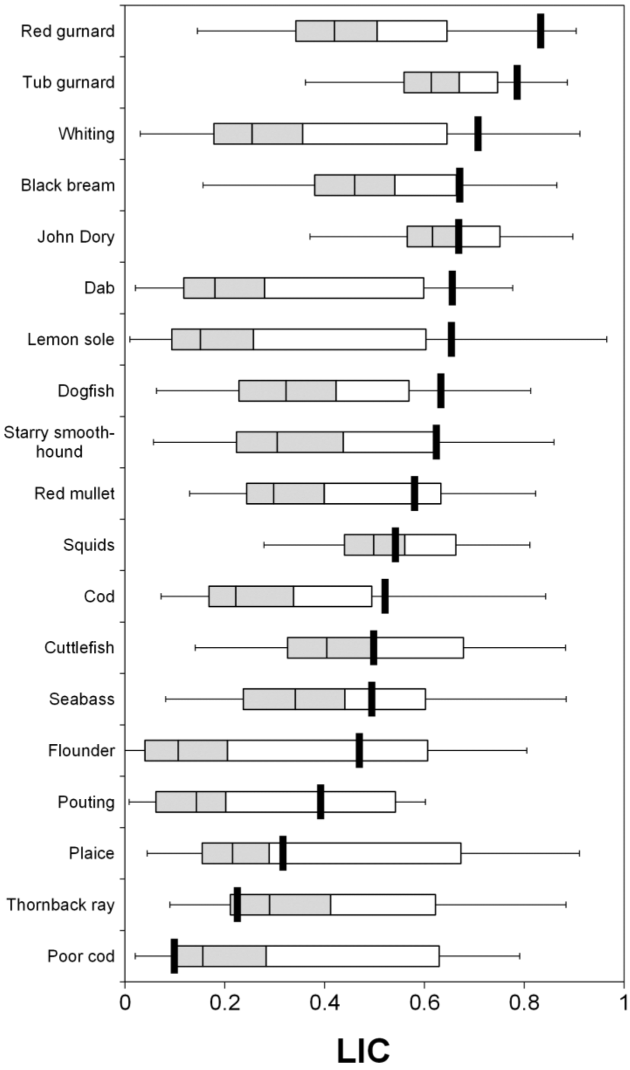

Figure 2.4. Actual Local Index of Collocation of the 19 species investigated in the Eastern English Channel (bold black line), compared to the distribution of 5000 randomly simulated LICs (permutation test). Minimum and maximum simulated LIC are represented by the short segments. Grey boxes represent Q1, median and Q3 ranges of simulated LICs. The white box represents the range of values between Q3 and the 95th percentile of simulated LICs.

Thornback ray, poor cod, plaice and pouting had the lowest LIC values, under 0.4. Cephalopods species, cuttlefish and squids, had intermediate LIC values of 0.50 and 0.54, respectively, and both were between the median and the 95th percentile. Finally, of the four flatfish species, i.e. common dab, lemon sole, European flounder and plaice, only common dab and lemon sole had a significant LIC.

2.3.3 Sensitivity to areas

In order to assess the sensitivity of the results obtained, a jackknife resampling was performed and results were analysed in regard to some characteristics of sensitive areas (Table 2.2). Of the 10 species for which no overlap could be evidenced, red mullet was the only one for which LIC became significant by removing one area. Red mullet original LIC significance value compared with permutations was close to 0.05, and dropped below that threshold with the removal of either the first or second top abundance areas as derived from CGFS information (ranked 8th and 4th building on OBSMER data).

Table 2.2. Jackknife results and main data attributes for species that did not initially demonstrate significant overlap between OBSMER and Channel Ground Fish Survey (CGFS) distributions. LIC: original value of Local Index of Collocation. p-value: situation of the LIC value related to the distribution of permutation tests (values below 0.05 indicate significant overlap). JK: number of areas which prevented from having significant overlap (with total number of areas). % abundance OBSMER & CGFS: percentage of abundance represented by these sensitive areas among all OBSMER and CGFS areas respectively (with ranking among all areas).

	LIC	p-value	JK	% abundance OBSM	% abundance CGFS
Seabass	0.49	0.156	0 (24)	/	/
Squids	0.54	0.440	0 (20)	/	/
Red mullet	0.58	0.063	2 (23)	5.8 (4/23) 3.7 (8/23)	12.4 (2/23) 19.2 (1/23)
Flounder	0.47	0.118	0 (21)	/	/
Plaice	0.32	0.194	0 (24)	/	/
Thornback ray	0.22	0.703	0 (22)	/	/
Cuttlefish	0.50	0.248	0 (21)	/	/
Pouting	0.39	0.108	0 (23)	/	/
Poor cod	0.10	0.768	0 (21)	/	/
John Dory	0.67	0.259	0 (24)	/	/

Among the nine species for which the LIC was significant for all areas being considered, the LIC of seven species became not significant when removing one area (Table 2.3). The LIC of tub gurnard, common dab, lemon sole, starry smooth-hound and lesser-spotted dogfish were thus sensitive to the absence of one particular area, ranked first or second in abundance. The LIC of cod and black seabream became not significant with the removal of one area among a list of 6 and 8, respectively. Their original p-values, close to the 0.05 threshold (i.e. 0.046 and 0.043), can partially explain the high number of sensitive areas.

Table 2.3. Jackknife results and main data attributes for species that did initially demonstrate significant overlap between OBSMER and Channel Ground Fish Survey (CGFS) distributions. LIC: original value of Local Index of Collocation. p-value: situation of the LIC value related to the distribution of permutation tests (values below 0.05 indicate significant overlap). JK: number of areas which allowed having significant overlap (with total number of areas). % abundance OBSMER & CGFS: percentage of abundance represented by these sensitive areas among all OBSMER and CGFS areas respectively (with rank among all areas).

	LIC	p-value	JK	% abundance OBSM	% abundance CGFS
Red gurnard	0.83	6e-04	0 (24)	/	/
Tub gurnard	0.79	0.016	1 (24)	11.1 (2/24)	11.3 (1/24)
Cod	0.52	0.046	6 (24)	1.9 (2/24) 0.0 (23/24) 45.3 (1/24) 0.2 (20/24) 3.8 (7/24) 0.0 (24/24)	0.7 (19/24) 1.2 (14/24) 10.6 (2/24) 0.5 (20/24) 2.3 (12/24) 3.4 (10/24)
Common dab	0.66	0.019	1 (23)	22.2 (1/23)	43.1 (1/23)
Whiting	0.71	0.030	0 (23)	/	/
Lemon sole	0.65	0.021	1 (22)	25.5 (1/22)	27.1 (1/22)
Starry smooth-hound	0.62	0.046	1 (22)	14.9 (3/22)	25.9 (1/22)
Lesser-spotted dogfish	0.63	0.020	1 (24)	27.9 (1/24)	12.2 (2/24)
Black seabream	0.67	0.043	8 (23)	0.2 (18/23) 0.0 (20/23) 0.0 (21/23) 0.2 (17/23) 0.0 (22/23) 7.8 (5/23) 0.0 (23/23) 14.8 (2/23)	1.0 (17/23) 0.1 (22/23) 0.1 (21/23) 0.3 (20/23) 0.0 (23/23) 12.6 (3/23) 1.6 (13/23) 12.7 (2/23)

2.3.4 Rebuilding of yearly abundance index

The *year* effect derived from each delta-GLM analysis can be considered as a yearly abundance index for each species. Figure 2.5 displays two examples of different levels of fit between survey and commercial data, ranging from good visual fit, for cod, to poor fit for black seabream. Cod abundance index shows consistent fluctuations in both survey and commercial data, with higher abundance from 2007 to 2009 followed by 4 years of lower abundance. Black seabream abundance index derived from survey displayed a general decrease from 2004 until 2014. in contrast, the index derived from commercial CPUEs shows an increase over this period. The Pearson's correlation index was computed to quantify the link between the two abundance indices produced for each species (Table 2.4). The results indicated that spatial overlap represented by LIC's significance is not necessarily related to concordant abundance indices time series, as most of the species with a significant LIC value have an intermediate correlation (Figure S2.1). Black seabream, with a significant LIC, has even the third lowest value for Pearson's correlation metrics.

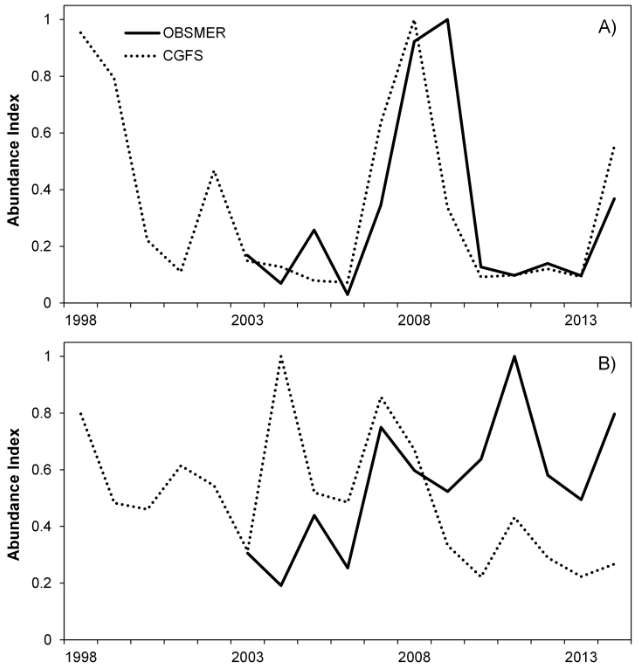

Figure 2.5. Annual abundance index estimated from Channel Ground Fish Survey (CGFS; dotted line) and OBSMER (solid line) for A) cod and B) black seabream.

Table 2.4. Correlation between Channel Ground Fish Survey (CGFS) and OBSMER annual abundance indices assessed by Pearson's correlation index (Pearson). LIC values are also reported for 18 species Eastern English Channel species. Tub gurnard is not represented because the year effect was not significant ($p > 0.05$) in the survey model. * emphasizes species for which spatial overlap was significant ($p < 0.05$).

Common name	Pearson	LIC
Poor cod	0.81	0.10
Cod	0.72	0.52*
John Dory	0.71	0.67
Red mullet	0.66	0.58
Plaice	0.65	0.32
Lemon sole	0.63	0.65*
Cuttlefish	0.51	0.50
Common dab	0.24	0.66*
Red gurnard	0.20	0.83*
Whiting	-0.01	0.71*
Starry smooth-hound	-0.05	0.62*
Thornback ray	-0.08	0.22
Squids	-0.12	0.54
Pouting	-0.13	0.39
Lesser-spotted dogfish	-0.22	0.63*
Black seabream	-0.23	0.67*
Flounder	-0.27	0.47
Seabass	-0.50	0.49

2.4 Discussion

2.4.1 Seasonal distribution patterns of the main fishing resources in the EEC

Our results show the usefulness of fisheries data to infer, in combination with surveys, the spatial and seasonal distributions of several species. The spatial and seasonal distribution of cuttlefish, one of the main commercial species for French fleets (Royer *et al*., 2006), is in agreement with literature. Indeed, from the examination of landings data, cuttlefish adults are known to start migrating in October to spend winter in the Central and Western English Channel, and to be inshore in the Eastern English Channel during summer for feeding and reproduction (Royer *et al*., 2006). Other remarkable life distribution can be derived from the maps (see Figures S2.2-S2.19), like the high winter abundance of squids in the EEC, confirming previous knowledge (Royer *et al.*, 2002), or the quasi-absence of red mullet in the East of the EEC in the beginning of the year while it concentrates in the East central part of the EEC in the end of the year, which adheres to the conclusions of Mahé *et al*. (2005) based on fishers' interviews. On the contrary the spatial distribution of other species remains more stable through the year, e.g. red gurnard in the centre of the EEC, or European flounder inshore except during the winter period, as described by Skerritt (2010). Finally punctual abundance or absence can be detected, like the high concentration of cod along the English coast in June and in the Dover Strait in November, or the high presence of black seabream in the centre of the EEC in February, contrasting with its absence in the eastern part, consistent with Pawson (1995).

2.4.2 Coherence between fisheries-dependent and -independent abundance indices

In addition to the accordance between the global seasonal pattern produced here and the available literature, our results also show that half of the species' spatial distributions exhibited good coherence at fine scale across the two data sources. This conclusion built on an analysis of the LIC overlap metric, the statistical significance of which was quantified using a permutation test. Prior to this study, LIC values were compared with and have been found very close to Horn index values. The Horn index is another overlap metric that is commonly used in trophic ecology, and for which a value > 0.6 is usually considered significant, without further testing (Scrimgeour and Winterbourn, 1987). Our results cross-checked this approach. Except for John Dory (i.e. LIC = 0.67) and cod (i.e. LIC = 0.52), every species' distribution with a LIC above 0.6 were significant. The unexpected outcome obtained for John Dory reveals a shortcoming of the method we applied to assess overlap significance. Indeed, when abundance is homogeneously spread in the entire study area (here the EEC), LIC can be above 0.6 and still non-significant when compared with values resulting from the permutation test. Actually, the LIC (as well as the Horn index) random permutation test can only be efficient with areas of contrasted abundance, as demonstrated by lemon sole or common dab with one area of high abundance contrasting with relatively low values. Therefore, for the evenly distributed John Dory spatial distributions derived from survey and fisheries data can be considered to be close.

Concerning the remaining half of species with lower coherence, a number of reasons can be invoked to explain the discrepancies observed. The results of jackknife analysis demonstrated the impact of some influential areas on the result of the LIC, which cannot be observed depending on the fishers' spatial distribution in October, and highlight the sensitivity of using fine scale comparison when high abundance areas are not available. Another issue is a possible non-proportionality between CPUE and abundance (Hilborn and Walters, 1992). Indeed, commercial fisheries are expected to concentrate their activities into attractive areas (Gillis, 2003). This issue was addressed by standardizing CPUEs using a delta-GLM, and by filtering out spatial auto-correlation. Owing to the limited amount of data, however, SAC correlations could not be computed separately for each year. This could be a concern, as species presence in a precise area/season may vary from one year to another. Thus, a more realistic approach could consist of computing SAC separately for each year, which could not be achieved in this study owing to the low number of observations in the dataset. For similar reasons, the CPUE delta-GLM could not be applied to each gear separately. Instead, observations from the different gears were analysed through the same model, where gear type was treated as an explanatory variable. This approach allowed to estimate the overall impact of gears on CPUE. However, more specific effects of gear types on CPUEs (e.g. selectivity, saturation) could not be fully addressed. In particular, the selectivity of large individuals could be a challenge, as the trawl selectivity ogive is sigmoid-shaped, while that of gillnets could be bell-shaped, or bi-normal, reducing the catch of larger individuals (Dickson *et al*., 1995). Among other potential limits, the soaking time of gillnets is much longer compared with trawls, and it is more subject to saturation effect, which could result in an asymptotic relationship between catches and fishing time (Hickford and Schiel, 1996).

Still, the lack of overlap between the spatial distributions derived from fisheries-dependent and -independent abundance indices for some species could also be explained by their actual biological and ecological characteristics. These could have strong impact on abundance estimations, particularly if only few observations are available within an area. Based on a scientific protocol, the CGFS sampling strategy is fixed and the timing of the survey almost does not vary from one year to the other. However, the EEC ecosystem constitutes for several species a migration path between the North Sea and the Atlantic Ocean, and this can lead to biased estimates of abundance based on survey conducted at a fixed period. For example, red mullet migrates during fall from the southern part of the North Sea to the Western English Channel (Mahé *et al*., 2005), but its migration timing appears variable across years (Carpentier *et al*., 2009), which could lead to high variance in some areas and thus causes difficulties to obtain a clear static mean distribution.

Pouting, poor cod, thornback ray and plaice have the lowest LIC in our results. Various species are known to change their behaviour between day and night (Pitcher, 1992), which may affect our results (Fréon *et al*., 1993). Indeed, pouting are known to have diel activity patterns, forming shoals near wrecks or rocks during the day and disperse during the night for feeding (Jensen *et al.*, 2000). Thornback rays predate also at night and burry in the sand during the day (Wilding and Snowden, 2008). There is evidence that poor cod is mainly caught at night (Gibson *et al*., 1996). Concerning plaice, differences in catches between day and night are less clear and vary across studies (De Groot, 1971; Arnold and Metcalfe, 1995). Surveys like CGFS occur only during daylight, while about half of the fishing operations are conducted during the night. Including explicitly the time of the day in our model would be a way forward, which would require a larger set of data (Benoît and Swain, 2003). Finally, variability in species distribution can occur by environmentally-driven spatial and annual shifts (Verdoit *et al*., 2003). As previously evoked, with sufficient data, dealing with these shifts would require interaction parameters, introduced by fixed effects (with associated restrictions, e.g. Thorson and Ward, 2013) or random effects (with corresponding bias-correction, e.g. Thorson and Kristensen, 2016). The high number of presence/absence models that did not converge with an area-by-year interaction can be explained by the small number of observations for each occurrence (i.e. on average 2 per area-by-year), often 0 or 1 for a substantial part of the new parameters. Increasing the number of iteration failed to improve model convergence.

In the coming years, the growing collection of data may allow for accommodating such processes, but also fine-scale targeting (e.g. Thorson *et al*., in press), and hence lead to more reliable abundance estimates per area for a broader coverage of the EEC. A next step could then be to derive spatially-explicit estimations of fish lengths, building on innovative approaches (e.g. Petitgas *et al*., 2011; Nielsen *et al.*, 2014). These could help to distinguish between mature and non-mature individuals, which are driving fish movement (Pittman and McAlpine, 2001).

2.4.3 Uses of data collected on-board commercial vessels

Another objective of this study was to provide annual series of abundance indices. The comparison between fisheries-dependent and -independent time series suggested contrasted results across species.

For species like cod (Figure 2.5a) and lemon sole, both the spatial and annual abundance distributions derived from fisheries and survey data were reasonably consistent. However, consistent annual trends across the two data sources were not necessarily linked with spatially overlapping distributions, e.g. cuttlefish or red mullet. Potential reasons for the lack of spatial overlap for such species were discussed above.

For other species, a good spatial overlap between fisheries-dependent and -independent abundance distributions was not necessarily associated with synchronous time series (e.g. black seabream, Figure 2.5b). This could be owing to data limitations, but also to some hyperstable relationship between abundance and CPUE (Hilborn and Walters, 1992), that could not be completely filtered out by our standardization approach. In addition, the species which present a good spatial overlap can be subject to intra-annual fluctuations of abundance owing to high exploitation, migrations and recruitment (Gillis and Peterman, 1998), that could strongly impact the mean annual abundance value.

Finally, abundance indices derived from fisheries data could be an appropriate source of information to provide seasonal and spatial distributions, particularly during periods where surveys do not operate. A better overview of species migrations is first a progress in current knowledge on species ecology, which could further be linked with seasonally-explicit abiotic and biotic environmental conditions. Secondly such information could be linked with fishers' movement throughout year, which could enhance our knowledge on fishers-resource interactions. Thirdly, seasonally- and spatially-resolved information such as that output from this study could also serve to calibrate complex end-to-end models such as Atlantis (Fulton *et al.*, 2007), OSMOSE (Shin and Cury, 2001), ISIS-Fish (Pelletier *et al.*, 2009) or Ecospace (Walters *et al.*, 1999), and enhance their capacity to evaluate ecosystem-based management strategies (e.g. closed areas and seasons). Finally, further studies could validate the assumptions that on-board commercial data give a better overview of spatial distributions than survey for a small portion of species (e.g. pouting). However, the distributions derived for species presenting strong variability in selectivity or behavioural pattern (e.g. diel variations or migrations) should be interpreted with caution.

In addition to spatial distributions, annual abundance indices derived from fisheries data could potentially complement the survey-based series used in stock assessments. This would require, as a follow-up to this study, to structure those fisheries-based annual indices by length and/or age, and perhaps to try to obtain such indices on a shorter duration than year. Previously, fisheries-based abundance indices should be closely examined, on a case-by-case basis, cognisant of the life cycle and exploitation features of the species under investigation.

2.5 Conclusion

This study shows the potential of combining fisheries-dependent and -independent data to increase our knowledge on the seasonal and spatial distribution of several marine species. Even if the comparisons realized during this study showed that fisheries-dependent data did not always mirror the time and spatial survey-based distribution of some species, they still remain a valid source of information. Fisheries-dependent data are relatively abundant, opportunistic and cheaper than survey data, and their use should be encouraged, especially to reflect abundance distributions in areas and seasons that are not covered by surveys. Moreover, some species are poorly sampled by surveys owing to their diel behaviour, and the use of at-night observations on-board commercial vessels could help better inferring their spatial distributions. The method we used here is relatively simple compared with, e.g. log-Gaussian Cox model method developed by Kristensen *et al.* (2014). Still, the quality of the resulting outputs we presented was assessed, and these provide valuable information on spatial and temporal species distributions, which concur with existing ecological knowledge. This approach would benefit from a better spatial representation along the English coastline, and further cooperation, data sharing and on-board observation program strengthening could substantially enhance our understanding of the spatio-temporal distribution of marine species in the Eastern English Channel.

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Chapter 3

Improving the perception of fishing effort and pressures in mixed fisheries using spatial overlap metrics

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Abstract

Managing mixed fisheries requires understanding fishers’ behaviour to allow predicting future fisheries distribution and impact on marine ecosystems. A new approach was developed to compare fine scale fishing effort distribution of Eastern English Channel (EEC) bottom otter trawlers, to the monthly- and spatially-resolved abundance distributions of commercial species, over the period 2008-2014. First, the added-value of using species-specific spatial overlap metric to quantify effective fishing effort and improve the relationship between fishing effort and fishing mortality was assessed. Second, based on the Ideal Free Distribution (IFD) theory, the species-specific weights (reflecting targeting intensity) given by fishers to different species were estimated by maximizing the overlap, measured by a Local Index of Collocation, between target species assemblage and fishing effort distributions in October. At a seasonal scale our results emphasized the importance of cuttlefish and red mullet for the global distribution of EEC bottom trawlers. In October, cuttlefish and red mullet were clearly more determining fishers’ location choice than historically harvested species such as cod or whiting, and also than the overall expected revenue. This is likely due to external constraints such as travelling costs or low cod quota, causing IFD assumptions violated. This study provided empirical evidence of the importance to get good insights into spatio-temporal distributions of stocks and fleets to understand fishers’ behaviour and *in fine* improve fisheries management advices.

Keywords: fishing effort, seasonality, target species assemblages, Eastern English Channel.

3.1 Introduction

The implementation of Ecosystem-Based Fisheries Management (EBFM) requires new methods to assess and manage exploited marine ecosystems (FAO, 2001; Pikitch *et al*., 2004; Long *et al.*, 2015). Successfully implementing the EBFM requires a thorough understanding of the mechanisms inherent in fishers' behaviour and particularly their relation with targeted stock assemblages (Fulton *et al*., 2011; Marchal *et al*., 2013; van Putten *et al*., 2013).

Understanding the relation between fishers and stocks' distributions is particularly challenging when it comes to (demersal) mixed fisheries. Mixed fisheries simultaneously harvest several species, the composition of which may change according to seasons (Poos *et al*., 2010), with a target-bycatch dichotomy in catches (Wilson and Jacobsen, 2009) associated with a poor selectivity (Marchal, 2008). Fishing fleets and gears operating in such fisheries interact technically (Ulrich *et al*., 2012; Cardoso *et al*., 2015), and are prone to high discard rates (Catchpole *et al*., 2005; Johnsen and Eliasen, 2011), particularly of undersized and/or over-quota fish catches (Andersen *et al*., 2010; Fernandes and Cook, 2013). These technical interactions make fisheries management challenging, and especially where species/stocks are managed individually.

To address this challenge and improve mixed fisheries management, numerous research studies have been carried out (i) to better quantify fishing effort and, (ii) to anticipate the dynamics of fishers' behaviour and its impact on the several species targeted or caught as bycatch when fishing patterns are changing.

With regards (i), a number of fishing effort analyses have focused on the identification of manageable fishing units (Laurec *et al.*, 1991; Marchal, 2008; Ulrich *et al*., 2012). Other fishing effort studies have focused on the quantification of fishing power and/or of the relationship between fishing effort and fishing mortality, with a focus on technical development (Kirkley *et al.*, 2004; Marchal *et al.*, 2007; Eigaard *et al*., 2014) and tactical adaptations (Hilborn, 1985; Rose and Kulka, 1999; Salthaug and Aanes, 2003). The metrics considered in these studies were used to standardise nominal fishing effort and calculate an effective fishing effort, thereby improving the estimation of the actual fishing pressure exerted on fish stocks. Such metrics, however, were derived from vessels, gears and/or skippers' characteristics only, and hence not explicitly considering the relative availability of the different targeted species. In our study, we calculate effective fishing effort including fish availability and fishers' ability to target and catch fish, which we quantify by the overlap between stocks and fishers' spatial distributions.

The first objective of this study will then be to quantify the effective fishing effort in a mixed fishery's context, and *in fine* improve the relationship between fishing effort and fishing mortality (Gascuel *et al.*, 1993; Winker *et al.*, 2013; García-Carreras *et al.*, 2015), using a combination of vessel characteristics and species-specific spatial overlap metrics.

With regards (ii), fleet dynamics has been subject to considerable attention in the past decades (see Van Putten *et al.*, 2012 for a review), a process largely supported by fine-scale and georeferenced data becoming increasingly available (e.g., Bastardie *et al.*, 2010; Hintzen *et al.*, 2012). Different theories have been proposed to explain the mechanisms of fishers' behaviour. The Ideal Free Distribution (IFD, Fretwell and Lucas, 1970; Fretwell, 1972) is one of the most widespread conceptual approaches that has been applied to predict the distribution of foragers (here fishers) in relation to available resources (Kacelnik *et al.*, 1992; Kennedy and Grey, 1993). The IFD states in particular that the number of foragers that will aggregate in various areas is proportional to the amount of resources these may supply. In a fisheries context, the spatial distribution of nominal fishing effort and of their harvested resource would then overlap (Abrahams and Healey, 1993; Gillis *et al.*, 1993; Rijnsdorp *et al.*, 2000). In mixed fisheries, where several fish species are harvested together, the amount of resources has often been translated into aggregated economic revenue metric like the value per unit of effort (VPUE) (e.g. Rijnsdorp *et al.*, 2000; Abernethy *et al.*, 2007; Gillis and van der Lee, 2012), making the hypothesis that fishers would try to maximize their expected revenue more than the volume of species they could catch.

Applying the IFD results in fishers' behaviour being fully driven by short-term economic consideration: the species with the largest expected return is the most targeted.

However, many studies have shown that species targeting could also be driven by other factors including regulations as well as longer-term economic and social considerations. For instance, valuable species may be avoided, when fishers do not have a sufficient quota provision to harvest them (e.g., choke species; Schrope, 2010; Ulrich *et al.*, 2011; Baudron and Fernandes, 2014), or because they do not have a market channel to sell them (Marchal *et al.*, 2009), or because targeting these species is not part of their habit (Vermard *et al.*, 2008; Marchal *et al.*, 2009; Girardin *et al.*, 2017), thereby inducing deviations from the basic IFD predictions. These results suggested in particular that the relative interest fishers give to the different species they harvest is not entirely reflected by their landed value.

The second objective of this study is then to quantify, using a novel method, the relative value fishers assign to their different targets, and to link it with current knowledge of their ecological, economic and regulatory environment.

The research pertaining the two objectives of this study will be evaluated for a typical EU mixed fishery, consisting of French otter trawlers harvesting demersal species in the Eastern English Channel.

3.2 Material and methods

3.2.1 Study area and fleet characteristics

The Eastern English Channel (EEC; ICES Division 27.7.d) is delineated by latitudes 49.3°N and 51°N and longitudes 2°W and 2°E (Figure 3.1). This shallow area constitutes a corridor between the northeast Atlantic Ocean and the North Sea, and is home to intense and diversified human activities including fishing, shipping, wind farms, aggregate extraction (Ulrich *et al.*, 2002; Dauvin, 2012). This area is also important for several commercially important migratory species, e.g., red mullet (*Mullus surmuletus*) (Mahé *et al*., 2005), mackerel (*Scomber scombrus*) (Eltink *et al*., 1986), herring (*Clupea harengus*) (ICES, 2015), European seabass (*Dicentrarchus labrax*) (Pawson *et al*., 2007) and cuttlefish (*Sepia officinalis*) (Boucaud-Camou and Boismery, 1991).

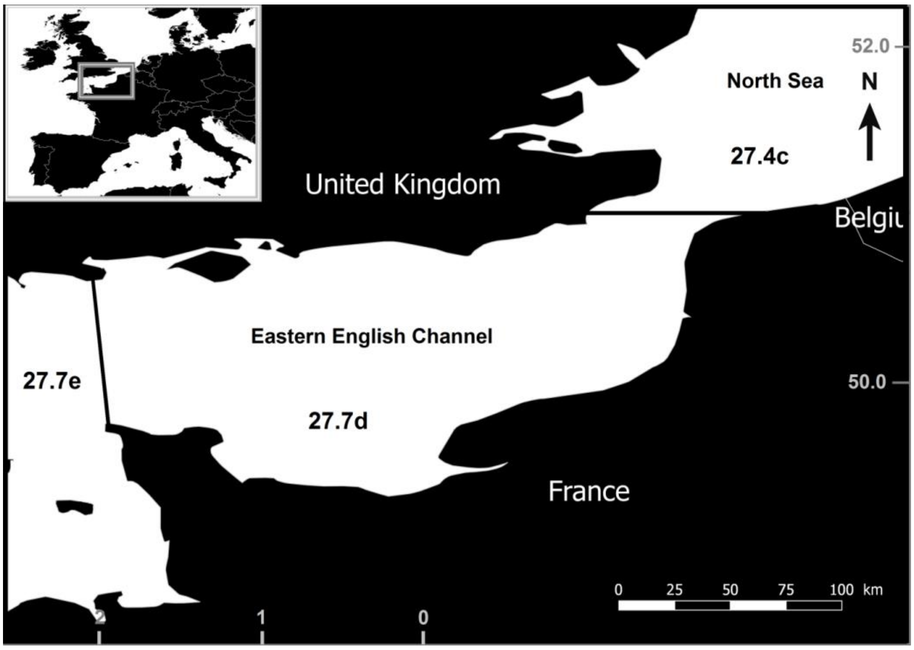

Figure 3.1. Study area of the Eastern English Channel, corresponding to the ICES Division 27.7d.

The current study focusses on French exclusive (i.e. keeping the same gear according to the season) bottom otter trawlers (OTB), of length above 18m and using a mesh size above 80mm. This fleet category is an archetype of mixed fisheries and is studied here for three reasons. First, this fleet category gets the bulk of yearly French bottom otter trawlers catches for the main demersal species in the EEC (Table S3.1). Second, as non-exclusive OTB are usually smaller than exclusive ones, they mostly operate in coastal areas close to their home harbor thus their spatial distribution is limited and only covers a limited portion of the EEC. Finally, exclusive otter trawlers above 18m generally use the same gear (with a mesh size above 80mm) all year round, making the exploration of their dynamics more tractable. Mesh sizes below 80mm are rarely used by this fleet (for only 5% of their landings in average, see Table S3.1), and only when targeting a reduced list of species (EC, 1998).

Table 3.1. List of Eastern English Channel species considered in this study, with their Minimum Landing Size (MLS, in cm) when existing, the minimum total length L_s (cm) above which individuals are considered to be equally selected by survey and commercial gears, and their closest code in commercial activity calendars.

Common name	**Scientific name**	**MLS (cm)**	**L_s (cm)**	**Activity calendars code**
European seabass	*Dicentrarchus labrax*	36	36	Bass (miscellaneous)
Atlantic cod	*Gadus morhua*	35	35	Cod
Squids	*Loligo* spp.	-	14*	Squids (miscellaneous)
Whiting	*Merlangius merlangus*	27	27	Whiting
Red mullet	*Mullus surmuletus*	-	15	Red mullet (miscellaneous)
European plaice	*Pleuronectes platessa*	27	27	Flatfishes (miscellaneous)
Atlantic mackerel	*Scomber scombrus*	20	20	Mackerel (miscellaneous)
Common cuttlefish	*Sepia officinalis*	-	13*	Cuttlefish, sepia (miscellaneous)
Black seabream	*Spondyliosoma cantharus*	-	17	Sparidae (seabream, dentex, sargo, …)

*mantle length

3.2.2 Data

This study requires spatial distributions of otter trawlers' nominal fishing effort and of the abundance of the main EEC stocks they harvest (Table 3.1). To estimate species distribution, both in time and space, we used the delta-GLM (Generalized Linear Model) approach described by Bourdaud *et al.* (2017) that combines survey and commercial data.

The input data sources for this delta-GLM are the Channel Ground Fish Survey (CGFS) for the fisheries independent data and on-board commercial fisheries observation (hereby named as OBSMER) data for the fisheries dependent information. These data sources are complementary, with CGFS data providing insights into inter-annual patterns (only in October, when the survey is operated), and OBSMER data being fit to investigate seasonal variability. Spatial distribution of species abundances are computed for each species above a length threshold (L_s; Ravard *et al.*, 2014), where individuals are considered to be well sampled. For species with a minimum landing size (MLS) in the EEC, L_s was assigned to that MLS. For others, L_s was approximately set for each species at the length of the highest mode of the length-frequency of combined catches from the different gears (Table 3.1). The delta-GLM applied to OBSMER data contains a maximum of six explanatory variables:

$$logit\left(\mathrm{p}_{\mathrm{v,i,m},y}^{>0}\right) = \beta_i\delta_m + \lambda_y + \omega_g l \ + \vartheta_s \tag{3.1}$$

$$\log\left(IA_{v,i,m,y}^{>0}\right) = \beta_i\delta_m + \lambda_y + \omega_g l \ + \vartheta_s + \varepsilon_{v,i,m,y} \tag{3.2}$$

where $p_{v,i,m,y}^{>0}$ is the mean presence probability and $IA_{v,i,m,y}^{>0}$ the CPUE of a species caught by vessel *v* of length *l* rigged with gear *g* (e.g. bottom otter trawl, trammel net), fishing in (0.3° x 0.3°) area i, year y and month *m*. β_i is the area effect of the fishing operation (treated as factor), δ_m is the month effect of the fishing operation, λ_y is the annual effect, ω_g is the gear effect, ϑ_s is a sediment effect, which accounts for small scale habitat variability and is decomposed into five categories *s*: mud, fine sand, coarse sand, gravel and pebble, based on a sediment map of EEC from Larsonneur *et al.* (1982), and $\varepsilon_{v,i,m,y}$ a term of residual error.

Sediments proved to have the strongest influence on the distribution of species in the shallow EEC, compared to, e.g., depth, temperature and salinity (see Carpentier *et al.*, 2009).

The final predicted abundance values $IA_{v,i,m,y}$ are obtained by the product of presence probabilities $p_{v,i,m,y}^{>0}$ and CPUE for positive values $IA_{v,i,m,y}^{>0}$.

CGFS survey data are always collected in October (i.e. no month effect) with the same research vessel (i.e. no vessel or gear effects), hence the previous formula was reduced to the following, with a maximum of three explanatory variables:

$$logit\left(\mathrm{p}_{\mathrm{i},y}^{>0}\right) = \beta_i + \lambda_y + \vartheta_s \tag{3.3}$$

$$\log\left(IA_{i,y}^{>0}\right) = \beta_i + \lambda_y + \vartheta_s + \varepsilon_{i,y} \tag{3.4}$$

The final predicted abundance values $IA_{i,y}$ are obtained by the product of presence probabilities $p_{i,y}^{>0}$ and CPUE for positive values $IA_{i,y}^{>0}$.

Access to all fishing effort information was provided by the French Directorate for Sea Fisheries and Aquaculture (DPMA). Nominal fishing effort is derived from the Vessel Monitoring System (VMS) and is here defined as an amount of fishing time for each month in a 0.3° x 0.3° area, a scale chosen to match the scale of the species abundance distributions computed above, and corresponding to a trade-off between the amount of data required and a sufficient level of precision (Bourdaud *et al.*, 2017).

To validate our results, we used monthly landings derived from combined logbooks and sales slips record (SACROIS) over the period 2008-2014. Landings data extracted from SACROIS were available by vessel, fishing trip, ICES rectangle and gear used. Activity calendars, collected directly from fishers on a regular basis by Ifremer, provided fishers' targeting intention, i.e. species assemblage targeted during each fishing operation. These assemblages were chosen to be the closest to the studied species (Table 3.1). For French exclusive OTB operating in the EEC during the period 2008-2014, 70% of the target assemblages in the calendars were classified as 'fishes (miscellaneous)', indicating no specific target. Among the remaining records, 79% mentioned targets corresponding to one of the species studied here. Numbers of fishing days are summed by month for each target species and were scaled to the year in order to obtain a monthly relative distribution of fishing time targeting this species.

3.2.3 From nominal fishing effort to spatially-derived species-specific fishing pressure

As defined by Mahévas *et al.* (2004) and Bordalo-Machado (2006), the effective fishing effort (*fe*) measures the real pressure exerted by fishers on a stock during a time unit. It can be defined as the product of the nominal fishing effort (*fn*) and a global fishing power, i.e., the ability of a fisher to catch available fish. The global fishing power combines the capacity of fishers to find the targeted fish (targeting capacity), and the catching capacity inferred from vessels' physical attributes, e.g., vessel length (physical capacity).

Physical capacity *Pc* is assumed to correspond to the $\omega_{g}l$ term for OTB gear in the delta-GLM equations 3.1 and 3.2, and characterizes the impact of vessel length and the gear effect on fish catchability. This parameter is used to weight nominal fishing effort per spatial unit by the length category of each vessel:

$$If_{k,i,m,y} = \sum_v fn_{v,i,m,y} \times Pc_{l,k} \tag{3.5}$$

Where $If_{k,i,m,y}$ is the integrated nominal fishing effort in area *i* for species *k* fished by a vessel *v* of length *l* during the month *m* and year y.

Targeting capacity is then measured for each species *k* as the similarity between the distributions of integrated nominal fishing effort and of harvested fish, using the spatial overlap index LIC (Local Index of Collocation, Woillez *et al.*, 2009):

$$LIC_{m,y} \frac{\sum_i If_{i,m,y} \times IA_{i,m,y}}{\sqrt{\sum_i {If_{i,m,y}}^2 \times \sum_i {IA_{i,m,y}}^2}} \tag{3.6}$$

Noting $IA_{i,m,y}$ the abundance of the species concerned in area *i* during month *m* of year *y*. The LIC was computed using R package {RGeostats} (Renard *et al.*, 2014), it ranges between 0, showing absolutely no match between the two spatial distributions, and 1, demonstrating a perfect match between them.

Finally the monthly relative fishing effort of each year (i.e. between 0 and 1, with the sum of fishing effort in each year = 1, see Figure 3.2) is weighted by the monthly LIC:

$$fe_{m,y} = \frac{LIC_{m,y} \times \sum_v \sum_i fn_{v,i,m,y}}{\sum_m \left(LIC_{m,y} \times \sum_v \sum_i fn_{v,i,m,y}\right)} \tag{3.7}$$

In order to evaluate the respective merits of *fe* and *fn*, in reflecting actual fishing pressure, both effort values were compared with available surrogates of fishing pressure: 2008-2014 averaged monthly landings (as extracted from SACROIS) and fishers' intentions (expressed for each month as the number of days targeting a given species, as extracted from activity calendars).

We computed the residual sum of squares (RSSQ) between the monthly resolved time series of, (1) nominal fishing effort (*fn*) and landings, (2) effective fishing effort (fe) and landings, (3) nominal fishing effort and fishers' intention and, (4) effective fishing effort and fishers' intention. Should the effective fishing effort we processed in this study reflect actual fishing pressure better than nominal fishing effort, we could then expect that *fe* would track monthly variations of both landings and fishers' intentions more closely than *fn*, for those species being targeted by otter trawlers. This improvement would also result in the RSSQ derived from (2) (respectively (4)) being lower than the RSSQ derived from (1) (respectively (3)).

3.2.4 Defining species targeting factors for mixed fisheries from spatial overlap metrics

While the monthly species-specific effective effort computed previously aims at better apprehending the variations of the fishing pressure exerted on each single species, it does not allow evaluating how variable the effort allocated to each species targeting is relative to the others. A combined-species approach is thus required to get better insights into the full dynamics of species targeting in a mixed fisheries context, including swaps from one target to another and their determinism. Combined-species targets were computed building on the maximization of the spatial overlap, measured with the LIC metric, between the distributions of fishing effort and of weighted combined-species abundances. Such approach requires a comprehensive and consistent spatial coverage across all species being considered, and therefore could only be realized for October, the only month covered by a scientific survey over the entire EEC, limiting the results to reflect inter-annual variations with no exploration of seasonal patterns. In order to maximize the LIC, each of the (*k*) species relative spatial distributions (i.e. scaled between 0 and 1) is multiplied by a combined-species targeting coefficient, *β*, which is bounded between 0 and 1 using the transformation:

$$\beta_k = \frac{e^{\alpha_k}}{\sum_k e^{\alpha_k}} \qquad (3.8)$$

Where α is the unconstrained coefficient to be optimized, using the 'optim' function of the R package {stats} (R Core Team, 2013) and the L-BFGS-B method. The objective function to be maximized with respect to α may then be formulated as:

$$\frac{\sum_i \left[fe_i \times \sum_k \left[\left(\frac{e^{\alpha_k}}{\sum_k e^{\alpha_k}} \right) \times \left(\frac{IA_{k,i}}{\sum_i IA_{k,i}} \right) \right] \right]}{\sqrt{\sum_i {fe_i}^2 \times \sum_i \left[\sum_k \left[\left(\frac{e^{\alpha_k}}{\sum_k e^{\alpha_k}} \right) \times \left(\frac{IA_{k,i}}{\sum_i IA_{k,i}} \right) \right]^2 \right]}} \qquad (3.9)$$

If fishers' foraging pattern was in consistency with IFD predictions, one could assume that fishing effort distribution would match EEC wealth distribution. The amount of available revenue *W* generated by each area *i* in year *y* may be computed by:

$$W_{i,y} = \sum_k (IA_{k,i,y} \times Price_{k,y}) \qquad (3.10)$$

knowing the abundance of species *k* in the area obtained from CGFS data and the mean price of the species in October in year *y* (Table S3.2).

Table 3.2. Overlapping LIC values between the distribution of fishing effort and the distribution of potential revenue (revenue-based LIC) or the combined distributions of species (maximized LIC). The difference between both metrics measures the deviation between actual fishing effort distribution and that predicted by IFD.

	Revenue-based LIC	**Maximized LIC**	**Difference**
2008	0.63	0.81	+ 0.18
2009	0.59	0.74	+ 0.15
2010	0.46	0.64	+ 0.18
2011	0.57	0.70	+ 0.13
2012	0.52	0.74	+ 0.22
2013	0.46	0.57	+ 0.11
2014	0.51	0.73	+ 0.22

The LIC values obtained from maximizing (9) are then compared to the LIC obtained from the comparison between fishing effort and available wealth in the EEC, one of the main hypothesis of fishing location driver (van Putten *et al*., 2012).

3.3 Results

3.3.1 Seasonal fishing pressure exerted on each commercial species

The seasonal variation of effective fishing effort is shown for each species separately in Figure 3.2. Fishing pressures (estimated from effective fishing efforts, *fe*) exerted on cuttlefish and seabass have the most pronounced pattern, with peaks reached in autumn for the former, and spring and autumn for the latter. Fishing pressure exerted on other species (see for example plaice or squids) exhibited a smoother seasonal pattern, with a peak in winter. Fishing pressure and landing seasonal patterns match for some species (cod, cuttlefish, plaice, red mullet, squids), but not for others (mackerel, seabass, whiting).

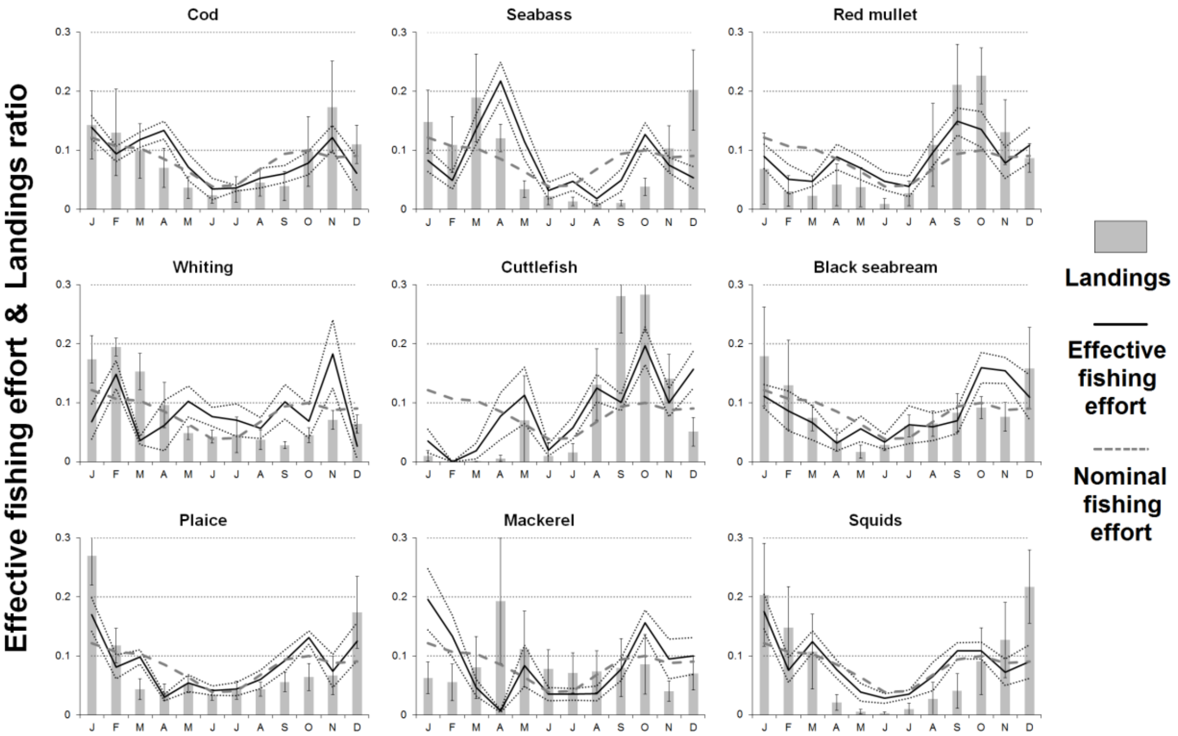

Figure 3.2. Average monthly nominal fishing effort, effective fishing effort and yearly standardized landings of exclusive bottom otter trawlers for nine main commercial species of the Eastern English Channel. Dotted lines and error bars indicate inter-annual variability over the period 2008-2014.

Fishers’ intention from activity calendars were strongly related to the landings, except for cod, red mullet, plaice and black seabream, but were subject to wider inter-annual fluctuations (Figure 3.3; ; see Figure S3.1 for the complete time series). There is a good match between fishing pressure and fishers’ intention for cuttlefish and seabass except in autumn, but not for the other species under consideration.

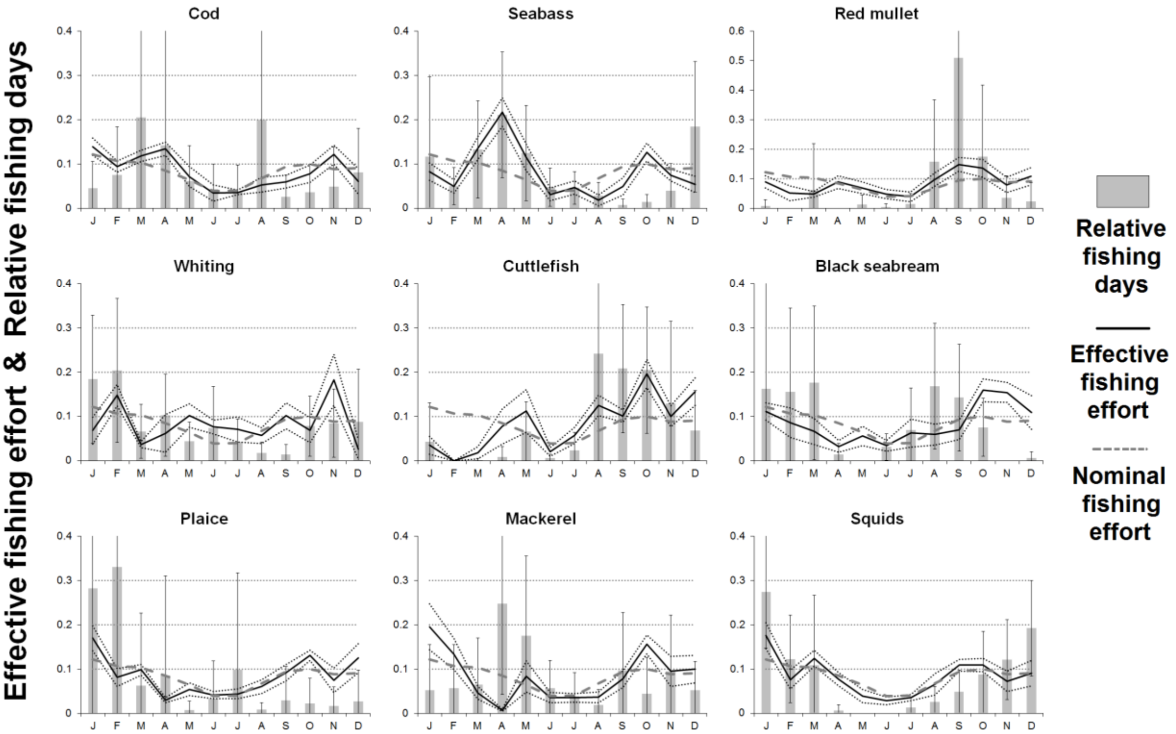

Figure 3.3. Average monthly nominal fishing effort, effective fishing effort and yearly standardized number of fishing days from activity calendars of exclusive bottom otter trawlers for nine main commercial species of the Eastern English Channel. Dotted lines and error bars indicate inter-annual variability over the period 2008-2014.

Considering monthly *fe* instead of *fn* improves substantially the correlation between fishing effort and landings for two species: red mullet and cuttlefish (Figure 3.4A; see Figure S3.2 for the complete time series). At the same time, substituting nominal by effective effort does not improve the correlation between effort and landings, and even deters it for mackerel, whiting, and seabass. Almost similar average results are obtained when investigating the effects of substituting nominal by effective fishing effort on the correlation with the species-targeted numbers of fishing days derived from activity calendars, but these were subject to large inter-annual fluctuations (Figure 3.4B).

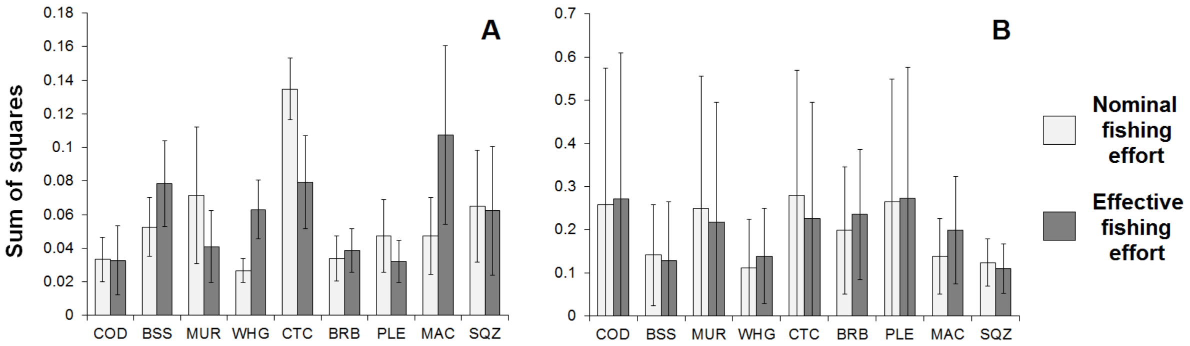

Figure 3.4. RSSQ between the monthly-resolved time series of (A) (i) nominal fishing effort and landings, (ii) effective fishing effort and landings; (B) (iii) nominal fishing effort and fishers' intention, (iv) effective fishing effort and fishers' intention; for nine key commercial species caught by exclusive bottom otter trawlers operating in the Eastern English Channel. Error bars indicate inter-annual variability over the period 2008-2014. COD: cod. BSS: Seabass. MUR: red mullet. WHG: whiting. CTC: cuttlefish. BRB: black seabream. PLE: plaice. MAC: mackerel. SQZ: squids.

3.3.2 Combined-species targeting

The relative target factors obtained by maximizing the (β-weighted) LIC are presented in Figure 3.5 for the six main October commercial species: cod, cuttlefish, mackerel, red mullet, squids and whiting. In October, the two main target species of French exclusive OTB are cuttlefish (44% of the annually averaged sum of target, with a peak of 78% in 2012), and secondly red mullet (22% on average, peaking to 59% in 2009). It is worth noting that the inter-annual variability can be very high for these species. For instance, the targeting factor for cuttlefish goes from 0% in 2009 to 78% in 2012, while the red mullet factor goes from 0% in 2012 to 59% in 2009. The targeting factors of mackerel, cod and squids are less variable over the years, and fluctuate between 0% and 29%. Finally, whiting never appears to be targeted.

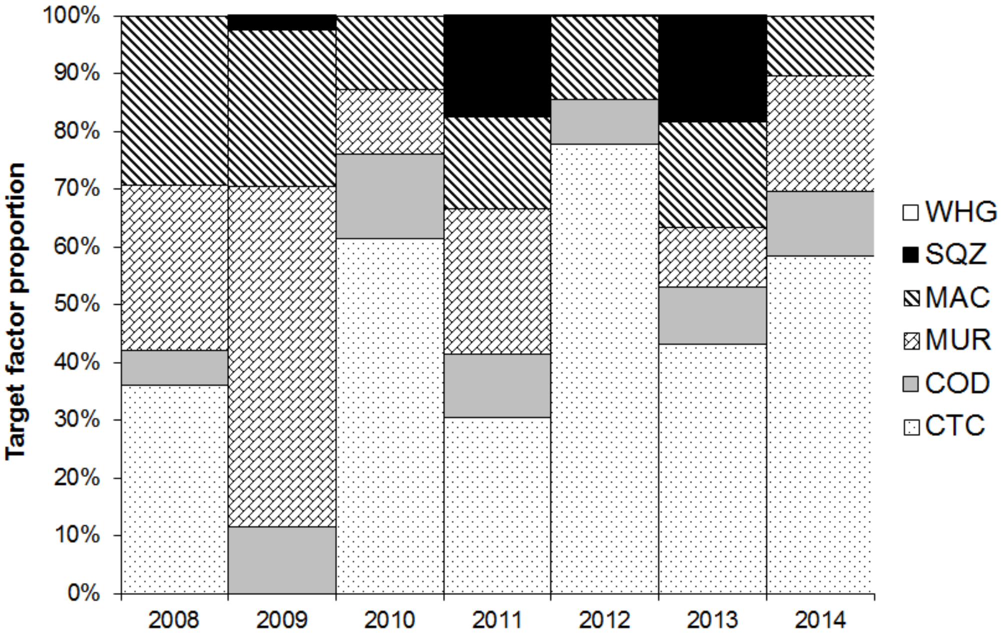

Figure 3.5. Relative target factor in October for whiting (WHG), squids (SQZ), mackerel (MAC), red mullet (MUR), cod (COD) and cuttlefish (CTC) for exclusive bottom otter trawlers in October over the period 2008-2014 in the Eastern English Channel, estimated by maximizing the Local Index of Collocation.

The maximized (β-weighted) LIC value was compared with the revenue-based LIC value, i.e., reflecting the overlap between fishers' distribution and the potential revenue W (Table 3.2). Every year the LIC value obtained by maximization was higher than the revenue-based LIC by at least 0.10, even reaching 0.22 in 2014. The range of maximized LIC is of 0.57-0.81, while the range of revenue-based LIC values is of 0.46-0.63, almost always below the 0.60 threshold below which spatial overlap is not meaningful (Scrimgeour and Winterbourn, 1987). This represents a substantial deviation from the IFD predictions.

3.3.3 Species targeting fluctuations and external factors

Cuttlefish abundance and economic attractiveness is highly fluctuating during the period, with peaks in 2010 and 2012 (Figure 3.6A). Cuttlefish targeting intensity follows economic attractiveness well, except for 2009 where there is no targeting. The correlation is particularly visible in the 2010-2014 fluctuations. Cod abundance and economic attractiveness show a clear decrease from 2008 to 2010, and then remain constant, while remaining quota shows at the same time an increase before being constant (Figure 3.6B). Cod targeting intensity increases from 6 to 15% between 2008 and 2010, when abundance and economic attractiveness both decrease. From 2010 onwards, the cod targeting factor is consistently above 10%, except in 2012. No clear pattern in abundance, attractiveness or remaining quota can be related to the low 2012 targeting. However, it may be noted that during 2012 the targeted species were dominated by cuttlefish (see Figure 3.5).

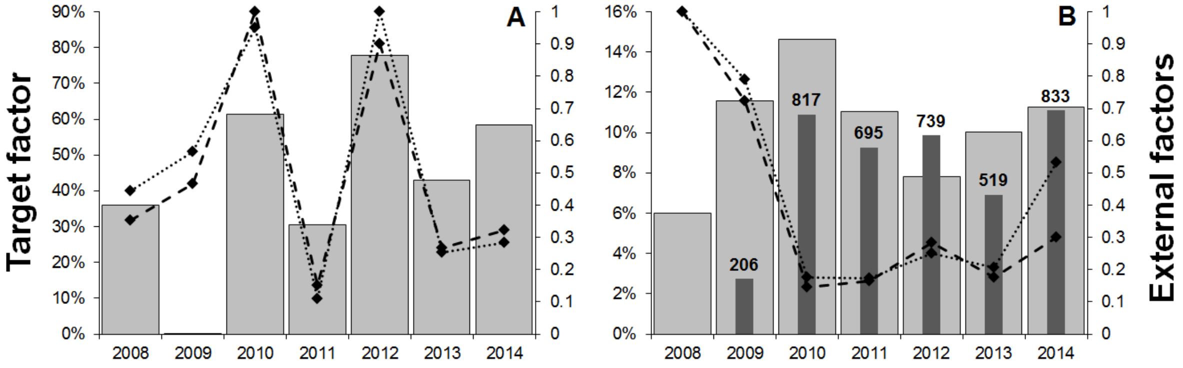

Figure 3.6. Relative A) cuttlefish and B) cod targeting factors in October, over the period 2008-2014 (light grey bars), compared to their relative abundances (dotted lines), relative economic attractiveness's (abundance x price; dashed lines) and remaining French quota in tons for cod (dark grey bars).

3.4 Discussion

3.4.1 An improved quantification of fishing pressure

Adjusting nominal fishing effort using the species-specific LIC improved our understanding of seasonal fishing pressure (here measured by relative landings and species targeting expressed by fishers) exerted by French otter trawlers on EEC cuttlefish and red mullet. These results have direct operational implications, as such effective fishing effort could be used to remove the seasonal effect in catch rates series used to calibrate cuttlefish and red mullet stock assessments. Such an improvement in the relationship between seasonal fishing pressure and fishing effort could not be observed for the other species under investigation, and particularly cod and whiting, which used to be traditional target species for French otter trawlers. Several reasons could explain a lack of improvement (or even a deterioration) in the relationship between fishing effort and estimated fishing pressure: i) high discards rate, which is not accounted for in landings data, ii) high spatial patchiness for some species, which increases landings variability, iii) high monthly fluctuation in biomass, which is not taken into account in landings data (e.g. migration from or to the EEC) and finally, iv) limited spatial coverage of abundance indices derived from fisheries-dependent OBSMER data (Bourdaud *et al*., 2017).

In their study, Sagarese *et al*. (2015) also quantified the overlap between fish distribution from survey data and fishing effort, in order to quantify the availability of spiny dogfish to sink gillnetters and otter trawlers. However, their approach was designed in a binary fashion (i.e. presence/absence), compared to ours, as they compared the number of cells with fishing effort and the number of cells with presence of spiny dogfish *Squalus acanthias*. Note that we assumed here a linear relationship between fishing pressure and our LIC spatial overlap index. Such a linear relationship is, however, a first proxy, and more work could be dedicated to finding either refined spatial overlap indices, or more realistic relationships relating the LIC to the real fishing pressure exerted on the different fish species.

Previous studies have been able to quantify other impacting factors on catchability, such as technical effects (Rijnsdorp *et al.*, 2006; Marchal *et al.*, 2007; Mahévas *et al*., 2011), individual vessel effects (Tidd, 2013; Thorson and Ward, 2014) or vessel competition (Gillis and van der Lee, 2012). The effects of technological creep could in principal enhance the perception of fishing pressure we obtained. In our case study, technological differences among vessels and among years are, however, expected to be relatively limited, as we only focus on one single fleet category, the French OTB >= 18m, and on a relatively short period of time (seven years).

3.4.2 Fishers' intentions and the IFD

The IFD theory builds on several key assumptions: i) interference competition among vessels exists in proportion to their local density, ii) fishers have equal competitive abilities, iii) no restrictions exist for effort allocation and iv) ideal knowledge of fishing grounds' local density (Gillis, 2003). We consider in this study that a poor spatial overlap between the distributions of fishing effort and of available wealth results from one or several of IFD assumptions being at fault. Deviations from IFD predictions are then related to factors that could potentially compromise the validity of these base assumptions. In doing so, we particularly considered assumption (iii), since additionally to external economic factors such as fuel costs (Poos *et al*., 2010; 2013) or spatial competition, fishing access to several of the EEC species being investigated (and hence effort allocation) has been restricted by Total Allowable Catches, direct effort (number of days at sea) limits, and minimum mesh size regulations. This is particularly true for cod, for which a recovery plan has been implemented since 2002 in the North Sea and the EEC. Departs from assumptions (ii) and (iv) are considered more limited, since we consider vessels belonging to one fleet category and no individual quotas are presently set for these boats (ii), and because the EEC is a small and shallow maritime domain, so we can reasonably assume that fishers have a good knowledge of their fishing grounds (iv). Although the legitimacy of assumption (i) is difficult to evaluate, previous studies did evidence that interference competition occurs between EEC fishing fleets (Girardin *et al.*, 2015; Tidd *et al.*, 2015).

Mixed fisheries in the Eastern English Channel target an assemblage of different species (Marchal, 2008, Girardin *et al.*, 2015, ICES, 2017), and our study proposed a novel approach, building on the optimization of a spatial species abundance / fishing effort overlap metric, to identify their key targets, and hence fishers' intentions. This approach was applied only in October as it required a good spatial coverage of both fishing effort and species distributions. Although Quirijns *et al.* (2008) also determined an explicit index for the targeting behaviour in a mixed fisheries context involving two species (i.e. sole and plaice in the North Sea), our approach is different as it explores fishers' intentions using fishery-independent data, and in an optimization fashion.

Our results evidenced that cuttlefish and red mullet have been the primary target species of the French EEC bottom trawlers over the period 2008-2014, which confirmed the strong fishing pressure exerted on both species in October (Figures 3.2 and 3.3). It is informative that cuttlefish and red mullet, the catch of which is not limited by quotas, are much more targeted than cod, whiting and mackerel, three species managed by TAC (Total Allowable Catches). This could result from an adaptation of fishers to increasingly restrictive TAC limitations, and more particularly in the context of the North Sea recovery plan (Horwood *et al.*, 2006), thereby confirming the decline of traditional targets and the emergence of valuable and poorly regulated species such as red mullet (Mahé *et al.*, 2005) and cuttlefish (Gras *et al.*, 2014). Concerning cuttlefish this can also be an adaptation to a gain in economic attractiveness during the same period (Figure S3.3).

It is noteworthy that mackerel has a significant target factor value every year in October. This could be seen as a surprise, as pelagic species such as mackerel are not usually targeted by bottom trawlers. This could be due to the nature of the EEC, a shallow sea (< 50m), with strong mixing and benthic-pelagic coupling processes (Giraldo *et al.*, 2017). The substantial mackerel targeting contrasts, however, with the weak (and even negative) effect of the LIC on the computed fishing pressure exerted on this species. This contrast may be explained by the larger intra- and inter-annual abundance fluctuations pelagic species are subject to, compared to the other species we considered.

The optimized spatial overlap between the distributions of fishing effort and the combined-species resource was achieved with species-specific weightings differing substantially from the available revenue coefficients used to derive VPUE as the aggregated resource metric. This difference measures the deviation between the actual spatial distribution of fishing effort and the one predicted under the IFD. In previous studies, the IFD provided a useful conceptual framework to predict fishing effort distribution patterns (e.g. Gillis and Frank, 2001; Swain and Wade, 2003). In several studies, however, the IFD did not predict fishing effort distribution well, which was interpreted as limited knowledge of fishing grounds, or external foraging constraints (Pet-Soede *et al*., 2001; Abernethy *et al*., 2007).

In our study, and without excluding other possible causes, we interpret here the deviation between observed and predicted effort patterns as IFD assumption (iii) (unrestricted access to the different EEC fishing grounds) being at fault and this for several reasons. First, while the large trawlers investigated here have the capacity to cover all the EEC, they might limit their visits to the closest fishing grounds to save fuel and time at sea costs. Second, weather and especially wind conditions could be poor in the EEC, and could influence the choice of fishing grounds (Wilen *et al.*, 2002; Respondek *et al*., 2014). Third, fishing habits may be more influential than economic opportunism in choosing fishing grounds (Salas and Gaertner, 2004; Holland, 2008; Girardin *et al.*, 2017), although these may be highly correlated (Van Putten *et al*., 2012). Fourth, the EEC is a particularly congested sea, where fisheries may compete for space with other fisheries, or other maritime activities (e.g., shipping, aggregate extractions), which could occasionally restrict their activities (Girardin *et al.*, 2015; Tidd *et al.*, 2015).

Finally, management is an obvious cause of restricted access to fishing grounds. This has been evidenced extensively in the case of Marine Protected Areas (e.g. Stelzenmüller *et al*., 2008; Dowling *et al*., 2012), although the fleet investigated in our study is only subject to limited spatial management measures within the 12 nautical miles coastal areas (EC, 1998). TAC management may also affect the spatial distribution of fishing effort (Batsleer *et al.*, 2013; Baudron and Fernandes, 2014), particularly when the TAC for a species is so low that this species becomes a choke species. This is an issue that we have investigated more thoroughly here, as cod has become a choke species in the EEC following the 2002 implementation of the North Sea cod recovery plan (Horwood *et al.*, 2006), with an impact on the spatial distribution of EEC bottom trawlers and their cod targeting.

3.4.3 Influence of external factors on species targeting fluctuations

The interpretation of cod targeting fluctuations is not straightforward. Thus, it seems at first glance difficult to capture why cod targeting increases over 2008-2010, while stock abundance reflected by CGFS decreases during the same time period. The rationale underlying these contrasted trends becomes, however, clearer when one considers the drastic increase in the unutilized cod quota, from 0 tons in 2008 to 817 tons available in October 2010. With cod quota becoming somehow less restrictive, it is not surprising that cod targeting increased somewhat. The 2011-2014 fluctuations in cod targeting, and the drop observed in 2012, are difficult to explain without considering the other species' targeting factors. Thus cuttlefish targeting, not restricted by quotas, varied synchronously with economic attractiveness, over 2008-2014, with a 2012 maximum corresponding to the sharp decrease in cod targeting concomitantly with a high economic attractiveness for cuttlefish during that year. Another illustration of the combined-species targeting complexity is the decline of red mullet targeting between 2008-2009 and 2010-2014. This could be due to increased spatial and market competition with Dutch fly-shooters, which targeted red mullet in the EEC from 2010 onwards (Marchal *et al*., 2014). The low red mullet targeting observed in 2012-2013 could also be related to the low abundance and economic attractiveness for this species during that year (Figure S3.4).

Future work could be dedicated to identifying groups of fishers according to their targeting patterns, leading to a more precise definition of métiers, and also to evaluate whether habits could be detected in these patterns. We also made a number of simplifications, which could be revisited. Thus, we neglected fishers' home harbour, although this has implications on travel costs, fishing grounds location, and hence the validity of IFD-based effort predictions (Gordon, 1953; see also Gillis, 2003 for a review). Furthermore, in combination with spatio-temporal distributions of species abundance and fish prices fluctuations, geographical features can induce traditional fishing patterns only revealed by fishers' interviews (Christensen and Raakjær, 2006; Boonstra and Hentati-Sundberg, 2014).

The method developed in this study is not aimed at forecasting fishers' intentions, as past choices are not causal (Van Putten *et al*., 2013). However, it could be included in individual-based models (IBM), which are considered particularly well-adapted for forecasting, especially in changing management regimes (Ulrich *et al*., 2012; Van Putten *et al*., 2012). Our approach could thus be combined to a number of existing integrated ecological-economic fisheries models (see Nielsen *et al*., 2017 for a review), by supplying knowledge on real fishers' intentions, which may contrast with preliminary modelling assumptions and choices.

A future development of this study could also be to consider extensions from the IFD conceptual framework, such as isodars (for 'iso-Darwin'; Morris, 1988, 2003). Isodars build on an ecological theory, predicting numbers in one area knowing numbers in another area and explicit expressions of local density-dependent per capita fitness. Isodars have been applied to fleet dynamics by Gillis and van der Lee (2012) and even proved to predict observations better than discrete choice models (van der Lee *et al.*, 2014). If determination of the nature of factors in isodars may not be easily interpretable, a challenge could be to develop the approach at a more disaggregated level (e.g. by home port) so to, (i) gain better knowledge of the basic desirability level of the different fishing areas at fine scale, in relation to associated operational costs and tradition aspects (see Figure S3.5) and, (ii) improve the estimation of species target factors, by including those area desirability factors identified in (i).

3.5 Conclusion

This study used spatial distributions collocations to improve the definition of fishing effort and our understanding of its determinism. Our results at seasonal scale emphasized the importance of cuttlefish and red mullet in determining the global distribution of Eastern English Channel bottom trawlers. These results have clear management benefits, in improving the definition of catchability, effective fishing effort, and how these relate to fishing mortality for red mullet and cuttlefish. We also used a metric measuring the optimized spatial overlap between fishing effort and combined-species abundances. It revealed the importance of cuttlefish, red mullet and, to some extent, mackerel targeting relative to the other species in October, which was in contrast with IFD predictions, probably owing to external factors including limiting quota, travelling costs, or competition with other sectors of activity. Our results could be validated by available fishers' knowledge (e.g. Neis *et al.*, 1999; McCluskey and Lewison, 2008; Hind, 2015).

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Chapter 4

Impact of a landing obligation on coupled dynamics ecosystem-fishers: individual-based modelling approach applied to Eastern English Channel

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Abstract

Within the new EU Common Fisheries Policy a landing obligation is gradually being implemented since 2015. This landing obligation aims at improving the size selectivity of fisheries by reducing the amounts of small fish in catches. However, the short- and long-term consequences of the landing obligation are largely unknown. Models could be used to explore the probable evolution of fishers' behaviour and subsequent knock-on effects on ecosystem structure, under several scenarios of the policy implementation. In the eastern English Channel, where many stocks are caught in mixed fisheries, the landing obligation is expected to have important impacts on the allocation of fishing effort. Low quota availability of choke species potentially constrains the quota uptake and harvest of fisher's target species. By integrating a fleet-dynamics model (DSVM) within the multi-species trophic model OSMOSE, an end-to-end model was developed and applied to the eastern English Channel ecosystem and fisheries to explore the future effects of this new policy. This mechanistic individual-based model focused on the bottom-trawl French fishers, catching the demersal fish community including two quota species, cod and whiting. The impacts of "Business as usual" and landing obligation scenarios are compared. First, the effects on the spatialization of fishing effort and fishers' revenue are addressed. Then the effects of the landing obligation on trophically-interacting species is evaluated using whole-ecosystem indicators. Results show that this new management policy would have short-term negative effects on fishers' revenue, induced by a large reallocation of their fishing effort to avoid quota over-shooting. By protecting quota-regulated species, global revenue can be profitable for fishers in the medium-term. However, the landing obligation induced an increase of the predatory pressure operated by cod and whiting on the other species, which does not improve the overall ecosystem health. Ultimately these results are based on the assumption of a full compliance by fishers but would depend on how efficient the enforcement of the landing obligation is.

Keywords: landing obligation; individual-based model, fleet dynamics; Eastern English Channel.

4.1 Introduction

Mixed fisheries harvest several species (Ulrich *et al*., 2012; ICES, 2017), composed of target species and bycatches (Wilson and Jacobsen, 2009). This particularity is mainly due to weak selectivity abilities of the gear used, and especially demersal ones (Marchal, 2008). A portion of bycatches is unwanted, and therefore discarded, for different reasons: undersized individuals, over-quota catches, absence of market channel, vessel's capacity, regulations on catch composition *etc.* (FAO, 1996; Catchpole *et al*., 2005; Feekings *et al.*, 2015).

This high volume of discarded fish is considered problematic for several reasons, both ecologically and from a management perspective. Indeed, as a major part of the discards is released dead (Broadhurst *et al*., 2006; Benoît *et al*., 2012), it represents a waste of resources for human consumption and might disturb the functioning of the ecosystem. Furthermore, if discards are not subject to regulatory processes and monitoring they can cause difficulties in the stock assessment process and pending management procedures. Not accounting discards in landings statistics can induce fishing mortality underestimations and management inefficiency, especially when Total Allowable Catches (TAC) are the primary management tool (Cotter *et al.*, 2004; Rätz *et al.*, 2007; Sardà *et al*., 2015).

In response to this situation, a landing obligation (or discard ban) is progressively being implemented in the European Union (EU), whitin the new Common Fisheries Policy, to promote more selective fisheries (EC, 2013; 2015; 2016; Borges, 2015). In the Northeast Atlantic, the landing obligation will apply to each quota regulated species, with exemptions for species, which are proved to survive after being discarded (e.g. *Nephrops norvegicus*; EC, 2016).

Fishers will be impacted by the landing obligation if they do not adapt their behaviour (Condie *et al*., 2014). Within the EU landing obligation, fishers reaching a quota for a species will have to stop immediately their activity if they have any chance to fish it during a trip. Additionally, they will have to sort and keep unwanted catches for further processing, which represents a non-negligible loss of time, and therefore revenue, but possibly also an increase of physical effort (Balazuc *et al*., 2016).

Fishers are expected to respond by two main manners to the landing obligation in order to limit their unwanted catches and losses of revenue: i) to use more efficient gears and thus limit the take of undersized or undesirable fish (technical response) and ii) to avoid areas known for their high discards rates depending on the time of the year (tactical response; Reid, 2016).

Spatio-temporal avoidance can already be observed without a landing obligation, as some fishers try to not have a volume/revenue ratio of landings too high (Holland, 2008; de Vos *et al*., 2016) and already try to adapt their activities according to unwanted species (Reid, 2016), such as choke species, the quota of which has been exceeded and hence cannot be landed anymore (Schrope, 2010; Ulrich *et al.*, 2011; Baudron and Fernandes, 2014).

Fishers' response must be cautiously examined to anticipate any undesirable effects of the landing obligation (Fulton *et al*., 2011). Fishers' future reaction can be partially anticipated by direct interviews, but this method would probably suffer from a poor anticipation of the future effects of this regulation by fishers (e.g. Fitzpatrick and Nielsen, 2016; Reid, 2016; Fitzpatrick *et al.*, 2017) and also from the limited number of fishers being sampled. Fleet dynamics modelling can provide answers to forecasting short-term fishers' behaviour (see Van Putten *et al*., 2012 for a review). One of the hypotheses used to model fishers' behaviour is that they will aim at maximizing their global revenue (Gordon, 1953). In a landing obligation context, they will thus try to reach the highest revenue possible without exceeding the quotas of concerned species and keep fishing as long as they can throughout the year. Some studies tried to predict the impact of a discard ban on fishers' revenue (e.g. Condie *et al*., 2013) or ecosystems (e.g. Heath *et al*., 2014) with very simple assumptions on the fishers' reaction to the ban. On the other side, the reaction of fishers to a discard ban has been predicted by Batsleer *et al*. (2013), but they worked with the assumption that availability of fish in an area was independent of the fishing pressure, thus ignoring any ecological impact of fishers' activity. Hence to our knowledge, no study tried to estimate the medium-term effects of a discard ban by modelling the interaction between fishers and ecosystem, and their co-evolution in a landing obligation context.

This study addresses the question by coupling a tropho-dynamic multispecies model, OSMOSE (Object-oriented Simulator of Marine Ecosystems Exploitation; Shin and Cury, 2001; 2004), and a fleet dynamic model, the Dynamic State Variable Model (DSVM; Houston and McNamara, 1999; Clark and Mangel, 2000) particularly adapted to forecast new management regimes (van Putten *et al*., 2012), and investigate the long-term changes on ecosystem and fisheries dynamics in the Eastern English Channel (EEC). To achieve this, a focus was made on exclusive French bottom otter trawlers (OTB) longer than or equal to 18m. This fleet will be highly impacted by the landing obligation due to their high species diversity found in catches (Marchal, 2008; ICES, 2017), and it is also one of the main fleets in the EEC (Carpentier *et al*., 2009). In this study we will only examine the spatio-temporal avoidance and not gear or selectivity improvements. The performances of current management regime and of the landing obligation on ecosystem health and fishers are evaluated and compared using different indicators such as species' biomasses, size spectra and fishers' revenue.

4.2 Material and methods

4.2.1 Eastern English Channel

The Eastern English Channel (ICES Division 27.7d) is a shallow sea delineated by latitudes 49.3°N and 51°N and longitudes 2°W and 2°E. It is subject to intense and diversified human activities including fishing, maritime traffic, wind farms or aggregate extraction, and it constitutes a corridor between the northeast Atlantic Ocean and the North Sea (Ulrich *et al.*, 2002; Dauvin, 2012). It is heavily exploited by fishing since the 20th century (Molfese *et al.*, 2014) and it is characterized by important mixed fisheries. This area is home to several commercially important species: sole (*Solea solea*), cod (*Gadus morhua*), whiting (*Merlangius merlangus*), scallops (*Pecten maximus*), plaice (*Pleuronectes platessa*), European seabass (*Dicentrarchus labrax*), red mullet (*Mullus surmuletus*), herring (*Clupea harengus*), mackerel (*Scomber scombrus*), cuttlefish (*Sepia officinalis*) and squids (*Loligo vulgaris* and *Loligo forbesii*).

In 2017, sole and whiting were the only species concerned by the discard ban in 27.7d (EC, 2016), but other quota species like cod or plaice should soon complete the list if they do not obtain a survivability exemption.

4.2.2 Biological simulation model: OSMOSE for EEC

OSMOSE is a multispecies, spatially-explicit, individual-based model. It models a community of fish species (including cephalopods), explicitly representing their full life cycle, from eggs to adults. Each species is decomposed in super-individuals, corresponding to groups of individuals with common characteristics, e.g., species, size, age, spatial location and trophic level.

OSMOSE is a tropho-dynamic model, meaning that the fish community fluctuations are led by predation and competition. Predation in OSMOSE is opportunistic and based on two key prerequisites: spatio-temporal co-occurrence and size suitability of a prey for a predator, without a priori diet setting. OSMOSE was previously applied to several type of ecosystems (e.g. Marzloff *et al.*, 2009; Travers-Trolet *et al.*, 2014; Halouani *et al.*, 2016; Grüss *et al.*, 2016), including the EEC (Travers-Trolet *et al.*, *in prep.*). For the current study, the existing EEC configuration (Travers-Trolet *et al*., *in prep.*) had to be adapted before being coupling with a fleet dynamics module. The original model is composed of fourteen species representing the major part of the biomass (75% of the biomass sampled by the EEC Channel Ground Fish Survey over 2000-2009) and catches (90% of fish catches in 27.7d over 2000-2009): lesser-spotted dogfish (*Scyliorhinus canicula*), red mullet, pouting (*Trisopterus luscus*), poor cod (*Trisopterus minutus*), cod, mackerel, horse mackerel (*Trachurus trachurus*), sole, plaice, whiting, dragonets (*Callionymus lyra*), sardine (*Sardina pilchardus*), herring and squids. For the current study, cuttlefish is added because it has recently become one of the main commercial species in EEC (Gras *et al*., 2014), especially for the bottom otter trawlers investigated here (Bourdaud *et al*., submitted). Furthermore, the model has been updated to represent the average state of the EEC ecosystem over 2008-2015 (rather than 2000-2009 as Travers-Trolet *et al*., *in prep*), in order to be the closest to the ecosystem state on which the European landing obligation is implemented.

Every time step (here two weeks), each super-individual can move into a square of 0.1° x 0.1° in the 2-dimensional grid representing EEC (Figure 4.1). They can remain in the same cell, but a random movement is also allowed in every adjacent cell according to two constraints: i) this cell is available for the super-individual in the input distribution maps provided for different species, ages and seasons (Figure S4.1) and ii) the distribution map remains the same between two consecutive time steps. If the latter condition is not fulfilled, the super-individual will be randomly distributed in the new distribution map. Here the presence-absence maps used by Travers-Trolet *et al.* (in prep.) were updated to density maps (Figure S4.1), based on monthly spatio-temporal distribution maps obtained by Bourdaud *et al*. (2017).

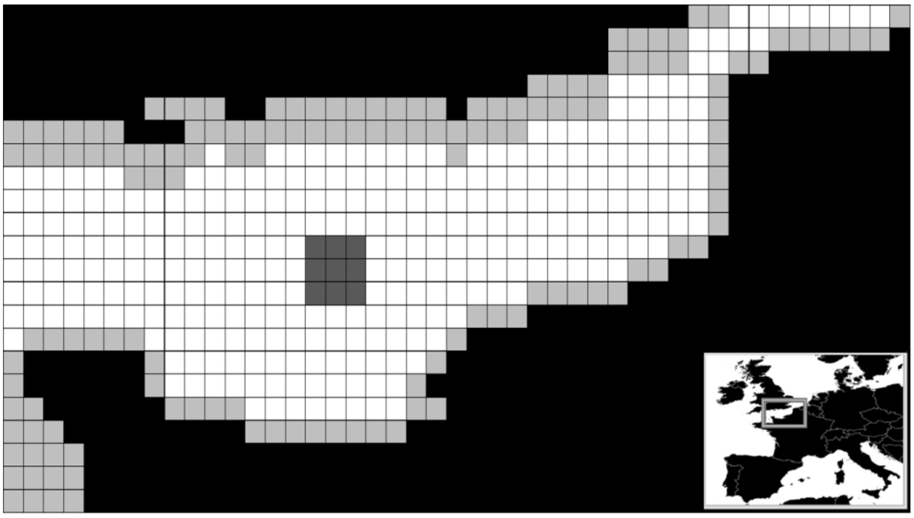

Figure 4.1. Grid of 0.1° x 0.1° cells representing Eastern English Channel in OSMOSE. The size of a fishing ground (dark grey) corresponds to 9 cells. The prohibited areas for the modelled fleet corresponding to the 12-miles limit along United Kingdom and 3-miles limit along the French coast is also indicated (light greys).

After a movement has occurred at the beginning of the time step, super-individuals interact between themselves and with their local environment following different processes (Figure 4.2).

Different mortalities can affect a super-individual: predation by other super-individuals, starvation, fishing, other natural mortalities (i.e. due to diseases or non-modelled organisms such as other fish or mammals) and even an approximation of the senescence if the super-individual exceeds the maximum longevity for the species. On the other side, a super-individual can predate other super-individuals but also plankton and/or benthic invertebrate groups.

Lower trophic levels (LTL) are not explicitly modelled in OSMOSE but forced as biomass prey fields. Phytoplankton (i.e. dinoflagellates and diatoms), microzoo- and mesozooplankton fields are provided by an ECO-MARS-3D biogeochemical model, with a grid of 2km horizontal resolution and 10 vertical layers. The plankton fields used represent the average state of the EEC pelagic ecosystem from 2000 to 2006 and are vertically-integrated before being added to OSMOSE. Additionally, homogeneously spatial distributions of six LTL are joined to complete the available food sources for fish: macrozooplankton and five different size classes of benthic invertebrates.

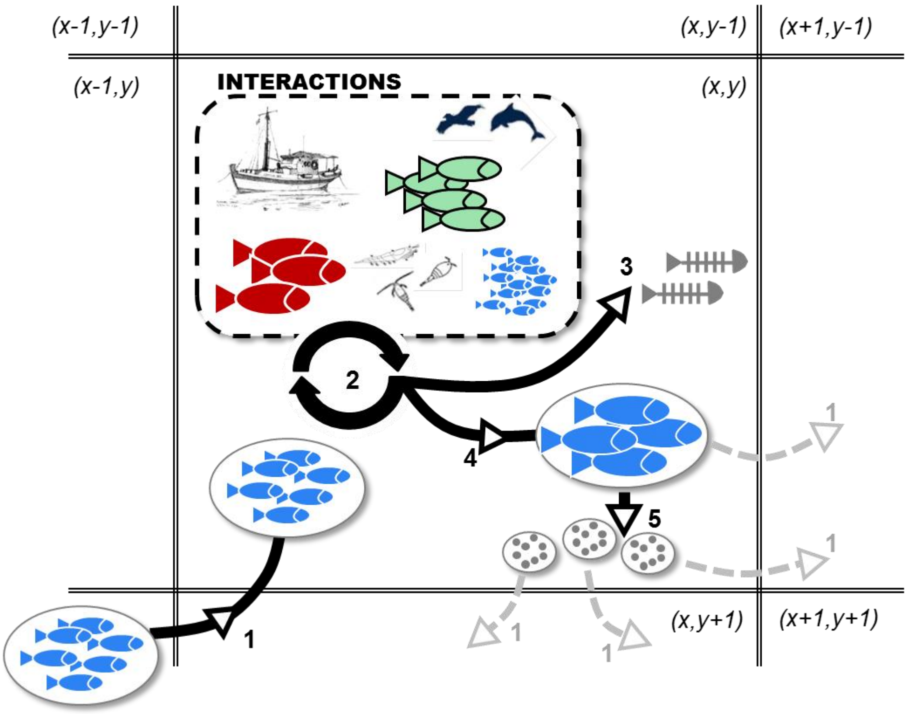

Figure 4.2. Processes undertaken by a super-individual during a 2-weeks time step : 1) movement of individual in a 2D grid either driven by input maps or due to random walk; 2) interactions and mortalities (3): explicit predation upon other super-individuals present in the same cell and upon LTL groups with associated predation for the prey groups, as well as fishing mortality and additional natural mortality; 4) growth and 5) reproduction which creates new super-individuals “eggs” for the next time step (source, Travers-Trolet *et al*., in prep.).

As the sequential order at which the different types of mortalities operate in the model is considered to have repercussions on its dynamics and outputs, each time step is divided in subdivisions further named *subdt* (here set to 10), in which the order of super individuals and mortalities are randomised.

All predation interactions respect the fundamental assumptions of OSMOSE: spatio-temporal co-occurrence and size suitability. However, as the model is only 2-dimensioned, accessibility coefficients can limit the vertical overlap of the pelagic and benthic compartments, but can also refine the predator-preys interactions if known morphological constraints exist between species. If the total biomass of prey accessible to a super-individual *i* during a *subdt* is greater than its maximum food requirement *Ing* (equal to the biomass of *i* times its maximum ingestion rate, here set to 3.5 g of food per g of individual and per year; Shin and Cury, 2001; 2004), the total biomass of each prey *j* predated by *i*, $Bpred_{i,j,subdt}$, varies between 0 if no suitable prey is available and *Ing*, and is proportional to the available biomass of each prey *j*:

$$Bpred_{i,j,subdt} = \frac{Baccess_{i,j,subdt}}{\sum_j Baccess_{i,j,subdt}} \times Ing_{i,subdt} \tag{4.1}$$

Otherwise if the total biomass of prey accessible to *i* is lower than *Ing*, $Bpred_{i,j,subdt}$ is equal to $Baccess_{i,j,subdt}$, obtained by the product of the biomass of *j* and the accessibility coefficient. For LTL prey, this accessibility coefficient representing vertical overlap is completed by an additional parameter (obtained by calibration) in order to take into account the portion of LTL biomass truly available to fish.

Despite involving mortalities for other super-individuals and LTL, the predation process has also repercussions on the super-individual itself, beginning with the starvation mortality evoked above. The predation efficiency ξ represents the amount of food eaten compared to the maximum edible food. For a given super-individual, if ξ falls below a threshold $\xi crit$ representing maintenance requirement at time step *t*-1, a starvation mortality M_ξ inversely proportional to ξ is computed:

$$M_{\xi_i} = M_{\xi max_i} \times \left(\frac{\xi_{i,t-1}}{\xi crit_i}\right) \tag{4.2}$$

In OSMOSE the different mortality rates M_x (i.e. starvation M_ξ, fishing *F* and other sources M_{oth}, see Table S4.1 for their values) are applied using the same formula:

$$N_{i,t+\Delta t} = N_{i,t} \times e^{-\Delta t \times M_x} \text{ with } M_x \in \{M_\xi, F, M_{oth}\} \tag{4.3}$$

On the contrary, if predation efficiency is higher than maintenance requirements, it can shape the growth of the super-individuals, i.e. the more a fish eat close to satiety and the more it will be able to grow. The length gain is thus depending on the predation efficiency and the average gain ΔL at a given super-individual's age according to the von Bertalanffy growth curve:

$$\begin{cases} \Delta L_{i,t} = 0 \quad if\ \xi i < \xi crit \\ \Delta L_{i,t} = \frac{1.5\Delta L}{1-\xi crit}(\xi i - \xi crit) \quad if\ \xi i > \xi crit \end{cases} \tag{4.4}$$

The maximum size reachable was limited to 1.5 times the asymptotic length (conversely to 2 in previous OSMOSE application) in order to better reflect the variability observed in length-at-age data collected in the EEC (www.fishbase.org).

Finally the reproduction process operates before the end of the time step. The production of new super-individuals (i.e. eggs) depends on the spawning stock biomass (computed with a sex ratio of 1:1 and from fish older than the age of maturity A_{mat}), the relative fecundity Φ of a species s and the seasonality of spawning γ (Figure S4.2):

$$N_{0,s,t} = \Phi_s \times \gamma_{s,t} \times \frac{1}{2} \sum_{a>A_{mat}} B_{a,t} \qquad (4.5)$$

In the EEC version of OSMOSE fishing pressure is implemented by fishing mortality rates. A species-specific fishing mortality rate is used, annual for twelve of the species and seasonally-varying for horse mackerel and squids, now completed by cuttlefish (Figure S4.3). Fishing is applied to recruited individuals, previously related to the age at recruitment but now converted in size to be more realistic (Table 4.1). The model is adapted to forecast the impact of some fisheries management measures (e.g. Marine Protected Areas with the use of spatial fishing mortality rates), but due to the absence of explicit discards and fleet dynamics in previous versions of the model, some changes had to be done to evaluate the effect of the landing obligation on French OTB. These improvements of the exploitation module of the OSMOSE model are done by coupling a fleet-dynamics model, DSVM (Dynamic State Variable Model), with the current OSMOSE model. Both models will exchange information: several inputs on fish dynamics will be given by OSMOSE to run DSVM, while outputs from DSVM will serve as inputs in the fishing mortality conducted in OSMOSE. For this reason, the DSVM model was directly coded in OSMOSE to facilitate the coupling using the object-oriented language Java (JdK 1.8.0_71).

During a simulation the fishing mortality applied to each species in OSMOSE is decomposed in two distinct parameters: F_{DSVM} and F_{others}. These two parameters allow for a decomposition of the total fishing mortalities in a fraction representing fishing mortality attributed to the simulated fleet (F_{DSVM}) and another fishing mortality (F_{others}) corresponding to the other French fleets and all other countries impacting these stocks. F_{others} is applied for each species as a proportion of the initial fishing mortality rate, while the computation of F_{DSVM} will be further presented. The proportion of the described fishing mortality over the total fishing mortality is obtained by the historical landings on average in the 2008-2015 period and set as a constant during the simulation (Table 4.1).

Table 4.1. Recruitment size, Minimum Landing Size (MLS), presence or absence of Total Allowable Catches (TAC), catchability (from calibration) and proportion of 2008-2015 landings due to exclusive bottom otter trawler for EEC species in OSMOSE.

	Recruitment size (cm)	Under TAC in the EEC	MLS (cm)	Catchability	Exclusive French OTB proportion of landings (%)
Lesser-spotted dogfish	39	NO	/	0.0140	59.9
Red mullet	14	NO	/	0.1555	43.6
Pouting	21	NO	/	0.0398	67.8
Whiting	21	YES	27	0.1359	79.2
Poor cod	/	NO	/	0	/
Cod	21	YES	35	0.0769	55.7
Dragonet	/	NO	/	0	/
Sole	22	YES	24	0.0004	0.9
Plaice	25	YES	27	0.0178	23.6
Horse mackerel	12	YES	15	0.0254	38.2
Mackerel	18	YES	20	0.0229	31.5
Herring	16	YES	20	0.0137	11.8
Sardine	10	YES	11	0.0485	12.2
Squids	14*	NO	/	0.0129	76.3
Cuttlefish	13*	NO	/	0.1108	40.1

*mantle length

4.2.3 Fisheries simulation model

DSVM individual-based model was first built to predict animal's behaviour but was further adapted and used to predict short-term fishers' behaviour (e.g. Gillis *et al.*, 1995; Poos *et al.*, 2010; Batsleer *et al.*, 2013). The model developed here is mostly derived from models developed by Poos *et al.* (2010) and Batsleer *et al.* (2013), in which individual fishers have a set of choices to select simultaneously at each time step: i) to go out fishing in a particular fishing ground or to stay in their home port, and ii) to discard or not a part of their catches in order to maximize their profit at the end of the year.

French large OTB are also *a priori* easier to model than smaller vessels because their activity covers a wider part of the EEC, more in adequacy with spatial species distributions represented in the OSMOSE model than coastal fleets. Fishers belong either to Boulogne-sur-Mer or Port-en-Bessin, the two main ports for the modelled fleet (i.e. more than 73% of French OTB fishing effort in hours in EEC in 2008-2015). The number of vessels explicitly modelled is set to 66, based on the average from the period 2008-2015. All vessels not belonging to one of these ports (i.e. 19 vessels) were equally spread in these two, for a total of 43 in Boulogne-sur-Mer and 23 in Port-en-Bessin.

A fishing ground is defined as an area of 0.3° x 0.3° (~700km²), corresponding to 9 OSMOSE cells, which approximately represents the mean observed dispersion of an OTB fishing operation during a fishing trip. French OTB activity is limited for the 12-miles area along United Kingdom coasts (~ 2 OSMOSE cells), and 3-miles along the French coast (~ 1 OSMOSE cell), prohibited for the majority of French trawlers' activity (Girardin *et al.*, 2015).These definitions lead to the consideration of 46 fishing grounds in the EEC (Figure 4.1).

Mechanisms driving fishers' choices

Fishers are expected to make their choices of fishing ground and discard strategy in order to maximize their revenue, but under the constraint of availability. In the model, quotas are updated only every five years to limit the computation time, before updating the evaluation of all fishers' possible choices, for two species: cod and whiting. Also for computation reasons, only these two species were considered to be regulated by quota here, as they are the most important species under TAC for French OTB in EEC, and don't have any survivability exemption (EC, 2016). The other 13 species considered in the model are not constrained by quota limitation, even if some of them are currently TAC species (see Table 4.1).

Cod quota computation is based on the Harvest Control Rules (HCR) from EC 1342/2008. *Fquota* is related to cod Spawning Stock Biomass and it is transformed to finally produce an estimated catch value which is used to form the quota. *Fquota* is computed as below:

$$\begin{cases} Fquota = F_{up} \text{ if } SSB > Bpa \\ Fquota = F_{up} - \left(F_{low} \times \frac{(Bpa-SSB)}{(Bpa-Blim)}\right) \text{ if } Blim > SSB <= Bpa \\ Fquota = F_{low} \text{ if } SSB <= Blim \end{cases} \tag{4.6}$$

However the current and limits values of F and Spawning Stock Biomass (SSB) from stock assessment cannot be input directly within OSMOSE due to the difference between observed and modelled values after the calibration process, and therefore were scaled by using ratios between limits and current F, and limits and current SSB from stock assessment averaged over 2008-2015, applied to F and SSB values in OSMOSE.

For whiting the quota is computed using HCR presented in ICES report (2017):

$$\begin{cases} Fquota = F_{MSY} \text{ if } SSB \geq MSY\ B_{trigger} \\ Fquota = F_{MSY} \times \frac{SSB}{MSY\ B_{trigger}} \text{ if } SSB < MSY\ B_{trigger} \end{cases} \tag{4.7}$$

The target F is finally divided into F_{DSVM} and F_{others}. In F_{DSVM}, the quota *Quota* in tons is computed by:

$$Quota = Brecru \times (1 - e^{-F_{DSVM}}) \tag{4.8}$$

where *Brecru* is the mean annual biomass of the quota-regulated species.

Finally *Quota* is divided into the number of vessels to obtain individual quotas *q* that will be used for the computation of optimal choices.

Fishers' choices at each time step are evaluated in order to maximize their annual realized net revenue φ, defined as the total quantity landed L (kg) of each species s weighted by each species price p_s (€ per kg) plus additional revenues R and minus variable fishing costs *Cost*, a fine Q for overshooting his individual quota and a fine O for discarding fish if existing.

$$\varphi(L, Q) = \sum_s \left[\sum_t \left(L_{s,t} \times p_{s,t}\right) - (Q_s + O_s)\right] + \sum_t (R_t - Cost_t) \tag{4.9}$$

During the simulations, the variations of the fines for discarding will ultimately affect O for fishers.

Computation of total revenue

Total landings L are the sum of the landings realized at each time step, according to the local conditions encountered within OSMOSE (i.e. the fish super-individuals present in the fishing ground). The fishing operation takes into account different components: the abundance A_i of each super-individual i, the catchability cat_s for the species s (estimated by calibration), the selectivity sel_l according to the length l of the super-individual, the time dedicated to fishing activity $Tfish_{g,\psi}$ for a vessel coming from a given port ψ to a particular fishing ground g *(see below)*, a fixed seasonal effort $E_{t,\psi}$ (between 0 and 1) representing the effort variability due to non-modelled phenomenon for vessels from each port (e.g. fishers' holidays; Table 4.2), and finally a ratio *rand* randomly selected in a normal distribution with a mean and standard deviation extracted from historical biomass from OSMOSE on the 10 precedent years. *rand* is used for each size class of each species along the time step.

$$C_{i,s,l,g,\psi,t} = A_{i,s,l} \times cat_s \times sel_l \times Tfish_{g,\psi} \times E_{t,\psi} \times rand \tag{4.10}$$

The selectivity ranging from 0% to 100% is applied according to the length of the super-individual using a logistic curve:

$$sel_l = 1/(1 + e^{(-\beta \times (l-\alpha))}) \quad (4.11)$$

Where β and α are two parameters equal to 0.3 and 17.5 (cm) respectively for OTB ≥ 18m in the EEC (Girardin *et al*., 2016).

For F_{others}, fishing mortality is applied upon individuals from a specific recruitment size.

Once total catches are computed, the landings and discards fractions are separated for quota-regulated species according to discarding options. The different choices are either discarding everything, discarding only the undersized fish, discarding only the legal-sized fish, or discarding northing. For species not regulated by quota in the model, all the catches are assumed to be landed.

In the model, species concerned by a Minimum Landing Size (MLS) are decomposed in two size classes, below and equal or above the MLS (Table 4.1), with undersized individuals' price set to 0 € per kg while the price of the larger size-class (or for all individuals for species without MLS) varies temporally according to observations from the EEC during the 2008-2015 period (Table 4.3).

Additional revenues R fishers can obtain are composed by the revenue of non-modelled species, considered to be linearly correlated to the explicit revenue, and a fixed seasonal revenue from areas outside EEC *Ext* (i.e. North Sea and Western English Channel; Table 4.2), based on data analysis over 2008-2015. For each time step, *R* is computed by

$$R_t = \left[\sum_s (L_s p_{s,t})\right] \times \frac{Add_{\psi,t}}{(1 - Add_{\psi,t})} + Ext_{\psi,t} \quad (4.12)$$

Where $Add_{\psi,t}$ is the percentage of revenues in EEC coming from species not explicitly represented in the model for each port ψ and at each time step t (Table 4.2).

Table 4.2. Percentage of catches not explicitly represented by the model, revenues from non-modelled areas and fixed effort ratio in each port of the modelled fleet in Eastern English Channel. Extra Area revenues for Boulogne-sur-Mer's vessels are exclusively coming from the North Sea while these for Port-en-Bessin's vessels are exclusively coming from Western English Channel.

		January	February	March	April	May	June	July	August	September	October	November	December
Extra species Revenue (%)	Boulogne-sur-Mer	16.8	19.9	27.2	34.9	23.1	23.0	21.1	16.4	13.9	17.2	23.4	20.6
	Port-en-Bessin	50.5	50.7	53.6	64.0	70.2	75.5	75.5	66.6	57.8	51.2	44.2	46.9
Extra Area revenue (€)	Boulogne-sur-Mer	122.8	23.8	29.8	210.7	157.4	91.9	89.2	252.0	233.1	404.3	720.6	397.3
	Port-en-Bessin	90.4	135.5	374.1	84.5	42.9	55.1	139.2	200.7	125.7	16.2	13.1	8.9
Fixed effort ratio	Boulogne-sur-Mer	0.93	1.00	0.96	0.66	0.39	0.27	0.32	0.53	0.83	0.83	0.66	0.60
	Port-en-Bessin	0.96	0.55	0.49	0.56	0.60	0.42	0.39	0.45	0.60	0.95	0.96	1.00

Table 4.3. Monthly prices (€ per kg) of the different species for the second size class. For the first size class, price is fixed to 0 €/kg.

	January	February	March	April	May	June	July	August	September	October	November	December
Lesser-spotted dogfish	0.68	0.77	0.74	0.48	0.38	0.32	0.32	0.33	0.39	0.47	0.54	0.49
Red mullet	5.96	7.01	7.65	8.36	7.86	5.39	6.92	6.53	5.36	4.31	4.41	5.15
Pouting	0.44	0.38	0.47	0.59	0.71	0.73	0.60	0.55	0.56	0.61	0.64	0.52
Whiting	1.24	0.98	1.01	1.18	1.14	1.08	1.43	1.38	1.69	1.61	1.65	1.63
Poor cod	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cod	2.94	2.75	3.10	3.32	3.63	3.30	3.24	3.10	3.44	3.47	3.25	3.73
Dragonet	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Sole	11.43	10.35	9.06	9.65	10.69	9.22	10.39	9.70	9.44	10.12	11.94	13.91
Plaice	0.84	0.94	1.10	1.20	1.14	1.08	1.56	1.61	1.52	1.70	1.91	1.01
Horse mackerel	0.43	0.49	0.49	0.27	0.28	0.35	0.43	0.37	0.37	0.34	0.49	0.40
Mackerel	2.00	1.80	1.24	0.97	1.19	1.04	1.13	0.86	1.07	1.28	2.35	2.09
Herring	0.43	0.41	0.35	0.34	0.41	0.44	0.29	0.39	0.65	0.54	0.37	0.34
Sardine	0.76	0.91	1.09	1.22	0.98	2.29	0.85	0.89	0.60	0.50	0.54	0.92
Squids	5.41	7.43	8.64	10.98	10.77	9.40	8.25	7.25	6.23	4.76	4.77	5.81
Cuttlefish	2.65	3.10	2.72	2.33	2.35	2.46	3.03	2.61	2.76	2.73	2.58	2.78

Computation of total cost

A fine Q_s is given for quota over-shooting, which hampers fishers to land over-quota species, and is calculated as in relation to the specific individual quotas q_s and a fine multiplier f_s applied to the price p_s of the species by

$$\begin{cases} Q_s = 0 \text{ if } L_s <= q_s \\ Q_s = (L_s - q_s) \times f_s \times p_s \text{ if } L_s > q_s \end{cases} \tag{4.13}$$

Here f_s is set to 10^9 for cod and whiting in order to prevent fisher to land more fish than they are allowed to, which mimics the final aim of the Landing Obligation.

If a landing obligation is implemented, fishers are penalized if they discard a quota-regulated species. A fine O is thus computed from the quantity of discards D in kg and the fine multiplier o_s applied to the price of the species following

$$O_s = o_s \times D_s \times p_s \tag{4.14}$$

For species not concerned by the landing obligation (i.e. the 13 species without quota or for which the quota is not considered to be restrictive), the multiplier o_s is set to 0 €, otherwise o_s is set to 10^9 in order to represent a potential exit of the métier in the full implementation of the landing obligation.

Costs for fishing are composed of the total fuel cost *Ctrav* of traveling to the fishing ground *g*, and the total fuel cost *Cfish* for the fishing operation *per se*. All these costs are weighted during each time step by the fixed effort coefficient $E_{t,\psi}$.

$$Cos_{g,\psi,t} = \left(Ctrav_{g,\psi} + Cfish_{g,\psi}\right) \times E_{t,\psi} \tag{4.15}$$

If a fisher stays at port during a time step, the cost is null.

We consider here that the further a fishing ground is, and the longer it takes to reach it (higher *Ctrav*) and the lesser fishers have time remaining for fishing in the time step (smaller *Cfish*). Based on the average distance $Dist_{ref}$ between fishing grounds and ports in the EEC (60km estimated from fishing effort data over 2008-2015, and considered as a reference in the following), reference travel time $Ttrav_{ref}$ is estimated to represent 8% of a fishing trip, leaving 92% of the time for effective fishing activity ($Tfish_{ref}$). The costs for travelling during a time step for a vessel to a fishing ground are thus computed as

$$Ctrav_{g,\psi} = \Lambda \times Dist_{g,\psi} \tag{4.16}$$

Where Λ is the cost of traveling per km, comprising the round trip and the different trip per time step, set to 24.3 € per km (estimated from Batsleer *et al.*, 2013). Costs for fishing during a time step are computed as

$$Cfish_{g,\psi} = Cfish_{ref} \times \left(\frac{1-\left(\frac{Dist_{g,\psi}}{Dist_{ref}} \times Ttrav_{ref}\right)}{Tfish_{ref}} \right) \tag{4.17}$$

Where $Cfish_{ref}$ is the cost for the reference fishing time, set to 16740 € (estimated from Batsleer *et al.*, 2013). In equation 4.17, the right part of the product is equivalent to the time ratio of fishing activity $Tfish_{g,\psi}$.

Evaluation of fishers' possible choices through backward calculation

The core of the DSVM is the backward calculation step. It consists of computing all fishing options fishers will have at each moment of the year, and assessing their relative interest according to their cumulated landings of quota-regulated species at this moment of the year. It is based on fishers' revenues, landings and costs perception.

Backward calculations are operated using the value function which represents the maximum expected net revenue between a time step and the end of the year T, expressed as $V(L\sim,q,f,t)$, knowing every possible state of landings $L\sim$ of the two quota-regulated species, their individual quotas q and the associated fines multipliers for quota over-shooting f. The optimal tactic is then computed by proceeding backward from the end of the year, and then linking each possible state of a fisher during a time step according to the maximum profit at the end of the year by the value function. The state of a fisher is dependent on its cumulated landings, remaining quotas and related fines, but is also influenced by estimated direct gains $R\sim$, estimated discard fines $O\sim$ and other costs of choosing one particular option. This utility value $U(L\sim,q,f,t)$ contains the choice of a fishing ground and discarding options for the two size classes of the two quota-regulated species (the four are here gathered under the name *dis*), for a total of 5 switching parameters:

$$U_{g,dis}(L\sim,q,f,t) = \sum_s \left(\text{Ł}_{g,dis,s,t} \times p_{g,dis,s,t}\right) - O\sim_{g,dis,t} + R\sim_{g,t} - Cos_{g,t} + Ex_{g,dis}\left[V_{g,dis}(L\sim',q,f,t+1)\right] \tag{4.18}$$

Where Ł are the expected landings resulting from the choice of a fishing area and discarding options, $L\sim'$ reflects the change of the state $L\sim$ resulting from the expected landings during the time step and $Ex_{g,dis}[V_{g,dis}(L\sim',q,f,t+1)]$ is the expected value taken over all possible states resulting from the choice of a particular fishing ground and discard options. The latter parameter is of main importance because it links every time step choices to final optimal revenue.

The estimate of expected landings Ł by fishers depends on a 10-years average fish biomass available from OSMOSE in each fishing ground. The mean biomass is estimated in the middle of the time step. Variability in catch rates probability is based on a Normal distribution with a mean and a standard deviation extracted from the simulated biomass of species during the 10 previous years in OSMOSE, which represents the memory fishers have of their environment, knowing distribution of species, their prices and the cost of travelling to each fishing ground at every time step of the year.

Given an optimal choice by the fisher at the time step,

$$V(L\sim, q, f, t) = max_{g,dis}\{U_{g,dis}(L\sim, q, f, t)\} \tag{4.19}$$

Finally a matrix of optimal choices for fishing location and discarding options is produced, containing every state a fisher can encounter at each time step of the year.

Individual variability of fishers' behaviour was included through the assumption that they do not always make the optimal choice to optimize their annual net revenue φ. As previously depicted, choices can differ between two individuals according to two conditions: an imperfect knowledge of the optimal choice and the influence of their own tradition.

Following Dowling *et al*. (2012), a degree of rationality is incorporated by a method of errors in decision-making. Knowing that the optimal choice at each time step is defined according to a fishing ground g and discarding options for the two size classes of the two quota-regulated species (thereafter named *w*, *x*, *y*, *z*), we can set

$$\Delta_{g,w,x,y,z}(L\sim, q, f, t) = V(L\sim, q, f, t) - U_{g,w,x,y,z}(L\sim, q, f, t) \tag{4.20}$$

Where $U_{g,w,x,y,z}$ $(L\sim,q,f,t)$ is any utility value of the time step, including $V(L\sim,q,f,t)$. This can give a probability P of fishing in a particular area with specific discarding choices

$$P_{g,w,x,y,z}(L\sim, q, f, t) = \frac{e^{-\Delta_{g,w,x,y,z}(L\sim,q,f,t)/\sigma}}{\sum_d \sum_h \sum_m \sum_r \sum_u e^{-\Delta_{d,h,m,r,u}(L\sim,q,f,t)/\sigma}} \tag{4.21}$$

Where σ is the degree of rationality tuning the distance of a choice to the optimal choice. If σ is large, probabilities will be uniform while if it is very small, all vessels will make the optimal choice. In our simulations σ is set to 15000 because this value resulted in a certain amount of variability in fishers' choices, without giving too much weight to the choice to stay at port at a given time step.

The final choice also includes a part of tradition. Indeed tradition proved to be an important part of fishers' drivers (Girardin *et al*., 2017). We define tradition as the combination of the presence of a fisher in a fishing ground during a given time step of the year and the resulting success derived from it (see van Putten *et al*., 2012 and literature in it). Resulting success is here computed as the ratio between realized and expected revenue.

Tradition information is computed for each fisher based on their individual history over the 10 previous years. Prior to the launch of DSVM, tradition for fishers is based on the effort distribution displayed on average in the 2008-2015 periods by French OTB in the EEC (Figures S4.4-4.5). For each vessel the final probability Pf for each choice is:

$$\begin{cases} Pf_{g,w,x,y,z}(L\sim,q,f,t) = 0 \text{ if } P_{g,w,x,y,z}(L\sim,q,f,t) = 0 \\ Pf_{g,w,x,y,z}(L\sim,q,f,t) = \left(\dfrac{\tau\times\left(\frac{T_{g,w,x,y,z}\times P^{*}}{T_t^{*}}\right)+(1-\tau)\times P_{g,w,x,y,z,t}(L\sim,q,f,t)}{\sum_d\sum_h\sum_m\sum_r\sum_u\left[\tau\times\left(\frac{T_{d,h,m,r,u}\times P^{*}}{T_t^{*}}\right)+(1-\tau)\times P_{d,h,m,r,u}(L\sim,q,f,t)\right]} \right) \\ \text{if } P_{g,w,x,y,z}(L\sim,q,f,t) > 0 \end{cases} \tag{4.22}$$

Noting T^* the maximum value for tradition during the time step for the fisher and P^* the maximum probability at this state from DSVM. Based on the Random Utility Models (RUMs) meta-analysis performed by Girardin *et al*. (2017), tradition proportion τ in fishers' choice is set to 65%. Choices from profit maximization therefore account for 35%, plus a variable part concerning the hindering of a given choice if $P_{g,w,x,y,z}(L\sim,X,f,t) = 0$.

Backward computation is realized every 5 years, corresponding to a compromise between computation time and the time needed for fishers to adapt their perception of the ecosystem over years. Even if DSVM is not updated during a 5-year period, tradition still impacts the final choice fishers make.

Forward calculation: realization and consequences of fishers' choices

The forward calculation operates on in the time step when fishers make their choices and participate to the global fishing mortality in OSMOSE, the trajectory of choices of each fisher is determined by their history (i.e. the results of their fishing activities) and the optimized paths from the backward calculations. At each time step, a choice is randomly made in the probability distribution created by the addition of all *Pf*. Fishers catch a certain amount of fish which are removed from the different fish population simulated in OSMOSE (these fish caught are then used to compute a yearly realized F_{DSVM}). In addition to the fishes removed by the explicit fishing activity modelled, some other fishes are removed from the simulated stocks due to fishing mortality from F_{others}.

4.2.4 Run setting Calibration

As the fleet dynamics module is computer-time-consuming, the model is first run 70 years without DSVM to let the ecosystem stabilize (thereafter named “spin-up”) and be no longer driven by the model initialization. After spin-up the model keeps running for an additional 20 years with DSVM activated, leading to a total of 90 years simulated. In the following, the results presented correspond to an average over these last 20 years, i.e. with fleet dynamics. During the spin-up of the model, the selectivity curve is used for the F_{DSVM} proportion of total catches. However to correct the overestimation of F_{DSVM} in this context, a correction is made to obtain the $F_{DSVM}\, su$ that will be applied before DSVM:

$$F_{DSVM}\, su = F_{DSVM} \times \frac{C_{recruitment}}{C_{selectivity}} \tag{4.23}$$

Where $C_{selectivity}$ are the total yearly catches that would result of a selectivity curve and $C_{recruitment}$ are the total yearly catches that would result of recruitment sizes alone.

Calibration of the unknown parameters of this model is also performed in two main phases. The first one allows the estimation of a set of biological parameters (i.e. LTL accessibility coefficients, larval mortality rates, global fishing mortality and additional natural mortality) and corresponds to the spin-up time needed for the ecosystem to reach a stable state.

This first part is therefore performed without DSVM, and using an automatic evolutionary algorithm previously developed and applied to OSMOSE (Oliveros-Ramos *et al.*, 2017). A range of values are explored for the parameters evoked above and likelihood objective functions select the optimal values according to biomass and catches target values. Biomass and catches values were extracted for sole, plaice, whiting and cod from stock assessment report (ICES, 2017), while these were provided by Alemany (pers. comm.) for cuttlefish. For sole and plaice the modelled stock was representative of the spatial distribution of the stock as defined in ICES. For other species, the modelled stocks were only representative of a fraction of the assessed stock that was covering a wider area. Therefore values were scaled according to the proportion of catches in EEC compared to the total catches of these stocks. Concerning the other species, landings extracted from ICES official database were used instead of catches, and no biomass estimates were available. The first phase of the calibration was performed in 3 sub-phases: first, the LTL accessibilities were adjusted using 30 generations of parameters, then the larval natural mortality rates were added to these estimations during 30 additional generations and finally natural mortality and fishing mortality rates were also considered (in addition with other parameters) using 200 more generations. A total number of 55 parameters were estimated.

The second phase of the calibration allowed to estimate species catchabilities, needed for simulating fleet dynamics, and was therefore realized with DSVM activated. To find the best catchability estimates, simulations were run by giving 100% weight to tradition in the choice of fishers, i.e. meaning that fishing grounds are chosen among historical fishing effort distributions, and by trying to manually fit the realized F_{DSVM} to the estimated ones from the previously calibrated fishing mortality.

4.2.5 Scenarios

A reference scenario, or “Business as usual” (BaU) scenario, is run with a discards fine equal to 0 €, meaning that fishers are allowed to discard regulated species as much as they want without overshooting their landing quota. For the landing obligation (LO) scenario, an extremely high value is tested to mimic the reality of European landing obligation, i.e. a fisher can’t go fishing anymore if he exceeds his catch quota. The impact of the management scenarios are compared for economic and biological compartments. Concerning fishers, revenue, quotas, landings and spatial distributions of effort are observed. Concerning the biological part, species biomasses and several marine ecosystem state indicators are compared: Large Fish Indicator (LFI; i.e. proportion of fish above a certain length here set to 40cm; Essington *et al*., 2006), Mean Maximum Length of the community (MML; Jennings *et al*., 1999; Nicholson and Jennings, 2004), slopes of size spectra (Rice and Gislason, 1996; Bianchi *et al*., 2000) and Marine Trophic Index (MTI; Pauly and Watson, 2005). Results are computed from the average of 30 run replicates for each scenario, due to the stochastic nature of OSMOSE.

Table 4.4. Values given to several parameters used in the Dynamic State Variable Model, and their source. DPMA: French Directorate for Sea Fisheries and Aquaculture. EEC: Eastern English Channel. When no source is indicated, the corresponding values were arbitrarily set for this study.

Definition	Symbol used	Value	Source
Number of exclusive bottom otter trawlers	/	66	DPMA
Boulogne-sur-Mer		43	DPMA
Port-en-Bessin		23	DPMA
Number of years forming fishers' memory	/	10	/
Parameters used for the selectivity curve applied to the modelled fleet			
1st parameter	β	0.3	Girardin *et al*. (2016)
2nd parameter	α	17.5 (cm)	Girardin *et al*. (2016)
Specific fine multiplier for quota over-shooting (for cod and whiting)	f_s	10^9	/
Specific fine multiplier for discards (in the Landing Obligation scenario for cod and whiting)	o_s	10^9	/
Average distance between fishing grounds and ports in the EEC	$Dist_{ref}$	60 (km)	Empirical
Average travel time percentage in the EEC for the modelled fleet	$Ttime$	8 (%)	Empirical
Average fishing time percentage in the EEC for the modelled fleet	$Ftime$	92 (%)	Empirical
Cost of traveling per km for the modelled fleet (for a time step of two weeks)	℩	24.3 (€)	Estimated from Batsleer *et al*. (2013)
Cost for the reference fishing time	$Cfish_{ref}$	16740 (€)	Estimated from Batsleer *et al*. (2013)
Degree of rationality	σ	15000	/
Number of years used to compute fishers' tradition	/	10	/
Tradition part in fishers' fishing ground choice	τ	65 (%)	Girardin *et al*. (2017)

4.3 Results

4.3.1 Calibration

The final configuration we retained is the best given by the evolutionary algorithm. Species biomasses and catches stabilized on average after 40 years of run, and at the end of the simulation (average of the last 20 years over the 70 simulated) approximately half of biomass and catch values are within the range of observed values over 2008-2015 (Figure 4.3). Simulated catches are within the range of observations for red mullet, whiting, plaice, mackerel and sardines, while those of lesser-spotted dogfish, cod, sole, horse mackerel and herring catches are underestimated and pouting, cuttlefish and squids catches are above the observation ranges. Concerning biomasses, whiting, sole, plaice, cuttlefish and mackerel fall in the range of stock assessment estimates, while average cod biomass is slightly below the minimum biomass estimated and herring is lower than the minimum biomass estimated. With the current set of parameters (Table S4.1), it was not possible to reach an ecosystem state with all variables within their range of observations. However, the current configuration was considered valid for whiting and cod, the two species under focus to evaluate landing obligation scenario.

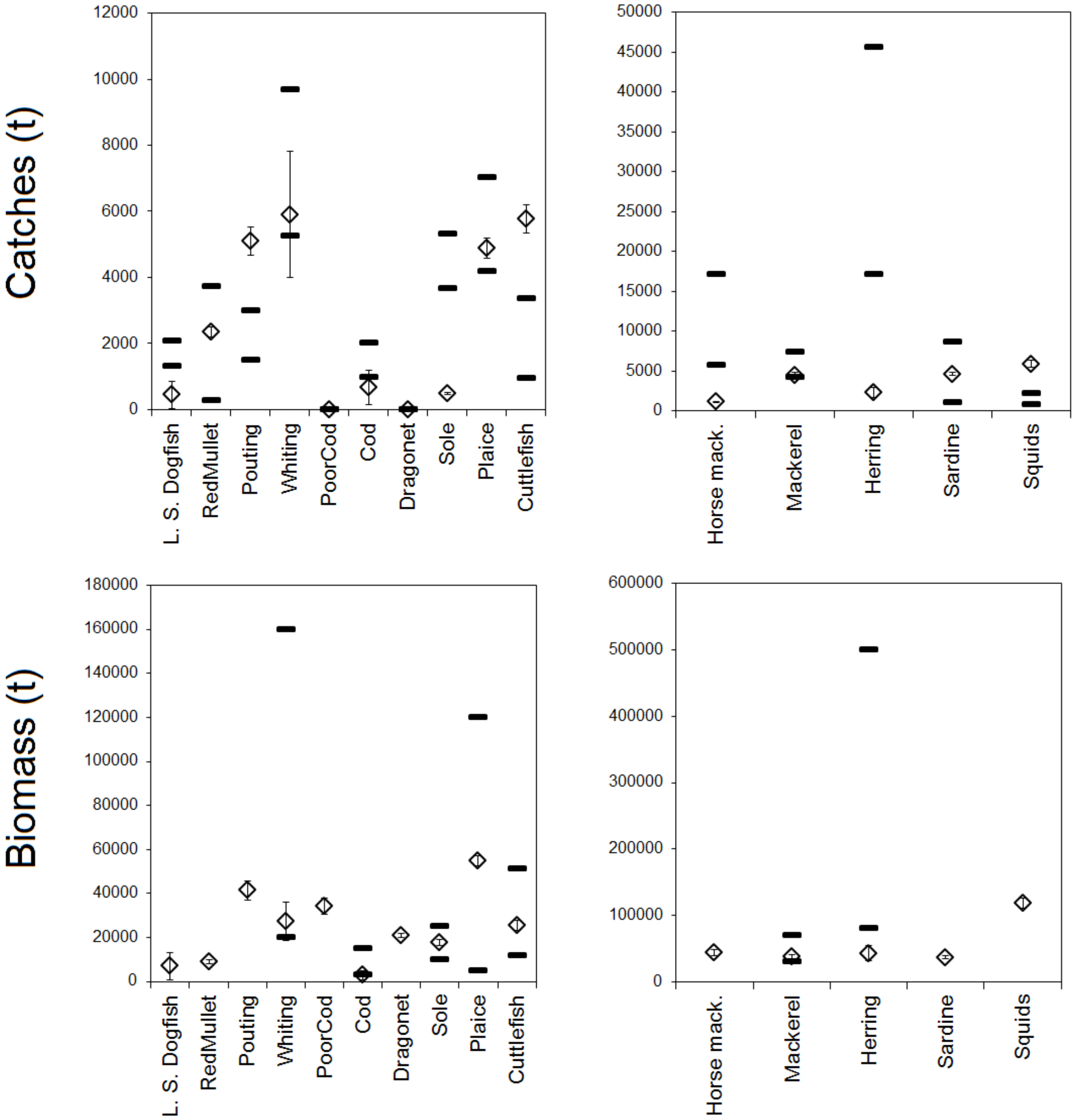

Figure 4.3. Distribution of the simulated catches (top) and biomass (bottom) over the 30 replicates (diamonds), and comparison with observations: minimal and maximum values of landings over 2008-2015 are indicated in the top panels by the black segments. Minimum and maximum estimated biomass values are only presented for assessed species in the bottom panels.

Using simulated values of biomasses and fishing mortalities, limits values used within HCR defining quotas can be estimated for cod and whiting, the two quota-regulated species of the model. This results for cod in F_{up} = 0.30, F_{low} = 0.15, Bpa = 1755 tonnes and Blim = 820 tonnes. With the simulated values, F is 26.7% higher than F_{up}. Using the same conversion with stock assessment values, this produces for whiting F_{MSY} = 0.27 and $MSY\ B_{trigger}$ = 34000 tonnes. F_{MSY} is here 22.2% lower than the simulated F. Thus both stocks are overexploited with these initial fishing mortalities.

4.3.2 Impact of the landing obligation on OTB fishers

Impact of the scenarios on quotas

For further indicators, only relative values are compared between “Business as usual” (BaU) and Landing Obligation (LO) scenarios. Quotas are observed on a five-years cutting due to their computation frequency in the model, and their evolution is presented relatively to the median value of the five first years in the BaU scenario (Figure 4.4). Quotas for whiting and cod had different trends according to the scenario considered. After 20 years, quotas were reduced by 55% and 80% for whiting and cod in the BaU scenario, respectively, indicating a decrease in their SSB coherent with their overexploited state. When the new policy was implemented, the quota for whiting increased by 75% in comparison to the base line scenario, and continued to rise, while the quota for cod was only reduced by 35% and remained stable.

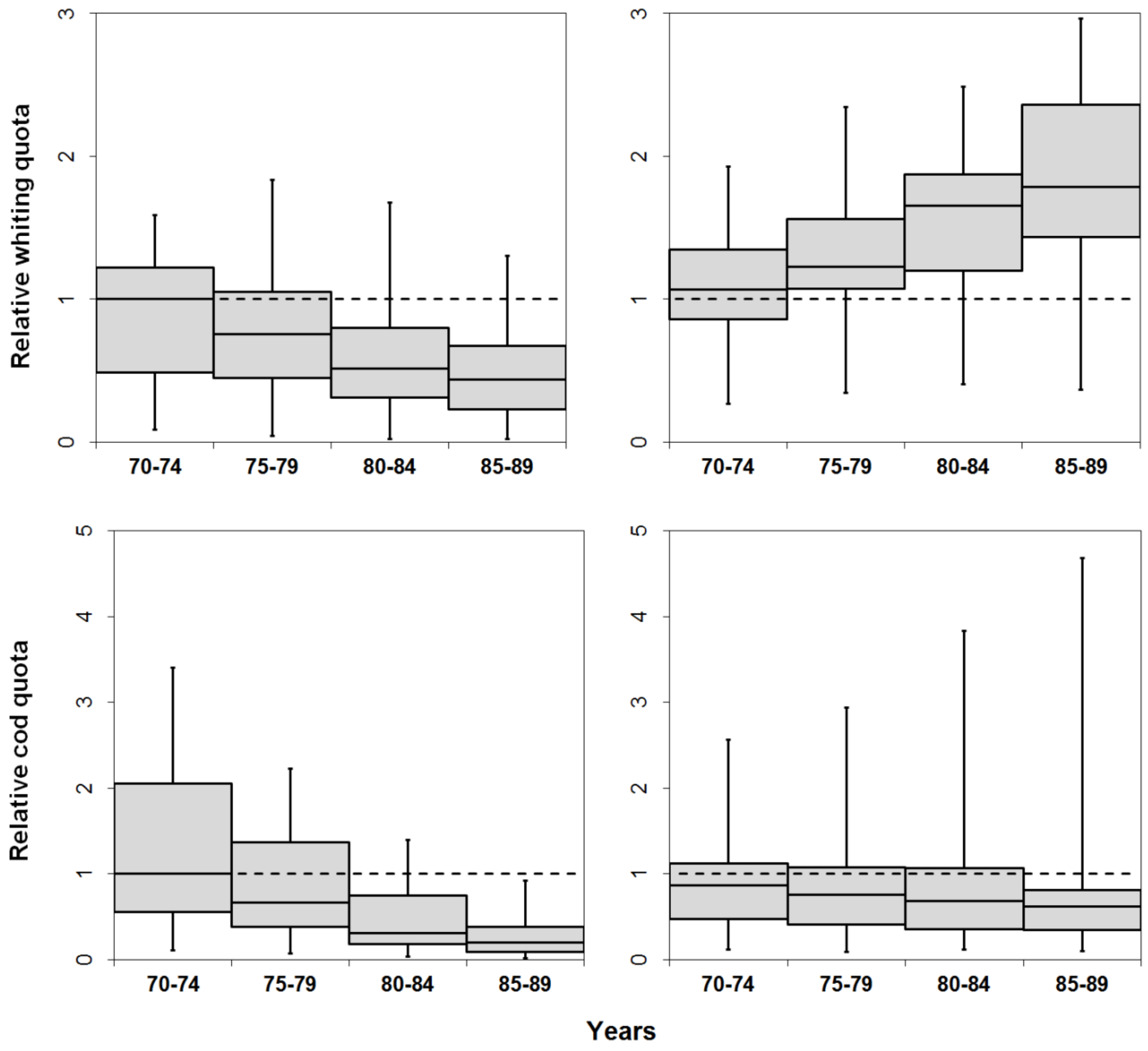

Figure 4.4. Relative changes in the simulated quotas of whiting (top) and cod (bottom) with "Business as Usual" (left) and Landing Obligation (right) scenarios. Minimum and maximum simulated quotas are represented by the short segments. Grey boxes represent Q1, median and Q3 ranges of simulated quotas (30 replicates).

Impact of the scenarios on fishers' effort distribution

Annual fishing effort resulting from the individual choices of the fishers is mainly distributed offshore in the center of the EEC, with two patches in front of Boulogne-sur-Mer (from vessels coming from this port) and Port-en-Bessin (corresponding to vessels coming from both ports) in the BaU scenario (Figure 4.5A). When LO is implemented, most of the fishing effort is concentrated offshore in the western part of EEC, while fishing effort in front of Boulogne-sur-Mer strongly decreased as vessels from this port moved westwards (Figure 4.5B). With the LO, both ports display a similar fishing effort distribution, contrary to the BaU situation (Figures S4.6-4.11).

At each time step, there is always a proportion of less than 1% of fishers on average who choose to stay at port in the BaU scenario (Figure 4.6). In the LO scenario, more fishers chose to stay at port. During the first two months of the year, about 20% of fishers choose to stay in their home port, and this proportion dropped to remain stable between 1% and 3% until the last two weeks of the year when 7% of fishers stayed in port. For both ports the winter pattern is similar, but all the fishers from Port-en-Bessin go fishing from April to October, contrary to a part of fishers from Boulogne-sur-Mer (Figure S4.12).

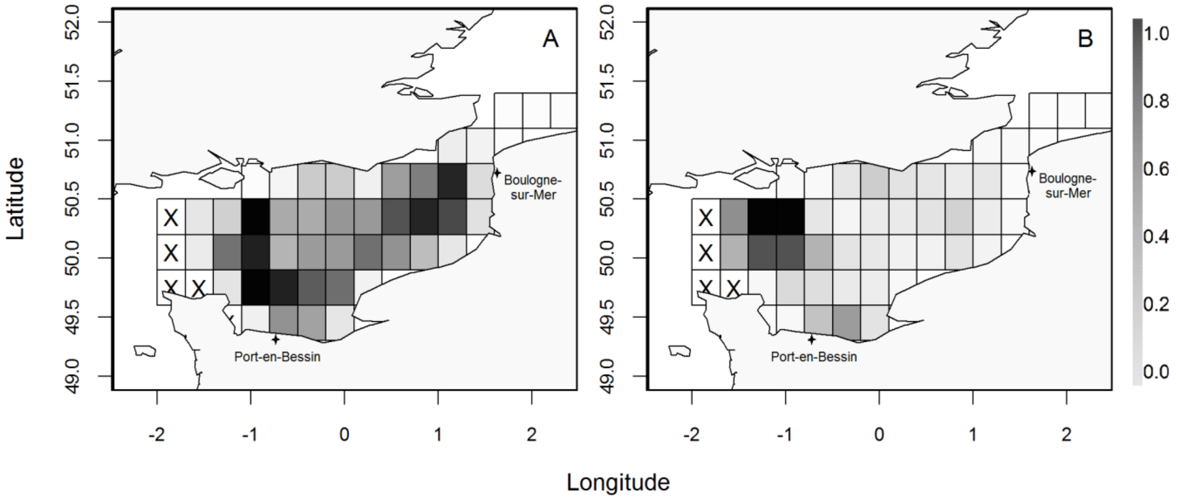

Figure 4.5. Mean annual relative distribution of fishing effort operated by exclusive bottom trawlers in Eastern English Channel with A) “Business as Usual” and B) Landing Obligation scenarios. “X” represents a null value.

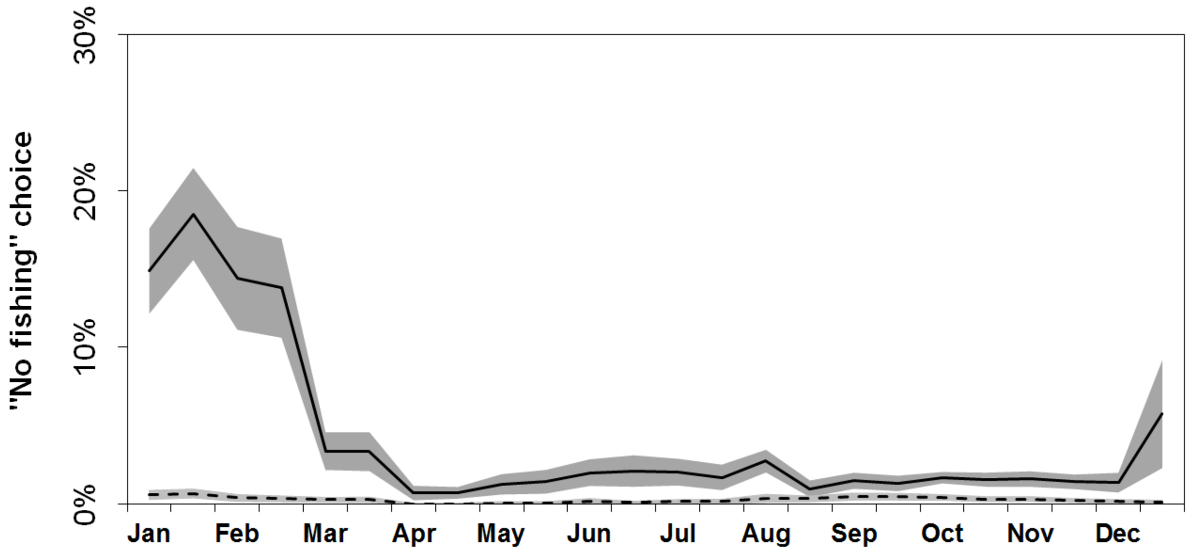

Figure 4.6. Mean percentage of "not to go fishing" choices for exclusive bottom trawlers operating in the Eastern English Channel with "Business as Usual" (dashed line) and Landing Obligation (full line) scenarios. Standard deviations are represented in grey.

Impact of the scenarios on fishers' catches

In comparison to the BaU scenario, landings realized by fishers under LO were more than doubled for whiting and cod on average (Figure 4.7). Landings slightly increased with LO also for squids and cuttlefish, while there were moderate decreases for horse mackerel and mackerel, and more severe ones concerning lesser-spotted dogfish, red mullet, pouting, sole, plaice, herring and sardine.

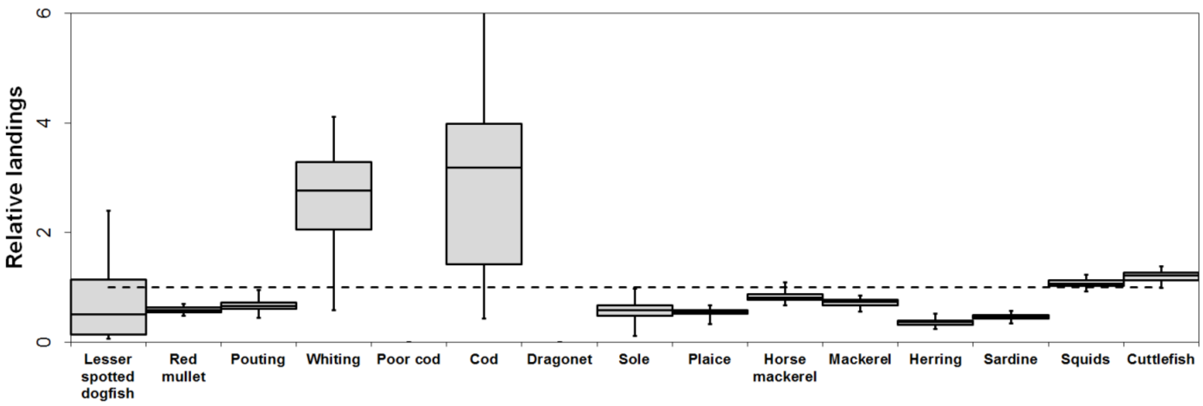

Figure 4.7. Relative changes in landings (averaged over the last 20 simulated years) of exclusive bottom otter trawlers in the Eastern English Channel with the Landing Obligation scenario in comparison to the "Business as Usual" scenario. Minimum and maximum simulated landings are represented by the short segments. Grey boxes represent Q1, median and Q3 ranges of simulated landings, over 30 replicates. Dashed line represents the basal value of median landings in the "Business as Usual" scenario.

Depending to the species considered, the consideration of a dynamic effort allocation led to catches in the BaU scenario being alternatively higher (e.g., whiting and cod; Figure 4.8), equal (e.g., red mullet and plaice) or lower (e.g., lesser-spotted dogfish and squids) than the catches simulated with a fishing mortality, i.e. without DSVM (for other species see Figure S4.13). For both quota-regulated species, discards represented the bulk of the catches in the BaU scenario. With the LO, the landings were slightly higher than the landings of the BaU scenario at the beginning of the simulation. They even increased (for whiting) or maintained (for cod) at the end of the simulation period, representing more than the total catches (i.e. including discards) of the BaU scenario. For the other species not subject to quota, different evolutions were observed (Figure 4.8 and Figure S4.13). For lesser-spotted dogfish, pouting and mackerel, landings were equivalent at the beginning of each scenario but increased in the BaU scenario and decreased with the LO. For plaice, sole, horse mackerel, red mullet, herring and sardine, landings were higher all along the 20 years with the BaU, even if they slightly decreased with time, while they decrease at a lower value with the LO scenario. Finally landings were equivalent for both scenarios at the beginning of the series for squids and cuttlefish, and slightly increased with the LO implementation.

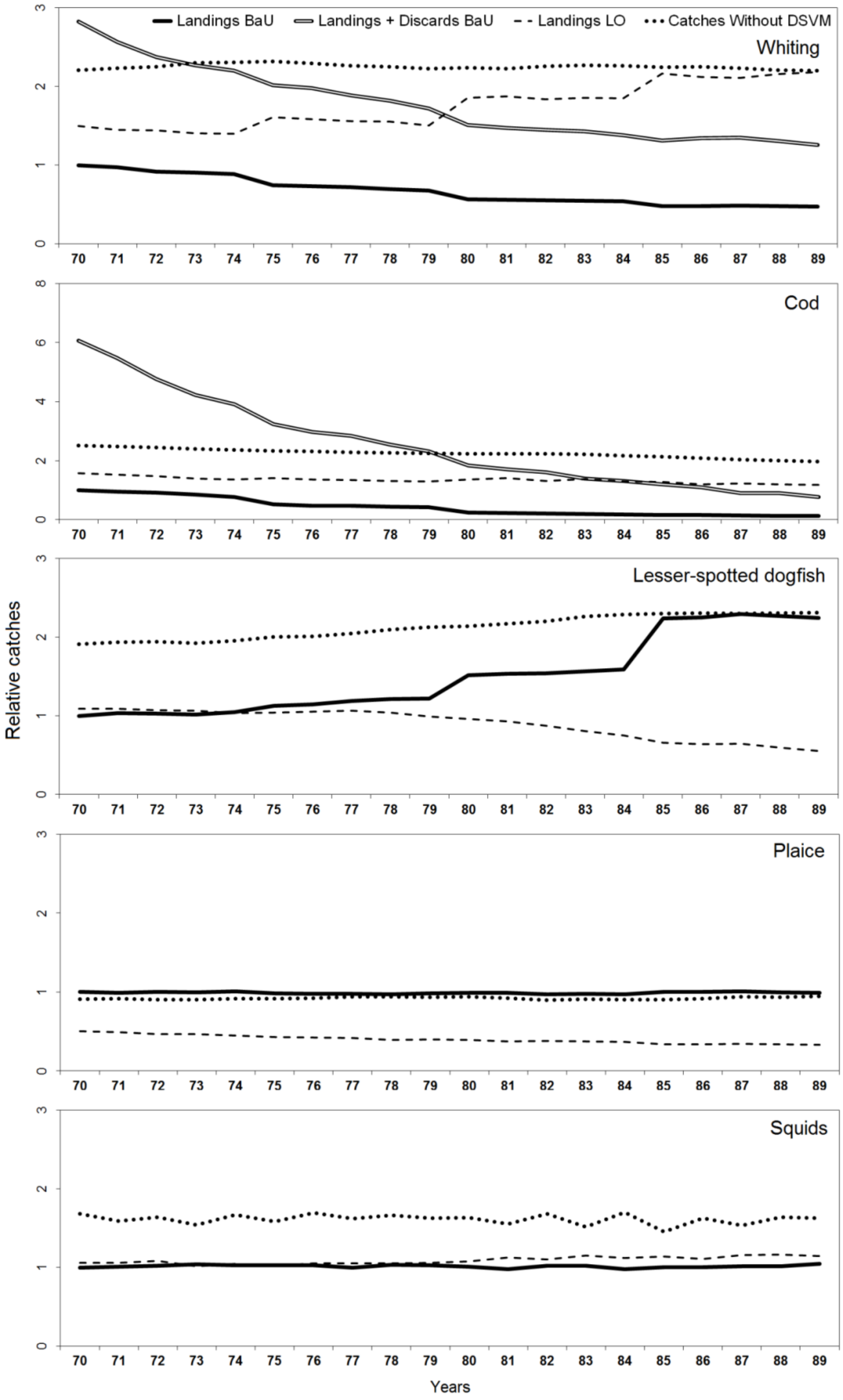

Figure 4.8. Temporal evolution of average catches of whiting, cod, lesser-spotted dogfish, plaice and squids by exclusive bottom otter trawlers in the Eastern English Channel. For quota-regulated species, landings (black lines) and catches (i.e. including discards, grey lines) are represented for the BaU scenario. Under LO, catches correspond to landings (dashed lines) as discards are banned. Catches simulated without DSVM (i.e. using a fixed fishing mortality instead of a dynamic effort allocation) are also represented for comparison (dotted lines).

Impact of the scenarios on fishers' revenue

The effects on the revenues of exclusive bottom otter trawlers of the landing obligation compared to BaU are shown in Figure 4.9. Revenues exhibited on average a 8.3% decrease from the beginning to the end of the 20 years simulated with the BaU scenario. The decrease is the most pronounced after 5-10 years of simulation and the revenue seems to stabilize during last ten years. With the LO implemented, fishers' revenues were stable during the first ten years, 2.9% lower than in the base line BaU scenario on average, with a high variability among replicates but increased during the last ten years to reach 7% increase above the base line BaU scenario at the end of the 20 years. During the last five years, fishers' average revenue is 15.3% higher with the LO than with the BaU scenario. The trends of revenues are similar for both ports (see Figure S4.14).

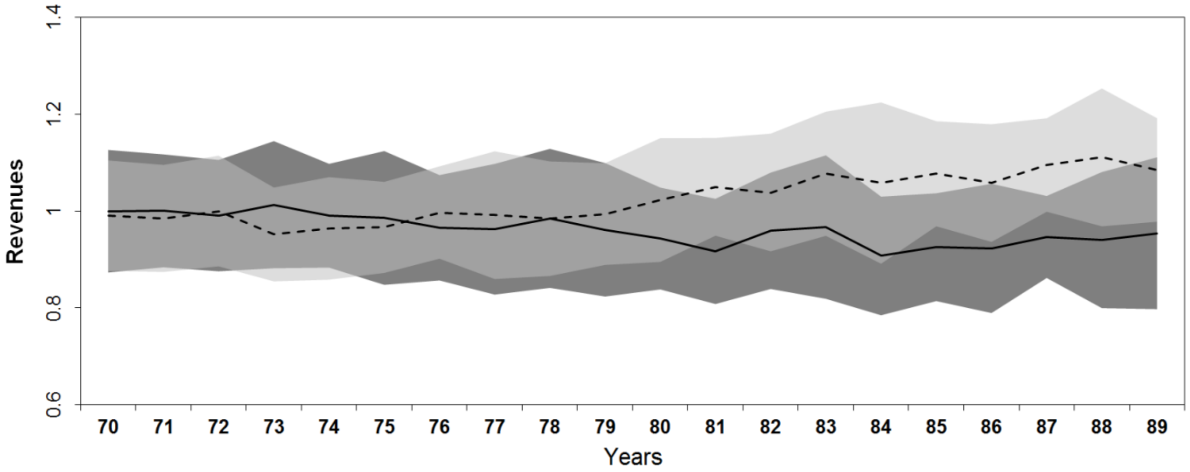

Figure 4.9. Relative changes of the revenue of exclusive bottom otter trawlers in Eastern English Channel with "Business as Usual" (full line) and Landing Obligation (dashed line) scenarios. Standard deviations are represented in grey.

4.3.3 Impact of the landing obligation on EEC ecosystem

Four size or trophic ecosystem indicators were compared between the BaU and LO scenarios: the slope of the abundance size spectra, the Large Fish Indicator (LFI), the Mean Maximum Length (MML) and the Marine Trophic Index (MTI) (Figure 4.10). The LFI and the MTI exhibited highly similar values in both scenarios. The slope of the size spectra (inverted on Figure 4.10) was slightly steeper in the BaU scenario than in the LO one, indicating a low improvement of the whole ecosystem state. Finally MML was slightly lower when the LO was implemented than in the BaU scenario, indicating an opposite trend.

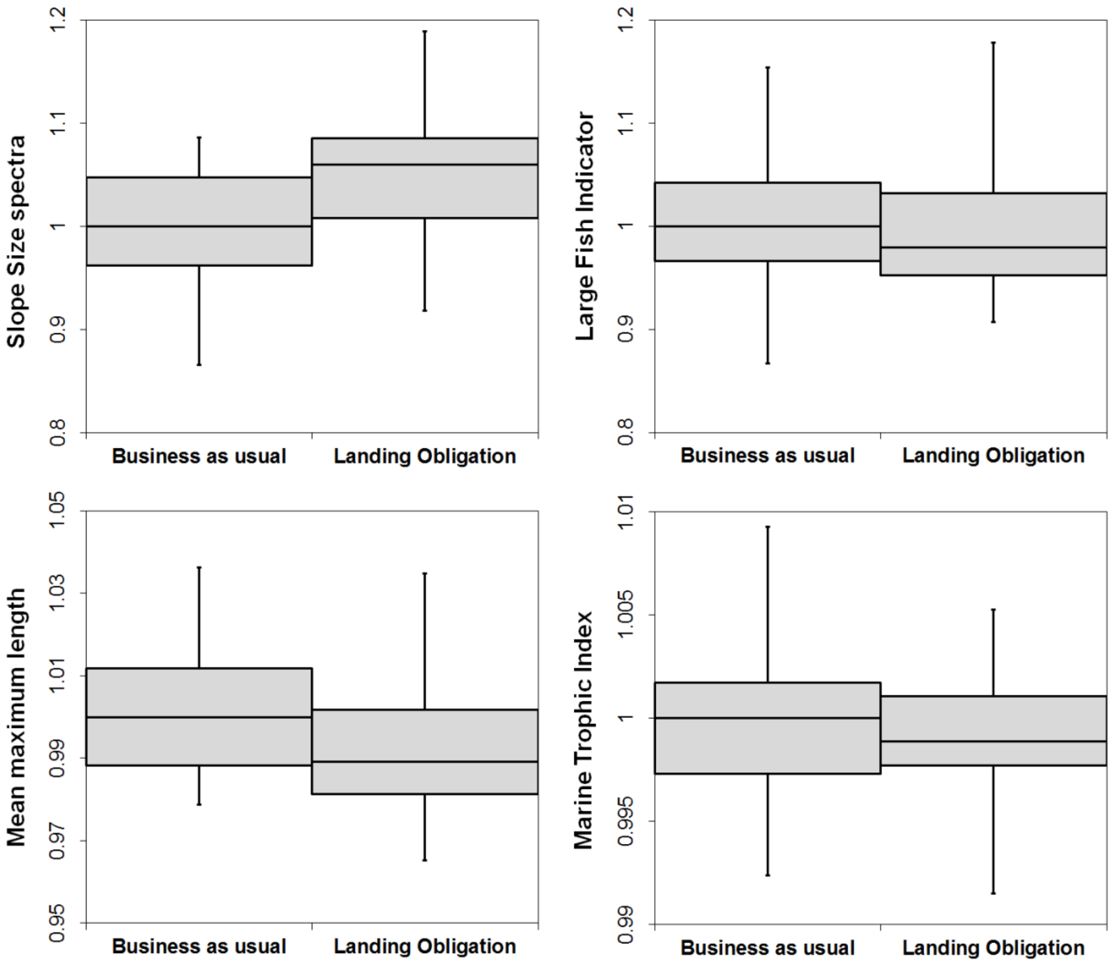

Figure 4.10. Mean relative values of slope of size spectra (inverted), Large Fish Indicator, Mean maximum length and Marine Trophic Index of the community with A) "Business as Usual" and B) Landing Obligation scenarios. Minimum and maximum simulated values of the indicators are represented by the short segments. Grey boxes represent Q1, median and Q3 ranges of simulated indicators.

With the LO scenario, biomasses were highly variable for lesser-spotted dogfish, whiting and cod (Figure 4.11), a pattern al in the calibration process. When LO is implemented, the biomass of lesser-spotted dogfish decreased by 75% in comparison to the BaU situation, while it was approximately three times higher for whiting and twice higher for cod. Concerning the other species, variations were +/- 20% beyond the base line value according to the species. The biomasses of pouting, sole, plaice, horse mackerel and mackerel decreased, while they expanded for, dragonet and cuttlefish and remained stable for red mullet, herring, sardine and squids.

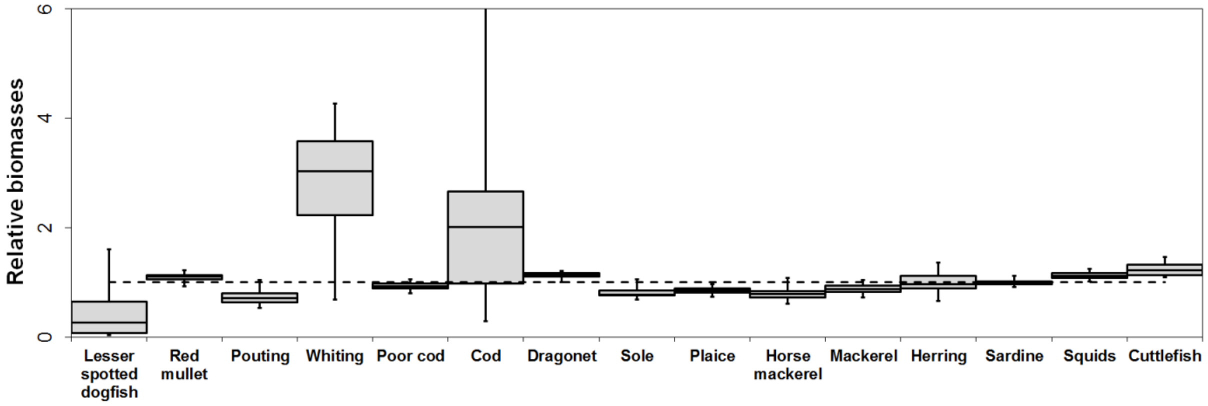

Figure 4.11. Relative changes in biomasses of species in OSMOSE with the Landing Obligation scenario in comparison to the “Business as Usual” scenario. Minimum and maximum simulated biomasses are represented by the short segments. Grey boxes represent Q1, median and Q3 ranges of simulated biomasses. Dashed line represents the base line value of median biomasses in the “Business as Usual” scenario.

4.4 Discussion

4.4.1 Landing obligation and its impacts

The coupled model developed here was able to predict long term deviations between ‘Business as Usual’ and Landing Obligation scenarios. When the LO is implemented, exclusive bottom otter trawlers exhibited strong changes in their fishing effort distribution. Even if tradition partly drives their choice, they moved away from their traditional fishing grounds, but also partly limited their fishing effort, particularly during winter. This shift of effort distribution resulted from the necessity for fishers to avoid areas where they faced a risk to exceed their quota when they were able to discard in the BaU scenario. The fact that the reduction of fishing activity mostly occurred during the first quarter of the year demonstrated that fishers have chosen to use their quota preferably towards the end of the year. This is *a priori* due to a combination of several parameters. First, the prices of quota-regulated species, such as whiting and cod, are prices are higher in the last quarter. Second, other species composing their landing profile, e.g., cuttlefish, one of the main commercial species in the EEC (Gras *et al.*, 2014), are increasingly accessible in the second part of the year (Bourdaud *et al.*, submitted). Fishers perhaps also had to deal with the fact that winter is the main season for discards in EEC, especially for whiting (Viðarsson *et al.*, 2016; see Figure S4.15 for model’s output). This quota reservation at the beginning of the year is typically a result on fishers’ behaviour that can only be predicted by the DSVM. It is noteworthy that the fishing effort reallocation is more strongly impacting trawlers from Boulogne-sur-Mer in their fishing habits and, as they had to move further offshore to avoid economically risky fishing grounds. Batsleer *et al.* (2013) included the North Sea in their analysis of the fishing effort reallocation of Boulogne-sur-Mer bottom trawlers, and the absence of the southern North Sea in OSMOSE is probably partly limiting the forecasting of the fishers' new behaviour. However, the results of their study with the LO scenario predicted a displacement of the fishing effort from the North Sea to the EEC, pronounced or moderate if the quota for cod was low or high, respectively. Even if the total fishing pressure is thus underestimated in EEC with the absence of fishing effort reallocation from the North Sea to the EEC, our observations of the EEC fishers demonstrated behaviour trends that could be applied to this percentage of unrepresented trawling activities.

The modification of fishing effort allocation had repercussions on the direct fishing performances of fishers, but also indirectly on fisheries management. When comparing the quotas trends between the two scenarios, there were evidences that the BaU scenario with DSVM was not beneficial for whiting and cod stocks sustainability, with a drop for both SSB. On the contrary the whiting SSB highly increased with the LO scenario, and the cod SSB remained stable. The BaU scenario is highly pessimistic on the quotas allowed for fishers, while the LO scenario is highly optimistic for whiting. Such measures are provoked by a stabilization of the SSB of quota-regulated species, due to management impact on fishing mortality. According to the simulations, total catches of whiting and cod were indeed strongly reduced during the first years of the simulations, mostly due to the reallocation of fishing effort. From this perspective, the LO seems to reach the goal of reducing fishing mortality for both whiting, and stabilizing the one for cod, without other incentives than quota limitation and landing taxes.

These events had consequences for the revenues of fishers, which were slightly reduced in the first ten years compared to the BaU scenario, but reached higher levels in the last ten years. This increase is mainly caused by the higher landings of whiting and cod, allowed by the increase of quotas, themselves following the increase of their biomass. Short-term consequences of a LO for fishers were explored in several models and our results corroborated their findings. In the short-term, economic losses were evidenced for Eastern English Channel and North Sea otter trawlers and netters (Batsleer *et al.*, 2013), English North Sea otter trawlers (Condie *et al.*, 2014b), and Basque trawlers in Bay of Biscay (Prellezo *et al.*, 2016). These short-term results were anticipated, considering that the landings obligation was expected to induce a reallocation of fishing effort. Indeed, if we consider that fishers are profit maximizers, a change in their habits should probably induce an economical loss. But the benefit of our methodology is to propose a medium-term forecast of the impact of the LO. Our results exhibited an inversion of the economic profitability for fishers at the term of the 20 years simulated, which was absent from these other models, due to the fact that trophic interactions and management evolution were together accounted for in the coupling of OSMOSE and DSVM. On long-term simulations with management but without trophic interactions, Simons *et al*., (2015) demonstrated an economic loss for international trawlers in the North Sea for a LO without additional measures, while Ono *et al*. (2013) suggested no profit loss with a discard ban for a theoretical mixed-stock fishery.

At the ecosystem level, no improvement could be evidenced. Clearly our results demonstrated that with the new policy, an improvement of state is highly directed on whiting and cod, while the main part of other species' biomasses was lower in the medium-term with the LO than with the BaU scenario. For most species, the implementation of the LO directly induced a reduction of the catches, due to the fishing effort reallocation. For sole, plaice, horse mackerel, red mullet, herring and sardine, the biomasses were stabilized, despite the lower fishing pressure, probably indicating a higher predatory pressure (indirect effect). For lesser-spotted dogfish, pouting and mackerel, LO seems to have induced a higher predation mortality reducing both biomass and landings. On the other hand, cephalopods seem to be less predated, which allowed their biomasses to increase and *in fine* increased their landing. Finally, unexploited poor cod and dragonet undergo an increase and a decrease of predatory pressures, respectively. Cod are high trophic level predators in the EEC, and their change in biomass (here there stabilization) has consequences on the mortality by predation exerted on a majority of potential preys in the ecosystem (Worm and Myers, 2003; Frank *et al*., 2005). Our results seemed to indicate that these managed stocks are the main levers of change in the community, as many small species' biomasses decrease while most of these did not suffer higher fishing mortalities. Travers-Trolet *et al*. (*in prep.*) using OSMOSE and Girardin *et al*. (2016) using Atlantis model emphasized that whiting and cod are key predatory species in the EEC ecosystem. Our results shared the same conclusions, any increase of these species biomass produces a decrease of prey and other predator groups. In the LO scenario, lesser-spotted dogfish was most impacted, by the emergence of cod and whiting as potential predators or competitors, while cuttlefish and squids were more resilient. While cephalopods' recruitments are mainly controlled by environmental conditions (Pierce *et al.*, 2008), this link could not be made explicit in OSMOSE. However the short-life specificity of cephalopods has probably something to do with this stability. Further works could particularly investigate their cannibalism behaviour and other possible density-dependence mechanisms which could explain this stability (Ibáñez and Keyl, 2010).

4.4.2 Fishers' behaviour in a new management policy context

Most of the studies which studied the impact of the LO insisted on the importance of incentives, enforcement and compliance for the benefit of such management policy (e.g. Borges *et al.*, 2016). If there are differences in the acceptance of diverse management measures in fisheries among Europe, there is a consensus against LO policies by fishers (Fitzpatrick *et al.*, 2017b), which could be a driver for non-compliance (Boonstra *et al.*, 2017). In the study, compliance was taken as a base line hypothesis, but further research should be done to incorporate the long-term impacts of non-compliance on the ecosystem. Contradictory results were observed on the ability for fishers to improve their targeting between Canadian British Columbia and US trawl fisheries in a LO context (Branch *et al.*, 2006; Branch, 2009). In the former a strict enforcement was implemented and induced a reduction of discards, whereas a poor enforcement resulted in extensive discarding for the latter (Fulton *et al.*, 2011). Condie *et al.* (2014) reviewed different LO experiences and also depicted the importance of compliance, with a reduction of discards when it was high for North America and Iceland. The authors also presented the difficulty to promote an efficient monitoring system (e.g., with an observer programme) due to the costs it would represent, but also that Remote Electronic Monitoring could at least alleviate the problem (Course *et al.*, 2011). It is noteworthy that it was tested and globally accepted in an experiment with Danish trawlers (Plet-Hansen *et al.*, 2017). Other solutions involving observable threshold for enforcement like move-on rules, real-time spatial management or direct fishers' collaboration seem to be promising alternatives to limit the catches of unwanted fish (Dunn *et al.*, 2014; Eliasen, 2014; Little *et al.*, 2015), but do not necessarily provide expected outcomes (Grafton *et al.*, 2005). While other methods are based on the control of fishers' activities, these methods have the advantage to involve fishers in the process, which is believed to improve the efficiency of fisheries management (e.g., Nielsen and Vedsmand, 1999).

In DSVM, the rationality degree is used as a black box to incorporate several processes not understood or not modelled. DSVM relies on a perfect knowledge of fishers, following the rational choice theory (Schlüter *et al.*, 2017), while the reality may be different (e.g. Abernethy *et al.*, 2007). Thus, in addition to tradition, the degree of rationality limits the perfect knowledge fishers could have of ecosystem processes. Even if information is not perfect because it is averaged over the previous years, more weight could be given to the exploration done by fishers for the probability estimates they have for each area in the model, particularly on areas poorly exploited according to historical observations of fishing effort distribution. However, the best choices fishers may operate for each state and each moment of the year is one of the outputs of DSVM backward computations. Our predictions could be compared with those from other methods used to determine the best areas suitable to fishing, which incorporate ecological and economic impacts (Paradinas *et al.*, 2016).

The model predicted a low reduction of average revenue during the first years after the implementation of the LO, but these results did not account for other costs. For instance, the potential increase of sorting time could not be quantified using our model. In addition, we neglected the effect of the longer travelling time fishers from Boulogne-sur-Mer should support to reach farther fishing grounds, while this could represent a physical risk but also a discomfort when accompanied by abrupt changes in their traditional working hours or habits. Also, only a part of the commercial species is modelled in OSMOSE, while other species may be of main importance for some regions of the EEC, e.g., scallops in the western EEC (Girardin *et al.*, 2015; Tidd *et al*., 2015). This is particularly true for fishers from Port-en-Bessin, although scallops is almost not taken by the modelled exclusive bottom trawlers (Leblond *et al*., 2014). However if smaller scale vessels had been modelled, it would have been necessary to add such species in OSMOSE. In our model, we choose to represent only exclusive bottom otter trawlers for the reasons previously evoked, but no distinctions were made between vessels, while these could have an influence on the catch potential. However adding finer definitions of boats would require computing more backward matrices, thereby inflating computing-burden.

Another important factor for fishers is the interference among vessels (Gillis and Peterman, 1998). In the DSVM structure, competition between vessels is not represented, while it can have consequences on fishers’ behaviours (e.g., Abrahams and Healey, 1993; Gillis, 2003; Poos and Rijnsdorp, 2007). However, a form of competition is created by the fact that visualization of fishing grounds by fishers is an average (on multiple years) of the biomass per fishing grounds before and after the fishing mortality process. Thus, highly targeted fishing grounds will endure higher biomass depletions, which will be reflected by the perception of catches fishers can have, but also by the direct results of their fishing activities on these grounds.

A limit of the current model is that our LO scenario only includes the possibility for fishers to change their fishing location or to stay at port to adapt to the new policy, but not to adapt their gear selectivity. Recent works on selectivity demonstrated the usefulness of selectivity devices to reduce the part of undersized whiting individuals in the English Channel and the North Sea (Vogel et al., 2017). Other work emphasized that Danish trawlers were able to change their catch composition to avoid unwanted individuals without loss of revenue, in an experiment of high gear flexibility (Mortensen *et al.*, 2017). Batsleer *et al*. (2016) also evidenced, using DSVM, that effective enforcement could enhance the gear selectivity of North Sea beam trawlers.

Flexibility in the broad sense is advocated as a requirement to produce positive biological and economic LO outcomes (van Putten *et al*., 2013), and it may also reduce revenue variability (Kaspersky and Holland, 2013), e.g., by allowing quota transfer between fishers (Johnsen and Eliasen, 2011; Sigurðardóttir *et al*., 2015), species (Simons *et al*., 2015), with and/or between countries ("swapping"; Ulrich, 2016; Catchpole *et al*., 2017b). Another kind of flexibility could be provided by the *de minimis* exemptions, which include whiting in the EEC (EC, 2016). While fleet dynamics were based on individual quotas' assumption with DSVM, French trawlers do not have explicit ones on the field. In reality each vessel has its own share of a cooperative's quota, which results in an approximation of the reality, but should be more precisely modelled in further applications.

In this work, only two species had their catches constrained by a quota. The addition of other quota-regulated species would have increasingly restrained fishers' flexibility. Plaice could have been another credible candidate to be modelled as quota-regulated species. In terms of direct ecological impacts, the forecasting would probably demonstrate lesser changes in the ecosystem due to the lower impact plaice have on other vertebrates groups (Girardin *et al*., 2016). Additionally recent works proved that plaice is a good candidate for survivability exemptions, as well as European seabass (*Dicentrarchus labrax*), skates and sole (Methling *et al*., 2017; Morfin *et al*., 2017a; 2017b). Among these species, sole is almost not caught by the modelled fleet while skates and European seabass are not modelled in OSMOSE. Other quota-regulated species present in OSMOSE like mackerel or herring are mostly considered as by-catches by the modelled fleet, and should not be limiting fishers' flexibility (e.g., Bourdaud *et al*., submitted).

Quotas computations created differences in the results obtained from our model. Indeed the computation done every five years was the strongest lever on fishers when the LO was implemented, because they impacted both F_{DSVM} and F_{other}. HCR used in the model were based on current policies, but were not adapted to LO. Quota uplifts, i.e. the upgrade of a quota to include the discarded part of catches, are not estimated in the model while they constitute a plausible accompanying measure to LO and could have strong consequences on fishers' flexibility. In terms of behaviour, we choose to define choices according to short-, medium-term profit, tradition and appreciations which evolve on different time scales. DSVM opportunities are modified every five years with the new set of quotas, while the ecosystem perceptions, tradition and evaluation are averaged on ten years patterns. It is difficult to estimate the influence of these time lapses, but we can guess that their importance is reduced using such scales than with year-by-year appreciations.

4.4.3 Predicting future using coupled fisheries-ecosystem modelling

Long-term forecasting is uncertain (Rochet and Rice, 2009; Planque, 2016). However, trends and outputs observed using our model can still inform on the potential reactions of the system according to diverse processes, especially when reasoning from relative values rather than absolute ones, as depicted by Pastoors *et al*. (2007). In the European H2020 DiscardLess project, several models are used to provide an assessment of the ecosystem impacts trends of a LO (Feekings *et al.*, 2015), and the addition of predictions will be able to give more robust predictions.

The imperfect knowledge we have on input values and their impact on final outcomes could only be observed using formal sensitivity analyses, which is still an ongoing research avenue for such complex models OSMOSE. The biological data implemented in OSMOSE mainly came from literature. If some values are well characterized and robust (e.g., von Bertalanffy's growth parameters), others are not well known (e.g., maximum ingestion rate) and/or are derived from the calibration. Before calibrating a model, routines such as the Pedigree from Ecopath (Funtowicz and Ravetz, 1990) could at least be used to quantify the quality of parameters used for the model.

Concerning outputs produced by the model, one of the solutions to strengthen the reliability of predictions would be to follow the pattern-oriented modelling approach (POM; Grimm *et al*., 2005). An example of the POM approach is the comparison between observed and simulated trends of commercial species landings in BaU scenario (Figure 4.12). Our results demonstrated relatively good fits for whiting and cod, moderate ones for cuttlefish and poor fits for red mullet and squids. These low fits are mostly due to low considerations of species' movements outside the EEC, which reduce the intra-EEC abundance of several species and are reflected in total landings operated by trawlers.

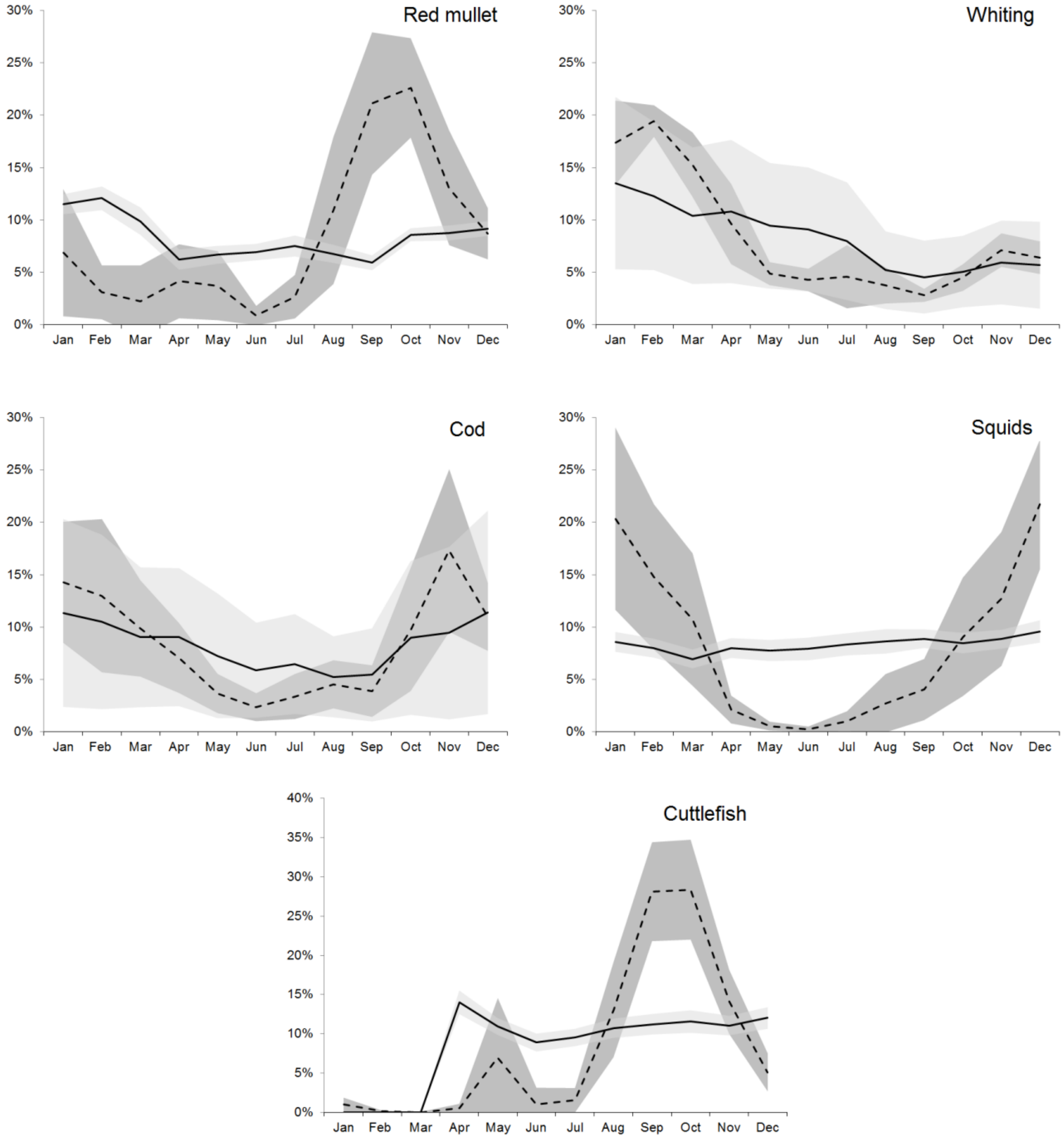

Figure 4.12. Simulated percentages of yearly landings of red mullet, whiting, cod, squids and cuttlefish caught by exclusive bottom trawlers with the "Business as Usual" scenario (full lines) and the same percentages obtained from empirical data (dashed lines).

Our results demonstrated the ambiguity of medium-term impacts of a LO. Fishers' revenues seem to reach acceptable thresholds after multiple years and the increases of biomass of whiting and cod. Condie *et al*. (2014) emphasized that a LO can produce better data collections and reduce waste, but it is not always inducing more sustainable fisheries. Our results demonstrated that it would be at least beneficial for highly commercial stocks managed by quotas, but would not have medium-term positive impacts on the whole ecosystem. Heath *et al.* (2014) demonstrated that implementing a LO without changing fishing practices would be negative for the ecosystem, while an avoidance of unwanted catches would be beneficial. The assumptions behind such results are simple concerning fishers' behaviour. However the trophic cascades used in this type of models include a greater part of ecosystem's compartments, which are currently not dynamically represented in OSMOSE. These authors suggested beneficial changes or negative changes on seabirds, marine mammals and seabed fauna according to the scenarios, which cannot be evaluated using OSMOSE. Additionally, Girardin *et al*. (2016), using Atlantis, observed that discard was an important source food for lesser-spotted dogfish and also had a high impact on the productivity of benthic invertebrates. In OSMOSE, discards do not represent a source of food for scavengers. Such impact could, however, have strong consequences on lesser-spotted dogfish biomass, but also on the trophic cascades induced from benthic invertebrates to demersal groups (Cachera, 2013). Another parameter not accounted for is the timeline the full policy implementation, knowing that an abrupt LO would have repercussions on scavengers, which are slow-adapting species (Fondo *et al*., 2015).

Concerning fishers, it is difficult to imagine that waiting multiple years could be attractive, without any financial compensation. Furthermore, our results are based on the assumption that seasonal prices are fixed, while these are known to be variable in time (Meuriot and Gilly, 1987; Loannides and Whitmarsh, 1987; Asche *et al*., 2015), and dependent on the quality and size of individuals, which is a driver of highgrading behaviour (Gillis *et al*., 1995; Batsleer *et al*., 2015). Dowling *et al*. (2012) modelled prices' fluctuations for one species, the broadbill swordfish (*Xiphias gladius*), but Prellezo *et al*. (2012) warned the difficulty to deal with multiple species price fluctuations in complex models. Finally unpredictable difficulties can divert the long-term evolution of fisheries and ecosystem at diverse levels: climate (e.g., Engelhard *et al*., 2014), societal events (e.g. Brexit) or environmental events inducing more or less pronounced recruitments for the managed species.

4.5 Conclusion

We coupled complex ecological and fleet dynamics models in order to forecast the medium-term impacts of the European Landing Obligation on the Eastern English Channel ecosystem and fisheries. Our results demonstrated that this new management policy would have short-term negative effects on fishers' revenue, induced by a large reallocation of their fishing effort to avoid quota over-shooting. Furthermore this model provided new results on the medium-term effects of the policy. By protecting quota-regulated species, global revenue can be profitable for fishers after several years. However, the LO indirectly increases the predatory pressure by cod and whiting on the other species, and hence does not improve the overall ecosystem health. Our results made the assumptions of a total compliance of fishers, and more research could still be undertaken to build in poor compliance in fishers' behaviour. Concerning the ecological compartments, further works could incorporate more compartments of the ecosystem, and multi-models approach should be promoted in order to strengthen the reliability of predictions (Peck *et al*., 2015).

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Chapter 5

General discussion and perspectives

5.1 Synthesis

The EU landing obligation and its impacts on fisheries and marine ecosystems is currently subject to debate. This thesis brings a contribution to the debate, by investigating the interactions between fishers and harvested resources in the Eastern English Channel (EEC), which required gaining extended knowledge on fisheries resource distributions (Chapter 2), developing novel approaches to get better insights into the mechanisms of fishers' species targeting (Chapter 3) and also building "eco-eco" models to simulate the impacts of LO implementation on both fish and fisheries (Chapter 4).

The second chapter provided a new approach to inform the spatial and seasonal distribution of marine commercial species. Species distributions are usually inferred from mandatory fishers' logbooks and/or fisheries-independent surveys. On the one hand, logbooks data are abundant but catch rates derived thereof often provide a biased perception of actual species abundance, since they generally do not account for discards and also due to their dependence on fishers' targeting behaviour. On the other hand, scientific surveys are subject to a well standardized protocol and provide for a broad spatial coverage, but often occur only once a year. I assessed the feasibility of using a relatively new source of data, on-board observers' data, to infer species' spatial and temporal distributions. Such data could provide a trade-off between reliability and coverage. For the third chapter, the added-value of using fine spatio-temporal biological and fishing effort data, with a focus on EEC exclusive bottom otter trawlers, was examined. The spatial overlap between the distributions of fishing effort and fisheries resources was explored to enhance the definition of effective fishing effort, with regards to two available surrogates: monthly trends in landings and fishers' intentions as derived from activity calendars. Furthermore, a newly-developed species targeting index was developed to quantify fishers' behaviour and response to co-existing commercial species abundance, prices and management measures. Finally, the fourth chapter evaluated some of the medium-term effects of a landing obligation on EEC exclusive bottom otter trawlers. This was done by coupling the trophodynamic multispecies model OSMOSE applied to the EEC with a fleet dynamics model able to anticipate fishers' reaction to a changing management environment, the Dynamic State Variable Model (DSVM).

The first study of this thesis, reported in Chapter 2, concluded that on-board observers' data, from the French OBSMER programme, can give valuable insights into the seasonal distribution of marine species. After standardizing catches per unit of effort (CPUE) from OBSMER, and combining them with abundance indices from the October Channel Ground Fish Survey (CGFS), monthly distributions of key species were mapped and compared to existing literature knowledge, with a focus on cuttlefish, one of the main commercial species displaying pronounced migrations in the EEC (Gras *et al*., 2014). The results were meaningful, with trackable patterns for a set of species for which information was available. The methodology used succeeded in providing seasonal distribution maps for a large number of species. The fine scale consistencies of the species distributions were assessed using a geostatistical index measuring the similarity between maps derived from OBSMER and CGFS data during the same month (October). This index was based on an overlap metric, the Local Index of Collocation (LIC), which gave a correlation value between both maps, together with a p-value derived from a permutation test. Many of the species sampled demonstrated a clear spatial distribution resemblance across the two data sources. However, results were more mitigated for some other species, partly due to their ecology, and day/night variations that could not be monitored from survey data. The monthly species distribution maps and/or the fishing power derived from Chapter 2 were further used to quantify interactions between fishers and fisheries resource, to improve fishing effort and pressure estimates (Chapter 3), and to build in seasonal distribution patterns in the OSMOSE model (Chapter 4).

The examination of fishers-resource fine scale interactions allowed adjusting nominal effort, and improved the ability of the resulting effective fishing effort to mimic the trends of available fishing pressure surrogates (monthly landings and fishers' intentions), particularly for cuttlefish and red mullet. These two species are highly seasonal, sold at a high price and not regulated by quotas. I evidenced that both species were driving fishers' spatial distribution, using a newly developed indicator to quantify fishers' targeting in a mixed fisheries context. The inter-annual variations in October targeting intensity were analysed for cuttlefish and cod, and related to species abundance, prices and quota management if existing. The results suggested that fishers' cuttlefish targeting was partly driven by species' availability, while cod targeting was limited by available quota. The results from Chapter 3 were partly used into the end-to-end model developed in Chapter 4, by building in the importance of quota availability as a driver limiting fishers' targeting. These results also provided empirical evidence of the importance to get good insights into fishers' resources spatial and temporal interactions, including avoidance behaviour for some choke species, which were useful to interpret how and why the Landing Obligation could affect future fisheries and the ecosystem they exploit.

In the fourth chapter, the coupling between OSMOSE and DSVM models allowed forecasting the impacts of the landing obligation on EEC ecosystem and fisheries. The overall impact on the ecosystem was assessed using species biomasses and several size-based and trophic-based ecosystem indicators. The impact on fishers' economics and activity, measured in terms of global revenue, landings and fishing effort distribution, indicated that fishers' medium-term revenue was increased with the landing obligation scenario in comparison to the "Business as Usual" one. However, this increase was only appearing after several years, due to an increase of both whiting and cod biomass which was inferred by fishers' avoidance behaviour. The landing obligation neither improved nor altered the overall ecosystem health, but the predatory pressure induced by cod and whiting decreased the biomass of several prey species in the EEC. This chapter constitutes a breakthrough from previous literature studies, as it presents the medium-term changes that could be induced by banning discards, building on individual variability leading to dynamic trophic interactions coupled with a complex representation of individual fishers' behaviour.

5.2 Beyond the results

5.2.1 Spatio-temporal resolution of fish distributions and fishing grounds

'*And it may be that what chiefly holds back our progress in natural science is the lack of relevant experiments and phenomena, which are often found only by chance, and sometimes when they are needed can't be found at all, even by the most persistent and careful enquiry.*' (Hume, 1748).

Chapters 2 and 3 were based on the assumption that marine species distributions could be well represented at the spatial scale chosen (i.e. 0.3° x 0.3° corresponding to 18nm x 18nm). Although this spatial scale did not able to completely track abrupt yearly distribution shifts, was below the "patchiness" limit defined by Poos and Rijnsdorp (2007) to be between

20 and 45 nautical miles (nm) for benthic species such as sole and plaice on a 2-3 weeks window in the North Sea. Most of the species examined in this thesis being also benthic or demersal species, with the substrate taken into account resulting maps can be trustful (Trenkel and Skaug, 2005), but more precautions must be taken with pelagic species.

The spatial scale of the fishing grounds considered in this thesis corresponded roughly to the mean spatial dispersion of EEC bottom trawlers' fishing activity during a trip according to historical data. However, there is also evidence that fishers may organize their activities and target fish at smaller scales 0.3° x 0.3° (e.g., Rijnsdorp *et al.*, 1998). Future work could refine the spatial resolution of fisheries' observations (Pet-Soede *et al.*, 2001), by explicitely building in the seafloor geomorphology as a driver of fishers' distributions (Piet *et al.*, 2000; Stelzenmüller *et al.*, 2008). Fishers probably also make choices at a finer temporal resolution to that considered in this dissertation, e.g., by targeting cod at night, an issue evoked in the second chapter (see also Thorson *et al.*, 2016). Considering such a fine temporal scale could also allow separating out exploration and exploitation phases within a fishing trip (Rijnsdorp *et al.*, 2000b).

In each chapter of this thesis, I used for simplicity a fixed regular grid to map the different variables under investigation (fishing effort, fisheries resources, CPUEs, *etc.*). Branch *et al.* (2005) suggested that 'the tyranny of the grid' could have adverse effects on the relevance of fisheries model outcomes, since fishing activity is not constrained into squares but rather related to real fishing opportunities (Ono *et al.*, 2015). Still, considering the analysis envisaged with geostatistical indicators and the data available, the use of such a regular grid was necessary, especially concerning the biological compartment.

If OSMOSE explicitly builds in the dynamics of fish super-individuals into spatial cells, fishers were not represented in the previous versions of the model. Estimating the performance of all possible fishers' choices within DSVM is time-consuming, therefore a compromise was made and led to model fishing choices at the scale of the fishing trip. This scale was considered appropriate, although it did not allow evaluating, in Chapter 4, how fishing effort could be affected by environmental factors or habitat preferences.

5.2.2 Fishers and fisheries resources in a changing environment

‘*Why is the old farmer more skillful in his calling than the young beginner if not because there is a certain uniformity in how the operation of the sun, rain, and earth affects the production of plants, and experience teaches the old practitioner the rules by which this operation is governed and directed?*’ (Hume, 1748).

In this thesis, and more particularly in Chapter 4, it was assumed that spatial fish distributions built in OSMOSE remain invariant across the year. Although the distribution of EEC fish communities is believed to have changed substantially since 1998 (Auber *et al.*, 2015), the use of constant spatial distributions was justified in that only data from the years of community stability were selected to feed in the model. However, this simplification limits the capacity of OSMOSE to anticipate regime shifts (Scheffer *et al.*, 2001; Scheffer and Carpenter, 2003), or climate change impacts (Hoegh-Guldberg and Bruno, 2010; Sumaila *et al.*, 2011).

Admittedly though, the objective of this work was not to forecast the effects of climate change on ecosystem trajectories. Still, building in ecosystem models processes such as climate-induced changes of commercial species distributions, recruitment successes, and their repercussions on fisheries management advice and measures could be the subject of further investigations.

In Chapter 4, costs were integrated in the form of simplified and aggregated fuel costs. Further works could refine cost estimates, including individual variability due to from vessel-specific maximal speed, fuel consumption rates or fuel tank capacity (e.g., Bastardie *et al.*, 2010), given fuel costs fluctuations can limit the extent of fishing effort distribution (e.g. Poos *et al.*, 2013).

Sociological aspects have not been accounted for in this thesis, although they could influence fishers’ behaviour (Liu *et al.*, 2007; Carpenter *et al.*, 2009; Lade *et al.*, 2013; Hentati-Sundberg *et al.*, 2015; Schill, 2017), and *in fine* ecological dynamics (Lade *et al.*, 2015). For example the list of species concerned by European Landing Obligation is evolving as a result of, e.g., survival tests that are currently experimented on species such as plaice and rays (e.g. Kingma and Walker, 2014). Brexit, when fully implemented, will probably also have strong (yet unknown) implications for French and English fishers’ effort distribution in the EEC, especially concerning scallops fishing. This will also apply when new sectors of activity occupy the EEC marine domain (e.g., wind farms).

5.2.3 Modelling fishers diversity

‘*Such complete uniformity is never found in nature. On the contrary, from observing the variety of conduct in different men we are enabled to form a greater variety of generalizations, which still presuppose a degree of uniformity and regularity ·underlying the variety.*’ (Hume, 1748).

In all research chapters, the between-fishers variability was taken into account in different ways. In Chapter 2, average fishers’ catch rates (CPUE) were used to infer the distribution of marine species, after being standardized to remove the effects of vessel and gear characteristics. This approach of CPUE standardization, however, neglects fishers’ individual ability to find fish, which may differ according to skipper skills (e.g. Abrahams and Healey, 1990; Marchal *et* al., 2006), or fine-scale vessel and gear adaptations (e.g. Robins *et al.*, 1998; Mahévas *et al*., 2004). It also neglects the possible bias incurred by the presence of on-board observers, i.e., non-random assignment of fishers (deployment effect) and change of fishers’ behaviour (observer effect) (e.g., in Australia: Liggins *et al*., 1997; in Canada: Benoît and Allard, 2009). When refining the definition of the fishing effort for exclusive bottom trawlers in Chapter 3, only total length differences across vessels were taken into account, in addition to the spatial overlap between effort and resource distribution.

When investigating species targeting, I interpreted the values of the species-specific coefficients in relation to those that would be anticipated if fishers were fully driven by expected revenue, building on the Ideal Free Distribution (IFD) theory. I interpreted the deviations from expectations by one of the IFD assumptions, i.e. fishers’ choices are unconstrained, being at fault due to, e.g., management. Another key assumption underpinning the IFD theory is that competition occurs in fisheries, directly through entanglement of fishing gears, or indirectly by local prey depletion and/or depression (i.e. the prey avoidance behaviour; Gillis, 2003; Poos, 2010 pp 154-155) (see Abrahams and Healey, 1993; Gillis and Peterman, 1998; Gillis, 1999; Rijnsdorp *et al*., 2000b; Poos and Rijnsdorp, 2007b). IPoos *et al.* (2010) suggested that direct interference between beam trawlers is limited in the North

Sea, which is unlikely to be true for a congested sea such as the EEC, where competition for space and resource should be high. The third chapter of this thesis did assume that competitive interaction among fishers existed, but it did not consider other competing human activities (e.g., shipping, windfarms). Further works could incorporate interactions between fisheries and other activities, to better identify drivers of fishers' behaviour (e.g., Pet-Soede *et al*., 2001; Abernethy *et al*., 2007; Daw, 2008).

Fishers adapt to different factors including fishing grounds preferences or management, which were in Chapter 3 combined into the choice of target species. In previous studies species targeting was determined using varied techniques ranging from simple threshold settings, e.g., using percentage of the catches above a pre-determined value (Lorance and Dupouy, 2001) to more complex multivariate analyses (Pelletier and Ferraris, 2000; Poulard and Léauté, 2002). The approach developed in this thesis avoids issues related to the use of landings data (Marchal, 2008). Christensen and Raakjaer (2006) and Boonstra and Hentati-Sundberg (2016) demonstrated the usefulness of direct interviews in understanding fishers' decisions and the diversity of their reactions in relation to their history, environment, tradition and own personality. They defined 'fishing styles', which could be an alternative to define métiers (Ulrich *et al*., 2012). In any case, defining métiers inevitably simplifies the reality of fishing intentions and activities. In the third chapter of this thesis, fishers' home harbour was not taken into account in the different analyses. However, home harbour may influence the distribution of fishing effort by affecting the distance to fishing ground, but also due to regional or familial traditions.

Chapter 4 explicitly accounted for fishers' diversity. Indeed, DSVM is an Individual-Based Model, where the response of fishers to management is processed individually. Even if physical differences between vessels such as length are not taken into account in the model, each fisher belongs to one of the two EEC home ports (Boulogne-sur-Mer or Port-en-Bessin), has its own catch entitlement and individual history. Although not explicitly built in the OSMOSE-DSVM model, exploitation competition emerges from the simulations, as described in Chapter 4's discussion.

One of the key sociological questions inherent to the Landing Obligation enforcement is fishers' compliance. It represents a real management issue (Mora *et al*. 2009; Pitcher *et al*. 2009) and a complex research field due to the broad diversity of fishers' response (Boonstra *et al*., 2017). In the model, we consider that all fishers comply with the policy, i.e., the landing tax, but a varying fine could also have been used to test the sensitivity of fishers to disincentives (Batsleer *et al*., 2013). In the context of the landing obligation, other non-compliance features could have been taken into account, such as the development of illegal markets in relation to poor enforcement measures (de Vos *et al*., 2016; Bellido *et al*., 2017). The EU landing obligation is currently far from being fully implemented, and it is difficult to anticipate whether current enforcement measures will suffice to prevent fishers' non-compliance. According to Branch *et al*. (2006) and Branch and Hilborn (2008), a full compliance would require full observation programmes, which is not currently possible (Condie *et al*., 2014; Sardà *et al*., 2015). In the OSMOSE-DSVM model, fishers are able to perceive stocks' abundance and the revenues they could expect from fishing trips. However, risk-taking attitudes may vary across fishers (Kahneman and Tversky, 1979; see 'prospect theory' in Poos, 2010 pp 155-157; van Putten *et al*., 2013), a process that could not be captured in the model. Risk preferences and individual variability in fishers' attitudes has been a well-explored field in fisheries sciences (Eggert and Martinsson, 2004; Eggert and Tveteras, 2004; Dwyer and Minnegal, 2006; Gelcich *et al*., 2007; Holland, 2008), with outcomes generally suggesting that fishers may be considered as risk-averse (Girardin *et al*., 2017). In DSVM I used linear distribution for cumulative probabilities, but in details the weighting fishers give to the extreme values of expectations could be different (van Putten *et al*., 2013). Finally, information exchange among fishers has proved to have importance in determining fishing effort distribution (Vignaux, 1996; Curtis and McConnel, 2004), and that process could be built in future OSMOSE-DSVM developments.

5.3 Perspectives

5.3.1 Forecasting marine ecosystems dynamics

'*But scientists, observing that in almost every part of nature there are vastly many different triggers and causes that are too small or too distant for us to find them, judge that it's at least possible that the contrariety of events comes not from any contingency in the cause—·i.e. the cause's being inherently liable to fail to produce the usual effect·—but from the secret operation of contrary causes.*' (Hume, 1748).

‘*We can have only a doubtful expectation of an outcome that is supported by a hundred instances or experiments and contradicted by fifty; though a hundred uniform experiments with only one that is contradictory reasonably generate a pretty strong degree of assurance. In all cases where there are opposing experiments, we must balance them against one another and subtract the smaller number from the greater in order to know the exact force of the superior evidence.*’ (Hume, 1748).

Our capacity to forecast ecosystem dynamics with high precision is constrained by two technical barriers. Firstly, he amount and quality of data available to statistical analyses and modelling was variable, an issue which could be addressed through new information and methodological developments brought about by future research projects.

Second, there are barriers inherent to the complexity of ecological, societal and mental systems, which could only be overcomed by innovative conceptual developments and paradigm shifts (see Prigogine and Stengers, 1984; Mullon *et al*., 2009; Planque *et al*., 2014; Michaelian, 2016). Fisheries sciences are complex because they combine these different fields, and their associated uncertainties. This is particularly true when it comes to anticipating the fishers’ response to a new management measure like the landing obligation. If the probabilistic nature of OSMOSE-DSVM captures part of the uncertainty inherent to marine systems, the model is still constrained by fixed parameters and forcing variables. Uncertainty, and how it propagates through complex models, needs to be examined to entrust the science supporting ecosystem approach to fisheries (Hill *et al.*, 2007). This could be achieved through sensitivity analyses approaches applied to complex models (Pantus, 2006). Often used as a successful example, weather forecasting has to deal with the chaotic nature of atmosphere (Lorenz, 1963) and could perhaps inspire ecosystem modelling in terms of both accounting for uncertainty using a multiple-modelling approach, and presenting results depending on the time scale of the predictions (e.g., Clark and Hay, 2004). Additionally, as depicted in the fourth chapter, an equivalent of the Pedigree (Funtowicz and Ravetz, 1990) from Ecopath could be created for OSMOSE to assess the quality of input data used to calibrate and run this kind of model. Food-web modelling, despite several shortcomings, is indispensable to project the long-term response of ecosystems to anthropogenic perturbations (Botsford *et al*., 1997; Peck *et al*., 2015; Grimm *et al*., 2017).

The environment is evolving and a realistic long-term forecasting would need a coupling with climate forecasting (Perry *et al.*, 2005; Tommasi *et al.*, 2017), which could have complex effects on EEC fish communities (Genu, 2017). However, long-term forecasting also imply long-term fishing strategies (see Christensen and Raakjear, 2006), which would require the complexification of the economic parts of OSMOSE-DSVM.

5.3.2 Understanding human behaviour

> '*The scientist, if he is consistent, must apply the same reasoning to the actions and decisions of thinking agents. The most irregular and unexpected decisions of men may often be explained by those who know every particular circumstance of their character and situation.*' (Hume, 1748).

The 'fishing style' approach (Boonstra and Hentati-Sundberg, 2016) previously evoked, as well as other attempts to deal with fishers' diversity (e.g. Hanna and Smith, 1993; Hind, 2012; Sønvisen, 2014), can help limiting the space of plausible fishers' behaviours. Additionally some refinements could be added in fleet-dynamics models by interviews, e.g. on the catches/revenues ratio interest for fishers (see Lehuta *et al.*, 2015), on compliance (e.g. Hønneland, 1999 for the theory), by defining more precise costs (e.g. Bastardie *et al.*, 2010), by including the importance of recent success (e.g., Beecham and Engelhard, 2007) and finally by adding the weight of information exchange (e.g., Little *et al.*, 2004; Millischer and Gascuel, 2006).

Research could also focus on the understanding of the sociocultural context of fleets and fisheries. Nowadays, social sciences apply to fisheries concepts developed in other fields, such as the adaptation of farming and agriculture sciences to move beyond the traditional *Homo economicus* picture ('the good fisher'; Gustavsson *et al.*, 2017). These authors found that for a small-scale fishing community interviewed in North Wales, licences limit the spatial extent of fishing effort. However, fishers' spatial distribution appears to be based on social relations between fishers, who know and respect the historical fishing grounds of each others. Socio-economic literature has advanced many theories about the behaviour of human agents (see van der Bergh *et al.*, 2000; Levine *et al.*, 2015), and fleet-dynamics models could benefit from some of these (Schlüter *et al.*, 2017). Our DSVM model could for instance fit in a MoHuB (Modelling Human Behavior) framework as depicted in Figure 5.1 (see Table S5.1 for description). Our model represents in fact a patchwork of different theories: Rational choice theory for the complete knowledge by fishers and the use of utility with tradition, Prospect Theory for the probabilities given to revenue according to choices, and Habitual/Reinforcement learning for the weights given to choices according to the satisfaction of past average choices. However, different aspects could be added or a least prospected, by building on some of the theories reviewed in Schlüter *et al*. (2017), and I provide a few examples below. The Bounded Rationality theory could be considered to limit information available to fishers (knowledge of species distribution), maybe geographically to a certain extent or in relation to empirical past fishing effort. Planned Behaviour theory could be considered by allotting a belief parameter to each fisher determining his attitudes against compliance. Habitual/Reinforcement learning could be strengthened by giving penalties to fishing tactics which have been unsuccessful. Descriptive Norm could introduce a relationship of conformism or non-conformism driving each fisher's choices toward other fishers' behaviour in a particular context, e.g., comply or not to a new regulation. Finally, the Prospect Theory could give more or less weight to extreme probability values in fishers' perception. As pointed out by the authors, these are just theories and the implementation belongs to modeller's choices, which could be done in two different manners. The first one would be to set parameters values according to literature, similar to what was done in this thesis for the tradition weight using meta-analysis from Girardin *et al*. (2017). The other approach would be to explore a panel of different values and their respective impact on fishers' behaviour and perhaps *in fine* on fish community.

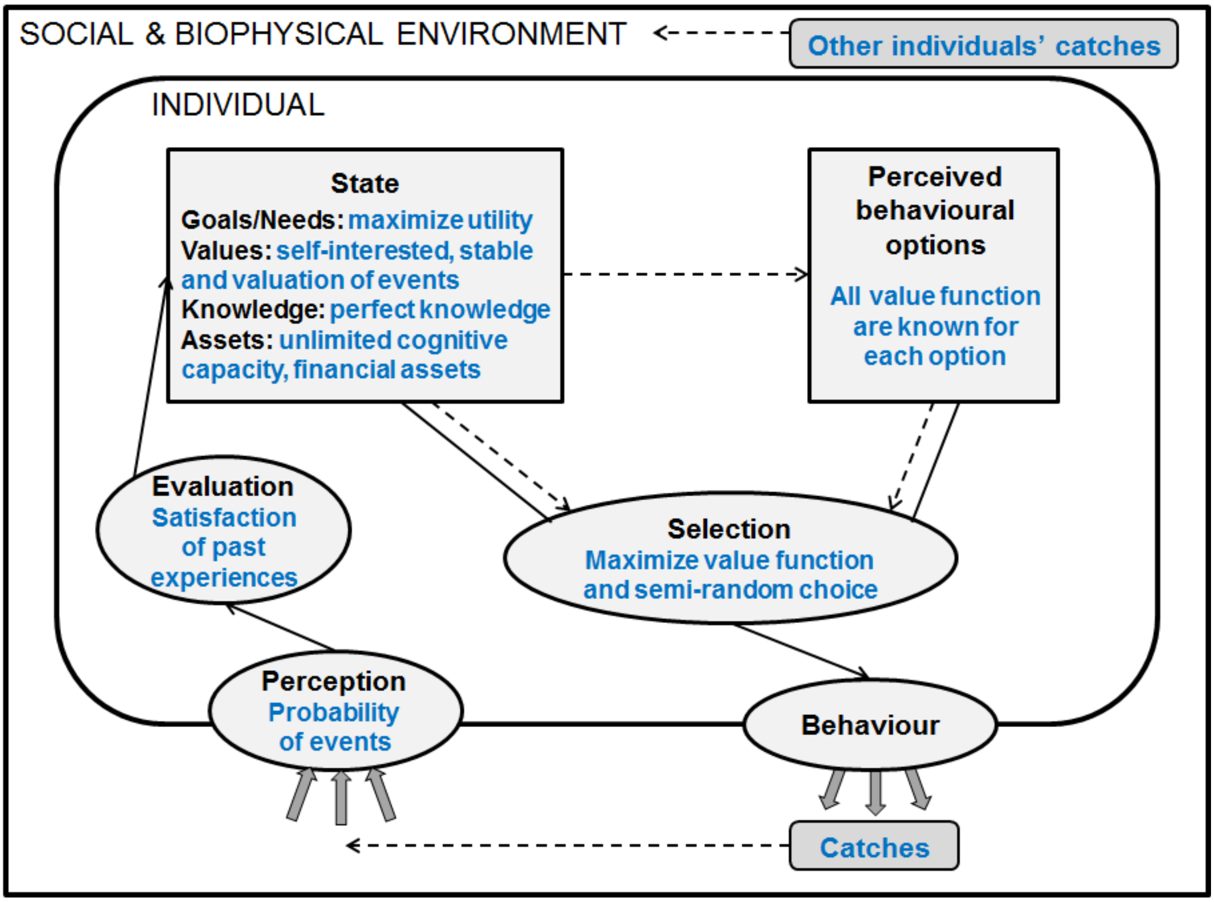

Figure 5.1. Representation of the DSVM model used in this thesis following the MoHuB framework (adapted from Schlüter *et al*., 2017; see Table S5.1 for description).

Involving more fishers in fisheries science is an indispensable step to successfully implementing the EAF (Pastoors *et al*., 2007; Pita *et al*., 2010). Fishers should be encouraged to participate more to the validation of results, e.g., by bringing their views on the outcomes of fleet-dynamics models, because their presence at sea can indeed bring valuable insights (Rochet *et al*., 2008). The involvement of fishers in the science process (e.g. Macdonald *et al.*, 2014) could have several merits including: i) increasing our knowledge of commercial species spatio-temporal distributions, ii) better understanding fishers' behaviour by, e.g., quantifying fishers' perception of catch opportunities (e.g. Abernethy *et al*., 2007; Holland, 2008), and thus reducing the uncertainty of their reaction and iii) involving fishers in the development of complex models. Given that complex system models are maybe easier to discuss with stakeholders than statistical ones due to their process representation (Lehuta *et al*., 2016), they are probably the best tool to interact with fishers. Such work would require a strengthened collaboration between fisheries modellers and social scientists.

There is on-going work aiming at implementing fisheries dynamics into OSMOSE and other models, and the relevance of using such a heavy time-consuming model as DSVM instead of another fleet-dynamics model, is clearly dependent on the number of fisheries to be considered and on the scientific question to be addressed. For instance, building in a large number of fleets and fisheries could be performed using ISIS-fish and relatively simple fleet dynamics models (Lehuta *et al.*, 2015), while the OSMOSE-DSVM model developed during this thesis was not designed to include a lot of different fleets, mainly due to computation limitations. The fleet-dynamics model selected should also be spatially adapted to fisheries. Indeed small-scale fisheries display smaller fishing dispersion, which may require a different scaling to that used in this thesis to fit the spatial distribution of large exclusive bottom trawlers.

With explicit fleet-dynamics being implemented, OSMOSE-DSVM offers now the potential to test the benefits of a wide range of management strategies, and notably newly developed ICES mixed-fisheries advice based on the Fcube approach (Fleet and Fisheries Forecast; Ulrich *et al.*, 2011; Iriondo *et al.*, 2012), target/limit reference points for mixed fisheries harvesting trophically-interacting species (Pascoe *et al.*, 2015), alternate management tools such as Individual Transferable Quotas (e.g., Squires *et al.*, 1998; Branch, 2009; Little *et al.*, 2009) and *in fine* Management Strategy Evaluation frameworks (e.g., Mapstone *et al.*, 2008; Thorpe *et al.*, 2016; Ono *et al.*, 2017). However, such work could require different adaptations in the code, and also probably increase the computation time. One of the perspectives in the landing obligation context would be to test the effects of higher selectivity of gears, in combination with avoidance behaviour, similar to earlier work carried out by O’Neill and Noble (2015; see Figure S5.1) for plaice or Madsen (2007; see Figure S5.2) for cod.

‘*Though it would be allowed, that reason may form very plausible conjectures with regard to the consequences of such a particular conduct in such particular circumstances; it is still supposed imperfect, without the assistance of experience, which is alone able to give stability and certainty to the maxims, derived from study and reflection.*’ (Hume, 1748).

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Supplementary material

Tables

Table S2.1. Parameters chosen and their respective percentage of deviance explained in the Delta-GLM applied to survey CPUEs with spatio-temporal interaction parameter forced for the 19 species of the study. MEV: Moran's Eigenvectors. * represents a significant effect of the spatio-temporal interaction parameter.

presence / absence	*area*	*year*	*sediment*	*area-by-year*	*MEV*
Chelidonichthys cuculus	33.6	2.1	5.4	0.0	11.4
Chelidonichthys lucerna	11.2			0.0	0.9
Dicentrarchus labrax	14.1	2.2	1.0	0.0	4.3
Gadus morhua	12.7	6.8		0.0	2.1
Limanda limanda	26.2	1.5	4.6	23.7	14.7
Loligo spp.	19.3	6.1	1.9	37.6	1.0
Merlangius merlangus	28.8		2.0	0.0	7.9
Microstomus kitt	22.1	4.0		0.0	4.5
Mullus surmuletus	11.3	10.1	2.4	0.0	3.9
Mustelus asterias	25.4	3.1		0.0	3.4
Platichthys flesus	19.0	4.1	3.8	0.0	13.5
Pleuronectes platessa	23.9	2.0	3.2	0.0	4.6
Raja clavata	22.7	2.2	0.9	0.0	3.0
Scyliorhinus canicula	33.0		1.3	30.1	9.3
Sepia officinalis	17.3	6.2	2.3	32.4	3.8
Spondyliosoma cantharus	29.9		2.5	30.7	10.0
Trisopterus luscus	19.5	2.8		0.0	1.2
Trisopterus minutus	16.9	1.6	1.0	0.0	3.5
Zeus faber	6.9	4.2	1.1	0.0	1.6
abundance	*area*	*year*	*sediment*	*area-by-year*	*MEV*
Chelidonichthys cuculus	20.3	2.7	1.9	38.0	5.9
Chelidonichthys lucerna	16.6			51.9	
Dicentrarchus labrax	20.5	3.5	1.4	39.0	4.7
Gadus morhua	23.2	13.1		38.5	
Limanda limanda	23.6	8.0	2.5	27.3	9.6
Loligo spp.	13.7	13.5		38.9 *	
Merlangius merlangus	25.1	4.3	2.4	29.8 *	15.6
Microstomus kitt	21.4	9.0	2.9	36.4	9.9

Mullus surmuletus	19.4	12.6		37.0	
Mustelus asterias	29.5	5.2	1.7	49.9 *	
Platichthys flesus	9.3			35.4	25.1
Pleuronectes platessa	19.9	3.5	5.7	22.2	22.7
Raja clavata	25.4	12.7		39.9	6.5
Scyliorhinus canicula	21.7	1.9	1.4	37.9 *	13.8
Sepia officinalis	26.6	6.7	3.6	30.7	
Spondyliosoma cantharus	26.5	8.4	1.5	34.8	
Trisopterus luscus	23.2	12.2		34.1	
Trisopterus minutus	21.3	5.3		50.9 *	5.1
Zeus faber	12.7	7.5		49.0	8.3

Table S2.2. Akaike information criterion (AIC) values and number of parameters (Nparam) for models with and without spatio-temporal interaction parameter. * represents the lowest AIC value. / denotes models with convergence failure. For those models which have a lower AIC with the interaction parameter, Local Index of Collocation (LIC) and significance of the overlap between models with and without interactions (obtained with 5000 random permutations) were added.

	Model	AIC with interactions	Nparam with interactions	AIC without interactions	Nparam without interactions	LIC	significance
Chelidonichthys cuculus	P / A	/	833	1234.33 *	88		
	> 0	6787.64	823	6527.23 *	69		
Chelidonichthys lucerna	P / A	/	783	1693.47 *	46		
	> 0	1883.13	782	1810.30 *	46		
Dicentrarchus labrax	P / A	/	879	1793.21 *	74		
	> 0	3380.31	872	3273.02 *	74		
Gadus morhua	P / A	/	819	1588.88 *	66		
	> 0	2926.03	816	2847.21 *	64		
Limanda limanda	P / A	1432.83	525	995.02 *	63		
	> 0	5600.96	517	5485.70 *	54		
Loligo spp.	P / A	1329.26	525	928.16 *	68		
	> 0	6207.94	520	6088.82 *	61		
Merlangius merlangus	P / A	/	764	1355.32 *	60		
	> 0	6166.69	755	6113.31 *	67		
Microstomus kitt	P / A	/	584	949.17 *	53		
	> 0	1641.15	584	1631.07 *	56		
Mullus surmuletus	P / A	/	873	1796.35 *	73		
	> 0	5544.09	867	5355.76 *	67		
Mustelus asterias	P / A	/	824	1342.01 *	73	0.61	0.02
	> 0	2414.01 *	820	2533.45	68		
Platichthys flesus	P / A	/	264	613.10 *	43		
	> 0	1161.68	256	1106.73 *	16		
Pleuronectes platessa	P / A	/	603	1233.55 *	67		
	> 0	6408.38	609	6236.19 *	65		
Raja clavata	P / A	/	853	1487.87 *	73	0.78	<0.0002
	> 0	2304.46 *	852	2330.36	68		
Scyliorhinus canicula	P / A	2125.25	882	1382.85 *	69		
	> 0	8419.57	881	8326.62 *	81		
Sepia officinalis	P / A	1309.59	457	989.01 *	61		
	> 0	1967.02	454	1855.43 *	59		
Spondyliosoma cantharus	P / A	2112.75	831	1409.78 *	68		
	> 0	8038.94	820	7800.40 *	69		

Trisopterus luscus	P / A	/	648	972.28 *	54	0.49	0.04
	> 0	8081.56	646	7812.70 *	55		
Trisopterus minutus	P / A	/	872	1750.77 *	78		
	> 0	5801.93 *	869	5872.73	69		
Zeus faber	P / A	/	801	697.03 *	69		
	> 0	1540.08	800	1503.07 *	64		

Table S2.3. Parameters chosen and their respective percentage of deviance explained in the Delta-GLM applied to commercial CPUEs for the 19 species of the study. MEV: Moran's Eigenvectors. * represents *area-by-month* replaced by *month* alone in the model.

presence / absence	*area-by-month*	*year*	*gear-by-length*	*sediment*	*MEV*
Chelidonichthys cuculus	15.5	2.2	0.5	3.9	1.8
Chelidonichthys lucerna	10.6	1.0	0.9	1.0	3.1
Dicentrarchus labrax	15.4	1.2	1.0	3.1	3.4
Gadus morhua	17.1	6.1	1.0	0.7	4.0
Limanda limanda	11.1	1.5	0.7	0.9	2.8
Loligo spp.	14.3	10.4	0.3	1.2	1.7
Merlangius merlangus	17.6	0.9	0.9	0.4	3.3
Microstomus kitt	15.0	2.1	0.3	1.1	4.0
Mullus surmuletus	14.4	2.3	1.7	0.9	2.3
Mustelus asterias	4.6 *	1.5	1.0	2.8	8.3
Platichthys flesus	5.1 *	1.5	0.5	1.9	7.8
Pleuronectes platessa	7.9	0.6	1.3	0.9	2.4
Raja clavata	14.8	1.0	0.2	1.8	4.8
Scyliorhinus canicula	15.4	3.5	0.8	1.7	3.3
Sepia officinalis	17.5	3.6	1.1		4.0
Spondyliosoma cantharus	16.9	2.5	1.2	2.9	2.9
Trisopterus luscus	9.8	0.9	1.7	1.3	2.7
Trisopterus minutus	17.4	6.4	2.2	2.4	6.2
Zeus faber	2.3 *	1.5	5.2	2.7	4.4

abundance	*area-by-month*	*year*	*gear-by-length*	*sediment*	*MEV*
Chelidonichthys cuculus	37.2	6.5	13.3	10.5	11.8
Chelidonichthys lucerna	33.1	7.3	16.4	0.8	20.8
Dicentrarchus labrax	45.9	2.7	6.7	4.8	17.9
Gadus morhua	32.6	5.4	9.5	1.5	27.1
Limanda limanda	26.3	4.8	13.8	0.4	26.0
Loligo spp.	49.0	10.1	1.0	8.9	12.9
Merlangius merlangus	41.0	6.5	12.7	0.5	23.8
Microstomus kitt	40.6	6.3	8.4	2.5	26.4
Mullus surmuletus	40.0	8.3	12.1	1.8	19.2
Mustelus asterias	50.7	11.8	8.7	12.1	7.6
Platichthys flesus	43.0	6.5	10.0	3.3	22.5
Pleuronectes platessa	21.4	3.8	17.0	0.3	22.3
Raja clavata	46.2	10.4	7.9	7.5	5.3
Scyliorhinus canicula	46.8	4.2	6.6	10.2	6.1
Sepia officinalis	55.5	11.6	6.4	9.4	5.2
Spondyliosoma cantharus	36.3	2.1	12.2	5.2	23.9

Trisopterus luscus	33.2	5.4	9.8	3.3	29.3
Trisopterus minutus	55.8	2.7	5.4	2.6	6.8
Zeus faber	49.4	7.6	10.7	7.1	

Table S2.4. Parameters chosen and their respective percentage of deviance explained in the Delta-GLM applied to survey CPUEs for the 19 species of the study. MEV: Moran's Eigenvectors.

presence / absence	*area*	*year*	*sediment*	*MEV*
Chelidonichthys cuculus	33.4	2.3	5.2	13.2
Chelidonichthys lucerna	11.8			
Dicentrarchus labrax	15.7	2.1	1.0	1.4
Gadus morhua	13.3	6.8		1.3
Limanda limanda	25.4	1.5	4.3	16.2
Loligo spp.	18.4	6.1	1.7	2.5
Merlangius merlangus	28.8		2.0	7.9
Microstomus kitt	22.2	4.1		4.6
Mullus surmuletus	13.2	10.1	2.1	1.8
Mustelus asterias	23.5	3.1		6.0
Platichthys flesus	19.5	4.0	3.7	12.6
Pleuronectes platessa	23.6	2.2	1.8	8.7
Raja clavata	22.7	2.2	0.9	3.0
Scyliorhinus canicula	33.3		1.3	9.7
Sepia officinalis	17.3	6.2	2.3	3.8
Spondyliosoma cantharus	30.5		2.5	10.9
Trisopterus luscus	18.4	2.7		
Trisopterus minutus	15.9	1.5	1.0	4.7
Zeus faber	6.9	4.2	1.1	1.6

abundance	*area*	*year*	*sediment*	*MEV*
Chelidonichthys cuculus	21.7	3.0	1.6	3.5
Chelidonichthys lucerna	16.6			
Dicentrarchus labrax	18.4	3.9	1.4	8.7
Gadus morhua	23.2	13.1		
Limanda limanda	22.3	7.5	2.7	11.6
Loligo spp.	13.1	14.0		
Merlangius merlangus	25.1	4.3	2.4	15.6
Microstomus kitt	22.0	8.5	2.9	9.9
Mullus surmuletus	19.4	12.6		
Mustelus asterias	28.8	5.9	1.7	
Platichthys flesus	9.3			25.1
Pleuronectes platessa	19.9	3.5	5.7	22.7
Raja clavata	27.8	10.4		6.5
Scyliorhinus canicula	21.4	2.2	1.4	13.8
Sepia officinalis	23.8	6.9	3.3	3.6
Spondyliosoma cantharus	26.4	7.5	1.2	1.9

Trisopterus luscus	23.2	12.2	8.6
Trisopterus minutus	22.7	3.9	5.1
Zeus faber	13.4	6.8	8.3

Table S3.1. Mean percentages of Landings owing to French bottom otter trawlers (OTB) with a vessel length above 18m and a mesh size (Φ) above 80mm on i) French OTB with a vessel length above 18m and ii) all French OTB for nine commercial species of the Eastern English Channel (2008-2014 average).

	OTB ≥ 18m & Φ > 80mm / OTB ≥ 18m	OTB ≥ 18m & Φ 80mm / OTB
Cod	95.7	82.1
Seabass	95.4	69.0
Red mullet	95.0	69.8
Whiting	95.6	87.0
Cuttlefish	95.2	63.7
Black seabream	96.0	73.3
Plaice	95.2	48.9
Mackerel	83.9	55.5
Squids	96.0	86.3

Table S3.2. Mean price (€ / kg) of the main commercial species in October in the Eastern English Channel in the period 2008-2014.

	2008	2009	2010	2011	2012	2013	2014
Cod	3.37	2.53	3.56	4.14	4.84	3.58	2.15
Cuttlefish	2.12	2.20	2.81	3.70	2.40	2.83	3.02
Mackerel	0.93	1.30	1.14	1.81	1.49	1.32	1.40
Red mullet	4.53	4.24	3.47	3.50	5.87	7.38	2.85
Squids	4.76	3.14	3.86	5.76	5.12	5.90	4.09
Whiting	1.65	1.61	1.32	1.83	1.85	1.65	1.59

Table S4.1. Input parameters of OSMOSE for the 14 fish species modelled explicitly. L_∞, K, and t_0 are the parameters of the von Bertalanffy growth model, with a linear growth before the threshold age a_{th} and a growth following the von Bertalanffy model after a_{th}; c is Fulton's condition factor and b the exponent of the L-W allometric relationship; L_{mat} is length at maturity and Φ is relative fecundity; a_{max} is longevity; F is the annual fishing mortality rate and L_{rec} is the recruitment size; M_{oth} is an additional mortality rate (resulting from predation by other species of the ecosystem that are not explicitly modelled); $M_{\xi\ max}$ is the maximum starvation mortality rate, M_L is the larval mortality rate applied to the first life stage; ξ_{crit} is the critical predation efficiency corresponding to maintenance requirements. Values reported in the table come from literature (references in Appendix 1) except from Moth, F and ML which come from calibration.

	GROWTH AND CONDITION						REPRODUCTION		SURVIVAL						PREDATION			
Species	L_∞	K	t_0	a_{th}	c	b	L_{mat}	Φ	a_{max}	F	L_{rec}	M_{oth}	$M_{\xi\ max}$	M_L	Min size ratio	Max size ratio	ξ_{crit}	max ingestion rate
	cm	*y^{-1}*	*y*	*y*	*$g.cm^{-3}$*		*cm*	*$eggs.g^{-1}$*	*y*	*y^{-1}*	*cm*	*y^{-1}*	*y^{-1}*	*$month^{-1}$*				*$g.g^{-1}$*
Lesser-spotted dogfish	87.4	0.118	-1.09	0.5	0.00308	3.029	57	0.14	10	0.103	39	0.018	0.3	2.45	50	3	0.57	3.5
Red mullet	53.3	0.18	-1.23	1	0.00716	3.178	16.7	500	11	0.984	14	0.063	0.3	3.60	125	10	0.57	3.5
Pouting	37.6	0.46	-0.77	0.5	0.00657	3.202	23	620	4	0.241	21	0.027	0.3	8.30	50	3.5	0.57	3.5
Whiting	40.2	0.63	-0.37	1	0.00621	3.103	20	797	20	0.379	21	0.043	0.3	9.63	30	1.5	0.57	3.5
Poor cod	22.2	0.462	-0.679	0.5	0.0092	3.026	13	100	3	0	13	0.413	0.3	1.86	50	3.5	0.57	3.5
Cod	103.9	0.19	-0.1	0.5	0.00835	3.053	56	800	25	0.330	21	0.054	0.3	10.61	50 / 20*	2.3 / 1.8*	0.57	3.5
Dragonet	28.3	0.471	-0.443	0.5	0.0262	2.442	17.4	255	6	0	21	0.112	0.3	0.14	125	10	0.57	3.5
Sole	37.3	0.35	-1.61	0.5	0.00391	3.264	29	482	20	0.516	25	0.092	0.3	2.68	125	10	0.57	3.5
Plaice	71.7	0.23	-0.83	0.5	0.0103	3.017	27	255	15	0.984	25	0.097	0.3	4.18	125	5	0.57	3.5
Horse mackerel	39.2	0.18	-1.515	1	0.0054	3.114	22	1655	15	0.117	12	0.017	0.3	0.38	100	2.5	0.57	3.5
Mackerel	42	0.24	-2.07	1	0.00338	3.241	29	1070	17	0.353	19	0.084	0.3	4.17	100	2.5	0.57	3.5
Herring	29.2	0.37	-0.67	0.5	0.00503	3.1	25	458	11	0.248	16	0.057	0.3	0.19	1000	5	0.57	3.5
Sardine	24.6	0.79	-0.22	0.5	0.00594	3.077	15	2228	15	0.386	11	0.035	0.3	3.62	1000	5	0.57	3.5
Squids	50	2	0.5	0.7	0.25	2.27	30	50	2	0.085	14	0.058	0.3	6.60	20	1.5	0.57	3.5
Cuttlefish	30.5	1.25	0	0.25	0.27	2.26	12.3	1.08	2	0.689	13	0.186	0.3	0.23	20	1.5	0.57	3.5

Table S5.1. Definition and specifications of the different elements of the MoHuB framework Definitions are adapted from the Merriam Webster dictionary (modified from de Schlüter *et al.*, 2017).

Context	
Social & biophysical environment	The environment the individual and her behaviour are embedded in
Structural elements	
State	The internal state of an individual physiological, psychological or material
State: knowledge	The information and understanding an individual has about her social-ecological environment and her own behaviour within this context
State: assets	Resources and other advantageous characteristics of an individual
State: values	Something (as a principle or quality) intrinsically valuable or desirable, i.e. not directly linked to the well-being of an individual or her motivational goals
Perceived behavioural options	The set of options the individual perceives and thus can choose from
Processes	
Perception	The process by which an individual senses the surrounding social and biophysical environment
Evaluation	The process by which an individual determines the significance, worth, or condition of the perceived state of the social and bio-physical environment
Selection	The process by which an individual chooses her behaviour from the set of perceived behavioural options taking its state into account, resulting in the executed behaviour
Behaviour	The behaviour that an individual executes as a result of the decision process

Figures

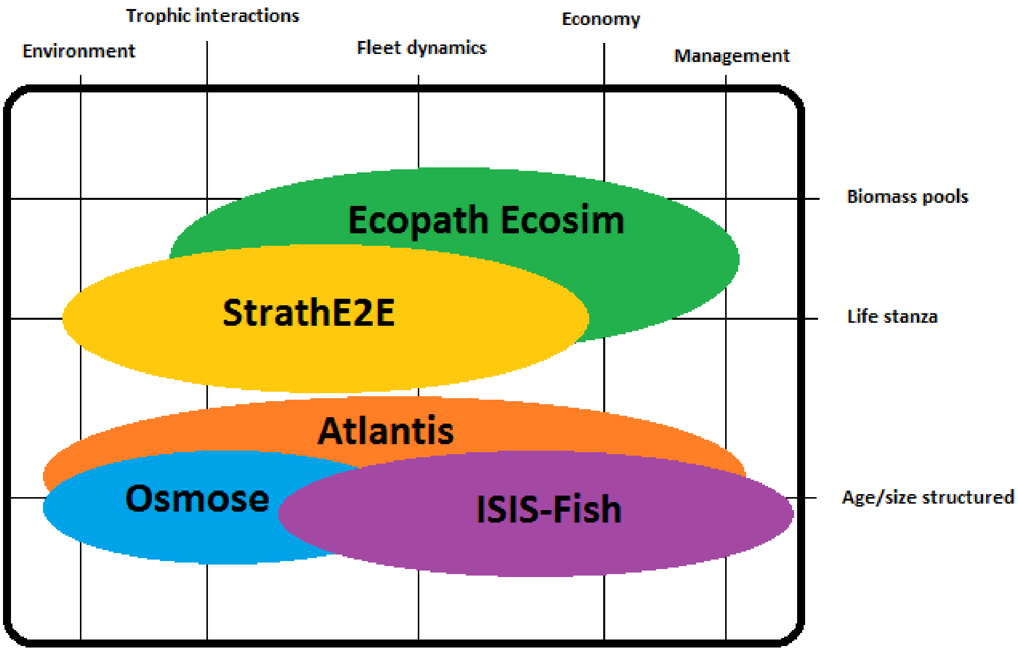
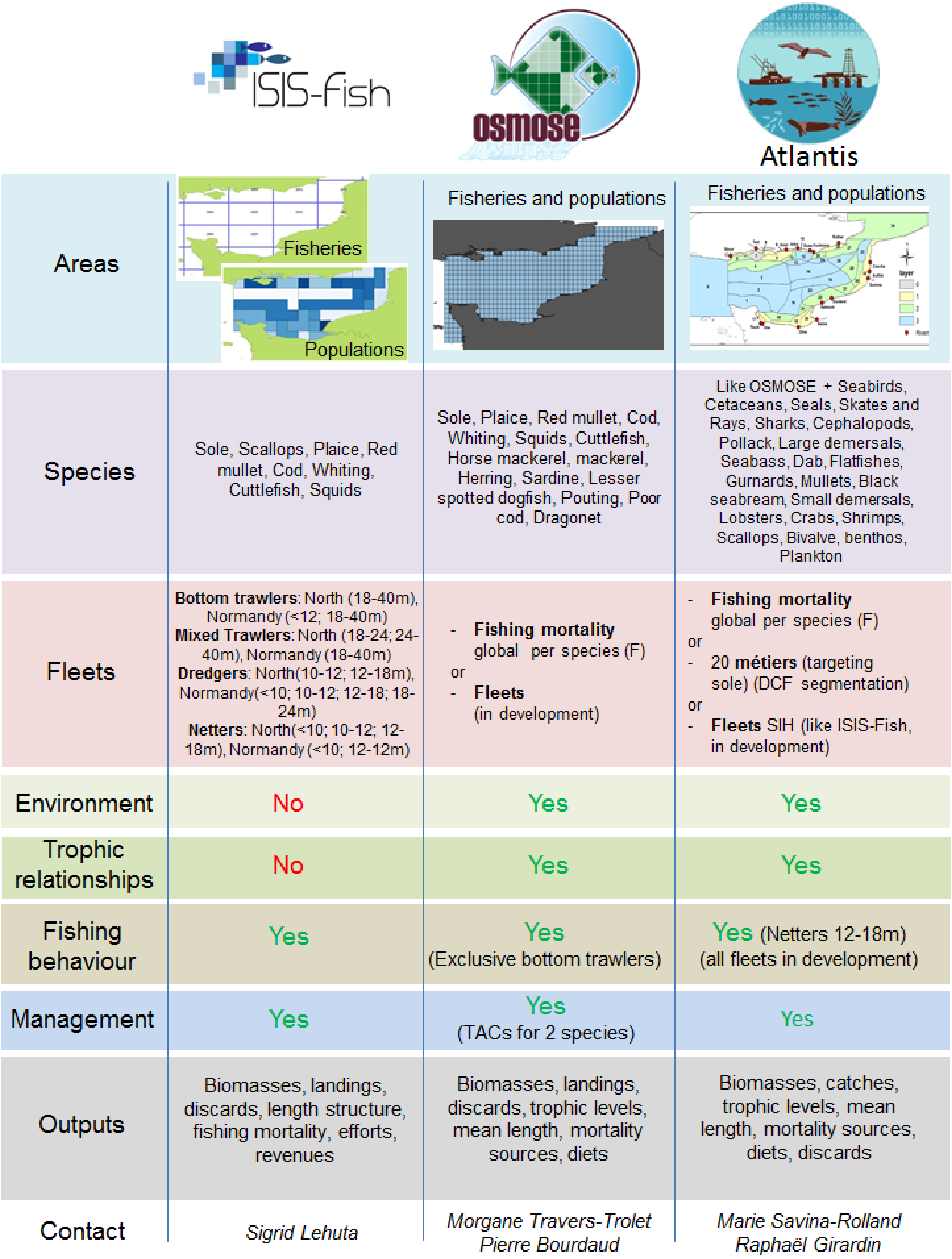
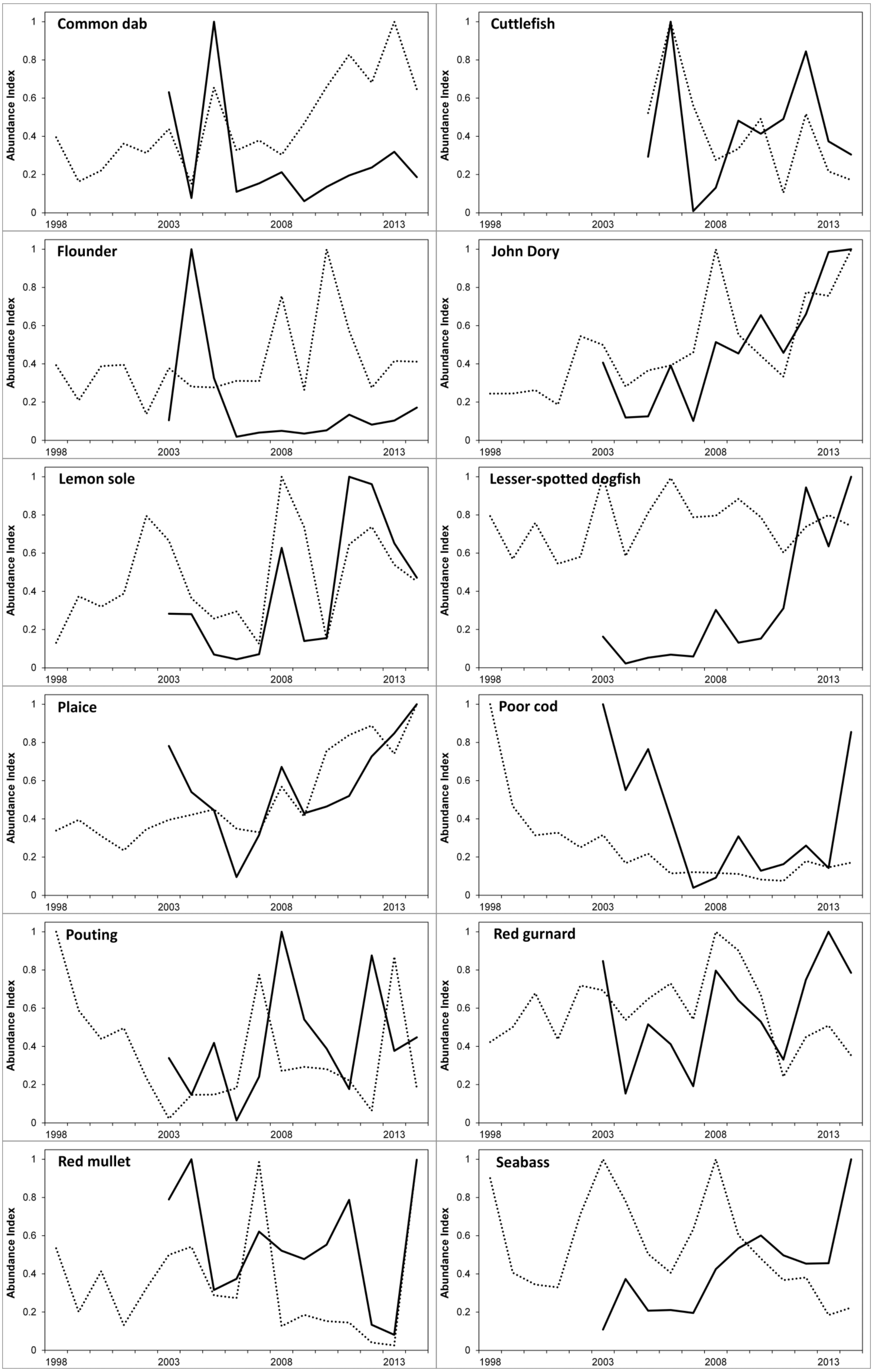


Figure S1.1. The five models used in DiscardLess and their range of complexity.



	ISIS-Fish	OSMOSE	Atlantis
Areas	Fisheries Populations	Fisheries and populations	Fisheries and populations
Species	Sole, Scallops, Plaice, Red mullet, Cod, Whiting, Cuttlefish, Squids	Sole, Plaice, Red mullet, Cod, Whiting, Squids, Cuttlefish, Horse mackerel, mackerel, Herring, Sardine, Lesser spotted dogfish, Pouting, Poor cod, Dragonet	Like OSMOSE + Seabirds, Cetaceans, Seals, Skates and Rays, Sharks, Cephalopods, Pollack, Large demersals, Seabass, Dab, Flatfishes, Gurnards, Mullets, Black seabream, Small demersals, Lobsters, Crabs, Shrimps, Scallops, Bivalve, benthos, Plankton
Fleets	**Bottom trawlers**: North (18-40m), Normandy (<12; 18-40m) **Mixed Trawlers**: North (18-24; 24-40m), Normandy (18-40m) **Dredgers**: North(10-12; 12-18m), Normandy(<10; 10-12; 12-18; 18-24m) **Netters**: North(<10; 10-12; 12-18m), Normandy (<10; 12-12m)	- **Fishing mortality** global per species (F) or - **Fleets** (in development)	- **Fishing mortality** global per species (F) or - 20 **métiers** (targeting sole) (DCF segmentation) or - **Fleets** SIH (like ISIS-Fish, in development)
Environment	No	Yes	Yes
Trophic relationships	No	Yes	Yes
Fishing behaviour	Yes	Yes (Exclusive bottom trawlers)	Yes (Netters 12-18m) (all fleets in development)
Management	Yes	Yes (TACs for 2 species)	Yes
Outputs	Biomasses, landings, discards, length structure, fishing mortality, efforts, revenues	Biomasses, landings, discards, trophic levels, mean length, mortality sources, diets	Biomasses, catches, trophic levels, mean length, mortality sources, diets, discards
Contact	*Sigrid Lehuta*	*Morgane Travers-Trolet* *Pierre Bourdaud*	*Marie Savina-Rolland* *Raphaël Girardin*

Figure S1.2. Comparison of ISIS-Fish, OSMOSE and Atlantis models of Eastern English Channel (source, Lehuta pers. comm.).


Common dab
Cuttlefish
Flounder
John Dory
Lemon sole
Lesser-spotted dogfish
Plaice
Poor cod
Pouting
Red gurnard
Red mullet
Seabass
Abundance Index
0
0.2
0.4
0.6
0.8
1
1998
2003
2008
2013

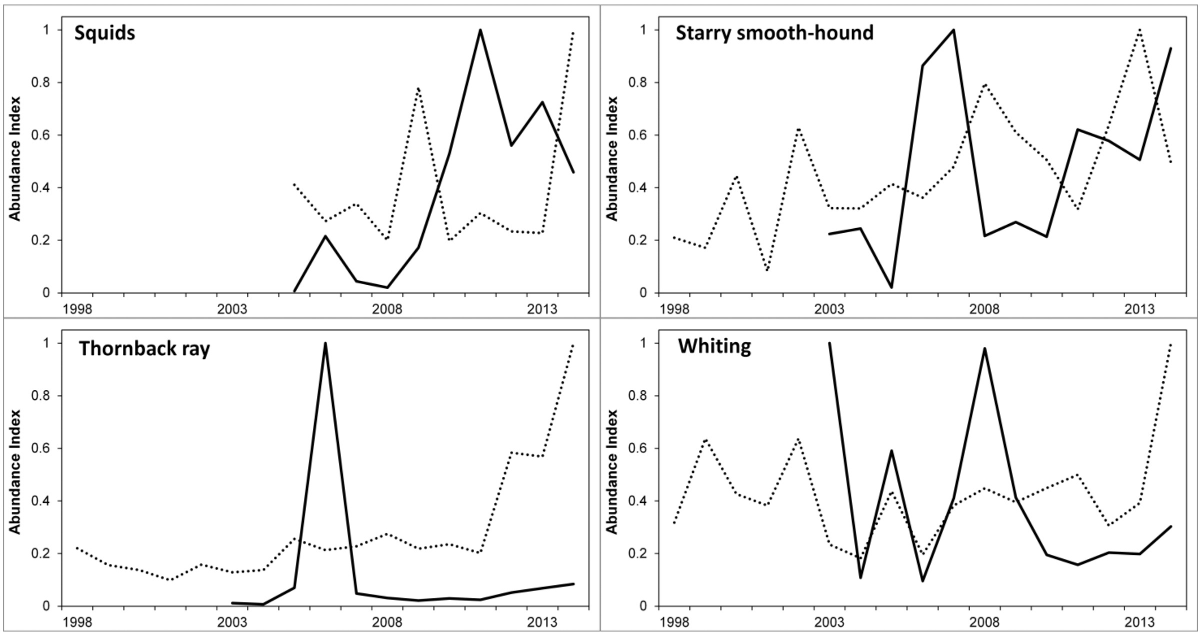


Figure S2.1. Annual abundance index estimated from Channel Ground Fish Survey (CGFS; dotted line) and OBSMER (solid line) for the 16 additional species.

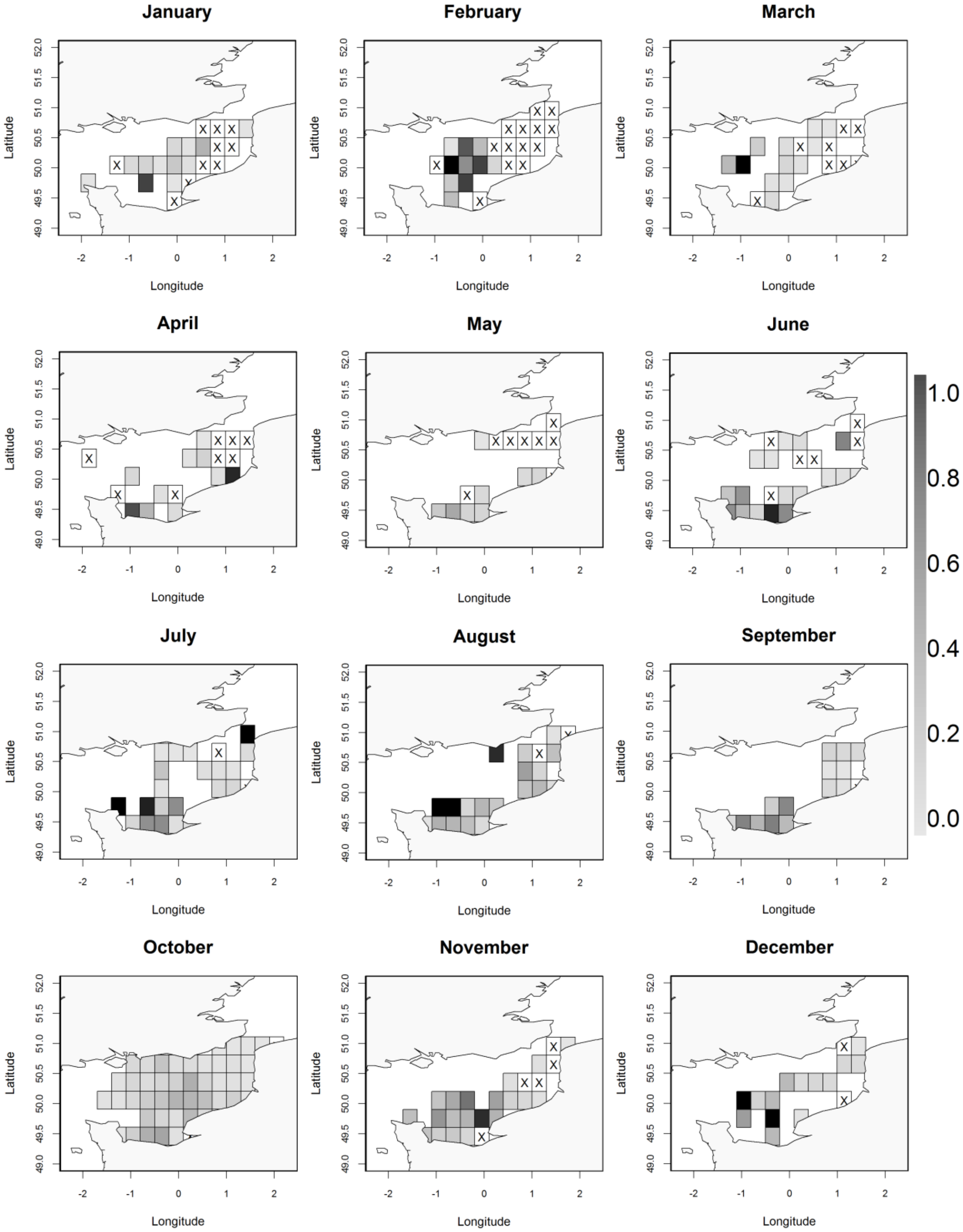


Figure S2.2. Monthly spatial abundance distribution estimated from OBSMER and CGFS for black seabream. 'X' represents areas where no individual of a species was ever fished during a month in the database.

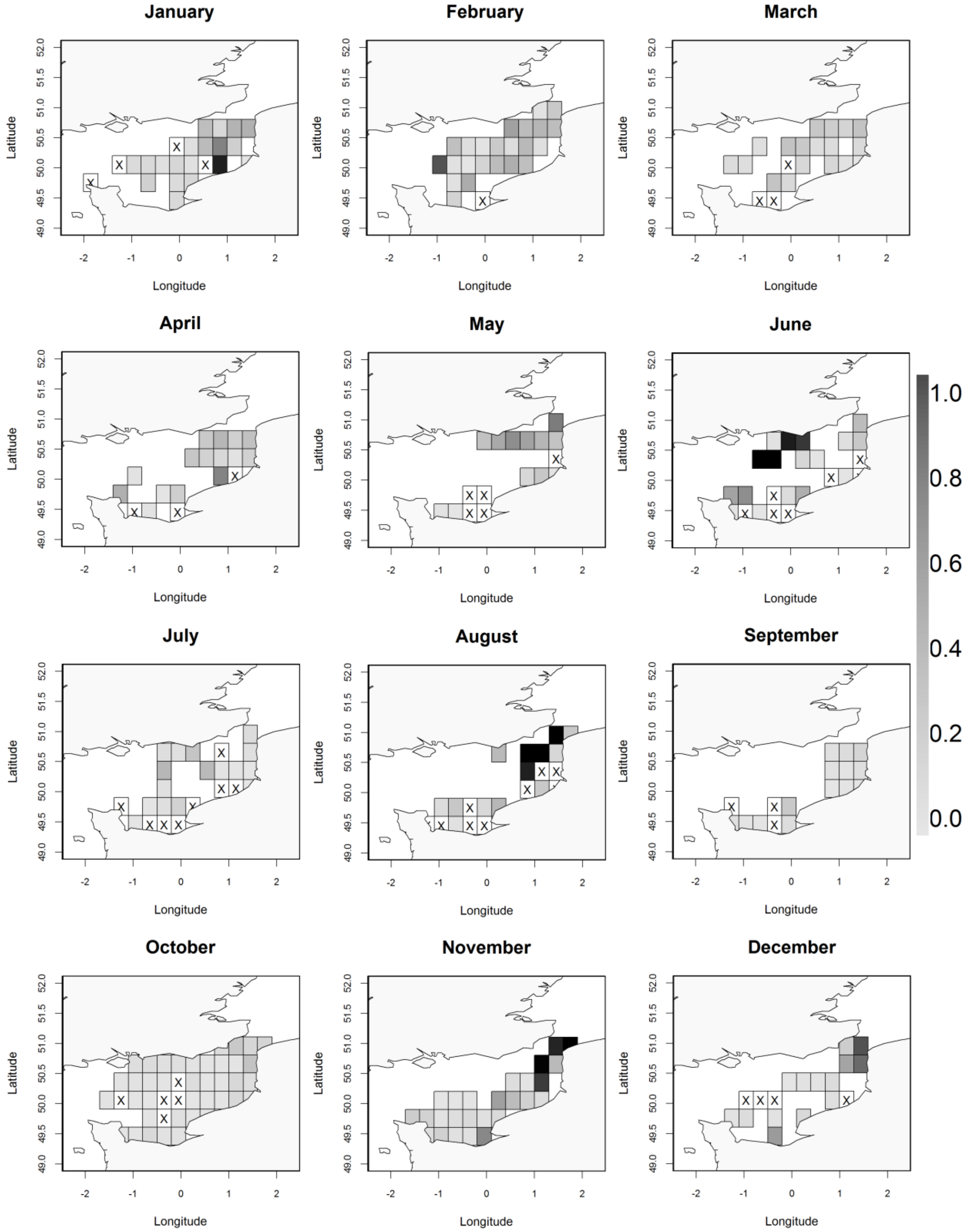


Figure S2.3. Monthly spatial abundance distribution estimated from OBSMER and CGFS for cod. ‘X’ represents areas where no individual of a species was ever fished during a month in the database.

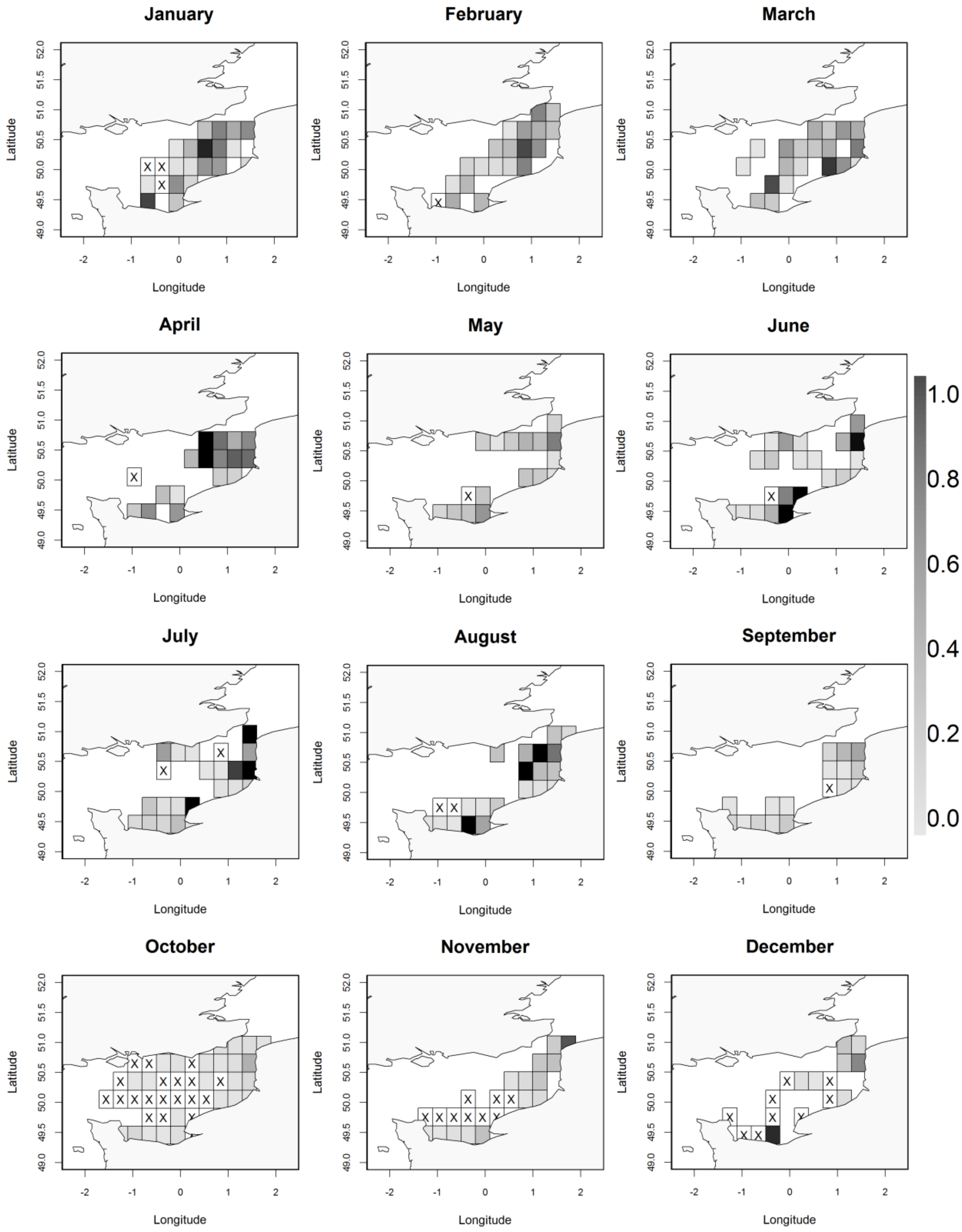


Figure S2.4. Monthly spatial abundance distribution estimated from OBSMER and CGFS for common dab. ‘X’ represents areas where no individual of a species was ever fished during a month in the database.

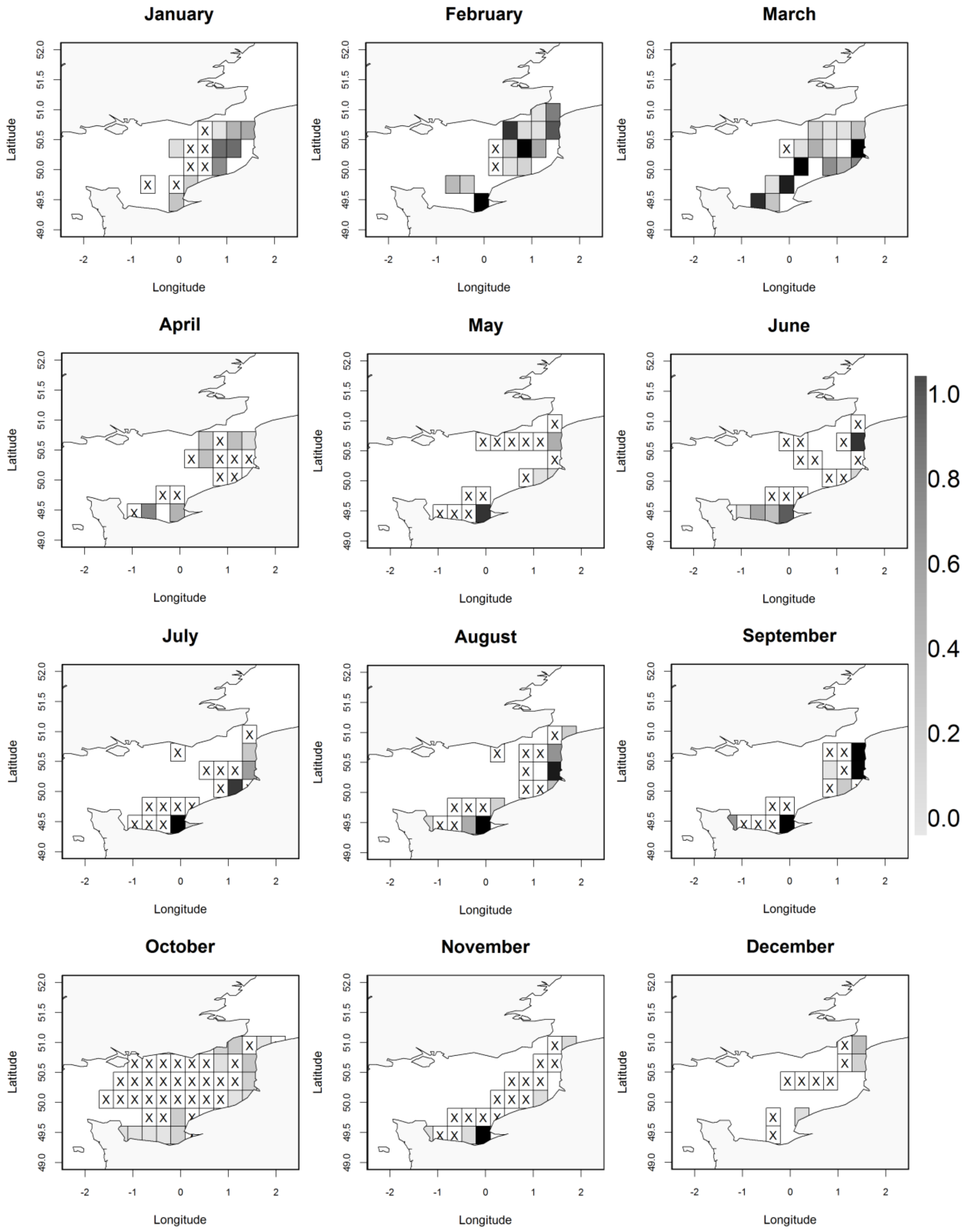


Figure S2.5. Monthly spatial abundance distribution estimated from OBSMER and CGFS for flounder. 'X' represents areas where no individual of a species was ever fished during a month in the database.

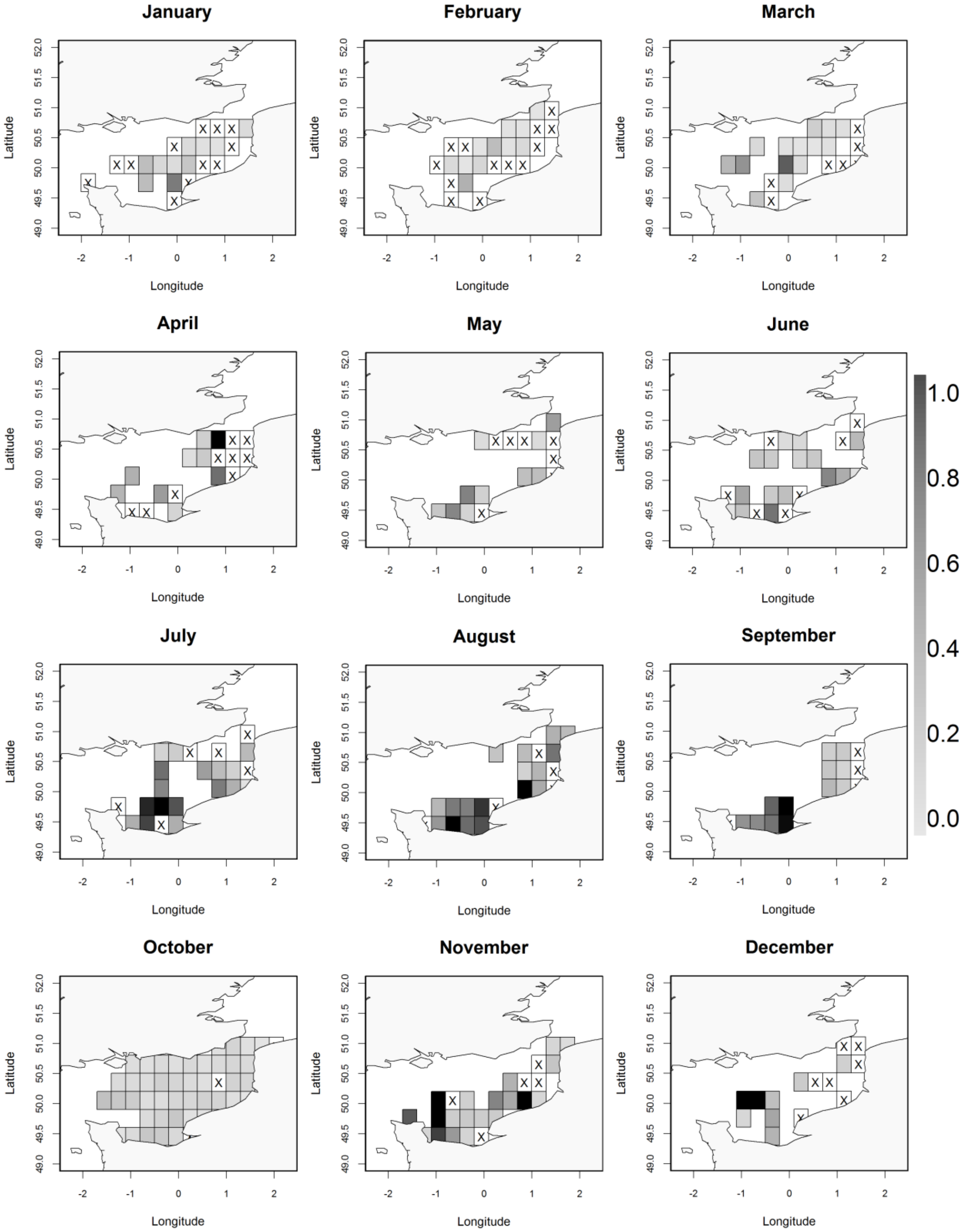


Figure S2.6. Monthly spatial abundance distribution estimated from OBSMER and CGFS for John Dory. ‘X’ represents areas where no individual of a species was ever fished during a month in the database.

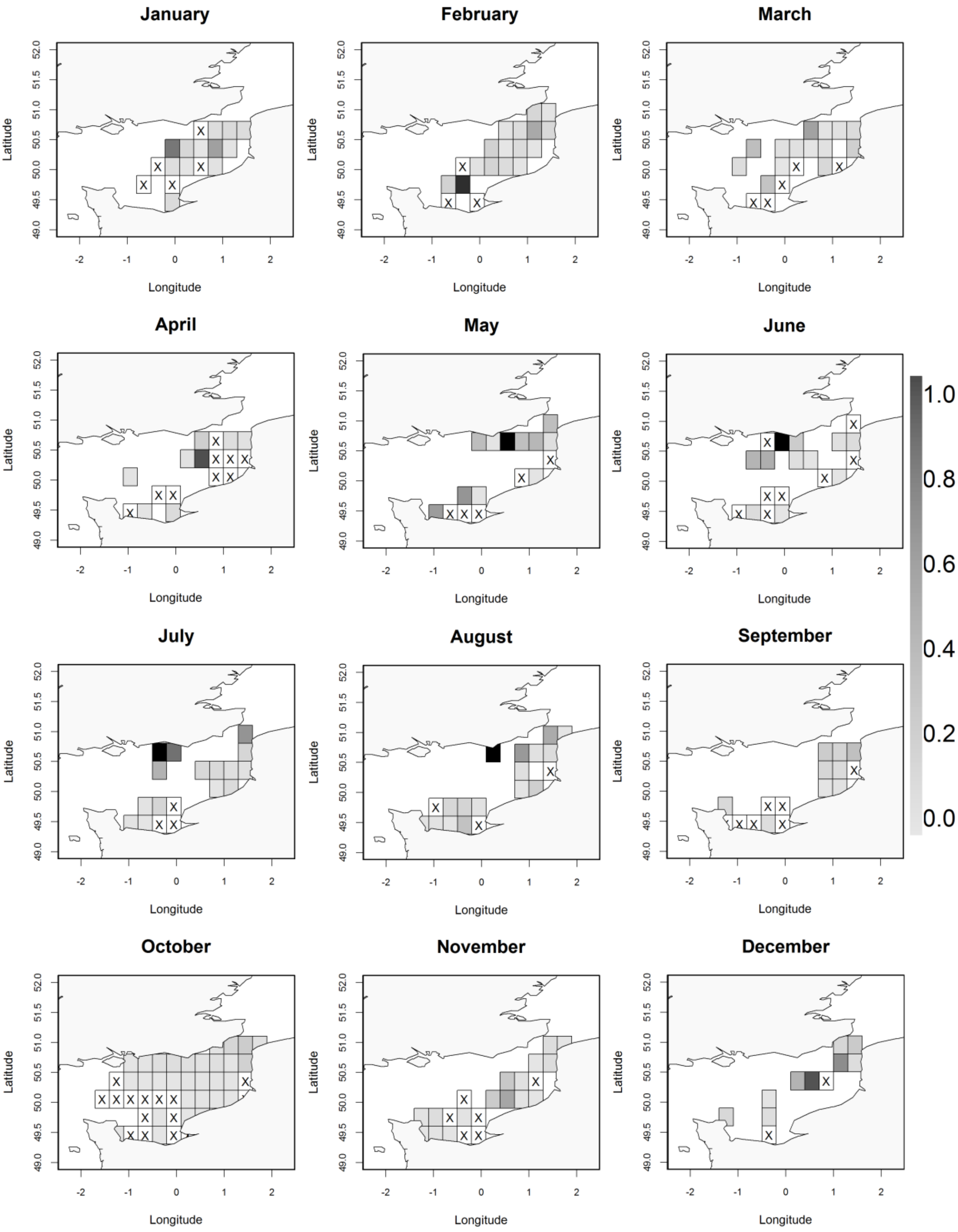


Figure S2.7. Monthly spatial abundance distribution estimated from OBSMER and CGFS for lemon sole. ‘X’ represents areas where no individual of a species was ever fished during a month in the database.

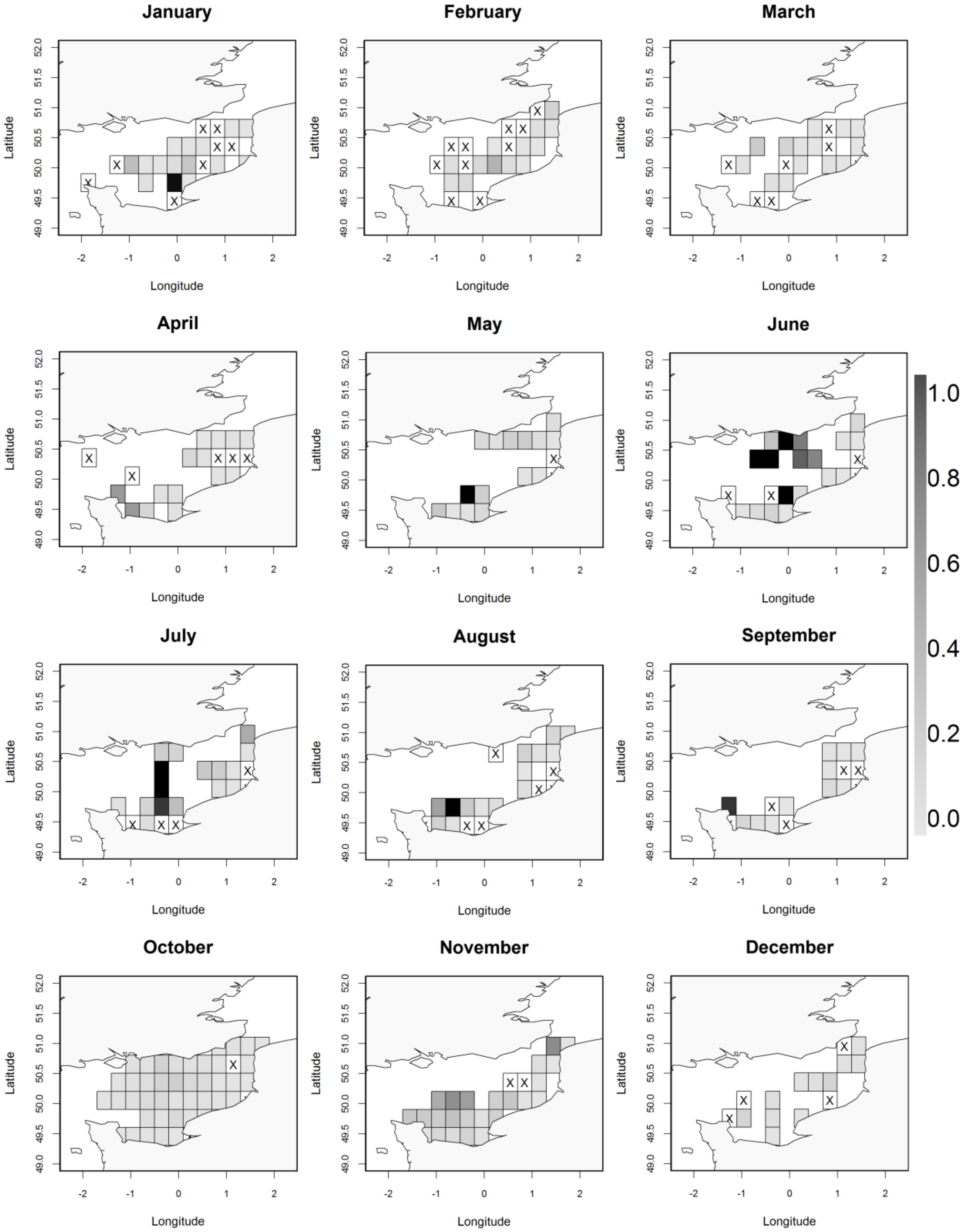


Figure S2.8. Monthly spatial abundance distribution estimated from OBSMER and CGFS for lesser-spotted dogfish. ‘X’ represents areas where no individual of a species was ever fished during a month in the database.

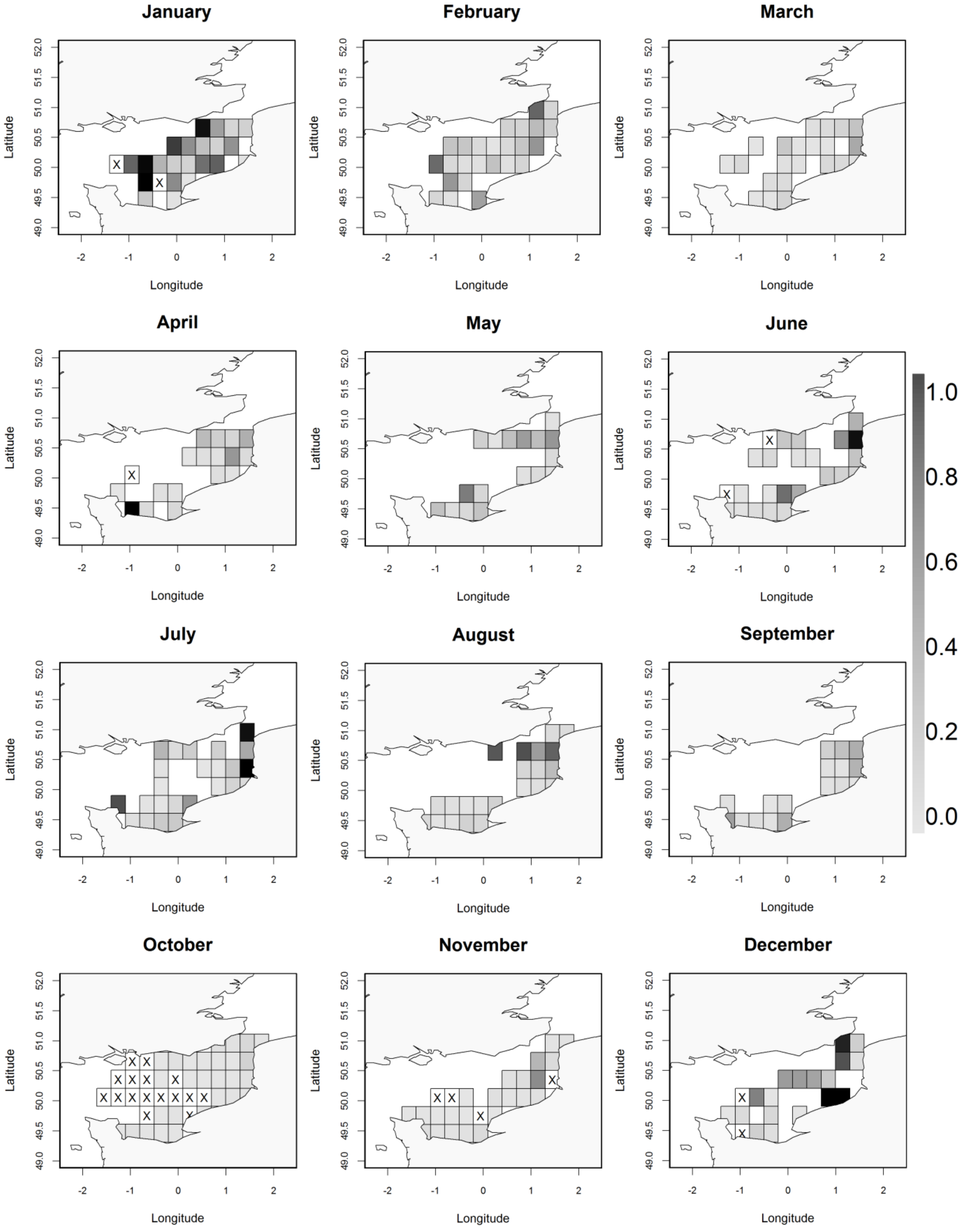


Figure S2.9. Monthly spatial abundance distribution estimated from OBSMER and CGFS for plaice. 'X' represents areas where no individual of a species was ever fished during a month in the database.

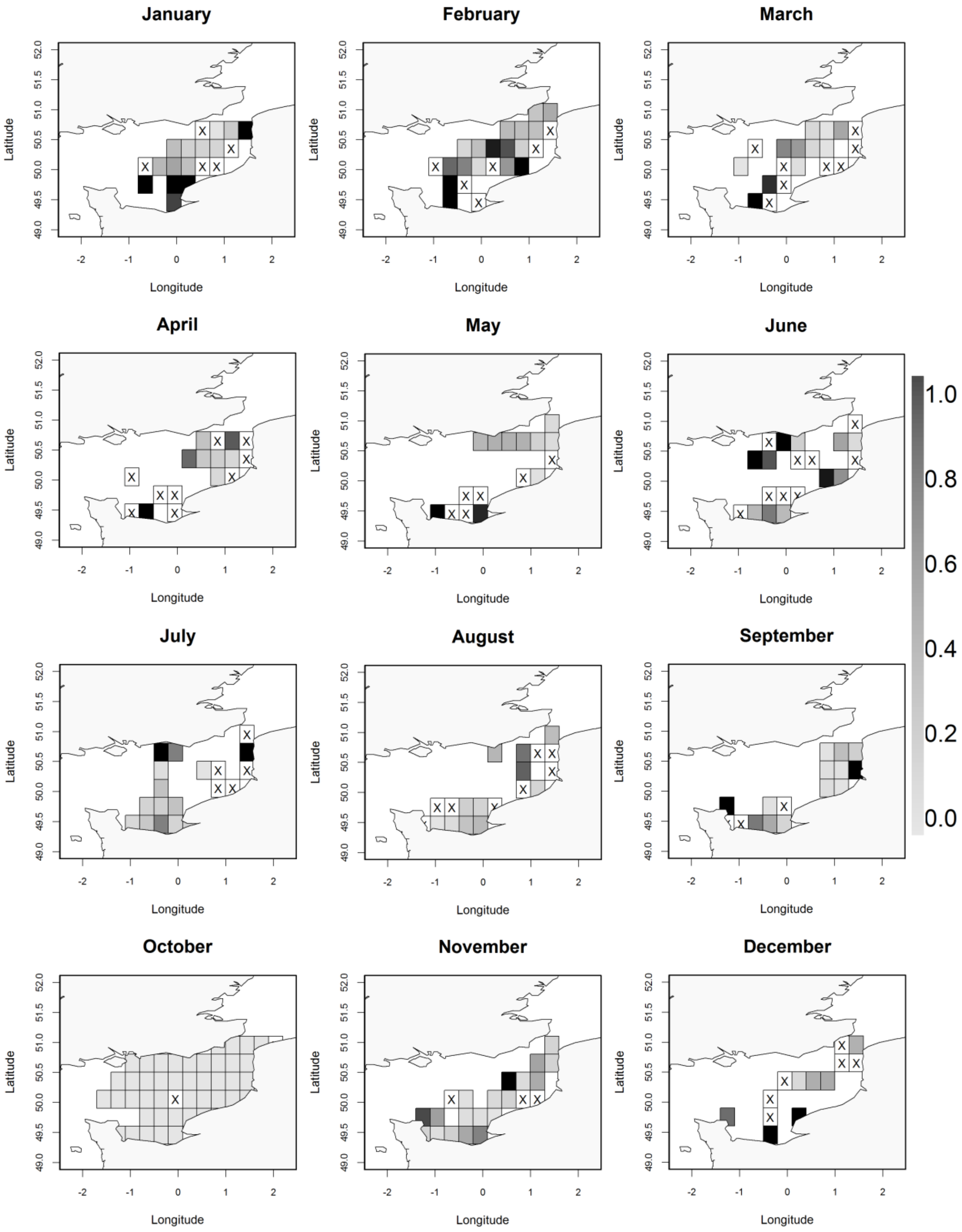


Figure S2.10. Monthly spatial abundance distribution estimated from OBSMER and CGFS for poor cod. ‘X’ represents areas where no individual of a species was ever fished during a month in the database.

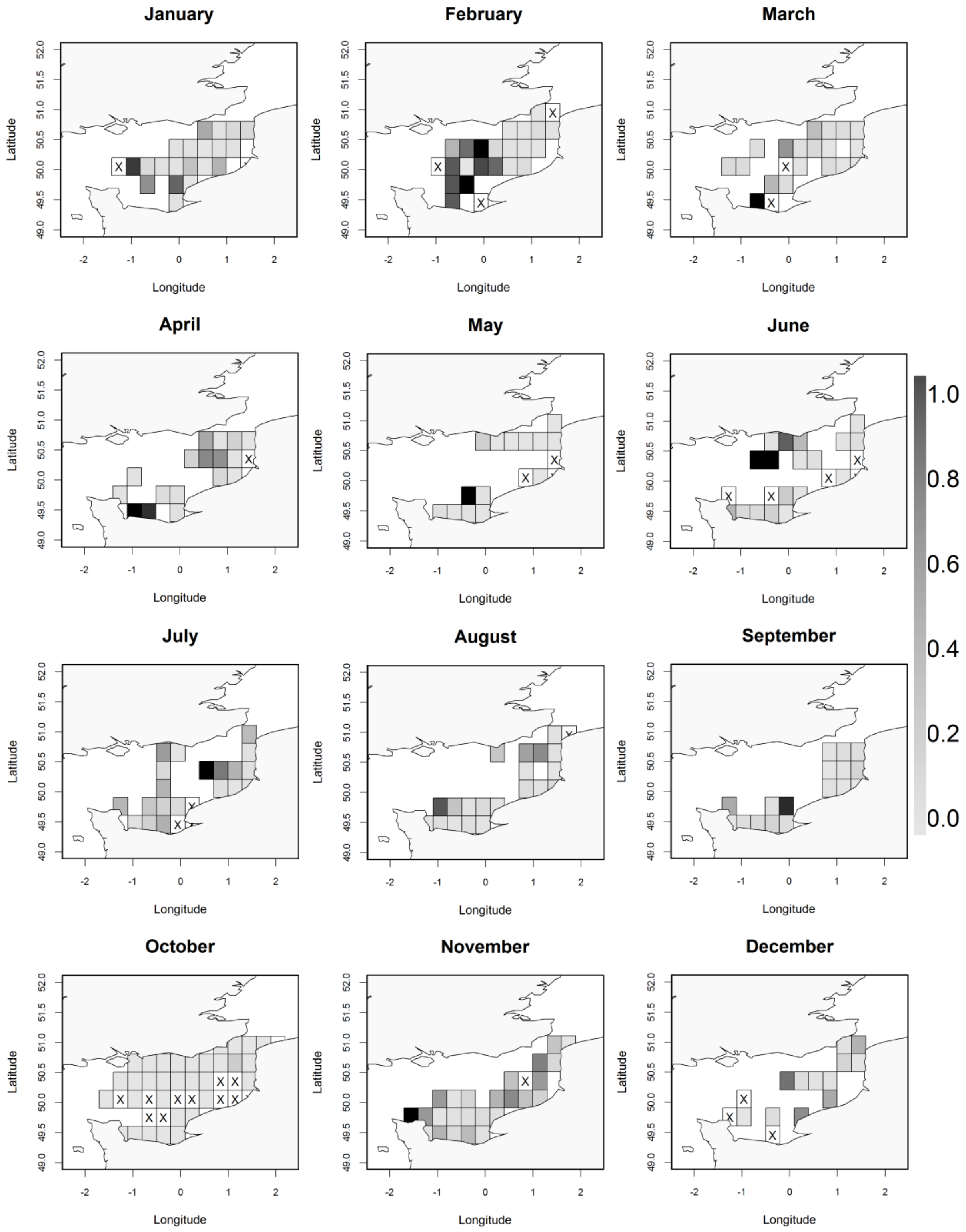


Figure S2.11. Monthly spatial abundance distribution estimated from OBSMER and CGFS for pouting. ‘X’ represents areas where no individual of a species was ever fished during a month in the database.

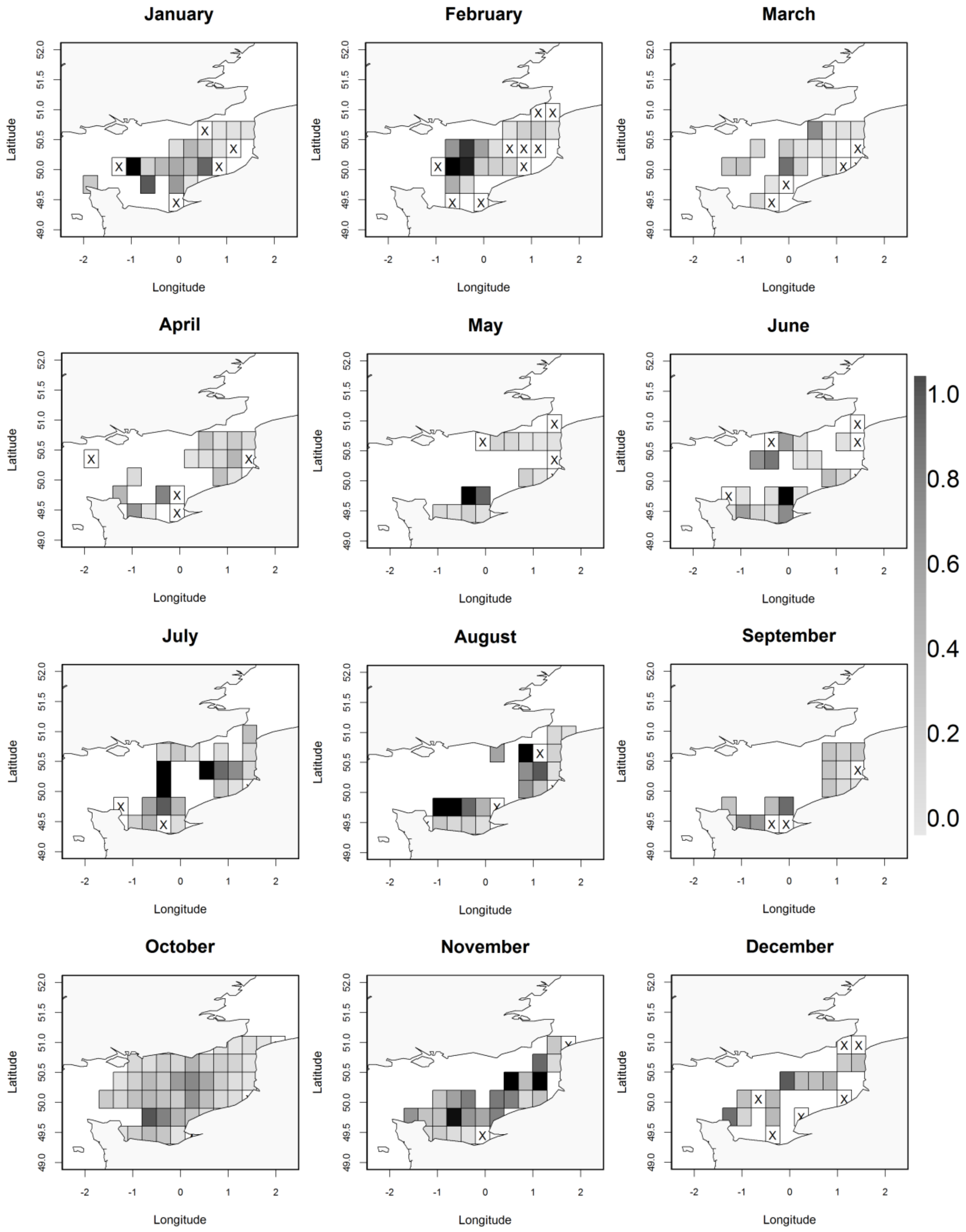


Figure S2.12. Monthly spatial abundance distribution estimated from OBSMER and CGFS for red gurnard. ‘X’ represents areas where no individual of a species was ever fished during a month in the database.

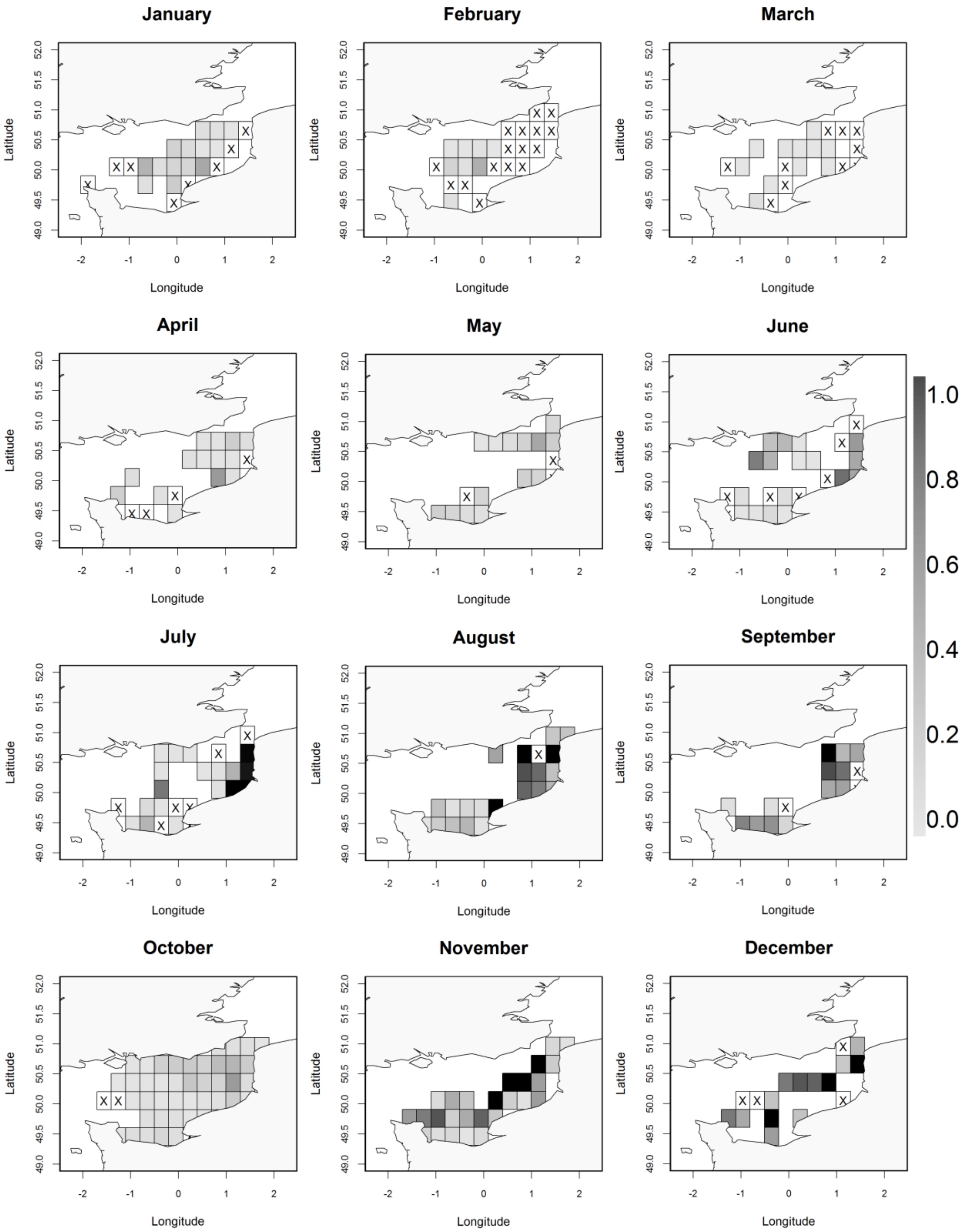


Figure S2.13. Monthly spatial abundance distribution estimated from OBSMER and CGFS for red mullet. ‘X’ represents areas where no individual of a species was ever fished during a month in the database.

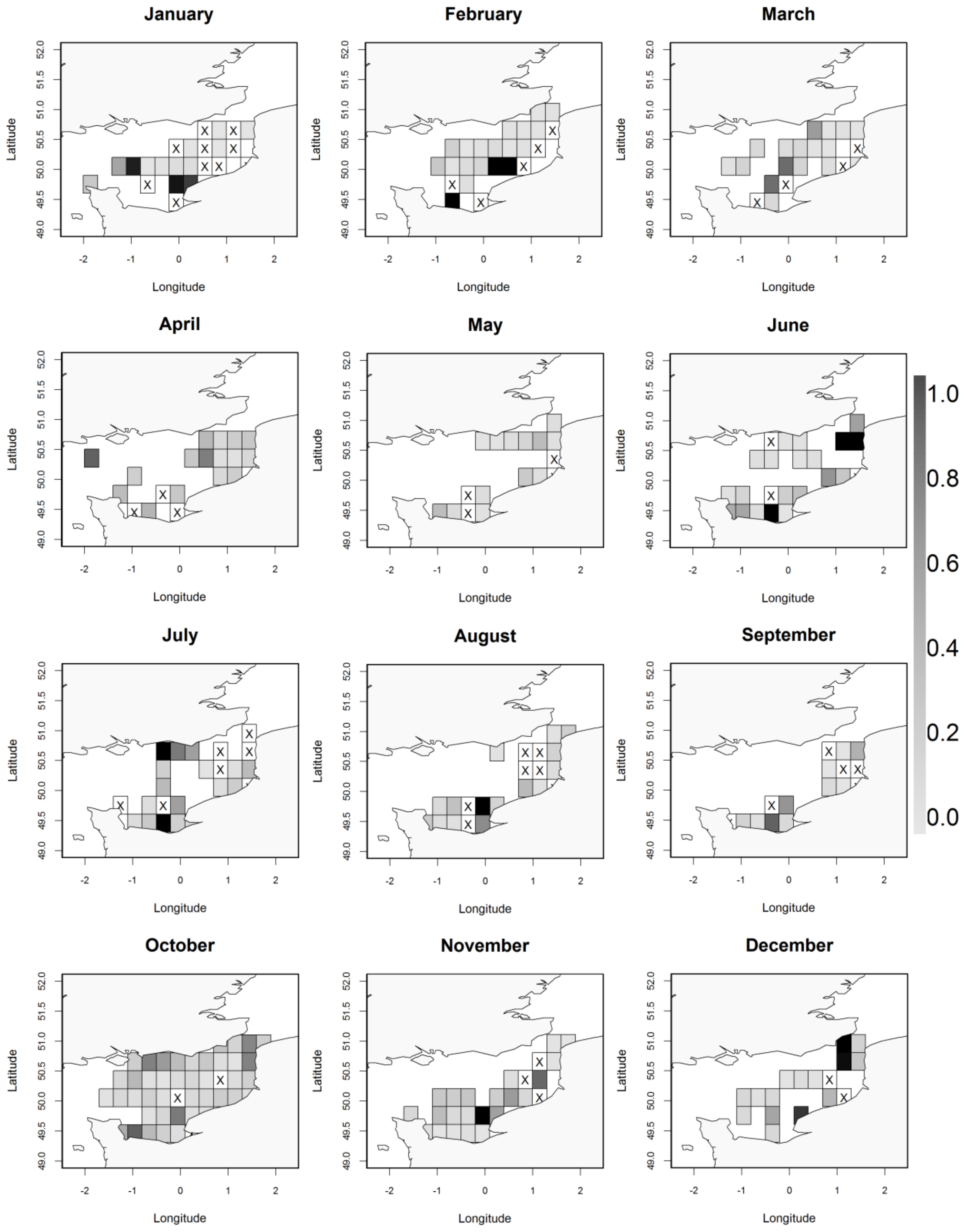


Figure S2.14. Monthly spatial abundance distribution estimated from OBSMER and CGFS for seabass. 'X' represents areas where no individual of a species was ever fished during a month in the database.

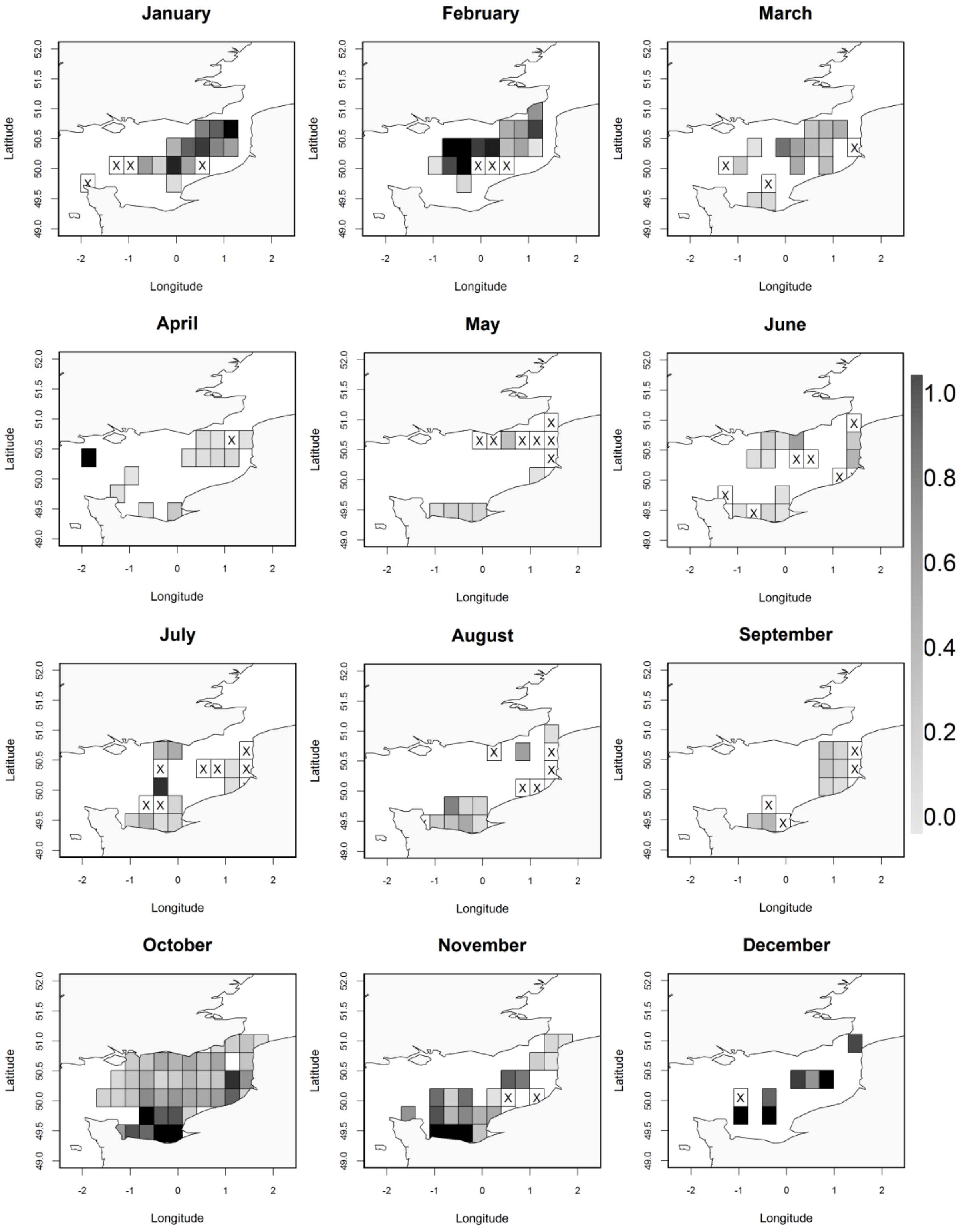


Figure S2.15. Monthly spatial abundance distribution estimated from OBSMER and CGFS for squids. 'X' represents areas where no individual of a species was ever fished during a month in the database.

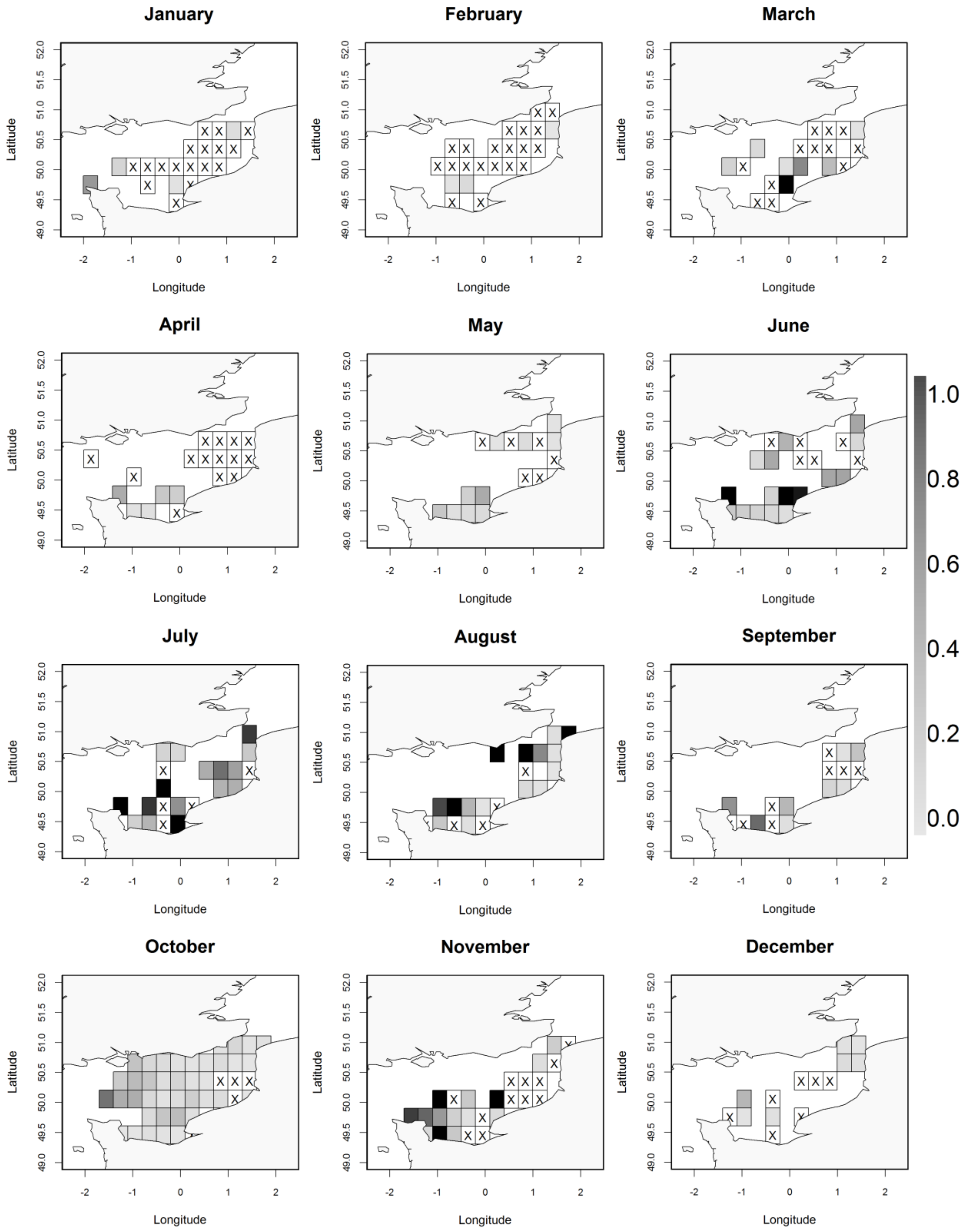


Figure S2.16. Monthly spatial abundance distribution estimated from OBSMER and CGFS for starry smooth-hound. ‘X’ represents areas where no individual of a species was ever fished during a month in the database.

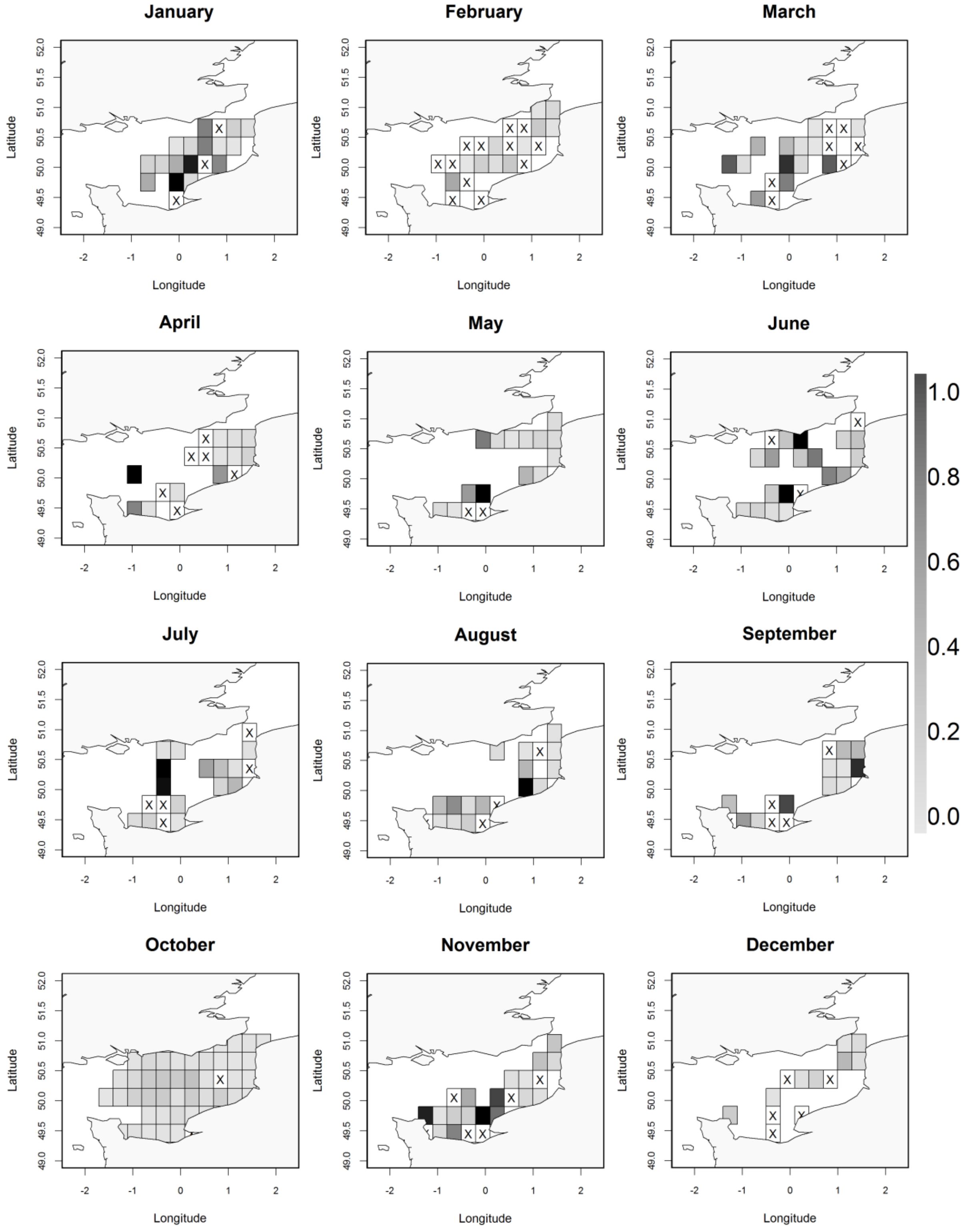


Figure S2.17. Monthly spatial abundance distribution estimated from OBSMER and CGFS for thornback ray. ‘X’ represents areas where no individual of a species was ever fished during a month in the database.

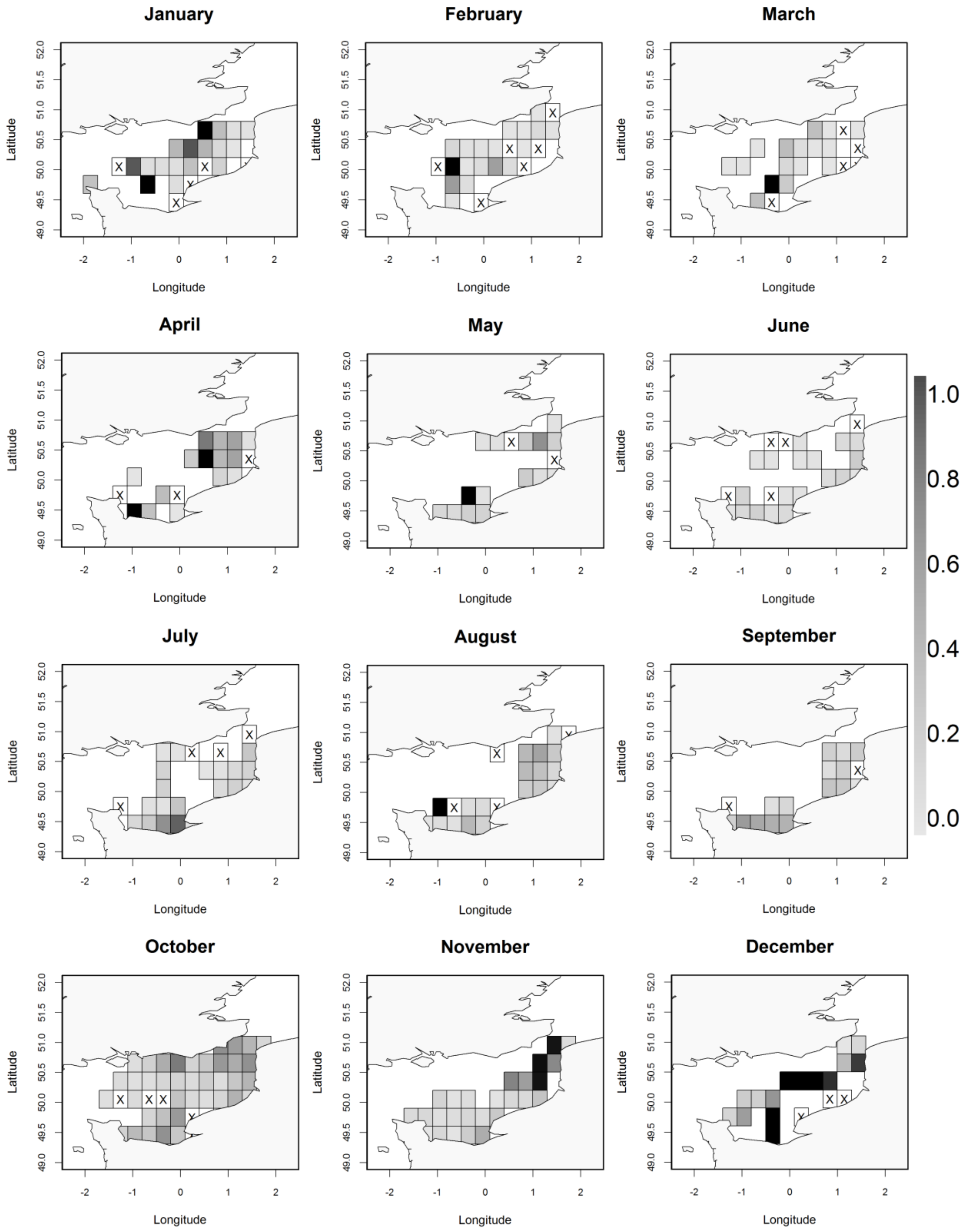


Figure S2.18. Monthly spatial abundance distribution estimated from OBSMER and CGFS for tub gurnard. ‘X’ represents areas where no individual of a species was ever fished during a month in the database.

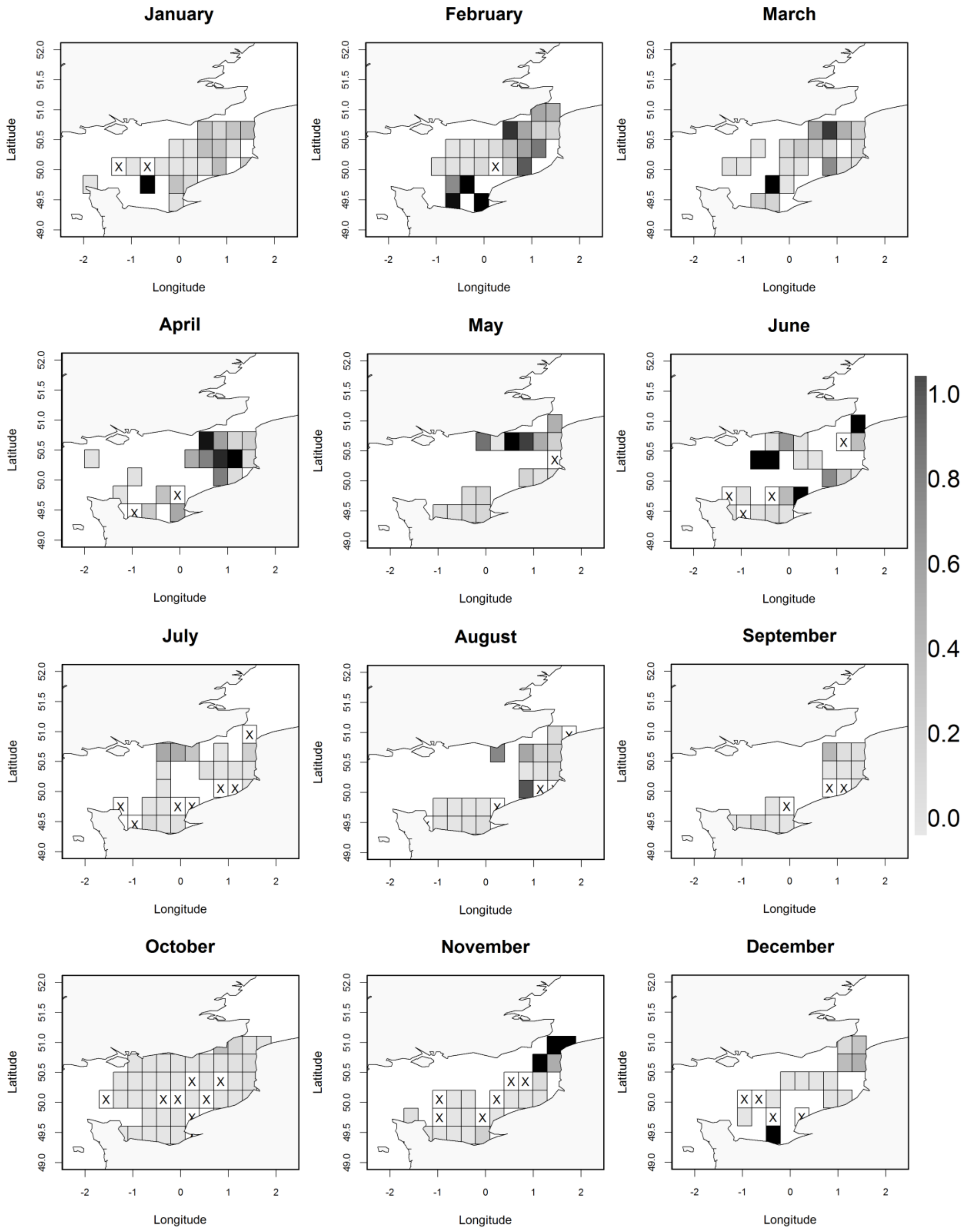
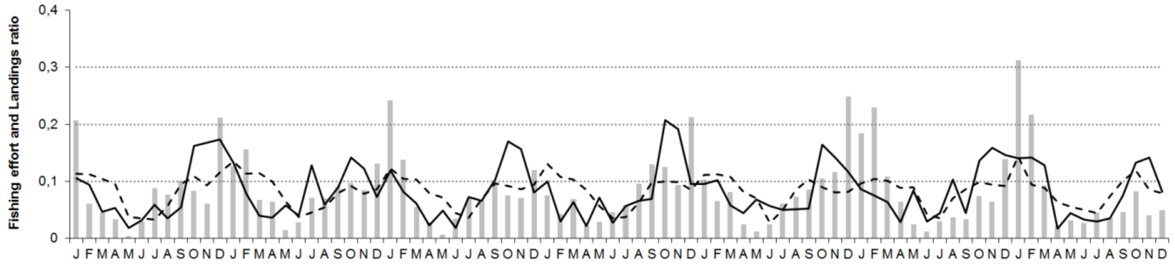
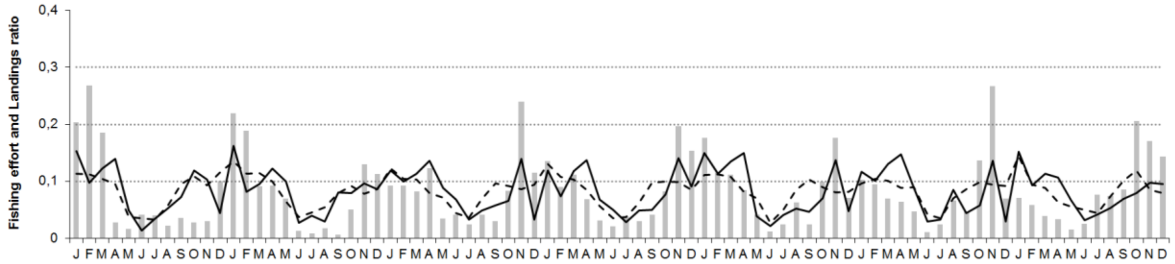
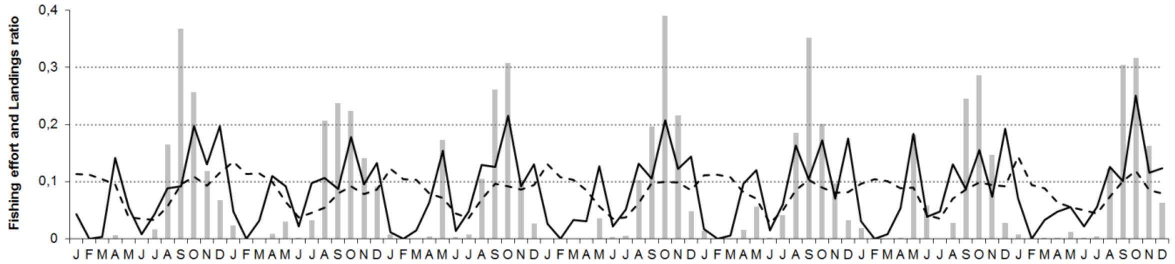
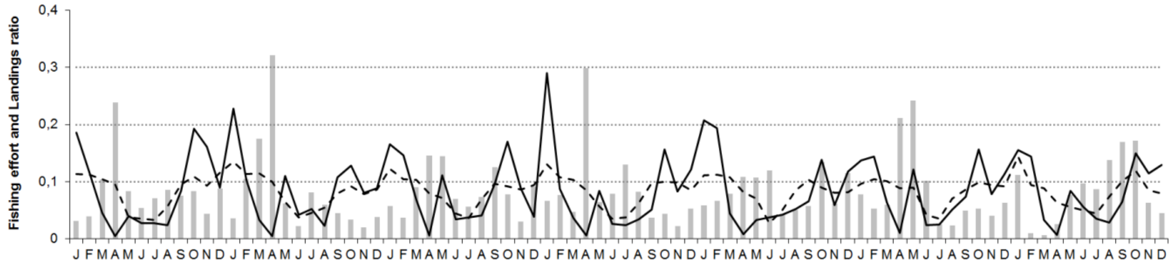


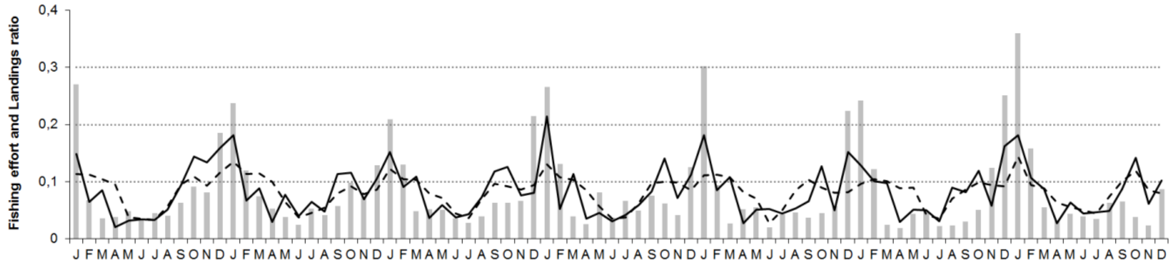
Figure S2.19. Monthly spatial abundance distribution estimated from OBSMER and CGFS for whiting. 'X' represents areas where no individual of a species was ever fished during a month in the database.

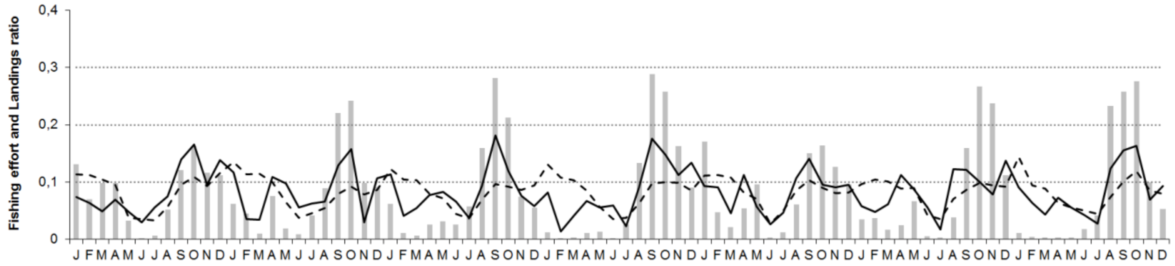

Black seabream

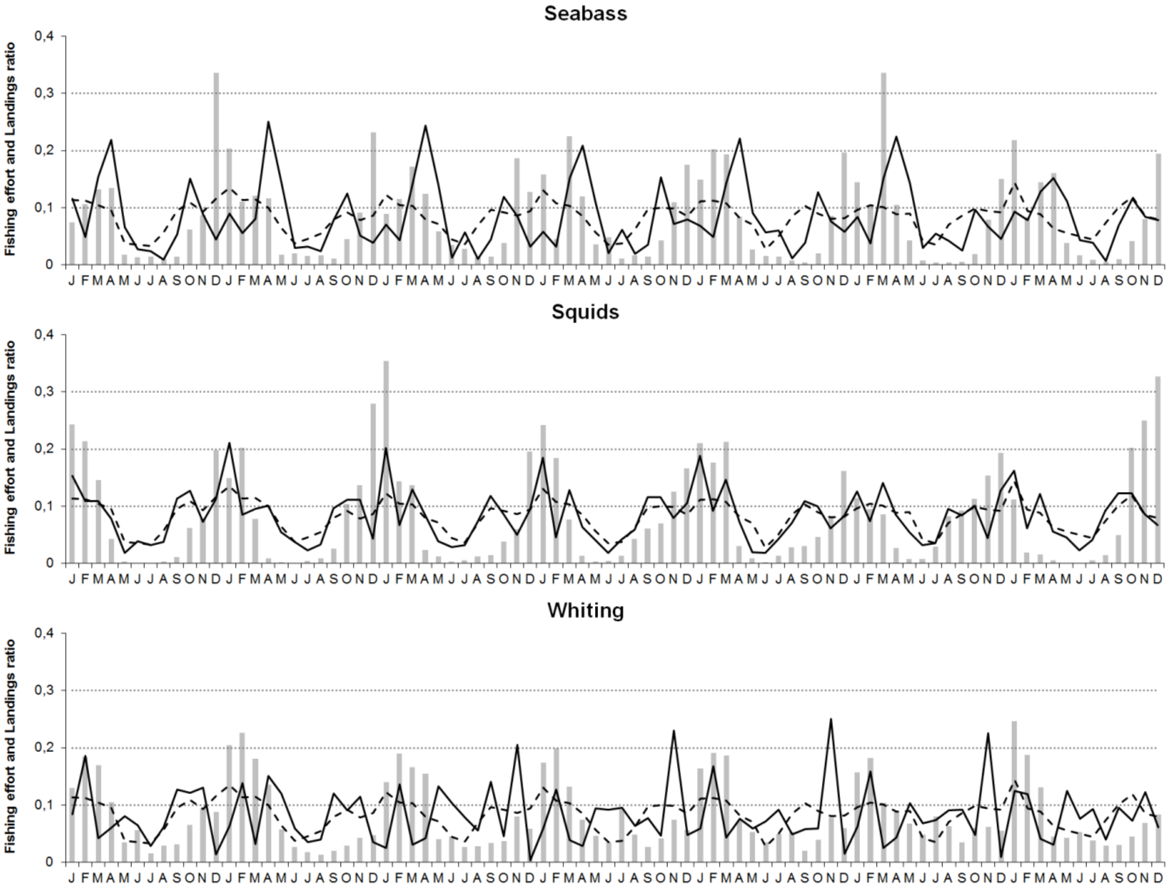

Cod


Cuttlefish


Mackerel


Plaice

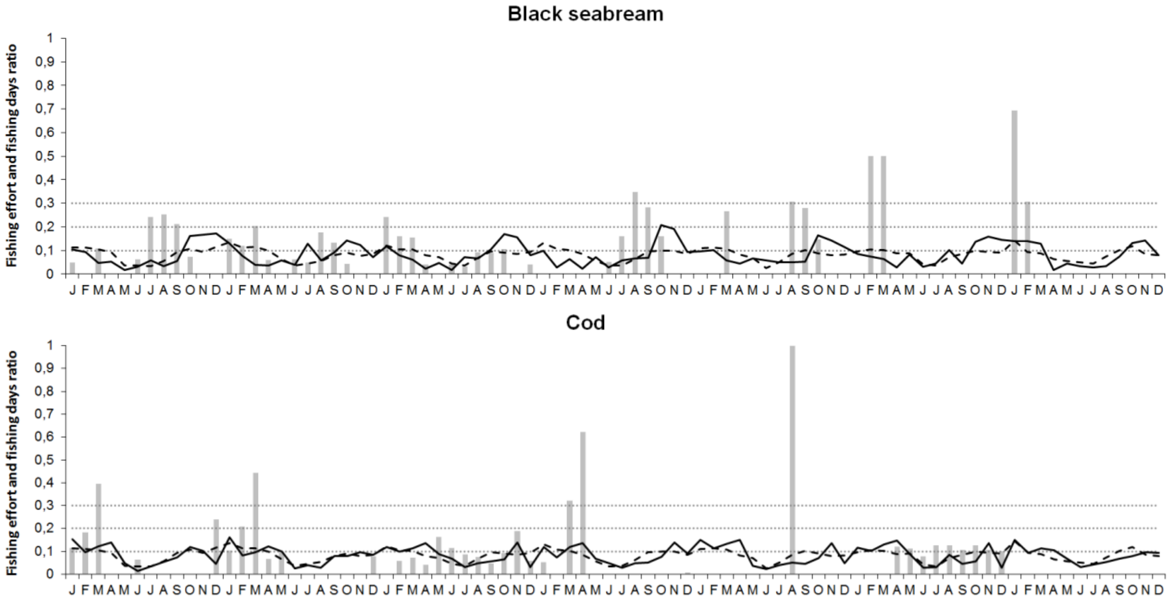

Red mullet


Seabass

Squids

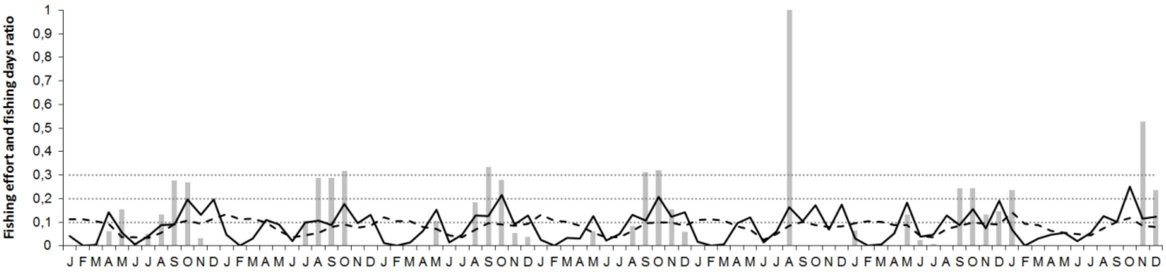
Whiting

Figure S3.1. Monthly nominal fishing effort, effective fishing effort and yearly standardized landings of exclusive bottom otter trawlers for nine main commercial species of the Eastern English Channel.

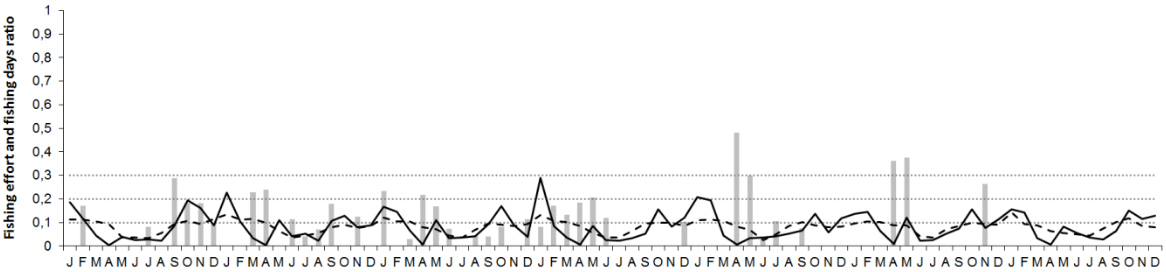

Black seabream

Cod

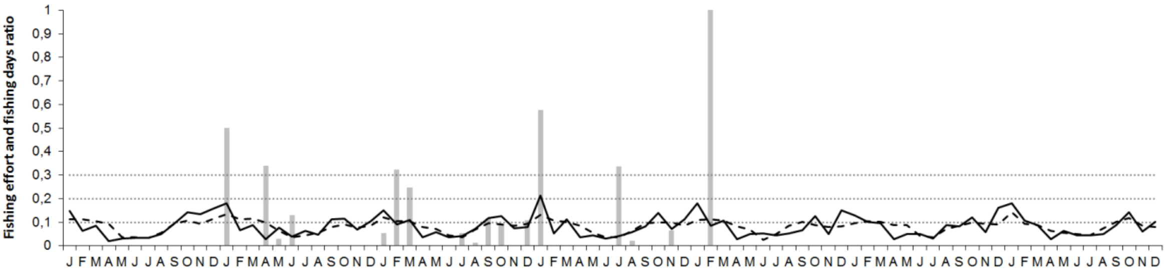
Cuttlefish



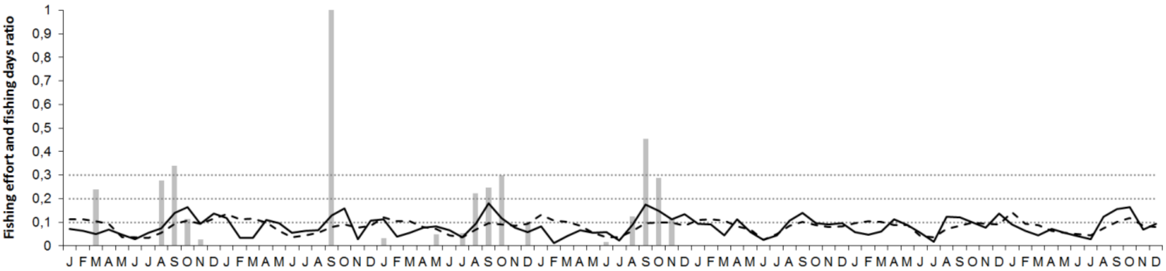
Mackerel



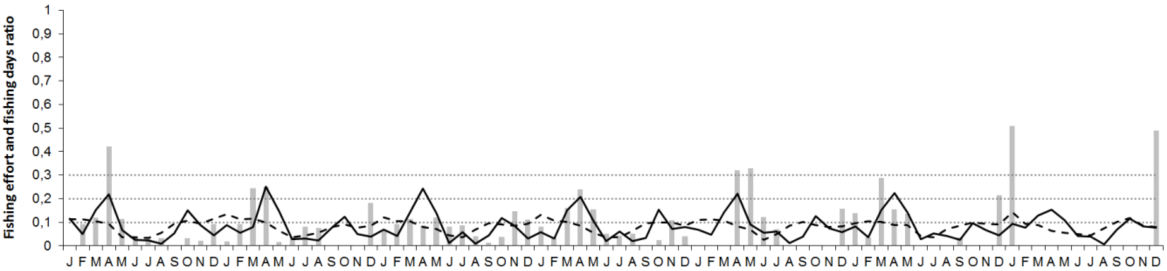
Plaice



Red mullet



Seabass



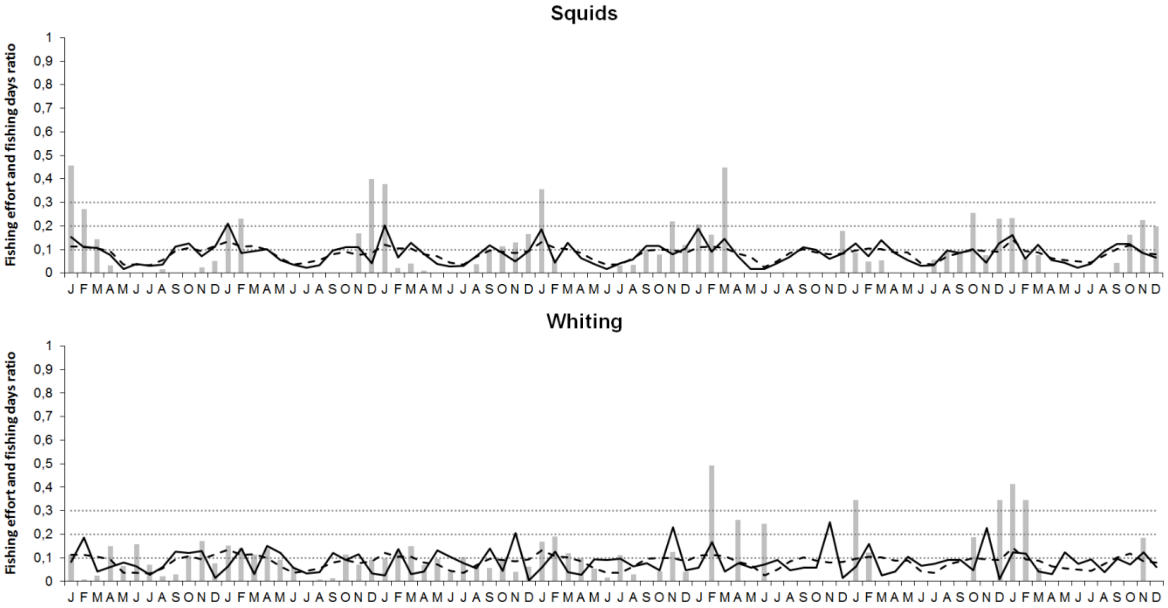


Figure S3.2. Monthly nominal fishing effort, effective fishing effort and yearly standardized number of fishing days from activity calendars of exclusive bottom otter trawlers for nine main commercial species of the Eastern English Channel.

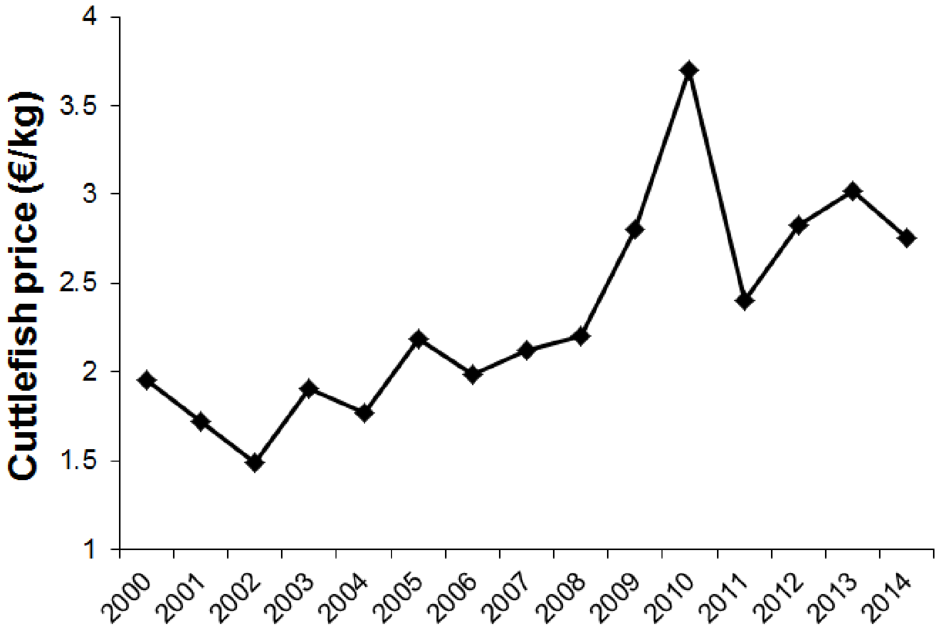


Figure S3.3. Average cuttlefish price (€ / kg) for bottom otter trawlers in October in Eastern English Channel during the period 2000-2014.

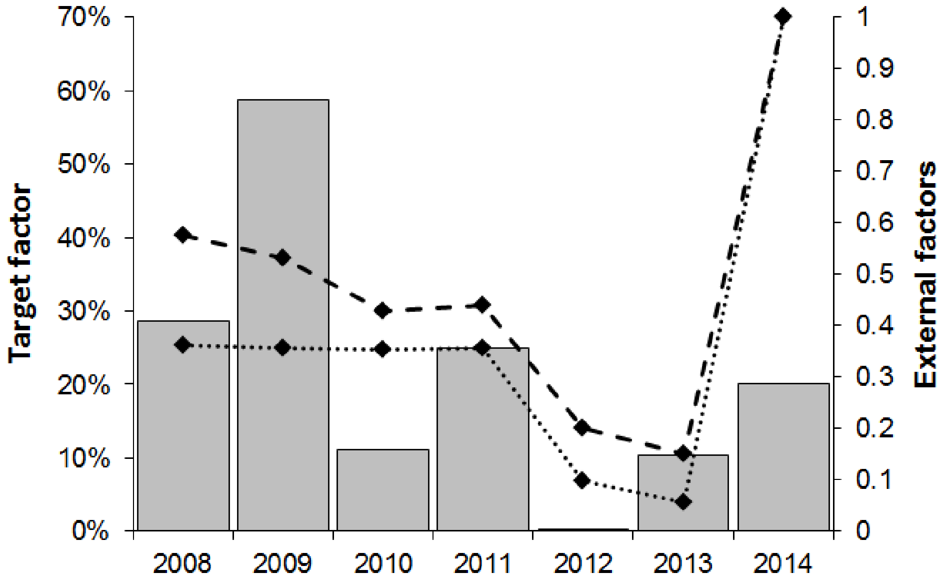


Figure S3.4. Relative red mullet targeting factor in October, over the period 2008-2014 (light grey bars), compared to its relative abundance (dotted lines), relative economic attractiveness (abundance x price; dashed lines).

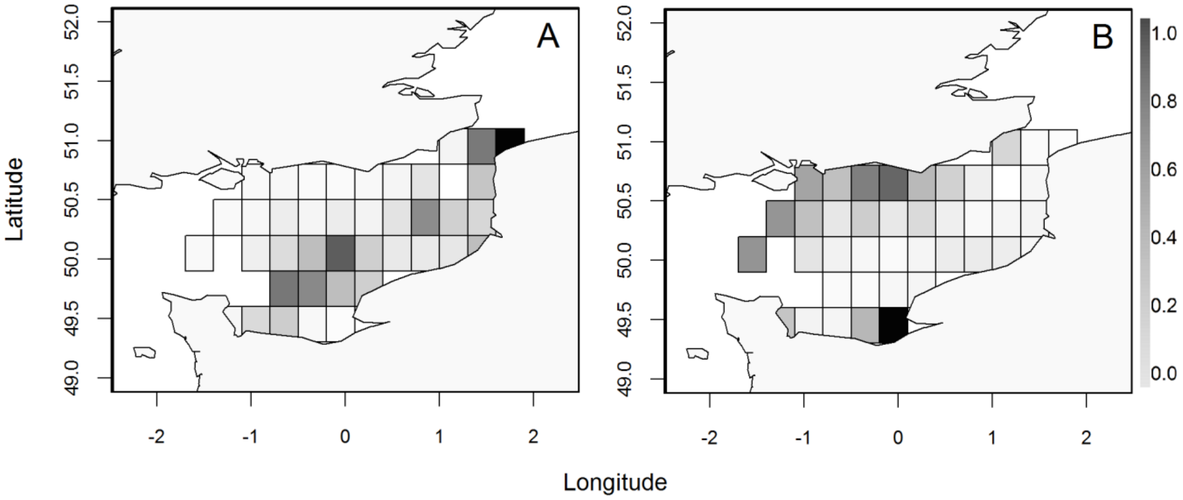


Figure S3.5. A) desirability and B) avoidance for fishing grounds in October for exclusive bottom trawlers in EEC. Obtain from Constant Catch rate Ratio model of isodars.

Isodars with Constant Catch rate Ratio (CCR)

Extensions of the Ideal Free Distribution (IFD), isodars (Gillis and van der Lee, 2012; van der Lee *et al.*, 2014), are used to examine the dynamics of effort related to target species distributions. Isodars (for 'iso-Darwin') can be developed with different models. For computation reasons but also due to the high number of activities in EEC, the Constant Catch rate Ratio model (CCR) was here chosen. It is based on the suggestion of Hilborn and Ledbetter (1979) that differences in the costs between areas can be explained by catch rates differing among areas but maintaining a constant ratio α. The ratio represents desirability for areas, associated to costs and risks. We use estimated abundance *A* of targets with effort *E* in areas 1 and 2 instead of catches. The model is represented by:

$$\frac{A_2}{E_2} = e^{\alpha}.\frac{A_1}{E_1}$$

$$\log(E_2) = \left[\alpha - \log(\frac{A_1}{A_2})\right] + \log(E_1)$$

Here isodars can only be used with percentages. Target distributions are obtained by the maximization of Local Index of Collocation (LIC) between relative abundance of main commercial species and fishing effort, as done by Bourdaud *et al.* (submitted) on exclusive bottom otter trawlers in October. But for isodars computation, an iterative process is used until reaching stabilization of LIC value (Figure S3.6). We consider the LIC stable when a variation of its value is less than 5% of the previous one. The median estimated value of α is kept at each step to estimate the target and compute the LIC. Desirability is finally converted by taking into account the percentage of the emerged part of the areas.

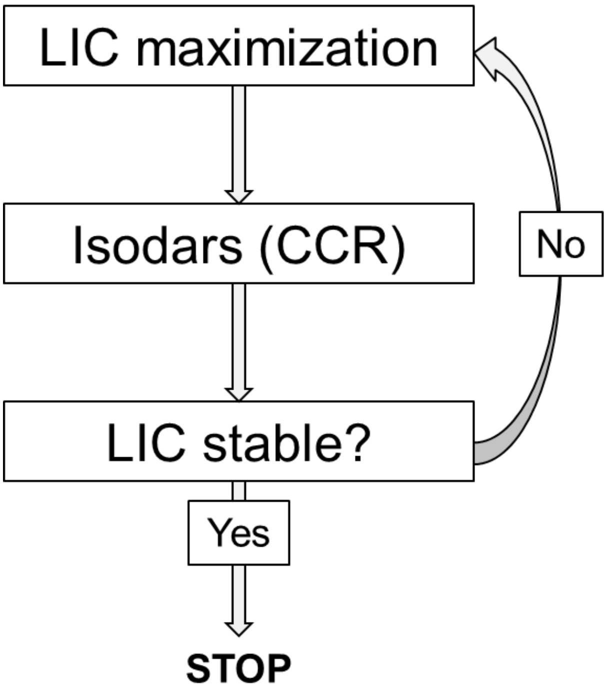
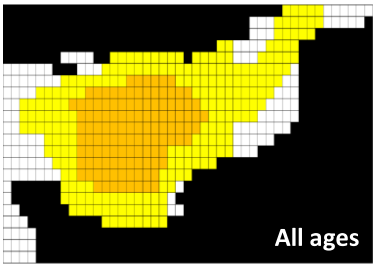


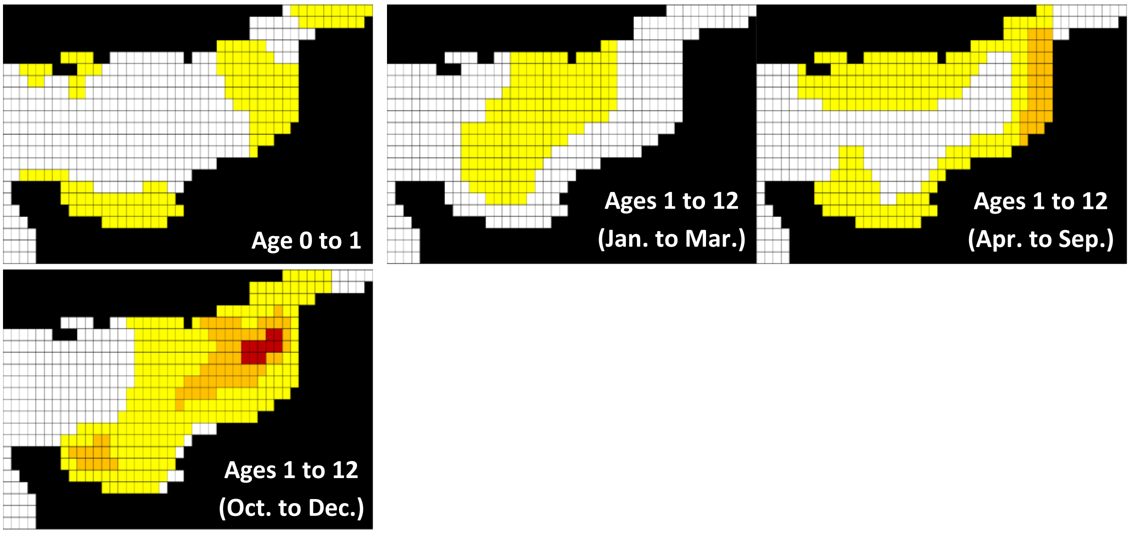
Figure S3.6. Iterative process used to obtain desirability of the different areas and target values.

The results of desirability and avoidance of exclusive bottom trawlers in EEC are exhibited on Figure S3.5. These confirm results obtained by Girardin *et al*. (2015) concerning the avoidance of United Kingdom coasts by French bottom trawlers. It is also noteworthy that fishers severely avoid the Eastern coastal part of Normandy, probably due to the shallowness in these areas (see Carpentier *et al*., 2009), and that three patches of desirability are observed in front of Port-en-Bessin, in front of Boulogne-sur-Mer and in the French coast of the Dover Strait at this time of the year.

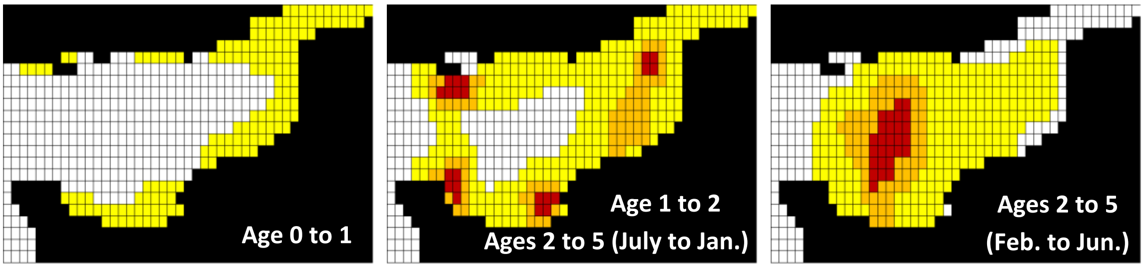
Lesser-spotted dogfish (Carpentier *et al*., 2009; Bourdaud *et al*., 2017)



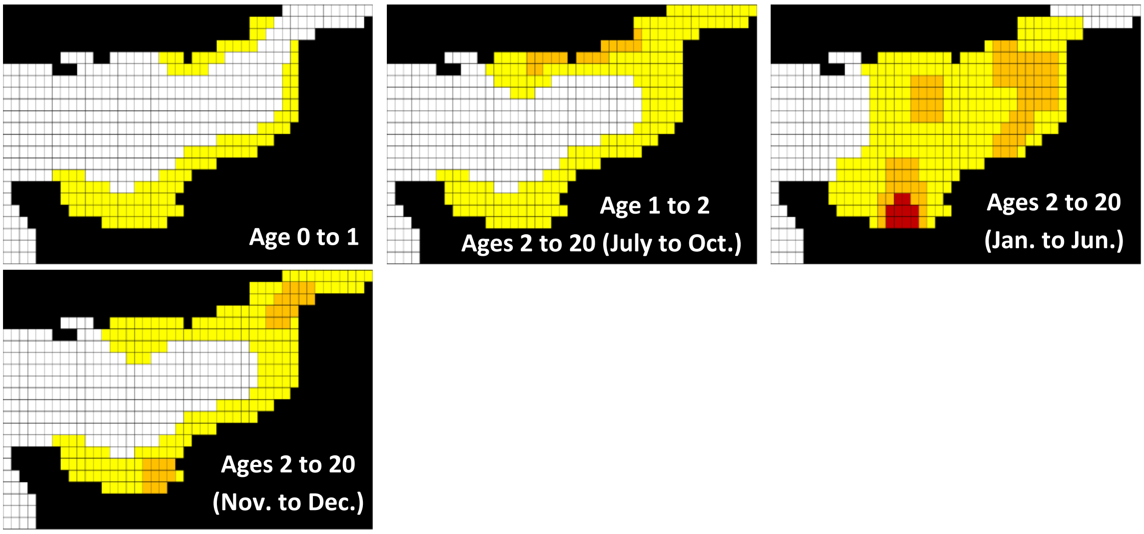
Red mullet (Carpentier *et al*., 2009; Bourdaud *et al*., 2017)



Pouting (Carpentier *et al*., 2009; Desmarchelier, 1986; Bourdaud *et al*., 2017)

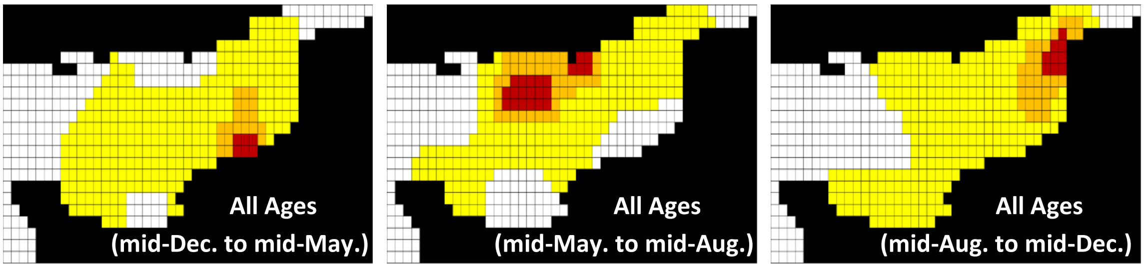


Whiting (Carpentier *et al.*, 2009; Lelièvre *et al.*, 2014; Pawson, 1995; Bourdaud *et al.*, 2017)

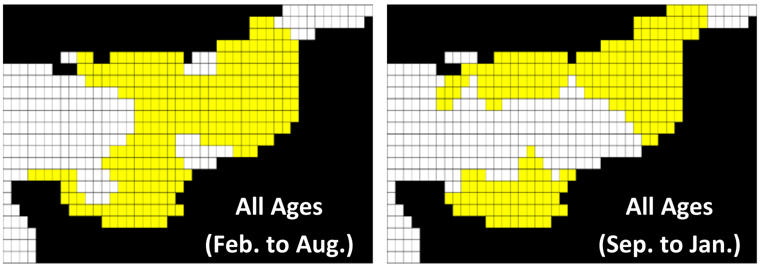


Poor cod (Carpentier *et al.*, 2009; Bourdaud *et al.*, 2017)

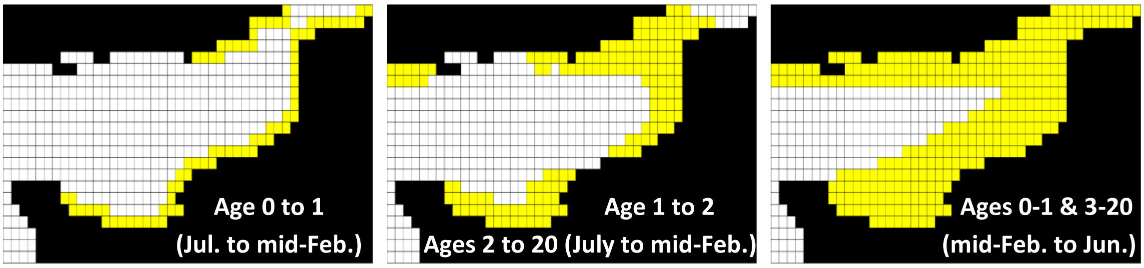
Cod (Carpentier *et al.*, 2009, Pawson, 1995, Lelièvre *et al.*, 2014; Bourdaud *et al.*, 2017)



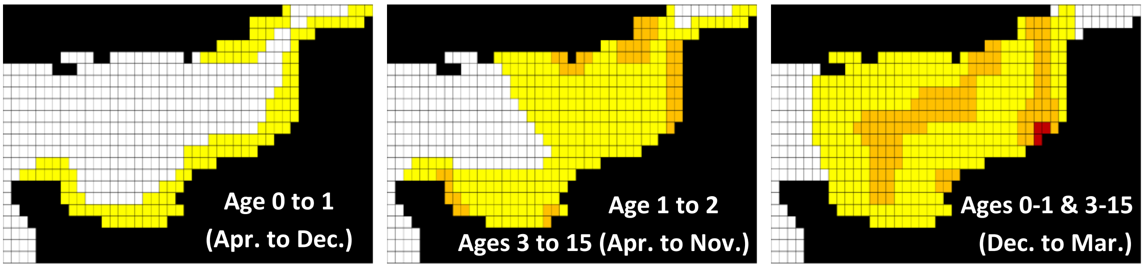
Dragonet (Carpentier *et al.*, 2009)



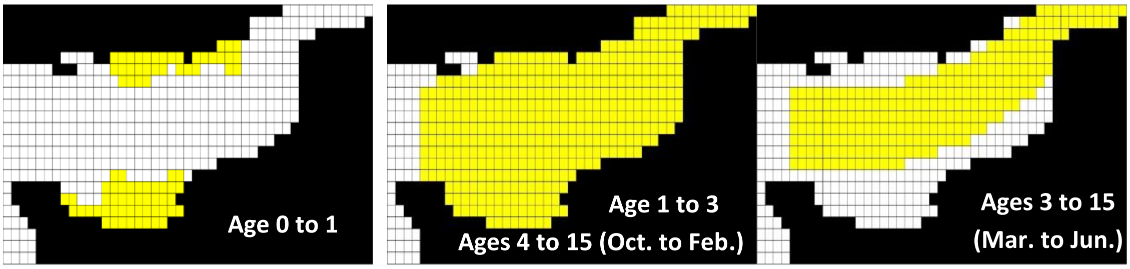
Sole (Carpentier *et al*., 2009, Rochette, 2011)



Plaice (Carpentier *et al*., 2009, Pawson 1995; Bourdaud *et al*., 2017)

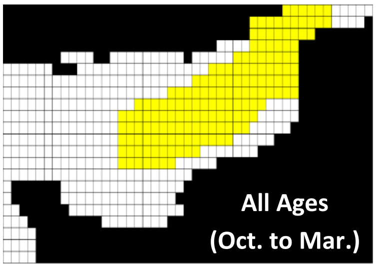


Horse Mackerel (Carpentier *et al*., 2009, Pawson 1995)

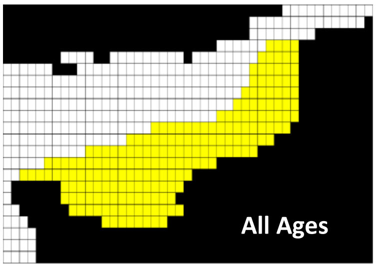


Mackerel (Carpentier *et al*., 2009; Bourdaud *et al*., submitted)

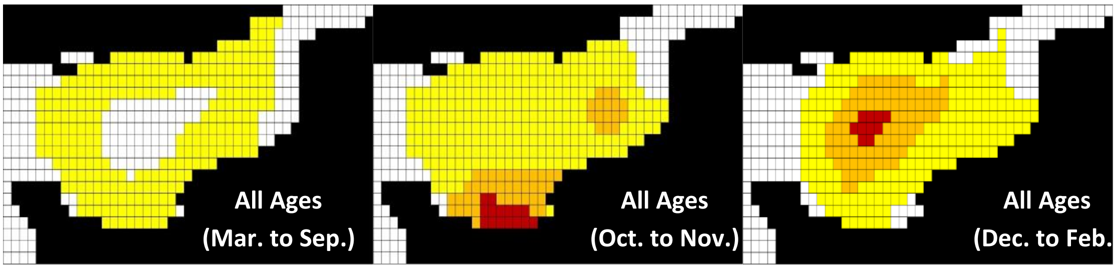
Herring (Carpentier *et al.*, 2009, Pawson 1995)



Sardine (Carpentier *et al.*, 2009)



Squids (Carpentier *et al.*, 2009; Bourdaud *et al.*, 2017)



Cuttlefish (Royer *et al.*, 2006; Bourdaud *et al.*, 2017)

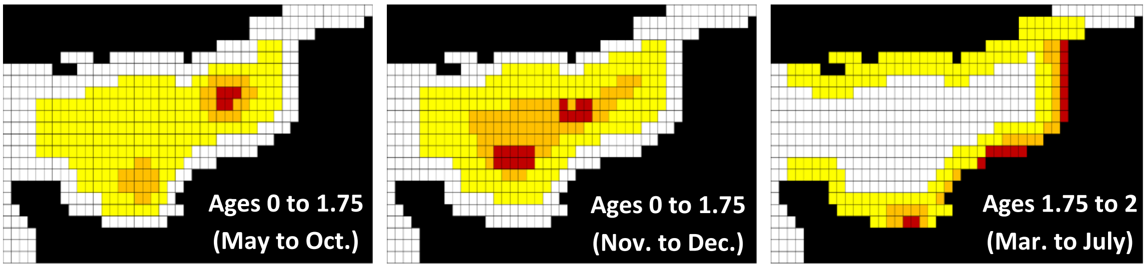
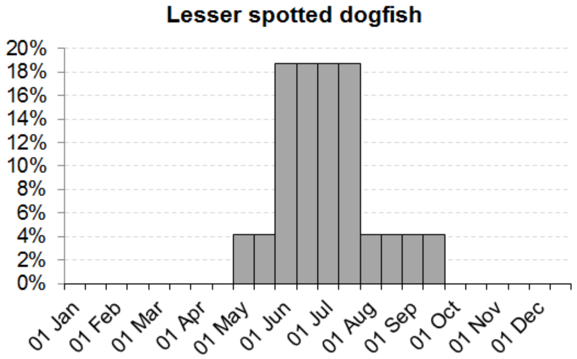


Figure S4.1. Distribution maps used as input of the model (with different abundance intensities: yellow for 1, orange for 2, red for 3 and white for absence). The age classes are specified for each map, as well as season when distribution changes within the year. Maps combined information from scientific survey CGFS (Coppin and Travers-Trolet, 1989) and from literature, with the references indicated for each species and including published maps as well as broader descriptions.


Lesser spotted dogfish
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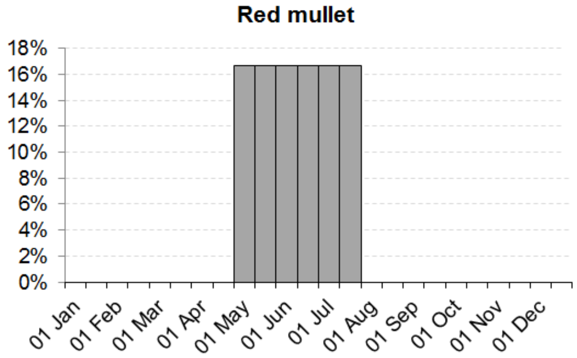
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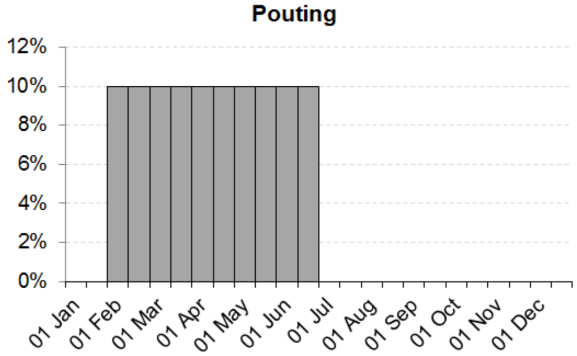
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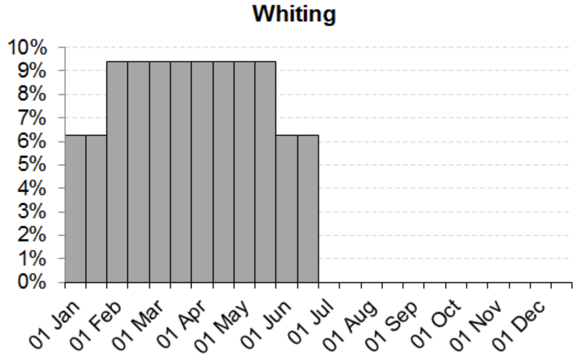
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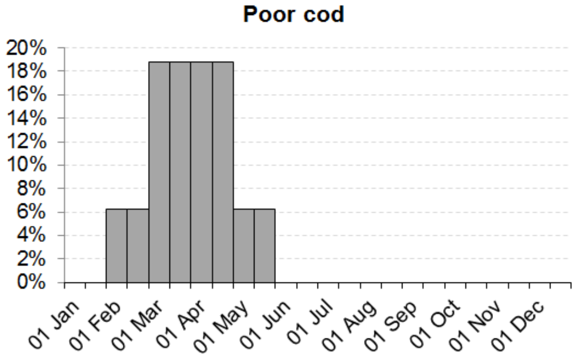
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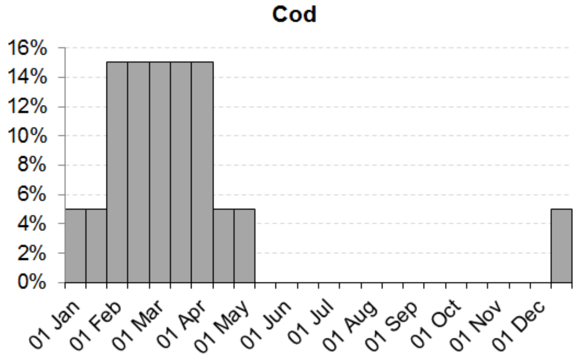
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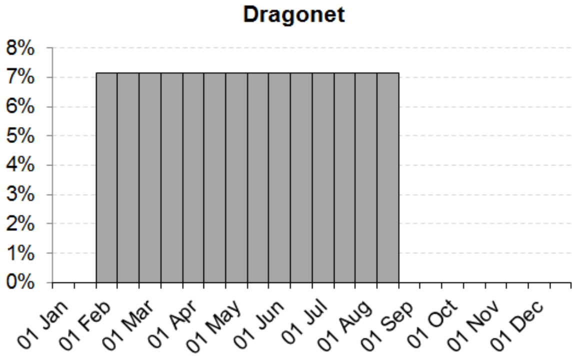
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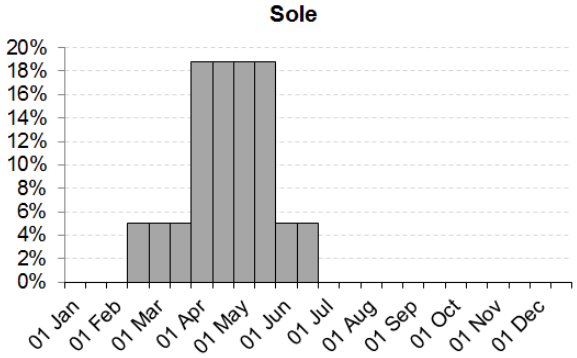
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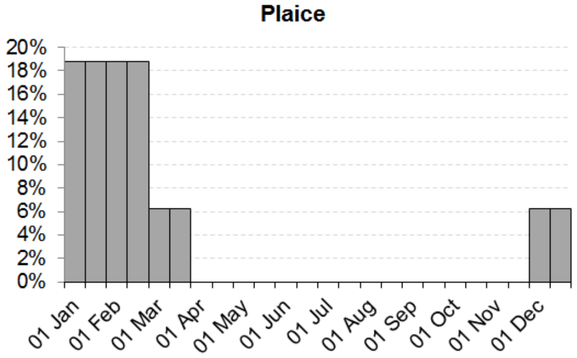
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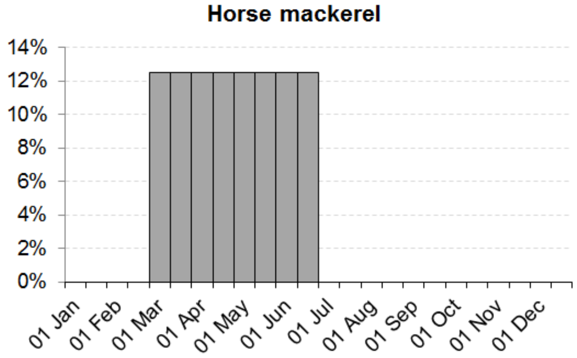
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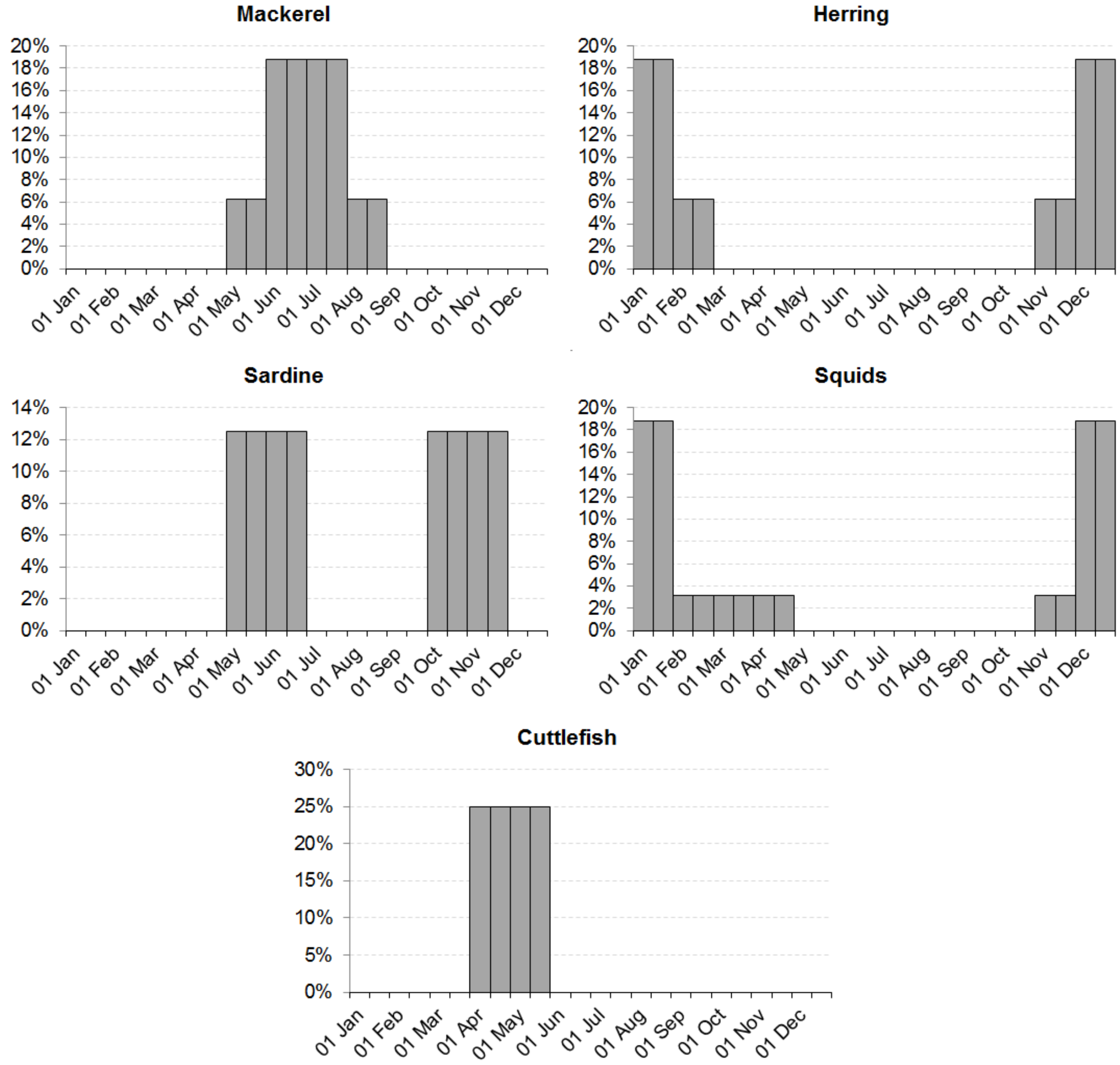


Figure S4.2. Reproduction seasonality γ for Eastern English Channel species in OSMOSE.

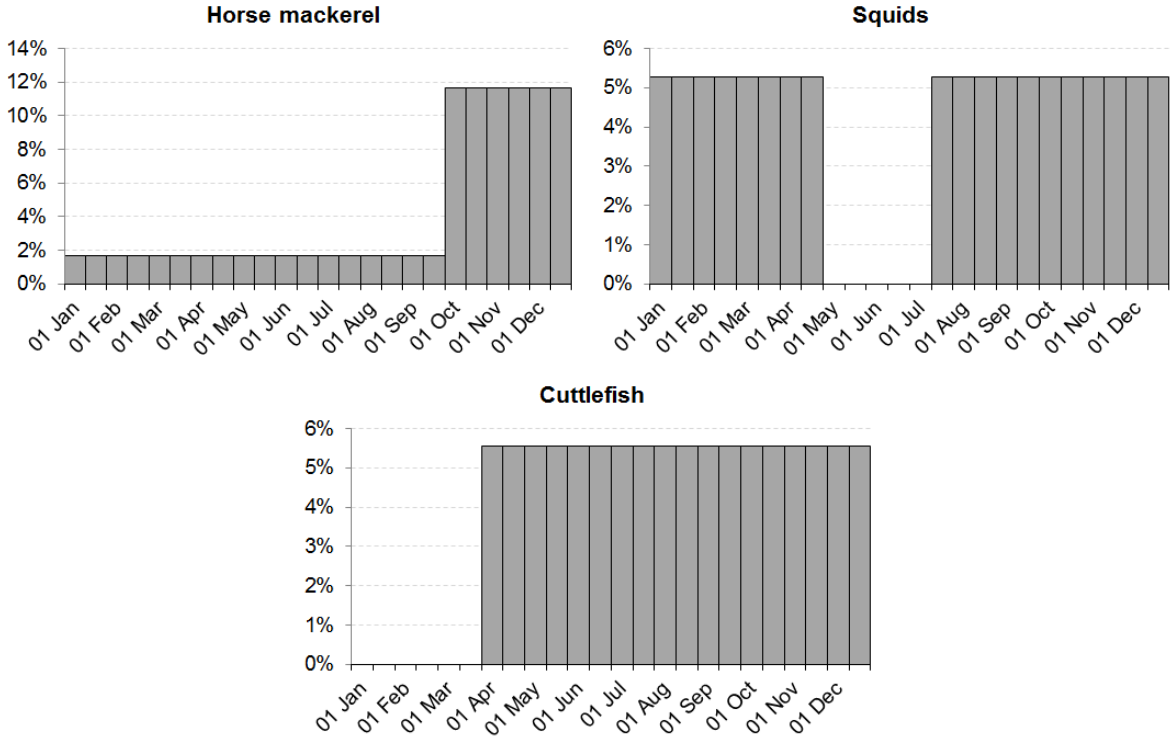


Figure S4.3. Fishing mortality seasonality for horse mackerel, squids and cuttlefish in OSMOSE.

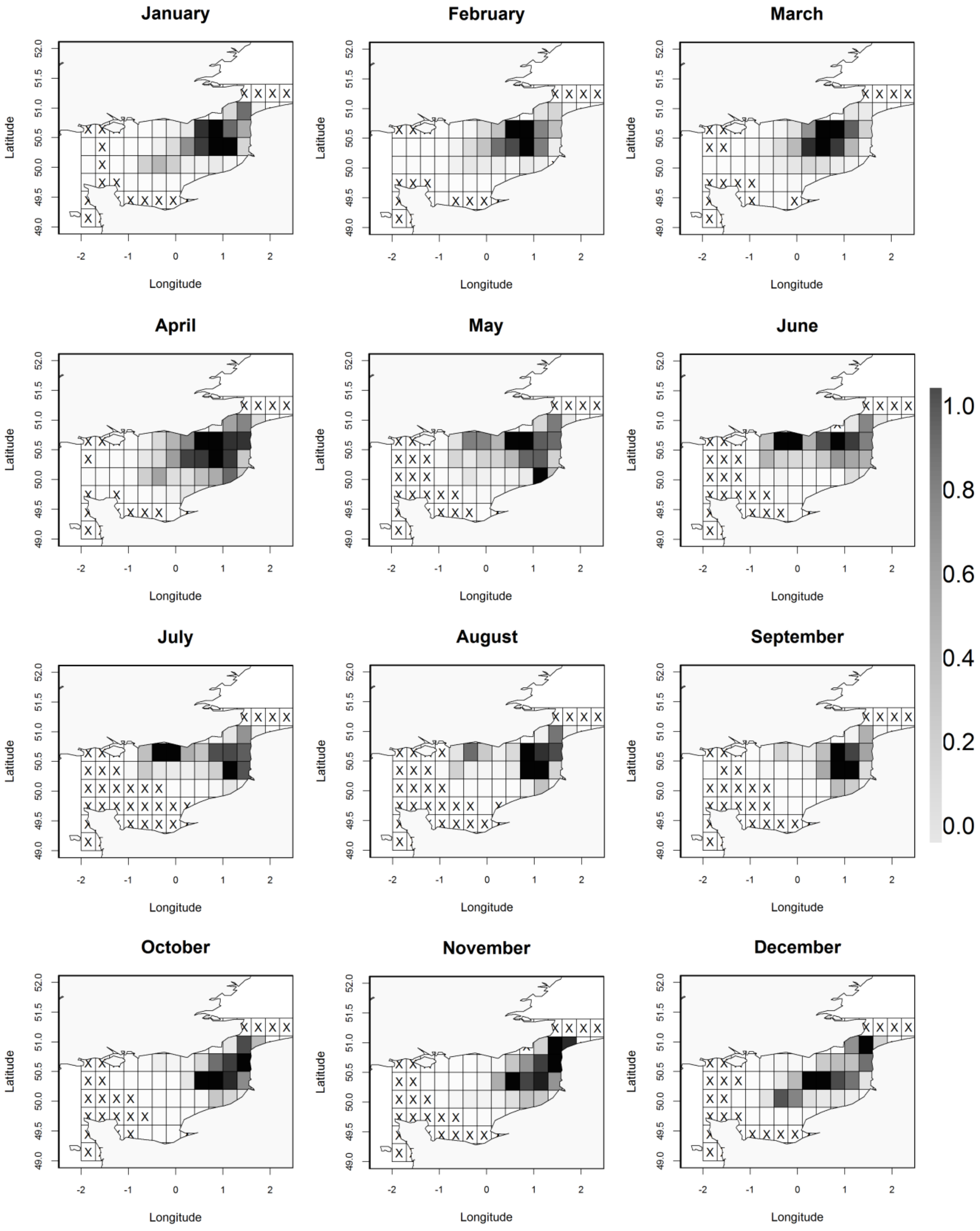


Figure S4.4. Mean monthly relative distribution of fishing effort operated by exclusive bottom trawlers from Boulogne-sur-Mer in Eastern English Channel in 2008-2015. “X” represents a null value.

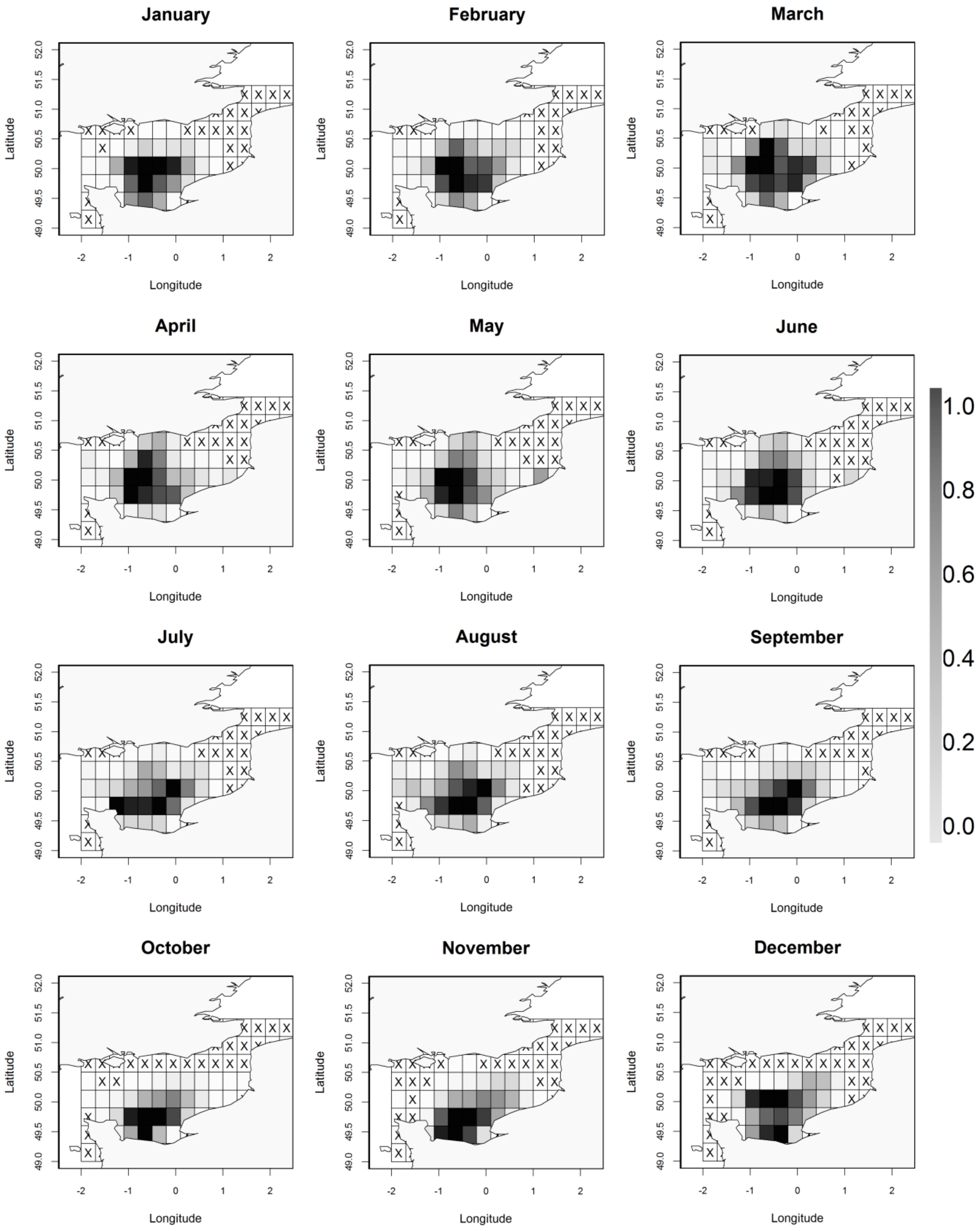


Figure S4.5. Mean monthly relative distribution of fishing effort operated by exclusive bottom trawlers from Port-en-Bessin in Eastern English Channel 2008-2015. “X” represents a null value.

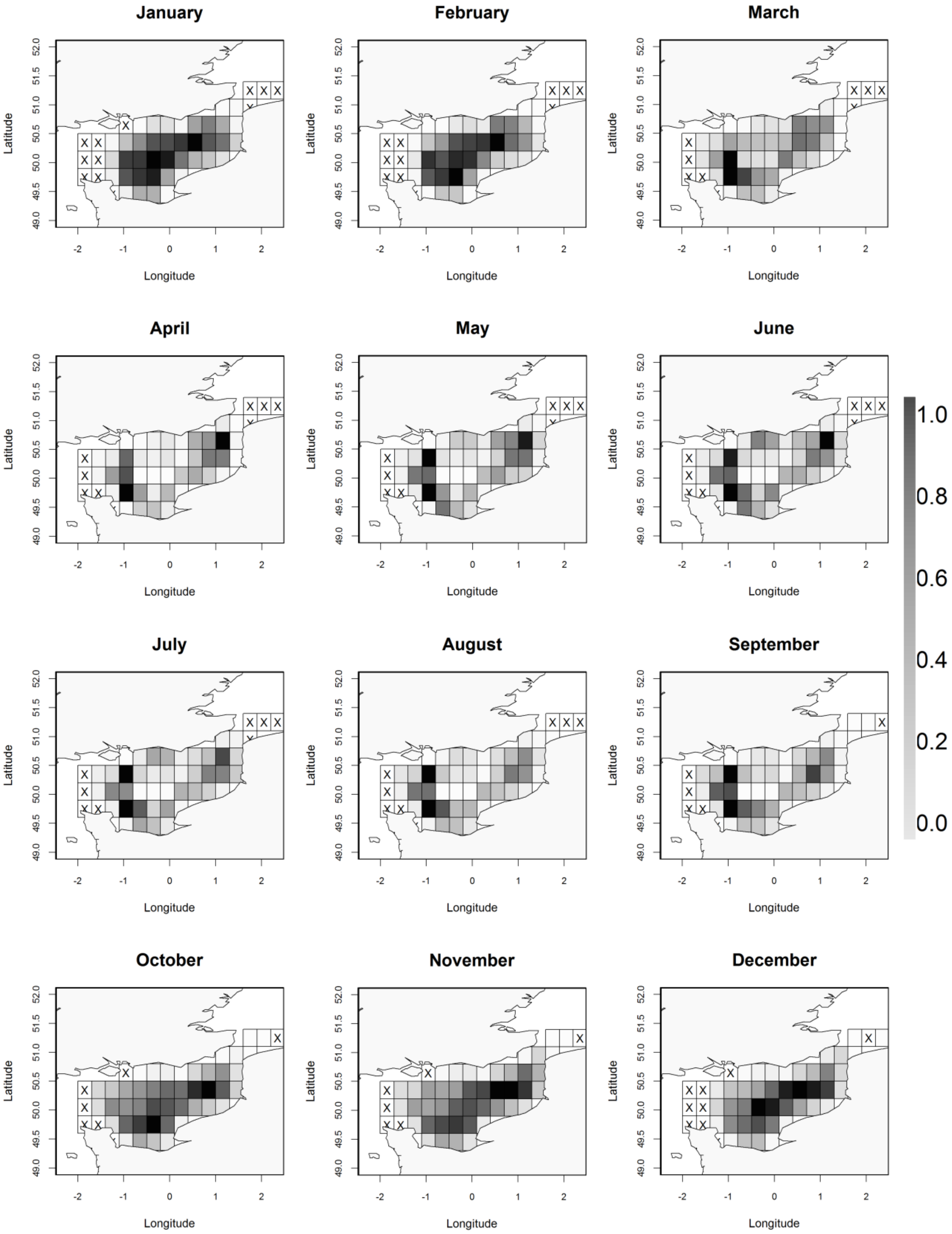


Figure S4.6. Mean monthly relative distribution of fishing effort operated by exclusive bottom trawlers in Eastern English Channel with the "Business as Usual" scenario. "X" represents a null value.

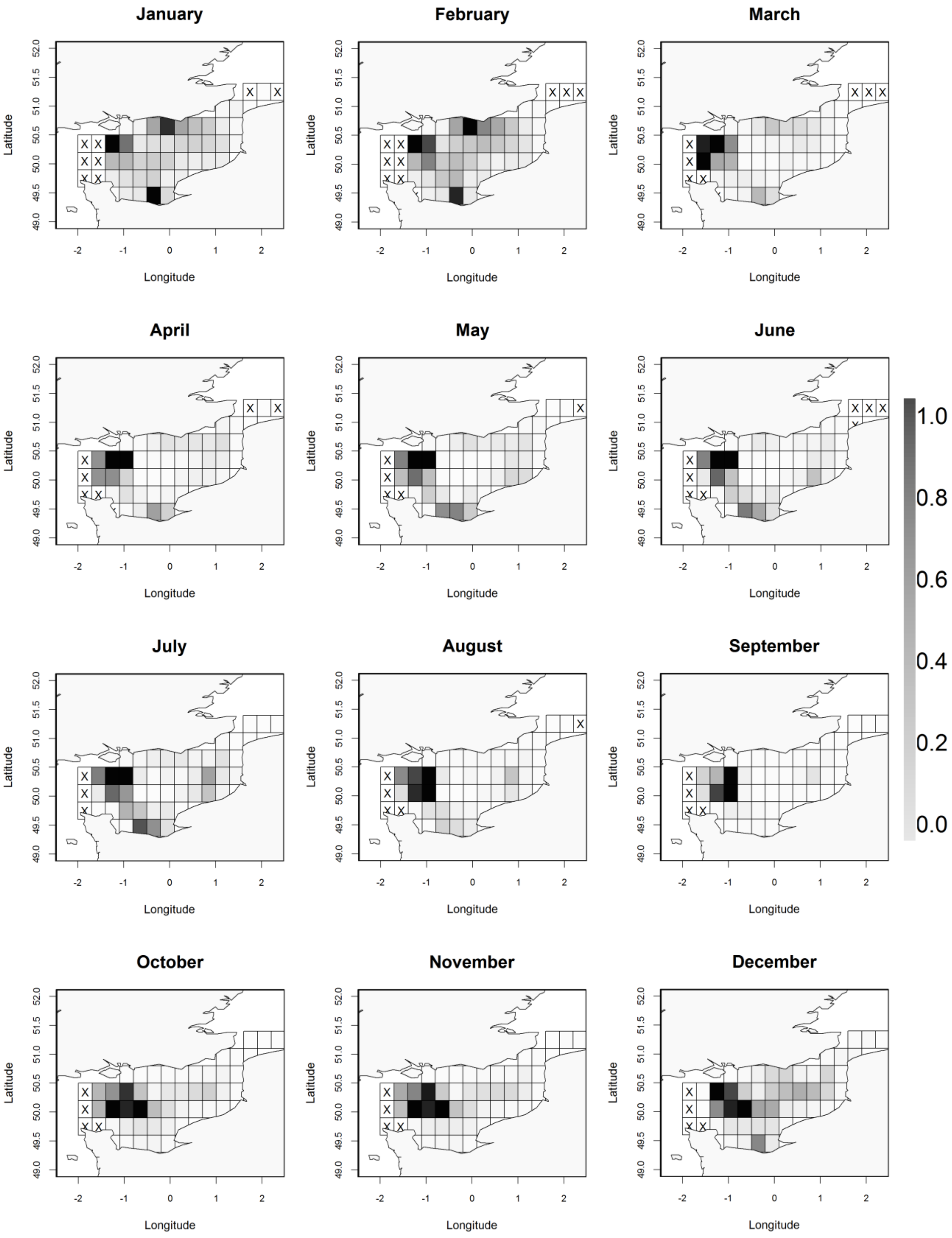


Figure S4.7. Mean monthly relative distribution of fishing effort operated by exclusive bottom trawlers in Eastern English Channel with the Landing Obligation scenario. “X” represents a null value.

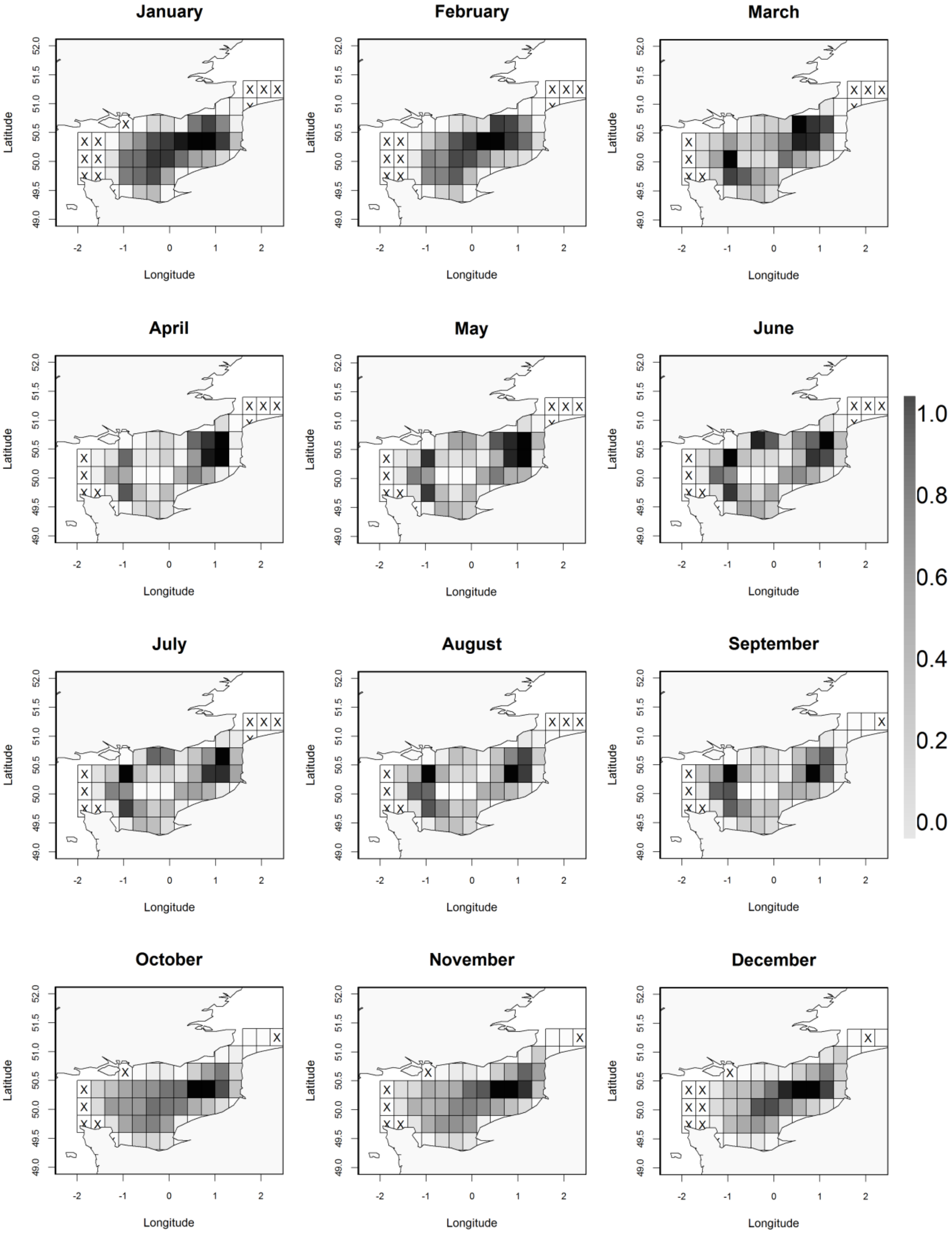


Figure S4.8. Mean monthly relative distribution of fishing effort operated by exclusive bottom trawlers from Boulogne-sur-Mer in Eastern English Channel with the “Business as Usual” scenario. “X” represents a null value.

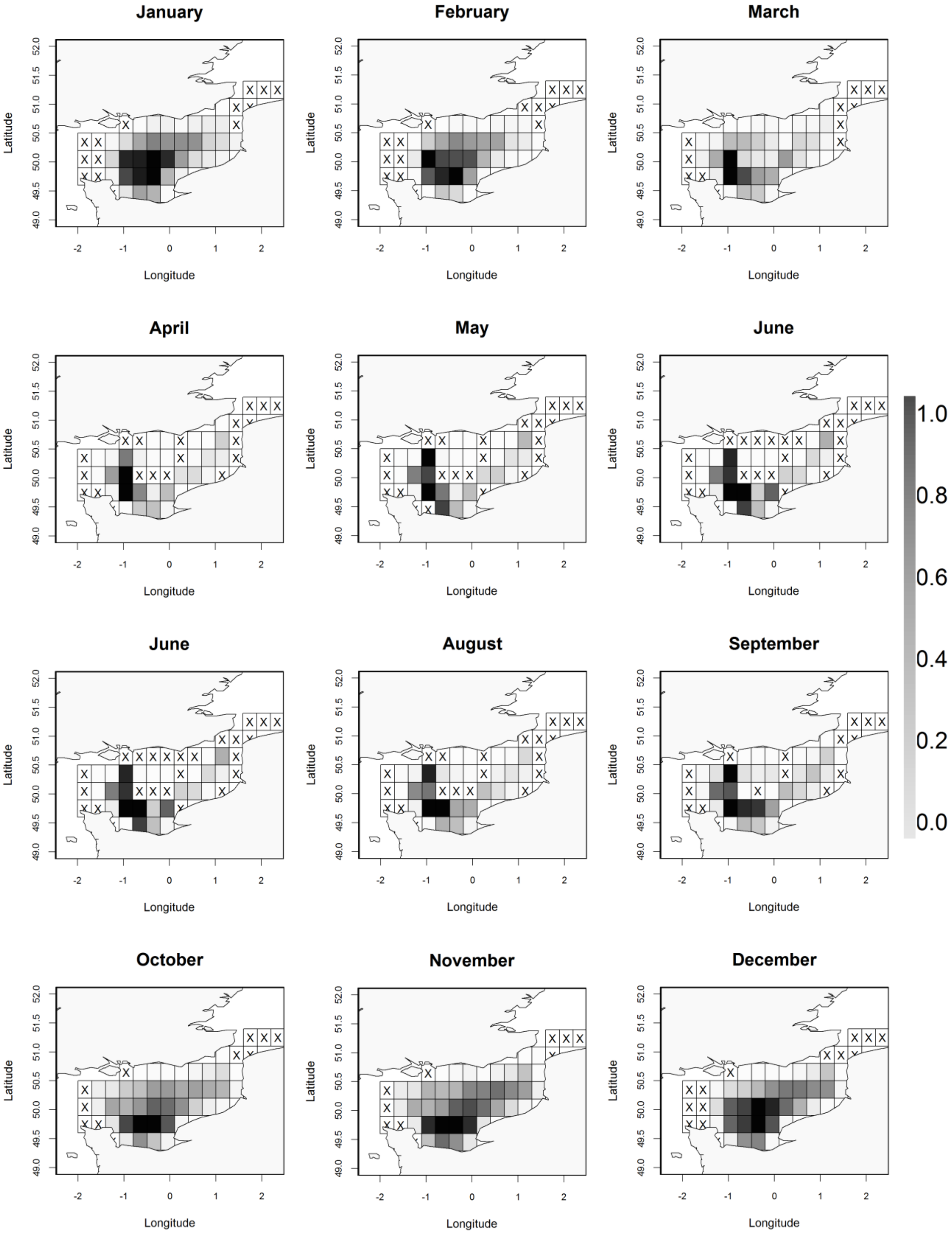


Figure S4.9. Mean monthly relative distribution of fishing effort operated by exclusive bottom trawlers from Port-en-Bessin in Eastern English Channel with the "Business as Usual" scenario. "X" represents a null value.

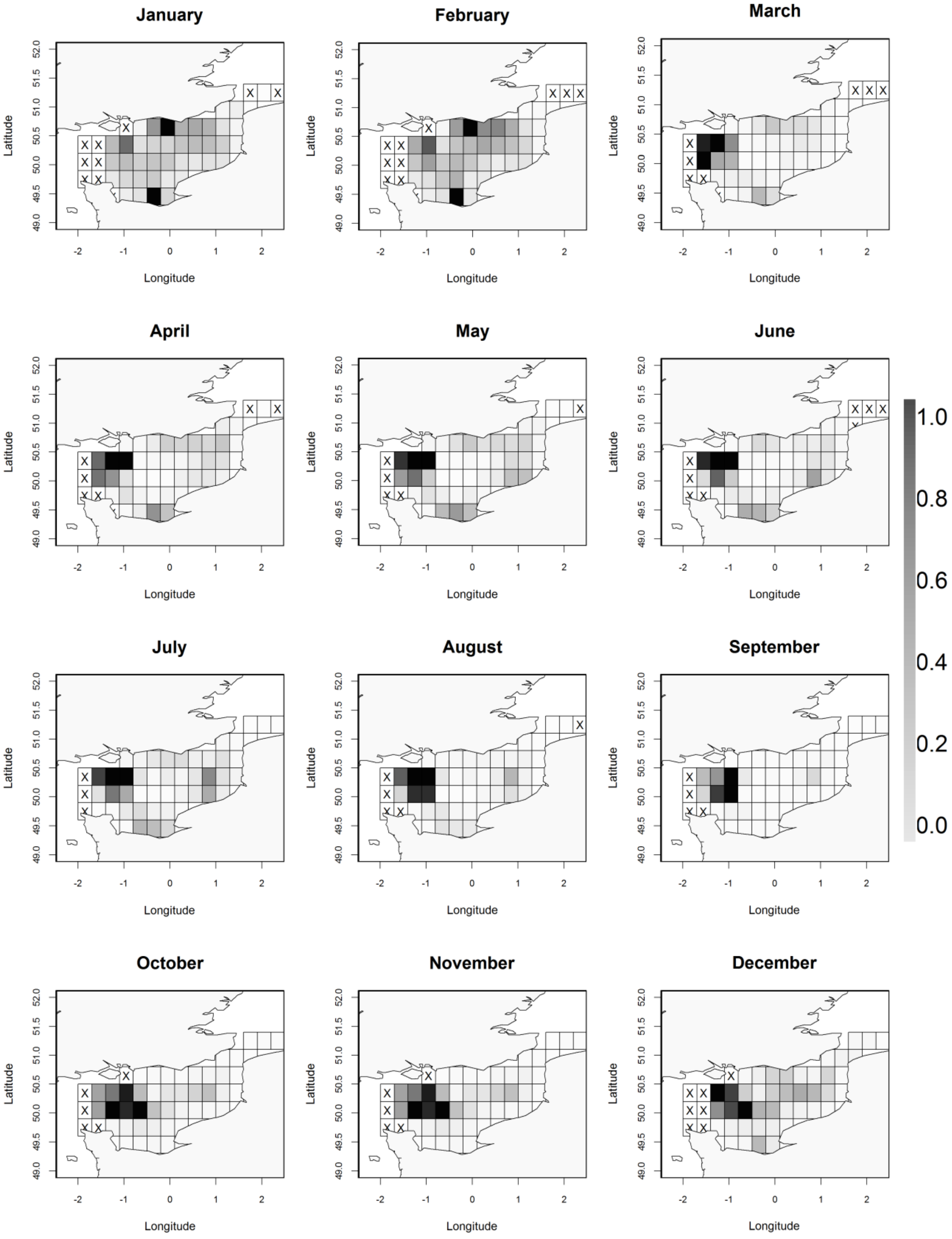


Figure S4.10. Mean monthly relative distribution of fishing effort operated by exclusive bottom trawlers from Boulogne-sur-Mer in Eastern English Channel with the Landing Obligation scenario. “X” represents a null value.

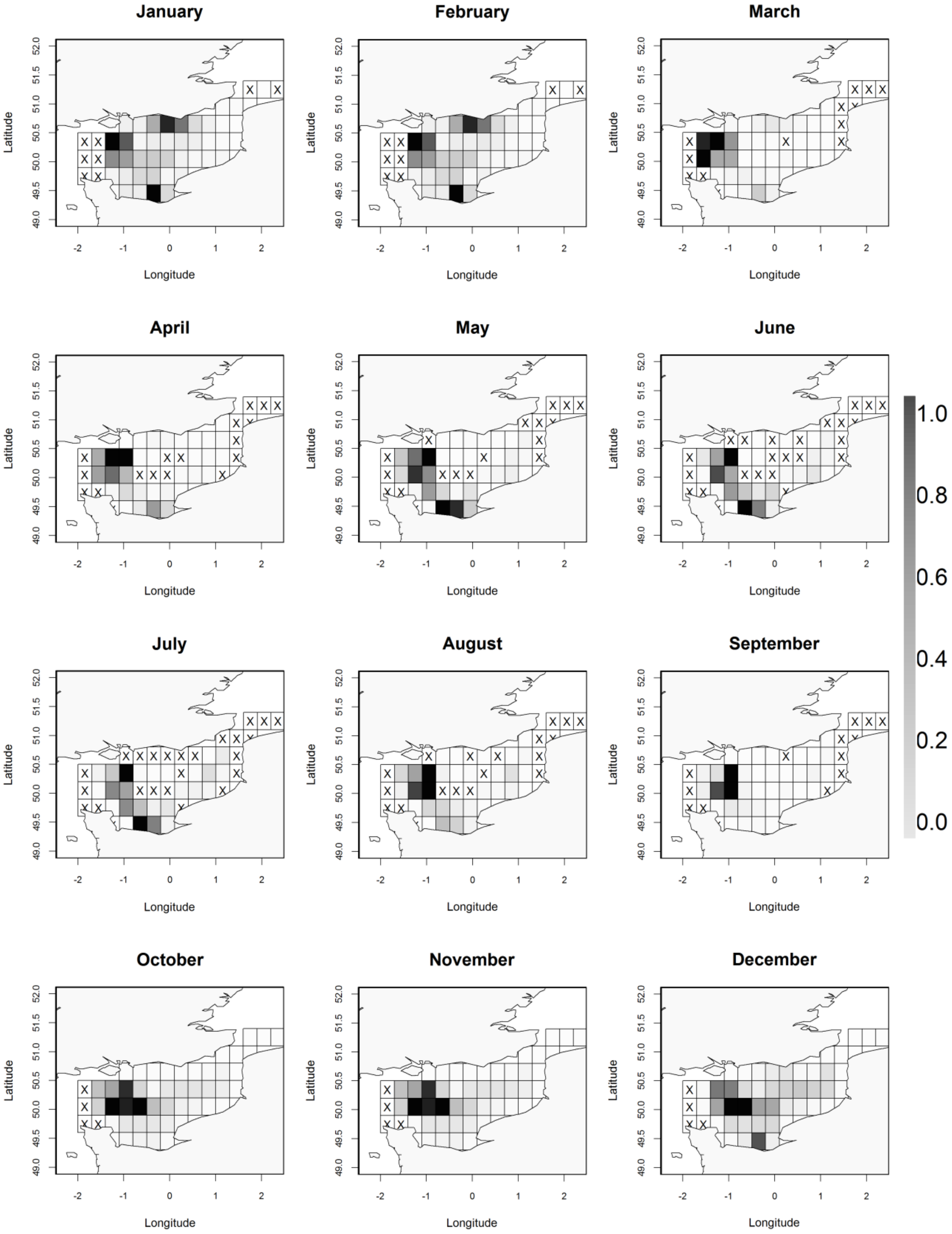


Figure S4.11. Mean monthly relative distribution of fishing effort operated by exclusive bottom trawlers from Port-en-Bessin in Eastern English Channel with the Landing Obligation scenario. “X” represents a null value.

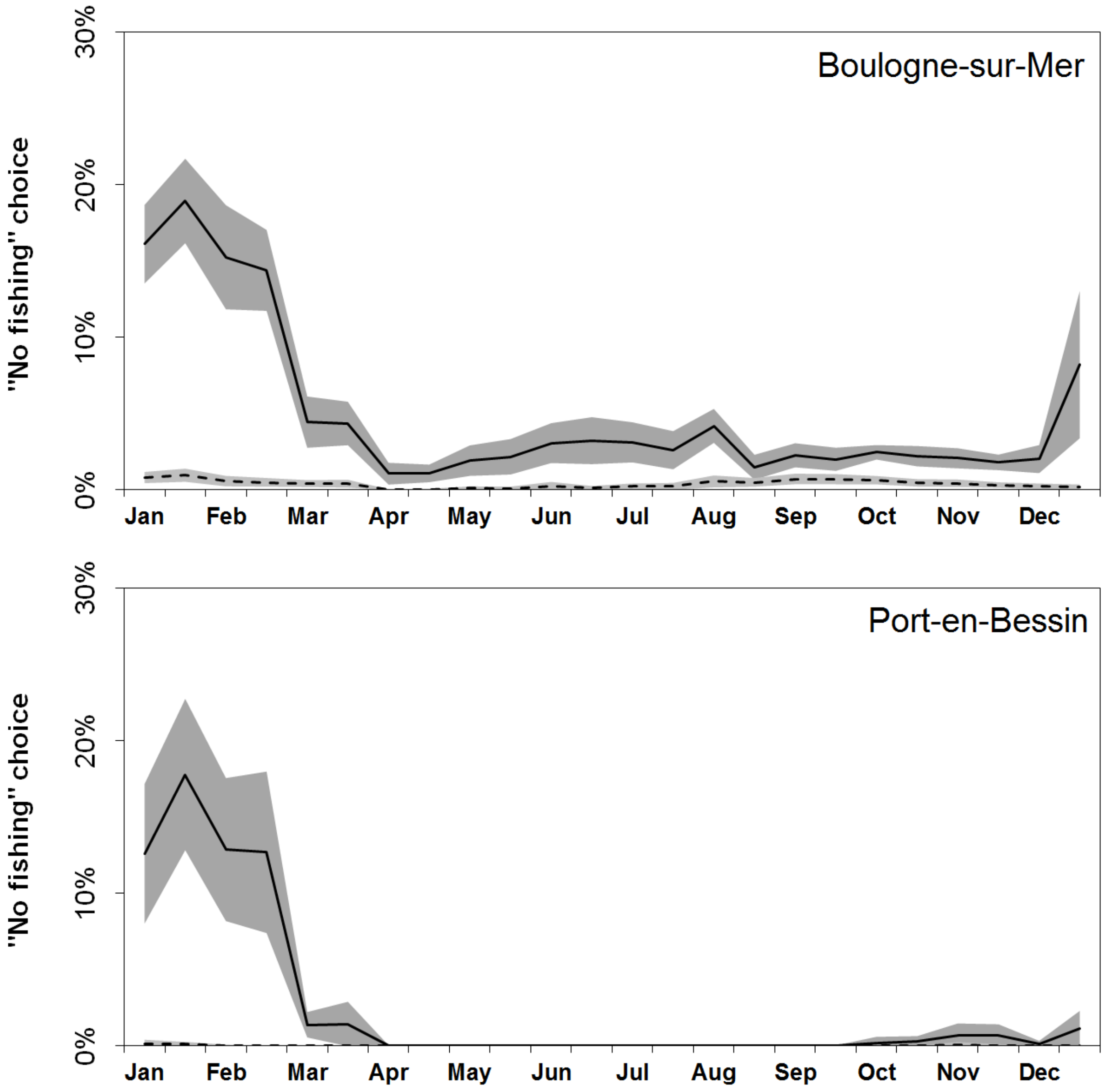
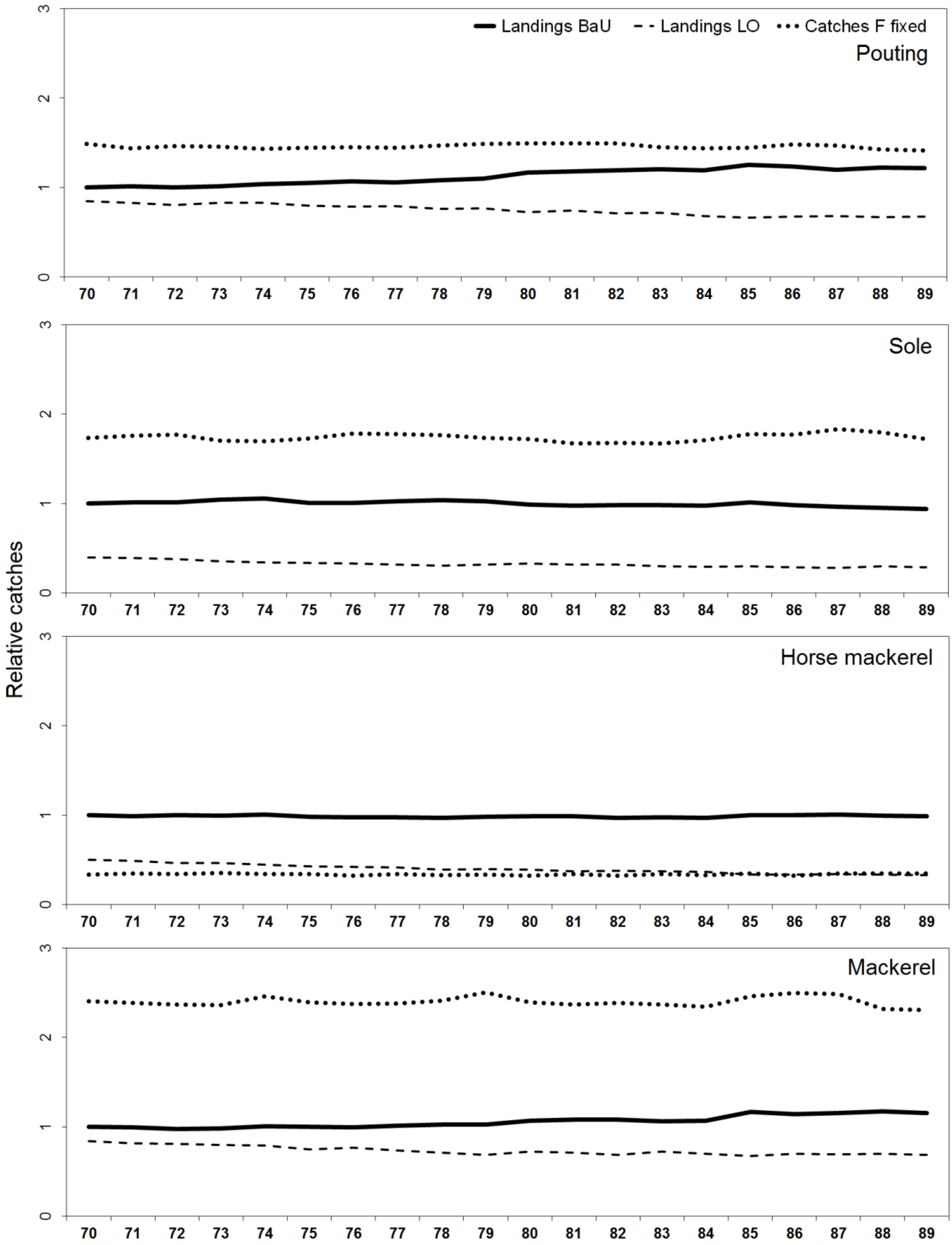


Figure S4.12. Mean percentage of choices not to go fishing done by exclusive bottom trawlers from Boulogne-sur-Mer and Port-en-Bessin in Eastern English Channel with "Business as Usual" (dashed line) and Landing Obligation (full line) scenarios. Standard deviations are represented in grey.


Landings BaU
Landings LO
Catches F fixed
Pouting
Sole
Horse mackerel
Mackerel
Relative catches
0
1
2
3
70
71
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73
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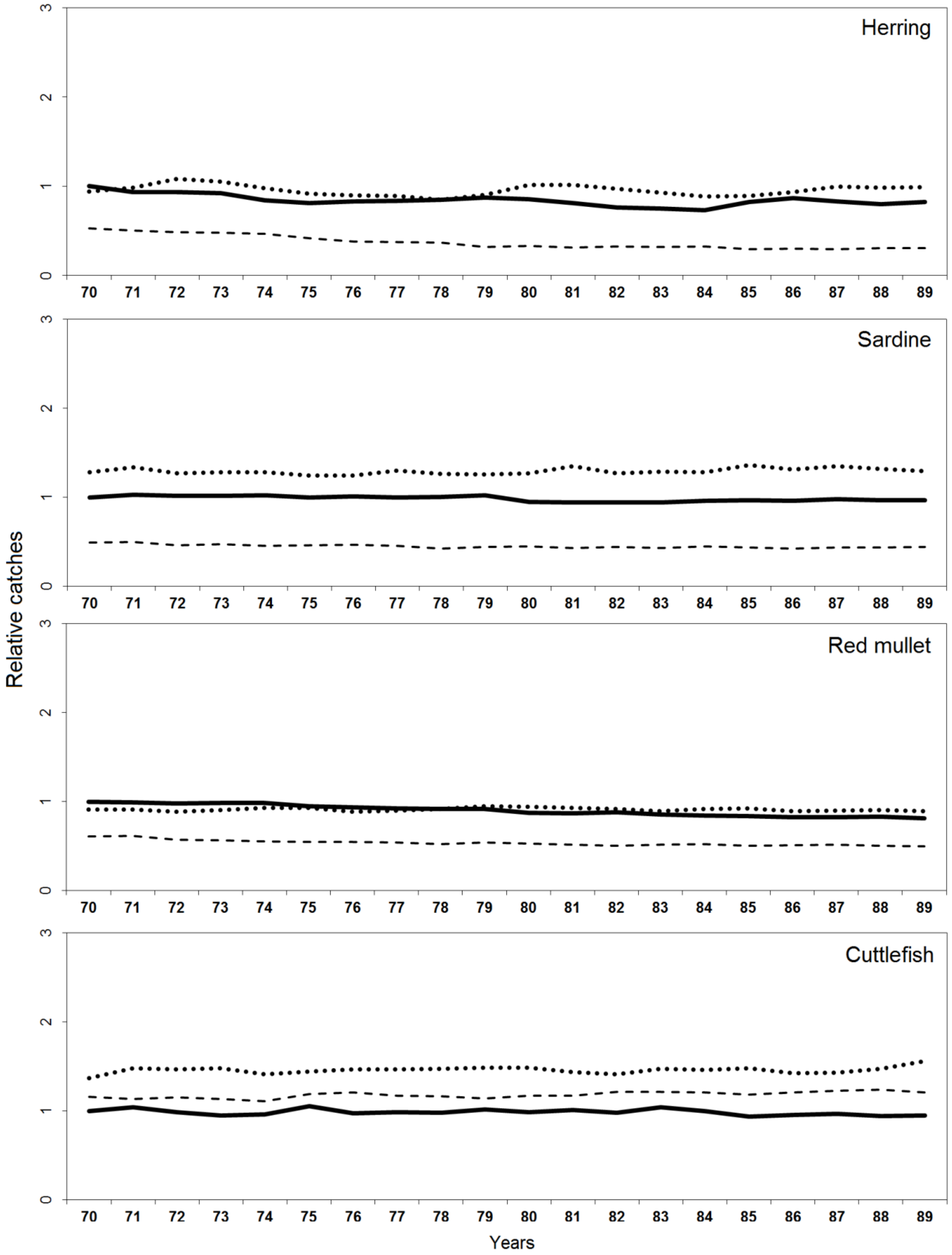

Figure S4.13. Temporal evolution of average catches of pouting, sole, horse mackerel, mackerel, herring sardine and cuttlefish by exclusive bottom otter trawlers in Eastern English Channel with the Business as Usual and Landing Obligation scenarios using DSVM and without DSVM.

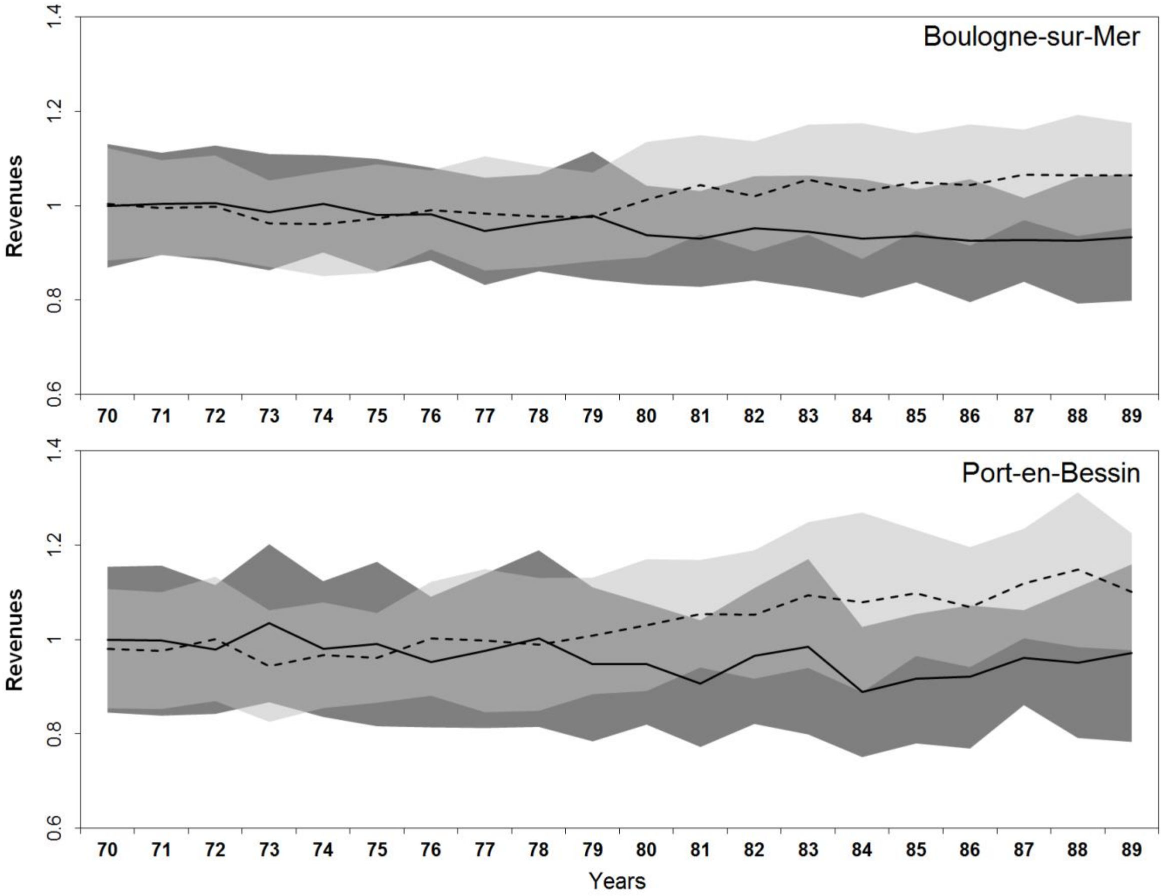

Figure S4.14. Relative changes of the revenue of exclusive bottom otter trawlers form Boulogne-sur-Mer and Port-en-Bessin in Eastern English Channel with "Business as Usual" (full line) and Landing Obligation (dashed line) scenarios. Standard deviations are represented in grey.

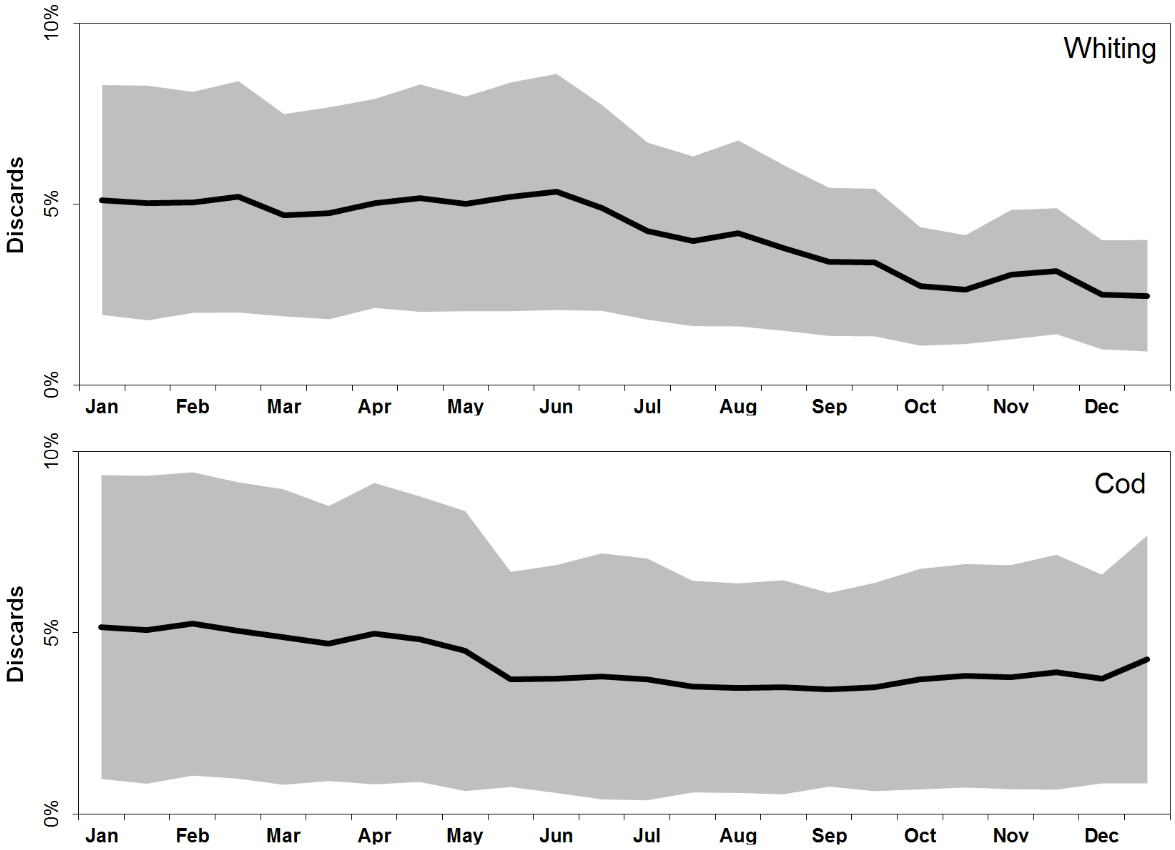

Figure S4.15. Mean percentage of discards of whiting and cod done by exclusive bottom trawlers in Eastern English Channel with "Business as Usual" scenario. Standard deviations are represented in grey.

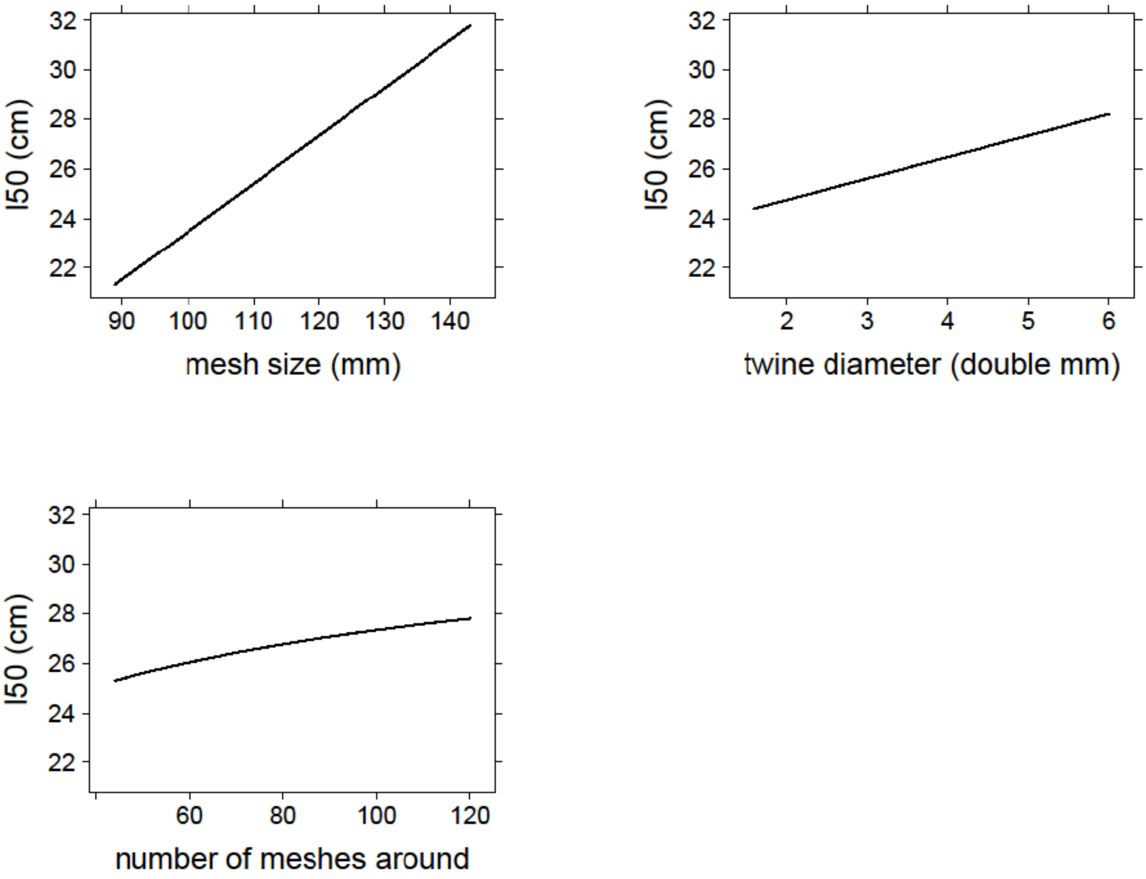

Figure S5.1. The dependence of plaice L50 on codend mesh size, twine diameter and number of meshes around the circumference, estimated from a statistical model based on meta-analysis data. (source; O'Neill and Noble, 2017).

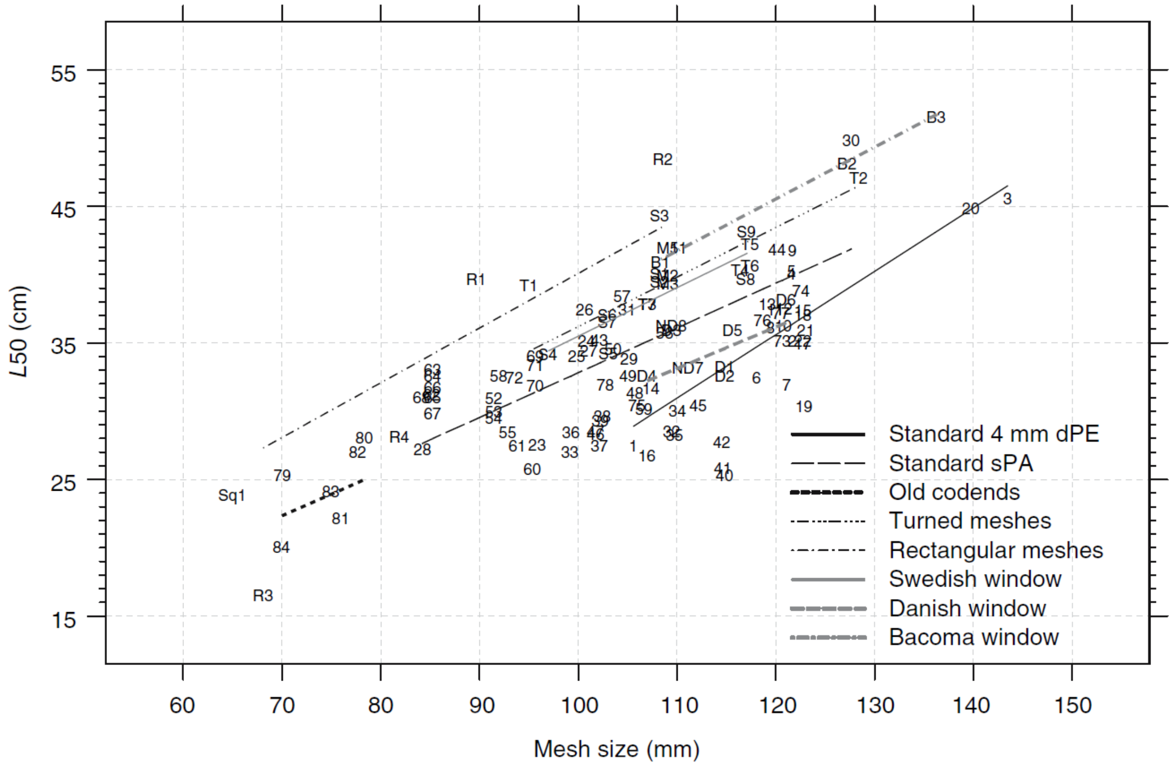

Figure S5.2. Relationship between selectivity parameters and mesh size for cod, estimated from linear regression in the Baltic cod fishery. (source; Madsen, 2007).

Publications

Bourdaud, P., Travers-Trolet, M., Vermard, Y., Cormon, X., Marchal, P. 2017. Inferring the annual, seasonal and spatial distributions of marine species from complementary research and commercial vessels' catch rates. ICES Journal of Marine Science. In press.

Bourdaud, P., Travers-Trolet, M., Vermard, Y., Marchal, P. Improving the perception of fishing effort and pressures in mixed fisheries using spatial overlap metrics. Submitted.

Bourdaud, P., Travers-Trolet, M., Poos, J.J., Vermard, Y., Marchal, P. Impact of a landing obligation on coupled dynamics ecosystem-fishers: individual-based modelling approach applied to Eastern English Channel. In preparation.

Presentations

Oral presentations

Bourdaud, P., Travers-Trolet, M., Vermard, Y., Cormon, X., Marchal, P. 2015. Spatio-temporal abundance index rebuilding from individual commercial data. “Les ecosystèmes marins dans tous leurs états” 12th scientific symposium of the French fisheries Association (AFH). Montpellier, France. (Long presentation)

Bourdaud P., Travers-Trolet, M., Vermard, Y., Cormon, X., Marchal, P. 2015. Spatio-temporal abundance index rebuilding from individual commercial data. Doctoral Day of the ‘Campus de la Mer’. Boulogne-sur-Mer, France. (Short & Long presentation)

Bourdaud P., Travers-Trolet, M., Vermard, Y., Cormon, X., Marchal, P. 2016. Inferring the annual, seasonal and spatial distributions of marine species from combined research and commercial vessels’ catch rates. ICES Annual Science Conference. Riga, Latvia. (Long presentation)

Bourdaud, P., Travers-Trolet, M., Vermard, Y. 2016. Effect of trophic and technical interactions on the definition of MSY reference points in a mixed-fisheries ecosystem. ICES Annual Science Conference. Riga, Latvia. (Long presentation)

Bourdaud, P., Travers-Trolet, M., Poos, J.J., Vermard, Y., Marchal, P. 2017. Fisher’s behaviour and ecosystem state under the EU landing obligation implementation in a mixed-fisheries ecosystem. 13th scientific symposium of the French fisheries Association (AFH). Nantes, France. (Long presentation)

Bourdaud, P., Travers-Trolet, M., Poos, J.J., Vermard, Y., Marchal, P. 2017. Fisher’s behaviour and ecosystem state under the EU landing obligation implementation in a mixed-fisheries ecosystem. Advances in Marine Ecosystem Modelling Research conference. Plymouth, UK. (Long presentation)

Poster presentation

Bourdaud, P., Travers-Trolet, M., Vermard, Y., Marchal, P. 2016. Determining seasonal target species assemblages of bottom otter trawlers from fishing effort and commercial species distributions. ICES Annual Science Conference. Riga, Latvia.

Résumé

Titre : Impact d’une obligation de débarquement sur les dynamiques couplées écosystème-pêcheurs : approche par modélisation individu-centrée appliquée à la Manche orientale

L’objectif de cette thèse était d’anticiper les effets de l’Obligation de Débarquement (OD) mise en place en UE depuis début 2015 en Manche Orientale (MO). Pour accomplir ces objectifs, il a été prévu de : i) mieux comprendre la distribution spatiale saisonnière d’espèces commerciales à l’aide d’observations embarquées sur des navires commerciaux, ii) les comparer avec la distribution de l’effort de pêche à fine échelle des chalutiers de fond (OTB), et iii) développer un modèle individu-centré de dynamique des flottilles, DSVM, à intégrer avec le modèle écosystémique OSMOSE pour simuler l’OD. L’utilité des données d’observations embarquées a été prouvée pour une majeure partie des espèces échantillonnées, en utilisant une validation par la bibliographie et un indicateur géostatistique. Ensuite la comparaison de l’effort de pêche à fine échelle a fourni une amélioration de la quantification de l’effort de pêche effectif et mis en valeur l’importance de la seiche et du rouget barbet pour la distribution des OTB en MO. De plus, l’intensité de ciblage des OTB a été quantifiée en Octobre à l’aide d’un nouvel indicateur, et démontré l’intérêt pour les mêmes espèces, mais aussi la contrainte d’un faible quota de cabillaud pour les pêcheurs. Les résultats du couplage OSMOSE-DSVM montrent que l’OD aurait des effets négatifs à court-terme pour le revenu des pêcheurs, induits par le déplacement de l’effort de pêche afin d’éviter les dépassements de quota, mais serait profitable à moyen-terme. Cependant, l’OD provoquerait une hausse de la pression de prédation produite par le cabillaud et le merlan sur les autres espèces, ce qui n’améliorerait pas l’état global de l’écosystème.

Mots-clés : distribution spatiale, saisonnalité, données commerciales, données de campagnes scientifiques, Manche Orientale, effort de pêche, assemblages d’espèces cibles, obligation de débarquement, modèle individu-centré, dynamiques des flottilles

Abstract

The objective of this thesis was to anticipate the effects if the EU Landing Obligation (LO) implemented since the beginning of 2015 in the Eastern English Channel (EEC). To achieve these objectives, it was planned to: i) better understand seasonal spatial distribution of commercial species using on-board commercial vessels observation data, ii) compare them with the fine scale fishing effort distribution of EEC bottom otter trawlers (OTB), and iii) develop an individual-based model of fleet-dynamics, DSVM, to be integrated within the ecosystem model OSMOSE to simulate a LO. The usefulness of on-board observation data was proved for a main part of a species sample, using validation from the literature and a geostatistical indicator. Then the comparison of fine scale fishing effort provided an improvement of the quantification of effective fishing effort and emphasized the importance of cuttlefish and red mullet for the global distribution of EEC OTB. In addition, the targeting intensity of OTB was quantified in October using a newly-developed indicator, and demonstrated the attractiveness of the same species, but also the constraint of low cod quota for fishers. Results of the OSMOSE-DSVM coupling show that the LO would have short-term negative effects on fishers’ revenue, induced by a large reallocation of their fishing effort to avoid quota over-shooting, but would be profitable in the medium-term. However, the LO would induce an increase of the predatory pressure operated by cod and whiting on the other species, which would not improve the overall ecosystem health.

Keywords: spatial distribution, seasonality, commercial data, survey data, Eastern English Channel, fishing effort, target species assemblages, landing obligation, individual-based model, fleet dynamics