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Application of the eco-viability approach for the management of mixed fisheries under output control

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Application of the eco-viability approach for the management of mixed fisheries under output control

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Submitted in fulfilment of the requirements
for the Degree of Doctor of Philosophy

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in cotutelle with
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"... la connaissance est une navigation
dans un océan d'incertitudes
à travers des archipels de certitudes."

Edgar Morin, *Les sept savoirs nécessaires à l'éducation du futur*

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ABSTRACT

There is now wide acknowledgement that traditional single-species approaches are not adapted to the management of mixed fisheries as they lack the systemic view required to adequately address the ecological, economic and social dimensions of sustainability. The thesis aims to advance the development of more integrated approaches to advise Total Allowable Catch (TAC) decisions in mixed fisheries by: (1) accounting for technical, but also economic interactions among jointly harvested species, and (2) addressing the variety of sustainability requirements faced in such fisheries. The developed methodology calling on the eco-viability approach has been applied to two mixed fisheries operating in different management contexts: the French demersal fishery in the Bay of Biscay (BoB) and the Australian Southern and Eastern Scalefish and Shark Fishery (SESSF).

Explicitly representing the technical and economic interactions structuring mixed fisheries, the ecological-economic simulation model IAM is used to assess the ecological, economic and social impacts of alternative harvest control rules in these fisheries. The thesis notably contributes to improving the representation of human dynamics in the type of models used to advise TAC decisions in mixed fisheries. It particularly discusses the implications in terms of provided advice of accounting for dynamics in the allocation of fishing effort, the trading of individual quotas and fish markets. The thesis also focuses on how the multiple dimensions of sustainability can be considered in the provision of fisheries management advice. The eco-viability approach is used here to identify future paths maintaining a fishery within ecologically, economically and socially acceptable bounds. Specifically, biological sustainability pertains to maintaining harvested stocks above a limit biomass threshold. Economic sustainability refers to the ability of fleets or vessels to ensure the remuneration of both physical and human forms of capital. Finally, an upper limit on the price of fish is considered to ensure that fish

remains affordable to the consumer.

Regarding modelling developments, the thesis presents a process-based model of ITQ (Individual Tradable Quotas) markets in mixed fisheries that captures the economic incentives that ITQ markets provide in a multi-species context, notably those to redirect fishing effort towards species not under quota or the TAC of which is not constraining. Numerical simulations in the SESSF also show the flexibility that exists in the fishery to adjust fishing practices to quota availability, thereby highlighting the importance of accounting for fishing dynamics when advising TAC decisions in mixed fisheries. Finally, eco-viability analyses carried out in both fisheries highlight trade-offs between conservation and economic objectives but also trade-offs pertaining to the distribution of benefits between capital owners, crews, and consumers.

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Presentations to management advisory groups

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Briton F., Thébaud O., Little L.R., Macher C., Gardner C.,Merzeréaud M., Le Grand C., Fifas S., Mobsby D. The eco-viability approach for integrated TAC advice in mixed fisheries. *ICES Working Group for Mixed Fisheries Advice Methodology (WGMIXFISH-METH)*, 22-26 June 2020.	Chapter 2, Chapter 4 and Chapter 5

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CHAPTER 1

GENERAL INTRODUCTION

1.1 Addressing the modern concept of sustainability

1.1.1 Definition and normative foundations

Scientists have mostly associated the notion of sustainability to that of continuance. In this sense, sustainability refers to the quality of being able to maintain oneself over time (e.g. see definitions from the Cambridge Advanced Learner's Dictionary, Oxford Advanced Learner's Dictionary, or Collins English Dictionary) and can apply to systems (e.g. ecosystems, socio-ecosystems), entities (e.g. a species, a business) or processes (e.g. ecological adaptation, resource exploitation, exchange of goods or services). The role of sustainability science is therefore to understand the dynamics of various systems, entities, or processes, and identify the condition(s) for their perpetuation. However, Becker (2012) notes that the modern concept of sustainability cannot be reduced to the sole meaning of continuance as it is also concerned with fundamental human-nature relationships, namely *(i)* those between humans and their contemporaries, *(ii)* those between present and future generations, and *(iii)* those between humans and nature. Such relations have been referred to in political statements on sustainability such as the Bründtland report (WCED, 1987) [1] which defines sustainable development as one "*that meets the needs of the present without compromising the ability of future generations to meet their own needs*" and states that "*the strategy for sustainable development aims to promote harmony among human beings and between humanity and nature*" (Chapter 2). Over the past fifty years, sustainability has been established as a major goal to

1. One can also mention the Johannesburg Declaration (U.N., 2002) Annex II

guide long-term human actions. This aspiration has brought to light and reinforced the ethical dimension of sustainability. The factual analysis of continuance thus only partially addresses the question of sustainability which now involves normative choices regarding, for instance, what ought to be maintained or how should humans relate to their contemporaries, future generations and nature. The reference to ethics is therefore necessary for scientists to state their position. In this regard, this thesis adheres to the normative positioning of Baumgärtner and Quaas (2010) which characterized sustainability as the pursuit of justice in each of the aforementioned relationships, that are, in respective order, *(i)* intra-generational justice, *(ii)* inter-generational justice, and *(iii)* physiocentric ethics. In the specific example of sustainability in the exploitation of a natural resource, intra-generational justice can be interpreted as justice in the distribution of benefits (material or immaterial) within a local community, society, or among nations. Inter-generational justice calls for a fair treatment of future generations regarding the presence or absence of the resource and associated services and benefits. Finally, the question of justice between humans and nature focuses on the importance given to nature in itself with respect to its human valuation. This aspect is at the heart of debates between advocates of weak and strong sustainability, the first assuming substitutability between natural and human capital, an assumption which is refuted by the second (Neumayer, 2003).

1.1.2 Sustainability in a context of uncertainties

In its meaning of continuance and duty regarding future generations, sustainability is deeply rooted in the future. Yet, its establishment as a guiding principle for human actions makes it a matter for present generations. In this regard, Baumgärtner and Quaas (2009) suggest to view sustainability as an attribute of present actions rather than one of future development. However, present decisions for sustainability are thwarted by what Faber et al. (1992) would call “surprises”.

Surprises can relate to the future evolution of the system, entity or process at stake. They can be the consequence of the present generation’s limited understanding of the dynamics involved and unpredictability of certain events. In the typology proposed by Faber et al. (1992), the former would fall under the “risk” (outcomes and associated probabilities are known) or “uncertainty” (outcomes are known without their associated probabilities) categories, whereas the latter would be referred to as “ignorance”. Political declarations on sustainability often refer to the precautionary principle, the application of which is legitimized by the scientific uncertainty regarding future events (González-Laxe, 2005). Defined in the Rio Declaration, the principle states that "*where* *there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation*" (U.N., 1992). Sustainable management is therefore one that must account for uncertainties regarding the consequences of present actions.

Relevant to the question of sustainability are also unknowns pertaining to the needs and preferences of future generations. In this regard, one can see in strong sustainability the recognition of our limited ability to speak for coming generations, justifying that we leave them with the ability to decide for themselves (Howarth, 1995; Baumgärtner and Quaas, 2009).

1.1.3 Viability theory to address the sustainability of renewable resource extraction

Mathematically formalized by Jean-Pierre Aubin in the early 1990's, viability theory has been recognized by many scholars as a relevant framework to address the question of sustainability in the exploitation of renewable resources (see for example a recent review carried out by Oubraham and Zaccour (2018)). Viability theory is a field of mathematics interested in the evolution of (controlled) dynamic systems whose state (and control) variables are subject to a set of constraints.

Solving a viability problem consists in the identification of trajectories X(.) and associated controls (or decisions) U(.) that meet the following dynamics and constraints:

$$\begin{cases} X(t+1) = f(t, X(t), U(t)) \\ U(t) \in \mathcal{B}(t, X(t)) & \qquad \forall t \in \mathcal{T} \\ X(t) \in \mathcal{A}(t) \end{cases}$$

where $\mathcal{A}(t)$ is the set of acceptable states of the system that can be time-dependent, $\mathcal{B}(t, X(t))$ the set of admissible controls which can be time- and state-dependent, $\mathcal{T}$ the time period over which the system is studied, and $f(t, X(t), U(t))$ the controlled dynamics function.

Viability theory introduces several mathematical objects of interest, among which the viability kernel (i.e. the ensemble of viable states, (Aubin, 1991)) or for a system out of its viability kernel, the notion of time of crisis (i.e. the time required for a system in crisis to reach its viability kernel (Doyen and Saint-Pierre, 1997)).

When the set of acceptable states $\mathcal{A}(.)$ reflects constraints pertaining to the continuation of a system's critical funds, functions or services, viability theory addresses the continuance aspect of sustainability. In addition to being strongly related to the idea of continuance, viability theory also deals with some of the ethical foundations embedded in the modern concept of sustainability. As highlighted by Baumgärtner and Quaas (2009), imposing constraints that aim at maintaining both natural and human forms of funds and services bears strong similarities with strong sustainability. Beyond justice between humans and nature, the framework can also incorporate objectives of intra-generational justice by ensuring that funds or services associated to the needs of different groups of people are maintained over time. Finally, when acceptability constraints $\mathcal{A}(.)$ are the same at all times, viable paths are by nature meeting inter-generational equity requirements (Martinet and Doyen, 2007).

Originally developed for deterministic systems, the viability framework was extended to formally address stochastic contexts by De Lara and Doyen (2008), thus enabling to address the uncertain nature of systems, entities or processes at stake.

1.2 Sustainability in fisheries management: the Ecosystem Approach to Fisheries and science in support to its implementation

1.2.1 Ecosystem Approach to Fisheries: the incarnation of sustainability in fisheries management

Concomitant to the progressive affirmation of sustainability as a major political engagement, the need to transition towards a more integrated approach to fisheries management, able to address the multiple facets of sustainability has made its way in the international political arena. The Ecosystem Approach to Fisheries (EAF) -also referred to as Ecosystem-Based Fisheries Management (EBFM)- was introduced in 2001 during the Reykjavik Conference on Responsible Fisheries in the Marine Ecosystem as the framework to operationalize sustainability in the management of fisheries (Garcia and Cochrane, 2005). Recognized by many jurisdictions as the new standard to manage fisheries (EU, 2013; NOAA, 2016; DFO, 2018; DAFF, 2018b) , its purpose was to "*plan, develop and manage fisheries in a manner that addresses the multiplicity of societal needs and desires, without jeopardizing the options for future generations to benefit from a full range of goods and services provided by marine ecosystems*". Primarily focusing on ensuring ecologically sustainable fisheries (Stephenson et al., 2017), fisheries management frameworks are increasingly transitioning towards an explicit integration of the four pillars of sustainability, namely ecological, economic, social and institutional (Benson and Stephenson, 2018; Hobday et al., 2018; Stephenson et al., 2018, 2019; Alexander et al., 2019; Foley et al., 2020).

EAF also aimed to "*balance diverse societal objectives, by taking account of the knowledge and uncertainties about biotic, abiotic and human components of ecosystems and their interactions and applying an integrated approach to fisheries within ecologically meaningful boundaries*" (FAO, 2003). The latter reference to knowledge and uncertainties highlights the crucial role that science has to play in the operationalization of EAF, further stressed by Garcia and Cochrane (2005). Smith et al. (2007) note that science can be involved in three steps of the fisheries management process, namely monitoring, assessment and decision-making. In subsequent developments, greater attention will be given to the two latter aspects as the areas covered in the present thesis. The assessment phase involves a large spectrum of methods, from quantitative models or empirical indicators assessing the status of fish stocks, to ecosystem indicators reflecting on ecosystem state or economic indicators evaluating the performance of fishing fleets. Scientific support to fisheries-related decision-making typically consists in the ex-ante evaluation of management options (e.g. Management Strategy Evaluation [2]) and the development of methods that can help structure and solve decision problems (e.g. Multiple Criteria Decision Analysis [3]).

1.2.2 Development of integrated models to operationalize the Ecosystem Approach to Fisheries

Biological single-species approaches have traditionally dominated the management of fisheries, with management objectives specified as species-specific reference points, stocks assessed individually, and, potentially, management strategies evaluated using single-species Management Strategy Evaluation frameworks (Punt, 2017). Meeting the ambitions of EAF yet requires the development of approaches that capture the ecological, economic and social complexities of the systems at stake, and assess the impacts of management decisions in these three dimensions. Integrated modelling has therefore become an active area of research to support decision-making in an EAF perspective.

2. Management Strategy Evaluation (MSE) refers to the simulation-based evaluation of management strategies to highlight the trade-offs associated to a set of alternatives and assess the consequence of uncertainty on the achievement of management objectives (Punt et al., 2016a).

3. Multiple Criteria Decision Analysis (MCDA) is a field of operations research that aims at solving decision-making problems involving multiple (and usually conflicting) criteria.

Integrated models such as the ones identified by Nielsen et al. (2018) or Melbourne-Thomas et al. (2017) enable the representation of complex systems by accounting for interactions at various scales and across several dimensions.

To support tactical decision-making, the choice has often been to tailor the model to the question asked by only representing the relevant components and processes. It is for instance the case of Models of Intermediate Complexity for Ecosystem assessment (MICE) (Plagányi et al., 2014) which opt for a simplified representation of the ecological realm, hence standing halfway between single-species and whole-of-ecosystem models. Human dynamics in such models have generally remained fairly minimal. In the same vein, bio-economic simulation models tend to focus on the representation of socio-economic dynamics and that of harvested stocks, often to the expense of ecological complexity.

1.2.3 Eco-viability modelling for holistic sustainability assessments of fisheries

Building on fisheries bio-economic models, applications of eco-viability modelling to fishery systems have outlined the potential of the approach to operationalize EAF (Doyen et al., 2017). As a framework assessing the sustainability of complex systems subject to uncertainties, eco-viability indeed addresses the main principles of EAF. Whereas early-age viability models were mostly stylized applications to allow for analytical solutions, more recent applications involving simulation models –generally of intermediate complexity as described in Section 1.2.2- have enabled greater realism in the representation of fisheries systems and accounting of uncertainties in the modelled processes.

As noted by Oubraham and Zaccour (2018), nearly half of the applications of viability theory to the management of renewable resources have been on fisheries study-cases. These applications have considered a variety of sustainability or acceptability constraints. Those pertaining to the ecological viability of the studied systems have mostly been lower bounds on stock abundance, but some authors have also considered minimum thresholds of biodiversity indices (Cissé et al., 2013, 2015) or maximum catch levels for TEP (Threatened, Endangered or Protected) species (Gourguet et al., 2016). Economic viability has generally been interpreted as the generation of positive profits (De Lara and Martinet, 2009; Gourguet et al., 2016), with sometimes consideration of minimum crew remuneration (Maynou, 2019). Finally, the social dimension of EAF has been addressed by constraints pertaining to matters of food security (Eisenack et al., 2006; Hardy et al., 2013; Cissé et al., 2013, 2015), employment (Péreau et al., 2012; Gourguet et al., 2013), recreational catch (Thébaud et al., 2014) or equality in access to the resource (Curtin and Martinet, 2013).

When focusing on the identification of viable management levers (i.e. controls in the viability formalism), viability modelling becomes a useful tool to support decision-making. The main management lever considered in viability studies of fisheries has been fishing effort (Oubraham and Zaccour, 2018), hence providing operational advice for fisheries under effort regulation. Although a large fraction of the world's fisheries, particularly in Europe, Oceania and North-America, are managed using output controls, fewer studies have considered the outputs as control variables (a few exceptions lie in the works of Eisenack et al. (2006); Péreau et al. (2012); Curtin and Martinet (2013) using catch levels or the allocation of catch quota as control variables), thereby underlining a current lack of eco-viability models to advise decision-making in fisheries under output controls.

1.3 Managing mixed fisheries

Mixed fisheries refer to those where the species caught are connected in various ways. Interactions among species in the harvesting process are referred to as technical and involve catching various species either simultaneously through the course of unselective fishing operations or sequentially throughout the season. Species can also be at the heart of economic interactions, with the revenue and costs of their harvest being interdependent [4].

Far from being anecdotal cases, mixed fisheries actually represent the dominant type of fishing worldwide (Cashion et al., 2018) and are found in all regions of the globe. In addition to their large commonality, they are also of great relevance to the study of sustainable fisheries management because of the multi-dimensional (i.e. ecological, economic and social) trade-offs that emerge from the numerous (and uncertain) interactions involved. In this regard, they make good candidates to explore the usefulness of eco-viability approaches to assist in identifying harvesting strategies that meet multi-dimensional sustainability requirements.

4. When attention is also given to the trophic interactions among caught species (i.e. the fact that they are connected in the food web), one usually refers to the term multispecies fishery (Santurtún et al., 2014). The present thesis focuses on mixed fisheries interactions and therefore does not address this latter aspect.

1.3.1 Limitations of single-species approaches for the management of mixed fisheries

The management of fisheries has been dominated by the application of single-species approaches but such approaches are problematic when applied in mixed fisheries for various reasons. First, objectives specified at the stock level are generally not achievable for all stocks simultaneously when technical interactions constrain the composition of catch in mixed fisheries (Vinther et al., 2004; Ulrich et al., 2011, 2017). This can lead to the so-called issue of « choke species » (Schrope, 2010), when low catch limits on a species prevent TACs for jointly caught species to be fully caught, leading to lost yield. Furthermore, low catch limits on species with no or little commercial value can act as an incentive to discard over-quota catches, so as not to lose fishing opportunities on more valuable species. With discard mitigation policies being increasingly adopted globally (Karp et al., 2019), it is important to ensure that TACs in mixed fisheries are consistent with the fleets' technical characteristics to reduce the incentive to discard over-quota catches.

Second, single-species approaches miss the fishery-wide view necessary to assess most of the human-related outcomes (i.e. economic and social) that manifest at this scale. Addressing economic or social management objectives in mixed fisheries therefore requires the development of integrated approaches to ensure that stock-specific regulations achieve fishery-wide objectives. Despite not being an issue specific to mixed fisheries, single-species management also lacks the breadth to address objectives that relate to the broader ecosystem in which fisheries operate.

1.3.2 Specific management arrangements in mixed fisheries

Management arrangements in mixed fisheries have been adapted in various ways to address the limitations of applying single-species approaches. In European mixed fisheries' management plans, targets are progressively being specified as reference ranges rather than points (EU, 2016, 2018b, 2019a,b). Defined as ranges of fishing mortality rates able to provide at least 95% of MSY, these so-called "MSY ranges" allow for some flexibility in the application of objectives still specified at the stock level. As discussed by Rindorf et al. (2017a), not only can they be used to reconcile catch limits with technical constraints, but they also give room for system-wide considerations (e.g. ecological objectives for the broader ecosystem or economic and social objectives for the fishery) to be discussed. Other jurisdictions have chosen to specify management objectives at the scale of the (mixed) fishery rather than that of the stock. This is the case for Australian federal fisheries, where the stated objective of Maximum Economic Yield (MEY) is defined as the maximization of fishery-wide net economic returns (DAFF, 2018b).

In order to avoid commercial fisheries being "choked" by species that have been overfished or the catch of which is prohibited, some jurisdictions have put in place catch limits that aim at covering unavoidable by-catch of the latter when harvesting target species. It is for instance the case of "incidental" Total Allowable Catch (TAC) limits for species under rebuilding strategy in the Australian Southern and Eastern Scalefish and Shark Fishery (AFMA, 2017), or "bycatch TACs" for prohibited species in groundfish fisheries off British Columbia and Alaska (Diamond, 2004).

Flexibility has also been pursued in the quota systems themselves, with for instance centralized (e.g. through Producer Organizations) or market-based (e.g. Individual Transferable Quota (ITQ) markets) transfers of quota, roll-over provisions (i.e. the possibility to save some amount of uncaught quota carried forward to the following season), retrospective balancing (i.e. the possibility to retrospectively acquire quota to match landings), deemed-value payments (i.e. a fee that is charged to fishermen for landings over quota holdings), species equivalence (i.e. the possibility to convert quota for a species into that for another one according to a pre-defined ratio) or basket quotas (i.e. quotas for a group of species rather than individual species) (Sanchirico et al., 2006).

1.3.3 Scientific advice for mixed fisheries: some current approaches and limitations

Scientific input is often sought to support tactical decision-making involved in some of these mixed fisheries management arrangements. The FCube approach has for instance been developed within the International Council for the Exploration of the Sea (ICES) to guide annual TAC decisions and the implementation of MSY ranges in European mixed fisheries (Ulrich et al., 2017; ICES, 2017b). So far, it has been used to provide mixed fisheries advice for the North Sea, Celtic Sea and Iberian waters (ICES, 2017b). In the Australian Northern Prawn Fishery, bio-economic modelling has become integral part of the scientific advice provided to set effort targets maximizing economic returns to the fishing industry (Dichmont et al., 2010).

Despite their adoption in advisory processes, these approaches are facing criticisms regarding, among others, assumptions made on the representation of fishing behaviour in the models. Indeed, the allocation of fishing effort among different activities practised in mixed fisheries [5] has currently been modelled as an exogenous process, an assumption criticized for its lack of realism (Pascoe et al., 2016; Weninger, 2019).

1.4 Thesis objectives

The implementation of EAF requires the development of decision-support approaches covering the complexity and uncertainties of the systems to be managed and able to reconcile multi-facetted management objectives. Mixed fisheries are typical examples of systems lacking adequate approaches to operationalize EAF. In this context, the thesis aims at proposing an operational approach building on eco-viability modelling to support TAC decisions in mixed fisheries. The approach involves: *(1)* capturing the system's processes that are relevant to the TAC setting question, *(2)* identifying appropriate sustainability constraints for the system and relating them to the considered management lever, here the TACs, and *(3)* conveying the results in a way that is informative and useful for decision-makers. Particular developments have been undertaken in relation to these three generic objectives.

With regards to the first point, the thesis aims at improving the representation of mixed fisheries' interactions (technical and economic) in the approaches used to advise TAC decisions in mixed fisheries. The integrated ecological-economic simulation model IAM developed by the French Research Institute for the Exploitation of the Sea (Ifremer) since 2009 to represent the dynamics of multi-species, multi-fleet and multi-metier fisheries (Merzereaud et al., 2011; Macher et al., 2018; Nielsen et al., 2018) is further developed to address this latter objective. In particular, the thesis investigates the implications of endogenizing three key processes pertaining to the human dimensions of fisheries systems which have mostly remained exogenous in models used to provide mixed fisheries TAC advice, namely fishing behaviour, dynamics of fish prices and allocation of quota.

With regards to the second point, the thesis aims at refining the set of sustainability constraints experienced in such systems, with a particular focus on economic and social dimensions. Furthermore, it aims at making eco-viability modelling an operational approach for the implementation of EAF in fisheries under TAC management, and therefore differs from most fisheries eco-viability modelling work in its consideration of

5. In Europe, the notion of metier has been introduced to describe fishing activity based on the gear used and the species targeted (EU 2008).

the fisheries' output (i.e. the catch) as the system's control variable.

Finally, comprehensively advising management with respect to the numerous facets of EAF comes with a particularly challenging communication task. In this regard, the thesis will work at conveying results in a way that is synthetic enough to facilitate decision-making, but analytical enough to appreciate trade-offs underlying TAC decisions in these fisheries.

1.5 Context of the PhD

The PhD has been conducted under a co-tutelle agreement between the University of Western Brittany (UBO) in Brest (France) and the University of Tasmania (UTAS) in Hobart (Australia). Is was co-funded by Ifremer and the UTAS-CSIRO (Commonwealth Scientific and Industrial Research Organisation) Quantitative Marine Science (QMS) PhD program. Additional financial support has also been provided by CSIRO in the form of a ResearchPlus Postgraduate Top-up Scholarship, as well as the Ecole des Docteurs of University Bretagne Loire and the Conseil Régional de Bretagne.

The first year of the PhD was carried out in Brest within the research unit Amure (Ifremer, UBO, CNRS). I was then hosted by CSIRO in Hobart for a year and a half before coming back to France for the final six months of the PhD. The time spent in both countries allowed the approach to be applied in two mixed fisheries under different management arrangements, namely the French demersal fishery in the Bay of Biscay (BoB) and the Australian Southern and Eastern Scalefish and Shark Fishery (SESSF).

1.6 Case studies

The thesis builds on two study-cases: the French demersal fishery in the Bay of Biscay and the Australian Southern and Eastern Scalefish and Shark Fishery. Both are mixed fisheries where several fleets, characterized by different fishing technologies and harvest strategies, capture a couple hundred of species more or less selectively. Joint productions are therefore a critical issue in the management of these fisheries, which mostly relies on output controls in the form of TACs and quotas. They are also important commercial fisheries in their respective regions. Aside from these common features, the two fisheries operate in different management contexts which are detailed in the following sections. The fisheries' main characteristics are summarized in Table 1.1.

Table 1.1 – Comparison of the two studied fisheries: the French demersal fishery in the Bay of Biscay (BoB) and the Australian Southern and Eastern Scalefish and Shark Fishery (SESSF)

Characteristic			Case study: BoB	Case study: SESSF
Landings	Weight (% national capture-fisheries)		66 kton (13%)	12 kton (7%)
	Value (% national capture-fisheries)		303 M€ (23%)	AU$ 55M (3%)
Number species	Retained		~250	~250
	90% value		22	18
Main target and by-product species			*Scalefish*: sole, hake, anglerfish, seabass	*Scalefish*: flathead, pink ling, blue-eye trevalla, orange roughy, blue grenadier
			Crustaceans: Norway lobster	*Crustaceans*: royal red prawn
			Cephalopods: common cuttlefish, inshore squids	*Cephalopods*: arrow squid
				Sharks: gummy shark
Fishing technologies			Bottom trawl, gillnet, Danish seine, longline	Bottom trawl, gillnet, Danish seine, manual and automatic bottom longline, dropline
Nb vessels			710	110
Employment			3187 crew members (3422 FTE)	340 crew members (166 FTE)
Markets			National + export	National
Management	Jurisdiction		European Union	Federal Australia
	Target		Single-stock MSY	Fishery-wide MEY
	Regulation instru-ments	Input controls	Fishing licences, minimum landing size, minimum mesh size	Fishing licences, minimum mesh size, spatial closures
		Output controls	Annual TACs on 16 stocks (13 demersal)	Multi-year (3 to 5 years) TACs on 30 stocks
			Quotas managed by Producer Organisations	Individual Transferable Quotas
	Regulation of discards		Landing obligation	By-catch mitigation measures triggered by Ecological Risk Assessment

1.6.1 Bay of Biscay French demersal fishery (BoB)

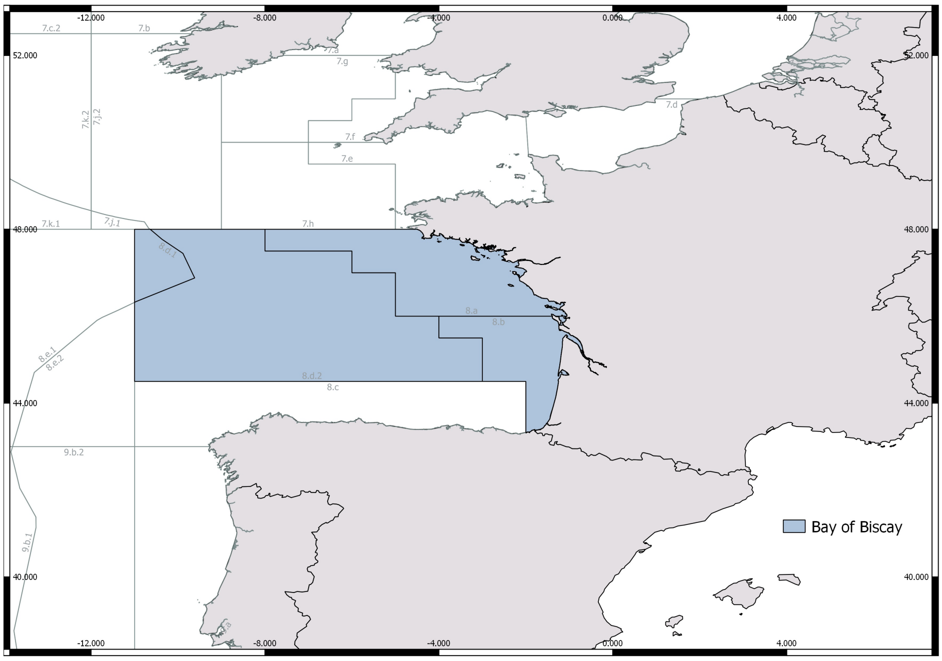


Figure 1.1 – Operating area of the Bay of Biscay demersal fishery (in blue).

The Bay of Biscay (ICES divisions VIIIabd) is characterized by a wide shelf extending west of France (Figure 1.1) supporting important pelagic and demersal fisheries. Although fleets operating in the area come from various European countries (France, Spain, Belgium), the thesis focuses on the French component of the demersal fishery, with the other countries contributions being taken into account as exogenously determined components of the system. Several fleets using different gears (e.g. trawl, gillnet, longline) and characterized by different specialization strategies are involved in the demersal fishery (Macher et al., 2015). As a consequence of poorly selective fishing practices, important technical interactions arise among fleets catching the same species (e.g. between specialized hake gillnetters and specialized Norway lobster trawlers catching hake juveniles).

In 2016, French demersal fisheries in the Bay of Biscay generated a gross value of production of 303 M€(23% of French wild fisheries GVP), 90% of which being accounted for by 22 species among the 250 retained. Demersal species in the Bay of Biscay are both destined to the French (e.g. the quasi totality of Norway lobster landings) and export (e.g. export of hake to Spain) markets. In 2016, 710 vessels participated in the fishery, and employed 3187 crew members.

Management of these fisheries mostly relies on output controls in the form of TACs on the main commercial stocks which are complemented by other conservation measures such as minimum landing sizes or minimum mesh sizes and access is regulated by fishing licenses. TACs are set at the European level and allocated among member States following the "relative stability" principle. States are responsible for the administration of and compliance with their respective quotas. In France, quotas are allocated in the form of collective sub-quotas to Producer Organisations, which manage them according to their own internal rules (Larabi et al., 2013). TAC setting in the EU is guided by ICES' scientific advice, which is consistent with the Common Fisheries Policy's objective of restoring and maintaining fish stocks above levels able to produce MSY (EU, 2013). ICES' TAC advice for demersal fisheries in the Bay of Biscay has so far only relied on single-species stock assessments, without accounting for mixed fisheries interactions. Yet, ICES Working Group on Mixed Fisheries Advice (WGMIXFISH-Advice) is currently working on the provision of mixed fisheries advice for the region using the FCube methodology.

Progressively implemented between 2015 and 2019, the European landing obligation requires for all species under TAC management to be landed (EU, 2013). Flexibility mechanisms have however been introduced such as inter-species quota flexibility or exemptions in the form of survivability or *de minimis* discard allowances.

1.6.2 Australian Southern and Eastern Scalefish and Shark Fishery (SESSF)

The Australian Southern and Eastern Scalefish and Shark Fishery ranges from Fraser Island (Queensland) to Cape Leeuwin (Western Australia) and expands from shallow to deep-water fishing grounds (Figure 1.2). In 2016, it was Australia's largest federal fishery in terms of volume caught and second most valuable (Patterson et al., 2017).

The fishery is composed of four sectors: the Commonwealth Trawl Sector (CTS), the Gillnet Hook and Trap Sector (GHTS) consisting of two sub-sectors, the Shark Gillnet Shark Hook Sector (SGSHS) and the Scalefish Hook Sector (SHS), the East Coast Deepwater Trawl Sector (ECDWTS) and the Great Australian Bight Trawl Sector (GABTS) (Figure 1.2). The thesis focuses on the first two as the latter two are independent and managed separately. The CTS and GHTS are complex multi-species and multi-fleet fishing sectors featuring the typical mixed fisheries interactions. Bottom trawl and Danish seine are used in the CTS to target scalefish and crustacean species

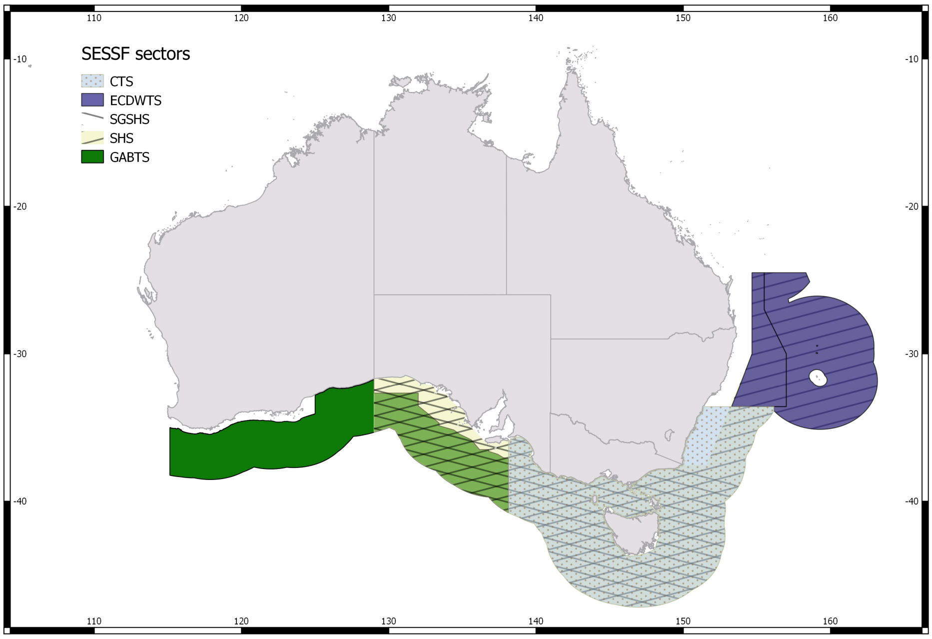


Figure 1.2 – Operating area of the Australian Southern and Eastern Scalefish and Shark Fishery and its different sectors.
CTS: Commonwealth Trawl Sector, SGSHS: Shark Gillnet Shark Hook Sector, GABTS: Great Australian Bight Trawl Sector, ECDWTS: East Coast Deep-Water Trawl Sector, SHS: Scalefish Hook Sector

from shallow waters to depths up to 1000m. In the GHTS, gillnets are mostly used to target shark species, whereas longlines and droplines are used to catch both shark and scalefish species.In 2015, both sectors landed 12,000 ton of fish mostly destined to the Australian market. The gross value of production was AU$55M, 90% of which being accounted for by 18 species.

Management of the fishery is primarily through output controls consisting of TACs generally revised every 3 to 5 years and allocated as individual quotas. The ITQ system was first introduced in 1992 and allows for the lease and permanent transfer of quota units through markets (Connor and Alden, 2001). TACs are complemented by input controls including limited entry, spatial closures and gear restrictions. Like other Australian federal fisheries, management in the SESSF is subject to objectives specified in the Commonwealth Fisheries Harvest Strategy Policy (DAFF, 2018b) for commercial species and the Commonwealth Fisheries Bycatch Policy (DAFF, 2018a) for by-catch species. In particular, the management of commercial species (including target and by-product species) shall aim at maximizing the fishery's net economic returns to the Australian community (i.e. MEY) while ensuring ecological sustainability (DAFF, 2018b). However in practice, management aims to maximise economic returns to capital owners with TACs set to maximise fishery's profits. TAC setting is advised by regular stock assessments undertaken by AFMA Resource Assessment Groups (RAGs). The latter provide Recommended Biological Catches (RBCs) to the South East Management Advisory Committee (SEMAC), which, in turn, make their recommendations to the AFMA Commission for final decision. Despite an explicit fishery-wide MEY objective for multi-species fisheries (DAFF, 2018b), RBCs still correspond to single-stock MEY proxies[6]. Joint productions are however accounted for in the determination of incidental TACs for species under rebuilding strategy, the latter being advised by companion species analyses (Klaer and Smith, 2012).

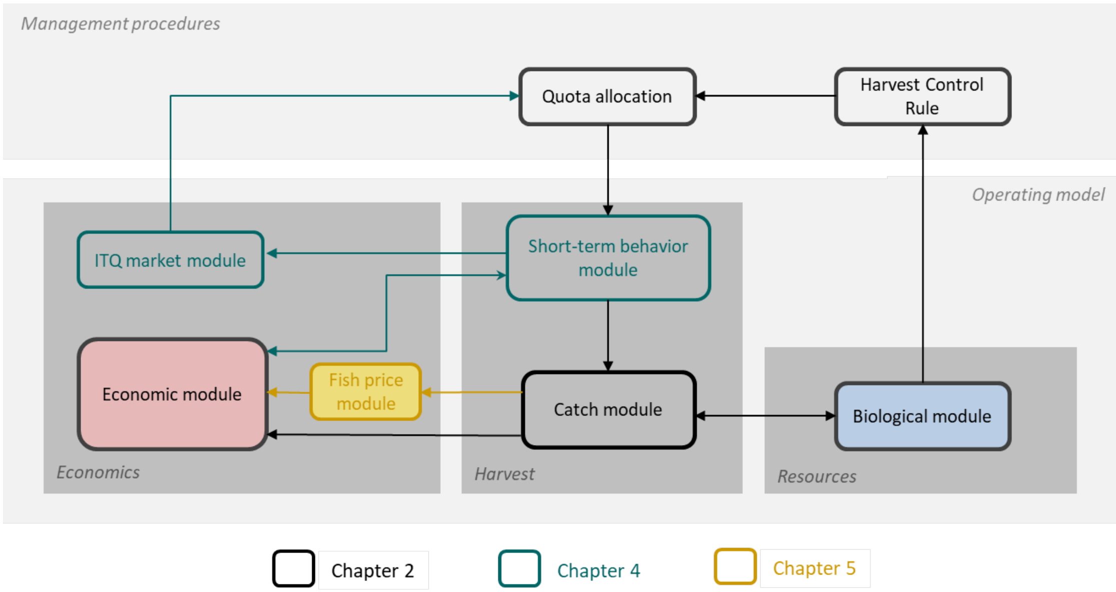
1.7 Structure of the thesis

Besides the introducing and concluding chapters, the thesis is composed of four stand-alone chapters collectively oriented towards the provision of mixed fisheries TAC advice embracing both the complex dynamics of these systems and the multiple constraints that their sustainability entails.

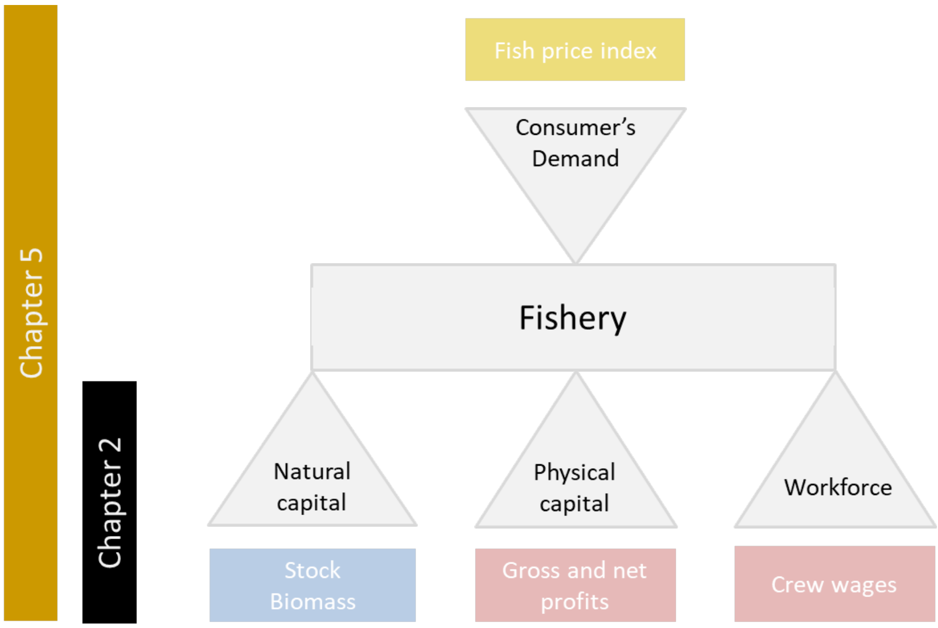
Chapters 2 and 5 apply the eco-viability approach to identify harvest rates for Harvest Control Rules[7] (HCR) in mixed fisheries that meet a set of sustainability/acceptability constraints. Both chapters follow the same methodology: (1) simulation of a range of HCR harvest rates for technically interacting stocks, (2) identification of achievable targets given technical interactions, and (3) evaluation of the latter regarding identified eco-viability constraints. Simulations are carried out using the integrated ecological-economic simulation model IAM. Besides their common purpose and methodology, the two chapters differ mostly on three aspects. First, the study-case: whereas the second chapter treats the French demersal fishery in the Bay of Biscay, the fifth is an application to the Australian Southern and Eastern Scalefish and Shark Fishery. The second difference lies in the set of sustainability, or acceptability, constraints considered. As shown on Figure 1.3B, the focus of the second chapter is on the fishery's inputs. The constraints considered here relate to: (1) the preservation of the fishery's natural capital, in the form of minimum (spawning) stock biomass thresholds, (2) the profitability of the fishery's investment in physical capital, addressed by con-

6. The proxy of stock biomass at MEY is 1.2 that at MSY ($B_{MEY} = 1.2B_{MSY}$). Its equivalent in terms of fishing mortality is $F_{MEY} = 0.8F_{MSY}$)

7. Harvest control rules are a set of pre-agreed rules that determine how much fishing effort or catch is allowed in a fishery. Such limits are a function of an indicator of stock status, which can be a model estimation of abundance derived from a quantitative stock assessment or a direct measurement based on survey or fishery data. The thesis considered a simple HCR defined as a constant fishing mortality rate (or harvest rate) applied to the stock's estimated abundance.



(A) IAM model
Outline colors relate to the chapters that led to the development and implementation of the different modules. Filling colors indicate the modules in which the various eco-viability indicators from Figure 1.3B are calculated.



(B) Eco-viability framework

Figure 1.3 – Evolution of the integrated ecological-economic simulation model IAM and eco-viability framework throughout the thesis

straints on gross and net profits, and (3) the ability of the fishery to maintain its human capital (i.e. its workforce), translated into constraints on the wage of fishing crews. In the fifth chapter, an upper threshold on the price of fish was added to the latter set of constraints to maintain acceptable prices for consumers. Finally, they differ in the degree of endogenization of the systems' "human" dynamics. Benefiting from the developments conducted in the third and fourth chapters, SESSF simulations involve an endogenous representation of the individual allocation of fishing effort among metiers, the trading of ITQs and the evolution of fish prices in response to the quantity of fish landed. Such developments were considered in reaction to the second chapter applied to the BoB, where the constant allocation of fishing effort among metiers, and of quotas among individual harvesters, were identified as potential limits to the conclusions drawn.

The third and fourth chapters contribute to improving the representation of "human" dynamics in the models used to advise TAC decisions in mixed fisheries, specifically highlighting how their consideration impacts the advice produced. The third chapter presents a theoretical analysis of the equilibrium of ITQ markets in mixed fisheries, followed by the numerical application to the SESSF of an algorithm mimicking the Walrassian tatônnement process to reach such equilibrium. The latter algorithm has then been introduced into IAM to model multispecies ITQ markets (Figure 1.3A). Using this new feature of the model, the fourth chapter investigates how the dynamics of ITQ markets might interact with that of fishing behaviour to determine the composition of catch in mixed fisheries.

CHAPTER 2

PROVIDING INTEGRATED TOTAL CATCH ADVICE FOR THE MANAGEMENT OF MIXED FISHERIES WITH AN ECO-VIABILITY APPROACH

This chapter is published as:
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Abstract

Well-established single-species approaches are not adapted to the management of mixed fisheries where multiple species are simultaneously caught in unselective fishing operations. In particular, ignoring joint production when setting Total Allowable Catches (TACs) for individual species is likely to lead to over-quota discards or, when discards are not allowed, to lost fishing opportunities. Furthermore, economic and social objectives have been poorly addressed in the design of fisheries harvest strategies, despite being an explicit objective of Ecosystem-Based Fisheries Management in many jurisdictions worldwide. We introduce the notion of operating space as the ensemble of reachable, single-species fishing mortality targets, given joint production in a mixed fishery. We then use the concept of eco-viability to identify TAC combinations which simultaneously meet biological and economic sustainability constraints. The approach is applied to the joint management of the European hake and common sole fishery in the Bay of Biscay, also accounting for the dynamics of the stocks of Norway lobster and European seabass. Results show that fishing at the upper end of the MSY range for sole and slightly above Fmsy for hake can generate gains in terms of long-term economic viability of the fleets without impeding the biological viability of the stocks, nor the incentives for crews to remain in the fishery. We also identify reachable fishing mortality targets in the MSY ranges for these two species, given existing technical interactions.

2.1 Introduction

Ecosystem-based approaches are increasingly being adopted for the management of natural resources, and fisheries make no exception, with the proposal of ecosystem-based fisheries management (EBFM) guidelines in the early 2000's (García, 2003; Pikitch, 2004), and their subsequent implementation in policy (Pitcher et al., 2009; Link and Browman, 2017).

Among other aspirations, EBFM aims at accounting for the technical interactions among jointly caught species in mixed fisheries. Joint production in mixed fisheries constrains the ability of fishing operators to fully use the quotas they have been allocated for different species. In a management scheme where individual quotas are not transferable, if harvesters stop fishing once they have reached their most limiting quota, any quota they have left for the other species is lost. The existence of unfished quotas may create an incentive for harvesters to continue fishing in order to fully use their fishing opportunities on valuable species while discarding or illegally selling the catches of their "choke species" (Schrope, 2010), i.e. the species for which they do not have quota. In addition to excessive pressure on the stocks of concern, discards of "choke species" can compromise the reliability of catch data underlying stock assessments as they make evaluation of the quantity of fish that is actually removed from the stock more difficult (Rijnsdorp et al., 2007). It is therefore particularly relevant to anticipate any quota under-consumption which may result from joint production. Indeed, minimizing such under-consumption is likely to facilitate compliance with quota regulations.

Attempts to account for the multispecies nature of fisheries in scientific catch recommendations have been developing in many jurisdictions, usually based on adaptations of the historical, and well established single-species assessment framework. In Europe, the FCube framework (Ulrich et al., 2011; ICES, 2015b) has been used by ICES since 2009 to reconcile single-species MSY (Maximum Sustainable Yield) catch recommendations for the North Sea, Celtic Sea, and Iberian Waters mixed fisheries. The introduction of a landing obligation as part of the latest reform of the Common Fisheries Policy (CFP) (EU, 2013) highlighted the potential mismatches between catch opportunities and fishing practices, and their determinant role for the economic viability of a number of European fleets (Simons et al., 2015; Prellezo et al., 2016). A degree of flexibility has been sought with the definition of target ranges around MSY, rather than single target reference points (EU, 2014) and discussions on how such ranges can be used in practice are ongoing (Ulrich et al., 2017). In Australia, the Commonwealth Harvest Strategy Policy identified Maximum Economic Yield (MEY) as a target for management (DAFF, 2007), which has been interpreted in a context of mixed fisheries as maximizing the economic returns from the fishery as a whole (Pascoe et al., 2015). This approach has the advantage of explicitly accounting for the technical interactions observed in mixed fisheries. The practical implementation of this approach, however, has proved difficult, as it requires both a good knowledge of all commercial stocks in a mixed fishery (where only the most valuable stocks are generally well known) and a reliable representation of its economic components (i.e. fleets' cost structures and market prices) (Pascoe et al., 2015; Hoshino et al., 2018).

The move towards EBFM also calls for the formulation of multi-dimensional objectives that integrate the preservation of biological resources as well as the services they provide to society. So far, fisheries management objectives have generally been formulated as the maximization of a quantity, be it the long-term production of the fishery when setting target reference points at the Maximum Sustainable Yield (MSY[1]), or

1. MSY is defined as a limit reference point in the US (Magnuson-Stevens Act, 2007) and in the European Union (Common Fisheries Policy, 2013).

the sustainable economic returns to the fishing industry when aiming at the Maximum Economic Yield (MEY). However, such maximizing approaches often fall short of embedding multi-criteria objectives as argued by Martinet et al. (2010). In Europe, despite the Common Fisheries Policy regulation specifically stating that "the Common Fisheries Policy shall ensure exploitation of living aquatic resources that provides sustainable economic, environmental and social conditions" (EU, 2009), ICES scientific catch recommendations are only based on the evaluation of the stocks status, with no account nor insight into the economic or social impacts of fishing. Based on the observation that ICES catch advice is often disregarded by EU member states fisheries ministers (Villasante et al., 2011; Carpenter et al., 2016), who are ultimately constrained by the social acceptability of their decisions, ICES has recognized the need to provide more integrated management advice (ICES, 2014a) in order to increase the transparency of the decision-making process. The work presented in this paper contributes to the ongoing reflexion among the scientific community on how to incorporate social and economic considerations in their recommendations to decision-makers, with particular emphasis on mixed-fisheries Total Allowable Catch (TAC) advice (ICES, 2016d; Rindorf et al., 2017a; Voss et al., 2017).

Viability theory (Aubin, 1991; Aubin et al., 2011) is particularly well-suited to account for the variety of sustainability requirements faced by a socio-ecosystem, and has been recognized as a relevant assessment framework to support management of renewable resources (review by Oubraham and Zaccour (2018)). Often referred to as co-viability or eco-viability when constraints of various types (e.g. biological, economic, social) are to be met simultaneously, the viability approach consists in identifying paths of a system's evolution that remain within predefined acceptability bounds. As opposed to optimization approaches which require the different objectives to be weighed one against another, the approach gives equal importance to the objectives identified as component parts of the system's sustainability. Indeed, an evolution of the system is considered viable if and only if **all** viability thresholds are respected at **any** time. Such an inter-temporal requirement also allows to account for inter-generationnal equity, as highlighted by Martinet and Doyen (2007); Doyen and Martinet (2012); De Lara et al. (2015). As it does not aim for a particular target but rather looks for a viable operating space, the viability approach seem particularly well suited to address the flexibility requirements that are strongly needed for the management of mixed fisheries (Rindorf et al., 2017a).

Since the mathematical formulation of viability theory in the early 1990's, nearly half of its applications have been fisheries study-cases (Oubraham and Zaccour, 2018). Most study-cases investigated how the inputs (e.g fishing effort, fleet size, investment in the fishery) should be set in order to maintain or restore a fishery's viability (Béné et al., 2001; Cissé et al., 2013, 2015; Gourguet et al., 2013, 2016; Martinet et al., 2010). Considering the fishery's inputs as the control variable in viability studies provides useful advice for the management of fisheries under effort regulation. However, it lacks operationality for fisheries managed under output-controls where the regulation exerts on catches via the definition of Total Allowable Catches (TACs) and quotas. This implementation of catch limits on exploited stocks has now become a keystone in the management of many fisheries worldwide (including in the USA, the EU except in the Mediterranean, Australia, New Zealand, Iceland, and South Africa, among others). Hence, it seems crucial to adapt the viability assessment framework to fisheries under output-controls (and the associated catch-share management systems). Scientific advice in such fisheries is generally given in terms of total catches that can be biologically sustained by the stock, the estimation of which derives from population models when possible, or from the application of some precautionary decision rules. Successfully managing a fishery from the output side not only requires setting appropriate caps on total catches (i.e. *how much can be fished*), but also identifying means of incentivizing efficient prosecution of the fishery (i.e *who gets which share of the catch*), which is widely absent from current ichtyocentric scientific advice. Thinking about by whom and how a quota is fished is far from anecdotal as this can impact *(1)* the stock status, as all gears are not equally selective, *(2)* the economic performance of the fishery, as all harvesters are not equally efficient, and *(3)* how the economic and social benefits (direct and indirect employment, supply of fish, persistence of local knowledge and tradition) generated by the exploitation of a common resource will be redistributed (Symes and Phillipson, 2009).

In the last two decades, the development of Integrated Ecological–Economic Fisheries Models (IEEFMs) has opened the way to a better understanding of the feedbacks between ecological and socio-economic dynamics, and the formulation of management advice embedding both biological and socio-economic assessments (Nielsen et al., 2018). The development of these models also enabled fishing behaviour to be explicitly modelled. Consequently, fishing effort and its direct effect on the resource (i.e. the fishing mortality) emerge as the response of harvesters to an ensemble of regulations and/or other incentives rather than being treated exogenously. Since fisheries management is about seeking to manage fishing activities, not resources (Hilborn, 2007; Fulton et al., 2011), representation of fishers' behaviour in the models provides better insights regarding the expected effectiveness of management options in meeting specified objectives. Our work follows in this vein by explicitly modelling the options fishers face, given catch limits on the different species they catch.

The objective of the present paper is to develop an approach to management advice that integrates **multiple objectives** for **mixed fisheries** under output controls. After describing the IEEFM used to simulate management scenarios, we define the eco-viability framework used to reconcile biological and economic management objectives. We apply the approach to the Bay of Biscay demersal mixed fishery, for which we present a first attempt at providing integrated TAC advice, accounting for multiple objectives. We show that gains can be expected in terms of long-term economic viability of the fleets without impeding the biological viability of the stocks, nor the incentives for crews to remain in the fishery.

2.2 The Bay of Biscay mixed demersal fishery

The Bay of Biscay has historically been an important fishing region in the North-East Atlantic, especially for France. More than 200 species are fished in the Bay but 80% of landings in value were accounted for by 21 species in 2016. The most valuable species are bentho-demersal species, namely Norway lobster, anglerfish, common sole, European hake and European seabass. In 2016, the landings of those 5 species from the Bay generated a gross value of 200 M€.

Fisheries in the Bay of Biscay are managed under the Common Fisheries Policy (CFP), with some managed by coastal states. Management mostly relies on conservation measures (TACs, minimum landing sizes), and about 1/4 of the stocks are managed through EU TAC. Multi-annual plans are also in place for common sole (Council Regulation (EC) No 388/2006) and the northern stock of European hake (Council Regulation (EC) No 388/2006), but should be both replaced by the multi-annual plan for the Western Waters (EU, 2018a), a single regulation embracing demersal and deep-sea stocks, and their fisheries in the Western Waters. TACs are set in line with the MSY objective stated in the CFP. France is allocated a share of the EU TACs following the “relative stability” principle and allocates its national quotas to Producer Organisations (POs) proportionally to their members’ historical catches. POs are responsible for managing their quotas and specifically making sure they are not over-caught (Larabi et al., 2013).Individual harvesters do not own the fishing rights but are limited to fishing what they have been allocated by the PO, which strongly contrasts with fisheries managed with individual (eventually tradable) catch shares.

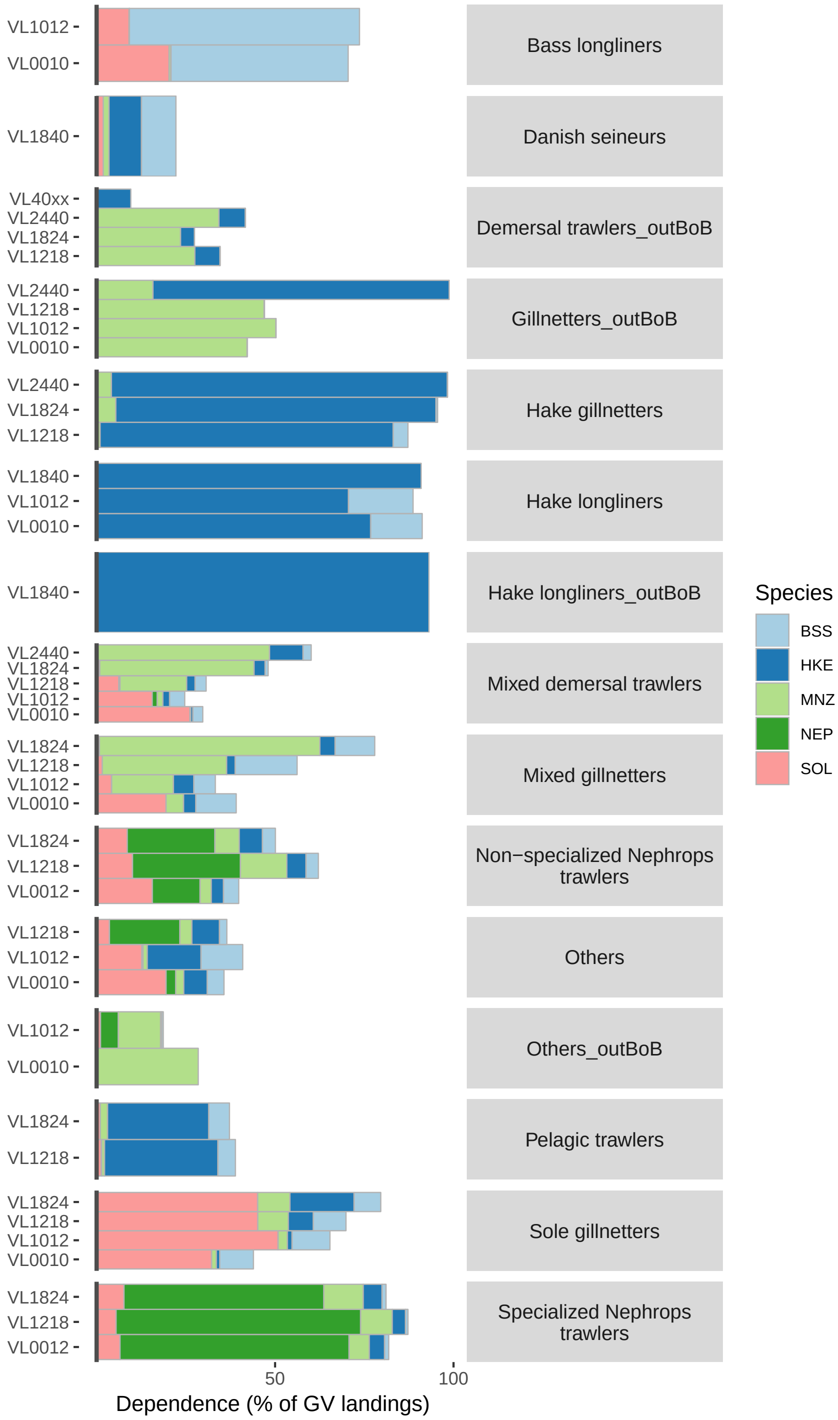

Figure 2.1 – Economic dependence of the demersal fleets on the five key bentho-demersal species. Economic dependence was calculated as the share of each species in the gross value of landings of the fleet. Codes on the left vertical axis correspond to segments of vessels of different lengths for each fleet. Source: Système d'Information Halieutique (2018)

The Bay of Biscay bentho-demersal fishery is a typical mixed fishery where many fleets operate in multiple fisheries, either sequentially because of fishing seasonality or simultaneously because of non-selective fishing practices. As shown on Figure 2.1, some fleets are specialized in targeting particular species which account for most of their revenue (e.g. hake longliners, hake gillnetters, bass longliners and specialized Norway lobster trawlers), whereas others depend on a broader range of species (e.g. mixed netters, sole gillnetters and trawlers). The technical interactions are not accounted for in TAC decisions made at the European level, which can cause discrepancies between fishing opportunities and what is technically achievable for the fleets. In 2016, French quotas for common sole, Norway lobster, whiting and megrim have been fully caught. Those for Pollock, anglerfish and blue whiting were respectively taken up by 87%, 85% and 72% , whereas there was less tension on the hake quota (60% uptake). So far, discards in the fishery have been more related to bycatch of undersized individuals (e.g. small hakes in the Norway lobster fishery) and quality rather than to major choke effects.

Besides ensuring a sustainable exploitation of fish stocks, interviews with representatives from Producers Organisations highlighted that an ageing fishing fleet and the volatility of crews are major concerns in the region. Issues of overcapacity in the region have also been highlighted in the past (Guillen et al., 2013; Gourguet et al., 2013; Bellanger et al., 2018) and addressed through limited entry and publicly funded decommissioning schemes (Quillérou and Guyader, 2012). Although we estimated that in 2016 all 44 fishing segments in the fishery had overall positive gross profits, only 41 showed positive net profits, which highlights the difficulties in renewing the fishing fleet. The economic performance of some segments (e.g. trawlers) is also highly sensitive to the variability in fuel price. Crew volatility is likely to be the reason for wages above the national average as our estimations show that all fishing segments were in average paying their crews a full-time equivalent wage above the mean wage of a French seaman.

2.3 Bio-economic simulation model

Simulations were run with the bioeconomic model IAM referenced in Nielsen et al. (2018) and Macher et al. (2018), and already used to assess the impacts of various management scenarios on the Bay of Biscay mixed fishery (Raveau et al., 2012; Guillen et al., 2013; Bellanger et al., 2018). IAM is a multi-species, multi-fleet or multi-vessel, and multi-metier model that can include stochasticity on recruitment and fish prices.

It runs with an annual time-step and is spatially aggregated. In line with the Management Strategy Evaluation (MSE) approach (Punt et al., 2016a), IAM was divided into an operating model which represents the biological and harvesting components of the system and a management procedure module which implements management regulations.

2.3.1 Operating model

Biological module

Depending on data availability and/or its importance in the modelled system, a stock can either be modelled using annual age-based dynamics, quarterly age-based dynamics, or a static model which assumes that the total biomass of the stock remains constant.

Age-based dynamics are governed by:

$$\begin{aligned} N_{s,a+1,t+1} &= N_{s,a,t}\, e^{-Z_{s,a,t}} \quad a \in [A_{min}; A_{max}-1], \\ N_{s,A_{max},t+1} &= N_{s,A_{max}-1,t}\, e^{-Z_{s,A_{max}-1,t}} + N_{s,A_{max},t}\, e^{-Z_{s,A_{max},t}}, \end{aligned} \tag{2.1}$$

where $N_{s,a,t}$ stands for the number of individuals of age a from stock s at time t, which experience a total mortality $Z_{s,a,t}$ equal to the sum of natural mortality $M_{s,a}$ and fishing mortality $F_{s,a,t}$. The fishing mortality applied to the stock is the sum of the fishing mortalities by vessel i and metier m ($F_{s,a,i,m,t}$) and the fishing mortality exerted by non-explicitly modelled fleets ($F_{s,a,t,OTH}$): $F_{s,a,t} = \sum_{i,m} F_{s,a,i,m,t} + F_{s,a,t,OTH}$.

The quarterly version of this age-based dynamics is detailed in Supplementary Material - Table S.6(a) .

Fishing mortalities at age by vessel and metier are proportional to individual fishing efforts by metier, assuming a constant catchability rate:

$$F_{i,m,s,a,t} = q_{i,m,s,a} \times E_{i,m,t}. \tag{2.2}$$

The spawning stock biomass (SSB) is calculated as:

$$SSB_{s,t} = \sum_{a} Mat_{s,a} w_{s,a} N_{s,a,t}, \tag{2.3}$$

with $Mat_{s,a}$ being the proportion of mature individuals of age a in stock s and $w_{s,a}$ the stock's mean weights at age.

Short-term fishing behaviour module

The short-term behaviour module determines fishing efforts at the metier level for each individual harvester. A metier is a combination of gear and targeted species, and aims at accounting for regional fishing specialisations. Individual fishing efforts at the metier level ($E_{i,m,t}$) ensuing from the quota constraint are calculated in a 2-step process:

1. Calculation of the effort $E_{i,m,s,t}$ required to catch the quota $Q_{i,s,t}$ for each individual harvester i, metier m and stock s . Assuming a constant allocation of the individual total effort $E_{i,t}$ among the different metiers used by harvester i, the problem to solve can be formulated as:

$$\text{Find } \lambda_{i,s,t} \text{ such that } \begin{cases} \sum_m L_{i,m,s,t} = Q_{i,s,t}, \\ E_{i,m,s,t} = E_{i,t} \times \alpha_{i,m}, \\ E_{i,t} = E_{i,t_0} \times \lambda_{i,s,t}. \end{cases} \tag{2.4}$$

Individual landings by metier for species the dynamics of which are explicitly modelled (hereafter referred to as "dynamic species") are given by:

$$L_{i,m,s,a,t} = (1 - d_{s,a})\, w_{s,a} \,\frac{F_{i,m,s,a,t}}{Z_{s,a,t}}\, N_{s,a,t}\,(1 - e^{-Z_{s,a,t}}), \tag{2.5}$$

with $d_{s,a}$ standing for the proportion of discarded individuals of age a and stock s (assumed constant), and $w_{s,a}$ being the individual weight at age in the landings of stock s.
Those for static species are given by:

$$L_{i,m,s,a,t} = LPUE_{i,m,s} \times E_{i,m,t}, \tag{2.6}$$

$LPUE_{i,m,s}$ being the landings of stock s per unit of effort for individual harvester i using metier m.
$\alpha_{i,m}$ is the proportion of total effort of individual i attributed to metier m. For the sake of simplicity, we assume that individual harvesters practice the different metiers in the same proportions as they did in the reference year, that is $\alpha_{i,m} = \alpha_{i,m,t_0}$. This assumption is justified as tradition was shown to be an important driver of fishing practices in other demersal fisheries(Marchal et al., 2013; Girardin et al., 2017). Fishermen's adaptation to seasonal dynamics of exploited stocks, weather conditions, seasonal gear restrictions count among likely explanations of the strong inertia often observed in fishing patterns. Introducing flexibility in the allocation of effort across metiers at the individual fisher level was beyond the

scope of the present study, but will be the focus of further research using the model.

2. Effort reconciliation at the metier level so that each fisherman stops fishing with metier m either when its most constraining quota is exhausted or when he has reached the upper limit E_{max}, i.e.:

$$E_{i,m,t} = \min(E_{max} \times \alpha_{i,m}, \min_{s} E_{i,m,s,t}). \tag{2.7}$$

Economic module

This module calculates for each individual harvester i a variety of economic outputs described in Bellanger et al. (2018).

Among them, three indicators are of particular interest:

— the Gross Operating Surplus (GOS), i.e. the gross income from landings minus all operating costs

$$GOS_{i,t} = (1 - cshr_i) \times rtbs_{i,t} - Cfix_i, \tag{2.8}$$

$$rtbs_{i,t} = GVL_{i,t} - \sum_{m} CvarUE_{i,m} \times E_{i,m,t}, \tag{2.9}$$

$rtbs$ being the "return to be shared" and $cshr$ its proportion allocated to the crew, GVL the gross value of landings, $CvarUE$ the variable costs per unit effort, and $Cfix$ the fixed costs.

— the Net Operating Surplus (NOS), i.e. the Gross Operating Surplus minus capital depreciation costs $Cdep$

$$NOS_{i,t} = GOS_{i,t} - Cdep_i. \tag{2.10}$$

(2.11)

— the Full Time Equivalent (FTE) wage of the crew ($Wage_{FTE}$):

$$Wage_{FTE_{i,t}} = \frac{cshr_i \times rtbs_{i,t}}{FTE_{i,t}}, \tag{2.12}$$

FTE being the full-time equivalent number of men on board a vessel.

2.3.2 Management procedures

The management procedure module is used to set and allocate TACs. At the end of each year, the EU TAC for the following year is calculated so that the stock is harvested under a fishing mortality $\overline{F}_{target}$ according to the procedure described in Appendix A.2. Hereafter, the term management strategy will refer to the specification of fishing mortality targets (and associated catch limitations) for the regulated stocks.

The French quota $Q_{s,t}$ is derived from the EU TAC according to the relative stability principle:

$$Q_{s,t} = TACshr_s \times TAC_{s,t}, \tag{2.13}$$

with $TACshr_s$ the French share of the EU TAC of stock s.

The national quota is then allocated to producer organisations (POs), and in turn to individual harvesters following an allocation key $Qshr_s$ provided as an input [2]

$$Q_{i,s,t} = Qshr_{i,s} \times Q_{s,t}. \tag{2.14}$$

2.4 Eco-viability evaluation

2.4.1 Eco-viability framework

Identifying appropriate acceptability constraints is a determinant step in the operationalization of the viability approach. It consists in:

1. Identifying the elements which determine the persistence of the system, i.e which variables are constrained;
2. Defining the acceptability threshold for the identified variables;
3. And identifying tolerance levels regarding the frequency with which these thresholds should be met in stochastic systems (Thébaud et al., 2014).

For this study we conditioned the viability of the fishing activity on the maintenance of its production factors. First, as any activity based on the exploitation of a natural resource, fishing can only persist if the resource, here the fish stocks, is present. In this regard, the spawning biomass of the stocks should not fall below a limit threshold B_{lim}, under which recruitment is likely to be impaired (ICES, 2015a).
The viability of stock s was thus calculated as:

$$\begin{aligned} V_{BIO}(s) &= 1 \quad \text{if} \quad SSB_s(t) \geq B_{lim_s} \quad \forall t \in [t_0; t_f], \\ &= 0 \quad \text{otherwise.} \end{aligned} \tag{2.15}$$

We also calculated a biological viability index aggregated accross all dynamically

2. We here assume that observed individual allocations of catch shares result from the management operated by POs

modelled stocks:

$$V_{BIO}(\text{ALL}) = 1 \quad \text{if} \quad SSB_s(t) \geq B_{lim_s} \; \forall t \in [t_0; t_f], \forall s \in \{\text{HKE, SOL, BSS, NEP}\}, \\ = 0 \quad \text{otherwise.} \tag{2.16}$$

Second, we consider the fact that fishing companies should be able to maintain their means of production, i.e. capital and labour. Ensuring the renewal of the physical capital for fishing (i.e. vessel, gears, motor...) was expressed as a need to maintain positive Net Operating Surplus. In other words, fishing companies should be able to make sufficient profits not only to cover operating costs (fuels costs, fixed costs and crew costs), but also to cover the depreciation of their capital, in order to be able to renew their equipments when needed. This can be seen as a long-term economic viability constraint, which was not applied on a yearly basis but rather evaluated over the 10-year simulation period ($\overline{NOS} = \frac{\sum_{t'=t_0}^{t_f} NOS(t')}{10}$), deemed a relevant time scale for capital renewal.
The long-term viability of fleet f was thus calculated as:

$$V_{LT}(f) = 1 \quad \text{if} \quad \overline{NOS}(f) \geq 0, \\ = 0 \quad \text{otherwise.} \tag{2.17}$$

In addition to the long-term economic viability of the fleets, we also considered their capacity to regularly cover their operating costs throughout the simulation period, i.e. to show a positive Gross Operating Surplus. Individual economic data (Service de la Statistique et de la Prospective, 2016) showed that having negative gross profits one year did not necessarily prevent fishing vessels from continued operation in the fishery in the following years. This profitability constraint was thus applied on the GOS averaged over a 2-year period ($\overline{GOS}(t) = \frac{\sum_{t'=t-1}^{t} GOS(t')}{2}$). This short-term economic objective is less constraining than the long-term objective defined supra since it does not account for capital depreciation costs. However, it evaluates the regularity of economic performance, which is considered important by the fishing industry.
The short-term viability of fleet f was calculated as:

$$V_{ST}(f) = 1 \quad \text{if} \quad \overline{GOS}(f,t) \geq 0 \quad \forall t \in [t_0+1; t_f], \\ = 0 \quad \text{otherwise.} \tag{2.18}$$

Third, a key production factor to maintain in the fishery is the workforce, especially as interviews with representatives from French producer organisations highlighted the extreme volatility of crews in the region. Keeping fishing crews active in the fishery

was ensured in this application by maintaining their annual full-time equivalent wage above a minimum threshold $Wage_{FTE_{min}}$.
The crew viability of fleet f was calculated as:

$$\begin{aligned} V_{CREW}(f) &= 1 \quad \text{if} \quad Wage_{FTE}(f,t) \geq Wage_{FTE_{min}} \quad \forall t \in [t_0; t_f], \\ &= 0 \quad \text{otherwise.} \end{aligned} \tag{2.19}$$

2.4.2 Eco-viability under uncertainty

Precautionary management requires articulating the acceptability constraints with possible uncertainties on the modelled processes. In our model, stochasticity applies to the recruitment of the dynamic species. De Lara and Doyen (2008) and De Lara et al. (2015) presented how uncertainties could be addressed in the viability framework, thanks to the concept of stochastic viability which is interpreted as maximizing the probability of respecting acceptability constraints. We estimated this probability through a Monte-Carlo simulation approach, which consists in running a number n_{rep} of replicates for which the value of the uncertain factor(s) is drawn in a probability distribution.

For each management strategy St, probabilities were derived for each type of viability: $P_{V_{BIO}}(St,s)$, $P_{V_{ST}}(St,f)$, $P_{V_{LT}}(St,f)$, and $P_{V_{CREW}}(St,f)$. As viability indicators defined in Section 2.4.1 are booleans, the probability of viability was calculated as the sum of the viability indicator over all replicates rep divided by the number of replicates n_{rep}, as :

$$P_V(St) = \frac{\sum_{rep=1}^{n_{rep}} V(St, rep)}{n_{rep}}. \tag{2.20}$$

Examples of stochastic economic trajectories and the associated viability probabilities are given in Figure E.1.

2.5 Model's dimensions and calibration

To select the fleets to model, we considered the four key demersal stocks under EU TAC management in the Bay of Biscay (northern stock of European hake, Bay of Biscay stock of common sole, Bay of Biscay stock of Norway lobster, and anglerfish in the Bay of Biscay and Celtic Sea). Modelled vessels were the French vessels contributing significantly to the landings of at least one of the four key stocks (in order to ensure that > 95% of the landings of each stock was accounted for by the model). In addition, we identified fleets depending economically on a stock as those for which more than 30% of the gross value of landings was made up of landings of this stock. Vessels that were economically dependent on one of the stocks were also included in the model, even if their contribution to landings was limited. In total, 710 vessels were identified and allocated to fleets adapted from the European Data Collection Framework typology of fishing fleets (EU, 2008), to account for regional specificities. The fleets were further divided into length categories to define segments sharing the same cost structures. Each vessel was modelled individually, but results were aggregated at the segment level by averaging economic indicators across all vessels in the segment, in order to represent regional differences in the structure of fishing activities from North to South of the Bay. Fishing activity of the vessels was described through 13 metiers referenced in Appendix - Table B.3.

The 21 most important species or group of species (e.g. anglerfish or megrim) caught by the modelled fleets in the Bay of Biscay were explicitly represented using IAM (list in Table B.1), and all remaining catches were pooled in an "Other species" category. As mentioned in Section 2.3, some species were modelled dynamically, whereas others are considered "static". The possibility to dynamically model a stock is first constrained by data availability: for many stocks, data is too scarce to assess the stock with an analytical model. In the European context, this concerns all stocks which are considered to be data limited (ICES categories 3 to 6 [3]- cf Table B.1). The rationale for the selection of stocks that were modelled dynamically was that:

- The modelled vessels should be important contributors to the total landings of the stock, since any change in the fishing effort of those vessels is likely to impact the stock status. On the contrary, all other things (effort of other fleets, environmental conditions) being equal, it is reasonable to assume a constant biomass of the stocks that are only marginally impacted by the effort of the modelled fleets.
- At least one fleet was economically dependent on the stock. The economic viability of such fleets is likely to be impacted by changes in catch rates (i.e. catches per unit of effort) consecutive to changes in stock biomass.

Only five species met those criteria as shown by Figure 2.2, namely Norway lobster, Common sole, anglerfish, European seabass and European hake. Among those, only Norway lobster, Common sole, European seabass and European hake could be dynamically modelled in IAM as the species of anglerfish were classified as a data-limited.

3. On the basis of available knowledge, ICES classifies the stocks into six main categories. Stocks in categories 1 and 2 are qualified as "data-rich" whereas others fall into the "data-limited" category

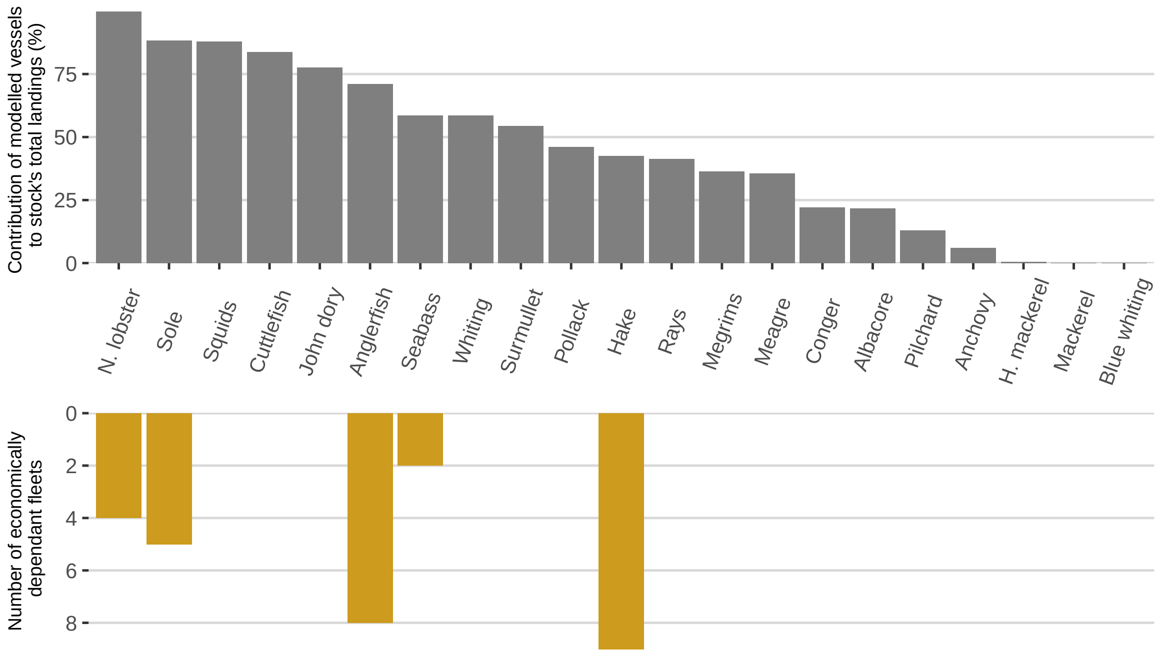

Figure 2.2 – Contribution of the modelled vessels to the total landings of the stocks (top panel) and number of fleets economically dependent on the important commercial species in the Bay of Biscay (lower panel). A fleet was considered economically dependent on a stock when the landings of the latter accounted for > 30% of the fleet's total value of landings. The absence of bar on the lower panel means that no fleet was economically dependent on the stock.
Sources: French landings and value of landings: Système d'Information Halieutique (2018)
Total landings of the stocks: ICES (2018a)

Parameters for the stock dynamics of European hake, common sole and European seabass were derived from the 2016 ICES stock assessments (ICES, 2017a). For Norway lobster, they were estimated from an XSA (Darby and Flatman, 1994; Shepherd, 1999) stock assessment. The recent UWTV survey "LANGOLF-TV", which started in 2014, was included as an additional tunning fleet compared to XSA stock assessments carried out before 2016 (see ICES (2017a) for more details on the survey). All parameters are given in Table B.2. Annual recruitments for those stocks were estimated from Hockey-Stick stock-recruitment relationships. Uncertainties on recruitment were accounted for by randomly sampling the parameters of the stock-recruitment relationship among a list of potential candidates estimated by softwares PlotMSY (sole) or EqSim (hake and seabass) as detailed in ICES (2014b).

Effort and production data by vessel and metier were calculated from the SACROIS database which is an algorithm crossing multiple existing data sources (auction halls, logbooks, dealer reports) to provide the best possible estimation of effort and production by vessel at the trip level (Système d'Information Halieutique, 2018). The maximal fishing effort E_{max} was set uniformly for all vessels at 300 days/year. Fish prices were kept constant and equal to the ones recorded in auction sales in 2016 (Système d'Information Halieutique, 2018). Prices were defined at the vessel and metier level in order to account for spatial variations in prices and differences in prices between gears (e.g. seabass fished with a longline is more expensive than fished with a net or a trawl).

Cost structures were estimated for each fleet segment from 2016 economic data (Service de la Statistique et de la Prospective, 2016). The vessels' cost structures were derived from those estimated for their segment, following the procedure described in Appendix B.3.

$TACshr_s$ was set as the proportion of French landings relative to total landings in 2016. French landings were provided by Système d'Information Halieutique (2018), and total landings by ICES (2018a).

Acceptability constraints and the value of the thresholds for this application are summarized in Table 2.1.

Table 2.1 – Acceptability constraints

Aim	Name	Aggreg. level	Time scale	Value	Source
Stock persistence	$B_{lim_{SOL}}$	stock	year	7,600 t	ICES (2016e)
	$B_{lim_{HKE}}$			32,000 t	ICES (2016a)
	$B_{lim_{BSS}}$			11,920 t	ICES (2018b)
	$B_{lim_{NEP}}$			5,557 t	lowest SSB since 1987 from XSA (pers.com S.Fifas)
Cover operating costs	GOS_{min}	fleet	2-year period	0 €	Varian (2010)
Renew capital	NOS_{min}	fleet	10-year period	0 €	Varian (2010)
Maintain crews	$Wage_{min}$	fleet	year	25,246 €	Mean wage of a French seaman actualized data from 2013 for 2016 - INSEE (cat 692a)

2.6 Management strategies

In order for the results to be displayed in two dimensions, we restricted our analysis to the joint management of two species, chosen for their historic importance in the Bay of Biscay demersal fishery, namely European hake and common sole. We recognize that it is a simplification of current management in the Bay of Biscay since many other stocks are actually under TAC regulation in the region. However, this stylized application has been beneficial both in terms of development and presentation of the approach as outputs were easily tractable and conveyable, and trade-offs between different objectives being made transparent.

In the remainder of the paper, a management strategy St will refer to a couple of target fishing mortalities $\overline{F}_{targ}$ for the 2 stocks under TAC management in the model, namely hake and sole, associated with TAC recommendations. Let $F_{targ_{SOL}}$ be an element of $I_{SOL}$and $F_{targ_{HKE}}$ an element of I_{HKE}, then St is an element of $I_{SOL} \times I_{HKE}$. For simulation purposes this 2D space was discretized in a grid of 20×20 which amounts to 400 simulated strategies. Based on preliminary analyses, the results presented here correspond to $I_{SOL} = I_{HKE} = [0.1; 0.8]$.

Additional strategies corresponding to status-quo targets (i.e. $(F_{targ_{SOL}}; F_{targ_{HKE}}) = (F_{SOL_{2016}}; F_{HKE_{2016}}) = (0.42; 0.27)$) and single-species F_{MSY} reference points ($F_{MSY_{SOL}} = 0.33$ and $F_{MSY_{HKE}} = 0.28$, (ICES, 2015c)) were also simulated. Simulated $\overline{F}_{targ}$ strategies were also compared to the stocks' MSY ranges defined by ICES (2015c) as the fishing mortality ranges resulting in long-term yield no less than 95% of MSY.

Strategies maximizing economic yield of the hake and sole fishery were identified among the simulated strategies. Dynamic maximum economic yield (MEY) refers to the maximization of the fishery's net present value calculated as:

$$NPV = \frac{1}{\sum_t (1+\delta)^{t-t_0}} \sum_{i,s \in (SOL;HKE)} GVL_{i,s,t} - C_{i,s,t}, \tag{2.21}$$

with δ being the discount factor, and $C_{i,s,t}$ the costs of individual i at time t associated to the harvest of species s, which were estimated as a fraction of total costs $C_{i,t} = C_{var_{i,t}} + C_{crew_{i,t}} + C_{fix_{i,t}} + C_{dep_{i,t}}$ equal to the share of species s in the gross value of landings of individual i:

$$C_{i,s,t} = \frac{GVL_{i,s,t}}{GVL_{i,t}} \times C_{i,t}. \tag{2.22}$$

Dynamic MEY was identified for discount factors of 0 and 5%. A static MEY was also calculated as the maximal value of the fishery's net profit at the end of the simulation period.

Strategies ensuring that the TAC of the two species can be simultaneously caught given the joint production formed what we called the *operating domain* of the fishery.

Each strategy was simulated over a 10-year period, and in order to account for uncertainties on the recruitment of dynamically modelled stocks, 200 replicates were run for each strategy [4].

2.7 Results

2.7.1 The joint production problem

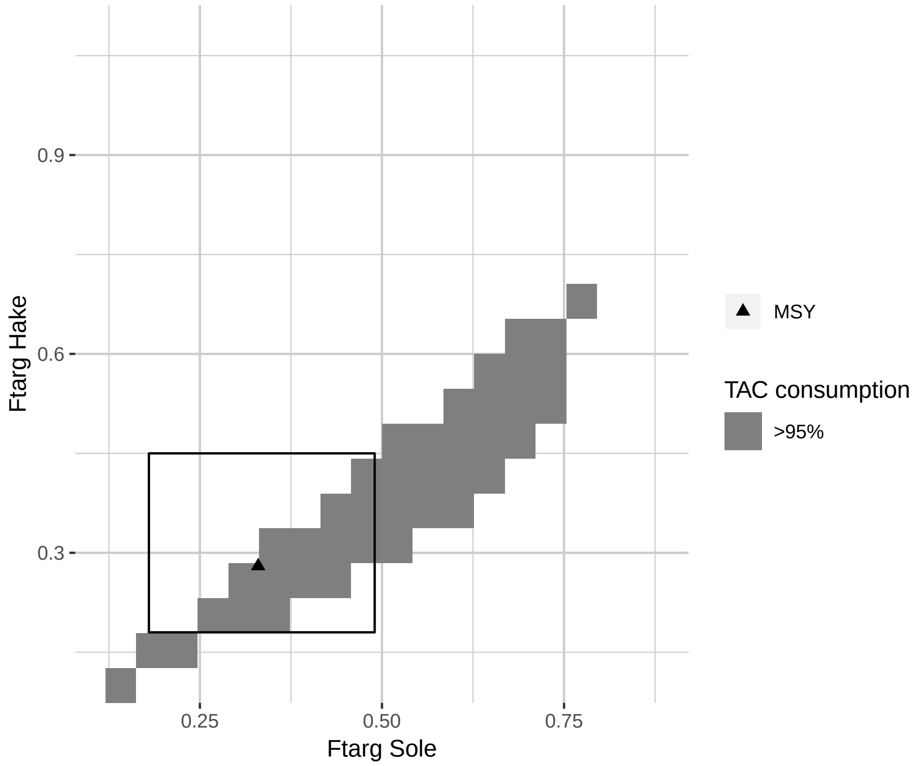

Figure 2.3 – Operating domain of the fishery defined as the domain where at least 95% of the quota for both species can be simultaneously caught given joint production constraints. The triangle shows the F_{MSY} reference points of the 2 species and the black rectangle their F_{MSY} ranges.
Source: output from IAM model

4. Increasing the number of replicates from 100 to 200 did not impact the results at the scale at which we present them

Figure 2.3 displays in grey the operating domain of the hake and sole fishery in the Bay of Biscay. If production of hake and sole for the modelled vessels was completely joint, i.e. every harvester fishing hake with one unit of effort of a given metier would also fish sole with the same unit of effort of this metier and vice versa, then the operating space would reduce to a line. In our model representation of the Bay of Biscay demersal fishery, it is not exactly a line, but is closer to a segment: the modelled vessels are not able to fully consume their quotas if the TACs are based on target fishing mortality rates greater than 0.77 for sole and 0.68 for hake. At this point, the vessels are not constrained by the quotas but by the maximal effort limit E_{max} (which truncates the theoretical perfect joint-production line into a segment). The operating space is not strictly uni-dimensionnal: some latitude around the perfect joint-production line exists. This room for manoeuvre is due to the fact that production of the two species is not perfectly joint: some vessels do catch both species jointly whereas others either only fish one species or do not fish both using the same metier. In the latter case, fishing operators are able to target both species separately and fully use their quotas on both species.

As shown on Figure 2.3, the single-species F_{MSY} reference points fall within the operating domain, which means that they are achievable targets given the joint production observed in the fishery. Figure 2.3 also shows that MSY ranges intersect with the operating domain, although they also include combinations that are unreachable.

2.7.2 Biological viability

We then proceed to the identification of biologically viable strategies. As shown on Figure 2.4, Norway lobster is the most vulnerable species among the 4 dynamically modelled stocks. Ensuring that its spawning biomass does not fall below B_{lim} with a probability of 95% restrains the operating domain presented in Figure 2.3 to the dark green domain on Figure 2.4, which corresponds to target fishing mortality rates below 0.6 for sole and 0.47 for hake.

2.7.3 Fleets' viability

The economic viability of the biologically viable operating domain (i.e. the dark green domain in Figure 2.4) was then evaluated. For each segment we calculated its probability to meet the 3 viability constraints described in Section 2.4.1. A segment was considered viable in a given strategy if its probability of viability was greater than 80%. Figure 2.5 shows the number of viable segments for each strategy with respect to the three constraints. The number of segments able to maintain positive gross or net

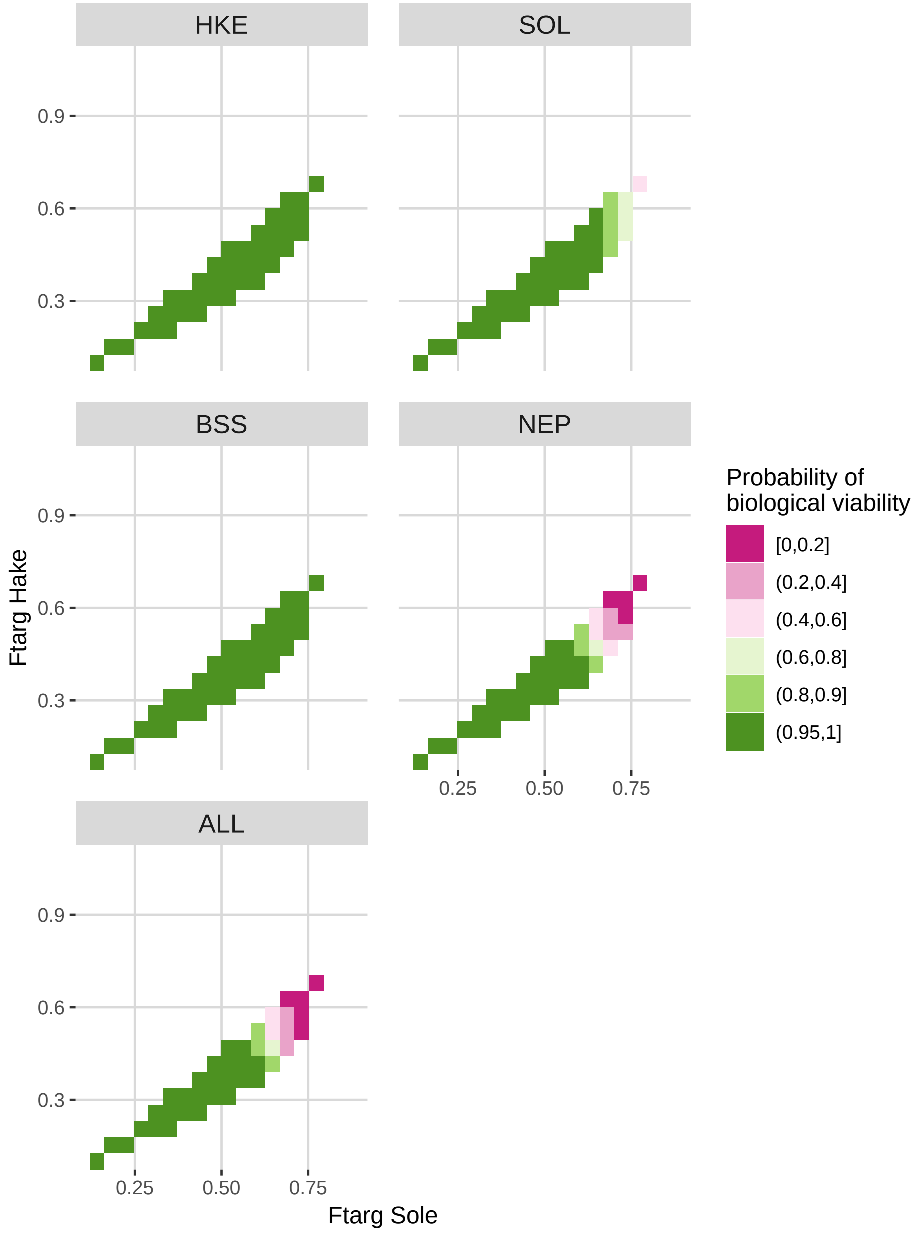

Figure 2.4 – Probability of biological viability of the operating domain by stock and aggregated for all stocks.
Source: output from IAM model

profits over the simulation period, increases with fishing mortality targets. In 2016, all 44 segments had a positive gross operating surplus, which means that setting fishing mortality targets lower than 0.31 for sole and 0.26 for hake will jeopardize the short-term viability of initially viable segments. Regarding long-term viability, 41 segments showed a positive net operating surplus in 2016. Thus, setting fishing mortality targets higher than 0.39 for sole and 0.31 for hake will enable more segments to meet long-term viability objectives than in 2016.

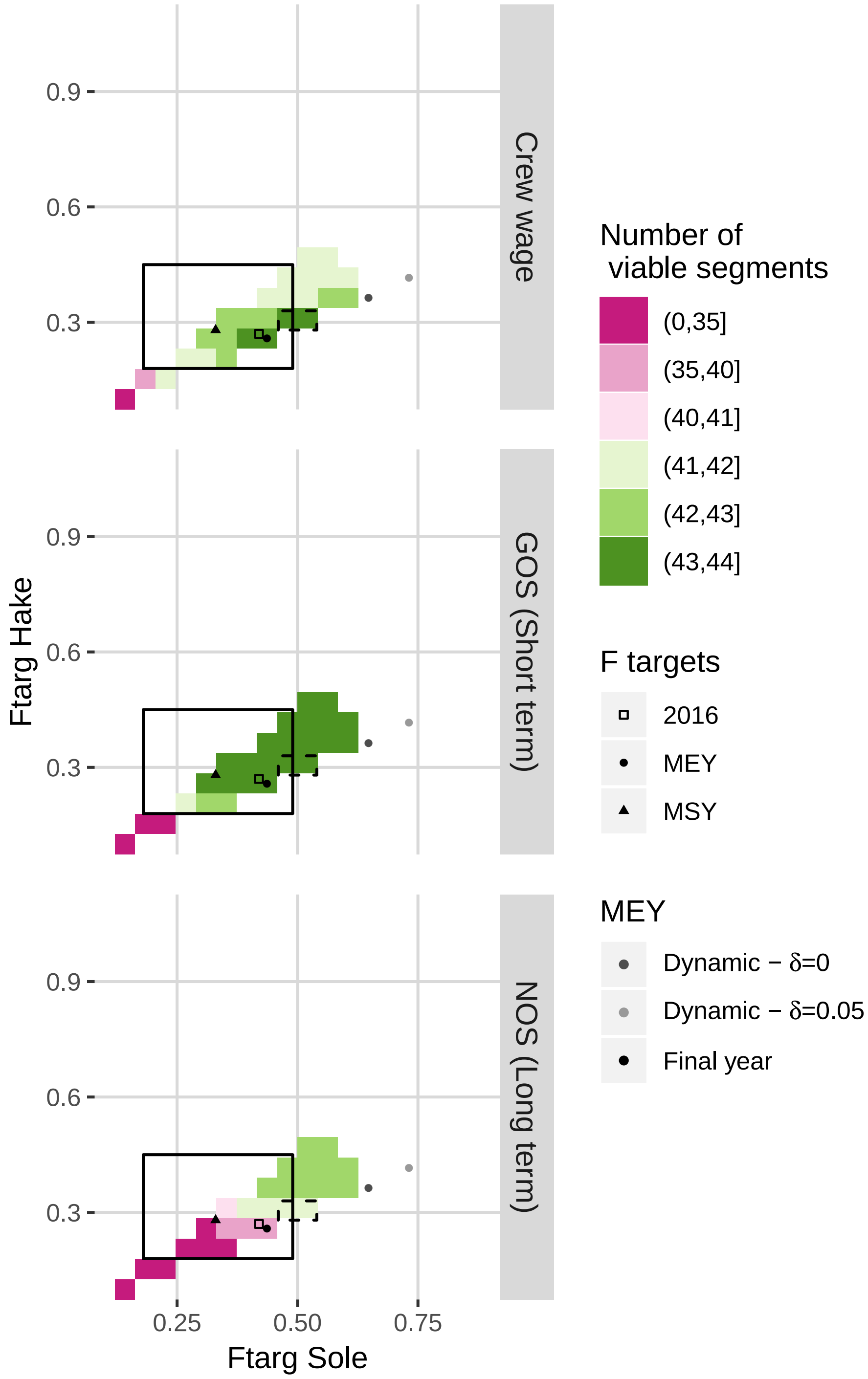

Figure 2.5 – Number of viable fishing segments regarding the three economic constraints: short-term viability, long-term viability, and the ability to maintain fishing crews. The solid lined rectangle refers to the 2 species' F_{MSY} ranges and the dashed lined rectangle shows the maximum viability space.
Source: output from IAM model

The ability to maintain crews in the fishery shows a somewhat different evolution.

The number of segments able to maintain their crews first increases with fishing mortality targets, until all segments are able to provide a sufficient FTE wage to their crews. Then, as fishing mortality targets increase above 0.52 for sole and 0.31 for hake, the decrease in labour productivity (i.e. the crew share per fishing hour) leaves some segments unable to guarantee the minimum FTE wage to their crews.

Interestingly, harvesting both stocks within their MSY ranges is relatively compatible with short-term economic viability and crew maintenance objectives since both criteria are always met by more than 41 segments within those ranges. However, targeting the lower end of the operating MSY ranges ($F_{targ_{SOL}} = 0.27$ and $F_{targ_{HKE}} = 0.21$) will prevent 17 segments from meeting long-term economic objectives, whereas fishing at its upper end ($F_{targ_{SOL}} = 0.48$ and $F_{targ_{HKE}} = 0.42$) allows 43 segments to reach long-term economic viability.

In no strategy can the three sustainability criteria be met simultaneously for all segments. However, with fishing mortality targets between 0.48 and 0.52 for sole and of 0.31 for hake (dashed-lined rectangle in Figure 2.5), all segments meet the short-term viability and crew wage constraints, and 42 over 44 segments generate sufficient profits to ensure the renewal of their capital. Targeting fishing mortality rates in those ranges therefore ensures that the fleets' viability at least improves compared to 2016. This domain of fishing mortality targets will be further referred to as the "maximum viability space".

On Figure 2.5 strategies that maximize economic yield from the hake and sole fishery are also identified. Maximizing the net present value of the fishery over a 10-year period leads to high harvest rates on both species ($F_{targ_{SOL}} = 0.73$ and $F_{targ_{HKE}} = 0.48$ for $\delta = 5\%$, and $F_{targ_{SOL}} = 0.65$ and $F_{targ_{HKE}} = 0.36$ for $\delta = 0$), which fall out of the biologically viable domain. This suggests the prevalence of short-term over long-term profits since the fishery's profits at the end of the simulation period are maximized under much lower harvest rates ($F_{targ_{SOL}} = 0.44$ and $F_{targ_{HKE}} = 0.26$). In other words, within a 10-year time frame, it is more profitable to highly fish now than to reduce fishing to increase tomorrow's profits. It is worth noting that the fishing mortality target which maximizes the fishery's net profits at the end of the simulation period is lower than F_{MSY} for hake but higher for sole.

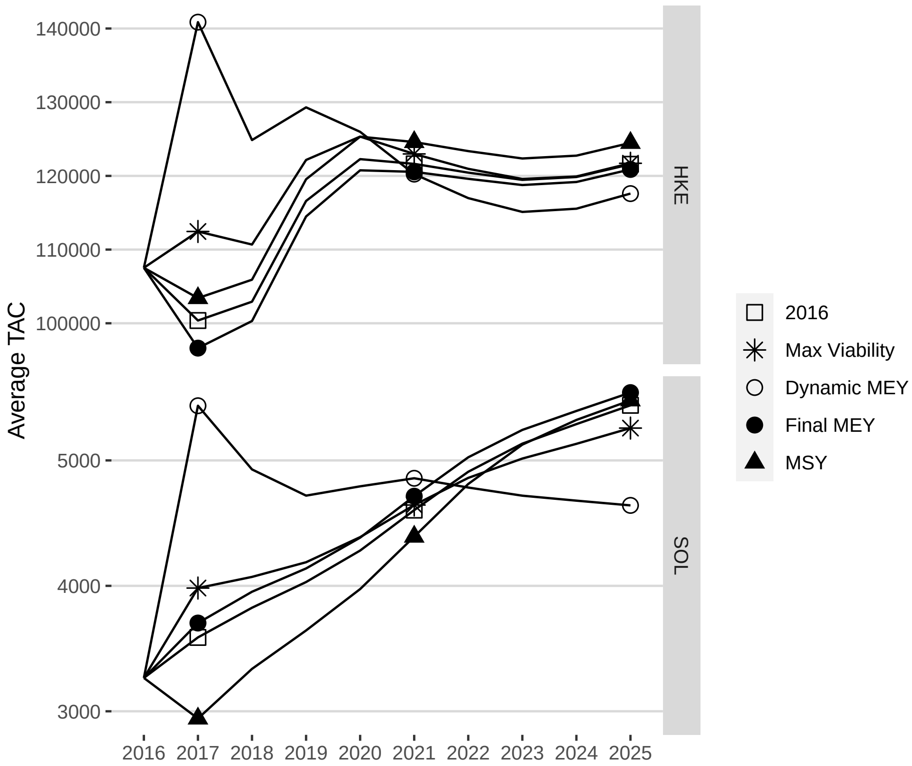

Figure 2.6 – Time series of predicted TACs for identified strategies: single-species F_{MSY}, Status quo F_{2016}, Maximum viability, and MEY.
Source: output from IAM model

2.7.4 From target reference points on fishing mortality to TAC advice

Of particular interest for decision-makers are the TACs associated with the recommended fishing mortality targets. Mean predicted TACs between 2017 and 2025 for sole and hake for some strategies are represented on Figure 2.6. The maximum viability strategy corresponds to $F_{targ_{HKE}} = 0.31$ and $F_{targ_{SOL}} = 0.48$, and the dynamic MEY strategy is the dynamic MEY with a discount rate of 5%.

In the first year the harvest control rule applies, the higher the fishing mortality target, the greater the resulting TAC. However, this correlation inverts in the longer term as more conservative strategies lead to more abundant stocks, and hence higher TACs. In the case of sole and hake, benefits from short-term restrictions on fishing mortality targets can be expected after 5 to 6 years, when the less conservative strategy (dynamic MEY) starts resulting in lower TACs than more conservative ones. In the long-term, all identified strategies result in higher TACs than in 2016 although some lead to lower catches in the short-term (MSY for sole and MSY, status quo and final

MEY for hake). In addition to these trends, inter-annual variations in TACs can also be quantified. In the status-quo, MSY and final MEY strategies, year to year variations in TACs never exceeded 15%, a threshold commonly requested by the fishing industry to stabilize catch possibilities in EU multi-annual management plans (Penas, 2007). The dynamic MEY strategy, however, led to TACs on sole and hake respectively increasing by 66% and 31% between 2016 and 2017. In the maximum viability strategy, the TAC on sole increased by 22% in the 1st year of implementation (Table 2.2).

The maximum viability strategy features as an intermediate strategy, neither too conservative to ensure the economic viability of the greatest number of segments during the entire projection period, nor too bold, so as not to impede the reproductive capacity of any single stock. Consequently, it allows for catches to be maintained at quite high levels over time, without deviating too much from the MSY strategy. Indeed, at the end of the 10-year projection period, catches of sole and hake in the maximum viability strategy are expected to be respectively 95% and 97% of those in the MSY strategy. It is also the strategy with the second highest NPV, after the MEY strategy as shown in Table 2.2.

Table 2.2 – Summary table of selected strategies, compared in terms of the Net Present Value (NPV) of the hake and sole fisheries (calculated with a discount rate of 5%) and the maximum inter-annual absolute TAC variation. The dynamic MEY strategy maximizes NPV at the 5% discount rate

Strategy	$\overline{F}_{targ}$ **HKE**	$\overline{F}_{targ}$ **SOL**	**Max** $\Delta TAC(\%)$	**NPV (M€)**
2016	0.27	0.42	13	25.2
Maximum viability	0.30	0.48	22	26.9
Dynamic MEY	0.36	0.65	66	28.6
Final MEY	0.26	0.44	14	24.9
MSY	0.28	0.33	13	24.2

2.8 Discussion

2.8.1 Towards operational eco-viable TAC advice for mixed fisheries

The approach presented here lays the foundations for more integrated advice for mixed fisheries under output controls.

First, it allows identifying the set of management strategies that are technically achievable in mixed fisheries with joint production. This is what we called the *operating domain* of the fishery, namely the set of fishing mortality targets ensuring that at least 95% of each TAC is consumed. Outside of this domain, quotas that are not fully consumed may create an incentive to continue fishing and discard over-quota catches of limiting species, undermining the monitoring and enforcement of catch regulations. The importance of limiting the mismatch between expected catches and TACs has also emerged in recent mixed fisheries advice (Ulrich et al., 2017). The operating domain is case specific : it depends on the structure of the fleet, and on the selectivity and catchability of the different metiers. Its extent will also depend on technical flexibilities: fisheries with productions which are joint in fixed proportions will show a narrower operating domain than fisheries with limited technical interactions. Flexibility offered by management can also broaden the operating space: in a system where individual quotas are tradeable, individual harvesters may have more possibilities to balance their catches with available quotas. Moreover the operating domain can change over time as fishing practices may adapt to changes in incentives resulting from external economic drivers (e.g. changes in input costs or in demand and fish prices), regulations, ecological factors and technical change.

Second, the application of the eco-viability approach allows evaluating how alternative management strategies perform in meeting predefined constraints, be they biological, economic or social. Those constraints are defined by thresholds on key variables of the model. Consequently, the type of constraints that can be accounted for in the evaluation will depend on the model's complexity. The present application considered a biological constraint applied to four stocks and three economic constraints (positive gross and net profits, and minimum crew wage) for the 44 fleets segments. Sustainability constraints could expand further upstream (e.g. preservation of the ecosystem's good environmental status) or downstream (e.g. ensuring the viability of the downstream fish supply chain), and range from fine-scale (e.g. ensure the economic viability of individual harvesters) to broad-scale (e.g. maintain global fish supplies) constraints.

In the best case, management strategies that simultaneously meet all constraints can be identified and form the *eco-viable space*. In remaining cases, the transparent evaluation of the strategies regarding the various constraints can still assist decision-makers in assessing trade-offs and identifying compromise strategies. It is important to highlight that increasing the number of constraints does not increase the computing time. However, it makes it more difficult to identify eco-viable management strategies.

Not only can the present framework help in the decision of management targets (here in the form of targeted fishing mortality) but it also provides the time series of TAC recommendations associated to those targets, thus giving more visibility to fishing firms and the opportunity to adjust their fishing strategies.

2.8.2 Its application to the Bay of Biscay demersal mixed fishery

We were unable to identify management strategies ensuring that the viability constraints proposed for this application were simultaneously met by all stocks and all segments over the 10-year simulation period. This did not result from the biological constraint of maintaining all stocks above their limit biomass reference points. Rather, our results show that it is not possible for all fleet segments to cover the depreciation of capital, thereby suggesting overcapacity in the fishery. Moreover, eco-viability was undermined by a tension between the interests of crews and capital owners. Indeed, generating sufficient profits to reach long-term economic viability implied harvesting stocks at a point where labour productivity was too low to ensure crews the minimal FTE wage. In our analysis, the share rate was assumed constant but vessel owners have proven to adapt crew shares to external factors (Guillen et al., 2017) in order to maintain crews when profits are low and to increase their share when profits increase again. Considering such adaptive share rates in the model might resolve the observed trade-off between crews' and capital owners' surplus. In addition, imposing the average wage of a French seaman as a minimum requirement is actually more demanding than maintaining current standards. Therefore, a more accurate estimation of the opportunity cost of labour in the fishery might also make eco-viable strategies emerge.

The possibility for individual vessels to alter their catch composition is also limited in the current version of the model since metier catchabilities are considered constant and individual harvesters are assumed to fish as they did in the reference year. Allowing more flexibility in fishing effort allocation (e.g. with a non-null coefficient α in equation 2.4 that is a weighted combination of profitability and tradition as suggested by Marchal et al. (2013)), will broaden the operating domain and possibly allow eco-viable strategies to emerge. The extent to which joint productions can be altered in the Bay of Biscay demersal fishery has not been specifically investigated but insights from other fisheries show that given the right incentives fishers are often able to change their fishing practices to avoid undesirable catches (Abbott et al., 2015; Little et al., 2015; Reimer et al., 2017).

We noted that single-species MSY reference points were achievable targets in the particular case of the hake and sole fishery in the Bay of Biscay. However, this result should not question the necessity to assess whether current management targets (by default F_{MSY} for data rich European stocks) are in line with technical limitations in other mixed fisheries or when more stocks are considered in the reconciliation process.

Maximizing the net present value of the hake and sole fishery over the 10-year simulation period with a discount factor of 5% led to harvest rates well above those that maximize the fishery's net profits after 10 years. This shows that long-term gains from fishing at low harvest rates are offset by short-term profits from fishing at high harvest rates. The difference between dynamic and static MEY should decrease as the projection horizon increases since short-term profits will weigh less in the calculation of the NPV. Moreover, whereas the stock of hake reached equilibrium at the end of the simulation period as evidenced by TACs levelling-off in Figure 2.6, the stock of sole was still in a transition phase. Consequently, harvest rates associated to MEY might also decrease with a longer projection horizon as long-term profits from fishing at low harvest rates are still expected to increase after 10 years. It is worth noting that maximizing the fishery's net profits after 10 years requires to harvest hake below and sole above their respective F_{MSY} reference points. The fact that fishery-wide reference points (Multispecies MSY or MEY) can lead to the under- or over-exploitation of individual stocks has already been highlighted by many authors (Guillen et al., 2013; Voss et al., 2014; García et al., 2016; Hoshino et al., 2018; Tromeur and Doyen, 2018). Finally, we observed that the dynamic MEY is sensitive to the value of the discount rate, and, as expected, the higher the discount rate, the higher the associated harvest rates.

2.8.3 Limitations and perspectives

In this work, we illustrated how to reconcile fishing mortality targets for two stocks at the heart of technical interactions, through the identification of operating and viable domains. Four main limitations of the approach warrant further research that was beyond the scope of the work presented here.

First, joint production in fisheries rarely is limited to two species. There is thus a need to generalize this approach to an unlimited number of stocks the catch of which technically interacts. Indeed, with operating and viable spaces dimensioned by the number of controls (here the number of stocks under TAC regulation), this methodology would quickly lack operationality for fisheries with more than three interacting stocks. The issue of dimensionality could be tackled through an optimized exploration of eco-viable strategies, similar to the approach developed by Gourguet et al. (2013), rather than through the exhaustive screening of all possible controls as done here.

Second, as highlighted earlier, the operating and viability domains presented here are likely to be overly constrained by rigid assumptions on how fishers adapt to regulations, in the model. Enabling more dynamic effort and/or quota allocation and investigating whether this provides more room for viability will be the focus of future work.

Third, in highly diversified mixed fisheries, the issue of "secondary species" that are caught as bycatch when targeting more valuable species often arises. Many of these are considered data-limited stocks (DLS) for which the lack of data prevents the calibration of stock dynamics models. So far, the integration of those species in mixed fisheries bio-economic models has been inconsistent: some consider that their landings remain constant (Guillen et al., 2013), some assume that their biomass remains constant and calculate catches using a Baranov production function (García et al., 2016) (which is equivalent to our constant LPUE hypothesis when the output elasticity for biomass in the Baranov equation equals 1), some scale the income from those species as a proportion of the revenue of the target species (Ulrich et al., 2002; Guillen et al., 2013; Gourguet et al., 2013). Different approaches to model the income derived from those species are likely to result in different economic outcomes for the modelled fleets, and ensuing viability evaluations. For instance, the income from stocks modelled with a constant LPUE will always increase with fishing effort and never show any stock effect. If this is the case, the income of fleets that mostly depend on those species may be overestimated under high fishing pressure (and conversely underestimated under low fishing pressure) which might wrongly bias the advice regarding harvest of the main stocks. Assuming the income associated to minor species is proportional to the income of the target species could resolve this bias, but is not straightforward to implement in fisheries like the mixed demersal fishery of the Bay of Biscay, where there is no unique target species. Progress in this space is likely to result from the systematic identification of the circumstances under which each approach bears more relevance, leading to model each species or group of species accordingly. For instance, as assumed in our analysis, a constant biomass of the stock is only -if at all, given natural variability- relevant if the modelled fleets do not account for much of the total landings of the stock, and thus to the fishing mortality exerted on the stock.

Finally, setting acceptability constraints should be the result of discussions with all stakeholders in the fishery, from fishers to consumers, fisheries managers, scientists, fish processors, harbour managers, etc. The involvement of stakeholders was beyond the scope of this work, more designed to be a proof of concept than actual alternative advice for the management of the Bay of Biscay demersal mixed fishery.

2.8.4 Articulation with current ICES mixed fisheries advice

The ICES Working Group on Mixed Fisheries Advice (WGMIXFISH-Advice) has been developing the FCube methodology to provide mixed fisheries TAC advice since 2009 (ICES, 2009), with first implementation in the North Sea demersal fishery in 2012 (ICES, 2017b). In its latest report, WGMIXFISH-Advice also extended its advice to the Celtic Sea and Iberian Waters mixed fisheries (ICES, 2017b). An application of the framework is also under way in the Bay of Biscay. The FCube approach provides short-term mixed fisheries catch forecasts under different scenarios of quota uptake (e.g. fishers stop once they reach their most constraining quota, or once they have consumed all quotas), assuming constant metier catchabilities and effort allocation among metiers (ICES, 2017b; Ulrich et al., 2011, 2017). In the FCube approach, biological sustainability is ensured a priori by restricting fishing mortalities to stock-specific MSY ranges, and economic viability is not assessed. FCube is only used to provide short-term mixed fisheries catch forecasts. Regarding the issue of catch-quota imbalance, FCube identifies the combination of fishing mortalities within the MSY ranges that minimizes the difference in catches summed across all stocks between the “min” and “max” scenarios. In our approach, we assumed that fishers would stop fishing once they have reached their most constraining quota, which corresponds to the “min” scenario in the FCube framework and the full implementation of the landing obligation. We also assumed constant catchabilities and effort allocation among metiers but our projections were run over a 10-year period to assess the bio-economic viability of various harvest scenarios. Because intergenerational equity is an intrinsic feature of viability approach, the approach can be used to advise yearly tactical TAC decisions as well as to inform on longer term consequences of short-term decisions, which is the horizon at which sustainability must be assessed. We adopted a more exhaustive approach than in FCube to the catch-quota imbalance question, by simulating all possible combinations and identifying the set of satisfying quota uptake where all quotas would be at least consumed by 95%. We thus feel that the present approach could effectively be applied to other European fisheries with technical interactions based on an operating model calibrated with ICES stock assessment models and DCF fleets economic data.

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Responsibility for the results set out in this manuscript lies entirely with the authors.

CHAPTER 3

MODELLING QUOTA UPTAKE IN MULTI-SPECIES FISHERIES MANAGED WITH ITQS

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Abstract

Models of markets for individual quotas have been developed to inform about the expected outcomes of market-based approaches to the allocation of fishing rights. However, these models have mostly concerned single-species fisheries and are not suited to represent the interactions between quota markets in multi-species fisheries where quotas for jointly harvested species are traded on separate markets. In this work, we present a process-based simulation model of perfectly competitive ITQ (Individual Tradable Quotas) lease markets for multi-species fisheries. The theoretical equilibrium of such markets is first presented using linear programming. Then, an iterative algorithm mimicking Walras' tâtonnement process is used to represent the convergence towards the equilibrium of quota markets in a simulation context. An application of the latter model to the Australian Southern and Eastern Scalefish and Shark Fishery highlights some of the economic incentives provided by competitive ITQ markets in a multi-species fishery. Typically, we expect economically rational fishers to redirect their fishing effort towards species with low demand on the quota market, hence increasing TAC uptakes for the fishery. We also show that ITQs create an economic incentive to redirect fishing effort towards species that are not currently under quota such as squids, frostfish and ocean jackets.

3.1 Introduction

In the 1970's, individual catch shares were suggested as a means to regulate access in a fishery and increase economic efficiency (Christy, 1973), the latter being enhanced by quota transferability (Moloney and Pearse, 1979). Active markets for quota shares would indeed ensure their reallocation to harvesters who can derive the highest profit from them (Grafton, 1996). Ever since, Individual Transferable Quotas (ITQs) have been given attention and implemented in several jurisdictions as a method to restrict access and enhance economic efficiency (see reviews from Thébaud et al. (2012) and Hoshino et al. (2019)). In multi-species fisheries, transferable fishing rights have also been suggested as a flexibility mechanism to reconcile fishing possibilities on jointly caught species and prevent discarding or illegal landings (Sanchirico et al., 2006). In these fisheries, quota markets shall also convey signals on quota availability and create an economic incentive to redirect fishing effort towards species with low demand on the quota market (Holland and Herrera, 2006). Yet, several multi-species fisheries managed with ITQs have experienced surprisingly low levels of Total Allowable Catch (TAC) uptake (Knuckey et al., 2018; McQuaw and Hilborn, 2020). Paying attention to how ITQ markets shape the composition of catch in multi-species is likely to shed some light on this phenomenon.

Theoretical developments on ITQ markets have primarily focused on their application in single-species fisheries. They have been used to address questions such as production externalities (Boyce, 1992), initial allocation of harvest rights (Heaps, 2003), rent distribution (Boyce, 2004; Grainger and Costello, 2012; Hatcher, 2014a), market power (Anderson, 1991, 2008), discarding and highgrading (Anderson, 1994; Vestergaard, 1996; Hatcher, 2005a), or compliance (Hatcher, 2005b). Except for a few papers challenging the assumption of Walrasian or competitive equilibrium (e.g. Rubinstein and Wolinsky (1985) or Binmore and Herrero (1988)), markets for quota units have generally been assumed to develop through Walras' "tâtonnement" process and eventually reach equilibrium where demand equals supply (Moloney and Pearse, 1979; Squires et al., 1995; Heaps, 2003; Hatcher, 2005a; Péreau et al., 2012). Far less attention has however been given to the formation of quota prices in multi-species fisheries, where quotas for jointly caught species are traded on separate markets. Theoretical developments in this area typically investigated the impact of the cost of selective fishing (Singh and Weninger, 2009; Hatcher, 2014b) or the compliance with potential discard policies or illegal landing schemes (Hatcher, 2014b; Thébaud et al., 2017) on the quota markets of multi-species fisheries.

With integrated ecological-economic models of fisheries being increasingly used as decision-support tools (Nielsen et al., 2018), the modelling of economic dynamics, and among them those of quota markets, has gained interest within the field of applied fisheries science. When representing the exchange of individual catch shares, simulation models of multi-species fisheries have dealt with the determination of the lease or selling value of quotas following either process-based or empirical approaches. Newell et al. (2005) proposed an empirical model of quota price for New Zealand ITQ markets. This econometric model relates quota price to fundamental variables such as the export price of fish, fishing costs, the demand for quota, and environmental variability, and has been calibrated for simulation purposes in other fisheries (Fulton et al., 2007; Kaplan et al., 2014). Among process-based models, the most widely used has been the one developed by Little et al. (2009) and taken up by subsequent work (Fulton et al., 2007; Marchal et al., 2011; Toft et al., 2011). In this model, the lease price of quota for a given species is exogenously set as the mean marginal profit from harvesting that species across individual harvesters. Because of the multi-species nature of the catch, the marginal profit of harvesting a particular species is a function of the ex-vessel price of this species, its relative proportion in the catch and the marginal cost per unit of all fish caught. This expression of the lease price of quota is essentially an attempt to extend theoretical results established for single-species ITQ markets (in particular the link between the lease price of quota and marginal profit) in a multi-species context. The quota lease price model proposed by Little et al. (2009) therefore captures some basic features of ITQ markets without explicitly modelling the underlying mechanisms. In particular, the process of market price formation is not explicitly represented and as a consequence calculated lease prices do not correspond to market clearing price. For instance, all species would have a strictly positive lease price of quota, even if demand for quota relative to supply is low. More recently, Bailey et al. (2019) proposed another process-based model of ITQ trading in multi-species fisheries. In the latter, fishers estimate a reservation price for each species, which is the benefit of owning an additional unit of quota for that species (given the associated catch) multiplied by the probability of needing quota (given expected catch and current quota holdings). The reservation price being a function of the probability of needing quota implies that lease prices will increase with quota scarcity. However, this model does not aim to represent the formation of a market price for quota leases. Instead, individual reservation prices are used to match askers and bidders and transactions are made at the asker's reservation price. This review highlights that the explicit representation of mechanisms underlying the dynamics of ITQ markets in applied simulation models of multi-species fisheries is still largely at its infancy.

The objectives of the present work are twofold: to describe the theoretical equilibrium of perfectly competing ITQ lease markets given joint productions in multi-species fisheries, and to develop a simulation model that explicitly represents the convergence towards such equilibrium. First, linear programming and duality methods are used to derive theoretical quota lease prices and fishing activities at the economic optimum. In a second part, a process-based simulation model of ITQ markets mimicking Walras' tâtonnement process of price formation is presented, followed by its numerical application to a multi-species fishery managed under ITQs, the Australian Southern and Eastern Scalefish and Shark Fishery. Two scenarios of fishing effort allocation at the individual level are considered : one where fishers are able to freely allocate fishing effort among different metiers they can practice and one where their effort is constrained to previously observed patterns. The latter scenario reflects the fact that fishing practices can be constrained by factors exogenous to the model such as seasonality in resource availability or access to fishing grounds. Particular attention is given to the composition of landings in both cases to shed light on possible shift towards species with under-caught TACs and species not currently under quota.

3.2 Equilibrium of multispecies ITQ markets

3.2.1 The model

We consider a fishery composed of n_v vessels practising n_m metiers to capture n_s species. In this section, we also introduce the term "activity" to refer to the practice of a metier by a vessel. Each vessel is able to practise several activities, but an activity can only be practised by one vessel (the same metier practised by several vessels corresponds to distinct activities). The number of activities is noted n_a and in the general case $n_v \leq n_a \leq n_v \times n_m$. The time frame is the fishing season and in this context, the number of vessels, their activities, catch and costs per unit of effort and ex-vessel prices of fish are exogenous. The fishery is under TAC and ITQ management. TACs are exogenous and, on the short term, the transfer of quota is ensured by the lease of quota units on separate markets for the different species.

Let us note X the vector (n_a) of fishing effort per activity, B the matrix $(n_v \times n_a)$ whose elements indicate whether a vessel is able to practise an activity ($B_{v,a} = 1$ if vessel v is susceptible of practising activity a and $B_{v,a} = 0$ otherwise), A the matrix $(n_s \times n_a)$ of catch per unit of effort by species and activity, C the vector (n_a) of costs per unit of effort per activity, P the vector (n_s) of ex-vessel price per species, Π the vector (n_a) of profits per unit of effort by activity ($\Pi = PA - C$), Q the vector (n_s) of TACs per species, and E the vector (n_v) of maximum fishing effort per vessel.

The equilibrium of perfectly competing ITQ markets is approached as a linear programming problem maximizing the fishery's profits under TAC and capacity constraints. The problem then formally writes as:

$$\max_{X \geq 0} \quad PX \tag{3.1}$$

$$\text{s.t.} \quad AX \leq Q \tag{3.1.1}$$

$$BX \leq E \tag{3.1.2}$$

The dual of the latter problem allows the determination of the quota lease prices and vessels' quasi-rents. Let us call L the vector (n_s) representing the dual variables associated to the quota constraint 3.1.1 (i.e. the lease price of quota units in an ITQ system) and R the vector (n_v) representing those associated to the capacity constraint 3.1.2 (i.e. the vessels' quasi-rents, or short-term profits).

The dual problem then reads as follows:

$$\min_{(L,R)\geq 0} \quad LQ + RE \tag{3.2}$$

$$\text{s.t.} \quad LA + RB \geq \Pi \tag{3.2.1}$$

3.2.2 Market equilibrium

Let us call X^*, L^* and R^* the solutions of the primal and dual problems. Application of the complementary slackness theorem gives the following relations.

Between primal constraints and dual variables:

$$(A_{s.}X^* - Q_s)L_s^* = 0 \quad \forall s \in \{1..n_s\}. \tag{3.3.1}$$

$$(B_{v.}X^* - E_v)R_v^* = 0 \quad \forall v \in \{1..n_v\}. \tag{3.3.2}$$

Between dual constraints and primal variables:

$$(L^*A_{.a} + R^*B_{.a} - \Pi_a)X_a^* = 0 \quad \forall a \in \{1..n_a\}. \tag{3.3.3}$$

Given that $B_{.a}$ has only one non-null coefficient $B_{v_a,a} = 1$, where v_a is the vessel susceptible of practising activity a, Equation 3.3.3 also writes as:

$$(L^*A_{.a} + R_{v_a}^* - \Pi_a)X_a^* = 0 \quad \forall a \in \{1..n_a\}. \tag{3.3.3$'$}$$

Quasi-rent

Dual variables R_a being non-negative, Equation 3.3.2 can be written as:

$$\begin{cases} B_{v.}X^* < E_v & \Longrightarrow \; R_v^* = 0 \\ R_v^* > 0 & \Longrightarrow \; B_{v.}X^* = E_v \end{cases} .$$

The latter conditions mean that only vessels employed at their full capacity at the optimum generate positive short-term profits.

Equilibrium lease price of quota units

Dual variables L_s being non-negative, Equation 3.3.1 can be written as:

$$\begin{cases} A_{s.}X^* < Q_s & \Longrightarrow \; L_s^* = 0 \\ L_s^* > 0 & \Longrightarrow \; A_{s.}X^* = Q_s \end{cases} ,$$

which means that only the species the TAC of which is binding at the optimum have a non-null lease price. Such species will also be referred to as binding or constraining species in the remainder of the text.

Primal variables X_a being non-negative, Equation 3.3.3′ can be written as:

$$\begin{cases} L^* A_{.a} + R^*_{v_a} > \Pi_a & \Longrightarrow X^*_a = 0 \\ X^*_a > 0 & \Longrightarrow L^* A_{.a} + R^*_{v_a} = \Pi_a \end{cases} ,$$

which means that the profits of an active activity at the optimum just cover the lease of quotas and the quasi-rent of the associated vessel.

Given that $\Pi = PA - C$, the latter relations also write as:

$$\begin{cases} L^* A_{.a} + R^*_{v_a} + C_a > PA_{.a} & \Longrightarrow X^*_a = 0 \\ X^*_a > 0 & \Longrightarrow L^* A_{.a} + R^*_{v_a} + C_a = PA_{.a} \end{cases} .$$

From these equations can be derived the lease price of quota for a species $\overline{s}$ whose TAC is binding at the optimum. For any activity a capturing species $\overline{s}$ and active at the optimum:

$$L^*_{\overline{s}} A_{\overline{s}a} + \sum_{s \neq \overline{s}} L^*_s A_{sa} + R^*_{v_a} + C_a = P_{\overline{s}} A_{\overline{s}a} + \sum_{s \neq \overline{s}} P_s A_{sa}$$

$$\Longrightarrow L^*_{\overline{s}} = P_{\overline{s}} + \frac{1}{A_{\overline{s}a}} \left[\sum_{s \neq \overline{s}} (P_s - L^*_s) A_{sa} - R^*_{v_a} - C_a \right] . \tag{3.4}$$

One can note from Equation 3.4 that the lease price of quota for species $\overline{s}$ at the optimum depends not only on its ex-vessel price but also on that of jointly caught species. As a consequence, the equilibrium lease price of quota of a binding species can exceed its ex-vessel price provided that ex-vessel prices of jointly caught species (diminished by their own quota lease price) are sufficiently high. The vessels' quasi-rents also appear in the expression of equilibrium quota lease prices. This highlights that the lease price of quota for a binding species approaches the marginal profit deduced from quota costs associated to jointly harvested species of the marginal vessel catching this species (i.e. the one that is indifferent between fishing or remaining in port).

3.3 The convergence towards ITQ market equilibrium

We now turn to the process which leads to the determination of equilibrium prices, and present a process-based model of the convergence towards the equilibrium of perfectly competitive ITQ lease markets. The model mimics Walras' tâtonnement process to determine the clearing lease price of quota units for jointly harvested species.

3.3.1 The tâtonnement algorithm

To the extent possible, we keep the notations introduced in Section 3.2. Nonetheless, exempted from the dimensionality constraint imposed by linear programming, activities are replaced by their vessel-metier combination to facilitate the writing of some equations. In the following sections, X is the vector (n_v) of fishing effort per vessel, A the array $(n_s \times n_v \times n_m)$ of catch per unit of effort by species, vessel and metier, C the matrix $(n_v \times n_m)$ of costs per unit of effort per vessel and metier, P the vector (n_s) of ex-vessel price per species, L the vector (n_s) of quota lease price per species, Π the matrix $(n_v \times n_m)$ of profits per unit of effort by vessel and metier (here, including the costs of leasing quota), Q the vector (n_s) of TACs per species, and E the vector (n_v) of maximum fishing effort per vessel. The superscript * refers to the value of the variable at equilibrium.

As represented in Figure 3.1, the tâtonnement algorithm progresses as follows:

1. Quota lease price are attributed an initial value.

   For each iteration *it* of the "tâtonnement" process (for the sake of clarity, *it* subscripts will be omitted in the following equations):

2. Individual harvesters decide on a fishing strategy by allocating their fishing effort to metiers.
   — Building on previous fleet dynamics models (review by van Putten et al. (2012) and Girardin et al. (2017)), the effort allocation is modelled as a function of a weighted average of the metiers' profitability and past effort allocation, with the weight given to profit attractiveness as opposed to habit being measured by the coefficient α. The profitability of metier m for vessel

v ($\Pi_{v,m}$) is calculated as follows:

$$\Pi_{v,m} = \sum_s [A_{s,v,m} \times (P_s - L_s)] - C_{v,m}, \tag{3.5}$$

Relative profitabilities $\overline{\Pi}$ (i.e. centred on the profitability of the vessel's least profitable metier) are used in the effort allocation function to avoid negative coefficients: $\overline{\Pi}_{v,m} = \Pi_{v,m} - \min_m(\Pi_{v,m})$. Following Marchal et al. (2011), the

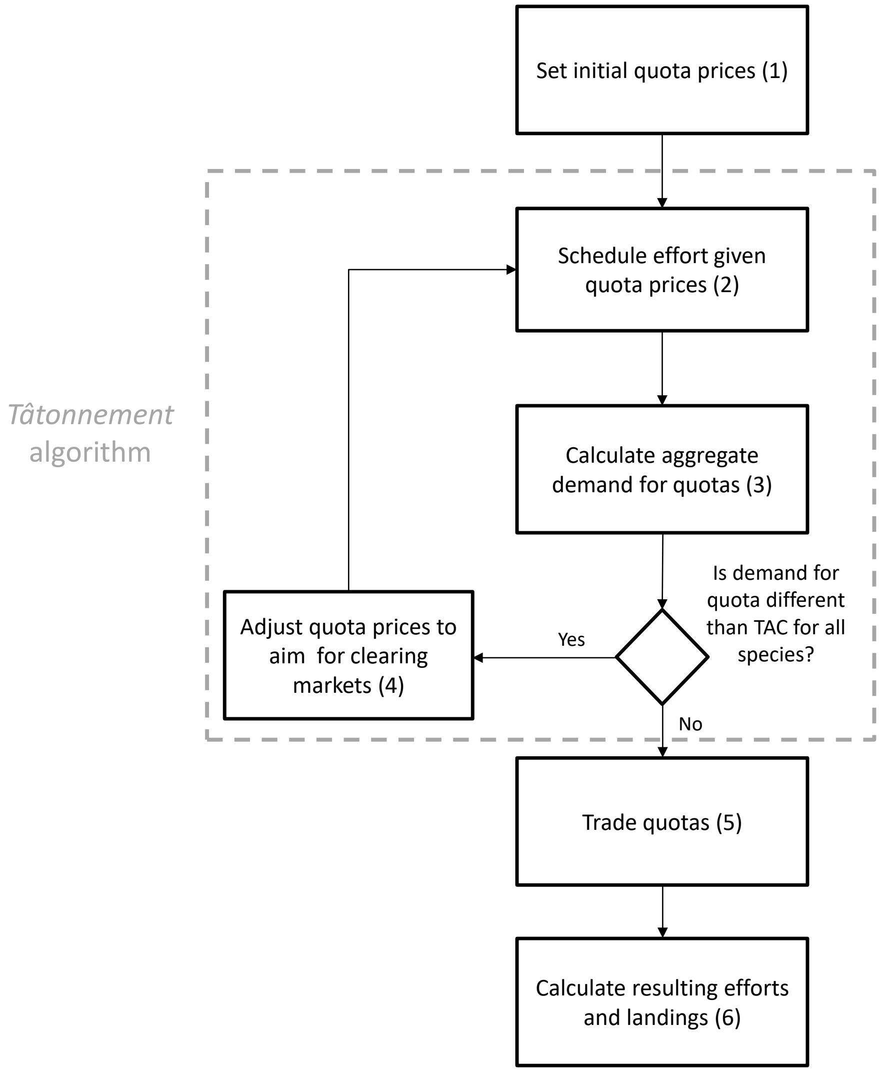


Figure 3.1 – Steps of the "tâtonnement" algorithm.
The algorithm keeps searching for quota clearing prices until the deviation of total demand for quota from the TAC is smaller than a tolerance level ϵ for all species under TAC or that a maximal number of iterations it_{max} has been reached.

proportion of effort the harvester of vessel v allocates to metier m, $pX_{v,m}$, is calculated as follows:

$$pX_{v,m} = \alpha \times \frac{\overline{\Pi}_{v,m}}{\sum_m \overline{\Pi}_{v,m}} + (1-\alpha) \times \frac{X_{0_{v,m}}}{\sum_m X_{0_{v,m}}}, \tag{3.6}$$

with $X_{0_{v,m}}$ being the historical effort of vessel v allocated to metier m. When $\alpha = 1$, effort allocation is entirely profit-driven [1], with the metier profitability being not only determined by its efficiency at catching fish but also the costs of leasing in quota to land the catch (Equation 3.5). The case $\alpha = 0$ is also considered as a baseline scenario against which the $\alpha = 1$ scenario will be compared. It corresponds to a scenario where the allocation of fishing effort is exogenously set as that in a reference period and represents a case where no effort shift is possible among metiers. Realized effort allocation is likely to lie between these two scenarios with possible operational constraints such as seasonality in resource availability or access to fishing grounds constraining the response of fishers to economic incentives.

— Given the planned fishing effort allocation, the harvester of vessel v assesses whether it is profitable to go fishing. If its average profitability per unit of effort is positive, then the vessel will operate at its maximum effort E_v, otherwise it will remain at port:

$$X_v = \begin{cases} E_v & \text{if } \sum_m (pX_{v,m} \times \Pi_{v,m}) > 0, \\ 0 & \text{otherwise.} \end{cases} \tag{3.7}$$

3. The demand for quota (D) is calculated for each species as follows:

$$D_s = \sum_{v,m} (A_{s,v,m} \times pX_{v,m} \times X_v). \tag{3.8}$$

4. Quota prices are then adjusted for the next iteration to aim for clearing markets for each species:

$$L_{s,it+1} = L_{s,it} \times (1 - \lambda \times \frac{Q_s - D_s}{Q_s}), \tag{3.9}$$

with λ the step coefficient of the convergence. [2]

1. One can note that allocating effort proportionally to the metiers' profitabilities does not exactly correspond to a profit-maximizing effort allocation, which would lead to the entire effort being allocated to the most profitable metier. Rather, this model can reflect a profit-maximizing allocation under uncertainties, where the probability to choose a metier is proportional to its expected profitability.

2. The value of the step coefficient is determined empirically to achieve a satisfying compromise between the precision of convergence and computing time. Precision will be greater for low values of

Steps 2 to 4 are iterated until total demand for quota is close enough to the TAC for all species (i.e $\frac{Q_s - \sum_v D_{s,v}}{Q_s} < \epsilon$) or after it_{max} iterations.

3.3.2 Implementation considerations

Quota lease prices are attributed a quasi-null value before entering the tâtonnement phase. The advantage of initiating quota prices at a null value is that at they will have to increase for markets to clear (at least for those who will clear given joint productions). However, given the form of the quota price adjustment function given in Equation 3.9, one cannot start with a null value. Therefore, quota lease prices were initiated at 0.1% of the species ex-vessel price.

Because of the discrete nature of the quota demand function (ensuing from vessels either remaining at port or fishing at their full capacity), market equilibrium is never reached perfectly and demand for quota for binding species will oscillate around the TAC [3]. In order to avoid TACs to be overshot due to the imperfect market equilibrium, quota distribution is explicitly represented in the model following the convergence process (Step 5 in Figure 3.1). Similar to the approach proposed by Little et al. (2009), quota for species s is allocated in priority to individuals with the highest incentive to lease in quota, that is to vessels with the highest marginal profit $\frac{\sum_m (pX^*_{v,m} \times \Pi^*_{v,m})}{\sum_m (pX^*_{v,m} \times A_{s,v,m})}$. Vessels are thus ranked by decreasing order of marginal profit on each market and quotas distributed according to this order until offer (for binding species) or demand (for non-binding species) expires.

As a result of this allocation, individual harvesters may not exactly hold the quota portfolio they demanded at equilibrium. To reconcile the multiple species' quota constraints, vessels' activity is capped by their most constraining quota ($X^*_v = \min_s \left(\frac{Q_s}{\sum_m (pX^*_{v,m} \times A_{s,v,m})} \right)$) and final landings are deduced from these efforts (Step 6 in Figure 3.1).

λ but at the cost of increased convergence time.

3. The extent of the oscillations around the TAC is related to the number of participants in the market. Typically, they are more important in thin markets where each participant potentially captures a higher share of the TAC.

3.3.3 Numerical application to the Australian Southern and Eastern Scalefish and Shark Fishery

Description of the fishery and quota system

The Australian Southern and Eastern Scalefish and Shark Fishery (SESSF) is a multi-sector and multi-species fishery that operates in Australian federally-managed waters as well as some state waters under specific arrangements. The fishery is divided in four sectors represented by different gear types targeting specific group of species, namely the Gillnet Hook and Trap Sector (GHTS), the Commonwealth Trawl Sector (CTS), the Great Australian Bight Trawl Sector (GABTS), and the East Coast Deepwater Trawl sector (ECDWTS). The present application focuses on the first two, at the heart of technical interactions. During the 2015 fishing season (May 2015-May 2016), 106 vessels from 12 fleets were operating in these sectors. Vessels in each fleet can target various species or species assemblages using the same gears. These different activities correspond to different metiers listed in Table 3.1. One can see that some fleets (e.g. trawlers) have a more diversified fishing activity than others (e.g. gillnetters).

Management in the fishery primarily relies on output controls with TACs being determined for 25 stocks in the selected sectors and allocated as individual transferable quotas. As shown by Figure 3.2, the TAC of many stocks were under-caught in 2015, with only the TAC of key economic stocks (e.g. flathead, orange roughy (Eastern and Southern stocks), school whiting, gummy shark, blue-eye trevalla and pink ling) or choke species (e.g. the rebuilding TAC for school shark constraining catches of gummy shark) being caught above 75% (AFMA, 2016). Full TAC uptake is extremely rare in the fishery partly due to frictions in quota markets and stocks with TAC uptakes above 75% are generally considered as constraining. The eleven stocks with TAC uptake above 75% in 2015, as identified in Figure 3.2, will therefore be referred to as the originally constraining stocks in the remainder of the text.

Model calibration

The model was calibrated based on economic and catch data from 2015. Catches and fishing effort (number of fishing days) at the vessel and metier level were calculated from SESSF logbook data. Ex-vessel price in 2015 for the various species were obtained from the Australian Fisheries Statistics (Mobsby, 2018). Cost structures were estimated per main gear type (trawlers, Danish seiners, gillnetters, and liners) based on the 2015 economic survey (Bath et al., 2018) and personal communications from

Table 3.1 – List of fleets and metiers for the Commonwealth Trawl Sector (CTS) and the Gillnet Hook and Trap Sector (GHTS) in the Southern and Eastern Scalefish and Shark Fishery.

Sector	Fleets	Metiers: Gear	Metiers: Target species/assemblage
CTS	Shelf trawlers East Mixed trawlers East Royal red prawn trawlers	Trawl (TW) East	Flathead Pink ling Royal red prawn Orange roughy Jackass Morwong Squids Frostfish Ocean jackets Mixed shelf Mixed slope
	Mixed trawlers West Blue grenadier trawlers	Trawl (TW) West	Blue grenadier Pink ling Squids Mixed deepwater Mixed shelf Mixed slope
	Danish seiners	Danish seine (DS)	Flathead School Whiting Mixed
GHTS	Gillnetters	Gillnet (GN)	Gummy shark
	Shark bottomliners Mixed bottomliners	Bottomline (BL)	Gummy shark Blue-eye trevalla Mixed scalefish
	Blue-eye dropliners	Dropline (DL)	Blue-eye trevalla Gummy shark Mixed scalefish
	Blue-eye auto-longliners Mixed auto-longliners	Automatic longline (AL)	Blue-eye trevalla Pink ling Mixed scalefish

the Australian Bureau of Agricultural Resource Economics and Sciences (ABARES), and used to calculate costs per unit of effort at the vessel and metier level as described in Appendix C.5. Maximal efforts E were assumed to be the vessels' maximal observed effort over the period 2010-2015. TACs and associated uptake levels were retrieved from AFMA (2016).

Metiers were defined as combinations of a fishing gear and targeted species or group of species and derived from a clustering analysis of individual shots based on their

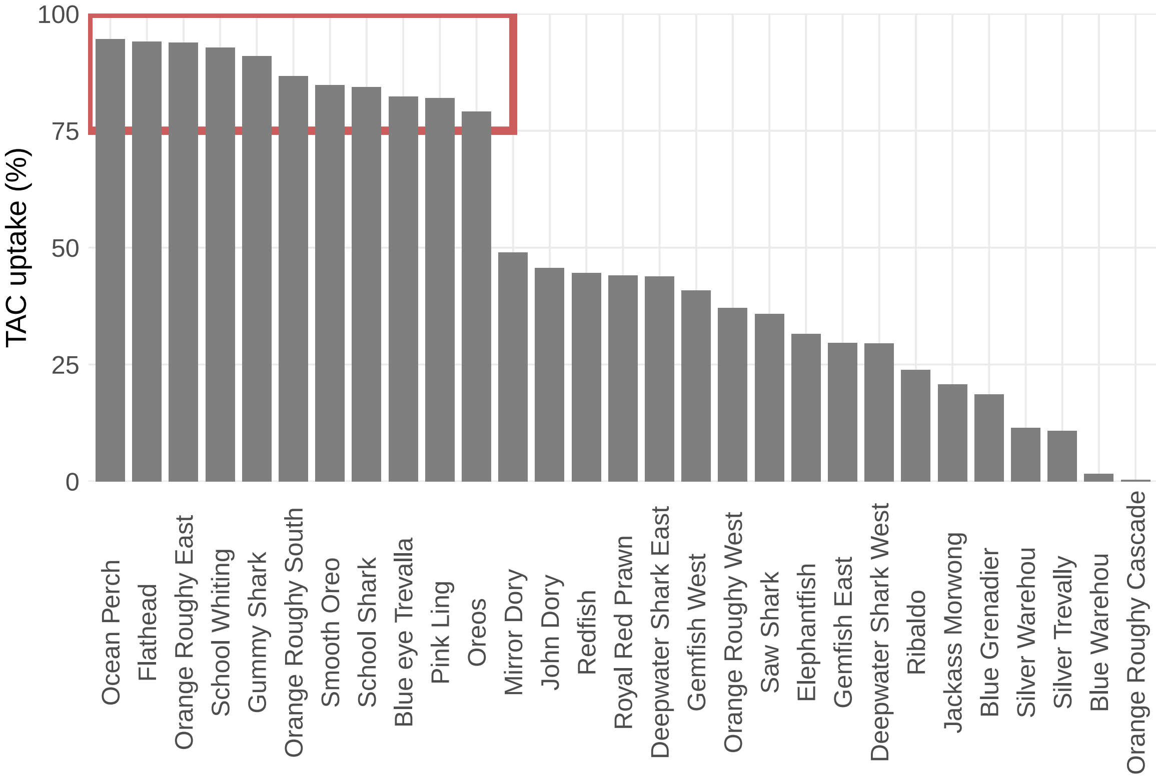


Figure 3.2 – Levels of TAC uptake in the Commonwealth Trawl and Gillnet Hook and Trap sectors of the Southern and Eastern Scalefish and Shark Fishery. Stocks within the framed area have a TAC uptake greater than 75%.
Source: AFMA (2016)

species composition. Fleets, on the other hand refer to groups of vessel having a similar fishing strategy. A multivariate statistical approach was also used to define fleets by clustering vessels based on their annual effort allocation among metiers. Further details on these analyses are provided in Appendix D.

A value of 0.1 for the convergence multiplier λ allowed convergence of the algorithm in a reasonable number of iterations (1000 in the present case).

Results

When fishers allocate their fishing effort based on profitability, quota lease markets clear for several but not all species in the fishery (Table 3.2). To be precise, indicated TAC uptake levels correspond to those observed once all quotas have been allocated and individual fishing efforts adjusted to the most constraining quota (i.e. after step 6 in Figure 3.1). The difficulty for vessel owners to meet their quota demand on all species simultaneously explains why the TAC of some species with positive quota lease prices at equilibrium (i.e. species for which the quota lease market clears) is not entirely

Table 3.2 – Quota uptake and lease prices of quota at equilibrium (model output) under the profit-driven effort allocation scenario ($\alpha = 1$). Bold highlighting indicates non-null equilibrium quota lease price, i.e. situations of market clearing.

Species	Ex-vessel price (AU$/kg)	TAC uptake (%)	Quota lease price (AU$/kg)
Flathead	6.18	91	0.00
Gummy Shark	6.29	99	**1.01**
Pink Ling	5.73	100	**0.31**
School Whiting	3.05	100	**0.52**
Orange Roughy East	5.59	100	**1.90**
Blue eye Trevalla	9.06	98	**0.01**
School Shark	5.99	100	**0.43**
Ocean Perch	5.08	100	**7.16**
Oreos	3.50	100	**1.06**
Orange Roughy South	5.59	94	**1.33**
Blue Grenadier	1.30	33	0.00
Silver Warehou	1.15	13	0.00
Mirror Dory	3.15	100	**1.50**
Royal Red Prawn	4.01	26	0.00
Saw Shark	1.91	43	0.00
Jackass Morwong	3.36	38	0.00
Silver Trevally	4.49	18	0.00
Ribaldo	3.50	21	0.00
Gemfish West	2.43	35	0.00
John Dory	8.66	92	0.00
Deepwater Shark West	3.64	34	0.00
Elephantfish	0.80	30	0.00
Redfish	3.43	49	0.00
Deepwater Shark East	3.64	42	0.00
Gemfish East	2.43	70	0.00
Smooth Oreo	3.50	1	0.00
Orange Roughy West	5.59	40	0.00
Blue Warehou	3.06	3	0.00
Orange Roughy Cascade	5.59	0	0.00

caught. The conclusions from Section 3.2.2 are nevertheless illustrated in this numerical application: the lease price of non-binding quotas are null and that of binding quotas captures the revenue from jointly captured species, which can lead to quota lease prices exceeding ex-vessel prices (e.g. ocean perch).

Under this scenario, markets clear for almost all species that were identified as constraining in the reference year (Figure 3.2). The only exception is flathead for which the market does not fully clear with a resulting TAC uptake of 91%. The TAC of mirror dory yet becomes constraining in this scenario.

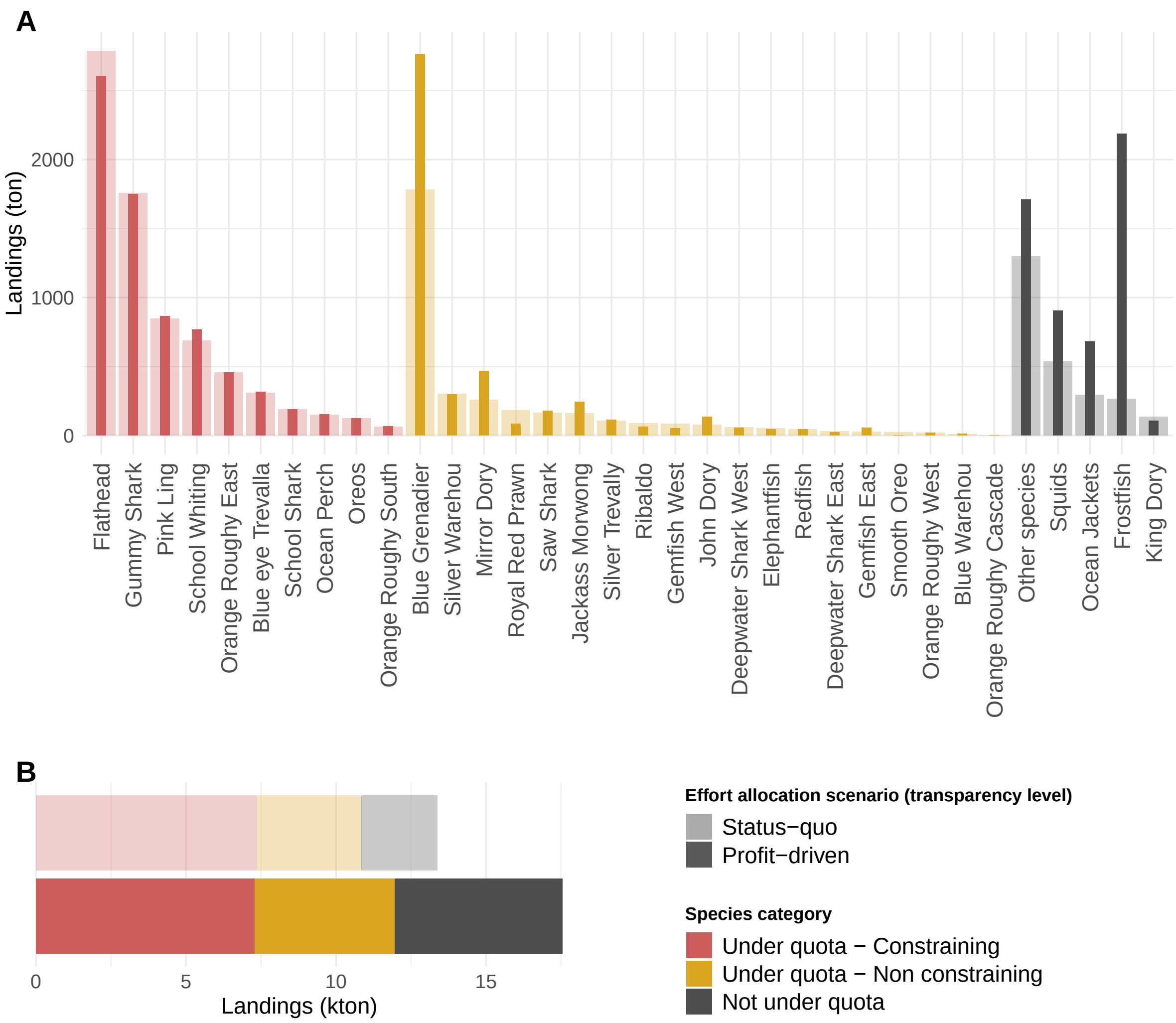


Figure 3.3 – Fishery's landings composition under the two scenarios of effort allocation: status-quo ($\alpha = 0$) and profit-driven ($\alpha = 1$)
A- Landings per species
B- Landings aggregated per species category
Light colourings relate to the tradition-based scenario and dark colourings to the profit-driven scenario. Species have been grouped into three categories: species for which the TAC was considered as constraining in the reference year, species that were not constraining in the reference year and species not under quota.
Source: model output

Let us now compare the composition of landings that would be obtained if fishers allocate their fishing effort based on profitability compared to when they keep fishing as they did in the reference year. Compared to the status-quo scenario, a profit-driven effort allocation leads to landings of originally non-binding species (in yellow) increase by 34% and those of species not under TAC (in black) increase by 121%, with only a slight decrease of 1% in landings of originally binding species (in red) (Figure 3.3-B).

Although overall landings of originally non-binding species and species not under quota increase from the status-quo to the profit-driven scenario, it is not necessarily the case at the species level, as shown by Figure 3.3-A. One can note that among originally non-binding species or species not under quota, those that can be targeted in the model (i.e. having a metier specifically targeting them - cf list of metiers in Table 3.1) generally see their landings increase. It is for instance the case of blue grenadier, jackass morwong, squids, ocean jackets and frostfish for which landings are respectively multiplied by 1.55, 1.51, 1.68, 2.33 and 8.25. There is however no generic pattern concerning by-product species (i.e. those that are not specifically targeted) as four see their landings increase and eight see their landings decrease from the habit-based to the profit-driven scenario.

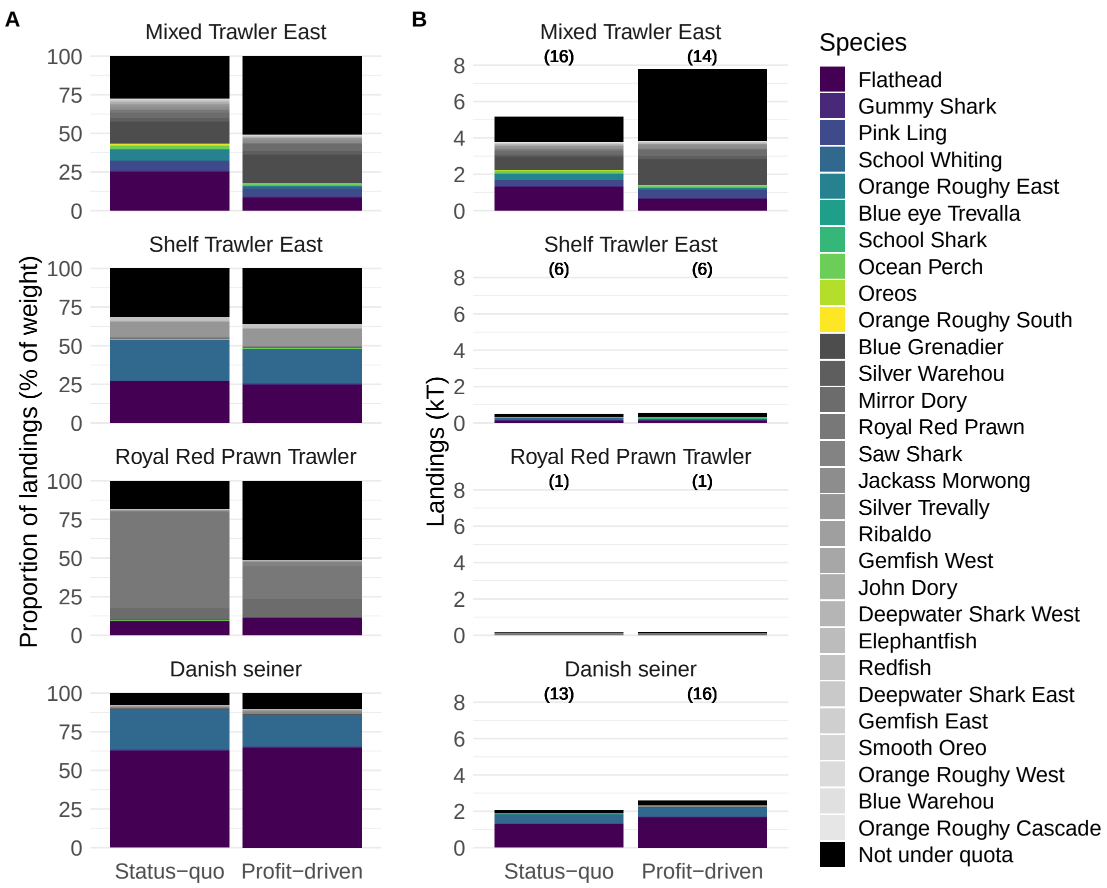


Figure 3.4 – Relative and absolute landings by the fleets harvesting flathead under status-quo and profit-driven effort allocation scenarios.
A- Proportion of species weight in the fleets' landings
B- Absolute landings per species per fleet
Constraining species are represented in colored scale, non-constraining species in gray scale and species not under quota in black. Numbers in parenthesis indicate the number of active vessels in each fleet.
Source: model output

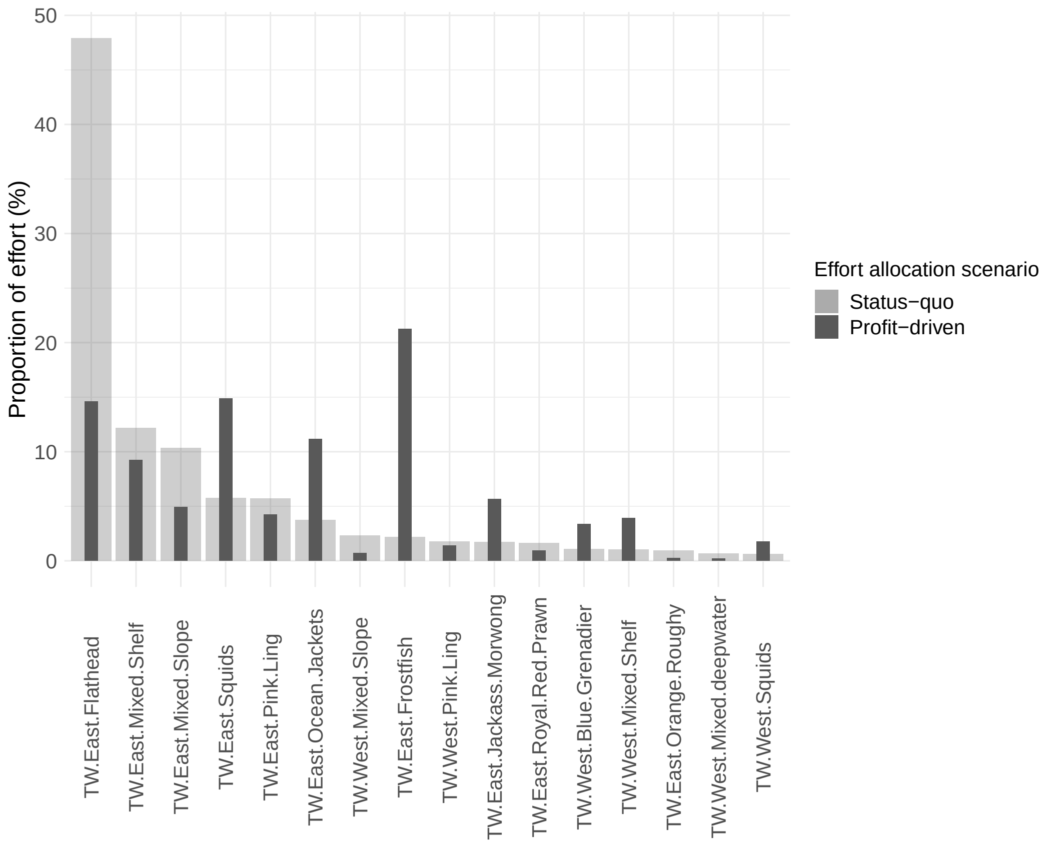


Figure 3.5 – Effort allocation of the eastern mixed trawler fleet in the two effort allocation scenarios: status-quo ($\alpha = 0$) and profit-driven ($\alpha = 1$).
Source: model output

The landings of originally constraining species (in red) remain fairly stable between both scenarios (Figure 3.3-A). Yet, the decreasing demand for flathead is worth expanding upon as it illustrates the incentive provided by ITQ markets to avoid species with high demand on the quota market. Figure 3.4-A shows that mixed trawlers considerably reduce the proportion of flathead in their catch between the status-quo and profit-driven effort allocation by shifting fishing effort towards other target species. Whereas nearly half of their fishing effort is dedicated at targeting flathead in the status-quo scenario, this activity only represents 15% of their time at sea when effort is allocated based on profitability (Figure 3.5). In this latter scenario, effort shifts to metiers targeting non-constraining species such as jackass morwong or blue grenadier, or species not under quota such as squids, ocean jackets or frostfish. The three other fleets harvesting flathead do not have more profitable activities to switch to and show a similar landings composition between both scenarios (Figure 3.3-A). Given the originally large contribution of mixed trawlers to flathead landings (Figure 3.3-B), their effort shift in the profit-driven scenario releases enough quota for more Danish seiners to operate in the fishery (16 compared to 13 vessels), which partly compensates for the trawlers effort shift and decrease in fleet size (loss of 2 vessels).

3.4 Discussion

3.4.1 Fishing incentives under perfectly competitive ITQ markets

In a fishery under ITQ, the quasi-rent of a vessel is not only determined by the value of its catch and its technical efficiency, but also depends on the cost of quota associated with the catch. If quota markets in a multi-species fishery are perfectly competitive, species with low demand for quota gain in attractivity compared to traditionally targeted species for which the shadow value of quota is high. In other words, perfectly competing ITQ markets generate an economic incentive to redirect fishing effort towards non-binding species. The results of our simulations highlight that if fishers can respond to such incentives, the fishery's TAC uptake would be improved compared to what results from current fishing practices.

It is also worth highlighting here that profit-driven fishing behaviour is also expected to redirect fishing effort towards species not under quota regulation, especially if they can be targeted. Problematic if putting the sustainability of unregulated stocks at risk, governance leakage, or spillover effects, complicate the design of quota systems in multi-species fisheries, where assigning quotas on all species would reveal extremely costly if even possible (Squires et al., 1998). Despite the issue not being new in the economics literature, it has not been quite addressed by management as leakages have been pointed at in multi-species fisheries where catch limits only apply on a few target species (Asche et al., 2007; Hutniczak, 2014).

3.4.2 Observations from multi-species ITQ markets

As outlined in our theoretical section, quota markets for jointly harvested species may not simultaneously clear. As a consequence, the equilibrium lease price of quota of binding species is a shadow value of their associated catch and that of non-binding species is null. Whether this is actually observed in multi-species ITQ markets is worth investigating. McQuaw and Hilborn (2020) report lease quota prices exceeding ex-vessel prices for several choke species in the US West Coast groundfish trawl fishery. In contrast, Holland (2013) noted that quota prices for binding species would not rise above ex-vessel price in the British Columbia groundfish fishery. Interviews with market participants from this fishery highlighted that prices of quota units higher than ex-vessel prices had a social stigma and was considered as gouging, hence preventing quota prices to fully reflect the value of quota. Another likely explanation suggested by Holland (2013) for not observing quota prices rising above ex-vessel price is the existence of barter or basket trades (i.e. quotas for multiple species being traded in a single transaction) that could be used to mask the price of the binding species. Basket trades are common in other multi-species fisheries as noted by Innes et al. (2014b) in the Australian Coral Reef Fin-Fish Fishery or Knuckey et al. (2018) in the SESSF. In New Zealand, quota prices are capped by their deemed value which is again likely to prevent quota price to fully express the implicit value of limiting quota (Hatcher, 2014b). However, Newell et al. (2005) noted that only 1% of the observed lease prices approached the level of the deemed value. Although they did not see quota prices for species without a binding TAC constraint drop to zero, they found a significant positive relationship between the demand for quota and quota price. Preliminary analyses of lease markets in the SESSF have shown similar relationship, with non-binding quotas also having a positive value on lease markets. Instead of the fishery's rent being only captured in the quota prices of constraining species, it is more evenly spread across species, with nevertheless more value attributed to constraining species.

3.4.3 Perspectives for process-based simulation models of multi-species ITQ markets

The present work advances the explicit representation of processes underlying ITQ market dynamics in simulation models of multi-species fisheries. However, as just noted, quota markets in multi-species fisheries feature more complex dynamics than that modelled, hence highlighting the need for further work in this domain. In this regard, we could mention three topics upon which research and modelling efforts could focus for a better representation of the dynamics of multi-species ITQ markets. First, because of the inherently uncertain nature of the catch in multi-species fisheries, fishers have a limited ability to anticipate the quota portfolio that they will need to cover their fishing season. In such context, the option value associated to holding a quota unit should be considered when estimating its market price. Second, basket trading tends to develop as multi-species ITQ markets mature and synergies among species establish since they can diminish transaction costs or reduce the risk of not obtaining the necessary quota compared to when engaging in several single-species transactions (Iftekhar and Tisdell, 2012; Tisdell and Iftekhar, 2013; Innes et al., 2014b). Yet, very little is known on how the value of a basket of quotas is determined, which hinders the representation of this trading practice in simulation models. Finally, there has also been multiple evidence of social networks developing in non-centralized quota markets (van Putten et al., 2011; Ropicki and Larkin, 2014; Oostdijk et al., 2019; Vasta, 2019), and influencing quota price formation (Pinkerton and Edwards, 2009; van Putten et al., 2011; Ropicki and Larkin, 2014). On one hand, the development of long-term trading relationships can foster trades as it reduces transaction costs associated to the search of trading partners (Connor and Alden, 2001; Innes et al., 2014a). On the other hand, trading habits could be seen as an obstacle to the optimal reallocation of fishing possibilities expected under ITQs as participants engaged in satisfying relationships have less incentives to search for the optimal deal.

3.5 Conclusion

This piece of work presents a process-based simulation model of perfectly competitive ITQ lease markets in multi-species fisheries where jointly caught species are traded on separate lease markets. It explicitly models Walras' tâtonnement process as a means to determine the market lease value of quota for multiple species, and is easily implementable in integrated fisheries simulation frameworks increasingly used to assess ex-ante management measures. Its application to the Australian Southern and Eastern Scalefish and Shark Fishery highlights some economic incentives that perfectly functioning quota markets provide in a multi-species context, such as encouraging fishers to increase their catch of species for which the TAC is not binding, hence leading to an increased uptake of catch opportunities. Our results also point to a risk of leakage towards species not currently under quota.

CHAPTER 4

FLEXIBILITY OF JOINT PRODUCTION IN MIXED FISHERIES AND IMPLICATIONS FOR MANAGEMENT

This chapter is under revision for publication in the ICES Journal of Marine Science as:

Briton, F., Thébaud, O., Macher, C., Gardner, C., Little, L.R. Flexibility of joint production in mixed fisheries and implications for management.

Abstract

Over the past decade, efforts have been made to factor technical interactions into management recommendations for mixed fisheries. Anticipating joint productions is a particularly challenging task which involves a good understanding of their environmental but also human determinants. Using an integrated ecological-economic simulation model, we explore the extent to which fishers are likely to alter the species composition in their catch in a mixed fishery managed with Individual Tradable Quotas (ITQs), the Australian Southern and Eastern Scalefish and Shark Fishery. We show that a significant part of joint productions is determined by the choices fishers make when deciding on when and where they go fishing, and that overlooking the incentives for fishers to alter the species mix in their catch strongly underestimates the flexibility of catch composition in the fishery. These results highlight the importance of capturing the mechanisms underlying fishing choices when advising TAC decisions in mixed fisheries. We also show the hierarchy of species in this fishery, with harvest targets set for primary commercial species determining most of its socio-economic performance. We finally discuss the implications of those results for the management of mixed fisheries.

4.1 Introduction

There is now wide recognition that traditional single-species approaches, still the basis for most tactical management decisions in fisheries, fall short of addressing the complexities observed in mixed fisheries, where a variety of species are simultaneously caught in unselective fishing operations. The issue with setting management targets at the stock level in fisheries with technical interactions is twofold: first the objectives are unlikely to be met for all stocks simultaneously, leading to situations of over-quota discards or lost catch opportunities (Ulrich et al., 2011, 2017; Patrick and Benaka, 2013), and second, they do not address the performance of the fishery *per se*, particularly in its economic and social dimensions (Dichmont et al., 2008; Rindorf et al., 2017a; Hoshino et al., 2018).

Economists were probably the first to argue that knowledge about the technological structure of a multi-output fishery is critical to successful regulation (Squires, 1987; Kirkley and Strand, 1988; Jensen, 2002). Particularly relevant to the regulator tasked setting catch limits in a mixed fishery is information about jointness in inputs and substitutability between outputs. In firm production analysis, a technology is said to be non-joint in input quantities when the production of single outputs can be represented as independent functions of inputs. Non-joint technologies in mixed fisheries represent one end of the regulator's spectrum where catch limits on individual species can be set independently as they relate to independent production processes. Although there is little chance for the regulator of a mixed fishery to effectively face this situation, as their production has generally been shown to be non-separable in inputs (Jensen, 2002), catch limits in mixed fisheries are still mostly set using single-species approaches, i.e. as if their catch was the result of independent production processes.

As highlighted by Pascoe et al. (2007), rejecting the assumption of separable production does not necessarily mean that the outputs are produced in fixed proportions (which could be referred to as purely joint production and would represent the other end of the spectrum); they can be substitutable to some extent. In this regard, the analysis of production functions in mixed fisheries has often evidenced fishery-specific (Kirkley and Strand, 1988; Jensen, 2002; Pascoe et al., 2007, 2010) and sometimes fleet-specific (Pascoe et al., 2007) levels of substitutability between species, hereby invalidating the assumption of purely joint production that is sometimes made when taking into account technical interactions in management advice of mixed fisheries(Ulrich et al., 2017). There is today a critical need for the science guiding the management of mixed fisheries to fully grasp the reality of their operation, which lies somewhere along the gradient between a collation of independent production processes and a purely joint production.

The extent to which multi-species catch composition is flexible at the individual level is the result of: (1) the **possibility** for fishers to alter their catch composition by changing their fishing practices (i.e. how they allocate their fishing effort among *metiers* using different gears and/or targeting different species in specific areas or periods (EU, 2008, 2010)) and (2) their **incentive** to do so. The possibility for fishers to change what they catch is constrained by the technology available to them and the ecosystem in which they operate. Consequently, margins in selectivity can be classified as either **technical**, which relates to improving the selectivity of fishing gears or practices, or **institutional** which pertains to providing incentives to fish more selectively, be they market or social-based (Pascoe, 2010; Abbott et al., 2015).

The present work explores the extent to which institutionally-driven incentives, here the lease value of quota units, can interact with technical constraints to determine the effective response in terms of output substitution. An integrated ecological-economic model representing the dynamics of the fishery and able to simulate different fishing behaviours and incentives was used to investigate the potential flexibility of catch composition resulting from changes in fishing practices in a fishery managed under Individual Transferable Quotas (ITQs), the Australian Southern and Eastern Scalefish and Shark Fishery. We present the outcome of different scenarios in terms of Total Allowable Catch (TAC) uptake and economic performance of the fishery and discuss some implications for management such as the possibility to simultaneously reach single-species reference points and whether management could rely on target reference points only set for some species.

4.2 The Australian Southern and Eastern Scalefish and Shark Fishery (SESSF)

The Australian Southern and Eastern Scalefish and Shark Fishery (SESSF) is a multi-sector and multi-species fishery that operates in Australian federally-managed waters as well as some state waters under specific arrangements, exploiting from shallow to deep-water fishing grounds. The SESSF is currently the largest Commonwealth fishery in terms of volume caught, and the second most valuable, accounting for 20% of the gross value of production of Australian federal fisheries (Patterson et al., 2017). Around 30 species of shark and scalefish are commercially harvested in the area, with a dozen accounting for more than 75% of the fishery's Gross Value of Production (GVP). Management in the fishery primarily relies on output controls on the key commercial stocks and several by-product species. TACs are currently determined for 34 stocks based on single-species target and limit reference points and allocated as individual transferable quotas (ITQs). Introduced in 1992, ITQs brought flexibility in the fishery (Connor and Alden, 2001) and a growing activity in the quota lease market indicates falling transactions costs facilitating the reallocation of quota units (Knuckey et al., 2018). The fishery is commonly divided in 4 sectors represented by different gear types targeting specific group of species, namely the Gillnet Hook and Trap Sector (GHTS), the Commonwealth Trawl Sector (CTS), the Great Australian Bight Trawl Sector - GABTS), and the East Coast Deep-Water Trawl Sector (ECDWTS). In this work, we focus on the first two since the latter two do not interact with other sectors and are managed independently. In 2015, there were 123 active vessels in the two sectors considered, with 110 of these vessels landing more than 1 ton and employing a total of 340 crew members (estimation based on personal communication from the Australian Bureau of Agricultural and Resource Economics and Sciences - ABARES)

The fishery is characterized by consistent quota latency, with the TACs of many species being regularly under-caught (Knuckey et al., 2018). The present work investigates whether there are technical impediments to fully catching the TACs of these species due to the multi-species nature of the catch. We do so by exploring the extent to which the joint production of 3 selected pairs of scalefish species is flexible, given the possibility for fishers to operate in different metiers characterized by different species composition. These pairs, referred to as sub-fisheries in the remainder of the text, represent various types of joint production summarized in Table 4.1 and are the following:

— flathead (*Neoplatycephalus richardsoni* and four other species) and john dory (*Zeus faber*) (sub-fishery A): flathead is found in continental shelf and upper-slope waters in the eastern part of the fishery with most of its commercial catch coming from trawlers and Danish seiners at depths between 50 and 200m on the continental shelf (Patterson et al., 2017). Similarly, john dory inhabits coastal and continental shelf waters with most of its catch taken in 50–200 m depth. John dory is generally not a targeted species with most of its catch being taken in the eastern part of the fishery when targetting flathead or catching a mix of species on the continental shelf.

— flathead and jackass morwong (*Nemadactylus macropterus* - eastern stock) (sub-fishery B): like flathead, jackass morwong is found in southern and eastern continental shelf and upper-slope waters, with greater abundance in the shallower part of the range, at depths between 100 and 200m (Patterson et al., 2017). In the eastern part of the fishery, flathead and jackass morwong are therefore at the heart of technical interactions, with jackass morwong mostly coming as a by-product of trawl and Danish seine operations targeting flathead. It is however possible for trawlers to target jackass morwong in specific areas, hence giving them the possibility to increase the proportion of jackass morwong in their catch.

— flathead and pink ling (*Genypterus blacodes* - eastern stock) (sub-fishery C): pink ling mostly occurs at depths between 200 and 1000m and is frequently targeted by trawlers, and longliners between 300 and 700m on the continental slope. As a consequence, flathead and pink ling are almost never caught during the same fishing operations, although they can be harvested by the same vessels. It is for instance the case of trawlers operating in the east who can either target flathead or pink ling depending on the fishing area. Danish seiners are restricted to shallow fishing grounds and hence do not catch pink ling, and longliners operate too deep to catch significant amounts of flathead.

These four species do not have the same economic contribution to the fishery, with flathead and pink ling being primary commercial species (respectively accounting for 44 and 11% of CTS and GHTS scalefish GVP in 2015 (Patterson et al., 2017)) and jackass morwong and john dory secondary commercial species (respectively 1 and 2% of GVP).

Table 4.1 – Selected pairs of species at the heart of technical interactions in the Australian Southern and Eastern Scalefish and Shark Fishery. Their commercial importance is indicated in parenthesis. Whether it is possible to target each species and/or simultaneously catch them in "mixed" metiers was deduced from the metier identification described in Appendix D

Sub-fishery	Species 1	Species 2	Available metiers		
			Targetting species 1	Mix species 1 and 2	Targetting species 2
A	Flathead (primary)	John Dory (secondary)	Yes	Yes	No
B	Flathead (primary)	Jackass Morwong (secondary)	Yes	Yes	Yes
C	Flathead (primary)	Pink Ling (primary)	Yes	No	Yes

4.3 Methods

4.3.1 IAM bio-economic model

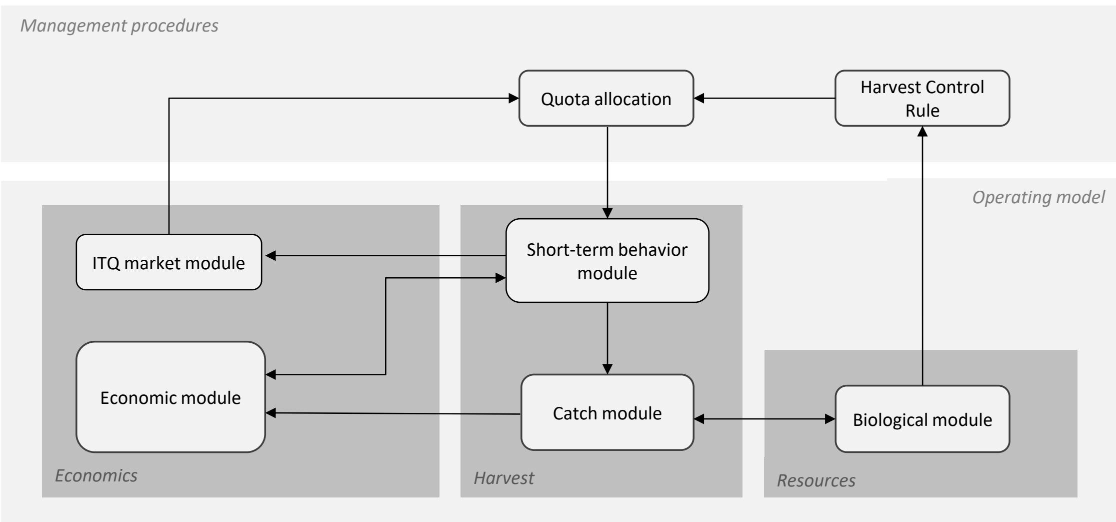


Figure 4.1 – Flow chart of the IAM model. Details of the different modules can be found in Appendix A.

Simulations were run with the integrated multi-species and multi-metier individual-based model IAM (Nielsen et al., 2018) which has been previously described in (Merzereaud et al., 2011; Bellanger et al., 2018; Macher et al., 2018; Briton et al., 2019). The equations of the model relevant for the present study are provided in Appendix A. The present work builds on the version described in Briton et al. (2019), which models the dynamics of mixed fisheries under output controls, with TACs set according to a harvest control rule (HCR) conditioned by fishing mortality targets specified as model inputs. For the purpose of the present work, this version was augmented with a module simulating the trading of individual quotas. As represented in Figure 4.1, the dynamics of the stocks are modelled in the *biological module* with catch information from the *catch module*. Stock dynamics include: age-based, age- and sex-based (Methot and Wetzel, 2013) or global surplus production (Fox, 1970) models. The stock abundance calculated by the biological module is then used to set the following year's TACs in the *Harvest Control Rule module*. Quotas are then allocated to individual operators who can decide to lease in/out quota in the *ITQ market module*. The resulting re-allocation of quota then constrains individual fishing effort in the *short-term behaviour module*, which determines the catch of various species (catch module) and a suite of economic indicators for individual operators (*economic module*). As a result of the reallocation of quota in the ITQ market module, only profitable vessels remain active in the fishery.

The ITQ market module models the formation of quota lease prices in purely competitive markets and their subsequent trading among market participants. Briton et al. (in prep.a) formally describes the model based on an iterative algortithm mimicking Walrasian "tâtonnement" process and provides a numerical application to the Australian Southern and Eastern Scalefish and Shark Fishery. Its implementation into IAM is detailed in Appendix A. As the model iteratively searches for market clearing prices, the lease price of quota units for which supply of quota (i.e. the TAC) exceeds demand for it (i.e. expected catches) is null by construction. The null lease price therefore creates an economic incentive to redirect fishing effort from species with constraining TACs towards those with excess TAC.

The short-term behaviour module allows for individual fishing practices to evolve towards what is profitable for vessel owners. Based on previous fleet-dynamics models (van Putten et al., 2012; Girardin et al., 2017), the allocation of fishing effort among various metiers is modelled as a function of a weighted average of two factors: the historical allocation, and the expected profitability of allocating effort to a particular metier. At the beginning of each annual fishing season, individual harvesters estimate the profitability of the metiers they practice ($ProfPUE^*$) based on the information available to them at the time, i.e. their past catch rates and costs but the current quota lease prices. At time t, the expected profitability $ProfPUE^*_{i,m,t}$ of metier m operated by individual i is simply its expected profits per unit of effort, i.e. the difference between expected income from fishing and operating costs (including variable costs such as fuel, crew costs, and fixed operating costs but also costs associated to leasing in quota):

$$\begin{aligned} ProfPUE^*_{i,m,t} =& (1 - cshr_i) \times \sum_s (\frac{L_{s,i,m,t-1}}{E_{i,m,t-1}} \times p_s) \\ & - \sum_s (\frac{L_{s,i,m,t-1}}{E_{i,m,t-1}} \times qp_{s,t}) \\ & - CvarUE_{i,m,t-1} - \frac{Cfix_i + Cdep_i}{E_{i,t-1}}, \end{aligned} \tag{4.1}$$

with $cshr_i$ the crew share of individual i, $\frac{L_{s,i,m,t-1}}{E_{i,m,t-1}}$ the landings per unit of effort of species s by individual i in metier m in the previous year, p_s the ex-vessel price of species s and $qp_{s,t}$ its lease price estimated by the ITQ market module. $CvarUE_{i,m,t-1}$ represents the variable costs per unit of effort for individual i in metier m in the previous year, and $Cfix_i$ and $Cdep_i$ individual fixed and depreciation costs. Relative profitabilities $ProfPUE_c$ (i.e. centred on the profitability of the individual's least profitable metier), are used in the effort allocation function to avoid negative coefficients: $ProfPUE^*_{c_{i,m,t}} = ProfPUE^*_{i,m,t} - \min_m(ProfPUE^*_{i,m,t})$.

Following Marchal et al. (2011), the proportion of effort $pE_{i,m,t}$ allocated by individual i to metier m at time t is calculated as:

$$pE_{i,m,t} = \alpha \times \frac{ProfPUE^*_{c_{i,m,t}}}{\sum_m ProfPUE^*_{c_{i,m,t}}} + (1 - \alpha) \times \frac{E_{0_{i,m}}}{\sum_m E_{0_{i,m}}}, \tag{4.2}$$

with $E_{0_{i,m}}$ being the original effort of individual i allocated to metier m. When $\alpha = 0$, individuals are assumed to operate according to past habits, whereas when $\alpha = 1$, effort allocation is entirely profit-driven.

As described in Briton et al. (2019), individuals stop fishing once they have reached their most constraining quota, which sets their annual fishing effort E_i.

Among the many indicators provided in the model output, we focus on the following selection for the purpose of this analysis:

— the number of active vessels Nbv_t in the sub-fishery at time t, i.e. the number of vessels catching at least one of the two species and which remained active in the fishery (as it was profitable) after the trading of quota:

$$Nbv_t = \sum_{i \in S_i(sp1,sp2)} \delta_{i,t}, \tag{4.3}$$

with $\delta_{i,t} = 1$ if $E_{i,t} > 0$ and 0 otherwise. $S_i(sp1, sp2) = \{i, L_{i,sp1} > 0 \vee L_{i,sp2} > 0\}$ is the set of individuals i in the sub-fishery ($sp1$,$sp2$), namely those landing at least one of the two species ($L_{i,sp1} > 0$ or $L_{i,sp2} > 0$).

— the annual tonnage of landings L_t in the sub-fishery, i.e. the annual amount landed by individuals and metiers catching at least one of the two species:

$$L_t = \sum_{(i,m) \in S_{i,m}(sp1,sp2)} L_{i,m,t}, \tag{4.4}$$

with $S_{i,m}(sp1, sp2) = \{(i, m), L_{i,m,sp1} > 0 \vee L_{i,m,sp2} > 0\}$ the set of individuals i and metiers m in the sub-fishery ($sp1$,$sp2$),

— the annual total wages of crews $Wage_t$ in the sub-fishery, calculated as a proportion of the revenue from fishing as generally observed in this fishery:

$$Wage_t = \sum_{(i,m) \in S_{i,m}(sp1,sp2)} (cshr_i \times GVL_{i,m,t}), \tag{4.5}$$

with $GVL_{i,m,t}$ the gross value of landings of individual i in metier m at time t and $cshr_i$ the share of the revenue of individual i distributed to its crew.

— and the annual Net Economic Returns NER_t of the sub-fishery:

$$NER_t = \sum_{(i,m) \in S_{i,m}(sp1,sp2)} NOS_{i,m,t}, \tag{4.6}$$

with $NOS_{i,m,t}$ the net operating surplus of individual i, metier m at time t:

$$\begin{aligned} NOS_{i,m,t} =& (1 - cshr_i) \times GVL_{i,m,t} \\ & - Cvar_{i,m,t} - (Cfix_i + Cdep_i) \times \frac{E_{i,m,t}}{E_{i,t}}, \end{aligned} \tag{4.7}$$

with $Cvar_{i,m,t}$ the variable costs of individual i in metier m at time t and $Cfix_i$ and $Cdep_i$ the fixed and depreciation costs of individual i. The latter two are allocated to metier m based on its share of fishing effort $\frac{E_{i,m,t}}{E_{i,t}}$.

These four indicators are used to quantify the socio-economic performance of the fishery and relate to different benefits that can be derived from fisheries. Concerning the production sector itself, the number of active vessels gives information about employment levels, net economic returns measure the surplus of capital owners, and wages measure the surplus of crew members. The sub-fishery's landings can be used to proxy the economic activity of the post-harvest sector (e.g. auction halls, processing plants, fishmongers) (Dyck and Sumaila, 2010). The amount of fish landed also directly affect food supply and therefore has implications for the wider society.

4.3.2 Model calibration

Biological parameters

The SESSF Harvest Strategy uses a tier-based approach conditional on data availability to assess stock status and recommend catch levels (Dowling et al., 2016; AFMA, 2017). Tier 1 assessments provide the highest quality assessments based on the estimation of age- and occasionally sex-based population dynamics. Outputs from those assessments were used to calibrate the dynamics of the Tier 1 stocks in IAM. Tiers 3 and 4 simply use indicators such as fishing mortality and catch rates to estimate stock status, without a population dynamics model being fitted to data. Consequently, important stocks in those tiers were modelled with a surplus production model whose parameters were either retrieved from Pascoe et al. (2018b) or specifically estimated for this work. Overall, 16 stocks were dynamically modelled (i.e. with either an (sex-and) age-based or surplus production model), accounting for 75% of the fishery's value in 2015, 4 with age-based dynamics, 6 with age- and sex-based dynamics and 6 with a surplus production model (Table C.1). Biological parameters are provided in Table C.2. Landings of the remaining stocks (also referred to as "static" stocks) were calculated assuming constant landings per unit of effort.

Annual stock recruitment was modelled using a Beverton-Holt stock-recruitment relationship with parameters specified in Table C.2. Uncertainties in the stock-recruitment relationship of the stocks were modelled as deviations from the stock-recruitment relationship. Formally, the observed recruitment $N_{0,t}$ was calculated as:

$$N_{0,t} = \frac{4hR_0 SSB_t}{SSB_0(1-h) + SSB_t(5h-1)} e^{\tilde{R}_t - \frac{\sigma_R^2}{2}} \quad \tilde{R}_t \sim N(\mu_R; \sigma_R^2) \tag{4.8}$$

with h the steepness parameter, R_0 the unfished equilibrium recruitment, SSB_0 the unfished equilibrium spawning biomass and R_t the deviation from the recruitment relationship drawn from a normal distribution of mean μ_R (representing recruitment shifts) and standard deviation σ_R^2. Bias in the estimation of the mean associated to the lognormal distribution is corrected by subtracting the factor $\frac{\sigma_R^2}{2}$ in the exponent

(Methot and Taylor, 2011).

Stock-specific reference points were also calculated from stock assessment outputs: F_{MSY} the fishing mortality rate maximizing yield at equilibrium and F_{20} the fishing mortality rate associated to an equilibrium biomass equal to 20% of its virgin value. The latter is the limit reference point specified by default in the Commonwealth Harvest Strategy Policy (DAFF, 2018b). Details about the calculation of both reference points are provided in Appendix C.1.

Fleet and metier characterization

Multivariate clustering analyses of catch composition at the haul level were carried out to define metiers by sector following the workflow developed by Deporte et al. (2012) to define metiers in European fisheries. Some clusters were then aggregated based on expertise from industry members to adequately describe fishing activity in the fishery. The same methodology, but this time based on the vessels' effort allocation among metiers, was used to define fleets within sectors. Both catch composition and effort data were extracted from SESSF logbooks. Details about these analyses are provided in Appendix D. A total number of 12 fleets and 29 metiers were identified in the fishery and given in Table 4.2. Figure 4.2 illustrates the fleets' fishing strategies by showing the proportion of fishing effort dedicated at targeting various species or groups of species. Fleets are not used *per se* in the model since vessels are represented individually, but are used to aggregate and present model outputs at a meaningful scale. Catches and fishing days were aggregated at the vessel and metier level to calibrate the model.

Economic parameters

Ex-vessel price for the various species in 2015 were obtained from Australian Fisheries Statistics (Mobsby, 2018). Cost structures were estimated at the fleet level based on the economic survey carried out in 2015 (Bath et al., 2018) and personal communications from ABARES, and details about the economic calibration are given in Appendix C.5. The maximal annual fishing effort of individual i (E_{max_i}) was assumed to be its maximal observed effort over the period 2010-2015.

The model endogenously determines equilibrium quota lease prices only for the species explicitly under TAC management in the simulated scenario. As described in Section 4.3.3, we looked at the joint management of pairs of species, with the control of

Table 4.2 – List of fleets and metiers for the Southern and Eastern Scalefish and Shark Fishery.

Sector	Fleets	Metiers: Gear	Metiers: Target species/assemblage
CTS	Shelf trawlers East Mixed trawlers East Royal red prawn trawlers	Trawl East	Flathead Pink ling Royal red prawn Orange roughy Jackass Morwong Squids Frostfish Ocean jackets Mixed shelf Mixed slope
	Mixed trawlers West Blue grenadier trawlers	Trawl West	Blue grenadier Pink ling Squids Mixed deepwater Mixed shelf Mixed slope
	Danish seiners	Danish seine	Flathead School Whiting Mixed
GHTS	Gillnetters	Gillnet	Gummy shark
	Shark bottomliners Mixed bottomliners	Bottomline	Gummy shark Blue-eye trevalla Mixed scalefish
	Blue-eye dropliners	Dropline	Blue-eye trevalla Gummy shark Mixed scalefish
	Blue-eye auto-longliners Mixed auto-longliners	Automatic longline	Blue-eye trevalla Pink ling Mixed scalefish

the catch of other species not being explicitly represented in the model. As explained in Chapter 3, when fishers are assumed to allocate fishing effort based on the profitability of the various metiers (i.e. $\alpha > 0$ in Equation 4.2), fishing effort tends to shift towards metiers catching more of the unregulated species because of the lower quota costs associated to those metiers. In order to avoid unrealistic effort shifts towards species not explicitly under TAC management in the simulated scenarios, we exogenously set non-null quota lease prices for the latter. This was done using data on quota lease prices collected by the Australian Fisheries Management Authority since July 2017 (Appendix C.5).

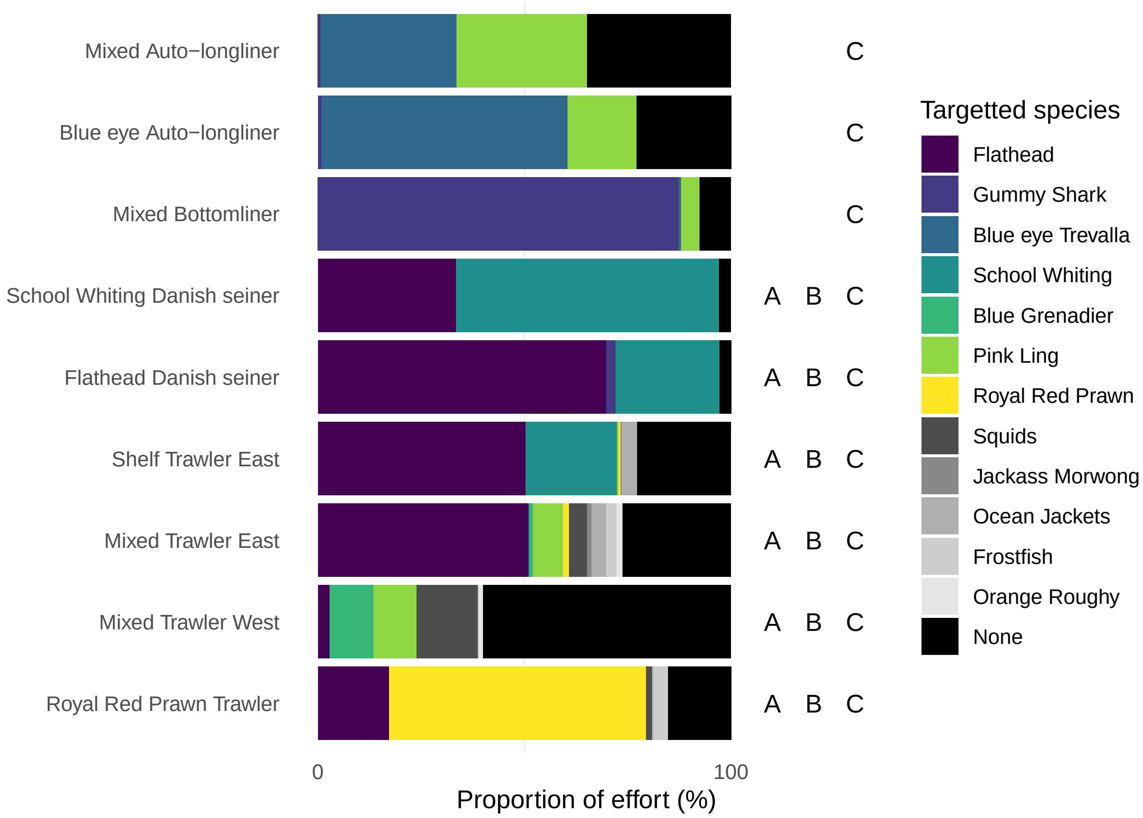


Figure 4.2 – Effort dedicated at targeting various species by fleet in the Australian Southern and Eastern Scalefish and Shark Fishery.
Primary target species are represented in colour scale, secondary target species in grey scale and black colouring refers to fishing effort allocated to "mixed" metiers, i.e. without a particular target species identified. The sub-fisheries (A: Flathead - John Dory, B: Flathead - Jackass Morwong and C: Flathead - Pink Ling) in which each fleet operates are indicated right to the plot.
Source: logbook data aggregated by fleet and metier as defined in Appendix D

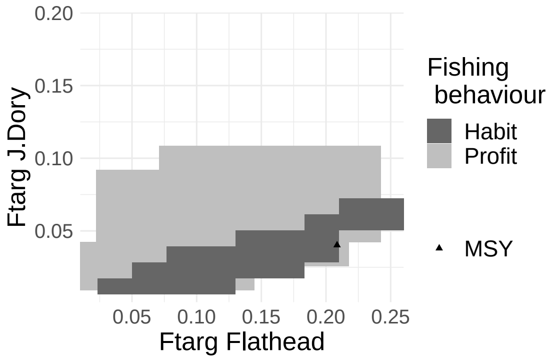
4.3.3 Exploring flexibility in joint productions

For each sub-fishery described in Section 4.2, we compared achievable catch compositions under two fishing behaviour scenarios: habit- ($\alpha = 0$ in Equation 4.2) and profit-driven ($\alpha = 1$) effort allocation. To do so, we simulated a set of combinations of fishing mortality targets used in the HCR to set annual TACs for the two species of interest. Simulations were run over a 10-year projection period and 100 replicates. The HCR is very simple and calculates annual TACs by applying a fishing mortality target $\overline{F}_{targ}$ to the current stock abundance (a detailed description of the HCR can be found in Appendix A). As in Briton et al. (2019), we identified the operating domain, defined as the subset of fishing mortality target combinations allowing the catch of at least 90% of both TACs.

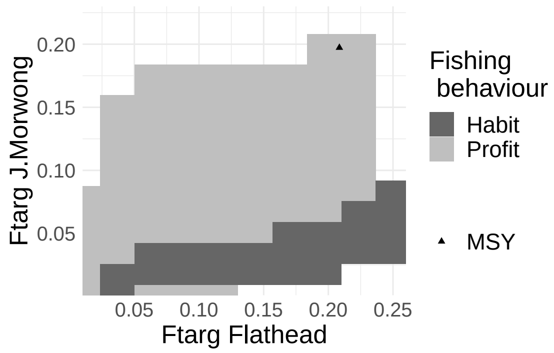
In addition to identifying the operating domain, we also calculated average performance indicators for the sub-fishery of interest, namely $\overline{Nbv}$, $\overline{L}$, $\overline{Wages}$ and $\overline{NER}$ respectively being the average through time and across replicates of Nbv_t, L_t,$Wages_t$, and NOS_t defined in Section 4.3.1.

4.4 Results

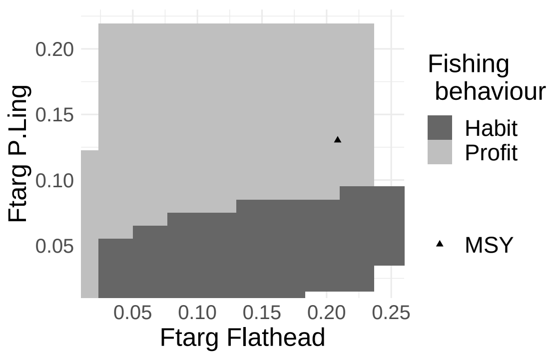
4.4.1 Flexibility in joint productions



(A) Flathead - J.Dory



(B) Flathead - J.Morwong



(C) Flathead - P.Ling

Figure 4.3 – Operating domains for three pairs of species (Flathead - John Dory, Flathead - Jackass Morwong and Flathead - Pink Ling) under two scenarios of fishing effort allocation: habit- ($\alpha = 0$) and profit-driven ($\alpha = 1$). The operating domain refers to the set of fishing mortality target combinations that allow at least 90% of the TACs of both species to be caught. Fishing mortality rates associated to single-species MSY are also represented.
Source: output from IAM model.

Starting with the operating domains (i.e. sets of achievable fishing mortality targets) resulting from a habit-driven effort allocation, one can see that whereas those of sub-fisheries A (Figure 4.3A) and B (Figure 4.3B) are narrow, that of sub-fishery C (Figure 4.3C) is wider. A linear operating domain under this hypothesis illustrates that all vessels catch both species at the annual level (but not necessarily with the same metiers), and as they operate under constant effort allocation among metiers, their fishing effort is constrained by the quotas of both species. A wider operating domain indicates that there are vessels in the fishery that only catch one of the two species, hence not being constrained by the TAC of the other one. This is the case in sub-fishery C as longliners only operate on the continental slope, targeting species such as pink ling or blue-eye trevalla, and therefore do not catch flathead present at shallower depths. This situation first requires that both species can be fished independently, i.e. by different metiers, but also that some fleets only participate in one of the two fisheries for economic (not competitive with other fleets), operational (unsuitable gear) or regulatory (exclusion from fishing grounds) reasons.

The extension of the operating domain under the profit-driven effort allocation shows that there is flexibility in achievable catch compositions in all three cases. Whereas it could be easily intuited for sub-fishery C since flathead and pink ling are caught by different metiers operating at different depths, such flexibility is more surprising for species that share the same habitat such as flathead, john dory (A) and jackass morwong (B). In the latter cases, fishers have nevertheless the possibility to alter the ratios in which they catch co-occuring species by fishing in different areas or at different times within the same habitat. In this case, the greater the targetability of each species, the greater the flexibility in achievable catch ratios. Indeed, whereas both the ratios of john dory and jackass morwong to flathead can increase when switching effort from metiers targeting flathead to more mixed metiers, that of jackass morwong to flathead can also increase when more fishing effort is allocated to targeting jackass morwong. As a result, there is more room for manoeuvre in the latter case, illustrated by a greater expansion of the operating domain in Figure 4.3B than in Figure 4.3A when allowing for effort shifts between metiers.

Interestingly, our simulations reveal that changes in fishing practices can bring single-species reference points such as F_{MSY} within the set of achievable targets. This is for instance the case of jackass morwong and pink ling being potentially harvested at MSY in the profit-driven effort allocation scenario (Figures 4.3B and 4.3C). In the 3 cases studied here, changes in fishing practices go towards higher harvest rates of john dory, jackass morwong and pink ling relative to flathead, as illustrated by operating domains extending towards the upper part of quadrant. This is a consequence of most of the fishing effort of the fleets catching flathead (mainly trawlers and danish seiners in the east) currently being dedicated at targeting flathead (Figure 4.2), hence constituting a significant amount of effort that can potentially shift towards metiers catching more of the other species (john dory, jackass morwong and pink ling) like mixed metiers or specific targeting of the latter.

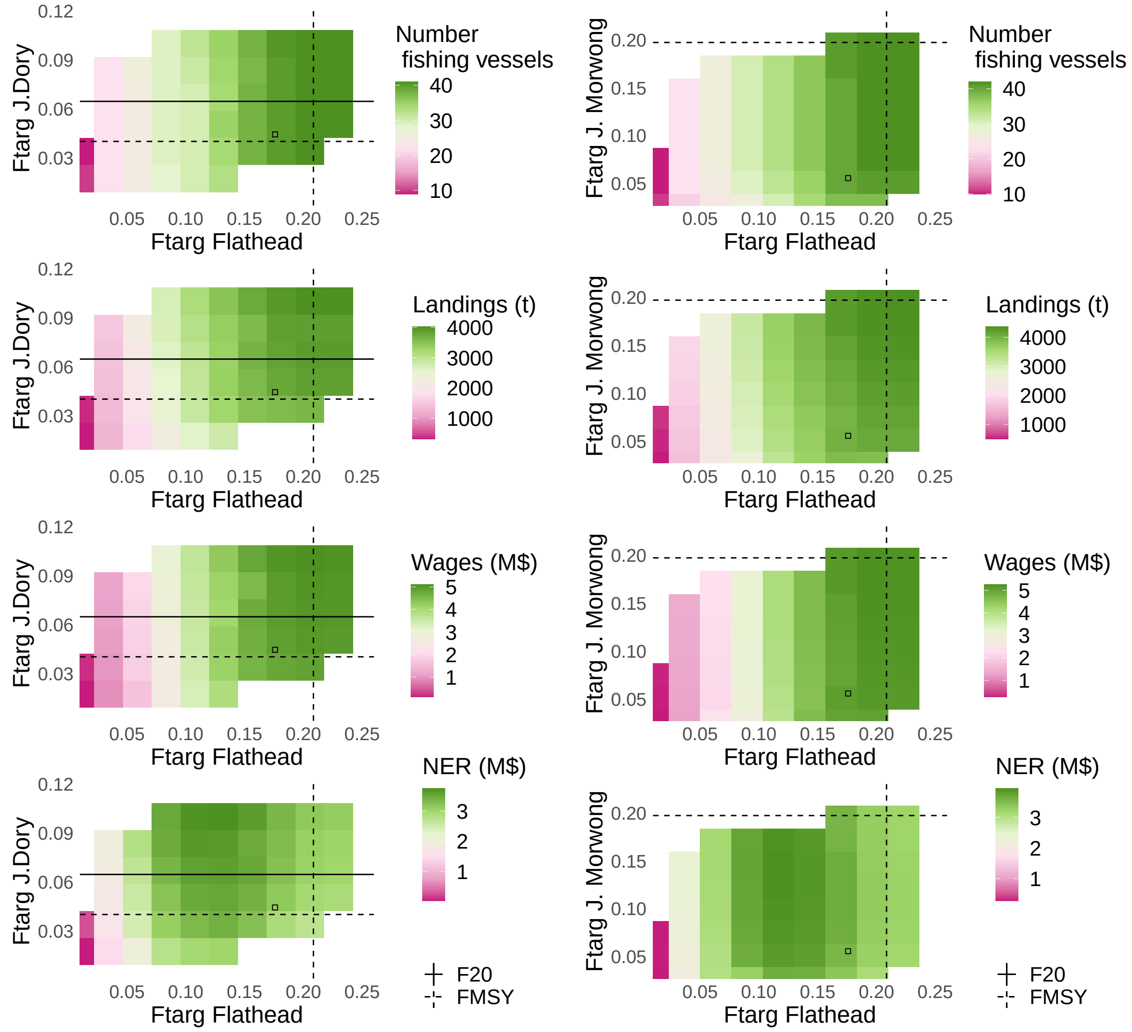
4.4.2 Socio-economic performance

Figure 4.4 presents the socio-economic implications of choosing specific management targets within the achievable set under the profit-driven fishing behaviour. We specifically show the number of active (profitable) vessels in the sub-fishery as well as the sub-fishery's landed weight, total wages, and net economic returns averaged over the 10-year projection period and across replicates.

One can notice very different patterns between sub-fisheries A (Figure 4.4A) and B (Figure 4.4B) on one side and sub-fishery C (Figure 4.4C) on the other side. Indeed, socio-economic indicators for sub-fisheries A and B are mostly driven by the fishing mortality target chosen for flathead, which is the key economic species of both sub-fisheries. This is illustrated by quasi vertical isolines for most indicators in Figures 4.4A and 4.4B, meaning that for a given harvest rate for flathead, the choice of the management target for either jackass morwong (B) or john dory (A) makes little difference to the fishery's socio-economic performance. One can also see that in these two sub-fisheries, the number of active vessels as well as the the sub-fishery's landings and crew wages are maximized under the highest harvest rates of flathead. Net economic returns, however, are maximized under lower harvest rates, a well-known result in fisheries economics (Gordon, 1954).

On the other hand, the performance of sub-fishery C is clearly dependent on the targets imposed on both species as shown by Figure 4.4C. This is a consequence of both flathead and pink ling being economically important for the fishery. Similarly to sub-fisheries A and B, the number of active vessels, total landings and wages in sub-fishery C are maximized at the highest harvest rates of both species, whereas the maximum of NER is reached under lower harvest rates of both species.

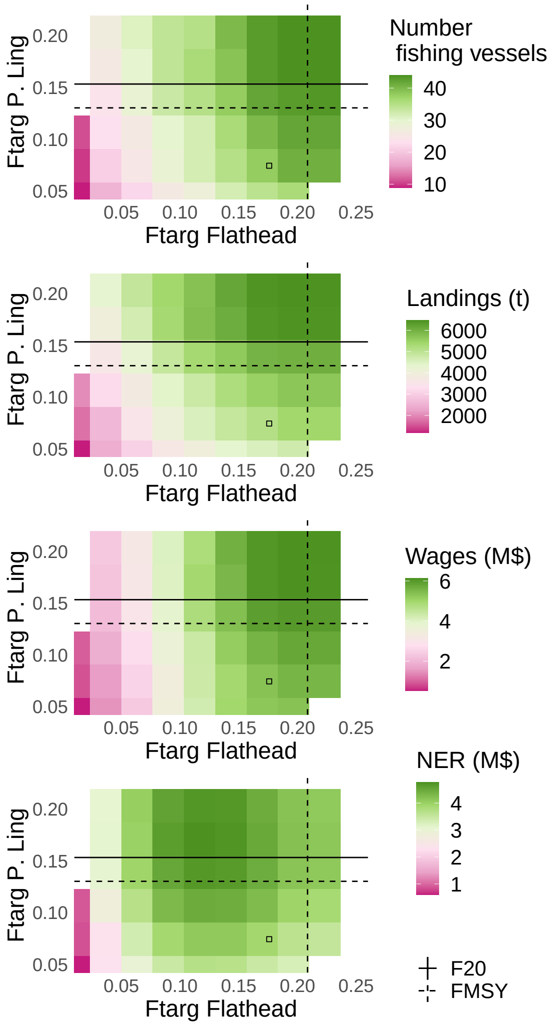
It is important to remember that the indicators' values displayed on Figure 4.4 are 10-year averages starting from the reference year and not long-term equilibria. This partly explains why sub-fishery's landings can be maximized under higher fishing mortality rates than the equilibrium F_{MSY} reference points. Another reason for landings to be maximized under fishing mortality rates higher than the single-species reference points is that landings presented here are not only those of the two species of focus but those of all species jointly caught with them. As most secondary species in the



(A) Flathead - J.Dory (B) Flathead - J.Morwong

Figure 4.4 – Fishery indicators in the profit-driven ($\alpha = 1$) fishing behaviour scenario for the three sub-fisheries.
From top to bottom: the number of active vessels in the sub-fishery, and the sub-fishery's landings, total crew wages and net economic returns (NER) averaged over the 10-year simulation period and across replicates. The blank squares show the fishing mortality rates in the reference year (2015), dashed lines the singles-species F_{MSY} reference points and the solid lines (when within the operating domain) the F_{20} limit reference points.
Source: output from IAM model

SESSF currently have catches and fishing mortality below that which would deliver MSY, overall MSY in the SESSF inevitably leads to harvesting some key stocks above their individual MSY, and sometimes above their limit reference point F_{20}.



(C) Flathead - P.Ling

Figure 4.4 – Fishery indicators in the profit-driven ($\alpha = 1$) fishing behaviour scenario for the three sub-fisheries, continued

4.5 Discussion

Our simulations show that there is significant flexibility in catch composition hidden in the choices that fishers make, and that changes in their use of already available options could allow the reconciliation of single-species objectives that cannot be simultaneously met under current fishing practices. Interestingly, such flexibility is not only observed for species that can be caught independently from each other, but also for species that spatially co-occur. The extent to which changes in fishing practices can extend the space of achievable catch compositions in the latter case is constrained by the possibility for fishers to increase or decrease the proportion in which they catch the different species by fishing in different areas or at different times within the same habitat. This is a strong message to bring to the attention of the scientific community providing TAC advice for mixed fisheries. In Europe for example, the ICES Working Group on Mixed Fisheries Advice (WGMIXFISH-Advice) provides the European Commission with alternative harvest rates within ranges around MSY so that TACs can better match the ratios in which individual species are currently caught (ICES, 2017b; Ulrich et al., 2017). By doing so, ICES has taken the path of accommodating management objectives to current fishing practices. However, this work shows that there is also room for fishing practices to adapt to the regulation, hence suggesting that appropriate approaches for the management of mixed fisheries are likely going to lie between traditional single-species approaches, simply ignoring mixed fisheries interactions, and those assuming that species are caught in absolutely fixed proportions. In this regard, TAC advice in mixed fisheries would benefit from an improved accounting of the mechanisms shaping joint productions.

Although our simulations suggest that catch composition in the SESSF is quite flexible, we would argue that the light grey operating domains in Figure 4.3 provide a rather optimistic estimation of such flexibility. First, they result from fishers only allocating their effort based on the profitability of the metiers. Fishers are however likely to be constrained by external factors such as seasonality in species availability or weather conditions restricting access to certain fishing grounds. Exogenous constraints on the time allocated to each metier could be imposed to reflect known constraints pertaining to non-modelled processes. Limited demand on the market of some species may also restrict what fishers catch, which could be addressed by accounting for market dynamics in the model and will be the object of future work. Finally, these operating domains result from perfectly functioning quota markets where all lessees find a leaser at equilibrium. However, frictions in quota markets caused by operators trading quotas in a context of uncertainty in what will be caught and without perfect information on quota supply and demand can limit the malleability of fishing practices (Innes et al., 2014a; Ropicki and Larkin, 2014). All these reasons could contribute to the perpetuation of fishing habits observed in many fisheries worldwide (Holland and Sutinen, 1999; Hutton et al., 2004; Marchal et al., 2009, 2013; Girardin et al., 2017), which could be accounted for by specifically estimating the parameter α driving fishing dynamics in the fishery, as done by Marchal et al. (2013) for the French deepwater trawlers in the North Sea. However, such estimation based on past realizations has a limited ability to capture the full potential of a system's adaptability. Another way to explore possible futures of a fishery, while acknowledging that fishers tend to adhere to habits for reasons that are difficult to capture in a model, would be to consider the cases $\alpha = 0$ and $\alpha = 1$ as bounding scenarios for its likely evolution.

Estimating the flexibility in achievable catch compositions also requires a good understanding of the production technology, without under- nor over-estimating the control fishers have over their catch composition. When defining metiers based on a statistical analysis of the fishing output, as we did, one faces the risk to miss potential options if they have not (or rarely) been exerted by fishers over the considered time period, but also that of over-estimating their control over the catch composition if a diversity in the latter is the result of environmental variability rather than an a fisher's intention. Both risks can be mitigated by calling on the industry's professionals to complement or reduce the set of metiers produced by statistical methods. In our case, we addressed the second issue by soliciting fishermen's expertise to filter out statistical clusters that did not result from a targeting intention, and merge them into a "mixed" metier as described in Appendix D. The spatio-temporal resolution at which catch data is available can also prevent an adequate determination of joint productions since catch composition can be the result of spatially and temporally fine-scaled decisions (Mateo et al., 2017). However, we do not believe it to be the case here as our clustering analysis was applied to catch data at the haul level. Finally, mixed fisheries being very dynamic systems, where fishing strategies regularly evolve in response to species availability, market demand or management regulations, it is important to regularly update analyses aiming at defining fishing strategies and their resulting catch. This would call for an institutional set-up for the drafting of advice which incorporates regular re-assessment of the structure of metiers in a fishery, involving expert input from the fishing industry.

Given a potentially important part of joint productions being born by fishers' choices (as suggested by this work but also highlighted by several empirical studies before us such as Branch and Hilborn (2008), Abbott et al. (2015) or Little et al. (2015)), we can only emphasize the need to pursue research aiming at better understanding fishing behaviour in mixed fisheries (drivers as well as limitations), but also to start using this knowledge to advise on the management of those fisheries through its implementation in decision-support modelling frameworks. In this regard, we support the conclusion drawn by Ulrich et al. (2011) that both metier and operator entities should be represented in integrated models of mixed fisheries since they respectively materialize the technical and behavioural determinants of joint productions. Whereas omitting the metier level is likely to under-estimate the flexibility in catch composition resulting from changes in fishing practices, representing a fishery as a set of independent metiers without accounting for their joint operation by individual operators is likely to over-estimate that same flexibility.

This work also shows a hierarchy among species in a mixed fishery, with the harvest targets imposed on the primary commercial species determining most of the fishery's production, employment and economic surplus. Secondary species only have a marginal influence on the overall performance of the fishery. This has practical implications for management as it justifies setting targets that align with socio-economic management objectives in priority for the primary commercial species. In this context, we could imagine the management of secondary species being "only" subject to ecological objectives and subsequent to decisions made for the primary species, with catch limits being set within the achievable range associated to a given catch limit on the primary species, and capped by a limit reference point pertaining to stock conservation or wider ecological objectives. Although not tested, we can reasonably expect these conclusions drawn for the SESSF to apply to other mixed fisheries constituted of a few key commercial species and a suite of secondary species.

4.6 Conclusion

This work advances the on-going development of management approaches specifically addressing the complexities faced in mixed fisheries. In particular, our simulations suggest that mixed fisheries are likely to feature an important latent flexibility in their catch composition, and that changes in fishing practices can broaden the space of achievable outcomes. Accounting for the behavioural determinants of joint productions is therefore critical to the provision of relevant TAC advice in mixed fisheries. Moreover, our results suggest that a hierarchical approach to management may be appropriate for mixed fisheries, with target reference points meeting socio-economic objectives being set for the key commercial species, and secondary species being managed so as to steer clear of ecological limit reference points.

Acknowledgements

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CHAPTER 5

FROM FISH STOCKS TO FISHERS AND CONSUMERS: ECO-VIABILITY IN THE AUSTRALIAN SOUTHERN AND EASTERN SCALEFISH AND SHARK FISHERY

This chapter is in preparation for submission to Fisheries Research as:
Briton, F., Thébaud, O., Macher, C., Gardner, C., Mobsby, D., Little, L.R. From fish stocks to fishers and consumers: eco-viability in the Australian Southern and Eastern Scalefish and Shark Fishery.

Abstract

While Maximum Sustainable Yield or Maximum Economic Yield are largely advocated as desirable targets for fisheries management, the analysis of trade-offs associated with achieving such objectives is generally absent from scientific advice to management. This work applies the eco-viability approach to a multi-species fishery, the Australian Southern and Eastern Scalefish and Shark Fishery, characterized by technical and economic interactions among harvested stocks. We identify output controls on two key species of the fishery (flathead and pink ling) that ensure its biological and economic viability, but also maintain the consumer's demand for fish. We highlight several trade-offs related to the distribution of benefits between vessel owners, fishing crews and consumers. Specifically, we show that maximizing the economic returns to vessel owners, which is the current management target in the fishery, does not maximize benefits to the society as it impedes the surplus of fishing crews as well as that of consumers.

5.1 Introduction

Since the Reykjavik Conference on Responsible Fisheries in the Marine Ecosystem in 2001, Ecosystem-Based Fisheries Management (EBFM) has been adopted by many jurisdictions worldwide as the new standard for managing fisheries. EBFM recognizes that harvested fish stocks are not independent resources, and that their connections in the socio-ecosystem should be accounted for when managing fishing activities (Leslie and McLeod, 2007; Marasco et al., 2007).

Technical interactions have long been put forward as a critical obstacle to successful output management in multispecies fisheries. First, they complicate the use of single-stock target reference points in mixed fisheries, simply because species cannot always be caught in the proportions prescribed by their respective targets (Vinther et al., 2004; Ulrich et al., 2011, 2017). Even in jurisdictions where policy objectives have been formulated at the fishery level, thereby acknowledging its multispecies nature, the complexity and additional data requirements of the models needed to operationalize such objectives have often impeded their application in practice (Dichmont et al., 2010; Pascoe et al., 2015; Hoshino et al., 2018) . Second, increasingly adopted discard mitigation policies (Karp et al., 2019), and discard bans in their strictest form, have been difficult to implement in mixed fisheries because of technical interactions. Indeed, the prospect of fisheries "choking" on species with low catch possibilities (Schrope, 2010; Patrick and Benaka, 2013) triggered important resistance to implementing effective discard policies in these fisheries (Fitzpatrick et al., 2019).

The growing awareness among scientific advisory bodies and management agencies that technical interactions can hinder the successful implementation of policy objectives in multispecies fisheries has motivated their representation in the models used to evaluate the impact of management decisions. The representation of multi-species economic interactions in such models has however been more sporadic (for a recent review of fisheries ecological-economic models see (Nielsen et al., 2018)). Economic interactions can concern the output from fishing, i.e. the captured fish, with various species acting as substitute or complementary products on markets (Sharma et al., 2003; Bose, 2004). They can also pertain to the activity's inputs. For instance, in fisheries managed with Individual Tradable Quotas (ITQs), quota costs associated to harvesting a species depend on the price of quota of jointly caught species. By affecting revenues and costs, interactions in the economic sphere orient fishing strategies (van Putten et al., 2012), upon which ultimately depend the ecological and socio-economic outcomes of the fishery. Economic interactions are therefore important to account for in the design and ex-ante evaluation of regulation measures in multi-species fisheries.

EBFM aims to embrace the four pillars of sustainable development, namely ecological, economic, social and institutional (De Young et al., 2008). However, progress in EBFM has so far been biased towards its ecological aspects, resulting in poorly defined objectives and management processes on its human dimensions (economic, social and institutional) (Symes and Phillipson, 2009; Link et al., 2017; Stephenson et al., 2017). Management objectives relating to the human dimensions are generally stated in aspirational terms in national (DFO, 2016; NOAA, 2016; DAFF, 2018b) and international (EU, 2009) policies, but have minor effect on management of individual fisheries.Hornborg et al. (2019) also highlighted that the inclusion of the human dimension in EBFM has primarily addressed economic sustainability, with less attention given to social and institutional components. Even the economic dimension is far from being addressed comprehensively as focus has generally been put on the industry's economic performance, to the expense of wider societal economic benefits such as the surplus of the downstream sector or that of consumers.

In addition to properly defining management objectives, EBFM requires the development of methods to embrace a variety of objectives in the management process. Particularly well-suited to the purpose, the eco-viability approach has been used to advise on the sustainable management of renewable resources, with nearly half of its applications being fisheries case studies as highlighted by a recent review from Oubraham and Zaccour (2018). Fundamentally different from optimization approaches which seek solutions optimizing an (multi-)objective function, eco-viability looks for solutions that are "good enough", i.e that maintain the system within acceptable bounds. This echoes Simon's theory of *satisficing* (versus optimal) decision-making (Simon, 1956). The multi-dimensional nature of sustainability has been well recognized by eco-viability studies of fisheries. All recognized the need to ensure its ecological viability, either through constraints on stock abundance (most frequently chosen) or levels of bio-diversity (Cissé et al., 2013, 2015). Explicit conservation constraints on by-catch species from the fishery have also been proposed by Gourguet et al. (2016), hereby accounting for the ecological impacts from fishing beyond that on harvested resources. Economic viability has also been given particular attention, although with different interpretations across case-studies. Some related it to the profitability of fishing fleets, either directly measured by the generated profits (Gourguet et al., 2013) or its expression in the lease price of quota in ITQ fisheries (Péreau et al., 2012), while others also accounted for economic constraints imposed on the remuneration of fishing crews (Briton et al., 2019; Maynou, 2019), or the depreciation of physical capital (Briton et al., 2019). Finally, social constraints have sometimes been considered, such as meeting food security requirements (Eisenack et al., 2006; Hardy et al., 2013; Cissé et al., 2013, 2015), maintaining some level of fishing activity as a proxy for employment in the fishery (Péreau et al., 2012; Gourguet et al., 2013), or answering the demand of recreational fishers for "trophy" catches (Thébaud et al., 2014).

The work presented here builds on previous modelling work carried out by Briton et al. (2019) looking for catch limits on jointly caught species allowing for the French demersal fishery in the Bay of Biscay to remain within an eco-viable space. The constraints considered there pertained to ensuring the viability of exploited stocks as well as the ability of the fleets to be economically profitable by accounting for the depreciation of the fishing capital and the opportunity cost of labour. Advancing in the comprehensive representation of sustainability in the eco-viability framework, the present application to the Australian Southern and Eastern Scalefish and Shark Fishery (SESSF) also explicitly accounts for the need to maintain affordable fish for the consumer when managing the fishery, and this through an upper limit on the price of fish.

5.2 The Australian Southern and Eastern Scalefish and Shark Fishery (SESSF)

The Australian Southern and Eastern Scalefish and Shark Fishery (SESSF) is a multi-sector and multi-species fishery that operates in waters under federal jurisdiction as well as some state waters under specific arrangements. The SESSF is currently the largest federal fishery in terms of volume caught, and the second most profitable, accounting for 20% of the gross value of production (GVP) of federal fisheries (Patterson et al., 2018). Around 30 species of shark and scalefish are commercially harvested in the area, with a dozen accounting for more than 75% of the fishery's GVP. Management in the fishery primarily relies on output controls on the key commercial stocks and several by-product species. TACs are currently determined for 34 stocks and allocated as individual transferable quotas. Catch limits are complemented by several input controls such as limited entry, gear restrictions, spatial closures and trip limits for certain species. The fishery is commonly divided in four sectors represented by different gear types targeting specific group of species, namely the Gillnet Hook and Trap Sector, the Commonwealth Trawl Sector, the Great Australian Bight Sector, and the East Coast Deep-Water Trawl Sector. The two latter were not included in the present work as they do not interact with other sectors and can thus be managed independently.

Like other federal fisheries, management in the SESSF is subject to the objectives and directives of the recently revised Commonwealth Harvest Strategy Policy. The objective of this policy is the "*ecologically sustainable and profitable use of Australia's Commonwealth commercial fisheries resources (where ecological sustainability takes priority)*" (Department of Agriculture and Water and Resources, 2018). Fisheries are to be managed towards their Maximum Economic Yield (MEY), which is defined as the "*[maximization of] net economic returns to the Australian community*" (Department of Agriculture and Water and Resources, 2018). Although the policy specifically states that fisheries management should benefit the whole Australian community, catch limits have so far been set so that rents from the fishery are private and the profits of the commercial fishery are maximized. Nonetheless, the desire to account for the broader society's interests in the valuation of fisheries economic returns has been recognized (Vieira, 2013) and recent academic work carried out by Pascoe et al. (2018b) proposed to include consumer surplus and non-market costs associated with by-catch in the estimation of MEY. Unlike other Australian fisheries, fish caught in the SESSF are primarily destined to the national market which makes the accounting of consumer surplus in the valuation of the fishery's economic returns particularly relevant.

5.3 Methods

5.3.1 IAM bio-economic model

Model description

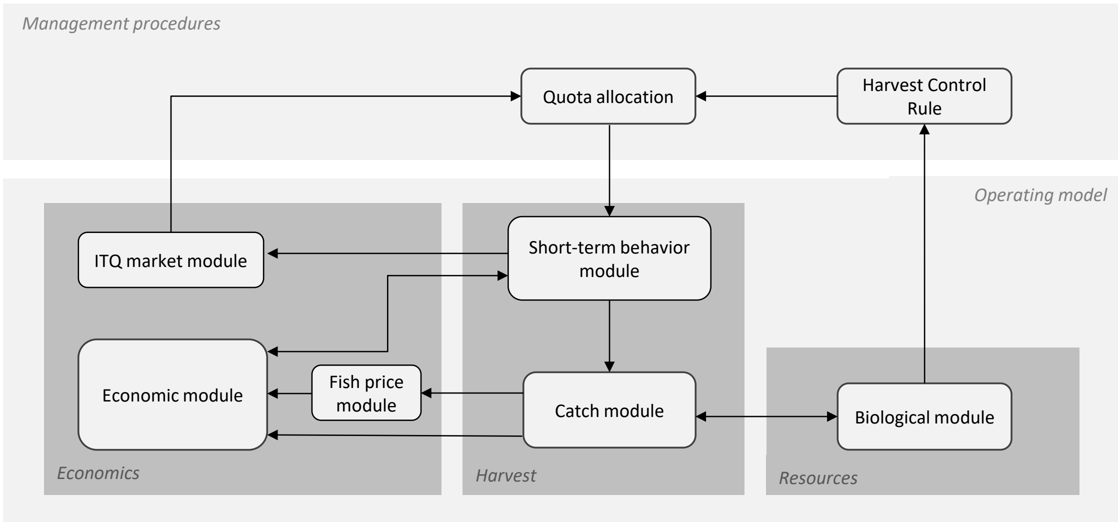

Figure 5.1 – Flow chart of the IAM model. Details of the different modules can be found in Appendix A

Simulations were run with the multi-species and multi-metier individual-based bio-economic model IAM which has been previously described in (Merzereaud et al., 2011; Bellanger et al., 2018; Macher et al., 2018; Briton et al., 2019). The present work builds on the version described in Chapter 4, which models the dynamics of mixed fisheries managed with Individual Tradable Quotas (ITQs) on several stocks. For the purpose of the present work, the version described in Briton et al. (in prep.b) was augmented with a module simulating the endogenous response of fish prices to the amount of fish landed. A flow chart of the model is presented in Figure 5.1 and the details of the modules already described in the publications mentioned above are provided in Appendix A.

The model is composed of an *operating model* representing the dynamics of the fishery, the latter being divided into its *resource*, *harvest* and *economic* components. Linked to the operating model, the *management procedures module* models the management of the fishery with stock-specific Total Allowable Catch (TAC) limits determined by a *Harvest Control Rule* (HCR) which applies a fishing mortality target $\overline{F}_{targ}$ specified in input to the stock's abundance calculated by the *biological module*[1]. The following options are currently available to model stock dynamics: a Fox global surplus production model, an age-structured model, a sex- and age-structured model, and a constant biomass model. TACs are then allocated in the form of individual quotas constraining individual fishing efforts in the *short-term behaviour module.* Over-quota discards are not allowed in the model, and fishers must stop fishing once they have reached their most limiting quota. The short-term behaviour module allows for individual harvesters to dynamically determine their fishing strategy, to be understood here as the allocation of fishing effort among a set of metiers they can practise. Following Marchal et al. (2011), effort allocation is modelled as a function of the weighted average between the expected profitability and the effort historically allocated to each metier, with the weight α given to profitability relative to habit being specified as a model input. To enable such changes in individual fishing strategies, the *ITQ market module* models the trade of quota units among operators. Briton et al. (in prep.b) compared two extreme scenarios of fishing behaviour, namely entirely profit-driven ($\alpha = 1$) and entirely habit-based ($\alpha = 0$), and highlighted the importance of modelling the fishers' endogenous response in decision-support simulation tools. To progress along this path, the present work focuses on the profit-driven fishing scenario ($\alpha = 1$).

In the present version of the model, the price of fish dynamically responds to the species' landings calculated by the *catch module.* Own- and cross-price flexibilities are used in the *fish price module* to calculate the price of fish as follows:

$$\ln(\frac{p_{s,t}}{p_{s,0}}) = \sum_{s'} \beta_{s,s'} \times \ln(\frac{L_{s',t}}{L_{s',0}}), \tag{5.1}$$

with $p_{s,t}$ the price of species s at time t, $L_{s,t}$ the landings of species s at time t and $\beta_{s,s'}$ the flexibility of the price of species s with regards to the landings of species s'.

Finally, the *economic module* calculates a variety of economic indicators at the vessel level, with an exhaustive list provided in Bellanger et al. (2018). In order to consider the MEY objective for fishery, while also considering some aspects usually not factored in, such as the capacity to maintain crews, the present work pays particular attention to the following indicators:

1. There is no observation uncertainty in the model in the sense that the HCR has perfect knowledge on stock abundance

— the individual gross operating surplus (GOS):

$$\begin{aligned} GOS_{i,t} = &(1 - cshr_i)\, GVL_{i,t} \\ &- \sum_s L_{s,i,t}\, qp_{s,t} - \sum_m (CvarUE_{i,m} \times E_{i,m,t}) - Cfix_i, \end{aligned} \tag{5.2}$$

with $cshr_i$ the crew share of vessel i, $L_{s,i,t}$ the landings of species s by individual i at time t, $GVL_{i,t} = \sum_s L_{s,i,t} \times p_{s,t}$ the gross value of landings of individual i at time t, $p_{s,t}$ the ex-vessel price of species s at time t and $qp_{s,t}$ its quota lease price as determined by the ITQ market module (see Appendix A for a detailed description of this module), $CvarUE_{i,m}$ the variable costs per unit of effort of vessel i in metier m and $Cfix_i$ the fixed costs of individual i [2].

— the individual net operating surplus (NOS):

$$NOS_{i,t} = GOS_{i,t} - Cdep_i - Copport_i, \tag{5.3}$$

with $Cdep_i$ and $Copport_i$ the depreciation and opportunity costs of capital of individual i.

— Crew wages at the vessel level, with crews being remunerated a share of the fishing income:

$$Wage_{i,t} = cshr_i \times GVL_{i,t}, \tag{5.4}$$

and its Full Time Equivalent (FTE):

$$Wage_{FTE_{i,t}} = \frac{Wage_{i,t}}{FTE_{i,t}}, \tag{5.5}$$

with FTE the full-time equivalent number of crew.

Some indicators are also calculated at the fishery level, by summing vessel-level indicators across all vessels operating in the fishery (here, participating in the flathead and/or pink ling fishery), namely:

— the fishery's Net Economic Returns (NER) at time t (i.e. the surplus of vessel owners):

$$NER_t = \sum_i NOS_{i,t} \tag{5.6}$$

— the fishery's net wages for crews ($NWage$) at time t (i.e. the surplus of fishing crews):

$$NWage_t = \sum_i (Wage_{i,t} - Wage_{opport} \times FTE_{i,t}), \tag{5.7}$$

2. In the absence of data on the vessels' initial quota holdings, we assumed in the model that the totality of quota required to operate had to be leased in.

with $Wage_{opport}$ the wage of opportunity for fishing crews.

Model calibration for the SESSF

The model was calibrated on calendar year 2015. Details on the calibration of the biological, catch and economic modules for the fishery can be found in Appendix C and Briton et al. (in prep.b). The dynamics of fish markets were parametrized based on own- and cross-price flexibilities estimated for the SESSF by Bose (2004) and only significant coefficients were kept.

Stock-specific reference points were also calculated from stock assessment outputs: F_{MSY} the fishing mortality rate maximizing yield at equilibrium and F_{20} the fishing mortality rate associated to an equilibrium biomass equal to 20% of its virgin value (see Appendix C for a detailed description of their calculation).

5.3.2 Eco-viability framework

The present analysis builds on viability thresholds proposed by Briton et al. (2019), with an additional constraint pertaining to satisfying the consumer's demand for fish.

Viability constraints

The first sustainability requirement is to ensure the biological viability of harvested stocks in the fishery by maintaining their biomass above a limit threshold B_{lim}, below which recruitment may be impaired. Depending on the representation of a stock s in the model, its biological viability was calculated as:

$$\begin{aligned} V_{BIO}(s) &= 1 \quad \text{if} \quad \begin{cases} SSB_s(t) \geq B_{lim_s} & \forall t \in [t_0; t_f] \quad \text{for age-based dynamics,} \\ B_s(t) \geq B_{lim_s} & \forall t \in [t_0; t_f] \quad \text{for global dynamics,} \end{cases} \\ &= 0 \quad \text{otherwise.} \end{aligned} \tag{5.8}$$

We used the limit reference point specified in the Harvest Strategy Policy, set at 20% of the stocks' virgin biomass, as the threshold B_{lim} below which there is an unacceptable risk of recruitment failure (Department of Agriculture and Water and Resources, 2018). Estimations of virgin biomass from stock assessment reports were used to calculate the limit reference points reported in Table 5.1.

Second, the fishing industry should be economically viable which requires fishing vessels to be able to maintain their means of production, i.e. capital and labour. A positive Net Operating Surplus (NOS) over a 10-year period ($\overline{NOS} = \frac{\sum_{t'=t_0}^{t_0+10} NOS(t')}{10}$) would secure the renewal of capital (i.e. cover its depreciation) and ensure the remuneration of physical capital at its opportunity cost. The 10-year period was chosen as it figures as a relevant time scale for the renewal of fishing capital. The long-term economic viability of any vessel i was thus calculated as:

$$\begin{aligned} V_{LT}(i) &= 1 \quad \text{if} \quad \overline{NOS}(i) \geq 0, \\ &= 0 \quad \text{otherwise.} \end{aligned} \tag{5.9}$$

In addition to the vessels' long-term economic viability, we also considered their capacity to regularly cover their operating costs, i.e. show a positive Gross Operating Surplus (GOS) every year. This constraint specifically ensures the regularity of economic performance, a common request from the fishing industry. The short-term economic viability of vessel i was calculated as:

$$\begin{aligned} V_{ST}(i) &= 1 \quad \text{if} \quad GOS(i,t) \geq 0 \quad \forall t \in [t_0; t_f], \\ &= 0 \quad \text{otherwise.} \end{aligned} \tag{5.10}$$

The last sustainability requirement for the fishing industry is to maintain fishing crews in the fishery, which was ensured by maintaining their annual full-time equivalent (FTE) wage above the opportunity wage of crews in the fishery ($Wage_{opport}$). The crew viability of vessel i was thus calculated as:

$$\begin{aligned} V_{CREW}(i) &= 1 \quad \text{if} \quad Wage_{FTE}(i,t) \geq Wage_{opport} \quad \forall t \in [t_0; t_f], \\ &= 0 \quad \text{otherwise.} \end{aligned} \tag{5.11}$$

The absence of wage statistics for the Australian fishing sector makes it difficult to estimate the opportunity wage of fishing crew in the SESSF. Squires (1988) estimated the opportunity cost of labour in the Pacific Coast Trawl fleet as the mean wage earned in manufacturing, transportation, and retail trade sectors. We estimated the opportunity wage of an Australian fishing crew using a similar approach based on statistics provided in Australian Bureau of Statistics (2016) to be AU$ 68,600 in 2015. This is above advertised wages found on the Internet in October 2019 and corrected for wage inflation (between AU$ 50,000 and AU$ 55,000), as well as the minimum wage for deckhands (which is not restricted to fishing deckhands) in Australian law as found in FairWork (2019) (AU$ 43,700). A compromise value of AU$ 60,000 was consequently chosen to represent the opportunity FTE wage of a crew member in the SESSF.

The last sustainability constraint pertains to satisfying the consumer's demand for fish by preventing the cost of fish to exceed an upper acceptability threshold. Paying attention to the consumer's demand is particularly relevant in our case-study as the recent review from Christenson et al. (2017) identified the price of fresh seafood and availability of local fish as substantial barriers to seafood consumption in Australia.

Calculating changes in the cost of fish for the consumer is not straightforward as it is a function of changes in the price of a set of products that have some level of substitutability. In this case, indices measuring the evolution of the price of a fixed basket of products, such as the Laspeyres or Paasche indices, are known to either positively or negatively bias inflation in this area of expenditure. The Fisher price index, calculated as the geometric mean of the Laspeyres and Paasche indices, corrects the positive and negative substitution biases respectively associated to these two indices, and provides therefore a good (and in some cases, perfect) estimate of changes in the cost of a product category (Diewert et al., 2009). The Fisher price index is defined as follows:

$$P_{F_t} = \sqrt{P_{L_t} \times P_{P_t}}, \tag{5.12}$$

with P_{L_t} and P_{P_t} being the Laspeyres and Paasche indices at time t and calculated as follows:

$$\begin{aligned} P_{L_t} &= \frac{\sum_s p_{s,t} \times L_{s,t_{ref}}}{\sum_s p_{s,t_{ref}} \times L_{s,t_{ref}}}, \\ P_{P_t} &= \frac{\sum_s p_{s,t} \times L_{s,t}}{\sum_s p_{s,t_{ref}} \times L_{s,t}}, \end{aligned} \tag{5.13}$$

with $p_{s,t}$ and $L_{s,t}$ respectively being the price and landings of species s at time t, and t_{ref} the reference year for the index, chosen here as the year of calibration, i.e. 2015.

The consumer's satisfaction, or consumption viability was then calculated as:

$$\begin{aligned} V_{CONS} &= 1 \quad \text{if} \quad P_{F_t} \leq P_{F_{max}} \quad \forall t \in [t_0; t_f], \\ &= 0 \quad \text{otherwise.} \end{aligned} \tag{5.14}$$

An upper threshold for the Fisher price index was estimated as the maximal value of the index observed between 2000 and 2015 in the SESSF. The index's time series was reconstructed based on price and production data from the Australian Fisheries Statistics (Mobsby, 2018), and adjusted for inflation with the Consumer Price Index for "Food and non-alcoholic beverages" provided by Australian Bureau of Statistics (2019). The maximal price index over that period was 1.2, which represents a 20% increase compared to the reference year 2015.

Table 5.1 summarizes the viability constraints used in the study.

Table 5.1 – Acceptability constraints

Aim	Name	Time scale	Related entity	Value
Stock persistence	B_{lim}	year	Blue-eye trevalla	2,475 t
			Blue grenadier	10,782 t
			Flathead	4,620 t
			Gummy shark	3,474 t
			Jackass morwong (East)	1,410 t
			Jackass morwong (West)	548 t
			John Dory	874 t
			Mirror Dory	2,678 t
			Ocean Perch (offshore)	238 t
			Orange roughy (East)	8,327 t
			Pink ling (East)	1,534 t
			Pink ling (West)	1,429 t
			Redfish	2,401 t
			School shark	7,215 t
			School whiting	1,509 t
			Silver warehou	3,790 t
Cover operating costs	GOS_{min}	year	vessel	AU$ 0
Remunerate physical capital	NOS_{min}	10-year period	vessel	AU$ 0
Maintain crews	$Wage_{opport}$	year	vessel	AU$ 60,000
Satisfy consumers	$P_{F_{max}}$	year	fishery	1.20

Eco-viability under uncertainty

De Lara and Doyen (2008) and De Lara et al. (2015) formalized the eco-viability framework under uncertainty. In this case, viability is assessed probabilistically through Monte-Carlo simulations covering the uncertainty range of the uncertain factor(s). Uncertainties in the present case relate to the estimation of the stocks' recruitment.

For each management strategy St described in the following section, probabilities were calculated for each type of viability ($P_{V_{BIO}}(St, s)$, $P_{V_{ST}}(St, v)$, $P_{V_{LT}}(St, v)$,

$P_{V_{CREW}}(St, v)$, and $P_{V_{CONS}}(St)$) as:

$$P_V(St) = \frac{\sum_{rep=1}^{n_{rep}} V(St, rep)}{n_{rep}}, \tag{5.15}$$

with the subscript rep referring to the replicate and n_{rep} being the number of replicates.

Each management strategy was run across 100 replicates to account for uncertainties in stock recruitments and projected over a 10-year period.

5.3.3 Simulation plan

Similarly to Briton et al. (2019), the integrated model IAM was used to identify achievable and eco-viable fishing mortality targets in a mixed fishery. In Chapter 4 we showed that the socio-economic dynamics of a multi-species fishery like the SESSF are mostly driven by its key commercial species, thus suggesting that the latter are the ones for which the allowed catch will be critical at meeting socio-economic objectives. Therefore, this application will focus on the joint management of two key species in the eastern part of the SESSF: flathead and pink ling.

We followed the same simulation approach as in Briton et al. (2019) by simulating a 2D grid of combinations of fishing mortality targets ($\overline{F}_{targ}$) used to set annual TACs on the two species of interest. A combination in this grid constitutes a particular management strategy. Within the simulated grid, we identified the operating domain, namely the area where both TACs are caught by at least 90%. Eco-viability and trade-offs within the operating domain were then assessed.

5.4 Results

5.4.1 Operating domain accounting for market dynamics

Figure 5.2 compares the space of achievable fishing mortality targets under 2 scenarios: one where the price of fish remains constant (results retrieved from Briton et al. (in prep.b)) and one where it responds to the quantity of fish landed. The dynamics on fish prices affect the economic incentives for fishers to harvest fish stocks. Indeed, as landings increase, the price of fish decreases, which diminishes economic returns compared to a situation where prices remain constant. As a consequence, it becomes unprofitable for fishers to fully catch the TAC when it is set with a harvest rate above $0.18yr^{-1}$ for flathead and $0.16yr^{-1}$ for pink ling, resulting in the operating domain shrinking on the top and right ends between the constant and dynamic price scenarios.

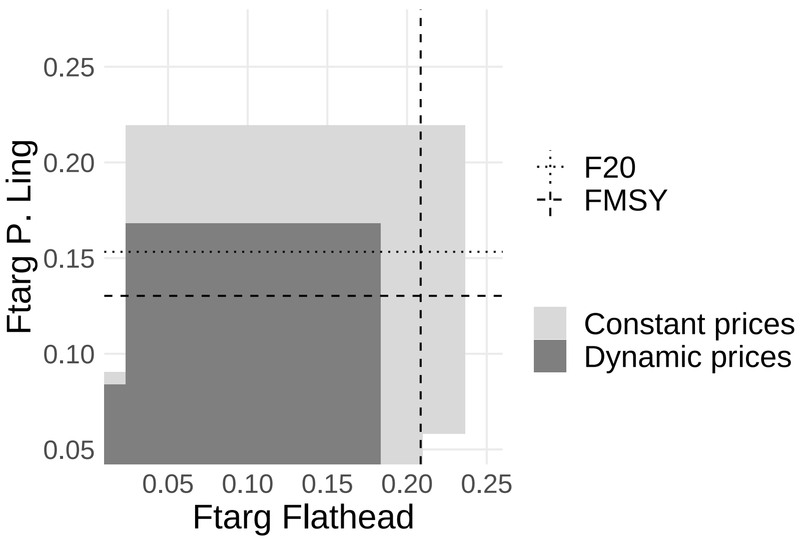

Figure 5.2 – Comparison of operating domains under the constant (i.e. no price reponse to changes in the quantities and composition of landings) and dynamic (i.e. the latter responses are considered) fish prices scenarios. The operating domain for the constant fish prices scenario is reproduced from Chapter 4. F_{MSY} is the fishing mortality rate maximizing yield at equilibrium for the stock and F_{20} the fishing mortality rate associated to an equilibrium biomass equal to 20% of its virgin value.
Source: output from IAM model

Eventually, the operating domain reduces down to below the MSY reference point for flathead and slightly above the F_{20} limit reference point for pink ling.

5.4.2 Eco-viability analysis

Biological viability

The number of biologically viable stocks in each scenario of the operating domain is given in Figure 5.3. Among the 16 modelled stocks, only 15 can be maintained above their limit biomass reference point throughout the projection period with a probability of 90%. The only stock which always fails at meeting the constraint is redfish. Concerns relative to this stock's sustainability are not new since it is estimated to have been overfished since the early 1990's according to the latest stock assessment (Tuck et al., 2017). In 2015, the stock was estimated at 4% of its virgin biomass, and depletion levels after 10 years range from 9 to 11% (median values) throughout the set of our simulated scenarios (data not shown). The present work was not designed to specifically address the rebuilding issue of that stock, and therefore, the constraint pertaining to its viability was released in the remainder of the study. The biologically viable domain refers to the domain ensuring the viability of the 15 remaining stocks (with a probability of 90%). The other stock whose viability is threatened under some scenarios is the eastern stock

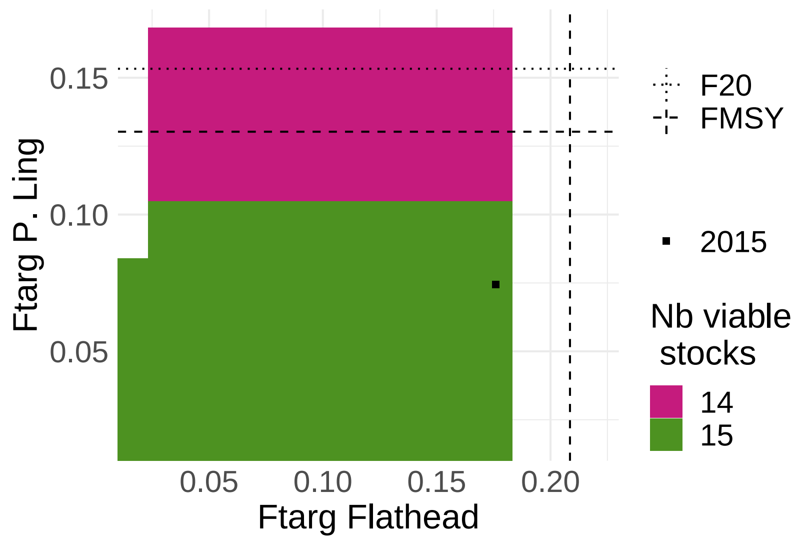

Figure 5.3 – Number of viable stocks (i.e. with a probability of remaining above their limit biomass reference point above 90%) in the operating domain. F_{MSY} is the fishing mortality rate maximizing yield at equilibrium for the stock and F_{20} the fishing mortality rate associated to an equilibrium biomass equal to 20% of its virgin value. The black square indicates harvest rates in the reference year (2015).
Source: output from IAM model

of pink ling, one of the two stocks under TAC regulation in our simulations. As shown by Figure 5.3, this stock's biological viability is ensured with at least 90% probability for harvest rates below $0.11 yr^{-1}$. It is worth noting that both F_{MSY} and F_{20} reference points are not precautionary with regards to the stock's sustainability when one starts accounting for uncertainties around recruitment. Indeed, although the mean spawning stock biomass of pink ling at MSY is estimated at 25% of its virgin biomass, which is above the 20% limit reference point, there is more than 10% chance of the stock falling below its limit reference point given the range of uncertainty on recruitment.

Socio-economic viability

Figure 5.4 shows the values taken by three economic indicators used to assess the socio-economic viability of the fishery over the range of management strategies in the operating domain. Values of these indicators that are robust to inter-annual variability as well as recruitment uncertainties (second column of Figure 5.4) are used to assess the viability of the fishery. Precisely, the guaranteed NER (Figure 5.4 A.2) and net crew wages (Figure 5.4 B.2) (resp. Fisher Price Index (Figure 5.4 C)) with a probability of 90% correspond to the 10th (resp. 90th) quantile of the 10-year minimum (resp. maximum) value. Positive guaranteed NER and net wages respectively indicate that the fishery is able to remunerate its physical and human capital. As shown by

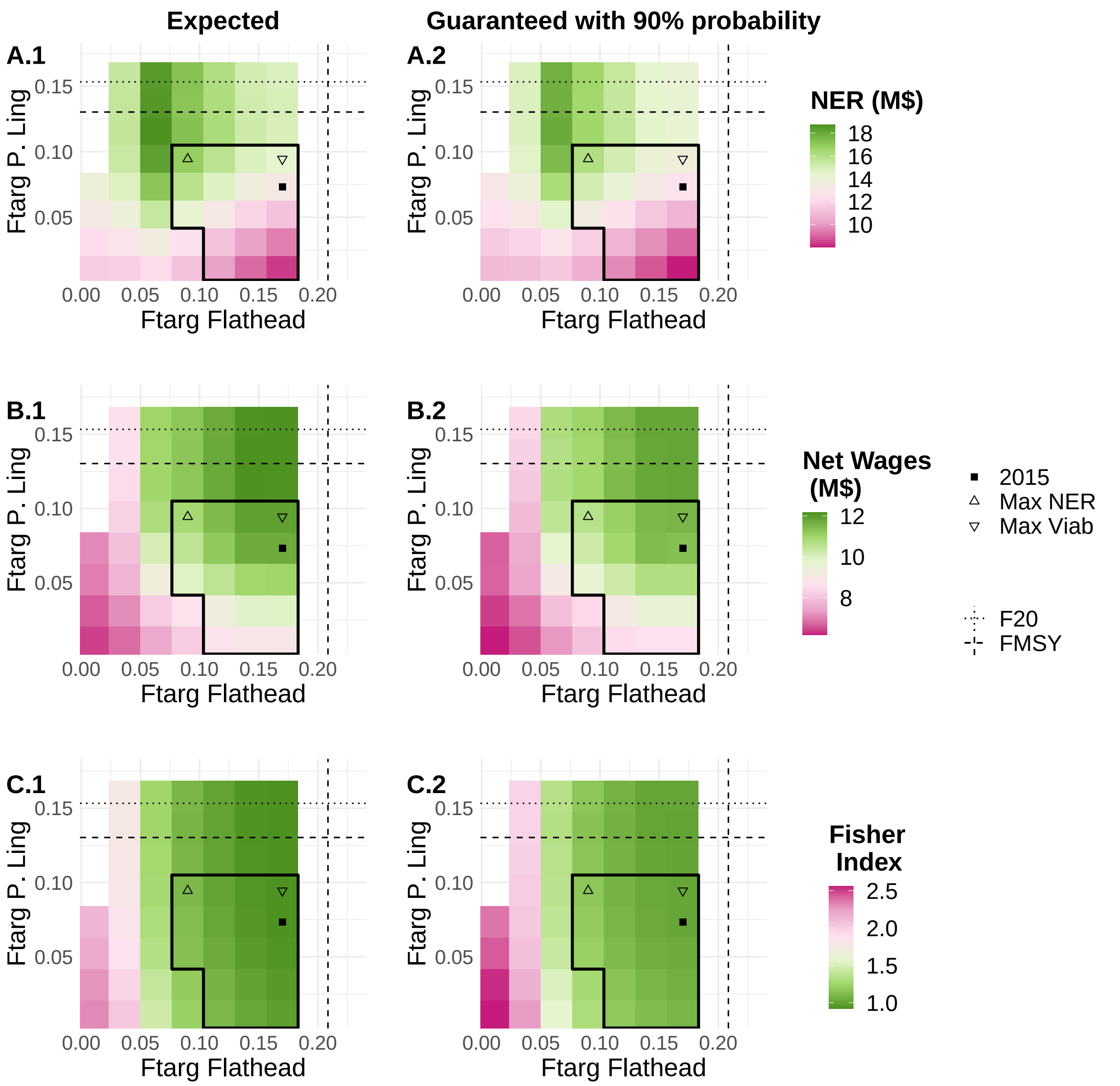

Figure 5.4 – Socio-economic indicators within the operating domain.
First column displays expected values while second column displays values that are guaranteed with a 90% probability.
A: Net Economic Returns (NER); B: Net wages; C: Fisher Price Index.
The thick black bounding delineates the eco-viable space, defined as the space ensuring biological and economic viability of the fishery and price acceptability. Dashed lines show stock-specific F_{MSY} and F_{20} reference points. Specific scenarios within the eco-viable space are also identified: the 2015 fishing mortality rates (black square),the one maximizing the industry's net economic returns (Max NER), and the one maximizing the viable fleet size (Max Viab - see Figure 5.5) but which also maximizes net crew wages and minimizes the price of fish.
Source: output from IAM model

Figures 5.4 A.2 and B.2, the economic viability of the fishery is guaranteed with a

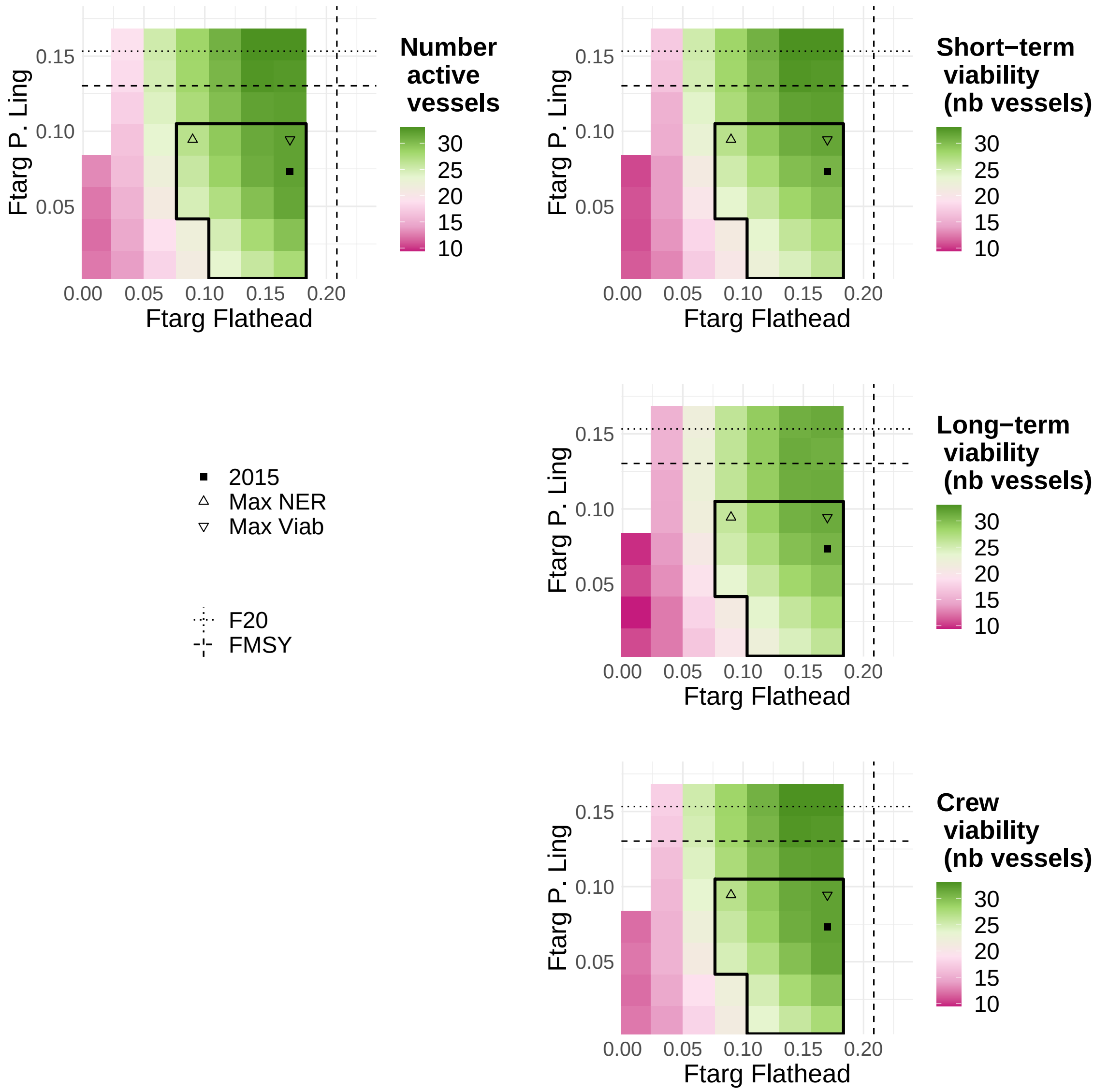

Figure 5.5 – Fleet economic viability within the operating domain.
The plots display the number of viable vessels with a probability of 90% - A: short-term financial viability - B: long-term financial viability - C: crew viability.
The thick black bounding delineates the eco-viable space, defined as the space ensuring biological and economic viability of the fishery and price acceptability. Specific scenarios within the eco-viable space are also identified: the 2015 fishing mortality rates (black square),the one maximizing the industry's net economic returns (Max NER), and the one maximizing the viable fleet size (Max Viab) but which also maximizes net crew wages and minimizes the price of fish (see Figure 5.4).
Source: output from IAM model

probability of 90% throughout the operating domain as both indicators, namely guaranteed NER and net wages, are positive. When interested in maintaining the consumer's demand for the fishery's products, one can look at the maximum price index experienced throughout the simulation period, shown on Figure 5.4 C.2. As expected, it decreases as harvest rates increase, as a result of higher TACs and thus landings. A guaranteed Fisher price index below the acceptable threshold value of 1.20 indicates that consumers' demand is ensured with a probability of 90% throughout the projection period. Unlike the fishery's economic viability, constraints pertaining to its biological (Figure 5.3) and consumption viability restrict the operating domain to the eco-viable space identified by the thick black lining on Figure 5.4. Within the eco-viable space, the biological and consumption viability of the fishery are guaranteed with a probability of 90%.

Economic viability is also assessed at the vessel level in Figure 5.5. Higher harvest rates, and thus TACs, in the operating domain allow more vessels to be active in the fishery (Figure 5.5 A). As detailed in Appendix A, individual harvesters who decide to lease in quota are those that expect positive NOS given the market lease values of quotas. However, as their expectations are based on the previous year's catch rates and fish prices, economic results may differ from the expected value. This explains why some active vessels do not meet the short- or long-term profitability constraint (Figures 5.5 B and C in comparison to Figure 5.5 A). Nonetheless, the number of viable vessels with regards to each constraint is correlated to the number of active vessels and the number of vessels simultaneously meeting the three viability constraints is maximal at the upper right corner of the eco-viable space (Max Viab scenario).

5.4.3 Trade-offs within the eco-viable space

The eco-viability framework allows to identify a safe operating space for the fishery. When the latter is not reduced to a single point, it does not completely solve the manager's problem, who is left with a set of options. In this context, knowledge of potentially conflicting aspirations within viable options can guide management decision. Several trade-offs pertaining to the distribution of benefits between vessel owners, crew members and consumers can be considered. Whereas the surplus to vessel owners (i.e. the fishery's expected NER) is maximized at the upper left corner of the eco-viable space (Max NER scenario - Figure 5.4 A.1), that of consumers (proxied by the inverse of the expected Fisher price Index) and crews (i.e. expected net crew wages) are maximized at its upper right corner (Max Viab scenario - Figures 5.4 B.1 and C.1). As already highlighted, this is also where the number of economically viable vessels is maximized (Figure 5.5).

These trade-offs are also represented on the radar plot in Figure 5.6 for the set

Figure 5.6 – Socio-economic trade-offs within the eco-viable space.
Variables have been scaled to the maximal value within the eco-viable space. The low price index refers to the inverse of the expected Fisher price index shown in Figure 5.4 C.1 and is used as a proxy for the surplus of consumers, NER is the expected NER as shown in Figure 5.4 A.1, Net Wages are the expected net crew wages as shown in Figure 5.4 B.1 and the viable fleet size is the number of vessels simultaneously meeting the three viability constraints and is deduced from Figure5.5.
Source: output from IAM model

of scenarios mentioned above. On this figure, the variable "Viable fleet size" refers to the number of vessels simultaneously meeting the three fleet viability constraints (with a probability of 90%), and all variables have been normalized relative to their maximum value within the eco-viable space. The inverse of the expected Fisher price index (referred to as "Low price index") was used as a proxy for the consumer surplus. Maximizing the fishery's profits to an expected NER of AU$16.9M is expected to increase the cost of fish by 11% compared to 2015, generate AU$10.9M of wages and sustain an economically viable fleet of 26 vessels. Maximizing consumer surplus, crew surplus and fleet size would decrease the cost of fish by 7% relative to 2015, generate AU$11.9M of net wages, allow a viable fleet of 31 vessels but generate profits of AU$14.4M, that is 86% of its maximal value.

5.5 Discussion

5.5.1 Definition of sustainability thresholds

The identification of thresholds separating safe from risky evolutions of the system is a critical, and certainly not trivial, step in the eco-viability approach. Regarding a stock's biological viability for instance, there is no real consensus emerging from the scientific literature on how to define reference points for recruitment overfishing. Haddon et al. (2012) traced the emergence of B_{20} as a limit reference point for Australian federal fisheries back to Beddington and Cook (1983), but mostly the report from Restrepo et al. (1998). They note that despite the lack of empirical basis for this value, actually being a proxy for the $0.5B_{MSY}$ reference point, it has been adopted for the management of Australian federal fisheries. This limit reference point is therefore more likely to represent a value people have agreed on at some point in time, rather than a viability threshold *stricto sensu*. Some good practices identified by Sainsbury (2008) could be considered to refine the biological thresholds used in viability analyses of the fishery.

Also linked to biological reference points, we noted that F_{msy} is not always a precautionary target in the sense specified by the Harvest Strategy Policy. Indeed, it can drive a stock below its limit reference point B_{20} more than 1 year out of 10 if uncertainties around recruitment are accounted for. This is something we observed for the eastern stock of pink ling in the SESSF and it is not the unique observation of the kind as it has also been reported for hoki in New Zealand, which contributed to B_{MSY} not being chosen as a management target for this stock (Punt et al., 2014).

The work presented here is, to our knowledge, the first attempt to incorporate a constraint pertaining to the prices to consumers in a fishery's eco-viability analysis. The two facets of the constraint, namely the indicator chosen and its threshold value, can be discussed. Regarding the indicator, we used the fishery's Fisher price index as a proxy for consumer surplus. The fishery's price index only partially reflects consumer surplus, which can be compounded by multiple drivers of changes in the availability and price of substitute and complementary products external to the fishery (e.g. imported fish (Ruello, 2011) or Australian farmed salmon). However, such external effects were out of scope of the present work which aimed at assessing the impact on consumers on management decisions made for the fishery, thus legitimating the use of a fishery-focused proxy. As the maximal observed value of the fishery's Fisher price index over the past 15 years, the considered threshold on price allows identifying TAC decisions that maintain consumer surplus within recent baselines. However, the present value shall not be interpreted as an acceptable threshold of loss of consumer surplus, whose definition is ultimately a political decision that goes beyond the task of the scientist.

5.5.2 From sustainability to trade-offs

A large scale European project working on the definition of fisheries management objectives encompassing the multiple dimensions of sustainability concluded that management objectives either take the form of constraints (referred to as "sustainability objectives") or quantities to maximize (referred to as "maximization objectives") Rindorf et al. (2017b). Importantly, the latter authors found a broad agreement among stakeholder groups that maximization should only occur within the "sustainable area", hereby highlighting the priority given to sustainability against the maximization of any specific objective. The two-step approach that we used here also prioritized sustainability constraints and could be used to operationalize prioritization of objectives. The eco-viability approach allows the identification of a safe control space for the system, respecting minimum standards with regards to the several pillars of sustainability (the "sustainable area" from Rindorf et al. (2017b)). In this particular application, we show that maximizing the fishery's profits is not en eco-viable trajectory as it breaches both the biological constraint on pink ling and the threshold imposed on the price of fish.

Our second step was to identify the options for maximizing particular quantities and present the trade-offs associated with each objective. Not only is the knowledge of potential trade-offs essential for being explicit with management decisions, but is also useful to the wider society as it provides transparency to the compromises underlying decision. In particular, we highlight a conflict between the surplus of vessel owners and that of consumers. Our results echoes the work carried out by Pascoe et al. (2018a) in the SESSF who showed that accounting for consumer surplus in the estimation of MEY resulted in a net transfer of benefits from producers to consumers.

We also highlight a divergence of interests between vessel owners and crew members. The extent to which both differ depends on the remuneration system of fishing crews. As noted by Guillen et al. (2017), shared remuneration systems (also called lay systems) have been commonly adopted in fisheries worldwide to make crews capture part of the rent from fishing, hence bringing their incentives in closer alignment to that of vessels owners. Such systems nevertheless vary in what is shared between vessel owners and crews. In some cases, crews get a share of the revenue and costs are born by vessel owners, which is the usual case in the SESSF. There are also situations where operating costs such as fuel, bait, ice or the lease of quota are jointly born by crews and vessel owners. Intuitively, the more costs are shared, the more the surplus of crews aligns with that of vessel owners. This is indeed what we see when simulating a remuneration of crew indexed on the income from fishing minus variable costs (Appendix F). It is nonetheless important to point out that both scenarios assume a constant rate for crew shares, which does not capture adjustments that can be made by vessel owners to align their remuneration to the labour market.

5.5.3 Market dynamics in multispecies fisheries: what is the added-value for management advice?

Using the same simulation approach as followed here, Briton et al. (in prep.b) determined achievable catch compositions for several species pairs in the SESSF and noted significant room to manoeuvre in the proportions these species were caught. For the purpose of the present work, IAM was augmented with the endogenous representation of fish market dynamics in the SESSF. Not only does it allow consideration of the surplus to consumers when evaluating management targets, but it also refined our estimation of the operating domain as presented in Section 5.4.1. Specifically, we show that typical market dynamics, with prices decreasing with supply, lessen the economic incentive to increase the landings of under-caught species, hence narrowing down the operating domain obtained under the assumption of constant prices. In addition to driving individual fishing strategies, market dynamics also determine overall economic outcomes of the fishery. In particular, they exacerbate the divergence of interests between producers and consumers, as an increase in price favours the former at the expense of the latter. These aspects being of notable interest to decision-makers, we emphasize the value of adequately capturing market dynamics in the models used to provide management advice. We would also like to highlight that these developments rely on regularly updated market dynamics, which is currently lacking for this fishery as their last estimation date back to Bose (2004).

5.6 Conclusion

Embracing the full spectrum of sustainability in the management of fisheries requires a systematic accounting of its economic and social dimensions in what has mostly been a biologically-focused process. We showed how the eco-viability approach can be used to identify TAC decisions in mixed fisheries which meet a multitude of biological, economic and social sustainability constraints. We also highlight specific trade-offs within sustainable options. In particular, we show that maximizing the fishery's profits, which figures as the management target in the Southern and Eastern Scalefish and

Shark Fishery, comes at a cost for crew members and consumers. Including the surplus of crews and consumers in the estimation of Maximum Economic Yield would therefore lead to different Total Allowable Catch decisions.

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CHAPTER 6

GENERAL DISCUSSION

The thesis worked at advancing the development of integrated approaches to support tactical decision-making in mixed fisheries under output-control. Such integrated advice typically involves accounting for the interactions structuring the dynamics of mixed fisheries, but also the various facets of their sustainability. To this purpose, the thesis developed an approach involving eco-viability analyses to evaluate TAC-setting options for mixed fisheries. The methodology relied on an integrated ecological-economic model, IAM, and a set of acceptability constraints for the system represented. The developed methodology was applied to two multi-species fisheries under TAC management: the French demersal fishery in the Bay of Biscay (BoB) and the Australian Southern and Eastern Scalefish and Shark Fishery (SESSF).

The first section of this concluding chapter summarizes the key results derived from each Chapter of the thesis. The two following sections are dedicated at presenting the methodological contributions of the thesis and discussing their implications for management and policy. The first concerns the representation of mixed fisheries dynamics in models used to support tactical decision-making and the second relates to the integration of multiple dimensions in the advice provided for tactical management decisions. These two sections also allow comparing how both aspects have been treated in the two case-studies. Perspectives for future work are finally proposed.

6.1 Key results of the thesis

The second chapter of the thesis presents a first application of the developed methodology to the BoB, with a particular focus on TAC-setting decisions for two key species of the fishery, common sole and European hake. The developed methodology allowed identifying the set of achievable harvest rates, referred to as the operating domain, given technical interactions between the two stocks. Specifically, we note that the operating domain of the fishery intersects with MSY ranges of both stocks and that MSY reference points are achievable targets. The biological viability assessment shows that only a subset of the operating domain ensures the biological sustainability of the four dynamically modelled stocks (common sole, European hake, Norway lobster and European seabass). Assessment of economic viability at the fleet level highlights overcapacity in the fishery as no management option allows all fleets to simultaneously meet the three economic viability constraints, namely the ability to cover operating costs, depreciation of fishing capital and opportunity cost of labour. We specifically highlight a trade-off between the remuneration of fishing crews and that of capital owners. Harvest rates above the upper bound of MSY ranges allow more fleets to cover the depreciation of capital at a 10-year horizon, but hinder their ability to retain crews. Yet, with TACs set around MSY for hake and in the upper range of sole's MSY range, we expect an increase in the number of economically viable fleets compared to 2016, the starting year of projections.

The third chapter is specifically dedicated to ITQ markets in mixed fisheries. The first part provides a theoretical analysis of perfectly competitive quota lease markets at equilibrium in a context of joint production. Linear programming and duality theory are used to derive quota lease prices at equilibrium. Lease prices for species the TAC of which is binding are shadow prices capturing marginal profit associated to the joint catch and vessels' quasi-rent. In a second part, an iterative algorithm mimicking Walrassian tâtonnement process is used to model the convergence towards lease market equilibrium. Simulations drawn under two scenarios of fishing effort allocation (status-quo and profit-driven) shed light on economic incentives provided by ITQ markets, namely effort shift towards species the TAC of which is not constraining but also towards species not under quota.

In Chapter 4, simulations involving different types of joint production in the SESSF highlight how assumptions of exogenous fishing behaviour, which typically underlies mixed fisheries TAC advice in Europe, underestimate the space of achievable catch compositions in a mixed fishery under ITQ, and even more so when species can be individually targeted. The simulations also show a hierarchy of species in the fishery, with socio-economic outcomes being mostly driven by key economic stocks. This result suggests that a hierarchical approach to TAC-setting could be envisioned in fisheries where a few stocks represent most of the produced value. Harvest targets incorporating socio-economic objectives could be defined for key economic stocks, while managing by-products and by-catch with reference points reflecting biological or ecological objectives.

The fifth chapter echoes the second in its application of a similar integrated assessment of TAC-setting options to the SESSF. Building on methodological developments carried out in Chapters 3 and 4, allocation of fishing effort, along with the response of fish prices to landings, were endogenized in the model. This allowed accounting for the surplus of consumers, alongside that of capital owners and crew members, in the evaluation of management options. We notably point to trade-offs between the interests of capital owners on one side, and crew members and consumers on the other side. This has important policy implications for a fishery whose management must aim at "maximizing economic returns to the Australian community", but where TACs are still set so as to maximize those to capital owners. Typically, accounting for the surplus of consumers and crew members in the calculation of Maximum Economic Yield (MEY) would lead to setting higher TACs than those set to maximize the surplus of capital owners.

6.2 The representation of mixed fisheries dynamics in the models used to support tactical decision-making in mixed fisheries

The thesis has contributed to improving the representation of mixed fisheries dynamics in the models used to provide TAC advice for fisheries. The same simulation framework, namely the integrated ecological-economic model IAM, has been used to represent the French demersal fishery in the Bay of Biscay and the Australian Southern and Eastern Scalefish and Shark Fishery. The model has increased in complexity throughout the thesis (Table 6.1) and conclusions drawn from developments undertaken are deemed of general interest to scholars interested in the sustainable management of a renewable resource, as well as to institutions in charge of advising TAC decisions in mixed fisheries.

6.2.1 Modelling the biological dynamics of mixed fisheries

The first difference between the two applications lies in the representation of the fisheries' biological components in the model (Table 6.1). In the BoB model, the stocks

Table 6.1 – Comparison of the models for the two studied fisheries: the French demersal fishery in the Bay of Biscay (BoB) and the Australian Southern and Eastern Scalefish and Shark Fishery (SESSF)

Model component		**Case study**	
		BoB	**SESSF**
Biology	**Stock dynamics**	**Age-based – Annual**: 3 stocks (common sole, Norway lobster, European seabass) **Age-based – Quarterly** : 1 stock (hake northern stock)	**Age-based – Annual**: 4 stocks (eastern school whiting, jackass morwong eastern and western stocks, silver warehou) **Age- and sex-based – Annual**: 6 stocks (flathead, blue grenadier, pink ling eastern and western stocks, orange roughy eastern stock, redfish) **Surplus production – Annual** :6 stocks (blue-eye trevalla, gummy shark, school shark, john dory, mirror dory, ocean perch)
	% fishery's GVP dynamically modelled	40%	80%
Fishing activity	**Number vessels**	710 vessels in 11 fleets	110 vessels in 9 fleets
	Number metiers	13 metiers	36 metiers
	Allocation fishing effort	Exogenous	Endogenous with expected profits and habit as drivers
Quota allocation		Exogenous	Endogenous through ITQ markets
Fish prices		Exogenous	Endogenous through own- and cross-species price-quantity flexibilities

the dynamics of which were explicitly represented (hereafter referred to as dynamically modelled stocks, as opposed to statically modelled stocks for which the catch per unit of effort (CPUE) is assumed constant throughout the simulation period) were either represented using an annual age-based model or a quarterly age-based model. In total, four key stocks for the fishery were dynamically modelled, accounting for 40% of the fishery's gross value of production (GVP). In the SESSF application, 16 stocks were dynamically modelled using either annual age-based, annual age- and sex-based or global surplus production models, accounting for 80% of the fishery's GVP. It is important to try to limit the static representation of stocks in bio-economic models as stock effects are simply not factored in, which leads to excessively optimistic or pessimistic anticipations of economic impacts of variations in TACs. The biological calibration of the model was mostly constrained by the availability of official stock assessments in both regions. In the Bay of Biscay, five demersal stocks had been quantitatively assessed by the EU Commission's advisory body, the International Council for the Exploration of the Seas (ICES) at the time of the study (ICES, 2017a). Among them, four were used to calibrate IAM (common sole, Norway lobster, European seabass and northern hake), leaving megrim as a static stock given its limited economic contribution for the modelled vessels and the absence of concerns regarding the status of the stock (ICES, 2016b). The main limitation of this application was the absence of population dynamics for anglerfish, an important stock for the fishery, and this despite (unsuccessful) attempts to fit a global surplus production model to the data. In the SESSF Commonwealth Trawl Sector (CTS) and Gillnet Hook and Trap Sector (GHTS), (sex- and) age-based population models derived from SS3 stock assessments were used to calibrate population dynamics for 10 stocks. These models were complemented by surplus production models for another 6 stocks, for which there was either no quantitative stock assessment or one that did not relate to the models already implemented in IAM [1]. Although not as reliable to assess stock status as their age- or length-based counterparts, surplus production models have the advantage of being less data-demanding, and therefore provide an interesting option to model data-limited stocks in bio-economic models. In the SESSF application, surplus production models enabled the dynamic representation of stocks accounting for 25% of the fishery's GVP, which would have otherwise been modelled as static stocks. Another option to model the catch of species that cannot be dynamically represented could be to scale the catch or income of static species on that of the dynamically modelled ones as in Ulrich et al. (2002); Gourguet et al. (2013); Guillen et al. (2013) or estimate non-linear responses of CPUEs to fishing effort as in Pascoe et al. (2015).

1. For instance, school shark assessment is based on a close kin mark recapture (CKMR) model that provides an estimate of absolute abundance without modelling the dynamics of the stocks and the impact of fishing (Thomson et al., 2020). Another example is gummy shark that is assessed using a quantitative stock assessment model but relying on different equations that those already coded into IAM (Pribac et al., 2005; Punt et al., 2016b)

6.2.2 Modelling fishing activity

The two applications also differed in their representation of fishing activity, both in terms of the scale at which it is represented and the modelling of underlying dynamics. They both involved vessel-based models, which is not common among decision-support models that generally work at the fleet level. Representing individual vessels allowed taking account of intra-fleet heterogeneity in fishing abilities. Agent-based models also make the representation of other facets of behavioural diversity possible, such as diversity in motivations, livelihoods or social interactions (Wijermans et al., 2020).

In both applications, fishing activity was modelled as the allocation of effort among metiers in which vessels can operate, depending on the fleet they belong to. As shown in Table 6.1, more metiers (relative to the number of fleets) were represented in the SESSF than in the BoB model. Whereas metiers in the BoB were an aggregation of EU's Data Collection Framework (DCF) typology (level 5) reflecting the main fishing strategies in the area (Macher et al., 2015), those in the SESSF were specifically defined in the thesis through a multi-variate clustering of catch data at the haul level. Along with inputs from industry members, this quantitative analysis allowed the identification of existing targeting strategies in the fishery, from the more to less frequent ones. The resolution at which metiers were defined in the SESSF allowed exploring the question of vessel-level flexibility in joint production across the fishery. Carrying out similar analyses in the BoB would likely involve refining the scale at which metiers are defined in the model in order to effectively match that at which fishing decisions are being made. In this regard, recent work from Mateo et al. (2017) involving a multivariate clustering analysis of catch data in the Celtic Sea shows that DCF's metier typology, based on gear type and dominant species in the catch, fails at capturing the clear spatial structure of the catch composition in the area.

Identification of achievable catch compositions in mixed fisheries being pivotal to their regulation by output controls, it was also deemed necessary to propose a model accounting for mechanisms by which fishers can alter the proportions in which they catch different species. Particularly relevant to the regulator charged to set TACs in a mixed fishery are information about targeting, avoidance or discarding behaviours, in order to prevent major "choke" or discard situations (Ulrich et al., 2011). These topics were addressed in the thesis by modelling how individual harvesters allocate their fishing effort among multiple metiers. Building on reviews of the drivers of fishing behaviour (van Putten et al., 2012; Girardin et al., 2017), effort allocation was modelled as a weighted average between so-called habit-based and profit-driven allocations.

Whereas there is nothing novel in the modelling approach, its potential for inclusions in existing advisory frameworks is worth highlighting. This model has the notable advantage of representing allocation of effort at the metier- and annual-level, which is the resolution at which catch and effort data is usually available, and thus that at which models used to support TAC advice are developed (e.g. FCube in the European context (ICES, 2016c)). The thesis focused on the simulation of two extreme scenarios of fishing behaviour (entirely habit-based or profit-driven) that provide bounds between which the fishery is likely to operate.

6.2.3 Modelling ITQ markets in multi-species fisheries

The modelling of ITQ markets in multi-species fisheries remains a largely under-explored topic despite their implementation in a number of multi-species fisheries worldwide (Sanchirico et al., 2006). A few models can be found in the literature (e.g. Newell et al. (2005); Little et al. (2009); Bailey et al. (2019)), yet none of them has explicitly represented the mechanisms underlying the formation of quota lease or sale prices. The thesis presents an iterative algorithm mimicking Walrassian tâtonement process as the mechanism determining general equilibrium lease prices for quotas on jointly harvested species. This model was developed in Chapter 3 and implemented into IAM for SESSF simulations in Chapters 4 and 5. The model captures core properties of ITQ markets, such as the allocation of quotas to the most efficient vessels, but also some specifically emerging in multi-species fisheries, such as the economic incentive to redirect fishing effort towards more catch of species the TAC of which is not binding, or of species not under quota regulation.

As illustrated in Chapters 4 and 5, the algorithm can be embedded in broader simulation frameworks of multi-species fisheries. Nonetheless, the computing time of the iterative approach proposed can reveal costly for its deployment within regular advisory procedures. Moreover, the quality of convergence of this model being correlated to the number, as well as diversity (in terms of catch and costs) of market participants, models working with fleets as operating units might not support such process-based modelling approach.

6.2.4 Modelling fish price dynamics

In Chapter 5, simulations include own- and cross-species ex-vessel price flexibilities, to enable representation of market responses to changes in the composition of landings, accounting for possible substitution or complementation effects among species. Accounting for such market dynamics allows broadening the scope of economic assessment, still mostly centred on the fishing industry (Hornborg et al., 2019), by considering the impact of TAC decisions on consumers aside from that on producers. It can also highlight inconsistencies between policy objectives (e.g. MEY in Australian federal fisheries) and those pursued by management in practice (e.g. maximizing the returns to capital owners).

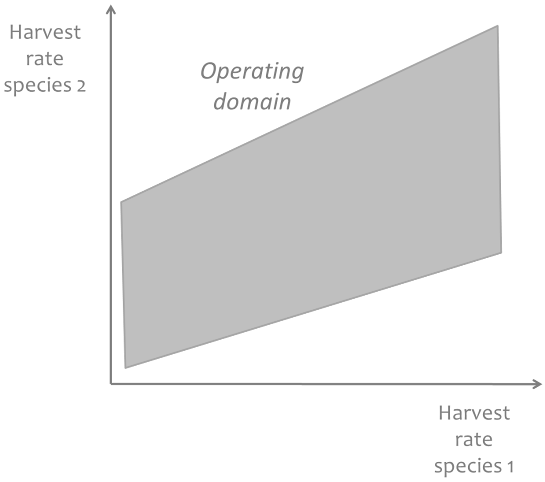
Dynamics in fish prices also directly impact the income from fishing and are therefore important drivers of harvesting strategies. In a mixed fishery, they will typically influence targeting decisions as shown in Chapter 5. Indeed, with ex-vessel prices decreasing with the amount of fish landed, there is less incentive for fishers to land more fish. This for instance diminishes the economic incentive provided by ITQ markets to redirect fishing effort towards more catch of non-binding species. Therefore, in an attempt to determine plausible multi-output production sets in a mixed fishery, it appears important to also account for dynamics in its output markets.

6.3 Integrating biological, economic and social considerations in the advisory process

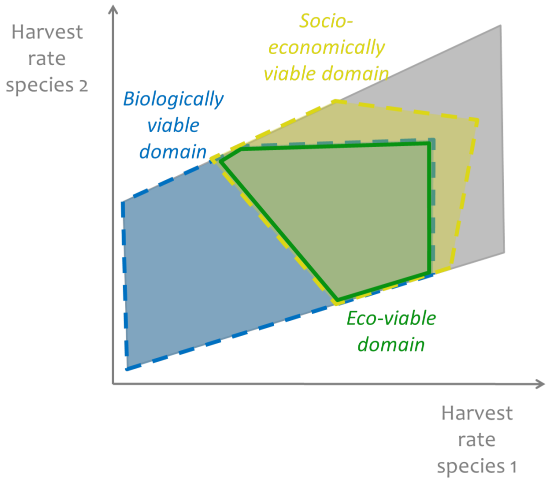
The thesis proposed an approach to integrate biological, economic and social considerations when advising TAC decisions in mixed fisheries. This step-wise approach, refined through the course of the thesis, allows the visualization of different domains and their interactions as illustrated in Figure 6.1. First, the operating domain delineates achievable harvest rates given technical interactions among jointly captured stocks (Figure 6.1A). Among technically feasible options, those ensuring biological sustainability define the biologically viable domain. Socio-economic viability at the fishery- (SESSF), fleet- (BoB), or vessel-level (SESSF) is assessed in a third phase and the intersection between biologically and socio-economically viable domains defines the eco-viable space (Figure 6.1B). Finally, the evaluation of management options across multiple dimensions allows the identification of trade-offs that are yet present among eco-viable options, or that prevent emergence of the latter (Figure 6.1C).

6.3.1 Assessing eco-viability

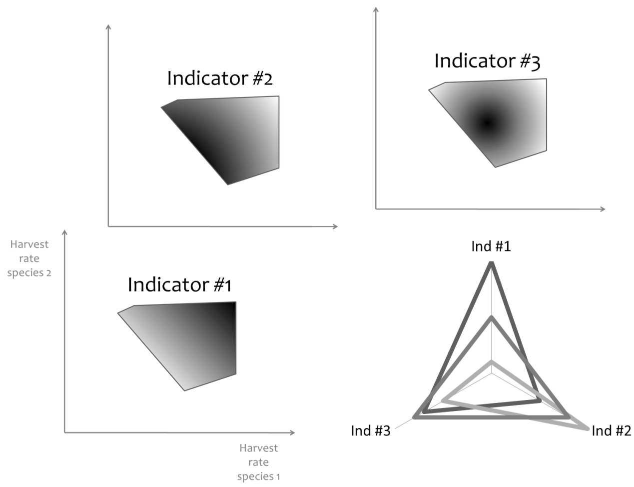
A common eco-viability framework accounting for biological, economic and social constraints was developed and applied in the two case studies, with yet some variants



(A) Identification of operating domain



(B) Viability assessment



(C) Evaluation of trade-offs

Figure 6.1 – Developed methodology for an integrated TAC advice for mixed fisheries

(Table 6.2). In both applications, biological sustainability constraints aimed at preserving the stocks' reproducing capacity and took the form of lower bounds on (spawning) stock biomasses. In the BoB application, these thresholds were the limit biomass reference points determined by ICES experts and in the SESSF the policy's limit biomass reference point B20 (namely 20% of the virgin (spawning) stock biomass) was used. Overall, biological viability assessment has relied on indicators and reference points that have often been considered in eco-viability studies of fisheries (Oubraham and Zaccour, 2018).

The thesis has however contributed to refining economic and social dimensions of eco-viability frameworks. Whereas most eco-viability applications to fisheries have interpreted economic viability as maintaining positive fishery profits (Oubraham and Zaccour, 2018), economic viability in the thesis has been decomposed into three constraints pertaining to the remuneration of both physical and human forms of capital. The remuneration of physical capital engaged in the fishery was addressed at two time scales. First, a short-term constraint on the operators' gross profits (annual for the SESSF and bi-annual for the BoB) assessed the ability of producers to cover their annual operating costs (fixed and variable). Second, a long-term constraint on the operators' net profits assessed their ability to cover capital costs in addition to operating costs. The remuneration of human capital is also critical to the fisheries' economic viability, which was addressed by an annual constraint on the Full Time Equivalent (FTE) wage of fishing crews. In the BoB, the minimum wage threshold was defined as the average wage of a French seaman for the year of calibration. In the SESSF, closer attention was given to estimating the opportunity wage of Australian fishing crews. To my knowledge, the work carried out in this thesis has been the only one, with that of Maynou (2019), accounting for the opportunity cost of labour in a fishery's eco-viability assessment.

The SESSF application also considered an additional social constraint limiting increases on prices paid by fish buyers due to changes in landings. Constraints pertaining to maintaining consumer surplus have, to my knowledge again, never been considered in eco-viability frameworks. The consideration of this constraint was motivated by the fact that management in the SESSF aims at maximizing profits of the fishing industry despite the fishery's production being primarily destined to the Australian market. This constraint took the form of an upper bound on the Fisher price index of the fishery. The value of the threshold was fixed at the maximal observed value over the past ten years, hereby assuming that past situations were deemed acceptable. Further refinements to this first approach could for instance aim at estimating price thresholds that would ensure that products of the fishery remain affordable for specific social groups.

Table 6.2 – Comparison of viability frameworks for the two studied fisheries: the French demersal fishery in the Bay of Biscay (BoB) and the Australian Southern and Eastern Scalefish and Shark Fishery (SESSF)

Type of constraint	Objective	Time frame	Case study	
			BoB	**SESSF**
Biological	Preservation of stocks' reproducing capacity	Annual	$SSB > B_{lim}$	$SSB > B_{20}$ for (sex-) and age based dynamics $B > B_{20}$ for surplus production dynamics
Economic	Remuneration physical capital	Annual	Gross profits > 0	
		Decadal	Net profits > 0	
	Remuneration human capital	Annual	FTE Wage > mean wage of French crew	FTE Wage > opportunity wage of Australian crew
Social	Maintain consumer demand	Annual		Fisher price index < 10-year maximum value

6.3.2 Highlighting trade-offs

Trade-offs between conservation and socio-economic objectives

Chapters 2 and 5 shed light on conflicts between conservation and socio-economic targets in both contexts. For instance, both highlighted the biological risk of opting for strategies maximizing the fishery's profits. Maximizing the net present value of the hake and sole fishery in the Bay of Biscay leads to both stocks being harvested above F_{MSY}, with harvest rates increasing with the value of the discount factor. This is a direct consequence of including short-term economic returns in dynamic versions of MEY [2], making the latter less conservative than its value at equilibrium (Clark, 1973; Grafton et al., 2007; Clark et al., 2010; Grafton et al., 2010). In this case, economic returns at equilibrium are maximized within MSY ranges for both species. With this issue of

2. Dynamic MEY aims at maximizing economic returns along a trajectory, rather than at a specific point in time, often its value at equilibrium.

inter-temporal compensation in mind, both the average and minimal Net Economic Returns (NER) over the 10-year projection period were given attention in the SESSF application (Chapter 5). It turns out that both are maximized at harvest rates that hinder the biological sustainability of the pink ling stock. Such an outcome is actually not unusual in a multi-species context, where the fishery's economic optimum can lead to the over-exploitation of some stocks in the species mix (Pascoe et al., 2015; Tromeur and Doyen, 2018).

Trade-offs related to the distribution of benefits: inter- but also intra-generational justice

Matters of inter- and intra-generational equity in the distribution of benefits were also given attention in the thesis. As highlighted by Martinet and Doyen (2007) or Doyen et al. (2017), inter-generational justice is inherent to eco-viability since all generations are imposed the same set of constraints. This ensures that there is no sacrified generation with regards to viability constraints. Yet, inter-annual variability in the flow of benefits remains and some generations are better off than others. In this regard, guaranteed outcomes (i.e. the minimal value of an indicator over the projection period) reported in Chapter 5 provide information on what can be ensured to the most disadvantaged generation. In SESSF simulations, both guaranteed and 10-year average economic indicators display similar patterns. This means that there is no trade-off between the fishery's average performance and that of the most disadvantaged generation.

The thesis also highlights several trade-offs pertaining to intra-generational equity in the distribution of benefits derived from common goods. First, conflicts between the interests of vessel owners and crew members were identified in both fisheries. In the BoB, the ability of fishing segments to ensure the long-term renewal of their physical capital conflicts with their ability to retain crews. As a consequence, the number of segments able to remunerate both their physical and human capital is maximized under lower harvest rates than those maximizing the number of segments only able to remunerate their physical capital. These results highlight the importance of accounting for the remuneration of crews in economic viability assessments as it can come to clash with other economic constraints for the fleets. One could however discuss the assumption of a constant crew share throughout the simulation period as there has been evidence of vessel owners adapting crew shares in order to maintain crews in difficult times (Guillen et al., 2017). Considering such adaptive share rates in the model could thus help reconcile the different facets of the fleets' economic viability. In the SESSF, the distribution of benefits between capital owners and crew members does not impede the vessels' economic viability. This is perhaps explained by the fact that the fishery harvests the two considered stocks below or at their respective MSY, and therefore does not reach the tipping point of labour productivity identified in the BoB around MSY for hake and slightly above MSY for sole. However, distributional concerns emerge at the scale of the fishery as maximizing the surplus of capital owners (i.e. fishery's NER) is associated to a loss in that of fishing crews (i.e. fishery's net wages), and vice-versa. This is the joint result of the fishery's NER being maximized under lower harvest rates than those maximizing the fishery's income (a classic bio-economic result due to the linear increase of fishing costs with effort) and crew remuneration being indexed on fishing revenues.

Second, the inclusion of fish market dynamics in the SESSF application has revealed another distributional issue, namely that between producers and consumers. Indeed, maximizing the fishery's NER comes at a cost for the consumer that exceeds the considered price constraint, hence excluding this management option from the eco-viable space. Among eco-viable options, maximizing the sub-fishery's NER is expected to increase the cost of fish by 11% compared to the reference year, whereas a decrease of 7% in the cost of fish would induce NER to be only 86% of its potential maximum. Using the fishery's price index as a proxy for consumer surplus shed light on this trade-off. Yet, this does not enable consumer surplus to be directly compared to that of capital owners or crew members. A full analysis of such trade-offs would require estimating variations in consumer surpluses associated with variations in fish prices.

6.3.3 Contributions of the approach to advisory procedures in both European and Australian federal contexts

In Europe, the FCube methodology is becoming the new standard of ICES for the provision of mixed fisheries TAC advice to the European Commission. Used in the North Sea for the first time in 2012, it has subsequently been applied to the Celtic Sea and Iberian waters and an application to the Bay of Biscay is currently under development (ICES, 2017b). In its current version, the advice provided by FCube informs on the expected consequences in terms of TAC under- or over-catch of various fishing behaviour scenarios (e.g. stop fishing at the most constraining quota, fish until all quotas are consumed, stop at the quota of a given species, etc. . .). It also identifies the set of fishing mortality rates within MSY ranges that minimizes the mismatch between TACs and expected catches given existing technical interactions, hereby guiding the election of harvest rates within these recently introduced ranges in mixed fisheries multiannual management plans (EU, 2016, 2018b, 2019a). To date, ICES mixed fisheries advice has mostly focused on the mismatch between TACs and expected catches, which would correspond to the "operating domain" step from the approach developed in the thesis (Figure 6.1A). What the latter essentially allows is an assessment of TAC options that goes beyond likely TAC under- or over-catch to include an evaluation of their biological, economic and social outcomes. Furthermore, ICES current advice for mixed fisheries builds on exogenously specified joint productions that do not capture the fleets' potential to reconcile catch opportunities and mitigate choke effects.

In the SESSF, despite an explicit fishery-wide MEY objective, there has not yet been an effort to reconcile TACs for jointly caught species. The first contribution of the present work has therefore been to factor technical interactions in the TAC advice. Second, the eco-viability approach allowed confronting a MEY target to a set of viability constraints spanning multiple dimensions. The current objective of the Harvest Strategy Policy for Australian federal fisheries being "*the ecologically sustainable and profitable use of Australia's Commonwealth commercial fisheries resources (where ecological sustainability takes priority)*" (DAFF, 2018b), the maximization of economic returns must take place within ecologically sustainable bounds. This objective aligns closely with the philosophy of the proposed approach which aims at identifying a safe space for the fishery within which management options can be envisioned. This safe space could be restricted to an ecologically viable space to meet this first policy requirement. Yet, including socio-economic constraints of the type that have been proposed in this work can ensure that society as a whole benefits from the exploitation of fish resources, which is another requirement of the present policy.

6.4 Perspectives

The thesis has broadened the dimensions and processes considered when providing TAC advice for mixed fisheries. Several perspectives can be suggested both for research and uptake by advisory bodies and decision-makers.

6.4.1 Modelling human behaviour under uncertainty

Precautionary management requires taking account of our limited understanding of the system when making decisions for the future. It therefore involves accounting for possible uncertainties on the processes represented in the models used to inform decision-making. In both thesis' applications, modelled uncertainties related to the stocks' recruitment, yet other types of uncertainties could be considered (Holland and Herrera, 2009). These could relate to key economic parameters such as fish or fuel prices, or catch composition, which is particularly relevant in a multi-species context. Not only is it critical for models to account for these uncertainties, but it is also important that they capture the effect that these uncertainties can have on individual decisions. Modelling human behaviour under uncertainty would indeed increase model realism. How catch or market uncertainties shape fishing choices or quota trading practices for instance would be particularly relevant developments to consider in a multi-species fishery's context.

6.4.2 Tackling the curse of dimensionality

Integrated ex-ante assessments carried out in the thesis have illustrated their applicability and usefulness in the restricted context of the joint management of two species. However, multi-species technical interactions and resulting TAC-setting issues usually involve more species, hence questioning the applicability of the approach in a more generic context. The curse of dimensionality associated with increasing the number of catch limits to be reconciled involves two main challenges. The first one relates to computing requirements as the time required to screen all possible TAC combinations is an exponential function of the number of stocks considered. A possible solution to this first issue could be to use optimization algorithms in order to efficiently identify search for solutions satisfying the set of viability constraints. Previous eco-viability studies involving a large number of control variables have for instance used genetic optimization routines to identify scenarios maximizing the probability of eco-viability, i.e. the probability that the system simultaneously meets all viability constraints given uncertainties in the model (Hardy et al., 2013; Cissé et al., 2013, 2015; Gourguet et al., 2013, 2016) . Single-objective (in these case the probability of eco-viability) optimization is however only useful when eco-viable solutions exist. More adapted to the alternative case, multi-objective optimization (also referred to as multi-criteria optimization or Pareto optimization) methods would allow identifying Pareto optima in regards to the multiple viability requirements (Emmerich and Deutz, 2018). The second challenge that emerges relates to the difficulty to process, analyse and convey information involving many dimensions. Again, the strategy could be to focus on the analysis and communication of scenarios performing well overall, or on certain aspects, in order to shed light on particular trade-offs.

6.4.3 Better accounting of ecological dimensions of EBFM

One could legitimately argue that eco-viability studies in the thesis have taken a rather anthropocentric view on the sustainability issue. Aiming at ensuring the viability of a fishery, an intrinsically human-oriented activity, the eco-viability framework proposed here does not really touch on the viability of the ecosystem in which it is embedded. Calling upon the continuance meaning of sustainability, ecological sustainability can be interpreted as the persistence through time of the processes critical to the functioning of an ecosystem. Because of their ability to integrate processes happening at and across various scales, system-level indicators are being given notable attention in the transition towards EBFM (Tam et al., 2017; Fu et al., 2019; McQuatters-Gollop et al., 2019). Moreover, recurrently emerging patterns in these indicators might facilitate the identification of perturbed states, and thus viability thresholds for ecosystems (Link et al., 2015; Libralato et al., 2019).

There are mainly two ways in which such indicators and thresholds could be used in an eco-viability framework. The first option could be to have the ecosystem dynamics underlying these indicators explicitly represented in the model. By increasing model complexity one might however step back in terms of reliability for tactical decision-making. The second option could be to quantitatively relate the value of identified ecosystem indicators to existing variables of the model. Ultimately, calling on the expertise of ecologists and ecosystem modellers will be critical to identify the option that bears the most potential.

6.4.4 Beyond eco-viability

Eco-viability analyses aim at identifying management options that meet a set of sustainability constraints. Yet, when eco-viable options do not exist, reporting on each dimension individually can help diagnose underlying conflicts. Notably, shedding light on conflicts that cannot be resolved with considered management levers can help identify those that may restore a system's viability.

Eco-viability assessment may also be not quite sufficient to comprehensively advise management when eco-viable options exists. Evaluating eco-viable options across multiple dimensions, as was done in the thesis, is one way to help decision-makers elicit among them.

A similar approach to eco-viability analysis could also be envisioned to reconcile aspirations rather than pure viability concerns. It may for instance be viable, but not acceptable, for the fishing industry to generate profits that are just positive. In this regard, one could define two strata of constraints, starting from those related to the system's viability, and increasing standards to what is satisfactory. Because of society's

(A) Eco-satisfactory space

(B) Diverging aspirations and distance from eco-satisfaction

Figure 6.2 – From eco-viability to eco-satisfaction

diversity of aspirations regarding the use of common resources, different satisfactory domains are likely to emerge and their intersection would define the *eco-satisfactory* domain (Figure 6.2A). Even if the latter space is reduced to an empty set, visualizing individual domains would help appreciate the extent to which aspirations are conflicting (Figure 6.2B). The mathematical distance between non-intersecting spaces could for instance be computed to quantify how far aspirations are from being reconcilable.

Appendices

APPENDIX A

DETAILED DESCRIPTION OF THE IAM MODEL

A list of the mentionned model's variables and associated subscripts is given in Table A.1.

Table A.1 – IAM variables and associated subscripts

(a) Subscripts

s	species
mo	morph
a	age
g	gender
t	year
se	season
i	individual harvester
f	fleet
m	metier

Table A.1 – IAM variables and associated subscripts, continued

(b) Variables

Variable	**Signification**	**Unit**
N	Number of individuals	$\emptyset$
N_0	Recruitment	$\emptyset$
B	Biomass	kg
K	Carrying capacity	kg
r	Growth rate	yr^{-1}
w	Individual weight	kg
wL	Individual weight in the landings	kg
Mat	% of mature individuals	$\emptyset$
$MatWt$	Mature weight	kg
SSB	Spawning Stock Biomass	kg
F	Fishing mortality rate	yr^{-1}
FLw	Mortality rate from landings in weight	yr^{-1}
FDw	Mortality rate from discards in weight	yr^{-1}
D	Discard rate	$\emptyset$
δ	Weighting coefficient for the calculation of $\overline{F}$	$\emptyset$
M	Natural mortality rate	yr^{-1}
Z	Total mortality rate ($F + M$)	yr^{-1}
E	Fishing effort	day
q	Catchability	$\text{year}^{-1}.\text{day}^{-1}$
L	Landings	kg
$LPUE$	Landings per unit of effort	kg.day^{-1}
p	Ex-vessel price	€.kg^{-1} or AU\$.kg^{-1}
qp	Lease price of quota	€.kg^{-1} or AU\$.kg^{-1}
GVL	Gross value of landings	€ or AU$
$cshr$	Crew share	$\emptyset$
$rtbs$	Return to be shared	€ or AU$
$Crep$	Repair costs	€ or AU$
$Cfix$	Fixed costs	€ or AU$
$opersc$	Other crew costs	€ or AU$
$Cdep$	Depreciation costs	€ or AU$
$Copport$	Opportunity cost of capital	€ or AU$
$CvarUE$	Variable costs per unit effort	€.day^{-1} or AU\$.day^{-1}
GOS	Gross operating surplus	€ or AU$
NOS	Net operating surplus	€ or AU$

Table A.1 – IAM variables and associated subscripts, continued

Variable	Signification	Unit
FTE	Full-time equivalent number of crew members	$\emptyset$
$Wage$	Wages	€or AU$
$Wage_{FTE}$	Wages per FTE	€or AU$
$NWage$	Net Wages	€or AU$
$\overline{F}_{targ}$	Fishing mortality target for the HCR	yr^{-1}
TAC	Total Allowable Catch	kg
Q	Individual quota	kg

A.1 Biological module

A stock is either modelled using annual age-based, or age- and sex-based dynamics, a surplus production model, or a static model which assumes that the total biomass of the stock remains constant.

Annual age-based dynamics are a simplified version of the age- and sex-based dynamics with only one gender. Both aim at incorporating stocks assessed with *Stock Synthesis 3 (SS3)* and use the equations provided in Methot and Wetzel (2013) given below.

Annual age- and sex-based dynamics are governed by:

$$\begin{aligned} &N_{s,g,a+1,t+1} = N_{s,g,a,t}\, e^{-Z_{s,g,a,t}} \quad a \in [A_{min}; A_{max}-1] \\ &N_{s,g,A_{max},t+1} = N_{s,g,A_{max}-1,t}\, e^{-Z_{s,g,A_{max}-1,t}} + N_{s,g,A_{max},t}\, e^{-Z_{s,g,A_{max},t}} \end{aligned} \tag{A.1}$$

where $N_{s,g,a,t}$ stands for the number of individuals from stock s of age a and gender g at time t, which experience a total mortality $Z_{s,g,a,t}$ equal to the sum of natural mortality $M_{s,g,a}$ and fishing mortality $F_{s,g,a,t}$. The fishing mortality applied to the stock is the sum of the fishing mortalities by vessel i and metier m ($F_{s,g,a,i,m,t}$) and the fishing mortality exerted by non-explicitly modelled fleets ($F_{s,g,a,t,OTH}$): $F_{s,g,a,t} = \sum_{i,m} F_{s,g,a,i,m,t} + F_{s,g,a,t,OTH}$. A_{min} (resp. A_{max}) is the age of the youngest (resp. oldest) modelled age class.

Fishing mortalities at age by vessel and metier ($F_{s,g,a,i,m,t}$) are proportional to individual fishing efforts by metier, assuming a constant catchability rate:

$$F_{s,g,a,i,m,t} = q_{s,g,a,i,m} \times E_{i,m,t}. \tag{A.2}$$

The spawning stock biomass of stock s at time t ($SSB_{s,t}$) is calculated as:

$$SSB_{s,t} = \sum_{a,g} Mat_{s,g,a} w_{s,g,a} N_{s,g,a,t} \tag{A.3}$$

with $Mat_{s,g,a}$ being the proportion of mature individuals of age a and gender g in stock s and $w_{s,a}$ their mean weight. $Mat_{s,g,a}$ for the non-spawning gender is equal to zero.

Recruitment can be modelled with Hockey-stick or Beverton-Holt stock recruitment relationships.

Beverton-Holt relationships calculate recruitment $N_{0,t}$ as follows:

$$N_{0,t} = \frac{4hR_0 SSB_t}{SSB_0(1-h) + SSB_t(5h-1)} e^{\tilde{R}_t - \frac{\sigma_R^2}{2}} \quad \tilde{R}_t \sim N(\mu_R; \sigma_R^2), \tag{A.4}$$

with h the steepness parameter, R_0 the unfished equilibrium recruitment, SSB_0 the unfished equilibrium spawning biomass and R_t the deviation from the recruitment relationship drawn from a normal distribution of mean μ_R (representing recruitment shifts) and standard deviation σ_R^2. Bias in the estimation of the mean associated to the lognormal distribution is corrected by subtracting the factor $\frac{\sigma_R^2}{2}$ in the exponent (Methot and Taylor, 2011).

Recruitment using Hockey-Stick relationships is given by:

$$N_{0,t} = \begin{cases} a \times SSB \times e^{\tilde{R}_t} & \text{if } SSB < SSB^* \\ a \times SSB^* \times e^{\tilde{R}_t} & \text{if } SSB \geq SSB^* \end{cases}, \tag{A.5}$$

with a the slope of the curve, SSB^* the SSB at the breakpoint and R_t being drawn from a normal distribution of mean $-\frac{\sigma_R^2}{2}$ to account for bias in the lognormal distribution and standard deviation σ_R^2.

The surplus production model is that of Fox (1970) with the dynamics of stock biomass being governed by:

$$B_{s,t+1} = B_{s,t} \times (1 + r_s \times \ln(\frac{K_s}{B_{s,t}}) - F_{s,t}), \tag{A.6}$$

with $B_{s,t}$ being the biomass of stock s at time t, r_s its growth rate, K_s its carrying capacity, and $F_{s,t}$ the fishing mortality exerted on the stock at time t. Similarly to age-based dynamics, the fishing mortality is the sum of fishing mortalities exerted by modelled ($\sum_{i,m} F_{s,i,m,t}$) and non-modelled vessels ($F_{s,t,OTH}$): $F_{s,t} = \sum_{i,m} F_{s,i,m,t} +$

$F_{s,t,OTH}$

A.2 Harvest Control Rule module

The management procedures module mimics the process of estimating each year the TAC for the following year so that the stock will be harvested at a fishing mortality rate $\overline{F}_{targ}$. It is a multi-step process:

1. Projection of :
   - — the numbers at age ($N^*_{s,g,a,t+1}$) for year $t+1$ given a mean recruitment R_{mean} and equation A.1 for (sex- and) age-based dynamics.
   - — the biomass ($B^*_{s,t+1}$) for year $t+1$ following equation A.6 for surplus production dynamics.
2. Calculation of current $\overline{F}$:

$$\overline{F}_{s,t} = \frac{1}{\sum_{g,a} p_{s,g,a}} \sum_{g,a} (p_{s,g,a} \times F_{s,g,a,t}) \tag{A.7}$$

   with $\delta_{g,a} = 1$ if age a of gender g is accounted for in the calculation of $\overline{F}$, and 0 otherwise

3. Calculation of fishing mortalities at age to reach $\overline{F}_{targ}$ (for (sex- and) age-based dynamics):

$$F_{targ_{s,g,a}} = F_{s,g,a,t} \times \frac{\overline{F}_{s,t}}{\overline{F}_{targ_s}} \tag{A.8}$$

4. Calculation of resulting landings using the equations from the catch module (Section A.5):

$$\begin{aligned} L^*_{s,t+1} &= \sum_{g,a} [(1 - d_{s,g,a}) w L_{s,g,a} \frac{F_{targ_{s,g,a}}}{F_{targ_{s,g,a}} + M_{s,g,a}} \\ &\times N^*_{s,g,a,t+1} \times (1 - e^{-F_{targ_{s,g,a}} + M_{s,g,a}})] \quad \text{for (sex-) and age-based dynamics} \\ &= (1 - d_s) \times \overline{F}_{targ_s} \times B^*_{s,t+1} \quad \text{for surplus production dynamics,} \end{aligned} \tag{A.9}$$

   which are used to set the TAC:

$$TAC_{s,t+1} = L^*_{s,t+1} \tag{A.10}$$

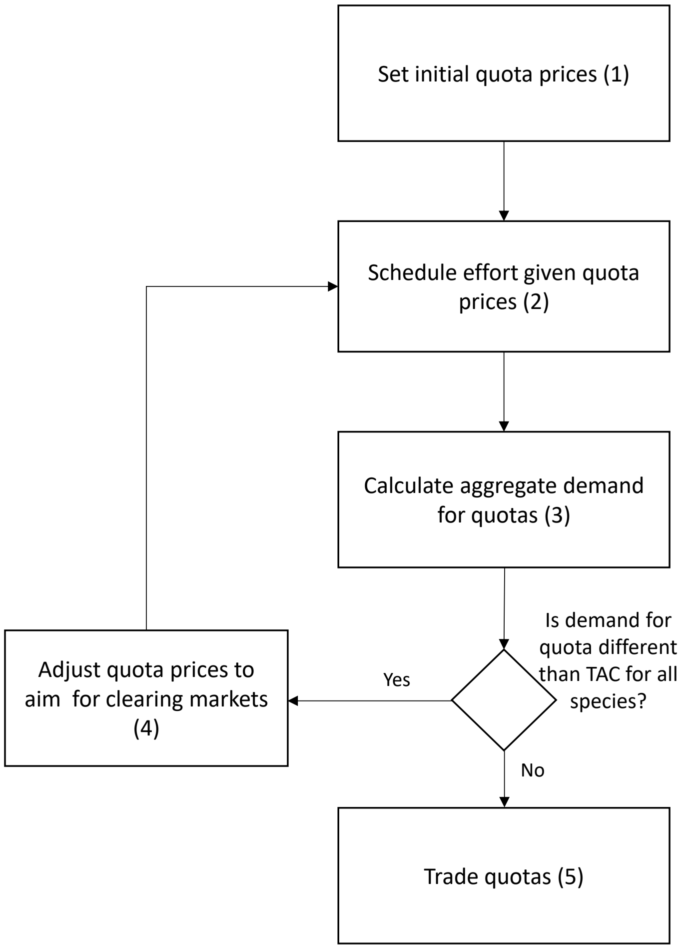


Figure A.1 – Flowchart of the ITQ market module

A.3 ITQ market module

This module simulates a walrassian tâtonnement process to determine market clearing lease prices for quota units. As summarized in Figure A.1, it builds on an iterative algorithm which progresses as follows:

1. Quota lease prices are given an initial value. [1]

   For each iteration *it* of the tâtonnement process (for the sake of clarity, *it* subscripts will be omitted in the following equations when not necessary):

2. Fishers decide on a fishing strategy by allocating fishing effort among their various

1. Quota lease prices are attributed a quasi-null value before entering the tâtonnement phase. The advantage of initiating quota prices at a null value is that at they will have to increase for markets to clear (at least for those who will clear given joint productions). However, given the form of the quota price adjustment function given in Equation A.15, one cannot start with a null value. Therefore, quota lease prices were initiated at 0.1% of the species ex-vessel price.

metiers.

— The effort allocation is modelled as a function of a weighted average of the metiers' expected profitability and past effort allocation, as detailed in Section A.4. The expected profitability of metier m for individual harvester i at time t ($ProfPUE_{i,m,t}$) is calculated as follows:

$$\begin{aligned} ProfPUE^*_{i,m,t} = &(1 - cshr_i) \times \sum_s (\frac{L_{s,i,m,t-1}}{E_{i,m,t-1}} \times p_{s,t-1}) \\ &- \sum_s (\frac{L_{s,i,m,t-1}}{E_{i,m,t-1}} \times \tilde{qp}_{s,t}) \\ &- CvarUE_{i,m,t-1} - \frac{Cfix_i + Cdep_i + Copport_i}{E_{i,t-1}}, \end{aligned} \quad (A.11)$$

if crews are remunerated as a share of the fishing income, and

$$\begin{aligned} ProfPUE^*_{i,m,t} = &(1 - cshr_i) \times (\sum_s (\frac{L_{s,i,m,t-1}}{E_{i,m,t-1}} \times p_{s,t-1}) - CvarUE_{i,m,t-1}) \\ &- \sum_s (\frac{L_{s,i,m,t-1}}{E_{i,m,t-1}} \times \tilde{qp}_{s,t}) \\ &- \frac{Cfix_i + Cdep_i + Copport_i}{E_{i,t-1}} \end{aligned} \quad (A.12)$$

if crews are remunerated as a share of the fishing income - variable costs (the "return to be shared").

$cshr_i$ represents the crew share of individual harvester i, $\frac{L_{s,i,m,t-1}}{E_{i,m,t-1}}$ the previous year's landings per unit of effort of species s by individual i in metier m, $p_{s,t-1}$ the ex-vessel price of species s in the previous year and $\tilde{qp}_{s,t}$ its lease price in the current iteration. $CvarUE_{i,m,t-1}$ represents the variable costs per unit of effort in the previous year for individual i in metier m, and $\frac{Cfix_i+Cdep_i+Copport_i}{E_{i,t-1}}$ the vessel's fixed and capital costs per unit of effort. Relative profitabilities $ProfPUE^*_c$ (i.e. centred on the profitability of the vessel's least profitable metier), are used in the effort allocation function to avoid negative coefficients: $ProfPUE^*_{c_{i,m,t}} = ProfPUE^*_{i,m,t} - \min_m(ProfPUE^*_{i,m,t})$

The proportion of effort individual harvester i allocates to metier m ($pE_{i,m,t}$) is calculated as follows:

$$pE_{i,m,t} = \alpha \times \frac{ProfPUE^*_{c_{i,m}}}{\sum_m ProfPUE^*_{c_{i,m}}} + (1 - \alpha) \times \frac{E_{0_{i,m}}}{\sum_m E_{0_{i,m}}}, \quad (A.13)$$

with $E_{0_{i,m}}$ being the historical effort of individual harvester i allocated to metier m and α the weight given to profitability in the allocation of effort.

— Given the planned fishing effort allocation, individual harvester i assesses whether it is profitable to go fishing. If its average profitability per unit of effort, $ProfPUE^*_{i,t} = \sum_m (pE_{i,m,t} \times ProfPUE^*_{i,m,t})$, is positive, then the individual harvester will operate at its maximum effort E_{max_i}, otherwise it will remain at port:

$$E_{i,t} = \begin{cases} E_{max_i} & \text{if } ProfPUE^*_{i,t} > 0, \\ 0 & \text{otherwise.} \end{cases} \tag{A.14}$$

3. Once fishing efforts are set, the biological and catch modules are called to estimate individual demands for quota on species s for the current iteration ($L^*_{i,s,t,it}$).
4. Quota prices are then adjusted for the next iteration to aim for clearing markets for each species:

$$\tilde{qp}_{s,it+1} = \tilde{qp}_{s,it} \times (1 - \lambda_q \times \frac{TAC_{s,t} - \sum_i L^*_{s,i,t,it}}{TAC_{s,t}}), \tag{A.15}$$

with $TAC_{s,t}$ being the total allowable catch for species s at time t and λ_q a fixed multiplier. [2]

Steps 2 to 4 are iterated until total demand for quota is close enough to the TAC for all species (i.e $\frac{TAC_{s,t} - \sum_i L_{s,i,t}}{TAC_{s,t}} < \epsilon$) or after it_{max} iterations.

5. Once quota markets have reached equilibrium, quota prices are set at their equilibrium value ($qp_{s,t} = \tilde{qp}_{s,t,equ.}$) and quotas for all species are traded. In a multi-species context, all quota markets may not clear which requires the trading process to be explicitly modelled as all leasers may not find a buyer. Individual net demands for quota (*Demand*) are calculated by deducing initial quota holdings (*Holdings*) from expected catches at equilibrium (L^*):

$$Demand_{s,i,t} = Holdings_{s,i,t} - L^*_{s,i,t}. \tag{A.16}$$

Similar to the approach proposed by Little et al. (2009), priority is given to trades between participants with the highest incentive to lease out or rent quota. The incentive to take part in a trade is measured by $\mid ProfPUE^*_i \mid$. For each species, quota leasers are ranked by decreasing order of profitability and quota lessors by

2. The value of the multiplier is determined empirically to achieve a satisfying compromise between the precision of convergence and computing time. Precision will be greater for low values of λ but at the cost of increased convergence time.

increasing order of profitability. Trades are conducted by order of priority under the limit of what is available or needed, i.e $\min(-Demand_{s,lessor}, Demand_{s,leaser})$, and so until offer or demand expires. Some quota may also be hold by external investors which are grouped into an additional market participant. Currently, external investors are given priority in trades but this could easily be modified. [3]

A.4 Short-term behaviour module

The short-term behaviour module determines fishing effort at the metier level for each individual harvester. It is a 2-step process:

1. determination of the fishing strategy, i.e. the allocation of fishing effort among various metiers
2. reconciliation of effort against quota constraints on several species

First, the allocation of fishing effort among several metiers is modelled as a function of a weighted average between habit and expected profitability. At the beginning of each year, individual harvesters estimate the profitability of the metiers they practice ($ProfPUE^{*}$) based on the information available to them at the time, i.e. their past catch rates and costs but the current quota lease prices. At time t, the expected profitability $ProfPUE^{*}_{i,m,t}$ of metier m operated by individual i corresponds to its expected profits per unit of effort, namely:

$$\begin{aligned} ProfPUE^{*}_{i,m,t} =& (1-cshr_i) \times \sum_s (\frac{L_{s,i,m,t-1}}{E_{i,m,t-1}} \times p_{s,t-1}) \\ & - \sum_s (\frac{L_{s,i,m,t-1}}{E_{i,m,t-1}} \times qp_{s,t}) \\ & - CvarUE_{i,m,t-1} - \frac{Cfix_i + Cdep_i + Copport_i}{E_{i,t-1}} \end{aligned} \tag{A.17}$$

if crews are remunerated as a share of the fishing income, and

$$\begin{aligned} ProfPUE^{*}_{i,m,t} =& (1-cshr_i) \times (\sum_s (\frac{L_{s,i,m,t-1}}{E_{i,m,t-1}} \times p_{s,t-1}) - CvarUE_{i,m,t-1}) \\ & - \sum_s (\frac{L_{s,i,m,t-1}}{E_{i,m,t-1}} \times qp_{s,t}) \\ & - \frac{Cfix_i + Cdep_i + Copport_i}{E_{i,t-1}} \end{aligned} \tag{A.18}$$

3. In the version in Chapters 4 and 5, all quota shares were assumed to be owned by external investors as information on individual quota holdings was not available.

if crews are remunerated as a share of the fishing income - variable costs (the "return to be shared").

$cshr_i$ represents the crew share of individual harvester i, $\frac{L_{s,i,m,t-1}}{E_{i,m,t-1}}$ the landings per unit of effort of species s by individual i in metier m in the previous year, $p_{s,t-1}$ the ex-vessel price of species s in the previous year and $qp_{s,t}$ its lease price estimated by the ITQ market module (Section A.3). $CvarUE_{i,m,t-1}$ represent the variable costs per unit of effort for individual i in metier m in the previous year, and $\frac{Cfix_i+Cdep_i+Copport_i}{E_{i,t-1}}$ the individual's fixed and capital costs (depreciation and opportunity) per unit of effort. Relative profitabilities $ProfPUE_c^*$ (i.e. centred on the profitability of the vessel's least profitable metier), are used in the effort allocation function to avoid negative coefficients: $ProfPUE^*_{c_{i,m,t}} = ProfPUE^*_{i,m,t} - \min_m(ProfPUE^*_{i,m,t})$

The proportion of effort $pE_{i,m,t}$ allocated by individual i to metier m at time t is then calculated as:

$$pE_{i,m,t} = \alpha \times \frac{ProfPUE^*_{c_{i,m,t}}}{\sum_m ProfPUE^*_{c_{i,m,t}}} + (1-\alpha) \times \frac{E_{0_{i,m}}}{\sum_m E_{0_{i,m}}}, \tag{A.19}$$

with $E_{0_{i,m}}$ being the original effort of individual i allocated to metier m and α the weight given to profitability against habit when allocating effort.

Second, individual fishing efforts at the metier level ($E_{i,m,t}$) reconciling the various quota constraints are determined in a 2-step process:

1. Calculation of the effort $E_{i,m,s,t}$ required to catch the quota $Q_{i,s,t}$ for each individual harvester i, metier m and stock s, which can be formulated by the following problem:

$$\text{Find } \lambda_{i,s,t} \text{ such that } \begin{cases} \sum\limits_m L_{i,m,s,t} = Q_{i,s,t}, \\ E_{i,m,s,t} = \lambda_{i,s,t} \times E_{i,t_0} \times pE_{i,m,t}. \end{cases} \tag{A.20}$$

   with $L_{i,m,s,t}$ the landings of stock s at time t by individual i in metier m as calculated by the catch module (Section A.5).

2. Effort reconciliation at the metier level so that each fisherman stops fishing with metier m either when its most constraining quota is exhausted or when he has reached the upper limit E_{max}, i.e.:

$$E_{i,m,t} = \min(E_{max_i} \times pE_{i,m,t}, \min_s E_{i,m,s,t}). \tag{A.21}$$

A.5 Catch module

Individual landings by metier for species with (sex- and) age-based dynamics are given by:

$$L_{i,m,s,g,a,t} = (1 - d_{s,g,a})\, wL_{s,g,a}\, \frac{F_{i,m,s,g,a,t}}{Z_{s,g,a,t}}\, N_{s,g,a,t}\, (1 - e^{-Z_{s,g,a,t}}), \tag{A.22}$$

with $d_{s,g,a}$ the proportion of discarded individuals of gender g, age a and stock s, and $wL_{s,g,a}$ the individual weight at age in the landings of stock s.

Landings of stocks modelled with a surplus production model are given by:

$$L_{i,m,s,t} = (1 - d_s)\, \times F_{i,m,s,t}\, \times B_{s,t} \tag{A.23}$$

Those for static stocks are given by:

$$L_{i,m,s,t} = LPUE_{i,m,s} \times E_{i,m,t}, \tag{A.24}$$

$LPUE_{i,m,s}$ being the landings of stock s per unit of effort for individual harvester i using metier m.

A.6 Fish market module

This module models the response of fish prices to changes in the species' landings using own- and cross-price flexibilities. The ex-vessel price of species s at time t, $p_{s,t}$, is given by:

$$\ln(\frac{p_{s,t}}{p_{s,0}}) = \sum_{s'} \beta_{s,s'} \times \ln(\frac{L_{s',t}}{L_{s',0}}), \tag{A.25}$$

where $L_{s,t}$ refers to the landings of species s at time t and $\beta_{s,s'}$ the flexibility of the price of species s with regards to the landings of species s'.

A.7 Economic module

This module calculates for each individual harvester i a variety of economic outputs, which are listed in Bellanger et al. (2018). Indicators used in the thesis are defined in each chapter.

APPENDIX B

CALIBRATION OF THE MODEL IAM FOR THE BAY OF BISCAY DEMERSAL MIXED FISHERY

B.1 Biological module

The list of stocks and dynamics used to represent them in the model is provided in Table B.1. The calibration of population dynamics models (Table B.2) was based on the following stock assessments:

— European hake: ICES (2017a) - Section 9

— Norway lobster: pers. comm. Spyros Fifas and ICES (2017a) - Section 11

— Common sole: ICES (2017a) - Section 7

— European seabass: ICES (2017a) - Section 14

1. This stock is assessed by ICCAT (see ICCAT (2017) for further information)

Table B.1 – Modelled stocks in the Bay of Biscay

Species	**FAO divisions**	**Quota**	**ICES cat.**	**IAM dynamics**
European hake (*Merluccius merluccius*)	IIIa, IV, VI, VII, VIIIabd	X	1	quarterly age-based
Norway lobster (*Nephrops norvegicus*)	VIIIab	X	1	annual age-based
Common Sole (*Solea solea*)	VIIIab	X	1	annual age-based
Anglerfish (*Lophius spp*)	VIIb-k, VIIIabd	X	3	static
European seabass (*Dicentrarchus labrax*)	VIIIab		3	annual age-based
Common cuttlefish (*Sepia officinalis*)	VIIIab		NA	static
European pilchard (*Sardina pilchardus*)	VII, VIIIabd		2	static
Inshore squids nei (*Loliginidae*)	VIIIab		NA	static
Albacore (*Thunnus alalunga*)	27	X	NA [1]	static
Pollack (*Pollachius pollachius*)	VIII, IXa	X	5	static
Surmullet (*Mullus surmuletus*)	VI, VIII, VIIa-c, e-k, IXa		5	static
Whiting (*Merlangius merlangus*)	VIII, IXa	X	5	static
Meagre (*Argyrosomus regius*)	VIIIab		NA	static
John dory (*Zeus faber*)	VIIIab		NA	static
European conger (*Conger conger*)	27		NA	static
Atlantic mackerel (*Scomber scombrus*)	27	X	1	static
European anchovy (*Engraulis encrasicolus*)	VIII	X	1	static
Megrims (*Lepidorhombus spp*)	VIIb-k, VIIIabd	X	1-5	static
Rays	VIII, IX	X	3-6	static
Blue whiting (*Micromesistius poutassou*)	I-IX, XII, XIV	X	1	static
Atlantic horse mackerel (*Trachurus trachurus*)	IIa, IVa, Vb, VIa, VIIa-c,e-k, VIII	X	1	static

Table B.2 – Biological parameters

Species	Parameter	Age						
		2	3	4	5	6	7	8+
Common sole	Initial abundance $N_{a,2016}$ $(*10^6)$	18.92	1.45	6.65	3.70	2.32	1.87	2.12
	Natural mortality rate M_a (yr^{-1})	0.1	0.1	0.1	0.1	0.1	0.1	0.1
	Initial fishing mortality rate F_a (yr^{-1})	0.07	0.28	0.55	0.41	0.42	0.56	0.56
	$\overline{F}$ weighting coeff. δ_a	0	1	1	1	1	0	0
	Weight at age w_a (kg)	0.2	0.25	0.3	0.37	0.39	0.4	0.56
	% of mature individuals Mat_a	0.32	0.83	0.97	1	1	1	1
	Discard rate D_a	0	0	0	0	0	0	0
	Mean recruitment $(*10^6)$	21.0						

Species	Parameter	Age								
		1	2	3	4	5	6	7	8	9+
Norway lobster	Initial abundance $N_{a,2016}$ $(*10^3)$	631351	559691	290669	131520	43031	16892	5977	3112	2500
	Natural mortality rate M_a (yr^{-1})	0.3	0.3	0.25	0.25	0.25	0.25	0.25	0.25	0.25
	Initial fishing mortality rate F_a (yr^{-1})	0.01	0.18	0.57	0.93	1.03	0.68	0.71	0.65	0.65
	$\overline{F}$ weighting coeff. δ_a	0	1	1	1	1	0	0	0	0
	Weight at age w_a (kg)	0.004	0.009	0.017	0.026	0.034	0.044	0.051	0.056	0.067
	Proportion of mature individuals Mat_a	0	0	0.75	1	1	1	1	1	1
	Discard rate D_a	1	0.93	0.42	0.18	0.11	0.08	0.08	0.18	0.08

Species	**Parameter**	**Age**																				
		0	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20+
Europ. seabass	Initial abundance $N_{a,2016}$ $(*10^3)$	36406	15650	16902	4824	6516	8643	2688	2447	2518	1769	1243	695	607	602	296	290	112	93	76	43	101
	Natural mortality rate M_a (yr^{-1})	0.24	0.24	0.24	0.24	0.24	0.24	0.24	0.24	0.24	0.24	0.24	0.24	0.24	0.24	0.24	0.24	0.24	0.24	0.24	0.24	0.24
	Initial fishing mortality rate F_a (yr^{-1})	0.00	0.00	0.00	0.00	0.03	0.08	0.12	0.13	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14
	$\overline{F}$ weighting coeff. δ_a	0	0	0	0	1	1	1	1	1	1	1	1	1	1	1	1	0	0	0	0	0
	Weight at age w_a (kg)	0.00	0.02	0.08	0.18	0.33	0.52	0.74	0.98	1.23	1.49	1.76	2.02	2.27	2.52	2.75	2.97	3.18	3.37	3.55	3.71	4.15
	Proportion of mature individuals Mat_a	0	0	0	0	0.04	0.21	0.51	0.75	0.89	0.95	0.98	0.99	0.99	1	1	1	1	1	1	1	1
	Discard rate D_a	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Species	**Parameter**	**Morph**	**Sem.**	**Age**															
				0	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15+
Europ. hake	Initial abundance $N_{a,2016}$ $(*10^4)$	M1	S1	13.77	5.06	2.13	2.54	1.55	0.24	0.09	0.06	0.07	0.01	0.00	0.00	0.00	0.00	0.00	0.00
			S2	12.45	4.48	1.86	2.15	1.28	0.20	0.07	0.05	0.06	0.01	0.00	0.00	0.00	0.00	0.00	0.00
			S3	11.22	3.92	1.60	1.76	1.02	0.16	0.06	0.04	0.05	0.01	0.00	0.00	0.00	0.00	0.00	0.00
			S4	9.87	3.44	1.36	1.44	0.82	0.13	0.05	0.04	0.04	0.01	0.00	0.00	0.00	0.00	0.00	0.00
		M2	S1	0.00	11.60	4.88	2.38	2.35	0.86	0.40	0.12	0.17	0.05	0.01	0.00	0.00	0.00	0.00	0.00
			S2	27.44	10.33	4.28	2.03	1.95	0.71	0.34	0.10	0.14	0.04	0.01	0.00	0.00	0.00	0.00	0.00
			S3	24.82	9.04	3.70	1.68	1.57	0.57	0.27	0.08	0.12	0.03	0.01	0.00	0.00	0.00	0.00	0.00
			S4	22.32	7.92	3.20	1.38	1.26	0.46	0.22	0.07	0.10	0.03	0.00	0.00	0.00	0.00	0.00	0.00
		M3	S1	0.00	5.61	2.36	2.96	1.96	0.31	0.11	0.08	0.10	0.02	0.00	0.00	0.00	0.00	0.00	0.00
			S2	0.00	5.03	2.08	2.54	1.63	0.26	0.09	0.07	0.08	0.02	0.00	0.00	0.00	0.00	0.00	0.00
			S3	11.74	4.41	1.80	2.13	1.32	0.21	0.08	0.05	0.07	0.01	0.00	0.00	0.00	0.00	0.00	0.00
			S4	10.62	3.85	1.57	1.78	1.06	0.17	0.06	0.04	0.06	0.01	0.00	0.00	0.00	0.00	0.00	0.00
	Initial fishing mortality rate in number Fq_a $(year^{-1})$	M1	S1	0.00	0.08	0.14	0.27	0.37	0.36	0.32	0.29	0.26	0.25	0.24	0.24	0.24	0.24	0.24	0.24
			S2	0.02	0.14	0.22	0.39	0.48	0.47	0.43	0.40	0.38	0.37	0.37	0.37	0.36	0.36	0.36	0.36
			S3	0.11	0.12	0.23	0.41	0.47	0.44	0.39	0.36	0.34	0.33	0.33	0.32	0.32	0.32	0.32	0.32
			S4	0.09	0.13	0.29	0.45	0.46	0.41	0.36	0.33	0.31	0.30	0.30	0.29	0.29	0.29	0.29	0.29
		M2	S1	0.00	0.06	0.12	0.24	0.36	0.37	0.33	0.29	0.27	0.25	0.25	0.24	0.24	0.24	0.24	0.24
			S2	0.00	0.13	0.18	0.35	0.47	0.48	0.44	0.41	0.39	0.38	0.37	0.37	0.37	0.36	0.36	0.36
			S3	0.02	0.13	0.18	0.38	0.47	0.45	0.41	0.37	0.35	0.33	0.33	0.32	0.32	0.32	0.32	0.32
			S4	0.06	0.12	0.24	0.42	0.47	0.43	0.37	0.34	0.31	0.30	0.30	0.29	0.29	0.29	0.29	0.29
		M3	S1	0.00	0.04	0.11	0.20	0.33	0.38	0.34	0.30	0.27	0.26	0.25	0.24	0.24	0.24	0.24	0.24
			S2	0.00	0.12	0.16	0.30	0.45	0.48	0.45	0.42	0.39	0.38	0.37	0.37	0.37	0.36	0.36	0.36
			S3	0.00	0.14	0.15	0.33	0.46	0.46	0.42	0.38	0.35	0.34	0.33	0.32	0.32	0.32	0.32	0.32
			S4	0.01	0.11	0.19	0.39	0.47	0.44	0.38	0.34	0.32	0.31	0.30	0.29	0.29	0.29	0.29	0.29

Species	Parameter	Morph	Sem.	Age															
				0	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15+
Europ. hake	Initial mortality rate from landings FLw_a ($year^{-1}$)	M1	S1	0.00	0.00	0.07	0.37	0.85	1.19	1.42	1.63	1.86	2.10	2.32	2.53	2.71	2.87	3.00	3.11
			S2	0.00	0.01	0.13	0.61	1.22	1.70	2.10	2.49	2.89	3.27	3.62	3.93	4.20	4.43	4.63	4.80
			S3	0.00	0.02	0.20	0.76	1.32	1.70	2.01	2.32	2.65	2.97	3.27	3.53	3.76	3.95	4.11	4.26
			S4	0.00	0.04	0.32	0.92	1.39	1.68	1.94	2.21	2.50	2.77	3.03	3.26	3.46	3.63	3.77	3.89
		M2	S1	0.00	0.00	0.04	0.26	0.74	1.12	1.37	1.58	1.80	2.04	2.27	2.48	2.67	2.83	2.97	3.08
			S2	0.00	0.00	0.08	0.46	1.08	1.60	2.00	2.39	2.79	3.17	3.53	3.86	4.14	4.38	4.58	4.76
			S3	0.00	0.01	0.12	0.60	1.20	1.61	1.93	2.24	2.57	2.89	3.20	3.47	3.70	3.91	4.08	4.22
			S4	0.00	0.02	0.21	0.77	1.29	1.62	1.87	2.14	2.42	2.71	2.97	3.21	3.41	3.59	3.73	3.86
		M3	S1	0.00	0.00	0.02	0.18	0.61	1.04	1.31	1.52	1.74	1.98	2.21	2.43	2.62	2.79	2.94	3.06
			S2	0.00	0.00	0.05	0.32	0.92	1.48	1.91	2.30	2.69	3.08	3.45	3.78	4.07	4.32	4.53	4.71
			S3	0.00	0.00	0.07	0.45	1.06	1.53	1.85	2.16	2.49	2.81	3.12	3.40	3.65	3.86	4.04	4.19
			S4	0.00	0.01	0.12	0.61	1.18	1.55	1.81	2.07	2.35	2.64	2.91	3.15	3.36	3.55	3.70	3.83
	Initial mortality rate from discards FDw_a ($year^{-1}$)	M1	S1	0.00	0.01	0.02	0.03	0.03	0.02	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
			S2	0.00	0.01	0.03	0.06	0.05	0.03	0.02	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00
			S3	0.00	0.01	0.04	0.05	0.04	0.02	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
			S4	0.00	0.02	0.04	0.05	0.03	0.02	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
		M2	S1	0.00	0.01	0.03	0.06	0.05	0.03	0.02	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00
			S2	0.00	0.01	0.02	0.05	0.05	0.03	0.02	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00
			S3	0.00	0.01	0.03	0.05	0.04	0.03	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
			S4	0.00	0.01	0.04	0.05	0.03	0.02	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
		M3	S1	0.00	0.00	0.01	0.03	0.03	0.02	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
			S2	0.00	0.01	0.02	0.05	0.05	0.04	0.02	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00
			S3	0.00	0.01	0.02	0.05	0.05	0.03	0.02	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00
			S4	0.00	0.01	0.03	0.05	0.04	0.02	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	$\overline{F}$ weighting coeff. δ_a	-	-	0.29	1	1	1	1	0.4	0	0	0	0	0	0	0	0	0	0

Species	Parameter	Morph	Sem.	Age 0	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15+
Europ. Hake	Natural mortality rate M_a (yr^{-1})	-	-	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4
	Mature weight at age $MatWt_a$ (kg)	M1	-	0.00	0.00	0.20	1.01	2.11	3.35	4.67	6.01	7.29	8.48	9.54	10.5	11.3	12.0	12.6	13.1
		M2	-	0.00	0.00	0.09	0.77	1.82	3.03	4.34	5.68	6.98	8.19	9.29	10.3	11.1	11.8	12.4	12.9
		M3	-	0.00	0.00	0.03	0.55	1.54	2.71	4.00	5.34	6.66	7.90	9.03	10.0	11.9	11.7	12.3	12.8
	Mean recruitment N_0 ($*10^4$)	M1	S1	9.22															
		M2	S2	16.4															
		M3	S3	7.87															

B.2 Fishing activity and catch module

Table B.3 provides the correspondance between represented metiers in the model and DCF level 5 categories. Aggregation aimed at representing the main fishing activities in the fishery identified by Macher et al. (2015).

Table B.3 – List of metiers

Name	**DCF code**
Demersal trawl Nephrops	OTBNEP, OTTNEP
Demersal trawl Sole	OTBSOX, OTTSOX
Demersal trawl Anglerfish	OTBMNZ, OTTMNZ
Demersal trawl Cephalopods	OTBSQU, OTTSQU, OTBCTL, OTTCT
Demersal trawl other species	OTTxxx, OTBxxx
Pelagic trawl other species	OTMxxx, PTMxxx
Danish Seine	SDNxxx
Net Sole	GTRSOX, GNSSOX
Net Hake	GNSHKE
Net other species	GNSxxx, GTSxxx
Longline Hake	LLSHKE
Longline other species	LLSxxx, LL_xxx
Other	All not mentionned above

Effort and catch data at the vessel- and metier-level was retrieved from the SACROIS database (Système d'Information Halieutique, 2018).

B.3 Economic module

Cost structures were estimated for each fleet segment from 2016 economic data (Service de la Statistique et de la Prospective, 2016). Fixed costs ($Cfix$) are the aggregation of repair costs $Crep$, crew costs in addition to wages $Ccrewoth$ (vacation and employer contribution calculated based on legislation in force in 2016), and other fixed costs $Cfixoth$. Variable costs are the sum of fuel costs $Cfuel$ and other variable costs $Cvaroth$ (ice, food supply ...). Crew share $cshr$ was calculated as the proportion of crew wages (calculated as crew costs $Ccrew$ minus the additional crew costs mentioned previously) relative to the "return to be shared" (i.e. fishing income minus variable costs). Depreciation costs were calculated according to the Perpetual Inventory Method (PIM) as described in (Onlus, 2006). Vessel prices at construction time were provided by DPMA.

Table B.4 provides the equations used to derive costs at the vessel level (indicated by subscript i) from costs structures estimated at the fishing segment level (indicated by subscript f). Cost structures at the fishing segment level are given in Table B.6.

Table B.4 – Individual economic calibration from fleets' cost structures

Cost	**Calculation**
Variable costs per unit effort	$CvarUE_i = (Cfuel_f + Cvaroth_f) \times \frac{GVL_i}{GVL_f} \times \frac{1}{E_i}$
Fixed costs	$Cfix_i = (Cfixoth_f + Crep_f) \times \frac{GVL_i}{GVL_f} + Ccrewoth_i$
Crew share	$cshr_i = cshr_f$
Value of vessel at construction time	$K_i = VP_{f,VL} \times VL_i$ $VP_{f,VL}$ = Vessel price per meter for vessels of fleet f and length class VL VL_i = length of vessel i
Depreciation costs	$Cdep_i = Cdep_{hull_i} + Cdep_{motor_i} + Cdep_{elec_i} + Cdep_{other_i}$ with element k linearly depreciated over its life length nb_yr_k, and its price p_k calculated as a fraction pK_k of vessel value K: $Cdep_k = \frac{p_k}{nb\_yr_k} = \frac{pK_k \times K}{nb\_yr_k}$

Table B.5 – Depreciation parameters

Element	**Parameter**	
	pK	nb_yr
Hull	0.6	40
Motor	0.2	10
Electronics	0.1	5
Other	0.1	7

Table B.6 – Cost structures for the Bay of Biscay demersal fleets. Costs per fishing segment are provided as a proportion of fishing income.
Source: Système d'Information Halieutique (2018) and SSP (Service de la Statistique et de la Prospective), Enquête sur la production de données économiques dans le secteur des pêches maritimes

Fleet	**Vessel length**	**Variable**					
		nb_crew	*Ccrew*	*Cfixoth*	*Crep*	*Cfuel*	*Cvaroth*
Bass longliners	VL0010	1.45	0.42	0.10	0.05	0.04	0.14
	VL1012	2.33	0.43	0.11	0.09	0.06	0.13
Danish seineurs	VL1840	4.73	0.36	0.08	0.14	0.12	0.12
Demersal trawlers_ outBoB	VL1012	2.64	0.38	0.10	0.07	0.09	0.19
	VL1218	3.41	0.37	0.10	0.09	0.13	0.16
	VL1824	4.95	0.34	0.08	0.11	0.17	0.18
	VL2440	7.50	0.29	0.15	0.11	0.23	0.15
	VL40xx	18.26	0.30	0.14	0.11	0.11	0.13
Gillnetters_ outBoB	VL0010	1.60	0.48	0.16	0.07	0.04	0.14
	VL1012	3.23	0.48	0.14	0.04	0.05	0.14
	VL1218	4.56	0.44	0.11	0.06	0.05	0.10
	VL2440	12.52	0.38	0.12	0.04	0.05	0.05
Hake gillnetters	VL1824	7.89	0.34	0.13	0.05	0.04	0.12
	VL1218	7.89	0.34	0.13	0.05	0.04	0.12
	VL2440	12.52	0.38	0.12	0.04	0.05	0.05
Hake longliners	VL0010	2.75	0.49	0.13	0.06	0.05	0.12
	VL1012	2.75	0.49	0.13	0.06	0.05	0.12
	VL1840	15.52	0.33	0.14	0.05	0.08	0.13
Hake longliners_ outBoB	VL1840	15.52	0.33	0.14	0.05	0.08	0.13
Mixed demersal trawlers	VL0010	1.45	0.39	0.14	0.06	0.10	0.12
	VL1012	2.13	0.39	0.10	0.09	0.12	0.12
	VL1218	3.49	0.40	0.08	0.09	0.14	0.14
	VL1824	4.77	0.36	0.08	0.09	0.15	0.13
	VL2440	7.48	0.33	0.10	0.05	0.16	0.10
Mixed gillnetters	VL1218	4.61	0.45	0.14	0.07	0.04	0.14
	VL1824	6.17	0.46	0.12	0.10	0.05	0.15
	VL0010	1.42	0.40	0.17	0.08	0.04	0.07
	VL1012	2.33	0.45	0.14	0.09	0.06	0.13
	VL2440	12.52	0.38	0.12	0.04	0.05	0.05
Non-specialized Nephrops trawlers	VL0012	2.02	0.39	0.06	0.07	0.13	0.13
	VL1218	2.97	0.40	0.07	0.10	0.14	0.16
	VL1824	4.87	0.40	0.06	0.13	0.14	0.13
Pelagic trawlers	VL1218	4.86	0.41	0.07	0.10	0.14	0.13
	VL1824	5.65	0.38	0.08	0.11	0.13	0.15
Sole gillnetters	VL0010	2.15	0.44	0.12	0.06	0.05	0.10
	VL1012	3.61	0.44	0.15	0.07	0.05	0.12
	VL1218	4.61	0.45	0.14	0.07	0.04	0.14

	VL1824	6.17	0.46	0.12	0.10	0.05	0.15
Specialized Nephrops trawlers	VL0012	2.25	0.39	0.08	0.09	0.11	0.09
	VL1218	3.48	0.45	0.08	0.12	0.12	0.10
	VL1824	4.87	0.40	0.06	0.13	0.14	0.13
Vessels using active and passive gears	VL0010	1.37	0.41	0.15	0.05	0.06	0.15
	VL1012	1.74	0.39	0.16	0.06	0.08	0.12
Vessels using active and passive gears_ outBoB	VL0010	1.44	0.35	0.11	0.07	0.04	0.16
	VL1012	3.27	0.49	0.17	0.08	0.04	0.18
Vessels using other active gears	VL0010	1.03	0.48	0.16	0.06	0.05	0.11
Vessels using polyvalent active gears	VL0010	1.39	0.38	0.11	0.05	0.06	0.10
	VL1012	2.70	0.39	0.09	0.05	0.10	0.09
	VL1218	3.10	0.42	0.10	0.13	0.12	0.15
Vessels using polyvalent active gears only_ outBoB	VL1012	2.70	0.39	0.09	0.05	0.10	0.09
Vessels using polyvalent passive gears	VL0010	2.83	0.48	0.15	0.08	0.04	0.11
	VL1012	2.63	0.49	0.09	0.05	0.04	0.13
Vessels using polyvalent passive gears_ outBoB	VL0010	1.13	0.48	0.20	0.05	0.07	0.14
Vessels using pots/traps	VL1012	1.25	0.42	0.12	0.03	0.02	0.08
	VL0010	1.34	0.38	0.14	0.05	0.04	0.12
Vessels using pots/traps_ outBoB	VL1012	3.07	0.43	0.09	0.05	0.05	0.17
	VL1012	2.97	0.45	0.08	0.06	0.05	0.22

APPENDIX C

CALIBRATION OF THE MODEL IAM FOR THE AUSTRALIAN SOUTHERN AND EASTERN SCALEFISH AND SHARK FISHERY

C.1 Biological module

The list of represented stocks and associated population dynamics models is given in Table C.1.

Table C.2 provides the calibration of stock dynamics. Parameters for the stocks modelled with a global surplus production model were either retrieved from Pascoe et al. (2018b) (School shark, gummy shark and mirror dory) or specially estimated for this work (Ocean perch, john dory and blue-eye trevalla). The later were carried out with the package *datalowSA* (Haddon, 2019) and using time series of catches from Castillo-Jordán et al. (2018) and catch rates from Sporcic and Haddon (2018). Model parameters for the (sex- and) age-based dynamics were obtained from stock assessments carried out by CSIRO Oceans and Atmosphere using the statistical framework *Stock Synthesis (SS3)* (Methot and Wetzel, 2013). References to those stock assessments are:

— School whiting: Day (2017),

— Silver warehou: Burch et al. (2018),

— Jackass morwong (East): Day and Castillo-Jordán (2018a),

— Jackass Morwong (West): Day and Castillo-Jordán (2018b),

— Tiger flathead: Day (2016),

— Blue grenadier: Castillo-Jordán and Tuck (2018),

— Pink ling (East): pers. com. Sandra Curin-Osorio,

Table C.1 – Modelled stocks in the Southern and Eastern Scalefish and Shark Fishery

Stock	**Species**	**Quota**	**IAM dynamics**
Blue-eye trevalla	*Hyperoglyphe antarctica*	Yes	Surplus production
Blue grenadier	*Macruronus novaezelandiae*	Yes	Age- and sex-based
Blue warehou	*Seriolella brama*	Yes	Static
Deepwater sharks	complete list in Patterson et al. (2018)	Yes	Static
Eastern school whiting	*Sillago flindersi*	Yes	Age-based
Elephantfish	*Callorhinchus milii*	Yes	Static
Flathead	*Neoplatycephalus richardsoni* and 4 other species	Yes	Age- and Sex-based
Gemfish	*Rexea solandri*	Yes	Static
Gummy shark	*Mustelus antarcticus*	Yes	Surplus production
Jackass morwong (East)	*Nemadactylus macropterus*	Yes	Age-based
Jackass morwong (West)	*Nemadactylus macropterus*	Yes	Age-based
John dory	*Zeus faber*	Yes	Surplus production
Mirror dory	*Zenopsis nebulosa*	Yes	Surplus production
Ocean jacket	*Nelusetta ayraud*	No	Static
Ocean perch	*Helicolenus barathri, H. percoides*	Yes	Surplus production
Orange roughy (East)	*Hoplostethus atlanticus*	Yes	Age- and sex-based
Orange roughy (South)	*Hoplostethus atlanticus*	Yes	Static
Orange roughy (West)	*Hoplostethus atlanticus*	Yes	Static
Oreodories	complete list in Patterson et al. (2018)	Yes	Static
Pink ling (East)	*Genypterus blacodes*	Yes	Age- and sex-based
Pink ling (West)	*Genypterus blacodes*	Yes	Age- and sex-based
Redfish	*Centroberyx affinis*	Yes	Age- and sex-based
Ribaldo	*Mora moro*	Yes	Static
Royal red prawn	*Haliporoides sibogae*	Yes	Static
Sawshark	*Pristiophorus cirratus, P. nudipinnis*	Yes	Static
School shark	*Galeorhinus galeus*	Yes	Surplus production
Silver trevally	*Pseudocaranx georgianus*	Yes	Static
Silver warehou	*Seriolella punctata*	Yes	Age-based

— Pink ling (West): pers. com. Sandra Curin-Osorio,

— Redfish: Tuck et al. (2017),

— Orange roughy (East): Tuck et al. (2018).

Stock-specific reference points were also calculated from stock assessment outputs. For (sex- and) age-based stock assessments, F_{MSY} was determined by identifying the multiplier μ_s of the current vector of fishing mortality at age $F_{s,g,a,0}$ maximizing total yield at equilibrium (which is the production of yield per recruit and recruitment, both being functions of fishing mortality): $F_{MSY_{s,g,a}} = \mu_s F_{s,g,a,0}$. A single F value for the stock was then estimated as a weighted mean of the values at age:

$$F_{MSY_s} = \frac{1}{nb_g \times \sum_a \delta_{s,a}} \sum_a (\sum_g (F_{MSY_{s,g,a}} \times \frac{N_{s,g,a}}{N_{s,a}}) \times \delta_{s,a}), \quad \text{(C.1)}$$

with $\delta_{s,a} = 1$ if age a is selected for the calculation and 0 otherwise. Youngest and oldest ages having very small contribution to the catch were removed from the selection, under the constraint of remaining ages accounting for at least 90% of the catch. Weighting coefficients $\delta_{s,a}$ are provided in Table C.2.

The same approach was followed to calculate the limit reference point F_{20} associated to an equilibrium spawning stock biomass equal to 20% of its virgin value.

For stocks represented with a global surplus production model, $F_{MSY_s} = r_s$ and $F_{20_s} = r_s \ln(\frac{1}{0.2})$.

Table C.2 – Biological parameters

Stock	Initial biomass (t) B_{2015}	Growth rate (yr^{-1}) r	Carrying capacity (t) K	Initial fishing mortality (yr^{-1}) F_{2015}
Ocean Perch	926	0.69	1190	0.16
Mirror Dory	12849	0.61	13000	0.02
John Dory	1660	0.04	4270	0.04
School shark	6943	0.08	36000	0.03
Blue-eye trevalla	3687	0.08	12375	0.08
Gummy shark	13148	0.38	17369	0.15

Stock	Sex	Age a	Initial abundance $N_{a,2015}$ $(.10^3)$	Natural mortality M_a (yr^{-1})	Init. fish. mortality $F_{a,2015}$ (yr^{-1})	$\overline{F}$ weighting δ_a	Weight in stock w_a (kg/ind)	Weight in landings wl_a (kg/ind)	Maturity Mat_a
School Whiting	Both	0	245083	0.59	0.00	0	0.00	0.01	0.00
		1	134054	0.59	0.01	0	0.02	0.03	0.00
		2	71429	0.59	0.16	1	0.03	0.05	0.21
		3	32363	0.59	0.38	1	0.05	0.07	0.77
		4	10939	0.59	0.51	1	0.07	0.09	0.96
		5	4587	0.59	0.56	1	0.09	0.10	0.99
		6	1050	0.59	0.59	0	0.11	0.11	1.00
		7	345	0.59	0.60	0	0.12	0.12	1.00
		8	116	0.59	0.60	0	0.13	0.13	1.00
		9+	63	0.59	0.60	0	0.14	0.15	1.00
Silver warehou	Both	0	9032	0.30	0.00	0	0.01	0.03	0.00
		1	4381	0.30	0.00	0	0.04	0.15	0.00
		2	3781	0.30	0.02	1	0.23	0.38	0.00
		3	1086	0.30	0.04	1	0.52	0.74	0.02
		4	895	0.30	0.06	1	0.85	1.11	0.54
		5	924	0.30	0.08	1	1.18	1.42	0.90
		6	545	0.30	0.09	1	1.48	1.69	0.98
		7	176	0.30	0.10	1	1.73	1.90	0.99
		8	152	0.30	0.10	1	1.93	2.07	1.00
		9	196	0.30	0.11	1	2.09	2.21	1.00
		10	70	0.30	0.11	1	2.21	2.31	1.00
		11	74	0.30	0.11	1	2.30	2.39	1.00
		12	58	0.30	0.11	1	2.38	2.45	1.00
		13	68	0.30	0.11	1	2.43	2.50	1.00
		14	34	0.30	0.11	1	2.47	2.53	1.00
		15	18	0.30	0.11	1	2.50	2.56	1.00
		16	14	0.30	0.11	1	2.52	2.57	1.00
		17	3	0.30	0.11	0	2.54	2.59	1.00
		18	1	0.30	0.11	0	2.55	2.60	1.00
		19	1	0.30	0.11	0	2.56	2.61	1.00
		20	0	0.30	0.11	0	2.57	2.61	1.00
		21	1	0.30	0.11	0	2.57	2.62	1.00
		22	0	0.30	0.11	0	2.58	2.62	1.00
		23+	0	0.30	0.11	0	2.58	2.62	1.00

Stock	Sex	Age a	Initial abundance $N_{a,2015}$ $(.10^3)$	Natural mortality M_a (yr^{-1})	Init. fish. mortality $F_{a,2015}$ (yr^{-1})	$\overline{F}$ weighting δ_a	Weight in stock w_a (kg/ind)	Weight in landings wl_a (kg/ind)	Maturity Mat_a
Jackass morwong (West)	Both	0	1110	0.15	0.00	0	0.07	0.10	0.00
		1	930	0.15	0.00	0	0.10	0.15	0.01
		2	780	0.15	0.00	0	0.14	0.22	0.05
		3	851	0.15	0.00	1	0.20	0.33	0.24
		4	905	0.15	0.00	1	0.29	0.44	0.63
		5	765	0.15	0.00	1	0.38	0.53	0.85
		6	508	0.15	0.00	1	0.46	0.61	0.94
		7	348	0.15	0.01	1	0.54	0.67	0.97
		8	144	0.15	0.01	1	0.60	0.72	0.98
		9	142	0.15	0.01	1	0.65	0.76	0.99
		10	80	0.15	0.01	1	0.70	0.80	0.99
		11	74	0.15	0.01	1	0.73	0.83	1.00
		12	95	0.15	0.01	1	0.76	0.85	1.00
		13	71	0.15	0.01	1	0.79	0.87	1.00
		14	83	0.15	0.01	1	0.81	0.89	1.00
		15	53	0.15	0.01	1	0.82	0.90	1.00
		16	52	0.15	0.01	1	0.83	0.91	1.00
		17	16	0.15	0.01	1	0.84	0.92	1.00
		18	14	0.15	0.01	1	0.85	0.92	1.00
		19	6	0.15	0.01	0	0.86	0.93	1.00
		20	9	0.15	0.01	0	0.86	0.93	1.00
		21	6	0.15	0.01	0	0.87	0.93	1.00
		22	9	0.15	0.01	0	0.87	0.94	1.00
		23	6	0.15	0.01	0	0.87	0.94	1.00
		24	5	0.15	0.01	0	0.87	0.94	1.00
		25	2	0.15	0.01	0	0.88	0.94	1.00
		26	2	0.15	0.01	0	0.88	0.94	1.00
		27	2	0.15	0.01	0	0.88	0.94	1.00
		28	2	0.15	0.01	0	0.88	0.94	1.00
		29	1	0.15	0.01	0	0.88	0.94	1.00
		30+	7	0.15	0.01	0	0.88	0.94	1.00
Jackass morwong (East)	Both	0	2342	0.15	0.00	0	0.07	0.11	0.00
		1	1978	0.15	0.00	0	0.10	0.16	0.01
		2	1702	0.15	0.00	1	0.14	0.23	0.05
		3	1343	0.15	0.01	1	0.20	0.31	0.24
		4	944	0.15	0.03	1	0.29	0.40	0.63
		5	947	0.15	0.04	1	0.38	0.48	0.85
		6	585	0.15	0.05	1	0.46	0.55	0.94
		7	338	0.15	0.06	1	0.54	0.61	0.97
		8	178	0.15	0.06	1	0.60	0.66	0.98
		9	183	0.15	0.06	1	0.65	0.71	0.99
		10	104	0.15	0.07	1	0.70	0.75	0.99
		11	116	0.15	0.07	1	0.73	0.78	1.00
		12	144	0.15	0.07	1	0.76	0.80	1.00
		13	114	0.15	0.07	1	0.79	0.82	1.00
		14	100	0.15	0.07	1	0.81	0.84	1.00
		15	28	0.15	0.07	1	0.82	0.85	1.00
		16	25	0.15	0.07	1	0.83	0.86	1.00
		17	16	0.15	0.07	1	0.84	0.87	1.00
		18	16	0.15	0.07	1	0.85	0.87	1.00
		19	4	0.15	0.07	0	0.86	0.88	1.00
		20	3	0.15	0.07	0	0.86	0.88	1.00
		21	7	0.15	0.07	0	0.87	0.89	1.00
		22	6	0.15	0.07	0	0.87	0.89	1.00
		23	4	0.15	0.07	0	0.87	0.89	1.00
		24	3	0.15	0.07	0	0.87	0.89	1.00
		25	2	0.15	0.07	0	0.88	0.89	1.00
		26	1	0.15	0.07	0	0.88	0.90	1.00
		27	1	0.15	0.07	0	0.88	0.90	1.00
		28	1	0.15	0.07	0	0.88	0.90	1.00
		29	1	0.15	0.07	0	0.88	0.90	1.00
		30+	2	0.15	0.07	0	0.88	0.90	1.00

Stock	Sex	Age a	Initial abundance $N_{a,2015}$ $(.10^3)$	Natural mortality M_a (yr^{-1})	Init. fish. mortality $F_{a,2015}$ (yr^{-1})	$\overline{F}$ weighting δ_a	Weight in stock w_a (kg/ind)	Weight in landings wl_a (kg/ind)	Maturity Mat_a
Flathead	Female	0	8612	0.27	0.00	0	0.00	0.01	0.00
		1	6436	0.27	0.00	0	0.03	0.10	0.00
		2	4735	0.27	0.03	0	0.16	0.38	0.00
		3	4898	0.27	0.10	1	0.48	0.69	0.57
		4	3453	0.27	0.14	1	0.73	0.93	0.76
		5	2573	0.27	0.17	1	0.99	1.20	0.87
		6	958	0.27	0.18	1	1.26	1.48	0.92
		7	1001	0.27	0.20	1	1.53	1.76	0.95
		8	367	0.27	0.21	1	1.79	2.04	0.97
		9	203	0.27	0.23	1	2.02	2.29	0.98
		10	91	0.27	0.24	1	2.24	2.52	0.98
		11	49	0.27	0.25	1	2.42	2.70	0.99
		12	95	0.27	0.26	1	2.58	2.86	0.99
		13	21	0.27	0.27	0	2.72	2.99	0.99
		14	12	0.27	0.28	0	2.83	3.09	0.99
		15	9	0.27	0.29	0	2.93	3.18	0.99
		16	8	0.27	0.30	0	3.00	3.25	0.99
		17	6	0.27	0.30	0	3.07	3.30	1.00
		18	2	0.27	0.30	0	3.12	3.35	1.00
		19	1	0.27	0.31	0	3.17	3.38	1.00
		20+	1	0.27	0.31	0	3.24	3.45	1.00
	Male	0	8612	0.27	0.00	0	0.00	0.01	0.00
		1	6436	0.27	0.00	0	0.03	0.08	0.00
		2	4738	0.27	0.02	0	0.14	0.31	0.00
		3	4954	0.27	0.08	1	0.39	0.57	0.00
		4	3589	0.27	0.11	1	0.56	0.73	0.00
		5	2767	0.27	0.14	1	0.72	0.88	0.00
		6	1071	0.27	0.15	1	0.88	1.03	0.00
		7	1161	0.27	0.17	1	1.03	1.16	0.00
		8	442	0.27	0.17	1	1.16	1.29	0.00
		9	254	0.27	0.18	1	1.28	1.40	0.00
		10	118	0.27	0.19	1	1.38	1.50	0.00
		11	67	0.27	0.19	1	1.47	1.59	0.00
		12	134	0.27	0.19	1	1.55	1.67	0.00
		13	31	0.27	0.20	0	1.61	1.73	0.00
		14	18	0.27	0.20	0	1.66	1.79	0.00
		15	15	0.27	0.20	0	1.71	1.83	0.00
		16	15	0.27	0.20	0	1.74	1.87	0.00
		17	11	0.27	0.21	0	1.77	1.90	0.00
		18	4	0.27	0.21	0	1.80	1.93	0.00
		19	1	0.27	0.21	0	1.82	1.95	0.00
		20+	3	0.27	0.21	0	1.86	1.98	0.00

Stock	Sex	Age a	Initial abundance $N_{a,2015}$ $(.10^3)$	Natural mortality M_a (yr^{-1})	Init. fish. mortality $F_{a,2015}$ (yr^{-1})	$\overline{F}$ weighting δ_a	Weight in stock w_a (kg/ind)	Weight in landings wl_a (kg/ind)	Maturity Mat_a
Blue grenadier	Female	0	7233	0.17	0.00	0	0.19	0.19	0.00
		1	17581	0.17	0.00	0	0.19	0.27	0.00
		2	22545	0.17	0.01	1	0.33	0.55	0.00
		3	15464	0.17	0.02	1	0.60	0.80	0.04
		4	14231	0.17	0.02	1	0.95	1.14	0.22
		5	8325	0.17	0.02	1	1.39	1.61	0.55
		6	1576	0.17	0.02	1	2.01	2.21	0.77
		7	212	0.17	0.02	1	2.37	2.52	0.81
		8	288	0.17	0.02	1	3.23	3.33	0.84
		9	492	0.17	0.02	1	3.51	3.59	0.84
		10	73	0.17	0.02	1	3.41	3.48	0.84
		11	225	0.17	0.02	1	3.88	3.92	0.84
		12	1319	0.17	0.02	1	3.81	3.85	0.84
		13	45	0.17	0.02	0	3.85	3.89	0.84
		14	32	0.17	0.02	0	3.91	3.94	0.84
		15	9	0.17	0.02	0	3.92	3.95	0.84
		16	10	0.17	0.02	0	4.20	4.21	0.84
		17	13	0.17	0.02	0	4.20	4.21	0.84
		18	47	0.17	0.02	0	4.22	4.23	0.84
		19	76	0.17	0.02	0	4.22	4.23	0.84
		20+	1278	0.17	0.02	0	4.15	4.16	0.84
	Male	0	7233	0.21	0.00	0	0.18	0.18	0.00
		1	16979	0.21	0.00	0	0.18	0.28	0.00
		2	21026	0.21	0.01	1	0.31	0.51	0.00
		3	13938	0.21	0.02	1	0.54	0.71	0.00
		4	12391	0.21	0.02	1	0.82	0.99	0.00
		5	6975	0.21	0.02	1	1.16	1.34	0.00
		6	1242	0.21	0.03	1	1.60	1.75	0.00
		7	156	0.21	0.03	1	1.84	1.95	0.00
		8	193	0.21	0.03	1	2.37	2.39	0.00
		9	315	0.21	0.03	1	2.52	2.51	0.00
		10	45	0.21	0.03	1	2.47	2.46	0.00
		11	134	0.21	0.02	1	2.70	2.67	0.00
		12	750	0.21	0.03	1	2.67	2.63	0.00
		13	24	0.21	0.03	0	2.68	2.65	0.00
		14	17	0.21	0.02	0	2.71	2.67	0.00
		15	5	0.21	0.02	0	2.72	2.68	0.00
		16	5	0.21	0.02	0	2.84	2.79	0.00
		17	6	0.21	0.02	0	2.84	2.79	0.00
		18	20	0.21	0.02	0	2.84	2.79	0.00
		19	30	0.21	0.02	0	2.84	2.79	0.00
		20+	452	0.21	0.02	0	2.81	2.76	0.00

Stock	Sex	Age a	Initial abundance $N_{a,2015}$ (.10^3)	Natural mortality M_a (yr^{-1})	Init. fish. mortality $F_{a,2015}$ (yr^{-1})	$\overline{F}$ weighting δ_a	Weight in stock w_a (kg/ind)	Weight in landings wl_a (kg/ind)	Maturity Mat_a
Pink ling (East)	Female	0	698	0.20	0.00	0	0.01	0.03	0.00
		1	420	0.20	0.00	0	0.07	0.29	0.00
		2	210	0.20	0.03	1	0.32	0.81	0.00
		3	227	0.20	0.10	1	0.87	1.27	0.03
		4	185	0.20	0.12	1	1.42	1.70	0.27
		5	131	0.20	0.11	1	2.04	2.18	0.63
		6	87	0.20	0.09	1	2.69	2.75	0.86
		7	56	0.20	0.08	1	3.34	3.40	0.96
		8	55	0.20	0.07	1	3.98	4.09	0.99
		9	41	0.20	0.06	1	4.58	4.74	1.00
		10	18	0.20	0.06	1	5.14	5.33	1.00
		11	21	0.20	0.06	1	5.65	5.85	1.00
		12	18	0.20	0.06	1	6.11	6.30	1.00
		13	9	0.20	0.06	1	6.53	6.70	1.00
		14	6	0.20	0.06	1	6.90	7.05	1.00
		15	3	0.20	0.06	1	7.22	7.36	1.00
		16	3	0.20	0.06	1	7.50	7.63	1.00
		17	2	0.20	0.06	0	7.75	7.86	1.00
		18	1	0.20	0.06	0	7.97	8.06	1.00
		19	2	0.20	0.06	0	8.15	8.24	1.00
		20	1	0.20	0.06	0	8.31	8.39	1.00
		21	1	0.20	0.06	0	8.45	8.52	1.00
		22	0	0.20	0.06	0	8.57	8.63	1.00
		23	1	0.20	0.06	0	8.68	8.72	1.00
		24	0	0.20	0.06	0	8.77	8.80	1.00
		25	0	0.20	0.06	0	8.84	8.88	1.00
		26	0	0.20	0.06	0	8.91	8.94	1.00
		27	0	0.20	0.06	0	8.96	8.99	1.00
		28	0	0.20	0.06	0	9.01	9.03	1.00
		29	0	0.20	0.06	0	9.05	9.07	1.00
		30+	0	0.20	0.06	0	9.11	9.13	1.00
	Male	0	698	0.20	0.00	0	0.01	0.03	0.00
		1	420	0.20	0.00	0	0.07	0.27	0.00
		2	210	0.20	0.03	1	0.29	0.78	0.00
		3	229	0.20	0.10	1	0.79	1.21	0.00
		4	188	0.20	0.12	1	1.33	1.61	0.00
		5	133	0.20	0.12	1	1.87	2.01	0.00
		6	88	0.20	0.10	1	2.36	2.41	0.00
		7	56	0.20	0.09	1	2.79	2.79	0.00
		8	54	0.20	0.08	1	3.14	3.13	0.00
		9	39	0.20	0.07	1	3.43	3.41	0.00
		10	17	0.20	0.07	1	3.66	3.63	0.00
		11	20	0.20	0.07	1	3.83	3.81	0.00
		12	17	0.20	0.07	1	3.97	3.94	0.00
		13	8	0.20	0.06	1	4.08	4.04	0.00
		14	5	0.20	0.06	1	4.16	4.12	0.00
		15	3	0.20	0.06	1	4.22	4.18	0.00
		16	3	0.20	0.06	1	4.27	4.23	0.00
		17	2	0.20	0.06	0	4.30	4.26	0.00
		18	1	0.20	0.06	0	4.33	4.29	0.00
		19	2	0.20	0.06	0	4.35	4.30	0.00
		20	1	0.20	0.06	0	4.36	4.32	0.00
		21	1	0.20	0.06	0	4.38	4.33	0.00
		22	0	0.20	0.06	0	4.38	4.34	0.00
		23	1	0.20	0.06	0	4.39	4.34	0.00
		24	0	0.20	0.06	0	4.40	4.35	0.00
		25	0	0.20	0.06	0	4.40	4.35	0.00
		26	0	0.20	0.06	0	4.40	4.36	0.00
		27	0	0.20	0.06	0	4.40	4.36	0.00
		28	0	0.20	0.06	0	4.41	4.36	0.00
		29	0	0.20	0.06	0	4.41	4.36	0.00
		30+	0	0.20	0.06	0	4.41	4.36	0.00

Stock	Sex	Age a	Initial abundance $N_{a,2015}$ $(.10^3)$	Natural mortality M_a (yr^{-1})	Init. fish. mortality $F_{a,2015}$ (yr^{-1})	$\overline{F}$ weighting δ_a	Weight in stock w_a (kg/ind)	Weight in landings wl_a (kg/ind)	Maturity Mat_a
Pink ling (West)	Female	0	2309	0.29	0.00	0	0.01	0.03	0.00
		1	2122	0.29	0.00	0	0.08	0.22	0.00
		2	1195	0.29	0.00	1	0.34	0.70	0.00
		3	1250	0.29	0.02	1	0.93	1.30	0.00
		4	803	0.29	0.02	1	1.40	1.81	0.16
		5	544	0.29	0.03	1	1.92	2.38	0.54
		6	329	0.29	0.04	1	2.47	2.99	0.82
		7	232	0.29	0.05	1	3.02	3.56	0.93
		8	157	0.29	0.05	1	3.55	4.09	0.97
		9	97	0.29	0.06	1	4.07	4.57	0.99
		10	67	0.29	0.06	1	4.55	5.01	0.99
		11	57	0.29	0.06	1	5.00	5.41	1.00
		12	42	0.29	0.07	1	5.42	5.78	1.00
		13	20	0.29	0.07	1	5.79	6.11	1.00
		14	13	0.29	0.07	1	6.13	6.41	1.00
		15	7	0.29	0.07	1	6.44	6.67	1.00
		16	3	0.29	0.07	1	6.71	6.91	1.00
		17	3	0.29	0.07	1	6.95	7.11	1.00
		18	2	0.29	0.07	0	7.16	7.30	1.00
		19	1	0.29	0.07	0	7.35	7.46	1.00
		20	2	0.29	0.07	0	7.52	7.60	1.00
		21	1	0.29	0.07	0	7.66	7.73	1.00
		22	1	0.29	0.07	0	7.79	7.84	1.00
		23	1	0.29	0.07	0	7.90	7.93	1.00
		24	1	0.29	0.07	0	7.99	8.01	1.00
		25	0	0.29	0.07	0	8.08	8.09	1.00
		26	0	0.29	0.07	0	8.15	8.15	1.00
		27	0	0.29	0.07	0	8.22	8.20	1.00
		28	0	0.29	0.07	0	8.27	8.25	1.00
		29	0	0.29	0.07	0	8.32	8.29	1.00
		30+	0	0.29	0.07	0	8.40	8.36	1.00
	Male	0	2309	0.29	0.00	0	0.01	0.03	0.00
		1	2122	0.29	0.00	0	0.07	0.21	0.00
		2	1195	0.29	0.00	1	0.30	0.64	0.00
		3	1252	0.29	0.01	1	0.81	1.19	0.00
		4	806	0.29	0.02	1	1.27	1.65	0.00
		5	549	0.29	0.03	1	1.74	2.12	0.00
		6	334	0.29	0.03	1	2.19	2.57	0.00
		7	237	0.29	0.04	1	2.60	2.98	0.00
		8	161	0.29	0.05	1	2.97	3.33	0.00
		9	101	0.29	0.05	1	3.28	3.61	0.00
		10	70	0.29	0.05	1	3.55	3.85	0.00
		11	60	0.29	0.06	1	3.77	4.04	0.00
		12	45	0.29	0.06	1	3.96	4.20	0.00
		13	22	0.29	0.06	1	4.11	4.33	0.00
		14	14	0.29	0.06	1	4.24	4.44	0.00
		15	8	0.29	0.06	1	4.34	4.53	0.00
		16	4	0.29	0.06	1	4.42	4.59	0.00
		17	3	0.29	0.06	1	4.49	4.65	0.00
		18	2	0.29	0.06	0	4.54	4.70	0.00
		19	2	0.29	0.06	0	4.58	4.73	0.00
		20	2	0.29	0.06	0	4.62	4.76	0.00
		21	2	0.29	0.06	0	4.64	4.78	0.00
		22	1	0.29	0.06	0	4.67	4.80	0.00
		23	1	0.29	0.06	0	4.68	4.82	0.00
		24	1	0.29	0.06	0	4.70	4.83	0.00
		25	0	0.29	0.06	0	4.71	4.84	0.00
		26	0	0.29	0.06	0	4.72	4.85	0.00
		27	0	0.29	0.06	0	4.73	4.85	0.00
		28	0	0.29	0.06	0	4.73	4.86	0.00
		29	0	0.29	0.06	0	4.74	4.86	0.00
		30+	0	0.29	0.06	0	4.74	4.87	0.00

Stock	Sex	Age a	Initial abundance $N_{a,2015}$ $(.10^3)$	Natural mortality M_a (yr^{-1})	Init. fish. mortality $F_{a,2015}$ (yr^{-1})	$\overline{F}$ weighting δ_a	Weight in stock w_a (kg/ind)	Weight in landings wl_a (kg/ind)	Maturity Mat_a
Redfish	Female	0	6678	0.10	0.00	0	0.00	0.00	0.00
		1	6033	0.10	0.02	1	0.00	0.01	0.00
		2	26621	0.10	0.04	1	0.01	0.01	0.05
		3	14499	0.10	0.05	1	0.01	0.02	0.29
		4	3868	0.10	0.05	1	0.02	0.02	0.56
		5	1846	0.10	0.06	1	0.02	0.02	0.73
		6	1712	0.10	0.06	1	0.02	0.02	0.82
		7	3663	0.10	0.06	1	0.03	0.03	0.88
		8	1596	0.10	0.06	1	0.03	0.03	0.91
		9	764	0.10	0.06	1	0.03	0.03	0.92
		10	649	0.10	0.06	1	0.03	0.03	0.93
		11	591	0.10	0.06	1	0.03	0.03	0.94
		12	524	0.10	0.06	1	0.03	0.03	0.94
		13	916	0.10	0.06	1	0.03	0.03	0.95
		14	1147	0.10	0.06	1	0.03	0.03	0.95
		15	880	0.10	0.06	1	0.03	0.03	0.95
		16	570	0.10	0.06	1	0.03	0.03	0.95
		17	161	0.10	0.06	1	0.03	0.04	0.95
		18	69	0.10	0.06	0	0.03	0.04	0.95
		19	68	0.10	0.06	0	0.04	0.04	0.95
		20	62	0.10	0.06	0	0.04	0.04	0.95
		21	96	0.10	0.06	0	0.04	0.04	0.95
		22	82	0.10	0.06	0	0.04	0.04	0.95
		23	83	0.10	0.06	0	0.04	0.04	0.95
		24	36	0.10	0.06	0	0.04	0.04	0.95
		25	20	0.10	0.06	0	0.04	0.04	0.95
		26	10	0.10	0.06	0	0.04	0.04	0.95
		27	7	0.10	0.06	0	0.04	0.04	0.95
		28	6	0.10	0.06	0	0.04	0.04	0.95
		29	3	0.10	0.06	0	0.04	0.04	0.95
		30	3	0.10	0.06	0	0.04	0.04	0.95
		31	2	0.10	0.06	0	0.04	0.04	0.95
		32	2	0.10	0.06	0	0.04	0.04	0.95
		33	1	0.10	0.06	0	0.04	0.04	0.95
		34	1	0.10	0.06	0	0.04	0.04	0.95
		35	1	0.10	0.06	0	0.04	0.04	0.95
		36	1	0.10	0.06	0	0.04	0.04	0.95
		37	1	0.10	0.06	0	0.04	0.04	0.95
		38	0	0.10	0.06	0	0.04	0.04	0.95
		39+	2	0.10	0.06	0	0.04	0.04	0.95
	Male	0	6678	0.10	0.00	0	0.00	0.00	0.00
		1	6034	0.10	0.02	1	0.00	0.01	0.00
		2	26727	0.10	0.03	1	0.01	0.01	0.00
		3	14656	0.10	0.05	1	0.01	0.01	0.00
		4	3933	0.10	0.05	1	0.01	0.02	0.00
		5	1887	0.10	0.05	1	0.02	0.02	0.00
		6	1763	0.10	0.06	1	0.02	0.02	0.00
		7	3807	0.10	0.06	1	0.02	0.02	0.00
		8	1672	0.10	0.06	1	0.02	0.02	0.00
		9	805	0.10	0.06	1	0.02	0.02	0.00
		10	689	0.10	0.06	1	0.02	0.03	0.00
		11	634	0.10	0.06	1	0.03	0.03	0.00
		12	569	0.10	0.06	1	0.03	0.03	0.00
		13	1003	0.10	0.06	1	0.03	0.03	0.00
		14	1271	0.10	0.06	1	0.03	0.03	0.00
		15	985	0.10	0.06	1	0.03	0.03	0.00
		16	642	0.10	0.06	1	0.03	0.03	0.00
		17	183	0.10	0.06	1	0.03	0.03	0.00
		18	79	0.10	0.06	0	0.03	0.03	0.00
		19	79	0.10	0.06	0	0.03	0.03	0.00
		20	71	0.10	0.06	0	0.03	0.03	0.00
		21	110	0.10	0.06	0	0.03	0.03	0.00
		22	95	0.10	0.06	0	0.03	0.03	0.00
		23	95	0.10	0.06	0	0.03	0.03	0.00

Stock	Sex	Age a	Initial abundance $N_{a,2015}$ $(.10^3)$	Natural mortality M_a (yr^{-1})	Init. fish. mortality $F_{a,2015}$ (yr^{-1})	$\overline{F}$ weighting δ_a	Weight in stock w_a (kg/ind)	Weight in landings wl_a (kg/ind)	Maturity Mat_a
		24	42	0.10	0.06	0	0.03	0.03	0.00
		25	23	0.10	0.06	0	0.03	0.03	0.00
		26	12	0.10	0.06	0	0.03	0.03	0.00
		27	8	0.10	0.06	0	0.03	0.03	0.00
		28	7	0.10	0.06	0	0.03	0.03	0.00
		29	4	0.10	0.06	0	0.03	0.03	0.00
		30	3	0.10	0.06	0	0.03	0.03	0.00
		31	3	0.10	0.06	0	0.03	0.03	0.00
		32	2	0.10	0.06	0	0.03	0.03	0.00
		33	1	0.10	0.06	0	0.03	0.03	0.00
		34	1	0.10	0.06	0	0.03	0.03	0.00
		35	1	0.10	0.06	0	0.03	0.03	0.00
		36	1	0.10	0.06	0	0.03	0.03	0.00
		37	1	0.10	0.06	0	0.03	0.03	0.00
		38	1	0.10	0.06	0	0.03	0.03	0.00
		39+	2	0.10	0.06	0	0.03	0.03	0.00
Orange roughy (East)	Female	0	3642	0.04	0.00	0	0.02	0.03	0.00
		1	3446	0.04	0.00	0	0.04	0.05	0.00
		2	3256	0.04	0.00	0	0.06	0.09	0.00
		3	3072	0.04	0.00	0	0.08	0.20	0.00
		4	2893	0.04	0.00	0	0.11	0.27	0.00
		5	2719	0.04	0.00	0	0.15	0.36	0.00
		6	2550	0.04	0.00	0	0.19	0.47	0.00
		7	2386	0.04	0.00	0	0.23	0.59	0.00
		8	2235	0.04	0.00	0	0.27	0.72	0.00
		9	2099	0.04	0.00	0	0.32	0.86	0.00
		10	1974	0.04	0.00	0	0.36	1.00	0.00
		11	1858	0.04	0.00	0	0.41	1.13	0.00
		12	1764	0.04	0.00	0	0.46	1.22	0.00
		13	1694	0.04	0.00	0	0.51	1.28	0.00
		14	1639	0.04	0.00	0	0.57	1.32	0.00
		15	1590	0.04	0.00	0	0.62	1.35	0.00
		16	1546	0.04	0.00	0	0.67	1.38	0.00
		17	1509	0.04	0.00	0	0.72	1.40	0.00
		18	1476	0.04	0.00	0	0.77	1.42	0.01
		19	1449	0.04	0.00	0	0.82	1.43	0.01
		20	1439	0.04	0.00	1	0.86	1.45	0.03
		21	1468	0.04	0.00	1	0.91	1.47	0.04
		22	1559	0.04	0.00	1	0.95	1.48	0.06
		23	1619	0.04	0.00	1	1.00	1.50	0.09
		24	1627	0.04	0.00	1	1.04	1.51	0.12
		25	1611	0.04	0.00	1	1.08	1.52	0.16
		26	1563	0.04	0.00	1	1.12	1.54	0.20
		27	1502	0.04	0.00	1	1.16	1.55	0.24
		28	1443	0.04	0.00	1	1.20	1.57	0.28
		29	1384	0.04	0.00	1	1.23	1.58	0.32
		30	1327	0.04	0.01	1	1.27	1.60	0.36
		31	1340	0.04	0.01	1	1.30	1.61	0.40
		32	1251	0.04	0.01	1	1.33	1.63	0.44
		33	1123	0.04	0.01	1	1.36	1.64	0.47
		34	982	0.04	0.01	1	1.39	1.65	0.51
		35	839	0.04	0.01	1	1.42	1.67	0.54
		36	728	0.04	0.01	1	1.44	1.68	0.56
		37	640	0.04	0.01	1	1.47	1.69	0.59
		38	570	0.04	0.01	1	1.49	1.71	0.62
		39	513	0.04	0.01	1	1.51	1.72	0.64
		40	462	0.04	0.01	1	1.53	1.73	0.66
		41	411	0.04	0.01	1	1.55	1.74	0.68
		42	358	0.04	0.01	1	1.57	1.75	0.69
		43	306	0.04	0.01	1	1.59	1.76	0.71
		44	259	0.04	0.01	1	1.61	1.78	0.72
		45	217	0.04	0.01	1	1.63	1.79	0.74
		46	182	0.04	0.01	1	1.64	1.80	0.75

Stock	Sex	Age a	Initial abundance $N_{a,2015}$ $(.10^3)$	Natural mortality M_a (yr^{-1})	Init. fish. mortality $F_{a,2015}$ (yr^{-1})	$\overline{F}$ weighting δ_a	Weight in stock w_a (kg/ind)	Weight in landings wl_a (kg/ind)	Maturity Mat_a
Orange roughy (East)		47	153	0.04	0.01	1	1.66	1.80	0.76
		48	128	0.04	0.01	1	1.67	1.81	0.77
		49	108	0.04	0.01	1	1.68	1.82	0.78
		50	92	0.04	0.01	1	1.70	1.83	0.79
		51	78	0.04	0.01	1	1.71	1.84	0.80
		52	66	0.04	0.01	1	1.72	1.85	0.80
		53	57	0.04	0.01	1	1.73	1.85	0.81
		54	49	0.04	0.01	1	1.74	1.86	0.81
		55	42	0.04	0.01	1	1.75	1.87	0.82
		56	37	0.04	0.01	1	1.76	1.87	0.83
		57	33	0.04	0.01	1	1.77	1.88	0.83
		58	29	0.04	0.01	1	1.77	1.88	0.83
		59	27	0.04	0.01	1	1.78	1.89	0.84
		60	24	0.04	0.01	1	1.79	1.89	0.84
		61	23	0.04	0.01	1	1.80	1.90	0.85
		62	21	0.04	0.01	1	1.80	1.90	0.85
		63	20	0.04	0.01	1	1.81	1.91	0.85
		64	19	0.04	0.01	1	1.81	1.91	0.85
		65	18	0.04	0.01	1	1.82	1.92	0.86
		66	16	0.04	0.01	1	1.82	1.92	0.86
		67	15	0.04	0.01	1	1.83	1.92	0.86
		68	14	0.04	0.01	1	1.83	1.93	0.86
		69	13	0.04	0.01	1	1.84	1.93	0.86
		70	12	0.04	0.01	1	1.84	1.93	0.87
		71	11	0.04	0.01	1	1.85	1.94	0.87
		72	11	0.04	0.01	1	1.85	1.94	0.87
		73	10	0.04	0.01	1	1.85	1.94	0.87
		74	10	0.04	0.01	1	1.86	1.94	0.87
		75	9	0.04	0.01	1	1.86	1.95	0.87
		76	9	0.04	0.01	1	1.86	1.95	0.87
		77	8	0.04	0.01	1	1.86	1.95	0.88
		78	8	0.04	0.01	1	1.87	1.95	0.88
		79+	121	0.04	0.01	1	1.87	1.95	0.88
	Male	0	3642	0.04	0.00	0	0.02	0.03	0.00
		1	3446	0.04	0.00	0	0.04	0.05	0.00
		2	3256	0.04	0.00	0	0.06	0.09	0.00
		3	3072	0.04	0.00	0	0.09	0.20	0.00
		4	2893	0.04	0.00	0	0.12	0.27	0.00
		5	2719	0.04	0.00	0	0.15	0.36	0.00
		6	2550	0.04	0.00	0	0.19	0.47	0.00
		7	2386	0.04	0.00	0	0.23	0.58	0.00
		8	2235	0.04	0.00	0	0.27	0.71	0.00
		9	2099	0.04	0.00	0	0.32	0.85	0.00
		10	1974	0.04	0.00	0	0.36	0.99	0.00
		11	1858	0.04	0.00	0	0.41	1.12	0.00
		12	1764	0.04	0.00	0	0.46	1.21	0.00
		13	1694	0.04	0.00	0	0.51	1.27	0.00
		14	1639	0.04	0.00	0	0.56	1.31	0.00
		15	1590	0.04	0.00	0	0.61	1.34	0.00
		16	1546	0.04	0.00	0	0.66	1.36	0.00
		17	1509	0.04	0.00	0	0.71	1.38	0.00
		18	1476	0.04	0.00	0	0.76	1.40	0.00
		19	1449	0.04	0.00	0	0.81	1.42	0.00
		20	1439	0.04	0.00	1	0.86	1.43	0.00
		21	1468	0.04	0.00	1	0.90	1.45	0.00
		22	1559	0.04	0.00	1	0.95	1.46	0.00
		23	1619	0.04	0.00	1	0.99	1.48	0.00
		24	1627	0.04	0.00	1	1.03	1.49	0.00
		25	1611	0.04	0.00	1	1.07	1.50	0.00
		26	1563	0.04	0.00	1	1.11	1.52	0.00
		27	1502	0.04	0.00	1	1.15	1.53	0.00
		28	1443	0.04	0.00	1	1.18	1.55	0.00
		29	1384	0.04	0.00	1	1.22	1.56	0.00
		30	1327	0.04	0.01	1	1.25	1.58	0.00

Stock	Sex	Age a	Initial abundance $N_{a,2015}$ $(.10^3)$	Natural mortality M_a (yr^{-1})	Init. fish. mortality $F_{a,2015}$ (yr^{-1})	$\overline{F}$ weighting δ_a	Weight in stock w_a (kg/ind)	Weight in landings wl_a (kg/ind)	Maturity Mat_a
Orange roughy (East)		31	1340	0.04	0.01	1	1.28	1.59	0.00
		32	1251	0.04	0.01	1	1.31	1.60	0.00
		33	1123	0.04	0.01	1	1.34	1.62	0.00
		34	982	0.04	0.01	1	1.37	1.63	0.00
		35	839	0.04	0.01	1	1.40	1.64	0.00
		36	728	0.04	0.01	1	1.42	1.66	0.00
		37	640	0.04	0.01	1	1.45	1.67	0.00
		38	570	0.04	0.01	1	1.47	1.68	0.00
		39	513	0.04	0.01	1	1.49	1.69	0.00
		40	462	0.04	0.01	1	1.51	1.71	0.00
		41	411	0.04	0.01	1	1.53	1.72	0.00
		42	358	0.04	0.01	1	1.55	1.73	0.00
		43	306	0.04	0.01	1	1.57	1.74	0.00
		44	259	0.04	0.01	1	1.59	1.75	0.00
		45	217	0.04	0.01	1	1.60	1.76	0.00
		46	182	0.04	0.01	1	1.62	1.77	0.00
		47	153	0.04	0.01	1	1.63	1.78	0.00
		48	128	0.04	0.01	1	1.65	1.79	0.00
		49	108	0.04	0.01	1	1.66	1.79	0.00
		50	92	0.04	0.01	1	1.67	1.80	0.00
		51	78	0.04	0.01	1	1.68	1.81	0.00
		52	66	0.04	0.01	1	1.69	1.82	0.00
		53	57	0.04	0.01	1	1.70	1.83	0.00
		54	49	0.04	0.01	1	1.71	1.83	0.00
		55	42	0.04	0.01	1	1.72	1.84	0.00
		56	37	0.04	0.01	1	1.73	1.84	0.00
		57	33	0.04	0.01	1	1.74	1.85	0.00
		58	29	0.04	0.01	1	1.75	1.86	0.00
		59	27	0.04	0.01	1	1.75	1.86	0.00
		60	24	0.04	0.01	1	1.76	1.87	0.00
		61	23	0.04	0.01	1	1.77	1.87	0.00
		62	21	0.04	0.01	1	1.77	1.87	0.00
		63	20	0.04	0.01	1	1.78	1.88	0.00
		64	19	0.04	0.01	1	1.79	1.88	0.00
		65	18	0.04	0.01	1	1.79	1.89	0.00
		66	16	0.04	0.01	1	1.80	1.89	0.00
		67	15	0.04	0.01	1	1.80	1.89	0.00
		68	14	0.04	0.01	1	1.81	1.90	0.00
		69	13	0.04	0.01	1	1.81	1.90	0.00
		70	12	0.04	0.01	1	1.81	1.90	0.00
		71	11	0.04	0.01	1	1.82	1.91	0.00
		72	11	0.04	0.01	1	1.82	1.91	0.00
		73	10	0.04	0.01	1	1.82	1.91	0.00
		74	10	0.04	0.01	1	1.83	1.91	0.00
		75	9	0.04	0.01	1	1.83	1.91	0.00
		76	9	0.04	0.01	1	1.83	1.92	0.00
		77	8	0.04	0.01	1	1.83	1.92	0.00
		78	8	0.04	0.01	1	1.84	1.92	0.00
		79+	121	0.04	0.01	1	1.84	1.92	0.00

Stock	Virgin recruitment $(.10^3)$ R_0	Virgin biomass (t) SSB_0	Steepness h	Recruitment shift μ_R	Recruitment deviation σ_R
School Whiting	273325	7546	0.75	0	0.35
Silver Warehou	11838	18949	0.75	-0.63	0.7
Jackass Morwong (West)	1207	2742	0.7	0	0.7
Jackass Morwong (East)	3103	7046	0.7	0	0.7
Flathead	21977	23100	0.62	0	0.4
Pink ling (East)	1845	7669	0.75	0	0.7
Pink ling (West)	4891	7143	0.75	0	0.7
Redfish	135929	12004	0.75	-0.91	0.7
Orange roughy (East)	8763	41634	0.75	0	0.7

C.2 Fishing activity and catch module

Metiers represented in the model were derived from a multivariate clustering analysis of catch data detailed in Appendix D. Catch and fishing effort at the vessel and metier level were calculated from SESSF logbook data.

C.3 ITQ market module

Quota lease prices for the species not explicitely under TAC management in the simulations were set as fixed parameters and estimated from data collected by the Australian Fisheries Management Authority since July 2017. Transactions between related entities (i.e. related through some type of control or ownership) were removed from the dataset to avoid bias in the estimation. The ratio of the yearly median lease price to the species' ex-vessel price was calculated for the period 2017-2018 and used to estimate a yearly lease price for 2015. Values of those ratios are given in Table C.5.

C.4 Fish price module

Ex-vessel prices for the various species in 2015 were obtained for each sector from Australian Fisheries Statistics (Mobsby, 2018) and are provided in Table C.6. Own- and cross-price flexibilities estimated for the SESSF by Bose (2004) were used to calibrate fish price dynamics in Chapter 5. Only significant coefficients were kept.

C.5 Economic module

Table C.7 summarizes the estimated cost structures for the four gear types in the SESSF. They are based on ABARES economic surveys from 2015 (Bath et al., 2018), which reports financial performance of the average boat for the Commonwealth Trawl Sector (CTS) and the Gillnet, Hook and Trap Sector (GHTS). Personal communication from ABARES allowed to break down sectoral costs per main gear type (trawl and Danish seine for the CTS and gillnet and hooks for the GHTS). Depreciation costs (C_{dep}) per sector were estimated by ABARES using the diminishing value method based on current replacement value and age of the items. Following ABARES methodology, the opportunity cost of capital (C_{opp}) per sector was estimated at 7% per year of capital value. This interest rate represents the long-term average rate of return that could be earned on an investment elsewhere. Crew share ($cshr$) is expressed as a percentage of

Table C.5 – Ratio between the median lease price and the ex-vessel price for species under quota in the SESSF between 2017 and 2018.
Source: AFMA

Species	Lease price/Fish price
Blue eye Trevalla	0.22
Blue Grenadier	0.05
Blue Warehou	0.11
Flathead	0.29
Gemfish East	0.17
Gemfish West	0.22
Gummy Shark	0.31
Jackass Morwong East	0.17
Jackass Morwong West	0.17
John Dory	0.04
Mirror Dory	0.15
Ocean Perch Inshore	0.16
Ocean Perch Offshore	0.16
Orange Roughy Cascade	confidential
Orange Roughy East	0.21
Orange Roughy South	0.24
Orange Roughy West	0.17
Pink Ling East	0.39
Pink Ling West	0.39
Redfish	0.03
Royal Red Prawn	confidential
Saw Shark	0.25
School Shark	0.54
School Whiting	0.12
Silver Trevally	0.02
Silver Warehou	0.15

income, variable costs (i.e. fuel costs (C_{fuel}) and other variable costs ($C_{var_{oth}}$)) were calculated per fishing day, and fixed costs and capital costs (depreciation and opportunity costs) estimated to be function of the vessel's income. Assuming fixed and capital costs to be function of the vessel's income allowed to account for the variability in vessels characteristics within sectors (e.g vessel length or engine power which are important determinants of fixed costs and often correlated to the level of catch and derived income). It also enabled to address the fact that some vessels are not only employed to fish in the SESSF (some also operate in state waters or other Commonwealth fisheries) and therefore their fixed costs and capital costs should be redistributed to each fishery accordingly. The ex-vessel price for the modelled species in 2015 was derived from Mobsby (2018) and values are given in Table C.6.

Table C.6 – Ex-vessel price of the modelled species in the SESSF in 2015.
Source: Mobsby (2018)

Species	Fish price (AU$/kg)	
	CTS	GHTS
Blue eye Trevalla	8.33	9.11
Blue Grenadier	1.30	1.30
Blue Warehou	3.07	3.00
Deepwater Shark	3.64	3.64
Flathead	6.18	6.18
Frostfish	3.50	3.50
Gemfish	2.43	2.43
Gummy Shark	6.29	6.29
Jackass Morwong	3.36	3.36
John Dory	8.66	8.66
King Dory	3.50	3.50
Mirror Dory	3.15	3.15
Ocean Jackets	1.72	1.72
Ocean Perch	5.08	5.08
Orange Roughy	5.59	5.59
Oreos	3.50	3.50
Other species	3.50	3.50
Pink Ling	5.73	5.73
Redfish	3.43	3.43
Ribaldo	3.50	4.35
Royal Red Prawn	4.01	4.01
Saw Shark	1.91	1.91
School Shark	5.99	5.99
School Whiting	3.05	3.05
Silver Trevally	4.49	4.49
Silver Warehou	1.15	1.15
Squids	3.79	3.79

Crew numbers in the fishery were found to be positively correlated with the amount of daily landings as reported in Table C.8.

Table C.7 – Cost structures for the four main gear types in the SESSF.
Source: Bath et al. (2018)

sector	$cshr$ (% income)	C_{fuel} (AU$/day)	$C_{var_{oth}}$ (AU$/day)	C_{fix} (% income)	C_{dep} (% income)	C_{opp} (% income)
TW	26	2513	2845	11	3	2
DS	46	436	1321	13	5	2
GN	42	564	403	20	7	9
HK	42	590	1208	18	6	5

Table C.8 – Estimation of crew numbers in the SESSF.
Source: pers. comm. ABARES

	Daily landings (kg/day)		
	< 350	$[350; 700[$	$\geqslant 700$
Crew numbers	2	3	4

APPENDIX D

METIER AND FLEET DEFINITION IN THE AUSTRALIAN SOUTHERN AND EASTERN SCALEFISH AND SHARK FISHERY

The aim of this analysis was twofold:

— define metiers for the South East Shark and Scalefish Fishery (excl. the Great Australian Bight and East Coast Deepwater Trawl sectors),

— define fleets for the South East Shark and Scalefish Fishery (excl. the Great Australian Bight and East Coast Deepwater Trawl sectors).

The metier analysis builds on the definition used by the European Data Collection Framework (EU, 2008), which is "a group of fishing operations targeting a similar (assemblage of) species, using similar gear, during the same period of the year and/or within the same area and which are characterised by a similar exploitation pattern". Multivariate statistical methods have often been used to identify groups of trips with similar landings compositions (Marchal, 2008; Deporte et al., 2012; Ziegler, 2012; Ono et al., 2018). A similar approach has been undertaken here, but on the species composition of the landed value rather than the landed weight since targeting is most likely to be driven by the value of landings than their weight.

A fleet, on the other hand, is defined here as a group of vessels showing similar fishing practices, in other words showing similar effort allocations among metiers. Here again, statistical clustering methods were used to identify groups of vessels showing similar fishing practices.

D.1 Material and Methods

D.1.1 Data

The species composition of landed catch was derived from the fishery's logbook data. In order to have a recent description of the fishery only records from calendar years 2012 to 2017 were used. Annual fish prices were retrieved from the Australian fisheries and aquaculture statistics report Mobsby (2018).

D.1.2 Definition of metiers

The definition of metiers consisted in 2 main steps:

1. A clustering of fishing hauls based on landings profiles (in value) using multivariate statistical methods,
2. A post-hoc refinement of the clusters identified by the clustering algorithm in order to: (2.1) group clusters that do not show significant differences in terms of value profile, and (2.2) make sure that clusters reflect an intended targeting based on expertise from members of the fishing industry.

Step 1: Statistical clustering:
The analysis follows the first 3 steps of the workflow developed by Deporte et al. (2012) and integrated in the R package *vmstools*:

1.1: Identification of the main species in order to reduce the dataset to these key species. Species were selected so that at least 90% of the total value and 70% of each haul value was represented in the final dataset to cluster.

1.2: Investigation of running a Principal Component Analysis (PCA) on the dataset prior to the clustering. The reason for applying a PCA to the dataset is that it reduces the number of variables in the dataset to cluster (initially, the variables are the individual species but when running a PCA variables become the relevant factors from the PCA).

1.3: Running a selection of clustering algorithms and dissimilarity measures. In order for large and small hauls to be given equal importance, the clustering was done on the species contribution to the haul value (the sum of which, across all species, equals 1), rather than absolute values of catch. The input to the clustering algorithm is therefore a 2D matrix M, with rows referring to logbook events (index le), and columns to species (index sp), and entry $M_{le,sp}$ defined as: $M_(le, sp) = \frac{p_{sp} * L_{le,sp}}{\sum_{sp} p_{sp} \times L_{le,sp}}$, with p_{sp} the price of species sp, and $L_{le,sp}$ the landed catch of species sp in logbook event le.

For the sake of brevity, only the most satisfactory results will be presented in this document. They were obtained by running the CLARA algorithm on the original dataset (i.e. without prior PCA) with the Euclidean distance used as the dissimilarity metric. Nonetheless, alternative choices for steps 1.2 and 1.3 were explored and the reason for discarding them provided hereafter:

— Step 1.2: The clustering analysis was performed on four different datasets: the original dataset with species as variables, and three PCA-transformed datasets respectively keeping factors representing 70% of the explained variance, 80% of the explained variance, and as determined by the Scree test. The Scree test retains fewer axes than thresholds of 70% or 80% of explained variance, which then results in fewer clusters. The clustering of PCA-transformed data also leads to fewer clusters than the clustering of original data and failed at identifying rare (but existing) metiers. This for instance concerns the targeting of orange roughy since the fishery has only reopened in 2015 in some areas. In order not to miss rare but important metiers throughout the clustering process, we decided to keep working with the original dataset, without prior dimension reduction.

— Step 1.3: Three clustering methods were implemented in the *vmstools* package: HAC, K-means and PAM (or its adaptation for the analysis of large datasets: CLARA). HAC and CLARA algorithms gave similar results, unlike K-means (in line with tests run by Deporte et al. (2012)). The CLARA algorithm was finally chosen since more efficient on large datasets. Sensitivity of the results to the distance metric used to cluster the data was also investigated. The Euclidean and Manhattan distances are two commonly used dissimilarity measures in clustering analyses. We observed that the Euclidean distance allowed identifying single species clusters (e.g. royal red prawn trawling) which did not emerge with the Manhattan distance. Since those single-species metiers are important for the fishery, we decided to work with the Euclidean distance rather than the Manhattan distance.

Multispecies technical interactions are also sensitive to the scale at which data is analysed and working with spatially or temporally aggregated data can only increase the level of perceived interactions. Therefore, data was kept at the finest scale as possible, i.e. at the haul level. Not aggregating data however requires the clustering algorithm to work on a large dataset which has been solved by using a variant of the PAM algorithm suited to the analysis of large datasets (CLARA). In addition, separate clustering analyses were run for each of the 5 groups specified in Table D.1 which reduced the size of the dataset to cluster and resulted in more relevant clusters. The East-West boundary is the 147-degree meridian (also used in the management of certain stocks).

Table D.1 – Gear classification and groups used for clustering

Sector	Gear	Abbreviation in logbook	Group for clustering	Nb. of events in 2012-2017
CTS	Otter trawl	TDO,TW	Trawl - East	59383
			Trawl - West	21044
	Danish seine	DS	Danish seine	54984
GHTS	Gillnet	GN	Gillnet	42528
	Automatic longline	AL	Hooks	17042
	Bottomline	BL		
	Dropline	DL		

Step 2: Refinement of clusters: This refinement phase consisted in two phases:

2.1: The grouping of clusters that had similar value profiles

2.2: Based on expert knowledge of the fishery, clusters whose landed value was dominated by species that are not identified as targeted species by members of the fishing industry were assigned to a "mixed" metier as they are likely to be the result of chance than intended targeting.

D.1.3 Definition of fleets

A similar approach using statistical clustering was used to define fleets for the SESSF. As metiers were clusters of hauls having similar landings profiles, fleets are clusters of vessels displaying similar fishing patterns (or metier profiles). A vessel's fishing pattern as the allocation of its annual fishing effort across the different metiers. Input of the clustering algorithm is therefore a 2D matrix F, with rows referring to vessels (index v), and columns to metiers (index m), and entry $F_{v,m}$ defined as: $F_{v,m} = \frac{Eff_{v,m}}{\sum_m Eff_{v,m}}$, with $Eff_{v,m)}$ the number of hauls vessel v operated in metier m.

This analysis builds on the allocation of hauls to metiers carried out previously. To be consistent with the methods used in the metier identification, the PAM algorithm was used to identify fleets (CLARA was not necessary as the sample size is now smaller, i.e. the number of vessels). The clustering algorithm was run separately for the 4 sectors: Trawl (East and West regions were not treated separately as some trawlers operate across the 2 zones), Danish seine, Gillnet and Hooks.

Similarly to the metier analysis, similar clusters were merged as a post-hoc refinement (step 2). This step was specifically important to ensure the inter-annual stability of fleets and not having vessels move between fleets from one year to another without having significantly changed their fishing practices.

D.2 Results

D.2.1 Metiers

Table D.2 describes the species composition of the clusters identified at Step 1 and provides the number of hauls in each cluster. It also specifies how each cluster has then been attributed to a metier through Steps 2.1 and 2.2. After Step 1, respectively 15, 6, 6, 7 and 6 clusters were identified for Trawl East, Trawl West, Danish seine, Gillnet and Hooks. After Step 2, those cluster have been merged into respectively 10, 6, 3, 1 and 4 metiers. When mixed clusters were found across a wide range of depths (e.g. cluster 3 of Trawl West), their hauls were attributed to the mixed shelf when operating at depths smaller than 250m and to the mixed slope metier when operating at depths greater than 250m.

D.2.2 Fleets

Table D.3 summarizes the fleet identification derived from the clustering analysis of vessels' fishing patterns. After Step 1, respectively 6, 10, 9 and 6 clusters were identified for Trawl, Danish seine, Gillnet and Hooks. After Step 2, those cluster have been merged into respectively 5, 2, 1 and 5 fleets.

Table D.2 – Description of clusters and attribution to metiers

Sector	Cluster Step 1	Species composition: Main species	Species composition: Secondary species	Depth zone	Zone	Main season	% hauls	Cluster Step 2.1	Cluster Step 2.2 (metier)	Remarks
Trawl East	13	Flathead (87%)		Shelf	10;20;30	All year	23	Flathead	Flathead	
	1	Flathead (61%)	John dory, squids, latchet, ocean jackets, other species	Shelf	10;20;30	All year	24	Flathead	Flathead	
	2	Royal Red Prawn (87%)	Mirror dory, other species	Slope	Ulladulla	All year	3	Royal Red Prawn	Royal Red Prawn	
	3		Flathead, John dory, squids, other species	Shelf	10;20;30	All year	12	Mixed Shelf	Mixed Shelf	
	4	Ocean jackets (43%)	Flathead,John dory	Shelf	10;20	Except winter	5	Ocean jackets	Ocean jackets	
	5	Jackass Morwong (53%)	Flathead, ocean jackets, squids, other species	Shelf	20	Summer	3	Jackass morwong	Jackass morwong	
	7	Silver trevally (62%)	Flathead, ocean jackets, other species	Shelf	10	Dec-Jan; April-June	3	Silver Trevally - Flathead	Mixed Shelf	Silver Trevally not targeted
	11	School whiting (57%)	Flathead, other species	Shelf	10	April - May	2	School Whiting	Mixed Shelf	School Whiting not targeted
	12		Other species (65%), flathead, squids	Shelf-Slope	10;20;30	All year	3	Mixed (shelf-slope)	Mixed (shelf-slope)	
	9	Squids (60%)	Flathead	Shelf-Slope	10;20;30	Summer - Autumn	5	Squids	Squids	
	10	Pink Ling (69%)	ocean perch offshore, blue grenadier, mirror dory	Slope	20	Not in summer	7	Pink Ling	Pink Ling	
	6	Ocean perch offshore (59%)	Pink ling, Mirror dory, gemfish, other species	Slope	10	July - October	3	Ocean perch - Pink Ling	Mixed Slope	Ocean Perch not targeted
	8		Blue grenadier, mirror dory, pink ling, ocean perch offshore	Slope	10;20;30	All year	6	Mixed Slope	Mixed Slope	
	14	Frostfish (60%)	mirror dory, pink ling	Slope	10;20;30;	Winter	2	Frostfish	Frostfish	
	15	Orange roughy (90%)	oreos	Deep	St Helens	Not in summer	1	Orange Roughy	Orange Roughy	

Description of clusters and attribution to metiers, continued

Sector	Cluster Step 1	Species composition		Depth zone	Zone	Main season	% hauls	Cluster Step 2.1	Cluster Step 2.2 (metier)	Remarks
		Main species	Secondary species							
Trawl West	4		deepwater flathead, squids, silver warehou, latchet, other species	Shelf-Slope	50	Winter - Spring	25	Mixed (shelf-slope)	Mixed (shelf-slope)	
	3	Squids (58%)	silver warehou, mirror dory, pink ling, blue grenadier	Shelf-Slope	50	Summer - Autumn	13	Squids	Squids	
	2	Pink ling (75%)	blue grenadier, king dory, silver warehou	Slope	40	Spring	13	Pink Ling	Pink Ling	
	5	Blue grenadier (79%)	Pink ling	Slope	40-50	Winter	14	Blue Grenadier	Blue Grenadier	
	1		blue grenadier, pink ling	Slope	40-50	Summer - Autumn	25	Mixed slope	Mixed slope	
	6		deepwater sharks, oreos, ribaldo, orange roughy	Deep	40-50	Spring - Summer	10	Deepwater basket	Deepwater basket	
Danish seine	1	Flathead (48%)	gummy shark, other species	Shelf	20	Drop in summer	13	Flathead	Flathead	
	2	Flathead (85%)		Shelf	20	Drop in summer	22	Flathead	Flathead	
	3	Flathead (98%)		Shelf	20	Spring - Summer	42	Flathead	Flathead	
	4	School whiting (91%)		Shelf	20	Drop in summer	12	School Whiting	School Whiting	
	5	School whiting (58%)	Flathead, other species	Shelf	20	Drop in summer	9	School Whiting	School Whiting	
	6		Other species (72%), school whiting, flathead	Shelf	20	Winter-Spring	2	Mixed	Mixed	

Description of clusters and attribution to metiers, continued

Sector	Cluster Step 1	Species composition		Depth zone	Zone	Main season	% hauls	Cluster Step 2.1	Cluster Step 2.2 (metier)	Remarks
		Main species	Secondary species							
Gillnet	5	Gummy shark (98%)		Shelf	50-60	Spring-Summer	33	Gummy Shark	Gummy Shark	
	1	Gummy shark (92%)	saw shark	Shelf	50-60	Spring-Summer	32	Gummy Shark	Gummy Shark	
	2	Gummy shark (76%)	saw shark	Shelf	50-60	Spring-Summer	18	Gummy Shark	Gummy Shark	
	4	Gummy shark (62%)	school shark	Shelf	50-60	Autumn	6	Gummy Shark	Gummy Shark	
	6	School shark (67%)	gummy shark	Shelf	50-60	Autumn	4	School Shark	Gummy Shark	school sharks not targeted
	7		other species, gummy shark	Shelf	50-60	More in winter	4	Mixed	Gummy Shark	not a specific metier
	3		saw shark, gummy shark, boarfishes	Shelf	60	More in winter	3	Mixed	Gummy Shark	not a specific metier
Hooks	2	Gummy shark (97%)		Shelf	50	All year	44	Gummy Shark	Gummy Shark	
	1	Gummy shark (71%)	school shark	Shelf	50	All year	26	Gummy Shark	Gummy Shark	
	5	School shark (65%)	gummy shark	Shelf	50	Autumn; Spring	9	School Shark	Gummy Shark	school sharks not targeted
	4	Blue-eye trevalla (90%)		Slope	30	November - March	9	Blue eye Trevalla	Blue eye Trevalla	
	3	Pink ling (78%)	blue-eye trevalla, ribaldo, ocean perch offshore	Slope	30;40	Spring	5	Pink Ling	Pink Ling	
	6		other species, blue-eye trevalla, pink ling, snapper	Shelf - Slope	30;40;50	Not Winter	8	Mixed Scalefish	Mixed Scalefish	

Table D.3 – Description of fleets

Sector	Step 1 Cluster	Metiers	Number of vessels x year	Step 2 (fleet)
Trawl	1	TW.West Slope (60%), TW.West.Shelf (15%), TW.West Blue Grenadier (9%), TW.West.Mixed.deepwater (8%)	67	Mixed Trawler West
	2	TW.East.Shelf (82%), TW.East.Slope (16%)	60	Mixed Trawler East
	3	TW.East.Shelf (63%), TW.East.Slope (30%)	50	Mixed Trawler East
	4	TW.East.Shelf (97%)	41	Shelf Trawler East
	5	TW.East.Royal.Red.Prawn (71%), TW.East Shelf (20%), TW.East.Slope (9%)	12	Royal Red Prawn Trawler
	6	TW.West.Blue.Grenadier (83%), TW.West.Slope (6%)	5	Blue Grenadier Trawler
Danish seine	1	DS.Flathead (90%), DS.School.Whiting (9%)	16	Flathead Danish seiner
	2	DS.Flathead (78%), DS.School.Whiting (20%)	27	Flathead Danish seiner
	3	DS.Flathead (84%), DS.School.Whiting (14%)	22	Flathead Danish seiner
	4	DS.Flathead (95%), DS.School.Whiting (4%)	7	Flathead Danish seiner
	5	DS.Flathead (100%)	5	Flathead Danish seiner
	6	DS.Flathead (70%), DS.School.Whiting (26%)	20	Flathead Danish seiner
	7	DS.School.Whiting (90%), DS.Flathead (8%)	3	School Whiting Danish seiner
	8	DS.School.Whiting (68%), DS.Flathead (31%)	2	School Whiting Danish seiner
	9	DS.School.Whiting (51%), DS.Flathead (48%)	3	School Whiting Danish seiner
	10	DS.School.Whiting (77%), DS.Flathead (20%)	2	School Whiting Danish seiner
Gillnet	1	GN.Gummy.Shark (97%)	48	Gillnetter
	2	GN.Gummy.Shark (90%)	29	Gillnetter
	3	GN.Gummy.Shark (82%)	12	Gillnetter
	4	GN.Gummy.Shark (93%)	30	Gillnetter
	5	GN.Gummy.Shark (100%)	45	Gillnetter
	6	GN.Gummy.Shark (75%), GN.Mixed (25%)	10	Gillnetter
	7	GN.Gummy.Shark (95%)	70	Gillnetter
	8	GN.Gummy.Shark (87%)	20	Gillnetter
	9	GN.Gummy.Shark (57%), GN.Mixed (43%)	3	Gillnetter
Hooks	1	BL.Gummy.Shark (99%)	102	Shark Bottomliner
	2	BL.Gummy.Shark (90%), BL.Mixed.Scalefish (6%)	60	Mixed Bottomliner
	3	BL.Gummy.Shark (66%), BL.Mixed.Scalefish (21%)	25	Mixed Bottomliner
	4	AL.Blue.eye.Trevalla (72%), AL.Mixed.Scalefish (14%), AL.Pink.Ling (12%)	18	Blue eye Autoliner
	5	DL.Blue.eye.Trevalla (91%)	44	Blue eye Dropliner
	6	AL.Pink.Ling (39%), AL. Mixed. Scalefish (22%), AL.Blue.eye.Trevalla (18%)	10	Mixed Autoliner

APPENDIX E

EXAMPLE OF STOCHASTIC TRAJECTORIES AND ASSOCIATED VIABILITY PROBABILITIES

In the example shown in Figure E.1, bass longliners have a 100% probability of meeting both short-term profits and crew wage constraints, whereas hake gillnetters have 0% probability of meeting the crew wage constraint since it is breached for all replicates in 2018.

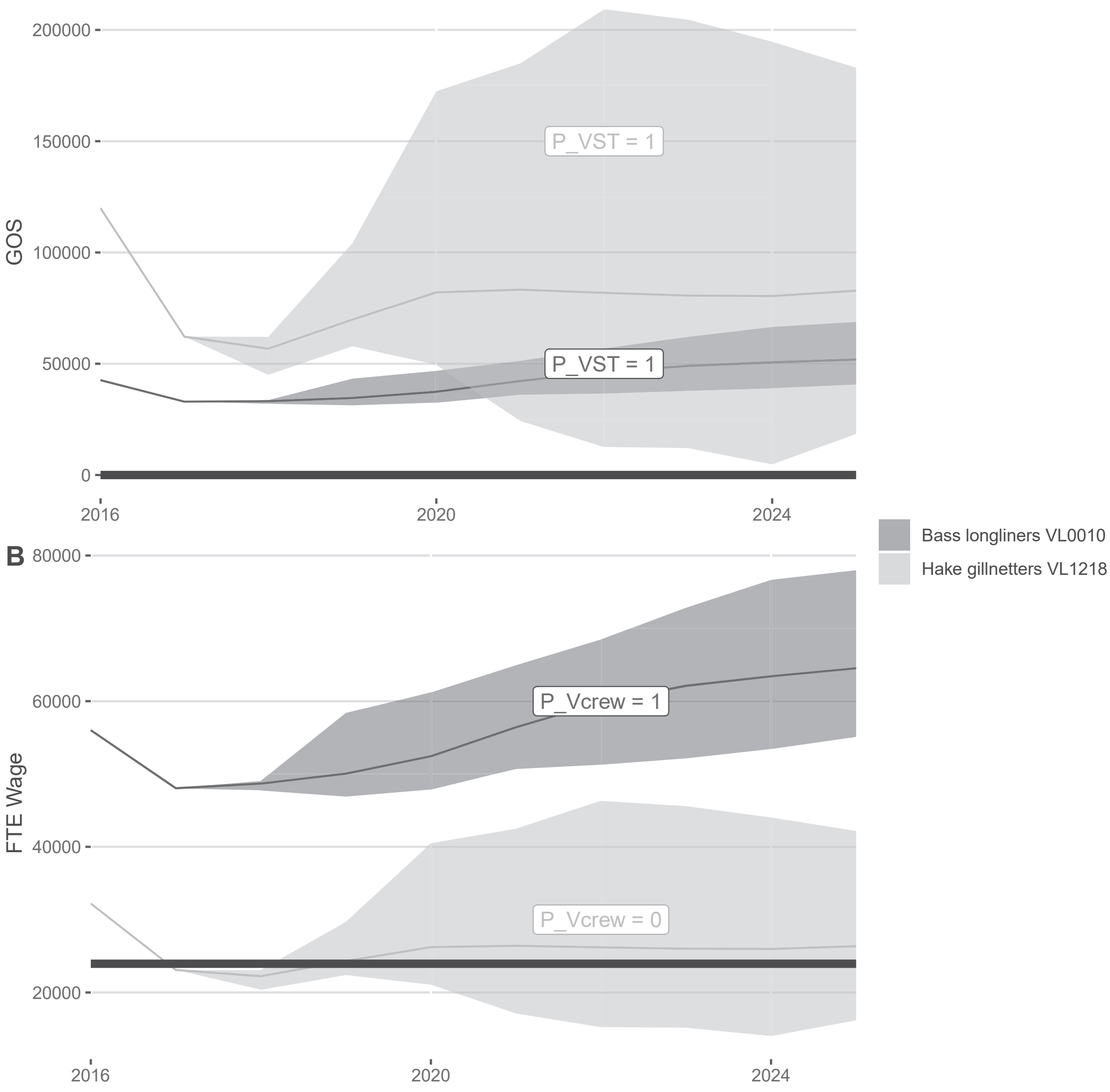


Figure E.1 – Examples of stochastic trajectories (200 replicates) of Gross Operating Surplus and FTE crew wage for 2 fishing segments. The solid black line shows the viability threshold used to estimate the probability of viability of each segment across the 10-year projection period. Source: simulations from IAM model

APPENDIX F

INFLUENCE OF REMUNERATION SYSTEM ON CREW SURPLUS

In the SESSF, crews are usually remunerated based on a share of the fishing income (GVL). A scenario of alternative shared remuneration system, where crews are also charged part of variable costs as can be observed in other fisheries (Guillen et al., 2017), was also simulated for this fishery. In this second scenario, crews get a proportion of the "return to be shared", which is equal to the GVL minus variable costs. Variable costs include fuel, ice, bait, packaging, and freight and marketing. Quota costs are generally covered by vessel owners as highlighted by Hatcher (2010). Crew shares (in %) that are applied in both scenarios are given in Table F.1.

Table F.1 – Crew shares in two systems of shared remuneration: indexed on the gross value of landings (GVL) or "return to be shared" (RTBS).
Source: Estimated from Bath et al. (2018)

Sector	Crew share	
	% GVL	% RTBS
TW	26	54
DS	46	64
GN	42	55
HK	42	54

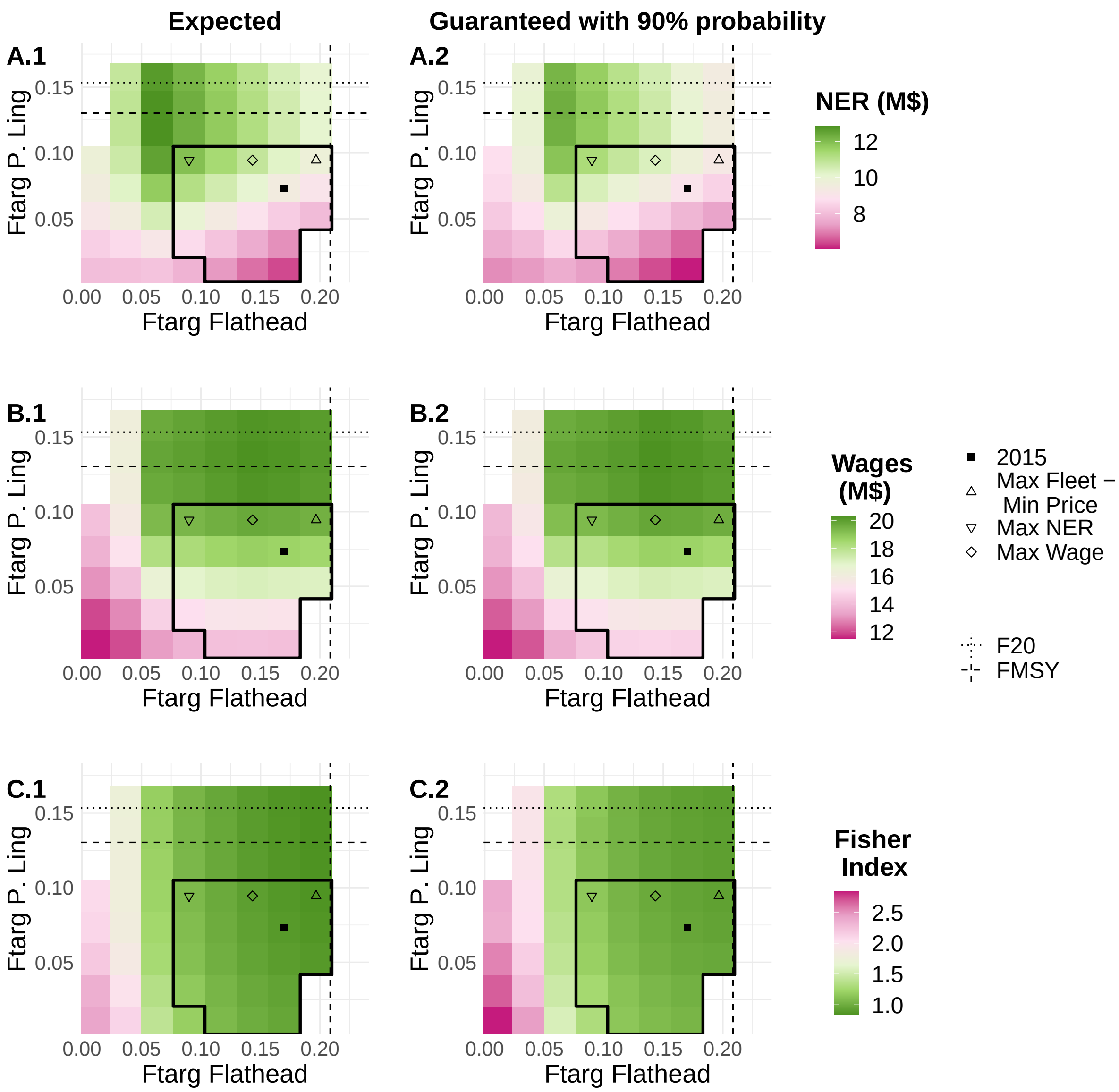


Figure F.1 – Socio-economic indicators within the operating domain when crews are remunerated as a share of the "return to be shared".
First column displays expected values while second columns displays values that are guaranteed with a 90% probability.
A: Net Economic Returns (NER); B: Crew wages; C: Fisher Price Index.
The thick black bounding delineates the eco-viable space, defined as the space ensuring biological and economic viability of the fishery and price acceptability. Dashed lines show stock-specific F_{MSY} and F_{20} reference points. Specific scenarios within the eco-viable space are also identified: the 2015 fishing mortality rates (black square),the one maximizing the industry's net economic returns (Max NER), the one maximizing crew wages (Max Wage) and the one maximizing the viable fleet size but which also minimizes the price of fish (Max Fleet - Min Price).
Source: output from IAM model

A comparison of the results under both scenarios (GVL scenario in Figure 5.4, and RTBS scenario in Figure F.1) shows that when crews are remunerated as a share of the RTBS rather than on a share of the GVL, their surplus aligns more closely with that of capital owners. Indeed, in the RTBS scenario, wages are maximized under lower harvest rates,, closer to the maximization of NER, than in the GVL scenario. This is a result of the sharing of costs between capital owners and crew members as highlighted by Guillen et al. (2017).

APPENDIX G

RÉSUMÉ LONG

G.1 Introduction générale

G.1.1 Durabilité des activités humaines et théorie de la viabilité

Dans son sens premier, la durabilité repose sur l'idée de continuité et se définit comme l'aptitude d'un processus, d'une entité ou d'un système à se maintenir dans le temps. Toutefois, le concept moderne de durabilité ne peut se réduire à son sens premier de continuité car il est aussi empreint de positionnements normatifs au regard de trois relations fondamentales: (i) celles entre êtres humains et leurs contemporains, (ii) celles entres générations présentes et futures, et (iii) celles entre Homme et Nature. Ces relations sont au cœur d'engagements politiques sur la question de la durabilité du développement des sociétés humaines. On peut notamment citer le rapport de Brüntland qui définit le développement durable comme "un mode de développement qui répond aux besoins des générations présentes sans compromettre la capacité des générations futures de répondre aux leurs" et qui a pour but de "promouvoir l'harmonie entre êtres humains et entre l'humanité et la nature".

Ayant vocation à orienter l'action humaine, la durabilité est une préoccupation du temps présent. Agir aujourd'hui pour la durabilité se heurte toutefois à de nombreuses incertitudes quant aux dynamiques des systèmes socio-écologiques. Dans ce contexte d'incertitudes, l'application du principe de précaution prévaut et requiert notamment que la prise de décision intègre les incertitudes relatives à ses effets. Par ailleurs, ne pouvant présumer des besoins et désirs des générations futures, les générations présentes ont également pour devoir de veiller à ce que leurs actions ne viennent pas restreindre la liberté de choix dans le futur.

Depuis sa formalisation mathématique dans les années 1990 par Jean-Pierre Aubin, la théorie de la viabilité fournit un cadre adapté à l'étude de la durabilité des activités d'exploitation des ressources renouvelables. La théorie de la viabilité est un champ des mathématiques ayant pour but l'identification de trajectoires permettant à un système de répondre à un ensemble de contraintes de durabilité. Lorsque les contraintes considérées sont relatives au maintien de fonds, fonctions ou services critiques au fonctionnement du système étudié, la théorie de la viabilité permet d'aborder la question de sa perpétuation dans le temps. On voit ici le lien étroit entre viabilité et durabilité, mais il convient aussi de souligner en quoi le cadre de la viabilité permet aussi d'aborder les dimensions éthiques de l'usage contemporain du terme. Tout d'abord, assurer le maintien à la fois du capital naturel et du capital humain permet une équité de traitement entre hommes et nature. Assurer le maintien de différentes formes de capital humain permet par ailleurs d'aborder des questions de justice intra-générationnelle. Enfin, l'approche intègre par nature le critère d'égalité intergénérationnelle en imposant des contraintes le long d'une trajectoire et non à un instant unique. Originellement formalisée en contexte déterministe, son extension plus récente aux évolutions stochastiques permet d'aborder la question de la durabilité des systèmes sous incertitudes.

G.1.2 Une gestion durable des pêches: l'approche écosystémique des pêches et la science en appui à sa mise en œuvre

Parallèlement à l'affirmation d'objectifs de développement durable sur la scène politique internationale, l'Approche Ecosystémique des Pêches (AEP) s'est vue introduite lors de la Conférence de Reykjavik sur une Pêche Responsable dans l'Ecosystème Marin en 2001 comme le cadre d'opérationnalisation d'une gestion durable des pêcheries. Reconnue dans de nombreuses juridictions comme le nouveau standard de gestion des pêches, son but est de "planifier, développer et gérer les pêcheries de manière à répondre aux multiples besoins et aspirations de la société, sans mettre en péril la possibilité des générations futures de bénéficier de l'ensemble des biens et services fournis par les écosystèmes marins". Elle vise aussi à "contrebalancer une diversité d'objectifs en prenant compte des connaissances et incertitudes relatives aux composantes abiotiques, biotiques et humaines des écosystèmes, ainsi que les interactions entre elles, et d'appliquer une approche intégrée dans les limites écologiques" (traduction de (FAO 2003)). Cette dernière référence à la connaissance et aux incertitudes souligne le rôle crucial porté par la science dans la transition vers l'AEP.

Répondre aux ambitions de l'AEP requiert le développement d'approches à même de prendre en compte les dimensions écologiques, économiques et sociales des systèmes socio-écologiques en question, mais aussi d'évaluer les impacts de mesures de gestion selon ces trois dimensions. La modélisation intégrée, permettant la représentation d'interactions à différentes échelles et à travers différentes dimensions, est devenue un domaine de recherche soutenant activement la transition vers l'AEP.

S'appuyant notamment sur la modélisation bioéconomique, les applications de l'approche d'éco-viabilité dans la pêche ont démontré son potentiel d'appui à l'opérationnalisation de l'AEP. Ces études ont permis de considérer une diversité de contraintes de durabilité, couvrant ses dimensions écologiques, économiques et sociales. Lorsque qu'utilisée pour identifier des contrôles viables du système, l'approche d'éco-viabilité peut être mobilisée en soutien à la gestion des pêches. Jusqu'à présent, les études appliquées aux pêcheries ont principalement considéré l'effort de pêche comme levier d'action, bien qu'une grande partie des pêcheries dans le monde soit aujourd'hui gérée en réglementant les captures plutôt que l'effort de pêche. La considération de TACs (Totaux Admissibles de Captures) et quotas est ainsi nécessaire afin de rendre l'approche d'éco-viabilité opérationnelle comme outil d'aide à la décision dans ce type de pêcherie.

G.1.3 La gestion des pêcheries mixtes

On dénomme pêcheries mixtes les pêcheries où les espèces capturées sont au cœur d'interactions qui peuvent être écologiques (e.g. relations dans le réseau trophique), techniques (e.g. capturées simultanément) ou bien économiques (e.g. produits substituables ou complémentaires). L'application d'approches dites mono-spécifiques, basées sur des objectifs de gestion spécifiés à l'échelle des stocks, se révèle inadaptée à la gestion des pêcheries mixtes. Les interactions mentionnées donnent en effet lieu à des arbitrages entre objectifs de conservation et objectifs socio-économiques, arbitrages qui ne peuvent réellement s'apprécier qu'en considération de la complexité de la dynamique de ces systèmes.

Certaines juridictions ont à cet effet adapté leurs systèmes de gestion au cas des pêcheries mixtes. En Europe par exemple, la réglementation offre désormais la possibilité de définir des quotas dans un intervalle autour du Rendement Maximal Durable (RMD) afin de permettre une plus grande cohérence entre quotas établis et capacité technique des flottilles. Dans les pêcheries fédérales australiennes, le choix a plutôt été d'adopter un objectif de gestion défini à l'échelle de la pêcherie (le Rendement Economique Maximal (REM)) plutôt qu'à l'échelle des différents stocks. Au-delà des objectifs de gestion, des mécanismes de flexibilité (e.g. transférabilité des quotas individuels, délai de régularisation. . .) ont vu le jour dans les systèmes de quotas afin de faciliter leur mise en œuvre dans les pêcheries mixtes.

L'appui scientifique à la décision a dû s'adapter à ces arrangements institutionnels. On peut notamment citer la méthodologie FCube développée par le Conseil International pour l'Exploration de la Mer (CIEM) pour orienter les décisions de TACs dans un intervalle autour du RMD, ou bien la modélisation bioéconomique pour opérationnaliser l'objectif de REM dans la pêcherie crevettière nord-australienne.

G.1.4 Objectifs de la thèse

La thèse propose un cadre d'évaluation multicritères de la durabilité des décisions relatives aux TACs en pêcherie mixte. L'approche d'éco-viabilité est au cœur de la méthodologie développée qui requiert : (1) la prise en compte des interactions critiques à la gestion des pêcheries mixtes à l'aide de TACs et quotas, (2) l'identification de contraintes de viabilité pour ces systèmes, et (3) la synthèse de résultats multidimensionnels en un avis clair, en mesure d'orienter la prise de décision.

Au regard du premier point, la thèse a pour but d'améliorer la représentation des interactions techniques et économiques dans les modèles utilisés en appui à la décision TAC en pêcherie mixte. Une attention particulière est donnée à la représentation des dynamiques d'allocation d'effort de pêche, d'échange de quotas et de prix du poisson dans ces pêcheries. Au regard du second, la thèse vise une meilleure représentation des contraintes de durabilité économique et sociale dans l'évaluation des scénarios de TACs en pêcherie mixte.

G.1.5 Cas d'étude

La thèse repose sur le développement et l'application de l'approche dans deux cas d'étude : la pêcherie démersale française dans le Golfe de Gascogne (GG) et la pêcherie démersale sud-est australienne (SESSF). Ces deux pêcheries mixtes d'importance commerciale notable dans leurs régions respectives capturent quelques centaines d'espèces de manière plus ou moins sélective. Elles sont aussi toutes deux sous gestion par TACs et quotas.

La pêcherie du Golfe de Gascogne est composée de différentes flottilles caractérisées par l'emploi de différents engins de pêche (e.g. chalut démersal, filet maillant) et différents degrés de spécialisation (e.g. chalutiers langoustiniers versus chalutiers mixtes). Les TACs de cette pêcherie sont établis dans le cadre de la Politique Commune des Pêches (PCP) dans un objectif d'exploitation des stocks au RMD. Depuis 2015, l'instauration progressive de l'obligation de débarquement dans les pêcheries européennes a mis en lumière le besoin de cohérence entre quotas d'espèces au coeur de productions jointes.

La pêcherie sud-est australienne est elle aussi composée de plusieurs flottilles déployant une variété d'engins de pêche afin de cibler différentes espèces ou groupes d'espèces. La gestion de la pêcherie se fait notamment à l'aide de Quotas Individuels Transférables (QIT) et la fixation des TACs doit permettre l'opération de la pêcherie au REM. Toutefois, les TACs sont encore fixés selon des points de référence mono-spécifiques, n'assurant donc pas la maximisation des profits à l'échelle de la pêcherie. De plus, le REM est interprété comme la maximisation du surplus des détenteurs de capitaux, négligeant par-là les surplus des équipages et consommateurs dans la prise de décision.

G.2 Chapitre 2

Le chapitre 2 illustre une première application de l'approche d'évaluation multicritère développée à la pêcherie démersale du Golfe de Gascogne. L'analyse se concentre en particulier sur la gestion par TACs de deux espèces clés de la pêcherie, le merlu européen et la sole commune. Elle met en œuvre une modélisation multi-espèce, multi-métier et multi-agent et comprend les étapes suivantes:

1. l'identification des combinaisons de mortalité par pêche atteignables étant données les interactions techniques entre les deux espèces. Ce domaine est appelé *domaine opérationnel*.
2. l'identification des combinaisons assurant la viabilité biologique des stocks dont la dynamique est explicitement représentée dans le modèle (stock nord de merlu européen, stock de sole commune du Golfe de Gascogne, stock de langoustine du Golfe de Gascogne et stock de bar dans le centre-nord du Golfe de Gascogne). Ce domaine constitue le *domaine biologiquement viable*.
3. l'évaluation des scénarios de TACs simulés au regard de trois contraintes de viabilité économique pour les flottilles. Les deux premières ont trait à la rémunération du capital physique et sont: la génération d'excédents brut d'exploitation (EBE) positifs chaque année, et la génération d'excédents net d'exploitation (ENE, i.e. EBE moins la dépréciation du capital) positifs sur une échelle de dix ans. La dernière

contrainte a trait à la rémunération du capital humain engagé et impose une rémunération (à équivalent temps plein) des équipages au moins égale au salaire moyen d'un marin en France.

Les résultats indiquent que les taux d'exploitation associés au RMD des stocks de merlu et sole sont atteignables étant données les interactions techniques entre ces stocks. Un domaine de viabilité biologique non vide est aussi identifié. En revanche, aucun scénario ne permet d'assurer la viabilité économique de toutes les flottilles, suggérant par-là une surcapacité dans la pêcherie. Nous montrons toutefois que fixer le TAC de merlu au RMD et celui de la sole à la borne supérieure de son intervalle de RMD permet d'augmenter le nombre de flottilles économiquement viables.

G.3 Chapitre 3

Le chapitre 3 est dédié à la modélisation des systèmes de marchés de quotas individuels en pêcherie mixte, où des quotas pour des espèces conjointement capturées sont échangés sur des marchés séparés. Une première modélisation s'appuyant sur la programmation linéaire met en lumière quelques résultats théoriques relatifs à la situation d'équilibre de ces marchés. Nous montrons notamment que le prix de marché de la location de quota pour une espèce dont le TAC n'est pas atteint à l'équilibre est nul (espèce non contraignante). En parallèle, celui pour une espèce dont le TAC est atteint à l'équilibre (espèce contraignante) tient compte du revenu associé aux espèces conjointement capturées. La deuxième modélisation permet de représenter la convergence vers cet équilibre de marché. Le modèle présenté imite le processus de tâtonnement walrassien de détermination des prix en compétition pure et parfaite et permet une modélisation mécaniste des marchés de location de quotas en pêcherie mixte. Ce modèle est ensuite appliqué à la pêcherie démersale sud-est australienne avec la considération de deux scenarios d'allocation de l'effort de pêche à l'échelle du navire (allocation historique et allocation motivée par la recherche de profits). Ces simulations mettent en évidence les incitations économiques sous-jacentes aux systèmes de QIT en pêcherie mixte, notamment celles de rediriger l'effort de pêche vers les espèces dont le TAC n'est pas contraignant mais aussi vers celles qui ne sont pas sous quotas.

G.4 Chapitre 4

Le chapitre 4 discute des implications pour la production d'avis TAC en pêcherie mixte de représenter de manière endogène les dynamiques d'allocation d'effort de pêche. La discussion repose sur la comparaison des domaines opérationnels obtenus sous deux types de représentation de l'allocation de l'effort de pêche. La première consiste en une allocation d'effort exogène, égale à celle observée sur une période de référence (le choix fait pour la production d'avis TAC en pêcherie mixte en Europe), et la deuxième en une allocation d'effort endogène motivée par la recherche de profits. L'exercice est répété pour trois types de productions jointes dans la pêcherie sud-est australienne, mettant en jeu des espèces d'importance économique plus ou moins grande ainsi que différents degrés de sélectivité. Nous montrons dans les trois cas que le domaine opérationnel obtenu en endogénéisant l'allocation d'effort est plus vaste que celui obtenu en modélisation exogène. Ce résultat montre qu'une prise en compte des dynamiques de pêche dans les modèles d'appui à l'avis TAC permet d'envisager un plus grand ensemble d'options compatibles avec les interactions techniques. La modélisation intégrée mise en œuvre dans ce chapitre met aussi en lumière la hiérarchie d'espèces dans la pêcherie. Nous montrons en effet que les performances socio-économiques de la pêcherie sont principalement déterminées par les TACs fixés sur les principales espèces commerciales. Ces résultats suggèrent qu'une approche de gestion hiérarchique pourrait être envisagée dans cette pêcherie, avec la considération d'objectifs socio-économiques impactant uniquement la définition de TACs pour les principales espèces commerciales. Les espèces secondaires pourraient quant à elles plutôt être gérées en réponse à des objectifs écologiques.

G.5 Chapitre 5

Le chapitre 5 fait écho au chapitre 2 en présentant une seconde application de l'approche d'évaluation intégrée des décisions de TACs, cette fois dans la pêcherie sud-est australienne. Nourrie des développements entrepris dans les chapitres 3 et 4, la modélisation intégrée mise en œuvre dans ce chapitre permet la prise en compte des dynamiques d'allocation d'effort, d'échange de quotas et de réponse des prix du poisson aux quantités débarquées. Cette dernière dynamique a permis la considération de l'impact des décisions de TACs sur les consommateurs. Un seuil maximal sur le prix du poisson est notamment venu compléter l'ensemble des contraintes de viabilité proposé dans le chapitre 2. Les simulations entreprises permettent l'identification d'un domaine de TACs éco-viables pour le couple *tiger flathead* - lingue rose. La prise de décision au sein du domaine d'éco-viabilité implique toutefois des arbitrages entre maximisation du surplus des détenteurs de capitaux d'un côté et surplus des équipages et consommateurs de l'autre.

G.6 Discussion générale

Les résultats clés de la thèse venant juste d'être soulignés, cette dernière section portera sur les développements méthodologiques apportés et leurs implications en matière de support aux décisions de TACs. Ces derniers sont principalement relatifs à: (1) la modélisation des dynamiques de pêcherie mixte dans une optique d'appui à leur gestion et (2) au développement d'avis TAC intégrant les différentes dimensions de la durabilité de ces systèmes. Enfin, des perspectives à la fois en matière de recherche et d'opérationnalisation d'avis TACs intégrées sont proposées.

G.6.1 La représentation des dynamiques de pêcherie mixte dans les modèles d'appui à la décision TAC

La thèse a contribué à améliorer la représentation des dynamiques de pêcherie mixte dans les modèles impliqués dans la production d'avis TAC. Le même cadre de simulation a été déployé dans les deux cas d'étude, le modèle de simulation intégré IAM, gagnant toutefois en complexité entre sa première application au cas GG et sa seconde dans la SESSF. Cette section a pour but de discuter des implications de ces développements en matière d'avis produit.

La première différence entre les deux applications du modèle a trait à la représentation des dimensions biologiques des pêcheries. Dans la SESSF, près de 80% de la valeur produite repose sur la capture de stocks dont la dynamique de population est prise en compte dans le modèle (le reste étant modélisé en tant que capture par unité d'effort constante, que nous réfèrerons comme de la modélisation "statique" dans le reste du texte), en comparaison au GG où ce chiffre n'est que de 40%. Il est important d'essayer de limiter la part de modélisation statique dans les modèles bioéconomiques car cela donne lieu à des anticipations excessivement optimistes ou pessimistes des impacts économiques de variations de TACs. Dans la SESSF, la modélisation statique a pu être limitée par l'utilisation de modèles de production globaux pour près de 25% de la production, complétant ainsi les modèles structurés en âge issus des processus officiels d'évaluation de stocks.

Les choix relatifs à la représentation de l'activité de pêche dans les deux cas d'étude peuvent aussi être analysés. Les deux applications reposent sur la représentation de chaque navire individuellement, un choix peu commun dans les modèles classiques d'appui à la décision, travaillant généralement à l'échelle de la flottille. Malgré un surcoût de calcul, considérer les navires comme unités d'opération dans le modèle a permis la prise en compte de l'hétérogénéité des aptitudes de pêche au sein d'une flottille et rendrait aussi possible l'intégration d'autres facettes de la diversité des comportements de pêche (e.g. diversité de motivations, interactions sociales...). Dans les deux applications, la représentation de l'activité de pêche fait appel à la notion de métier, qui se définit comme le déploiement d'un engin de pêche particulier dans une zone ou à un moment spécifique, dans le but de cibler une certaine espèce ou assemblage d'espèces. Dans le GG, les métiers représentés dans le modèle avaient été définis dans un projet en partenariat avec les professionnels de la pêche antérieur au travail de thèse, et avaient pour vocation de rendre compte des principales stratégies de pêche observées dans la région. La définition des métiers pratiqués dans la SESSF a quant à elle fait partie intégrante du travail de thèse. L'emploi de méthodes statistiques multidimensionnelles de regroupement a permis l'identification des stratégies de pêche pratiquées dans la pêcherie, des plus au moins fréquemment observées. Cela a donné lieu à une représentation de l'activité de pêche à une échelle plus fine dans la SESSF que dans le GG, permettant ainsi plus de marche de manœuvre dans l'ajustement des pratiques de pêche à la disponibilité en quotas. Le dernier point en lien avec l'activité de pêche concerne la modélisation de l'allocation de l'effort de pêche à l'échelle du navire. Dans la SESSF, cette dernière a été représentée de manière endogène, alors qu'elle demeurait exogène dans l'application au GG. Le modèle considéré fait intervenir un nombre réduit de facteurs (habitude et recherche de profit) et permet de modéliser l'allocation de l'effort aux échelles habituellement considérées dans les modèles d'appui à la gestion des pêches (métier et année). Les travaux de la thèse montrent que des moyens existent déjà pour endogénéiser relativement simplement les comportements de pêche dans les modèles actuels d'appui à la gestion et ainsi améliorer la prise en compte de ces dynamiques dans la prise de décision.

La thèse a aussi permis des développements relatifs à la représentation des dynamiques de marchés de location de quotas en pêcherie mixte. Le modèle développé représente le processus de tâtonnement walrassien pour déterminer des prix d'équilibre de loyer de quota en compétition pure et parfaite. Il rend compte des propriétés clés des systèmes de QITs, telles que la réallocation des droits de pêche vers les opérateurs les plus efficients, mais aussi d'autres, spécifiques à leur mise en œuvre en pêcherie mixte, telles que l'incitation économique à rediriger l'effort de pêche vers les espèces non contraignantes du système (i.e. les espèces dont le TAC est en excès ou celles dont les captures ne sont pas soumises à quota). Les chapitres 4 et 5 illustrent son intégration dans le modèle de simulation IAM. Toutefois, le coût en temps de calcul de cette approche de modélisation (approche itérative et représentation des navires comme agents du modèle) peut constituer un frein à son déploiement dans les modèles de support à la décision.

Enfin, les simulations du chapitre 5 prennent en compte les variations de prix du poisson en réponse aux variations des quantités débarquées. Considérer ces dynamiques a notamment permis de mettre en lumière des arbitrages entre intérêts des producteurs et des consommateurs. Les dynamiques de marché du poisson ont aussi un effet direct sur les revenus des pêcheurs, et par conséquent sur leurs stratégies de pêche. Ainsi, il apparaît important de tenir compte des facteurs de marché dans l'anticipation des réponses des pêcheurs à des changements de TACs.

G.6.2 L'intégration des dimensions biologiques, économiques et sociales de la durabilité dans le développement d'avis TAC

L'approche d'évaluation intégrée développée au cours de la thèse permet la visualisation de différents domaines ainsi que leurs possibles intersections. Le premier, appelé domaine opérationnel, permet la visualisation des taux d'exploitation atteignables étant données les interactions techniques entre les stocks. L'approche d'éco-viabilité permet dans un second temps l'identification des sous-ensembles biologiquement et socio-économiquement viables, avec, à leur intersection, le domaine d'éco-viabilité. La visualisation des arbitrages entre différents objectifs vient compléter l'identification des domaines de viabilité.

En matière d'analyse de viabilité, la thèse a surtout œuvré à préciser les contraintes relatives à ses dimensions économiques et sociales, traitant la dimension biologique de manière analogue à la majorité des travaux portés sur la question. La viabilité économique des flottilles/navires a quant à elle été décomposée en plusieurs contraintes: (1) la rémunération du capital physique à court et long terme, et (2) la rémunération du capital humain. Une dernière contrainte, visant à satisfaire la demande des consommateurs a été considérée dans le chapitre 5.

Les analyses d'éco-viabilité ont mis en lumière des conflits entre objectifs de conservation et d'exploitation. La maximisation de profits à l'échelle de la pêcherie peut notamment amener à la surexploitation de certains stocks. L'actualisation des profits dans le calcul du REM peut aussi pousser la pêcherie hors de son domaine de viabilité biologique. L'attention portée aux différentes dimensions de la viabilité a aussi permis de souligner des arbitrages entre les intérêts des détenteurs de capitaux, des équipages et des consommateurs. Dans le GG, les contraintes de rémunération des équipages empêchent notamment certaines flottilles de couvrir la dépréciation de leur capital. Dans la SESSF, la contrainte de prix sur le poisson restreint la rémunération du capital dans la pêcherie.

L'approche développée permettrait de faire progresser l'avis à la décision TAC dans les contextes européens et australiens. En Europe, l'avis TAC en pêcherie mixte se focalise sur la cohérence entre TACs et interactions techniques, ce qui serait l'équivalent de la première étape de l'approche développée dans la thèse (identification du domaine opérationnel). Les dimensions économiques et sociales de la durabilité de ces pêcheries n'ont jusqu'à présent pas été intégrées dans l'avis scientifique. Dans la SESSF, malgré un objectif de REM à l'échelle de la pêcherie, l'avis TAC est toujours produit selon des approches mono-spécifiques. La première contribution de la thèse a donc été la prise en compte des interactions techniques dans la production d'avis TAC. L'analyse d'éco-viabilité a par ailleurs permis une prise en compte explicite des conséquences économiques et sociales des décisions de TACs, dimensions aujourd'hui traitées de manière implicite à travers des approximations mono-spécifiques du REM.

G.6.3 Perspectives

G.6.4 La modélisation du comportement humain sous incertitudes

Une gestion précautionneuse requiert la prise en compte des incertitudes dans la prise de décision. Les simulations mobilisées dans la thèse ont permis la prise en compte des incertitudes liées au recrutement des stocks exploités, qui pourraient par ailleurs être complétées par celles liées aux paramètres économiques du modèle (e.g. prix du poisson, du carburant) ou à la composition des captures, ce dernier point étant particulièrement pertinent en pêcherie mixte. Il est non seulement nécessaire que les modèles en appui à la décision prennent en compte les incertitudes relatives à l'évolution des systèmes, mais il est aussi important qu'ils représentent les effets que ces incertitudes peuvent avoir sur les comportements humains. Dans le cas particulier des pêcheries mixtes, il serait notamment pertinent de représenter la manière dont les incertitudes liées à la composition des captures ou aux dynamiques de marché des différentes espèces influencent les décisions en matière de stratégie de pêche ou d'échange de quotas.

G.6.5 Le défi de la dimensionnalité

Les évaluations intégrées réalisées dans la thèse ont démontré leur applicabilité et apport dans le contexte restreint de la gestion par TACs de deux espèces au cœur d'interactions techniques. Toutefois, les interactions techniques en pêcherie mixte mettent généralement en jeu plus de deux espèces. Ceci appelle à la poursuite des réflexions quant à la généralisation de ce type d'approche à plus d'espèces. Le premier défi se pose en termes de temps de calcul car le temps nécessaire à la simulation exhaustive des combinaisons de TACs est une fonction exponentielle du nombre d'espèces considérées. L'optimisation multicritère, permettant la recherche efficiente d'optima de Pareto au regard d'un ensemble de contraintes de viabilité, pourrait être une voie à explorer en ce sens. Le second défi a trait à l'analyse et communication de résultats mettant en jeu un grand nombre de dimensions. Là-encore, la stratégie pourrait être de se focaliser sur un nombre restreint de scénarios choisi de manière à mettre en lumière d'éventuels arbitrages.

G.6.6 Une meilleure prise en compte des dimensions écologiques de l'AEP

La dimension écologique de la durabilité a ici principalement été abordée sous un angle anthropocentré. Ayant pour but d'assurer la durabilité des stocks exploités, les contraintes de viabilité biologiques considérées ne permettent pas une analyse de la durabilité de l'écosystème dans lequel s'inscrit la pêcherie. La santé de l'écosystème pourrait notamment être quantifiée à l'aide d'indicateurs écosystémiques qui permettent l'intégration de processus s'observant à et entre différentes échelles écologiques. De plus, des tendances récurrentes dans ce type d'indicateurs faciliteraient l'identification de seuils de perturbation pour les écosystèmes.

G.6.7 Au-delà de l'éco-viabilité

Les analyses d'éco-viabilité visent à l'identification d'options de gestion répondant à un ensemble de contraintes de durabilité. Elles sont donc un outil intéressant, mais guère suffisant pour aiguiller la prise de décision. Dans le cas où le domaine d'éco-viabilité est vide, il peut par exemple être intéressant de mettre en lumière les conflits qui ne peuvent être résolus à l'aide des leviers de gestion considérés. Ceci permettrait d'aider à identifier ceux qui permettraient de restaurer la viabilité du système. Même lorsque des solutions viables existent, l'évaluation transparente de leurs conséquences au regard des différentes dimensions de la durabilité permet d'aiguiller la prise de décision au sein de l'ensemble viable. On pourrait aussi imaginer utiliser le cadre d'éco-viabilité pour réconcilier des aspirations qui vont au-delà des exigences de viabilité. Différents types de contraintes pourraient alors être envisagées, à commencer par celles ayant trait à la viabilité du système, complétées par des contraintes reflétant ce qui est considéré comme acceptable.

LIST OF ACRONYMS

AFMA	Australian Fisheries Management Authority
ABARES	Australian Bureau of Agricultural and Resource Economics and Sciences
BoB	Bay of Biscay
B_{lim}	ICES Limit Biomass reference point
B_{20}	Limit Biomass reference point in the Commonwealth Harvest Strategy Policy
CFP	Common Fisheries Policy
CPUE	Catch Per Unit of Effort
CSIRO	Commonwealth Scientific and Industrial Research Organisation
CTS	Commonwealth Trawl Sector
DCF	Data Collection Framework
DLS	Data-Limited Stock
DPMA	Direction des Pêches Maritimes et de l'Aquaculture
EAF	Ecosystem Approach to Fisheries
EBFM	Ecosystem-Based Fishereis Management
ECDWTS	East Coast Deep-Water Trawl Sector
EU	European Union
FCube	Fleet and Fishery Forecast
F_{20}	Limit fishing mortality reference point in the Commonwealth Harvest Strategy Policy
F_{MSY}	Fishing mortality rate at MSY
FTE	Full Time Equivalent
GABTS	Great Australian Bight Trawl Sector
GHTS	Gillnet Hook and Trap Sector

GOS	Gross Operating Surplus
GVP	Gross Value of Production
HCR	Harvest Control Rule
IAM	Integrated Assessment Model for fisheries management
ICES	International Council for the Exploration of the Sea
IEEFM	Integrated Ecological-Economic Fisheries Models
Ifremer	Institut Français de Recherche pour l'Exploitation de la Mer
ITQ	Individual Tradable Quota
LPUE	Landings Per Unit of Effort
MSY	Maximum Sustainable Yield
MEY	Maximum Economic Yield
NER	Net Economic Returns
NOS	Net Operating Surplus
NPV	Net Present Value
PO	Producers Organisations
SESSF	Southern and Eastern Scalefish and Shark Fishery
SIH	Système d'Information Halieutique
SSB	Spawning Stock Biomass
TAC	Total Allowable Catch
TEP	Threatened, Endangered or Protected (species)
UBO	Université de Bretagne Occidentale
UTAS	University of Tasmania

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Titre : Application de l'approche d'éco-viabilité pour la gestion des pêcheries mixtes sous quotas

Mots clés : pêcheries mixtes, éco-viabilité, modélisation intégrée, appui à la décision, Total Admissible de Capture, Quotas Individuels Transférables

Résumé : La thèse contribue au développement d'approches intégrées en appui aux décisions relatives aux Totaux Admissibles de Capture (TAC) dans des pêcheries mixtes en : (1) prenant en compte les interactions à la fois techniques et économiques de ces pêcheries, et (2) tenant compte des différentes dimensions de leur durabilité. Les développements de la thèse permettent notamment une meilleure représentation des dynamiques humaines dans le type de modèles impliqués dans l'élaboration des recommandations de gestion pour ces pêcheries. Une attention particulière est donnée à la modélisation des dynamiques de pêche, d'échange de quotas et de prix du poisson. La thèse oeuvre en outre à une meilleure prise en compte des dimensions économiques et sociales dans l'évaluation des options de gestion. Elle fait notamment appel à l'approche d'éco-viabilité afin d'identifier des niveaux de captures assurant la viabilité biologique, économique et sociale de deux pêcheries mixtes opérant dans des contextes de gestion différents : la pêcherie démersale du Golfe de Gascogne et la pêcherie démersale sud-est australienne.

Title: Application of the eco-viability approach for the management of mixed fisheries under output control

Keywords: mixed fisheries, eco-viability, integrated modelling, decision-support, Total Allowable Catch, Individual Tradable Quotas

Abstract: The thesis contributes to the development of more integrated approaches to advise Total Allowable Catch (TAC) decisions in mixed fisheries by: (1) accounting for technical, but also economic interactions among jointly harvested species, and (2) addressing the variety of sustainability requirements faced in such fisheries. The thesis notably contributes to improving the representation of human dynamics in the type of models used to advise TAC decisions in mixed fisheries, with a particular attention given to the representation of dynamics in the allocation of fishing effort, the trading of individual quotas and fish markets. The thesis also focuses on how the multiple dimensions of sustainability can be considered in the provision of fisheries management advice. The eco-viability approach is used here to identify future paths maintaining a fishery within ecologically, economically and socially acceptable bounds. The developed methodology has been applied to two mixed fisheries operating in different management contexts: the French demersal fishery in the Bay of Biscay and the Australian Southern and Eastern Scalefish and Shark Fishery.