Ecological and economic viability for the sustainable management of mixed fisheries

By

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"Remember to look up at the stars and not down at your feet. Try to make of what you see and wonder about what makes the universe exist. Be curious. And however difficult life may seem, there is always something you can do and succeed at. It matters that you don't just give up."

Prof. Stephen Hawking

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A small anecdote to finish: I remember, when I was around 10, wanting for a short time to be fishmonger on markets, because they were talking loudly with a lot of people and were looking happy...as if I knew somehow that I would eventually work with fishes...

Abstract

Empirical evidence and the theoretical literature both point to stock sustainability and the protection of marine biodiversity as important fisheries management issues. Decision-support tools are increasingly required to operationalize the ecosystem-based approach to fisheries management. These tools need to integrate (i) ecological and socio-economic drivers of changes in fisheries and ecosystems; (ii) complex dynamics; (iii) deal with various sources of uncertainty; and (iv) incorporate multiple, rather than single objectives.

The stochastic co-viability approach addresses the trade-offs associated with balancing ecological, economic and social objectives throughout time, and takes into account the complexity and uncertainty of the dynamic interactions which characterize exploited ecosystems and biodiversity. This thesis proposes an application of this co-viability approach to the sustainable management of mixed fisheries, using two contrasting case studies: the French Bay of Biscay (BoB) demersal mixed fishery and the Australian Northern Prawn Fishery (NPF). Both fisheries entail direct and indirect impacts on mixed species communities while also generating large economic returns. Their sustainability is therefore a major societal concern.

A dynamic bio-economic modelling approach is used to capture the key biological and economic processes governing these fisheries, combining age- (BoB) or size- (NPF) structured models of multiple species with recruitment uncertainty, and multiple fleets (BoB) or fishing strategies (NPF). Economic uncertainties relating to input and output prices are also considered. The bio-economic models are used to investigate how the fisheries can operate within a set of constraints relating to the preservation of Spawning Stock Biomasses (BoB) or Spawning Stock Size Indices (NPF) of a set of key target species, maintenance of the economic profitability of various fleets (BoB) or the fishery as a whole (NPF), and limitation of fishing impacts on the broader biodiversity (NPF), under a range of alternative scenarios and management strategies.

Results suggest that under a status quo strategy both fisheries can be considered as biologically sustainable, while socio-economically (and ecologically in the NPF case) at risk. Despite very different management contexts and objectives, viable management strategies suggest a reduction in the number of vessels in both cases. The BoB simulations allow comparison of the trade-offs associated with different allocations of this decrease across fleets. Notably, co-viability management strategies entail a more equitable allocation of effort reductions compared to strategies aiming at maximizing economic yield. In the NPF, species catch diversification strategies are shown to perform well in controlling the levels of economic risk, by contrast with more specialized fishing strategies. Furthermore analyses emphasize the importance to the fishing industry of balancing global economic performance with inter-annual economic variability.

Promising future developments based on this research involve the incorporation of a broader set of objectives including social dimensions, as well as the integration of ecological interactions, to better address the needs of ecosystem-based approaches to the sustainable harvesting of marine biodiversity.

Scientific production

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Gourguet S., Macher C., Doyen L., Thébaud O., Bertignac M., Guyader O., (2013). *Managing mixed fisheries for bio-economic viability*, Fisheries Research, 140, 46-62. Available online.

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Congress presentations

Gourguet S., Thébaud O., Dichmont C., Jennings S., Little L.R., Pascoe S., Deng R.A., Doyen L., *Risk versus economic performance in a mixed fishery: the case of the Northern Prawn Fishery in Australia*. Paper accepted for oral communication at the 20th Annual Conference of the European Association of Environmental and Resource Economics, to be held in Toulouse from 26 to 29 June, 2013.

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economic model for the viable management of the Bay of Biscay mixed demersal fisheries. Oral communication at the 13th International BIOECON Conference, Geneva, Switzerland, Sept 2011.

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Contents

Chapter 1. General introduction

1.1 General context

1.1.1 Marine biodiversity and ecosystem services

The growing importance of biodiversity and ecosystem services (MEA, 2005) in environmental policy-setting is demonstrated by the establishment in 2012 of the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES), a global science-policy interface which is expected to enhance global conservation policy (IISD, 2012). The Platform objective is to strengthen the science-policy interface for biodiversity and ecosystem services for the conservation and sustainable use of biodiversity, long-term human well-being and sustainable development (UNEP, 2012). Worldwide, sustainable use of the environment is seen as a crucial issue in managing biological resources (FAO, 1999; Heino and Enberg, 2008). For instance the Implementation Plan of the World Summit on Sustainable Development ('Johannesburg Summit', 2002) promotes sustainable use of biological diversity in general and sustainable forestry, agriculture and fisheries in particular.

The oceans are a significant source of animal protein for the global population (Pauly et al., 2005; FAO, 2012). These resources also underpin substantial economic activity, through harvesting, production, and transport of seafood and the associated secondary activities of ship building, maintenance, fishing supplies, research and administration. In 2010, fisheries and aquaculture provided livelihoods and income for an estimated 54.8 million people engaged in the primary fish production sector alone, of which an estimated 7 million were occasional fishers and fish farmers (FAO, 2012). Furthermore, total employment in fish production (including dependants), is estimated to support the livelihoods of 660–820 million people, or about 10–12 % of the world's population (FAO, 2012). Capture fisheries and aquaculture produced about 148 million tonnes of fish in 2010 (with a total value of US$217.5 billion), of which about 128 million tonnes was used for human consumption (figure 1.1).

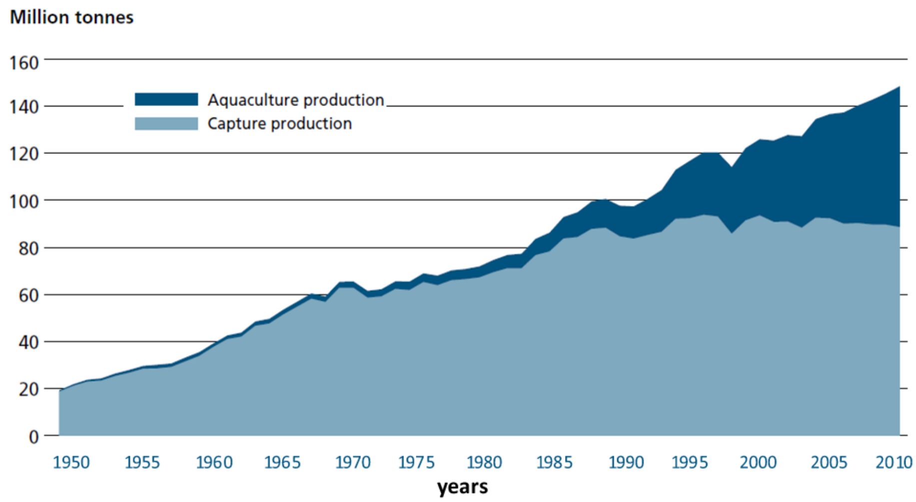

Figure 1.1: World capture fisheries and aquaculture production. Source: FAO (2012)

1.1.2 Marine biodiversity under pressure

The past few decades have been characterised by accelerating loss of marine populations and species (Hutchings and Reynolds, 2004), with largely unknown consequences. Extensive declines over the past 50 years have been reported across a broad range of predatory marine fishes (Christensen et al., 2003; Myers et al., 2003). The sequential economic over-harvesting of high order trophic species and subsequent focus on lower trophic level species has been called ‘fishing down the food chain’ (Pauly et al., 1998). There is little doubt that part of the biodiversity loss observed in coastal and marine ecosystems is related to human activities, particularly fisheries.

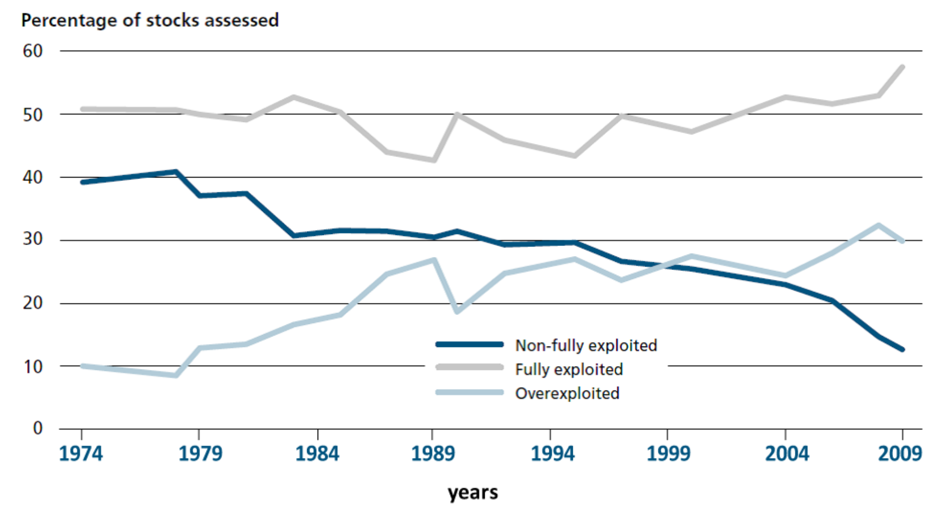

Figure 1.2: Evolution of status of worldwide stocks. Source: FAO (2012)

Despite the recognized importance of marine resource sustainability (Heino and Enberg, 2008), Worm et al. (2009) estimate that 63 % of assessed fish stocks worldwide require rebuilding; while only 22 % of the world's fisheries are considered sustainable (UN, 2008). The proportion of stocks not fully exploited has decreased gradually since 1974 when the first FAO assessment was completed (Figure 1.2). In contrast, the percentage of overexploited stocks increased, from 10% in 1974 to 30% in 2009 (FAO, 2012).

1.1.3 Open access resource exploitation issues

Common pool resources that are exploited by more than one party are vulnerable to the 'tragedy of the commons' (Hardin, 1968). Wild fish stocks, that have open access harvesting (Gordon, 1954; Hardin, 1968), provide a common illustration of this. When fishers can freely enter and capture a resource (open access), resource users do not consider the external effects their behaviour imposes on others. Such 'externalities' can be either positive or negative. Fishing externalities are commonly negative (Smith, 1969; Agnello and Donnelley, 1976). In such cases the private cost of using the resource is less than the social cost; therefore negative externalities created by each fisher on the rest of the fleet incite fishers to be the first to catch the fish. The consequence of this is a 'race for fish' as individuals increasingly attempt to maximise their take of the common pool resource, leading to overcapacity and overexploitation (Beddington et al., 2007; UN, 2008). Such overexploitation can reduce fish stocks to unprofitable levels, and to levels from which they can not recover resulting in fishery collapse. Overcapacity, which occurs where significantly more fishing effort (eg. vessels or fleets) is employed than is required to produce a given output (e.g. harvested catch) level, can result in substantial financial losses. The economic loss to society resulting from the overexploitation and degradation of marine resources globally, estimated by Arnason, Kelleher and Willman (Bank and FAO, 2009) as the difference between actual and potential economic benefits of marine resource exploitation, was found to range between 24 and 72 billion US$/year (average: 50 billion US$). This loss could be avoided if fish stocks were rebuilt to higher levels, catch capacity reduced and perverse subsidies ended.

1.1.4 Marine resource and managements

Measures for fisheries management

The effective management of common pool resources necessitates public intervention and careful management. Fishery management relies on two major sets of measures: technical and access regulation measures (Boncoeur et al., 2006). Technical, or conservation, measures aim at the protection of the productive and reproductive capacity of stocks via two types of tools. Some tools concern the selectivity of catches, and others involve direct and indirect limitation of total catch. Access regulation measures are concerned with the regulation of individual access to fish stocks and operationally require the selection of operators who are allowed to fish a given stock (or group of stocks) and/or the determination of each operator's share (Boncoeur et al., 2006). These fisheries management tools can also be classified according to the control method ('administrative' versus 'economic' methods) and the control variable ('input' versus 'output' controls) on which they operate.

Toward integrated approaches

A wide range of stakeholders are typically involved in fisheries, including industrial, artisanal, subsistence and recreational fishermen, suppliers and workers in allied industries, managers, scientists, environmentalists, economists, public decision makers or the general public (Hilborn, 2007b). Each of these groups has an interest in particular outcomes, which may conflict among groups (Hilborn, 2008). There has also been increasing widespread acceptance that an integrated perspective is needed at the ecosystem scale which embraces marine biodiversity preservation, and economic and social objectives across a broad range of interests (FAO, 2003; Pikitch et al., 2004; Nomura, 2008; Kempf, 2010). In line with this, numerous scientists and stakeholders advocate an Ecosystem-Based Fishery Management (EBFM) approach. One definition of EBFM is given by FAO (2003) as: "an ecosystem approach to fisheries strives 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".

Implementation of EBFM requires decision-support tools to highlight trade-offs in management options by assessing their potential biological, social and economic impacts. Given the complexity of fisheries systems, these decision-support tools are increasingly based on computer simulation models.

1.2 Bio-economic models

Economics is one of the most important factors influencing fishing and other marine based activities. Models that combine biological and economic considerations (bio-economic models) have been developed in fisheries since the middle of the 1950's (Gordon, 1954; Schaefer, 1954). Bio-economic models can show the implications of management decisions (Anderson and Seijo, 2010) and are increasingly used as a decision-support tool for fisheries management where the aim is to integrate human activities with ecological models.

1.2.1 Schaefer model

The 'Schaefer model', originally introduced in Schaefer (1954), includes harvesting activities in a population dynamics model, which in discrete time is given by:

$$\mathrm{x}(t+1) = f\Big(\mathrm{x}(t)\Big) - \mathrm{Y}(t), \qquad 0 \leq \mathrm{Y}(t) \leq \mathrm{x}(t). \tag{1.1}$$

where $\mathrm{x}(t)$ represents the biological resource stock at time t, $\mathrm{Y}(t)$ is the harvesting flow or catch at time t and f captures the regeneration and growth of the stock generally involving a density dependence mechanism through an intrinsic growth rate r and a carrying capacity K. Most usual population dynamics include the logistic[1], Beverton-Holt[2], or Ricker[3] functions. In equation (1.1), regeneration is assumed to occur at the beginning of time t, while harvesting is assumed to take place at the end of the time t. It is frequently assumed that the catch $\mathrm{Y}(t)$ depends on both biomass and fishing effort, through some relation:

$$\mathrm{Y}(t) = H\Big(\mathrm{x}(t); \mathrm{E}(t)\Big). \tag{1.2}$$

where the catch function H increases in both arguments stock x and effort E. The most usual form of this production function is bilinear as proposed by Schaeffer with $H = q\mathrm{Ex}$ where parameter

[1] $f(\mathrm{x}) = \mathrm{x} + \mathrm{rx}(1 - \frac{\mathrm{x}}{\mathrm{K}})$.
[2] $f(\mathrm{x}) = \dfrac{(1+\mathrm{r})\mathrm{Kx}}{\mathrm{K}+\mathrm{rx}}$.
[3] $f(\mathrm{x}) = \mathrm{x}\exp\Big(\mathrm{r}(1 - \frac{\mathrm{x}}{\mathrm{K}})\Big)$.

q refers to catchability.

Bio-economics can be used to explore different processes in both the biological and economic sub-systems, and also to measure and assess the effects of resource management activities on various management objectives and performance criteria (biomasses, biodiversity, catches, profits, etc.). These models provide a more comprehensive indication of the linkages and feedback effects between socio-economic activity and natural resources (Prellezo et al., 2012). There is growing interest in using them for policy analysis to better understand pathways of development and to assess the impact of alternative policies on the natural resource base and human welfare.

1.2.2 Maximum Sustainable Yield

As pointed out by Mardle et al. (2002), to develop fisheries management policy, objectives must be defined and targets for achievement must be set. Fisheries management catch targets vary across worldwide fisheries. To manage fisheries, there is a need to know how much can be safely taken without depleting the resource, and without otherwise negatively impacting the environment. A first step consists therefore of reasoning at equilibrium, when a stationary level of exploitation induces a steady population x_E. In this context, the Schaefer model defines the so-called 'sustainable yield' $\sigma(x_E)$ by an implicit equation whenever, for given x_E, there exists a harvesting Y_E giving:

$$\sigma(x_E) := Y_E \iff x_E = f(x_E) - Y_E. \tag{1.3}$$

Meaning that a "surplus production exists that can be harvested in perpetuity without altering the stock level" (Clark, 1990).

A common objective of fisheries management has been to maintain steady population so as to afford the largest yield (or catch) that can be taken from the population stock over an indefinite period under constant environmental conditions, which is called the Maximum Sustainable Yield (MSY). The MSY is therefore the maximum catch Y_{MSY} expressed by:

$$Y_{MSY} = \max_{x \geq 0, Y = \sigma(x)} Y = \max_{x \geq 0} \sigma(x). \tag{1.4}$$

The associated effort E_{MSY} derives from catch Y_{MSY} and stock x_{MSY}.

Box 1 gives graphical details on the estimation of MSY. As an apparently simple[4] and logical

[4]MSY is simple to explain, as it is a purely physical measure, i.e. not tied to any economic or social and hence political

management objective, and in the absence of other management goals with similar qualities, MSY was an important concept in the 1982 United Nations Convention on the Law of the Sea, thus paving the way for integration into national fisheries acts and laws (Mace, 2001). Fishing the biomass to below that at which MSY is achieved has been a traditional definition of biological overexploitation, and MSY can (in principle at least) be estimated using limited data and computing resources (Punt and Smith, 2001). However, the use of MSY in fisheries management has been criticized especially for its single species focus, equilibrium implication, the fact that it is not actually sustainable over the long run because of natural fluctuations in resource stock (Conrad and Clark, 1987), but also for its economic implications (e.g. Larkin, 1977; Sissenwine, 1978).

Box 1: MSY.

The key assumption behind sustainable yield concept is that populations grow and replace themselves, and it is assumed that they produce a surplus of biomass that can be harvested (i.e. surplus production). The case of a logistic function population dynamics is represented in figure 1.3. Here, MSY is defined as the peak value of a surplus production curve.

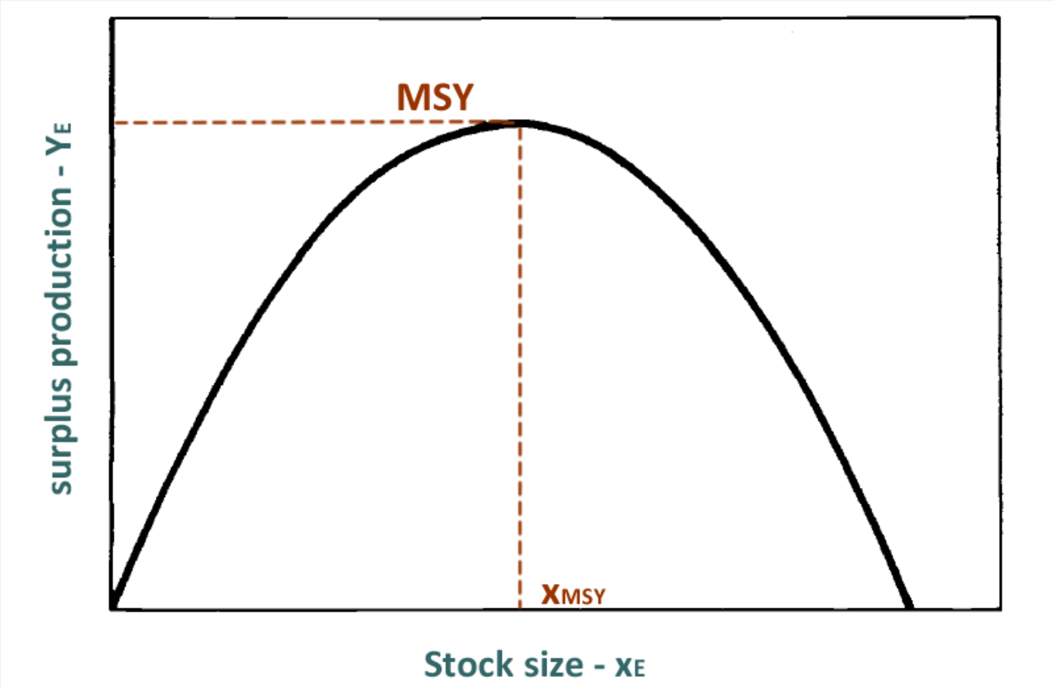

Figure 1.3: MSY in the case of a logistic function.

doctrine (Punt and Smith, 2001).

1.2.3 Maximum Economic Yield

Among economists a dominant view of marine resources is simply as a type of asset which should be managed so as to maximize its value to society. Therefore, some fisheries in the world are committed to the Maximum Economic Yield (MEY) target (Grafton et al., 2010), especially in Australia[5].

Static x_{MEY} corresponds to an equilibrium that maximizes the expected economic profit π in a fishery. Effort E_{MEY} is the solution of:

$$\max_{(x,E) \text{ such that } Y=qEx=\sigma(x)} \pi(x, E). \tag{1.5}$$

with

$$\pi(x, E) = \text{Inc}\Big(x, E\Big) - c^{var}(E) - c^{fix}. \tag{1.6}$$

where the function Inc expresses the gross revenue of the fishery (often assumed to equal $pY(t)$), the function c^{var} defines the total variable cost depending on effort E, and c^{fix} corresponds to the total fixed cost associated to one period of time.

In most cases, this catch criteria results in effort levels smaller than at MSY and in stock biomass levels greater than at MSY (Clark, 1990; Grafton et al., 2007) as it is shown in box 2.

Clark (1973) and Clark and Munro (1975) developed steady-state expressions for dynamic MEY in an inter-temporal setting with discounting and assumed that harvesting costs were proportional to the biomass. Dynamic MEY ($dynMEY$) can therefore be calculated from the sum of discounted profits, as E_{dynMEY} is the solution of:

$$\max_{E(0),E(1),\dots,E(T)} \sum_{t=t_0}^{T} \frac{1}{(1+\delta)^t} \pi\Big(x(t), E(t)\Big). \tag{1.7}$$

where T is the time horizon and δ represents the discount rate (the rate at which future income or expenditure is discounted relative to the present value (Grafton et al., 2006)). In the particular case where $\delta = 0$, the dynamic MEY is equal to the static MEY.

[5]The Australian Fisheries Management Act 1991, which relates to fisheries in Commonwealth waters, includes maximizing economic efficiency as an explicit objective, and since 2007 the associated Australian Fisheries Harvest Policy (DAFF, 2007) states that harvest strategies “will be designed to pursue MEY in the fisheries”.

Box 2: Static MEY.

Figure 1.4 illustrates a typical production surplus model (as in figure 1.3) expressed in terms of economic relationships. The total revenue (Inc) curve shows the relationship between effort and economic yield. Every point along this curve represents an effort and economic yield combination that is sustainable (i.e. associated to the surplus production, as in figure 1.3), with effort at MSY generating the largest total gross revenue (i.e. corresponding of the peak value of the surplus production function). The total cost curve is taken as the total variable cost of fishing (in this case, total fixed costs are assumed to be equal to 0), assumed to be increasing and linear in effort, for convenience. MEY in figure 1.4 occurs at the effort level E_{MEY} and corresponding value of catch Y_{MEY} that creates the largest difference between the total revenue and total cost, thus maximising profits, given by the difference between gross revenues and costs. The value of E_{MEY} will change given a change in market prices, which shifts the total revenue curve up or down, or the cost of fishing, which rotates the total cost curve (Kompas, 2005).

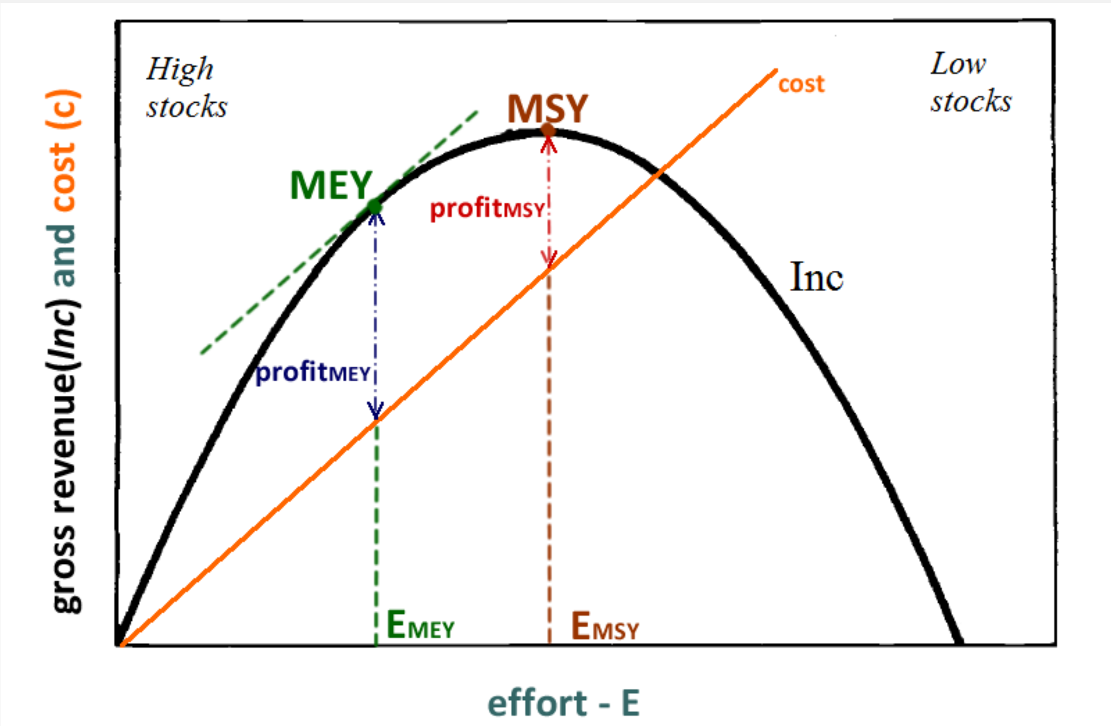

Figure 1.4: MEY and MSY.

MEY has also become the object of criticism by both biologists and economists. It has been noted that producing effort levels lower than the MSY result in fewer benefits further along the value chain (i.e. processing, retail, etc.) that may outweigh the additional profits generated through the fishing process itself (Christensen, 2010). Opponents to MEY argue then that, potentially, achieving MEY in fisheries – as it is traditionally defined – may result in a net economic loss relative to MSY when the flow on effects to the rest of the economy are considered (Bromley, 2009). While Grafton et al. (2010) and Norman-López and Pascoe (2011) respond to these criticisms, managing fisheries for MEY remains controversial and operationalizing MEY requires strong industry commitment and involvement (Dichmont et al., 2010).

1.2.4 Risk management

It has also been recognized that wise use of fish resources over time should incorporate the inherent risk and uncertainty of fishery systems (Garcia, 1996; Hilborn and Peterman, 1996). Hilborn and Peterman (1996) identified a set of sources of uncertainty associated with stock assessment and management, including uncertainty in resource abundance (for instance, uncertainty in recruitment), in model structure, in model parameters, in the behaviour of resource users, in future environmental conditions, and in future economic, political and social conditions. Implementation of risk management is therefore crucial to reduce the risk of collapse of fishing communities. As Hilborn et al. (2001) said: "if we are to succeed at management – if we are to maintain stable fishing communities – we have to begin to manage risk". One way of reducing risk is by reducing fishing pressure in order to have larger average stock sizes, which would serve as a buffer for natural fluctuations (Hilborn et al., 2001). To deal with the variety of uncertainties, the Precautionary Approach, which involves the application of prudent foresight by taking account of the uncertainties in fisheries systems and the need to take action with incomplete knowledge (Garcia, 1996), has been suggested. The approach is used by the International Council for the Exploration of the Sea (ICES) for fixing quotas.

The use of reference points is also recommended (Caddy and Mahon, 1995) as a guide for fisheries management. A reference point indicates a particular state of a fishery indicator corresponding to a situation considered as desirable ('target reference point'), or undesirable and requiring immediate action ('limit reference point' and 'threshold reference point') (Caddy and Mahon, 1995; Garcia, 1996). Reference points may be general (applicable to many stocks) or stock-specific. Defining reference points is not an easy task. Annex II of the UN (1995) suggests that the fishing mortality rate which generates MSY should be regarded as a standard for limit reference points. However, in implementing the Precautionary Approach, the ICES defined limit reference points in terms of the maximum fishing mortality and minimum biomass thresholds associated with stock collapse rather than with MSY. Furthermore the ICES has been using precautionary reference points in addition to the limit reference points to frame its advice to the European Commission, the Northeast Atlantic Fisheries Commission, and the North Atlantic Salmon Conservation, to cite some. Indicators for fisheries performance are an integral part of fisheries management plans providing dynamic signs of the relative position of such indicators with respect to predetermined reference points (Seijo and Caddy, 2000). The figure 1.5 illustrates how the precautionary approach can be implemented, through Harvest Control Rules (HCR), by specifying when a rebuilding plan is mandatory in terms of precautionary and limit reference points for spawning biomass and fishing mortality rate. The HCR describes how the harvest is intended to be controlled by management in relation to the state of some indicator of stock status.

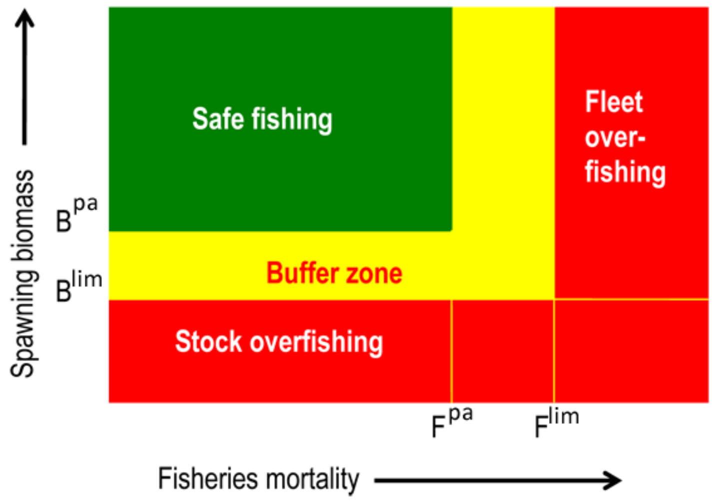

Figure 1.5: Harvest Control Rules and precautionary approach.

1.2.5 Multiple management objectives

In any given context, fisheries management is clearly characterised by multiple objectives, some of which may be conflicting (Crutchfield, 1973; Charles, 1989). While management objectives are often not clearly stated, Charles (1989) summarized some of the most commonly declared objectives of fisheries management as: (i) resource conservation, (ii) maintaining the viability of fishing communities, (iii) food production, (iv) generation of economic wealth, (v) generation of reasonable income for fishers, and (vi) maintaining employment for fishers. These objectives can be grouped according to whether they reflect ecological, economic or social objectives.

An important issue is thus to determine management procedures that give acceptable results with respect to the sustainability objectives while being robust to uncertainties (De Lara and Martinet, 2009). Management Strategy Evaluation (MSE) provides a framework for comparing different fisheries management strategies with respect to conflicting objectives, and taking into account uncertainties (Sainsbury et al., 2000; Kell et al., 2007). However, even if MSE allows us to describe trade-offs in management objectives and to characterize potential management procedures with respect to a set of performance statistics, due to the absence of an agreed 'common measure' or conflicting performance measures, the decision-makers are left with clearer perspectives but without tools to rank the various management procedures (De Lara and Doyen, 2008).

Therefore, even if the current approaches for sustainable management of fisheries have been and are still useful; there is a need for alternative approaches that can help to deal with multiple objectives under uncertainty.

1.3 Viability models

Viability theory, introduced by Aubin (1990), aims at identifying decision rules or controls for dynamical systems (in particular non-linear control problems), such that the systems are maintained at each instant inside a given set of admissible states of diverse nature, called the viability constraints set. Although dynamic optimisation problems are usually formulated under constraints, the role played by the constraints poses difficult technical problems and is generally not tackled as a specific issue (De Lara and Doyen, 2008). Furthermore, the optimization procedure reduces the diversity of feasible forms of evolution by, in general, selecting a single trajectory (De Lara and Doyen, 2008). In contrast, viability analysis, instead of maximizing an objective function, focuses on the role of constraints and on characterizing the safe paths and decisions.

1.3.1 Non-linear dynamical system and viability constraints

Solving the viability problem relies on the consistency between a controlled dynamic and acceptability constraints applying both to states and decisions of the system. Viability constraints for a non linear dynamic system x are represented by:

$$\begin{cases} \mathrm{x}(t+1) = f\Big(t, \mathrm{x}(t), \mathrm{u}(t)\Big), & t = t_0, \ldots, T, \\ \mathrm{u}(t) \in \mathbb{B}\Big(t, \mathrm{x}(t)\Big), & \forall t = t_0, \ldots, T, \\ \mathrm{x}(t) \in \mathbb{A}(t), & \forall t = t_0, \ldots, T. \end{cases} \tag{1.8}$$

where $\mathrm{u}(t)$ is the control or decision, $\mathbb{B}\Big(t, \mathrm{x}(t)\Big)$ is the set of admissible and 'a priori' feasible decisions, and $\mathbb{A}(t)$ corresponds to a non empty state or target (according the objective of the analysis) constraint domain and represents the safety, the admissibility or the effectiveness of the state for the system x at time t .

This approach has been applied to a number of environmental management and sustainability issues (Bene et al., 2001; Bonneuil, 2003; Eisenack et al., 2006; Rapaport et al., 2006; Aubin and Saint-Pierre, 2007; Martinet and Doyen, 2007; Tichit et al., 2007; Baumgärtner and Quaas, 2009; Bene and Doyen, 2008; De Lara and Doyen, 2008; De Lara et al., 2011; Doyen and Martinet, 2012; Doyen and Péreau, 2012) and to fisheries management problems in particular (Béné and Doyen, 2000; Mullon et al., 2004; Cury et al., 2005; Doyen et al., 2007; Martinet et al., 2007; De Lara et al., 2007; Chapel et al., 2008; Bendor et al., 2009; Martinet et al., 2010; Doyen and Péreau, 2012).

1.3.2 Co-viability analysis

As fisheries management is characterized by multiple management objectives, the co-viability approach combines multiples constraints, like biological and economic viability constraints, as defined by the set of equations (1.9) inspired by Bene et al. (2001):

$$\begin{cases} \mathrm{x}(t) \geq \mathrm{x}_{min} \geq 0 & \forall t = t_0, \ldots, T, \\ \pi\Big(\mathrm{x}(t), \mathrm{E}(t)\Big) \geq \pi_{min} \geq 0 & \forall t = t_0, \ldots, T. \end{cases} \tag{1.9}$$

where x_{min} is the resource minimum level to maintain, $\pi\Big(\mathrm{x}(t), \mathrm{E}(t)\Big)$ represents the net benefit (or profit) from the harvesting of the resource x and π_{min} is the minimum profit to guarantee at all time (typically, the lower viable bound on the profit can be zero).

Co-viability analysis shows the ability of management actions to maintain natural and economic capital stocks above some minimum levels. This approach (Le Fur et al., 1999; Bene et al., 2001; Eisenack et al., 2006; Martinet et al., 2007) can be particularly useful in multi-criteria management problems, as it can highlight the domain of possibilities, feasibility and trade-offs between potentially conflicting objectives or constraints that are required to be fulfilled both in present and future time periods.

1.3.3 Stochastic co-viability

Risk and uncertainty of fishery systems constitute major issues in fisheries management, and acceptability constraints in a co-viability context have to be articulated with uncertainty in a probabilistic or stochastic sense. The probability of co-viability CVA of a fishery system, regarding control or decision $\mathrm{u}(t)$ and considering the multiple constraints defined in section 1.3.2, is expressed by:

$$\mathrm{CVA}\Big(\mathrm{u}(t_0),\dots,\mathrm{u}(T)\Big) = \mathbb{P}\Big(\text{ constraints (1.3.2) are satisfied for } t = t_0,\dots,T\Big). \tag{1.10}$$

The idea underlying stochastic viability is to require the respect of the constraints at a given confidence level. Therefore, of particular interest are the set of control or decision $\mathrm{u}(t)$ such that the probability of co-viability CVA is above a certain level as:

$$\mathrm{CVA}\Big(\mathrm{u}(t_0),\dots,\mathrm{u}(T)\Big) \geq \beta. \tag{1.11}$$

With β some confidence level (typically 90%, 95% or 100%).

By compiling ecological and economic goals from stochastic simulation models, stochastic co-viability analysis (De Lara and Doyen, 2008; Baumgärtner and Quaas, 2009; Doyen and De Lara, 2010) can be used to address important issues of vulnerability, risk, safety and precaution, and to determine the ability of a particular resource system to achieve specified multiple sustainability objectives with sufficiently high probability. In contrast to the MSE approach, stochastic co-viability analysis proposes a 'sustainability metric' to rank alternative management strategies through the co-viability probability. Indeed, as viable management requires all constraints (and hence objectives) to be satisfied, the approach is not based on arbitrary weights that may reflect priorities in the objectives. This approach therefore provides a useful tool to inform policy makers about the trade-offs involved in managing fisheries under multiple constraints in a stochastic environment. As depicted in box 3, the viability kernel plays a major mathematical role in the viability analysis.

Box 3: Viability kernel.

A major mathematical tool to study the whole viability of the system is provided by the so-called viability kernel, denoted by Viab. It corresponds to the set of all initial conditions such that there exists at least one trajectory starting from the initial conditions that stays in the set of constraints $\mathbb{A}$. Figure 1.6 gives a graphical representation of the viability kernel.

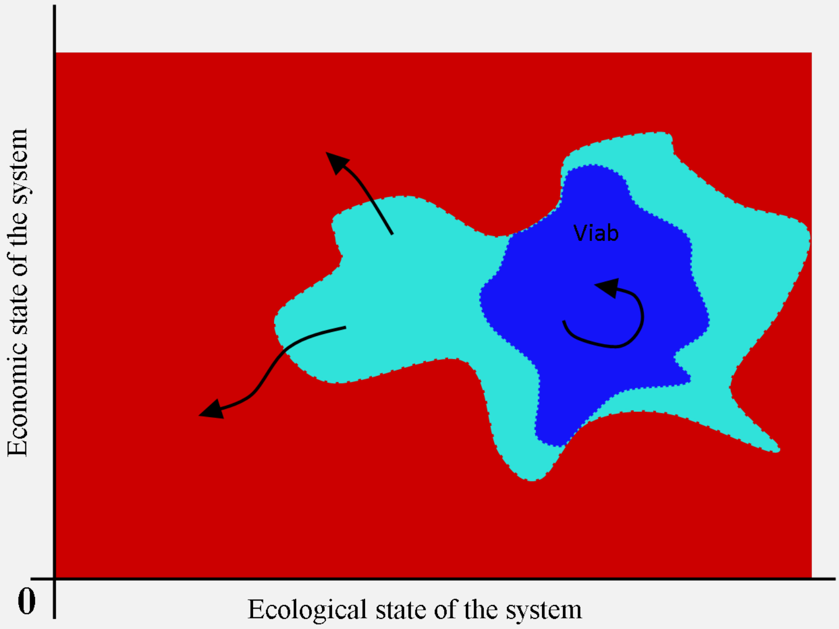

Figure 1.6: The state constraint set $\mathbb{A}$ defined by a set of ecological and economic viability constraints corresponds to the large blue set. It includes the smaller viability kernel Viab (in dark blue).

For decision makers, knowing the viability kernel has practical interest since it describes the states from which controls can be found that maintain the system in a desirable configuration until the horizon time T.

Only a few applied studies (Béné and Doyen, 2000; De Lara et al., 2007; Doyen et al., 2007; Martinet et al., 2007; De Lara and Martinet, 2009; Martinet et al., 2010; De Lara et al., 2011; Péreau et al., 2012) have made use of the co-viability approach to integrate economic and ecological objectives for fisheries management. Among these studies, only Doyen et al. (2007) and De Lara and Martinet (2009) integrate uncertainty affecting biological dynamics. While these studies integrate several species in their analysis, the diversity among the fishery industry is not taken into account.

1.4 Thesis objective

There is a growing need for decision-support tools to assist in evaluating management policies for the regulation of marine fisheries. These tools need to (i) integrate ecological and socio-economic drivers of changes in fisheries and ecosystems; (ii) include the complex dynamic dimensions of these drivers; (iii) deal with various sources of uncertainty and (iv) incorporate multiple, rather than single objectives. In this context, this thesis investigates the use of stochastic co-viability analysis as a decision-support tool to assist in evaluating management strategies for sustainable regulation of mixed fisheries in different contexts. This analysis is developed for two case studies (the French Bay of Biscay mixed demersal fishery and the Australian Northern Prawn Fishery) each of which displays different features and thereby allowing the evaluation of the approach for different ecosystems. Consistent with the sustainability issue, the dynamic modelling integrates biological and economic components of the systems calibrated with ecological and economic data, and accounts for uncertainties and multiple objectives.

This thesis has four main objectives:

- Development of bio-economic models of the French Bay of Biscay (BoB) demersal mixed fishery and of the Australian Northern Prawn Fishery (NPF) which capture the key biological and economic processes governing these fisheries. The generality of their modelling is deliberately maintained, thereby allowing potential future extensions and their application to other fisheries and economic and management contexts.

- Comparison of fishing strategies in terms of viability and bio-economic risks for BoB and NPF fisheries.

- Identification of viable fishing strategies for both case studies.

- Identification of main drivers of ecological-economic viability regarding both fisheries.

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1.5 Case studies

Two mixed fisheries are analysed in this thesis: the French Bay of Biscay (BoB) demersal mixed fishery and the Australian Northern Prawn Fishery (NPF). These two fisheries are relevant to investigate the question of how to manage multi-species fisheries in a stochastic context taking into account multiple objectives regarding specifically the biological and economic sustainability of the fisheries. These two fisheries are multi-species, and both make use of multiple fishing strategies, with direct and indirect impacts on the ecosystems. Both fisheries use indeed non-selective trawl technology inducing important by-catches and discards. Furthermore they are of great commercial and industrial interest. Therefore their sustainable management is a major societal concern. While drawing on a common methodology, this thesis examines the two cases in a way that recognises their different histories, social and political contexts and captures differences in management strategies and objectives. A summary of the main features of both case studies is given in box 4.

1.5.1 French Bay of Biscay demersal fishery (BoB)

The Bay of Biscay demersal mixed fishery operates in divisions VIIIa and b of the ICES grid and includes French, Spanish and Belgian fishery fleets. This thesis focuses on the French fleets. The main gears used in these fisheries are trawl, gill-net and longline, and all induce variable levels of impacts on a wide range of species. Among the 200 species caught in the Bay of Biscay, three of the most important species in percentage of the total French national landing value include Norway lobster (*Nephrops norvegicus*, 6%), Hake (*Merlucius merlucius*, 7%) and Sole (*Solea solea*, 11%). The management of the Bay of Biscay demersal fisheries mainly relies on conservation measures: a Total Allowable Catch (TAC) revised each year, a minimum landing size (MLS), and a minimum trawl mesh size. In accordance with the framework of the Common Fishery Policy reform (EC, 2009), a multi-species management plan for the Bay of Biscay is to replace the former mono-specific management plan on Sole and Hake. The analysis implemented in this thesis will provide important evidence of management effectiveness that can inform the development of the new management plan.

1.5.2 Australian Northern Prawn Fishery (NPF)

The NPF, located off Australia's northern coast and established in the late 1960s, is a multi-species trawl fishery based on several tropical prawn species, each with different biology. The main revenue of the fishery (95% of the total annual landed catch value) comes from an unpredictable naturally fluctuating resource, the white banana prawn (*Penaeus merguiensis*), and a more predictable resource, comprising two tiger prawn species (*Peaneus semisulcatus* and *Penaeus esculentus*).

The fishery operates over two 'seasons' spanning the period April to November with a mid-season closure of variable length from June to August. The fishery effectively consists of two sub-fisheries that are (to a large degree) spatially and temporally separate. The 'banana prawn sub-fishery' is a single species fishery based on the white banana prawn, while the 'tiger prawn sub-fishery' is a mixed species fishery targeting grooved and brown tiger prawns, as well as blue endeavour prawns (*Metapenaeus endeavouri*) which are caught as by product. The NPF was the first fishery in Australia to adopt biomass at maximum economic yield (MEY) as its management target (Woodhams et al., 2011). The fishery is primarily managed through input controls, mainly in the form of restricted quantities of tradeable units of effort (based on the length of trawl net headrope) and seasonal closures. Seasonal closures are in place to protect small prawns (closure from December to March), as well as spawning individuals (June to August closure). In the future, the fishery will be moving to an individual transferable quota (ITQ) management regime. This fishery is one of Australia's most valuable federally managed commercial fisheries, and since its establishment in the late 1960s has regularly returned positive profit (Rose and Kompas, 2004). However, in recent years the fishery has experienced a decline in value as a result of the increased supply of aquaculture-farmed prawns to both domestic and international markets, strong Australian currency and increasing fuel prices (Punt et al., 2011). The bio-economic model and co-viability analysis developed for this fishery allows for the exploration of alternative management strategies for different economic scenarios.

Box 4: BoB and NPF features.

The table 1.1 summarized the main features of both fisheries studied in this thesis.

Table 1.1: Features of Bay of Biscay demersal mixed fishery and Northern Prawn Fishery.

Features	**Bay of Biscay demersal fishery**	**Northern Prawn Fishery**
Countries involved	France, Belgium, Spain	Australia
Fleets	4 main fleets (French vessels)	2 sub-fisheries
Number of vessels	577 in 2008, 548 in 2009	52 since 2007
Annual effort (days)	101 441 in 2008	8 044 in 2010
Fishing methods	Bottom otter trawl and gill net	Otter (prawn) trawl
Key target and by-product species	Hake, Sole, *Nephrops*, Anglerfish, Megrim, Sea bass, Pollack, cephalopods	White banana prawn, Brown tiger prawn, Grooved tiger prawn, Blue endeavour prawn, Red endeavour prawn, Red-legged banana prawn
Other by-product species	More than a hundred species including cuttlefish, whiting and small-spotted catshark	Bugs, Finfish, Redspot king prawn, Western king prawn, Scallops, Scampi, Squid
Gross Value of production	Turnover amounted to 200 million €in 2009	AU$ 91.6 million in 2009–10
Net economic return	24 million €in 2009	AU$ 11.9 million in 2009-10
By-catch	Benthos and demersal elasmobranch as small sharks	Sea snakes, sharks, rays, sawfish, turtles, sponge, bryozoans, and gorgonians
Impacts of trawling	Impact from *Nephrops* trawling on the sediment compound, especially of the 'Grande Vasière' (nursery)	Identified impacts on seabed and benthic communities seem to operate at smaller local habitat.
Management objectives	Management objective by species for sustainable exploitation of Northern Hake and Sole (management plans).	(i) Ensuring sustainable exploitation of the resource, (ii) maximising economic efficiency , (iii) implementing efficient and cost effective management of the fishery, and (iv) reducing to a minimum the by-catches
Management target	Biological reference point: Biomass limit of precaution (B_{pa}) for Hake and Sole. Approach toward Maximum Sustainable Yield (MSY)	Maximum Economic Yield (MEY) for the tiger prawn sub-fishery as a whole (including by-product of endeavour prawns)
Management methods	Licences for Sole and *Nephrops* fleets, Total Allowable Catch (TAC) revised each year, minimum landing size (MLS) and a minimum trawl mesh size	Input controls: individual tradeable gear units, limited entry, gear restrictions, area closures, seasonal closures and time-of-day closures

1.6 Structure of the thesis

This thesis investigates the use of bio-economic risks management methods and especially stochastic co-viability approach (CVA) as a decision-support tool to assist in evaluating management strategies for sustainable regulation of mixed fisheries in different contexts. Bio-economic models of two contrasted case studies are developed and stochastic analyses, especially in terms of co-viability, are applied to them.

First, chapters 2 and 3 study the French Bay of Biscay (BoB) demersal mixed fishery. The issue of joint production within mixed fisheries is raised in chapter 2, which is based on a simplified multi-species and multi-fleet bio-economic model with technical interactions. Two species and two aggregated 'fleets' are explicitly represented. Stochastic co-viability analysis is used through a prospective approach to identify fishing strategies that satisfy both biological conservation and economic sustainability in this multi-species and multi-fleet context. But what would be the consequences for our results of the addition of another species in the analysis? Moreover, would it change the conclusions on management options for sustainability if, instead of having 'aggregated' fleets, the French vessels were grouped based on their main gear used, structure of landings, costs and length-class? To answer these questions, the bio-economic model presented in chapter 2, is improved and extended in chapter 3. Main extensions include the integration of a third species and set of vessels targeting this species. The aggregation of vessels in sixteen sub-fleets allows a more social analysis of management strategies. Viable management strategies preserving Spawning Stock Biomass (SSB) of every species and maintaining the economic profitability of the various fishing sub-fleets are investigated. The simulations allow also comparison of the trade-offs associated with different allocations of fishing effort across sub-fleets, under various economic scenarios.

The use of the co-viability approach to assist mixed fisheries sustainable management, is further explored in chapter 4 and 5 through another case study: the Australian Northern Prawn Fishery (NPF). Like the BoB fishery, the NPF is an industrial fishery with a great economic importance. However its history and policy managements differ markedly from BoB (notably management objective aiming for MEY for the NPF, compared to MSY for the BoB fishery), which makes its study relevant to assessing the general interest of using the CVA approach as a decision-support tool for mixed fisheries management. Chapter 4 describes the bio-economic modelling undertaken to represent the major biological and economic processes governing the NPF. This multi-species, bio-economic model with dynamic allocation of fishing effort across multiple sub-fisheries and fishing strategies is used to examine the trade-offs between mean economic performance of the fishery and the variance of this performance, under a range of economic scenarios and strategies with respect to fleet capacity and effort allocation. Then, chapter 5 analyses the application of the stochastic co-viability approach to the NPF, integrating a set of management objectives reflecting current issues in the fishery. This set includes biological, economic and biodiversity conservation objectives. To allow the implementation of a biodiversity conservation objective regarding the impacts of trawling on the broader biodiversity, the bio-economic model presented in chapter 4 is extended. Trade-offs between biological, economic and biodiversity conservation management objective constraints in highly uncertain context are investigated and viable management strategies identified.

Finally, chapter 6 provides a discussion of the results presented in the thesis, with emphasis on the two bio-economic models developed, on the viability analyses results obtained, and on the viable management strategies identified. Perspectives for future research are then provided.

1.7 Context of the thesis

This PhD is a co-tutelle between France and Australia, more specifically between the French Université de Bretagne Occidentale (UBO) in Brest and the Australian University of Tasmania (UTAS) in Hobart.

This thesis was co-funded by the French Institute for the Exploration of the Sea (Ifremer) and the UTAS-CSIRO (Commonwealth Scientific and Industrial Research Organisation) conjoint PhD program in quantitative marine science (QMS). This thesis is also part of the project funded by the ANR (Agence Nationale de la Recherche, the French national agency for funding research) and entitled ADHOC (Co-viability modelling for marine biodiversity and fisheries). Additional financial supports for travel were provided by the Australian National Network in Marine Science, the Fisheries Research and Development Corporation through the Australian Fisheries Economics Network, the University of Tasmania and the 'Région Bretagne'.

During the time of this thesis, I was hosted by various laboratories and institutions. While in France, two French joint research units hosted me : the UMR CERSP, department of Ecology and Biodiversity Management, MNHN-CNRS (French National Museum of Natural History - French National Center for Scientific Research) in Paris and the UMR AMURE, department of marine economics, Ifremer-UBO, Brest. The Australian CSIRO marine and atmospheric research department in Brisbane, the Institute for Marine and Antarctic Studies (IMAS) in Hobart and the School of Economics and Finance in Hobart hosted me while in Australia.

Chapter 2. A stochastic viability approach to ecosystem-based fisheries management

Published in *Ecological Economics*:

Doyen L., Thébaud O., Béné C., Martinet V., **Gourguet S.**, Bertignac M., Fifas S. (2012).A stochastic viability approach to ecosystem-based fisheries management, *Ecological Economics*, 75, 32–42.

For the sake of consistency between chapters, the initial published terms of ‘ecological viability’ and ‘economic viability’ have been respectively changed in ‘biological viability’ and ‘socio-economic viability’; and the following variables have been changed as described:

Variable	Ecological Economics	Thesis
Number of vessels	k	K
Catch	C	Y
Discount rate	ρ	δ
Baseline strategy	BLA	SQ
Net present value	PV	NPV

This published paper presents a bio-economic model developed for the Bay of Biscay (BoB) mixed demersal fishery. The annual multi-species and multi-fleet, age-structured bio-economic model includes technical interactions involving Norway lobster (*Nephrops norvegicus*) and Hake (*Merlucius merlucius*) and two explicit fleets harvesting these species. A stochastic co-viability analysis is used through a prospective approach to identify fishing strategies that satisfy both biological conservation and socio-economic sustainability in this multi-species and multi-fleet context.

Abstract

Academia and management agencies show a growing interest for Ecosystem-Based Fishery Management (EBFM). However, the way to operationalize this approach remains challenging. The present paper illustrates how the concepts of stochastic co-viability, which accounts for dynamic complexities, uncertainties, risk and sustainability constraints, can be useful for the implementation of EBFM. In the present case, this concept is used to identify fishing strategies that satisfy both biological conservation and socio-economic sustainability in a multi-species, multi-fleet context. Socio-Economic Viability Analysis (EVA) and the broader Co-Viability Analysis (CVA), are proposed to expand the usual Population Viability Analysis (PVA) and precautionary approach. An illustration is proposed, using data of the fisheries of Bay of Biscay (France) exploiting the stocks of *Nephrops* and Hake. Stochastic simulations show how CVA can guarantee both biological (stock) and economic (profit) sustainability. Using 2008 as a baseline, the model is used to identify levels of fishing intensity that ensure such co-viability.

Keywords: Ecosystem based fisheries management, Viability, Stochastic, *Nephrops*, Hake

2.1 Introduction

Marine fisheries resources are under extreme pressure worldwide. According to recent estimates (Garcia and Grainger, 2005; FAO, 2010), three quarters of the world's fish stocks are fully exploited or over-exploited and the proportion of those stocks that are too intensively exploited is growing. As a consequence, the sustainability of the world fisheries is now becoming a major concern for national and international agencies. As a consequence, fisheries management increasingly involves restoration and conservation objectives, along with the more conventional biological and economic objectives that are identification of desirable levels of fish resources, catches, and profitability from fishing. Examples are the restoration plans discussed and/or adopted by the European Commission for several collapsed stocks in the E.U. waters, or the international commitment by the countries present at the 2002 Johannesburg Summit on Sustainable Development to return fisheries to levels allowing their maximum sustainable yield by 2015.

Indicators and their associated reference points are key to the implementation of such sustainability strategies. For example, the objective of the precautionary approach promoted by the

International Council for the Exploitation of the Sea (ICES) is to maintain spawning stock biomass above a limit reference point B^{lim}, while keeping fishing mortality below a limit F^{lim}. To achieve this in a context of high uncertainty on the current level of both indicator and reference point, operational precautionary reference points B^{pa} and F^{pa} are used. This approach proposes to preserve a minimum quantity of reproducers to avoid recruitment accidents that would endanger the sustainability of the stock and consequently the fishery. Although such a precautionary approach has had positive effects in Europe on some severely depleted stocks, the overall state of European fish stocks remains grim. A first criticism of this approach is that it adopts a viewpoint which is too 'ichthyocentric', as it focuses on the conservation of fish populations and stocks only. Social and economic considerations are not included and left to the fishery managers' calls. Excluding these considerations from the assessment of trade-offs associated with alternative management strategies leaves managers with limited scientific ground on which to base decisions, a lack of clarity in the objectives pursued, and greater potential for stakeholder conflict and resistance to the implementation of management options, particularly in the face of uncertainty. Formal approaches to including multiple objectives in evaluation of management options, such as Management Strategy Evaluations (Sainsbury et al., 2000), has been shown to positively contribute to agreement being reached on fisheries regulation options. Increasingly, it is also stressed that single stock assessments should be replaced by more complex multi-species and/or multi-fisheries analyses that account for interactions between species and/or fisheries. For example, in the United States, the National Marine Fisheries Service (NMFS) began to include ecosystem effects on stocks and fishery effects on ecosystems in its individual groundfish stock assessment reports to the North Pacific Fisheries Management Council in 2002. As part of its strategic goals, the NMFS has now replaced single-species management with ecosystem-based management, balancing ecological and social objectives. There is nowadays widespread acceptance that more integrated perspectives are needed to manage marine fisheries sustainably.

Ecosystem-based fishery management (EBFM) approaches advocate an integrated management of marine resources (FAO, 2003; Jennings, 2005). Such an approach requires accounting for the impacts of fishing on the wider ecosystem, and considering the complexity of ecological mechanisms, encompassing fish population and fish community dynamics, spatial processes, and environmental (habitat, climatic) uncertainties. Attention must also be paid to complexities and uncertainties related to economic drivers of the fisheries, including non-compliance and effort adjustment in multi-fleet context. In the face of such diversified, difficult and ambitious goals, a large number of models have been proposed for the exploration of possible scenarios for fisheries all around the world. Plagányi (2007) provides an overview of the main types of relevant modelling approaches and analyzes their relative merits and limitations in an ecosystem approach context. modelling approaches and metrics useful for planning, implementing, and evaluating EBFM are also discussed in Marasco et al. (2007), with particular emphasis on Management Strategy Evaluation. Hall and Mainprize (2004) argue that the expansion of single-species reference points to take account of the non-target species of a fishery is tractable and desirable. The need to develop indicators that account for the ecosystem-wide impacts of fishing has also attracted growing attention, as pointed out in Cury and Christensen (2005) -see also Rice (2000). Sanchirico et al. (2008) argue that risk management is a major ingredient for EBFM and propose to use the portfolio theory to operationalize the concept, while Link (2005) emphasizes the need for multi-criteria consideration to achieve ecological, economic and social objectives.

This article deals with the sustainable management of multi-species, multi-fleet fisheries, following the EBFM approach. For this, it adopts a general modelling approach relying on the stochastic viability framework (De Lara and Doyen, 2008) and proposes to illustrate its applicability to the integrated management of a mixed fishery, based on the case study of the Bay of Biscay fisheries (France), in which several fishing fleets exploit a set of species including *Nephrops* (*Nephrops norvegicus*) and Hake (*Merluccius merluccius*). The *Nephrops* trawler fleet is one of the largest French fleet segments of the Bay of Biscay[1]. In 2003, the 234 *Nephrops* trawlers (1/4 of the total trawler fleet of the Bay) generated a sales value of 82,4 million Euros, of which approximately 40% was from *Nephrops* (Macher et al., 2008). The technique of *Nephrops* trawling, however, lacks selectivity, both in terms of species, and in terms of catch size. This results in important quantities of by-catch of various age-classes affecting several species present in the fishing ground, including juvenile *Nephrops*, and Northern Hake. Part of this by-catch is discarded, particularly those fish smaller than the legal landing size limit.

[1]Divisions VIIIa and VIIIb of the ICES statistical areas.

Northern Hake is conjointly caught by several mixed demersal fisheries, and is an economically important species for several major European fishing fleets[2]. Spain accounts for the main part of the landings with nearly two thirds of landings in recent years, while French fleets contribute to a quarter of the total landings, and other countries (UK, Denmark, Ireland, Norway, Belgium, Netherlands, Germany, and Sweden) contributing small amounts (STECF, 2008). Total landings decreased steadily from 66,500 tonnes in 1989 to a low 35,000 tonnes in 1998, and fluctuated at around 40,000 tonnes since then. During the same period, the estimated biomass of reproducers (spawning stock biomass SSB) decreased to levels close to the minimum biomass level recommended by the ICES, $B^{\text{lim}} = 100,000$ tonnes. In 2003, the ICES considered that this stock presented a risk of collapse. Various management regulations were then introduced to attempt to restore the sustainability of the Hake fishery. In particular a series of technical measures was proposed (EC Council Regulations No 1162/2001, 2602/2001 and 494/2002) to improve the selectivity of the fishing gear and protect juveniles. Subsequently a Hake recovery plan was introduced (Council regulation EC Reg. No 811/2004), recommending a reduction in fishing mortality to the precautionary level $F^{\text{pa}} = 0.25$ to allow for a recovery of the SBB above $B^{\text{pa}} = 140,000$ tonnes.

This article builds directly on these concerns. It deals with the sustainability of both Hake and *Nephrops* fisheries in the Bay of Biscay. It considers a typical problem encountered in attempting to adopt an ecosystem-wide perspective which involves managing simultaneously the harvest of several species, rather than adopting a single stock approach. In addition, the analysis accounts for age-structured population dynamics with uncertainty on recruitment, together with interactions between fisheries through by-catch. The aim is to determine how the economic viability of both *Nephrops* and Hake fisheries can be maintained while, at the same time, allowing for the biological conservation of both stocks and in particular the recovery of the Hake population. To achieve this, we adopt a viability framework of analysis.

The viability (or viable control) approach aims at identifying desirable combinations of states and associated controls that ensure the ‘good health’, safety or effectiveness of the system (Bene et al., 2001). By identifying the conditions that allow desirable objectives to be fulfilled over time, considering both present and future states of a renewable resources system, the viability

[2]The Northern Hake stock spreads across divisions IIIa, IV, V, VI, VII and VIII of the ICES.

approach conveys information relevant for policy and decision makers. Viability does not aim at identifying optimal or steady state paths for the co-dynamics of resources and exploitation but, instead, provides feasible trajectories and controls satisfying both economic and environmental constraints. In this respect, it is a multi-criteria approach (Baumgärtner and Quaas, 2009). The approach is also closely related to the maximin, or Rawlsian, approach (Heal, 1998) with respect to intergenerational equity (Martinet and Doyen, 2007). Tichit et al. (2007) show how the so-called Population Viability Analysis (PVA) developed in conservation biology (e.g. Morris and Doak (2002)) addresses issues comparable to those of the viability approach. Viability analysis has been applied to renewable resources management and especially to fisheries (see, e.g., Béné and Doyen (2000); Bene et al. (2001); Doyen and Béné (2003); Eisenack et al. (2006); Martinet et al. (2007)), but also to broader (eco)-system dynamics (Mullon et al. (2004); Chapel et al. (2008); Doyen et al. (2007)). Cury et al. (2005) illustrate how the viability approach can potentially be useful to integrate ecosystem considerations into fisheries management. Relationships between viability, sustainable management objectives and reference points as adopted in the ICES precautionary approach are discussed in De Lara et al. (2007).

In the present paper the viability framework is used to analyse a multi-species and multi-fleet model inspired by Bay of Biscay fisheries. The focus is on Hake and *Nephrops* species impacted mainly by *Nephrops* trawlers and gill-netters targeting Hake. First a Population Viability Analysis (PVA) identifies the appropriate viable combinations of fishing intensity that ensure the biological conservation of the two stocks considered simultaneously. A socio-economic viability approach (EVA) is then developed to identify the combinations of harvesting mortality that ensure the socio-economic viability of the two fisheries. Finally, a co-viability approach (CVA) aimed at reconciling both PVA and EVA objectives is proposed.

The paper is organized as follows. Section 2.2 describes the bio-economic model of the system, together with the biological and socio-economic constraints. Section 2.3 presents the results related to the PVA, EVA and CVA approaches. The following section discusses those results and explores the potential usefulness of co-viability in relation to ecosystem-based fisheries management. A series of conclusion follows.

2.2 The bio-economic model

We consider the two species, *Nephrops* and Hake, exploited by several fleets using different 'metiers' (trawlers targeting *Nephrops* or targeting demersal fish, gill-netters targeting Hake, long-liners, etc.) based on information provided in ICES (2009). To capture this complexity but keep it manageable, we group these fleets into three generic sets of fleets operating in the Bay of Biscay: one constituted of the *Nephrops* trawlers, one constituted of gill-netters targeting Hake grouped together under a general Hake fleet category and a third 'fleet' termed 'others' constituted by all other vessels impacting Hake or *Nephrops*. The economic analysis focuses on the two first fleets.

2.2.1 A multi-species, multi-fleet and age-structured dynamics

We develop an age-structured population model derived from the standard fish stock assessment approach (Quinn and Deriso, 1999). Time $t \in \mathbb{N}$ is measured in years. Let $A = 9 \in \mathbb{N}^*$ denote a maximum age limit, and $a \in \{1, \dots, A\}$ an age class index, all expressed in years. The state variables $N_{s,a}(t) \in \mathbb{R}_+^{2A}$ are the abundances of species $s = 1, 2$ at age[3] a, where index $s = 1$ refers to Hake and $s = 2$ refers to *Nephrops*. Similarly the index $f = 1$ is used to denote the fleet targeting Hake while index $f = 2$ denotes the fleets targeting *Nephrops*. The third fleet $f = 3$ encompasses all other vessels involved. For age $a = 1, \dots, A-1$ and each species $s = 1, 2$, the dynamics of the two species are assumed to follow the discrete equation system:

$$N_{s,a+1}(t+1) = N_{s,a}(t) \exp\left(-M_{s,a} - \sum_{f=1}^{3} u_f(t) F_{s,a,f}\right), \tag{2.1}$$

where

- $M_{s,a}$ is the natural mortality rate of individuals of species s at age a;
- $F_{s,a,f}$ is the current (here 2008) fishing mortality rate of species s at age a due to fleet f, and
- the controls $u_f(t)$ are multipliers of the current fishing mortality $F_{s,a,f}$ for fleets f. As we assume that there is no control on the third fleet 'others', we fix its fishing mortality by writing

$$u_3(t) = 1.$$

[3] The last class $N_{s,A}(t)$ is the number of individuals of age greater than $A-1$.

More globally, the vector $u = (1, 1, 1)$ represents the fishing baseline for year $t_0 = 2008$.

The parameter values used in the analysis are detailed in appendix (table 2.1 for Hake and table 2.2 for *Nephrops* respectively). They are derived from ICES databases[4], working group WGHMM (ICES, 2009) and the Ifremer databases[5]. Note that the mortality of Hake ($s = 1$) due to the *Nephrops* fleet's ($f = 2$) by-catch is included in the dynamics through the positive parameters $F_{1,a,2}$.

Recruitment involves complex biological and environmental processes that vary over time. The recruits $N_{s,1}(t+1)$ for each species are therefore supposed to be uncertain functions of the spawning stock biomass

$$N_{s,1}(t + 1) = \varphi_s\Big(\mathrm{SSB}_s(N_s(t)), \omega(t)\Big), \tag{2.2}$$

where

- $\mathrm{SSB}_s(N_s)$ is the spawning stock biomass of species s

  $$\mathrm{SSB}_s(N_s) = \sum_{a=1}^{A} \gamma_{s,a} \upsilon_{s,a} N_{s,a}\,, \tag{2.3}$$

  with $(\gamma_{s,a})_{a=1,\dots,A}$ being the proportions of mature individuals at age a and $(\upsilon_{s,a})_{a=1,\dots,A}$ being the weights of individuals at age a,

- the function φ_s represents the specific stock-recruitment relationship of each species s,

- $\omega(t)$ stands for the uncertainties (environmental or demographic) affecting the stock recruitment relationships through different possible scenarios Ω.

In our case, following STECF (2008), the recruitment relationship for the Hake stock is set through an Ockham-Razor function as in O'Brien et al. (2002), that is,

$$\varphi_1(\mathrm{SSB}, \omega) = \begin{cases} \omega_1 \rightsquigarrow \mathcal{N}(\overline{B}_1, \sigma_1) & \text{if} \quad \mathrm{SSB} \geq B_1^{\mathrm{lim}} \\ \mathrm{SSB}\,\frac{\overline{B}_1}{B_1^{\mathrm{lim}}} & \text{if} \quad \mathrm{SSB} < B_1^{\mathrm{lim}} \end{cases} \tag{2.4}$$

where $B_1^{\mathrm{lim}} = 54,521$ tonnes, $\mathcal{N}(\overline{B}_1, \sigma_1)$ stands for a Gaussian distribution[6] and $\overline{B}_1 = 241,776$ for the estimated mean of the Hake recruitment (in tonnes) over the period 1992-2006 while the

[4]http://www.ices.dk/datacentre/StdGraphDB.asp
[5]http://wwz.ifremer.fr/peche/Le-role-de-l-Ifremer/Observation/
[6]We have also tested a uniform distribution but it does not significantly modify the whole results.

standard deviation $\sigma_1 = 58,459$ measures the dispersion of the recruitment. Note that the risk of collapse of the stock is captured by the linear declining value for SSB_1 below the critical level $B_1^{\mathrm{lim}} = 54,521$ tonnes.

The *Nephrops* recruitment is also assumed to be subject to uncertainty but with no density dependence as explained in (FAO, 2006). The intuition underlying such assumption is that the quality of the environment (the sea bottom) affects more the recruitment process for *Nephrops* than the spawning stock biomass. Consequently, using the 1992-2006 data, the following recruitment Gaussian relationship is used for the *Nephrops* stock:

$$\varphi_2(\mathrm{SSB}, \omega) = \omega_2 \rightsquigarrow \mathcal{N}(\overline{B}_2, \sigma_2) \tag{2.5}$$

where $\overline{B}_2 = 699,387$ tonnes is the mean of the 1987-2008 *Nephrops* SSB and the standard deviation $\sigma_2 = 166,158$ represents the dispersion of the recruitment. Random variables w_1 and w_2 are assumed to be independent.

2.2.2 Catches and gross incomes

For each period t, the exploitation of the two species is described by the catches $Y_{s,a,f}(t)$. These catches are function of the fishing mortality intensity $u_f(t)$ and abundances $N_{s,a}(t)$ through the Baranov catch equations :

$$Y_{s,a,f}(t) = N_{s,a}(t) u_f(t) F_{s,a,f} \frac{1 - \exp\left(-M_{s,a} - \sum_{f=1}^{3} u_f(t) F_{s,a,f}\right)}{M_{s,a} + \sum_{f=1}^{3} u_f(t) F_{s,a,f}} \tag{2.6}$$

The gross income of each fleet's catch is then estimated by incorporating the market prices of the species, recorded for different commercial categories (corresponding to different age groups), along with the estimates of the discard rates (see tables 2.1 and 2.2 for details), so that:

$$\mathrm{Inc}_f(t) = \sum_s p_{s,a} \sum_{a=1}^{A} \upsilon_{s,a}\, Y_{s,a,f}(t)(1 - d_{s,a,f}) \tag{2.7}$$

where

- $p_{s,a}$ is the market price of individuals of species s at age a,
- $\upsilon_{s,a}$ is the mean weight of individuals of species s at age a, and

- $d_{s,a,f}$ is the discard rate of individuals of age a by the fleet f.

Fish price data used in the model are based on first sale prices for the two species and for different market categories (defined in terms of the size/age of fish) recorded in French harbors, and obtained from the fisheries information system operated by Ifremer (see footnote 5). Discard ratios were calibrated based on the data available in the ICES working group in charge of the assessment of the two stocks.

2.2.3 Profits

The economic value of each fleet relies on its profitability accounting for both gross income and fishing costs including fixed and variable costs. The focus is here on fleets 1 and 2. The profit π_f of the fleets $f = 1, 2$ is estimated as follows:

$$\pi_f(t) = \alpha_f \mathrm{Inc}_f(t) - (c_f^{var} e_f(t_0) + c_f^{fix}) \mathrm{K}_f(t_0) u_f(t). \tag{2.8}$$

where

- c_f^{var} is the total variable cost by fishing effort unit (day at sea) and by vessel of fleet f including fuel cost, oil, supplies, ice, bait and device cost,
- $e_f(t_0)$ is the mean value of fishing effort (number of day at sea) by vessel of the fleet f for the year of reference (2008),
- c_f^{fix} correspond to the fixed costs by vessel of the fleet f including licenses, maintenance and repair costs, insurance premium, amortizing and interests. Their values are also set through the reference year 2008,
- $\mathrm{K}_f(t_0)$ is the number of vessel by fleet f for the reference year 2008,
- α_f stands for the rate of income of the fleet f derived from the catches of other species not taken into account in the current model. The values are based on the data of gross incomes 2008 (Ifremer, SIH, DPMA) and assumed to be constant over the simulation period.

The whole set of these economic parameters is displayed in the table 2.3 in appendix.

The connection between fleets' fishing mortality, effort and number of vessels is captured by the following relation:

$$F_{s,a,f} = q_{s,a,f} e_f(t_0) \mathrm{K}_f(t_0), \tag{2.9}$$

where catchability $q_{s,a,f}$ is the fishing mortality of species s at age a by unit of fishing effort and by vessel of fleet f. The catchabilities are supposed constant over the simulation period.

2.3 A viability diagnosis

We now examine the sustainability of the two fisheries through three approaches. The first one, termed PVA (Population Viability Analysis), is basically an ichthyocentric approach which puts emphasis on stock conservation through the adoption of a precautionary approach. The second approach, called EVA (Socio-Economic Viability Analysis), gives priority to economic sustainability through the adoption of guaranteed profit constraints. The third approach, CVA (Co-Viability Analysis), considers both population and socio-economic viability objectives conjointly.

PVA: Under the PVA approach, the objective is to maintain the sustainability of the marine resources through the adoption of constraints on minimum spawning biomass, as it is the case in the ICES precautionary approach, namely:

$$\mathrm{SSB}_s(N_s(t)) \geq B_s^{\mathsf{pa}}, \qquad s = 1, 2 \qquad t = t_0, \ldots, T, \tag{2.10}$$

where $t_0 = 2008$ stands for the baseline year and $T = 2028$ is the final time horizon (20 years). In our case, the minimum spawning biomass requirement concerns the stock of Hake only, and we write

$$\mathrm{SSB}_1(N_1(t)) \geq B_1^{\mathsf{pa}} = 75,784, \qquad t = t_0, \ldots, T. \tag{2.11}$$

where B_1^{pa} corresponds to the (new) precautionary level estimated for the Hake by the ICES. As there is no ICES precautionary or limit spawning biomass level for *Nephrops*, we set its viability threshold to zero with $B_2^{\mathsf{pa}} = 0$.

EVA: Under the EVA approach, the objectives are related to the socio-economic viability of the fleets. Here we choose to represent this socio-economic viability through the profit $\pi_f(t)$. The objective is to maintain this profit positive for both fleets 1 and 2, namely

$$\pi_f(t) > 0, \qquad f = 1, 2, \qquad t = t_0, \ldots, T. \tag{2.12}$$

where again $t_0 = 2008$ and $T = 2028$.

CVA: The co-viability approach requests that both stock conservation and socio-economic viability of the fleets are guaranteed conjointly. This requires complying both with the SSB constraints (2.10) and the socio-economic constraints (2.12).

For all three approaches, we choose to deal with uncertainty in a probabilistic sense. We therefore perform a stochastic viability analysis. For this, we consider a probability $\mathbb{P}$ on scenarios $\omega(.) \in \Omega$, a confidence level $\beta \in]0, 1]$, and we aim at identifying the controls (fishing multipliers u_1 and u_2) that satisfy the following condition[7]:

$$\mathbb{P}_{\omega(.)}\Big(N(t) \text{ satisfies the constraints, } t = t_0, \ldots, T\Big) \geq \beta \tag{2.13}$$

In the case of the PVA, using the stochastic viability approach means that we consider the constraint (2.10) and compute the viability probability $\mathrm{PVA}(u_1, u_2)$ associated to the SSB constraint (2.10):

$$\mathrm{PVA}(u_1, u_2) = \mathbb{P}_{\omega(.)}\Big(N(t) \text{ satisfies (2.10), } t = t_0, \ldots, T\Big) \tag{2.14}$$

Similarly, in the EVA context, $\mathrm{EVA}(u_1, u_2)$ denotes the socio-economic viability probability associated to the socio-economic constraints (2.12), namely:

$$\mathrm{EVA}(u_1, u_2) = \mathbb{P}_{\omega(.)}\Big(N(t) \text{ satisfies (2.12), } t = t_0, \ldots, T\Big) \tag{2.15}$$

Finally for the CVA perspective, $\mathrm{CVA}(u_1, u_2)$ denotes the co-viability probability associated to both biological requirements (2.10) and socio-economic constraints (2.12):

$$\mathrm{CVA}(u_1, u_2) = \mathbb{P}_{\omega(.)}\Big(N(t) \text{ satisfies (2.10) and (2.12), } t = t_0, \ldots, T\Big) \tag{2.16}$$

In terms of decision, given a level of risk $1 - \beta$, we aim at identifying viable fishing intensity

[7]In a more formal way, stochastic viability analysis refers to the identification of the stochastic viability kernel Viab_β De Lara and Doyen (2008) defined as

$$\mathrm{Viab}_\beta(t_0) = \left\{ N(t_0) \;\middle|\; \mathbb{P}_{\omega(.)}\Big((N(t), u(t)) \text{ satisfies the constraints, } t = t_0, \ldots, T\Big) \geq \beta \right\}.$$

vectors, namely $u(t) = (u_1, u_2)$ expressed as multipliers of the baseline $u = (1, 1)$, that satisfy viability condition (2.13). In this context, of particular interest are the controls that maximize the viability probabilities, that is, $\max_u \mathrm{PVA}(u)$, $\max_u \mathrm{EVA}(u)$, and $\max_u \mathrm{CVA}(u)$.

2.4 Results

We use numerical computations performed with the Scilab software[8]. We focus first on estimations of viability probability PVA(u), EVA(u) and CVA(u) (see below). Based on those preliminary results, we describe in greater detail the outcomes of five specific scenarios. The first one corresponds to a 'status quo' scenario u^{SQ} where fishing efforts are kept at the 2008 baseline level. The next three scenarios depict situations corresponding to the maximisation of the three viability approaches described above: the biological strategy u^{PVA}, the socio-economic strategy u^{EVA} and the co-viability strategy u^{CVA}. Finally, we examine the viability performances of a more conventional scenario termed present value u^{NPV} relying on the maximisation of discounted total rents. Projections and viability probability are computed for $T = 20$ years starting from the initial stock abundance $N(t_0)$ at year $t_0 = 2008$ estimated as in tables 2.1 and 2.2 from ICES (2009). For each fishing strategy $u(t) = u_0$, the viability probability is approximated by calculating the percentage of viable trajectories among 100 simulated trajectories[9]. Each trajectory corresponds to different recruitment levels $\omega(t) = (\omega_1(t), \omega_2(t))$ initiated randomly. We first examine the shape of viability probabilities. Then we compare the outcomes of the five scenarios.

2.4.1 Population viability analysis

Figure 2.1(a) shows the numerical approximation of the viability probabilities for the PVA case. A viability 'frontier' appears in dark blue, delimiting a viability control space within which combinations of fishing intensity (u_1, u_2) are such that the viability probability to satisfy (2.10) is close to 100%, namely $\mathrm{PVA}(u_1, u_2) \approx 100\%$. Outside this viability space, i.e. above the frontier, the viability probability declines progressively toward zero in the red zone where fishing intensity is too high and condition (2.10) is not satisfied any longer namely $\mathrm{PVA}(u_1, u_2) \approx 0\%$. This corresponds

[8] Scilab is a free software (similar to Matlab) available on line at www.scilab.org. It is dedicated to scientific computations and is especially well-suited for the analysis and control of dynamic systems.

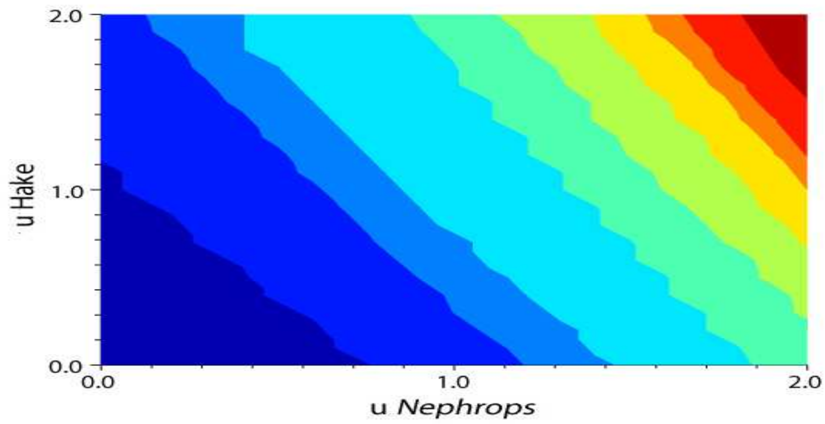
[9] By 'viable trajectories', we mean trajectories satisfying the constraints (2.10) and/or (2.12), depending on PVA, EVA or CVA contexts.

to combinations of high catch mortality that drive the stock of Hake below the precautionary level $B_1^{\mathrm{pa}} = 75,784$ tonnes. The 'edge' of the viability frontier indicates intermediate situations where $\mathrm{PVA}(u_1, u_2)$ lies between 100% and 0%. This corresponds to 'risky' fishing strategies for which the sustainability of the Hake stock is at stake. In particular, the status quo strategy $u^{\mathrm{sq}} = (1, 1)$ which is slightly above the frontier has not a satisfying PVA (in fact $\mathrm{PVA}(u^{\mathrm{sq}}) \approx 75\%$) informing on its underlying biological risk.

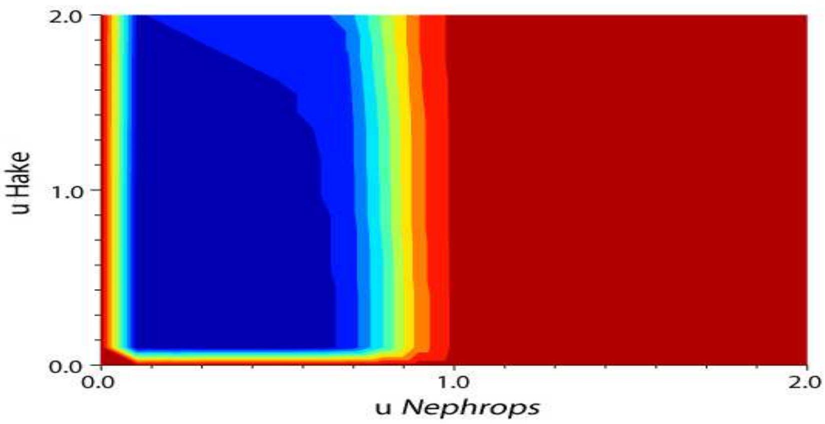
The position of the frontier is determined *inter alia* by the level of the SSB precautionary threshold. In particular, if one were to relax slightly the precautionary level (i.e. setting it lower than the current $B_1^{\mathrm{pa}} = 75,784$ tonnes, say, 70,000 tonnes), the frontier would shift upwards *ceteris paribus*.

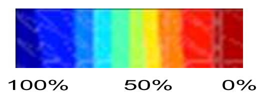
2.4.2 Socio-economic viability analysis

The results of the socio-economic viability probability EVA are shown in figure 2.1(b). A viability domain exists (in blue), which contains control states (u_1, u_2) for which the viability probability to guarantee profit condition (2.12) is close to 100%, i.e. $\mathrm{EVA}(u_1, u_2) \approx 100\%$. Outside the boundary of this viability space, the probability to maintain the socio-economic viability of the system decreases. Intermediate situations appear around of the viability space, where the probability declines rapidly from 100% to 0%. Beyond this, in color red, the probability to maintain the two fisheries' profitability is nil. The right part of the red area corresponds to non viable socio-economic situations due to *long-term* (indirect) effects of fishing on the stocks: excessive fishing mortality drives the stocks down to levels which are too low to ensure profitabilities. This especially occurs for large fishing mortality of Hake fleet. The dynamics at work on the two other sides of the viability space are of a different nature. The two sharp-edged boundaries running parallel to the axes reflect the *short-term* (direct) effects of the socio-economic constraints (2.12). They illustrate in particular the fact that minimum fishing mortality are necessary to ensure that constraint (2.12) is satisfied. Note that the status quo strategy $u^{\mathrm{sq}} = (1, 1)$ has an EVA close to 0%.

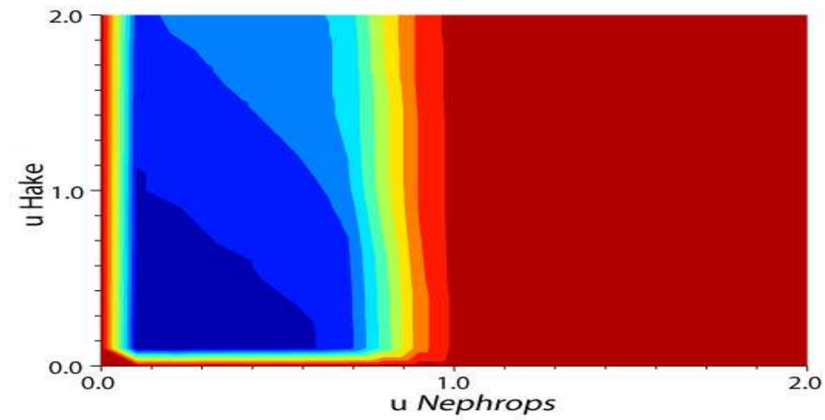


(a) Population viability probability PVA($u_1 = u_{Hake}, u_2 = u_{Nephrops}$)





(b) Socio-economic viability probability EVA(u_1, u_2)



(c) Co-viability probability CVA(u_1, u_2)

Figure 2.1: Viability probabilities PVA(u_1, u_2), EVA(u_1, u_2) and CVA(u_1, u_2) as a function of effort multipliers u_1 and u_2. The viability control space (probability$\approx$ 100%) is in blue. The non viability space (probability$\approx$ 0%) is in red.

2.4.3 Co-viability analysis

Finally, combining the biological and socio-economic constraints, figure 2.1(c) illustrates the results of the computation of the co-viability analysis CVA. In that case, the viability space is reduced to an area close (but not exactly) to the intersection of the PVA and EVA viability spaces. The short-run and long-run effects of the constraints described for the PVA and EVA cases still hold. In particular we recognize the direct short-term economic effect of constraint (2.12) on the lower and left side of the viability space and the effect of the biological constraint (2.10) on the upper side. The long-term effect of (2.12) is not visible as the constraint (2.10) affects the system before it can come into play. Note however that situations may occur where the long-term effect of (2.12) may appear before constraint (2.10). The fact that the co-viability control space does not exactly coincide with the intersection of the PVA and EVA viability spaces proves that complex and non linear mechanisms occur through the dynamics, interactions and uncertainties at play.

2.4.4 Status quo strategy: neither viable nor sustainable

We now examine in more detail the particular case of the status quo (or baseline strategy). This case corresponds to a scenario where the fishing intensity of the fleets is maintained at the 2008 level, that is

$$u^{\mathrm{sq}} = (1, 1).$$

Figure 2.2 displays ten trajectories randomly generated under this scenario reflecting recruitment uncertainties and stochasticity. The figure first shows that, this scenario is not biologically viable since the Hake biomass trajectories frequently pass under the ICES precautionary threshold $B_1^{\mathrm{pa}} = 75,784$ tonnes. Conjointly, the figure also reveals that this scenario is not viable from the socio-economic viewpoint especially for the *Nephrops* fleet: over the 20-year simulations, numerous profit trajectories become negative. The profitability of Hake fleet is also at stake although the risk is very low compared to the *Nephrops* trawlers. Overall, this indicates that 2008 has been a favourable year for catches and profits, but that this was not sustainable. Catch intensity could not be maintained at this level if biological and socio-economic viability of the system are to be ensured.

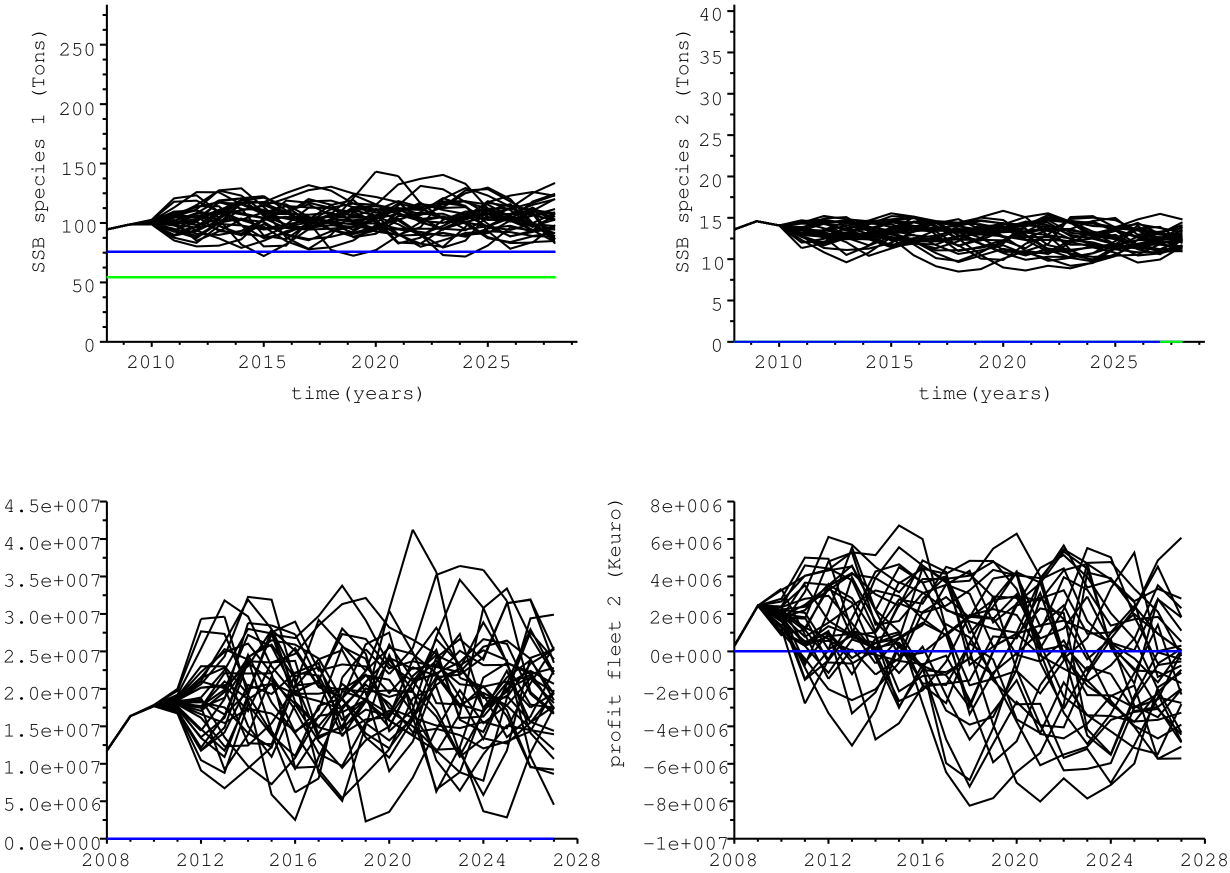


Figure 2.2: Trajectories (in black) under the 2008 baseline scenario $u^{SQ} = (1, 1)$ and viability thresholds (in blue). Top diagrams: spawning biomass SSB(t) for Hake (left) with its precautionary biomass level B^{pa} (in blue) and its limit level B^{lim} (in green), and *Nephrops* (right). Bottom: profits $\pi(t)$ for the two fleets with zero viability threshold in blue. See text for comments.

2.4.5 Optimizing scenario: a high bio-socio-economic risk

We then tested a fishing scenario maximizing the expected sum of discounted total net incomes of both fleets:

$$\mathrm{NPV} = \max_{u_1,u_2} \mathbb{E}\left[\sum_{t=0}^{T}\left(\frac{1}{1+r}\right)^t \sum_{f=1,2} \pi_f(t)\right] \tag{2.17}$$

where the discount rate is set to $r = 0.1$. The associated optimal effort multipliers take the values $u^{NPV} \approx (2.5, 0.25)$. Figure 2.3 displays ten trajectories randomly generated for this scenario. The figure shows that the paths are not very safe from the biological viewpoint, as the spawning biomass trajectories fall below the Hake limit reference point at several occasions over the 20 year horizon. This is due do the significant increase in the Hake fishing effort induced by the adoption of the maximization strategy. This large increase in catch effort does not, however, lead to a sustainable profitability of the Hake fleet. In other words, the strategy which consists in maximizing the present value of the total income is not simply biologically risky; it also implies that the socio-economic viability of the Hake fleet is not warranted. Note also the severe reduction in the *Nephrops* fleet's effort imposed by the strategy with $u_2 \approx 0.25$.

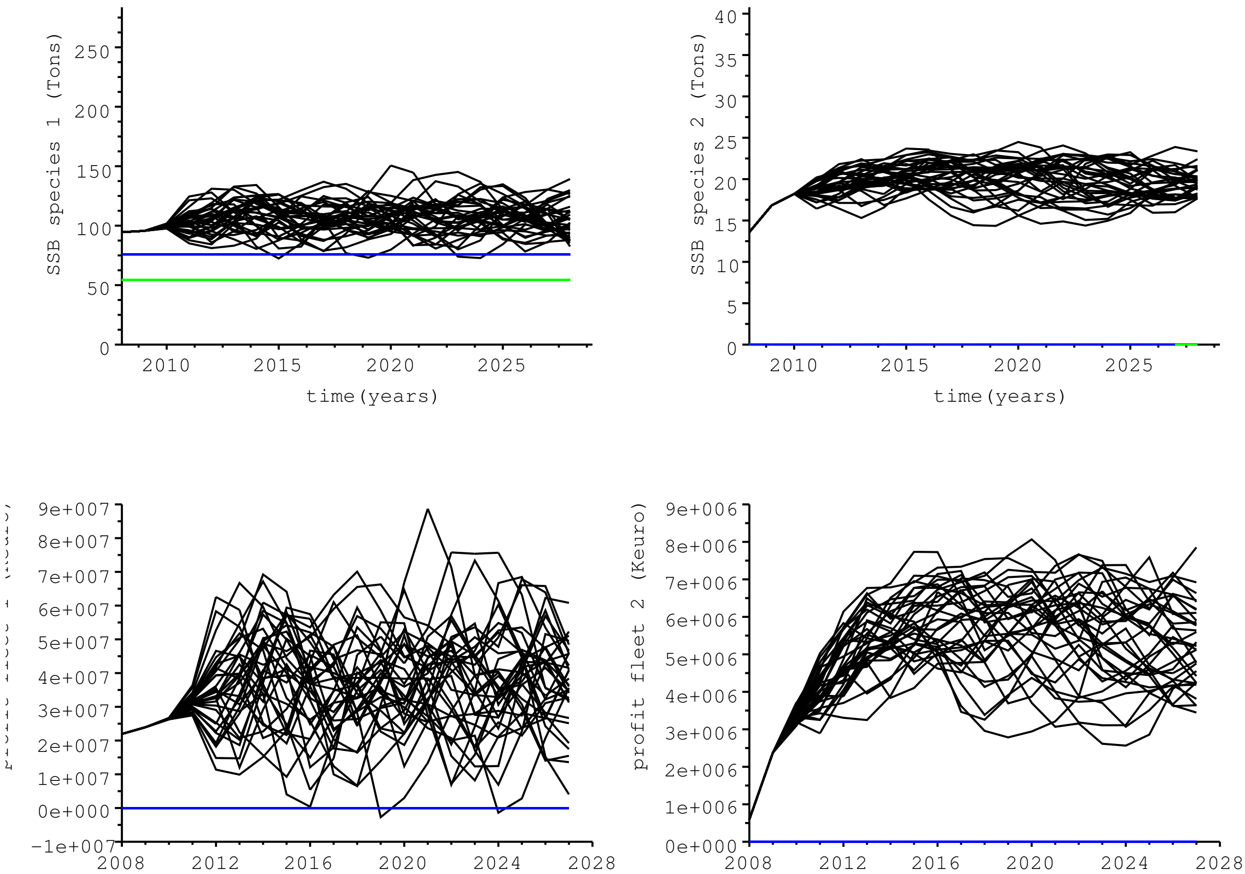


Figure 2.3: Trajectories under the optimizing present value scenario NPV with effort multiplier $u^{\text{NPV}} = (2.5, 0.25)$. Top diagrams: spawning biomass SSB(t) for Hake (left) and *Nephrops* (right). Bottom: profits $\pi(t)$ for the two fleets. See text for details.

2.4.6 Biological Conservation scenario: viable but not sustainable

The third scenario we explore derives from the population viability analysis PVA. Within this framework we consider the specific 'maximum PVA' case corresponding to the 'extremal' conservation strategy $u^{\text{PVA}} = (0, 0)$ which also maximizes the biological viability probability PVA:

$$\text{PVA}(u^{\text{PVA}}) \quad = \max_{u} \text{PVA}(u)$$

This strategy is equivalent to a no-take strategy $u^{\text{PVA}} = (0, 0)$, where no catch is extracted for both fisheries. This strategy obviously satisfies the biological conditions as illustrated by the very high SSB trajectories in figure 2.4 for both Hake and *Nephrops* stocks. This, however, is clearly not a satisfying solution in socio-economic terms as both fleets incomes are nil.

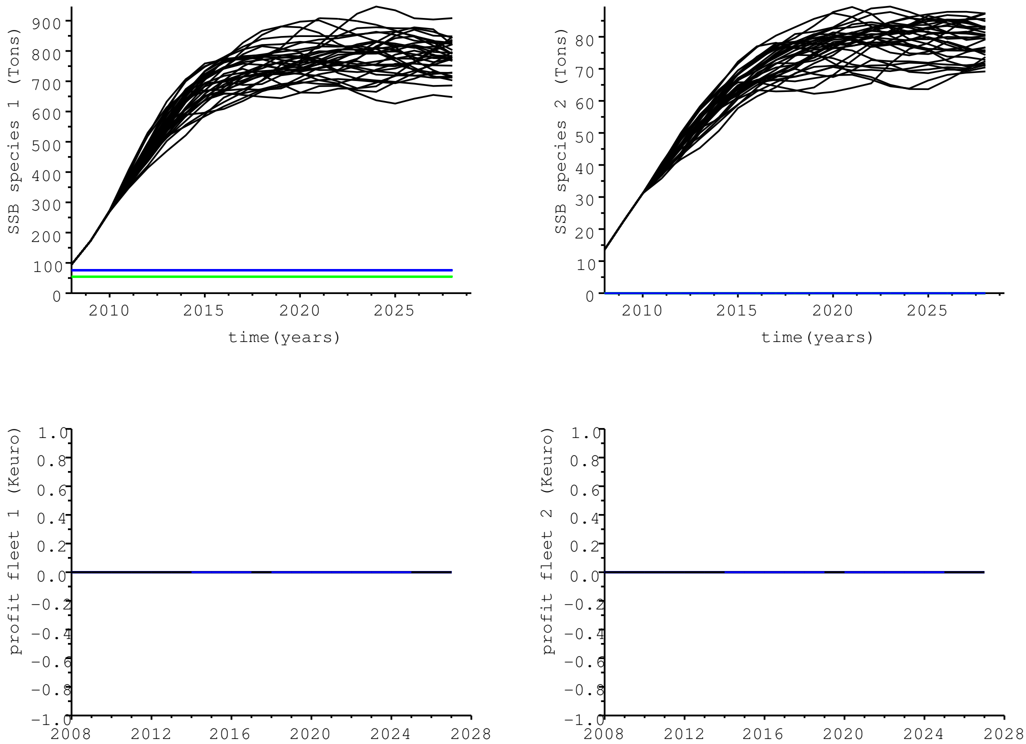


Figure 2.4: Trajectories under the biological (no-take) strategy, i.e. $u^{\text{PVA}} = (0, 0)$. Top diagrams: spawning biomass SSB(t) for Hake (left) and *Nephrops* (right). Bottom: no profit is generated.

2.4.7 Economic scenario: sustainable and almost viable

Symmetrically to the previous biological scenario, we now consider ‘extreme’ viable strategies u^{EVA} characterized by minimal socio-economic risk as follows:

$$\text{EVA}(u^{\text{EVA}}) = \max_{u} \text{EVA}(u)$$

Among the different possible solutions to this optimality problem, we select the largest inertial multiplier $u^{\text{EVA}*}$ which corresponds to the fishing mortality with the smallest difference compared to status quo:

$$|u^{\text{EVA}*} - u^{\text{SQ}}| = \min\left(|u^{\text{EVA}} - u^{\text{SQ}}|,\ \text{EVA}(u^{\text{EVA}}) = \max_{u} \text{EVA}(u)\right)$$

This EVA scenario corresponds to a strategy where we aim at identifying the controls (fishing multipliers) u that maximize the probability of socio-economic viability EVA(u), and then, amongst those solutions, to retain these with the values closest to current level namely $u^{\text{SQ}} = (1, 1)$. The rationale underlying such an option is to minimize the 'costs' of changes and thus account for inertia or rigidity in behaviours. Computation indicates that one solution to achieve this maximum EVA strategy is $u^{\text{EVA}*} \approx (0.92, 0.54)$. Figure 2.5 shows one series of trajectories obtained under this strategy. As expected, the levels of profit generated for both fleets remain positive throughout time. An interesting result is that, despite the potentially detrimental nature of this strategy from a biological conservation point of view, both Hake and *Nephrops* average SSB levels appear to be higher under this particular EVA strategy than they were under the initial baseline scenario SQ. This outcomes emerge because maximizing $\text{EVA}(u)$ necessitates fishing mortality levels which do not impact too severely the stocks, so as to maintain long-term catch rates. The fact that socio-economic viability constraints may imply biological viability has also been stressed by Bene et al. (2001); Martinet et al. (2007) and Martinet et al. (2010).

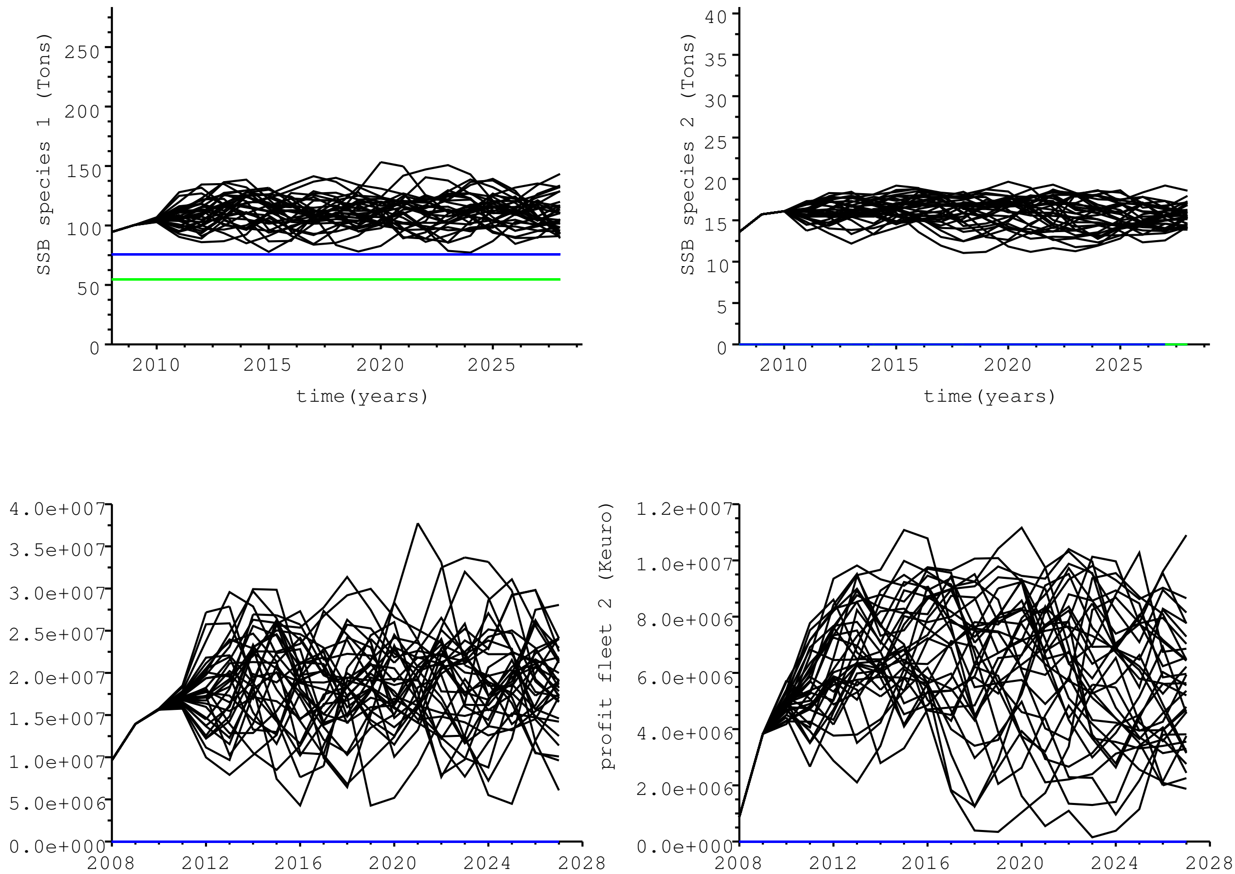


Figure 2.5: Trajectories under the economic scenario EVA with fishing multiplier $u^{\text{EVA}*} = (0.92, 0.54)$. Top diagrams: spawning biomass $\text{SSB}(t)$ for Hake (left) and *Nephrops* (right). Bottom: profits $\pi(t)$ for the two fleets. See text for details.

2.4.8 Co-viability scenario: A win-win situation

Finally, we consider a co-viability strategy u^{CVA}, that is, a strategy that maximizes CVA probability mixing biological and socio-economic constraints:

$$\mathrm{CVA}(u^{\text{CVA}}) = \max_{u} \mathrm{CVA}(u)$$

Again, among the possible fishing mortalities solution of this optimality problem, the multiplier $u^{\text{CVA}*}$ with the smallest difference compared to status quo is choosen as follows:

$$|u^{\text{CVA}*} - u^{\text{SQ}}| = \min\Big(|u^{\text{CVA}} - u^{\text{SQ}}|,\ \mathrm{CVA}(u^{\text{CVA}}) = \max_{u} \mathrm{CVA}(u)\Big).$$

Figure 2.6 displays a series of trajectories obtained under this $u^{\text{CVA}*}$ strategy. In the particular case studied here, the combination of fishing intensity used for the simulations is $u^{\text{CVA}*} \approx (0.9, 0.2)$. Under this strategy, viability probabilities are maximum, that is, $\mathrm{CVA}(u^{\text{CVA}*}) = \mathrm{EVA}(u^{\text{CVA}*}) = \mathrm{PVA}(u^{\text{CVA}*}) = 100\%$. When compared to the $u^{\text{EVA}*}$ scenario above, it is worth emphasizing the very important reduction of fishing intensity requested under $u^{\text{CVA}*}$ for *Nephrops* fleet. The intuition here is that the CVA strategy accounts for the fact that catches made by the *Nephrops* fleet weaken the viability probability of the Hake stock through the impact of the by-catch. In this sense, CVA permits to better balance the stock and fleet requirements, and more generally biological and economic goals, than any other strategy. However the severe reduction in fishing for the *Nephrops* fleet imposed by this co-viability strategy may question its acceptability.

Finally note that although the probabilities $\mathrm{EVA}(u^{\text{CVA}*}) = \mathrm{PVA}(u^{\text{CVA}*}) = 100\%$ resulting from the co-viability strategy are unique, co-viability itself can be obtained through different combinations of fishing controls, as illustrated in figure 2.1(c). This characteristic constitutes a notable difference with optimal control theory where single control solutions are usually identified. In the case of co-viability, a space of solution is identified.

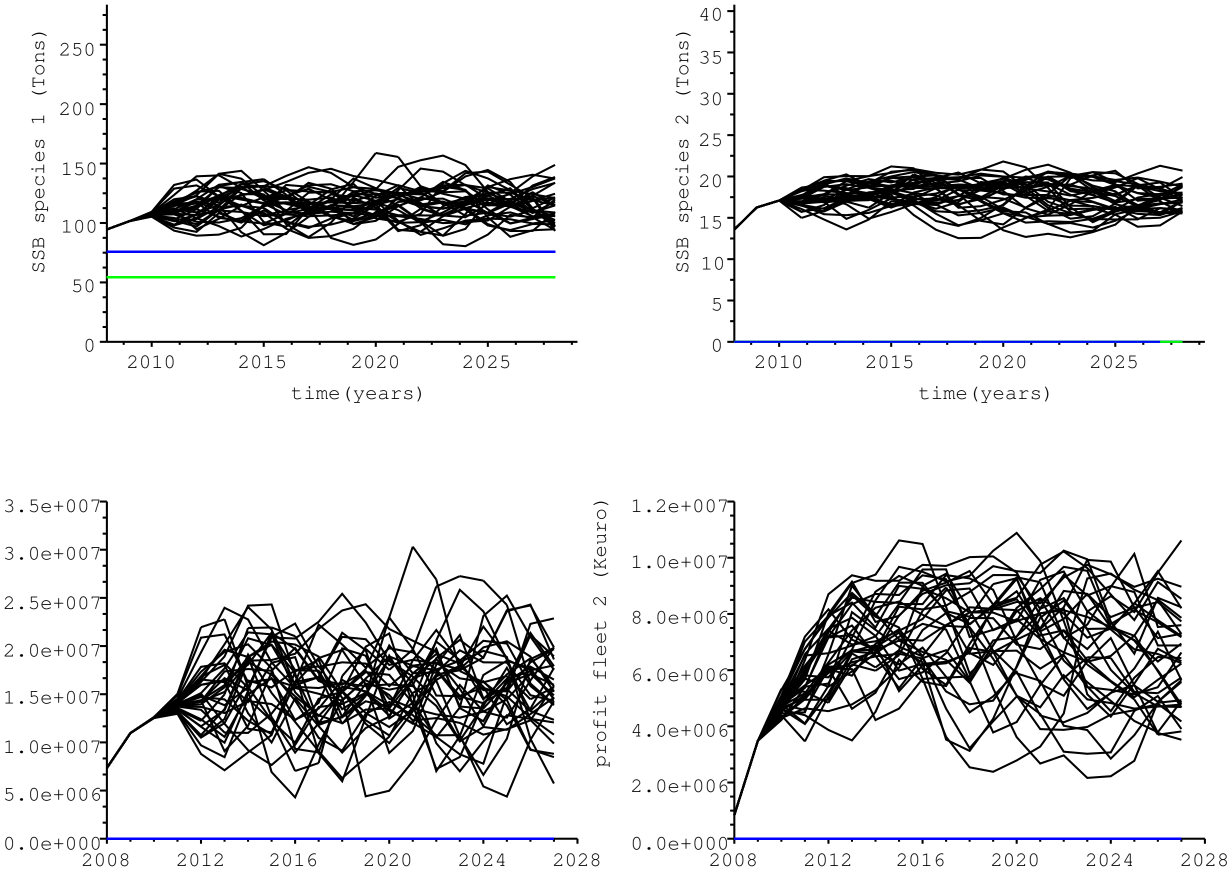


Figure 2.6: Trajectories under the co-viability scenario CVA with fishing multiplier $u^{\text{CVA}*} = (0.9, 0.2)$. Top diagrams: spawning biomass $\text{SSB}(t)$ for Hake (left) and *Nephrops* (right). Bottom: profits $\pi(t)$ for the two fleets. See text for details.

2.5 Discussion and perspectives

This article gives insights on the implementation of the Ecosystem-based Fisheries Management (EBFM) in the case of the Northern Hake and *Nephrops* fisheries of the Bay of Biscay. A bio-economic model is developed, which integrates age-structured dynamics together with uncertainty on recruitment of both species. Interaction between the two fisheries is also accounted for through the by-catch of juvenile Hake by the *Nephrops* fleet. Attempts to integrate such complexity constitute a first step toward the development of EBFM for multi-species fisheries. The viability of various levels of fishing intensity is examined both for the stocks (through a Population Viability Analysis PVA) and the fleets (through a Socio-Economic Viability Analysis EVA). Following the ICES precautionary approach imposed recently on the Northern Hake stock, the PVA adopts a minimum biomass level, while the socio-economic viability is considered through a guaranteed profitability of the different fleets at play. Projections and simulations over 20 years starting from the 2008 baseline year are used to perform viability assessments. The analysis reveals the existence of viable control spaces without bio-economic risk where the probability to maintain the viability of the system is close to 100%.

Five specific fishing strategies are then investigated more thoroughly. The analysis shows that the status quo strategy, consisting of maintaining fishing intensity at the level of the 2008 baseline is not sustainable, as both the biological and socio-economic constraints can be violated for some recruitment scenarios. The no-take strategy aiming at only maximizing the PVA probability appears biologically viable as expected, but is not a viable economic option since no fish is landed and no profit is generated. In contrast, the EVA strategy aiming at maximizing the probability of the fleets' rent does not entail catastrophic biological performances. Not only are the profitability constraints satisfied, but the risk to violate the biological precautionary threshold remains moderate. Finally a co-viability strategy CVA combining both biological and socio-economic objectives is explored. The simulations show that a severe reduction of *Nephrops* fishing mortality is necessary to guarantee a co-viability of the system. The more conventional strategy relying on optimizing the present value of incomes turns out to be risky from the biological viewpoint, as it imposes a major increase of Hake harvest which would alter the Hake stock. It also requires a very important reduction of *Nephrops* fishery which questions its acceptability. Moreover, it does not warrant profits for Hake fishery throughout time. Overall the analysis was therefore useful at contrasting the potential outcomes of different management scenarios both in terms of biological and socio-economic considerations.

A more thorough analysis of these issues would require however to refine the description of the economic structure of the fishery by expanding for instance the number of fleets included in the model. This would allow to analyse the distributional implications of alternative strategies, and the implications of setting minimum profitability constraints for different sub-fleets. A further step could involve accounting for the behavioral response of fleets to changes in their economic performance, adding through this feedback loop another challenging level of complexity. Finally the extension of the model towards more dynamic controls could be another challenging goal.

From a wider perspective, the present research was motivated by the growing interest for

Ecosystem-based Fisheries Management (EBFM). Fisheries scientists and regulating agencies are now encouraging and starting to implement this approach in an increasing number of fisheries. The way to operationalize EBFM remains, however, unclear and challenging. The present paper contributes to this on-going effort. It illustrates how the concepts of stochastic viability and co-viability (CVA) can provide policy-relevant information for the implementation of EBFM. Stochastic CVA is especially equipped to cope with risk, precaution and sustainability in dynamic systems, elements that are central in the EBFM approach (Sanchirico et al., 2008). Stochastic CVA allows to account for the complexities and uncertainties of biological dynamics and interactions that encompass community dynamics (as in the present case) but also trophic webs, or environmental (habitat, climatic) uncertainties as addressed in other works (Doyen et al., 2007). Furthermore, through the use of distinct constraints, CVA provides a multi-criteria framework that accommodates ecological, economic and social objectives for present and future generations. As such it is an integrated and interdisciplinary modelling framework that can be used to explore alternative regulation scenarios and provide policy-relevant informations for the sustainable management of natural resources. From an ecological economics point of view, CVA can deal with a large range of goods and services provided by ecosystems. In this context it has been recently used to address issues related to biodiversity valuation (Bene and Doyen, 2008). The generalization and application of such ideas, concepts and methods to more complex systems is very promising, but remains a challenging task.

Acknowledgments This work was supported by programs of the French agency ANR's (Agence Nationale pour la Recherche) through the research projects Chaloupe (www.projet-chaloupe.fr) and Adhoc. We are grateful to Clotilde Lebreton and Stanislas Prigent for their help in developing some of the Scilab codes of the model used in this research.

2.6 Appendix

Age a	1	2	3	4	5	6	7	8	9
$N_1(2008)$	236062	132608	61571	25195	5219	1606	497	162	45
Maturity $\gamma_{1,a}$	0	0.11	0.73	0.93	0.99	1	1	1	1
Weight $\upsilon_{1,a}$	0.03	0.25	0.72	1.57	2.5	3.45	4.39	5.77	6.75
Natural mortality $M_{1,a}$	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4
Fishing mortality $F_{1,a,1}$	0	0	0.02	0.1	0.17	0.09	0.03	0.01	0.01
Fishing mortality $F_{1,a,2}$	0.09	0.05	0.01	0	0	0	0	0	0
Fishing mortality $F_{1,a,3}$	0.08	0.3	0.47	0.75	0.79	0.85	0.73	0.88	0.88
Discard $d_{1,a,1}$	0	0	0	0	0	0	0	0	0
Discard $d_{1,a,2}$	1	0.37	0	0	0	0	0	0	0
Prices $P_{1,a}$	2	2	2.9	4.1	5.5	6.9	6.9	6.9	6.9

Table 2.1: Hake parameters $s = 1$: *source : ICES; Ifremer, SIH, 2008*

Age a	1	2	3	4	5	6	7	8	9
$N_2(2008)$	642616	650008	328988	180528	65279	23173	8304	4257	4679
Maturity $\gamma_{2,a}$	0	0	0.75	1	1	1	1	1	1
Weight $\upsilon_{2,a}$	0	0.01	0.02	0.03	0.04	0.05	0.06	0.07	0.09
Natural mortality $M_{2,a}$	0.3	0.3	0.25	0.25	0.25	0.25	0.25	0.25	0.25
Fishing mortality $F_{2,a,1}$	0	0	0	0	0	0	0	0	0
Fishing mortality $F_{2,a,2}$	0.01	0.14	0.21	0.21	0.18	0.18	0.19	0.19	0.19
Fishing mortality $F_{2,a,3}$	0.01	0.19	0.28	0.28	0.24	0.25	0.25	0.25	0.25
Discards $d_{2,a,1}$	1	0.97	0.34	0.06	0.02	0.01	0.01	0.02	0.01
Discards $d_{2,a,2}$	1	0.97	0.34	0.06	0.02	0.01	0.01	0.02	0.01
Prices $P_{2,a}$	10.1	10.1	9.1	9.1	9.1	9.1	14.6	14.6	17.6

Table 2.2: *Nephrops* parameters $s = 2$: *source : ICES; Ifremer, SIH, 2008*

Fleets	$K_f(t_0)$	$e_f(t_0)$	c_f^{var}	c_f^{fix}	α_f
Nephrops trawlers $f = 1$	87	180	379	257604	1.48
Gill-netters $f = 2$	116	180	481	211432	1.52

Table 2.3: Economic parameters for fleets $f = 1, 2$: Initial number of vessels $K_f(t_0)$, effort by vessel $e_f(t_0)$ (day at sea), variable c_f^{var} (€ by vessel by day), fixed costs c_f^{fix} (€ by vessel) and multiplier of extra fishing income α_f . *source : Ifremer, SIH, DPMA, 2008*

Chapter 3. Managing mixed fisheries for bio-economic viability

Published in *Fisheries Research*:

Gourguet S., Macher C., Doyen L., Thébaud O., Bertignac M., Guyader O., (2013). Managing mixed fisheries for bio-economic viability, *Fisheries Research*, 140, 46-62.

For the sake of consistency between chapters, the initial published terms of 'ecological viability' and 'economic viability' have been respectively changed in 'biological viability' and 'socio-economic viability'; and the following variables have also been changed as described:

Variable name	Fisheries Research	Thesis
Catch	C	Y
Discount rate	r	δ
Biological viability probability	$\mathbb{P}_{\text{PVA}}$	PVA
Socio-economic viability probability	$\mathbb{P}_{\text{EVA}}$	EVA
Co-viability probability	$\mathbb{P}_{\text{CVA}}$	CVA

To note: in chapter 2, species $s = 1$ corresponded to Hake and species $s = 2$ to *Nephrops*, however in this chapter it is the opposite: $s = 1$ stands for the *Nephrops* and $s = 2$ for the Hake.

This chapter extends the bio-economic model presented in chapter 2, by the integration of another species, the Sole (*Solea solea*) and the separation of the French fleet vessels in sixteen groups of vessels (sub-fleet) based on their main gear use, structure of landings, costs and length-class. A co-viability analysis examines the trade-offs between preserving Spawning Stock Biomass (SSB) of every species and maintaining the economic profitability of the various fishing sub-fleets. The simulations allow also comparisons of the trade-offs associated with different allocations of fishing effort across sub-fleets, under various economic scenarios. Optimal management strategies are provided thanks to a genetic algorithm adapted for the co-viability analysis with optimisation. The genetic algorithm is presented in appendix of the thesis.

Abstract

Management of fisheries for sustainability requires dealing with multiple and often conflicting objectives. A stochastic viability approach is proposed to address the trade-offs associated with balancing ecological, economic and social objectives in regulating mixed fisheries, taking into account the complexity and uncertainty of the dynamic interactions which characterize such fisheries. We focus on the demersal fishery in the Bay of Biscay and more specifically on the fleets harvesting Norway Lobster (*Nephrops norvegicus*), Hake (*Merluccius merluccius*) and Sole (*Solea solea*). A bio-economic multi-species and multi-fleet model with technical interactions is developed to examine the trade-offs between preserving Spawning Stock Biomass (SSB) of every species and maintaining the economic profitability of the various fishing fleets. Different management strategies are tested and compared. Results suggest that ensuring viability of this demersal fishery requires a significant decrease in fishing capacity as compared to the reference year. The simulations allow comparing the trade-offs associated with different allocations of this decrease across fleets.

Keywords: Bay of Biscay, bio-economic model, co-viability, fisheries, uncertainty.

3.1 Introduction

Marine biodiversity is under extreme pressure worldwide as a result of overexploitation, pollution and habitat loss (Ye et al., 2012). Overcapacity and overfished populations, as well as the indirect effects of fisheries on marine ecosystems, reflect the difficulties faced by management in achieving the principal goal of sustainability. One of the reasons put forward to explain the limited success of fisheries management is their frequent focus on single targeted species, rather than on the entire set of species affected by fishing. Because such approaches ignore multi-species and multi-fleet interactions, their effectiveness is limited where such interactions are an important driver of fishing mortality and of economic profitability, particularly in mixed fisheries. Moreover, understanding the trade-offs between ecological, economic and social objectives is important in designing policies to manage ecosystems and fisheries (Cheung and Sumaila, 2008). As stressed by Pikitch et al. (2004) and Kempf (2010), there is nowadays widespread acceptance that a more integrated perspective is needed, if these multiple objectives are to be successfully addressed in designing fisheries management regulations for sustainable use of marine living resources. However, the way to operationalize integrated approaches to the management of mixed fisheries remains controversial as pointed out in Sanchirico et al. (2008) and Doyen et al. (2012). Single-species targets and reference points may still be appropriate, but need to be adapted (Pikitch et al., 2004; Hall and Mainprize, 2004). Bio-economic models have been proposed as a means to explore these issues, taking into account socio-economic dimensions, and the complexity of feedback effects between anthropogenic activities and natural resources (Prellezo et al., 2012). Growing efforts have been made to develop modelling approaches allowing to assess alternative management options for complex fishery systems. Plagányi (2007) provides an overview of the relative merits and limitations of the different modelling approaches in this domain. Management Strategy Evaluation has been widely recognised as a relevant framework to test the robustness of alternative management procedures to the uncertainties that characterize fishery systems (Punt and Smith, 1999; Kell et al., 2007). Using this framework, several applications have been developed for the northern Hake fishery (Murua et al., 2010; Garcia et al., 2011). Viability modelling is also proposed by several authors (Bene et al., 2001; Cury et al., 2005; Eisenack et al., 2006; Bene and Doyen, 2008; Baumgärtner and Quaas, 2009; Doyen et al., 2012) as a relevant bio-economic modelling framework. Viability theory - introduced mathematically by Aubin (1990) - aims at identifying decision rules such that a set of constraints, representing various objectives, is respected at any time. It can be useful in multi-criteria contexts as this approach exhibits a domain of possibilities, feasibility and trade-offs between potentially conflicting objectives or constraints (Baumgärtner and Quaas, 2009). The approach is also closely related to the maximin, or Rawlsian, approach with respect to intergenerational equity (Martinet and Doyen, 2007; Doyen and Martinet, 2012) as constraints can be assumed to apply throughout both present and future time periods. Furthermore, stochastic viability (Doyen and De Lara, 2010) can handle issues of bio-economic vulnerability, risks, safety and precaution by compiling ecological and economic goals in a random context and expanding the Population Viability Analysis (PVA) used in conservation biology to address extinction risks for populations. The viability approach has been applied to the bio-economic management of renewable resource systems, especially fisheries in Bene et al. (2001); Eisenack et al. (2006), only a few of which are dedicated to case studies with real data (Mullon et al., 2004; Martinet et al., 2007; De Lara et al., 2007; Chapel et al., 2008; Doyen et al., 2012).

The objective of the present paper is to use the framework of viability analysis to address the formal modelling of trade-offs between conflicting objectives in the management of a mixed fishery. We apply the framework to the Bay of Biscay demersal mixed fishery. A discrete-time stochastic bio-economic model is developed and calibrated based on the data available on the fishery. The model is used to explore alternative management strategies, with particular emphasis on those which allow joint biological-socio-economic viability of three main exploited fish species - Norway Lobster (*Nephrops norvegicus*), Hake (*Merluccius merluccius*) and Sole (*Solea solea*) – and of the sixteen sub-fleets (trawlers and gill-netters) harvesting these species.

3.2 Material and methods

3.2.1 The Bay of Biscay case study

The Bay of Biscay demersal mixed fishery operates in divisions VIIIa and b of the ICES grid (Figure 3.1). French, Spanish and Belgian fishery fleets operate in this area. The main gears used in these fisheries are trawl, gill-net and longline, and all induce variable levels of impacts on a wide range of species. Under the European data collection framework for fisheries, a number of fishery-independent surveys, data collection programs, stock assessments based on virtual population analysis (vpa) models (ICES, 2009) and research projects have been carried out over the years (ICES, 2009, http://www.ices.dk/datacentre/StdGraphDB.asp, http://www.umr-amure.fr/pg_partenarial_bioeco.php), which provide both biological and economic information that can be used to calibrate a bio-economic model of the French component of the fishery[1]. According to Daurès et al. (2008) among the 200 species caught in the Bay of Biscay, 20 species correspond to 80% in volume of the landings in 2007. Three of the most important species in percentage of the total French national landing value include *Nephrops* (6%), Hake (7%) and Sole (11%). The model we develop aims to represent the dynamics of these three species.

The French fleets which target these species can be separated in four main groups of vessels based on their main gear used and structure of landings: *Nephrops* trawlers, various fish trawlers,

[1]Only French economic data were available for this study.

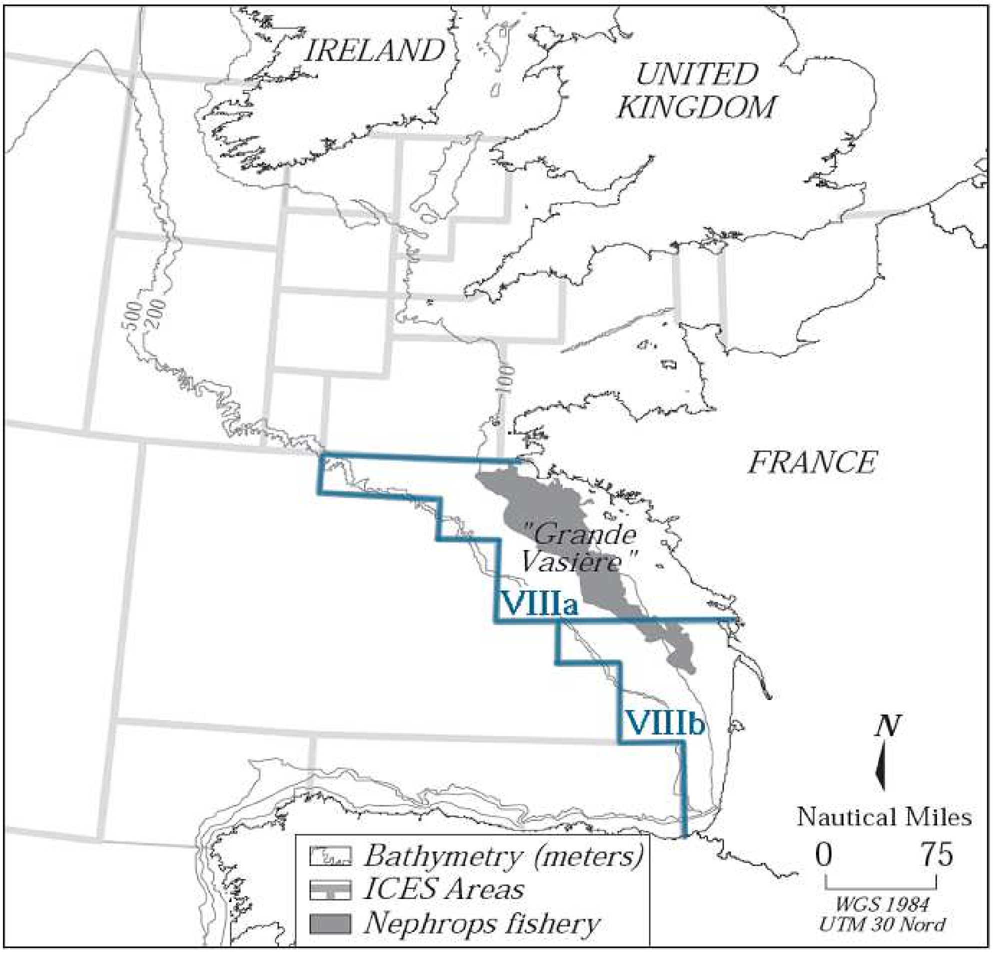


Figure 3.1: ICES Divisions VIIIa,b Source: Macher et al. (2008)

Sole gill-netters and various fish gill-netters (Macher et al., 2011). These four fleets involved 577 vessels in 2008 and their total turnover amounted to 206 million €. The four fleets can be further sub-divided into sixteen sub-fleets according to the length-class of vessels and their associated cost structure. A 17th fleet is also considered in the model to account for the fishing mortalities caused by vessels that do not belong to the sixteen sub-fleets, particularly Spanish and Belgian vessels.

Figures 3.2 and 3.3 capture the major interactions in the Bay of Biscay mixed demersal fishery which are taken into account in our analysis. Technical interactions between fleets are illustrated in the figure 3.2, which shows the estimated number of individual fish caught by each fleet during the year 2008. These numbers include landed but also discarded individuals. Bottom trawls, in particular, are poorly selective gears and their use induces catches of non-targeted fishes (by-catch and by-product) or unwanted length grades of the targeted species. Most of these catches are usually discarded (ICES, 2009). In the Bay of Biscay, *Nephrops* and Hake are the most discarded species in weight and numbers. While the *Nephrops* trawlers target mainly *Nephrops*, they also have an important impact on juvenile Hake (as shown by figure 3.2) due to the fact that the *Nephrops* fishing grounds are located on a Hake nursery area. Discarding leads to negative impacts on stock renewal as discards have a high mortality rate (Guéguen and Charuau, 1975; Alverson et al., 1994). However, neither the *Nephrops* trawlers nor the various fish trawlers depend on Hake for their revenue, as shown in figure 3.3 which illustrates the contribution of each species to the gross income of each fleet. This is characteristic of a technical interaction in which the fishing mode of *Nephrops* trawlers has an unsought joint impact on Hake resources, which affects mainly other fleets for which this species is an important source of revenue. Many other similar interactions exist between sub-fleets, as illustrated by the figure 3.3.

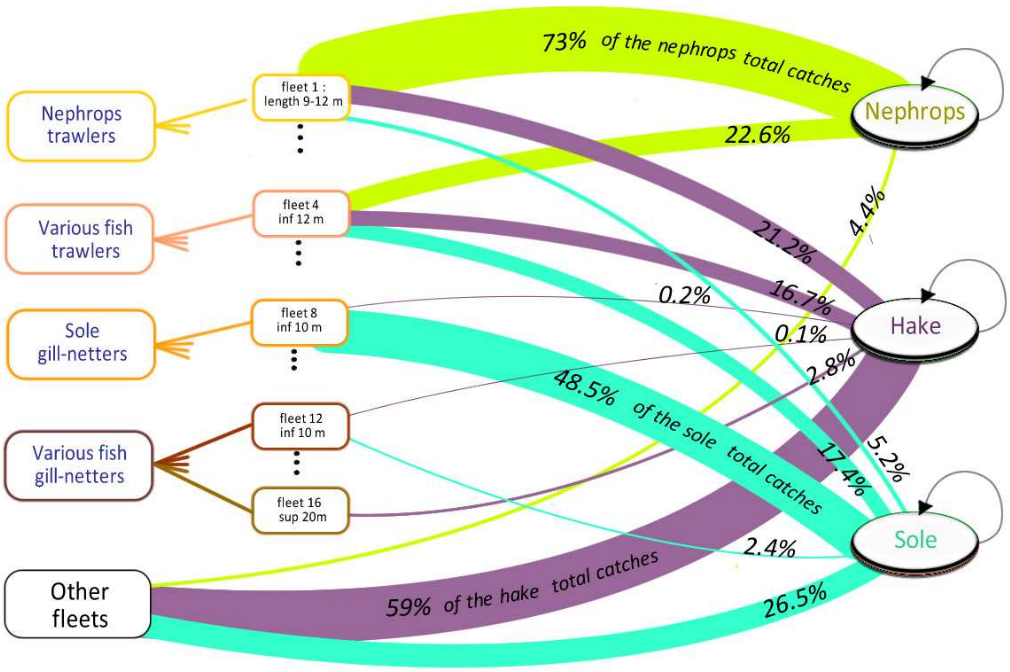


Figure 3.2: Stylized representation of the Bay of Biscay mixed demersal fishery used as a basis to develop the bio-economic model. The width of the arrows is proportional to the percentage of total number of individual fish caught by the fleets in 2008, including both landings and discards.

The management of these fisheries mainly relies on conservation measures: a Total Allowable Catch (TAC) revised each year, a minimum landing size (MLS) and a minimum trawl mesh size. *Nephrops* are targeted by bottom trawlers on a sand-muddy area called 'La Grande Vasière' (Figure 3.1). A major part of the *Nephrops* landings (in weight) from VIIIa,b are taken by French trawlers. The figure 3.2 shows that a large amount of *Nephrops* are caught by the 'other fleets' fleet, however

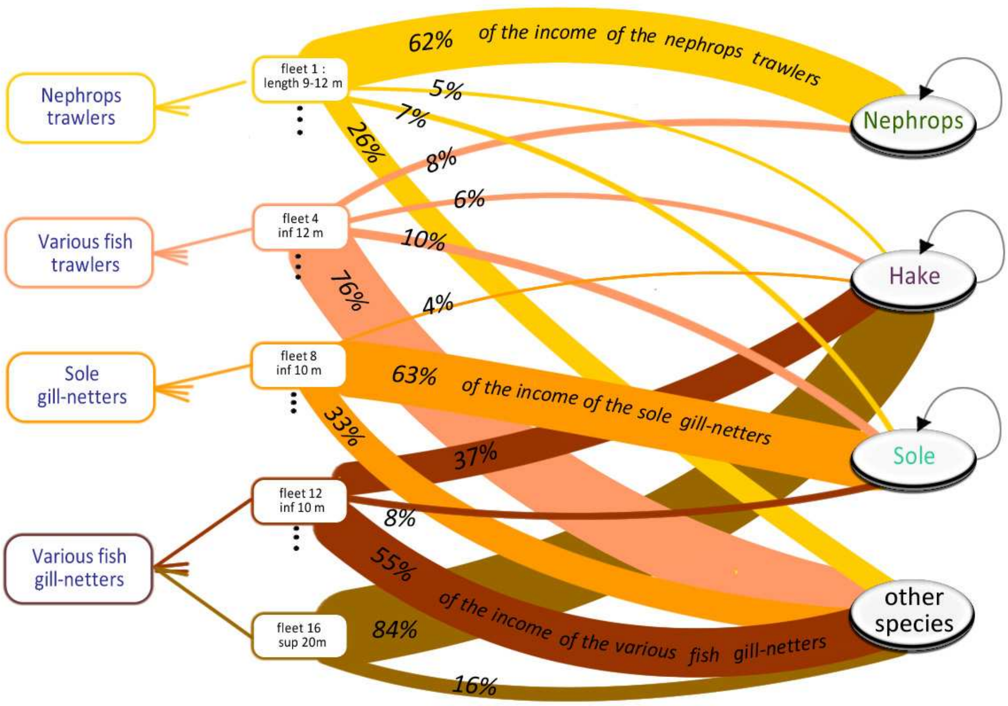


Figure 3.3: Stylized representation of the contribution of each species to the gross income of each fleet in 2008. The width of the arrows is proportional to the contribution (in percentage) of each species to the total gross income of each fleet.

most of these individuals are not landed. They correspond for most of them to individuals below the minimum landing size. The *Nephrops* trawler fleet is one of the most important segments of the French fleet in the Bay of Biscay (ICES, 2010). The fleet indeed represents about one quarter of the French trawlers in this area. With regards to access regulations, a limited entry license system has been enforced since 2005. However, it does not include individual limitations of effort or catches. The high level of catches and discards of younger age groups below the MLS contributes to economic inefficiency of the exploitation (Macher and Boncoeur, 2010). The stock was assessed in 2008 and ICES concluded that its SSB was relatively stable and advised to maintain current landings. The agreed TAC for 2008 was 4320 tonnes. The important hake by-catch previously mentioned mainly affects the fleets that depend strongly on Hake, which include French gill-netters (accounting for around 30% of total landings) as well as Spanish fleets (accounting for around 53% of total landings in 2008) (Macher et al., 2011). The Hake TAC in 2008 was set at 54,000 tonnes, including 20,196 tonnes for the Bay of Biscay area. The Bay of Biscay Sole fishery has two main components: a French gill-net fishery directed at Sole and a French and Belgian trawl fishery. The French fleet is the most important participant in the fishery with landings being close to 90% of the total international landings, over the available historical series since 1979. The Sole landings in the Bay of Biscay are subject to a TAC regulation in combination with technical measures. The 2008 TAC was set at 4582 tonnes.

3.2.2 The bio-economic model

The bio-economic model we develop captures the main features of the technical interactions which exist between the various components of the Bay of Biscay demersal fishery as described in the previous section. Due to data availability constraints, the focus of the analysis is on the bio-economic outcomes of alternative management strategies for the French fleets. However, the model also captures the influence of foreign fleets on the dynamics of the system, and on the potential outcomes of alternative management strategies targeted at the French fleets. The model relies on the mathematics of controlled dynamic systems (Clark, 1990) and more specifically of discrete time systems (De Lara and Doyen, 2008). It extends the bio-economic model presented in Doyen et al. (2012) in which only Hake and *Nephrops* were taken into account, and only two aggregated fishing fleets were explicitly represented. Based on the existing understanding of trophic structures in the Bay of Biscay demersal ecosystem Le Loc'h and Hily (2005), no trophic interactions are assumed between these species.

A multi-species, multi-fleets, age-structured dynamic framework

Fish population dynamics are modelled using an age-structured population model derived from the standard fish stock assessment approach (Quinn and Deriso, 1999). Population dynamics are described on a yearly basis and integrate uncertainties regarding recruitment. The age-structured dynamics of the three species are governed by :

$$\begin{cases} \mathrm{N}_{s,a}(t+1) = & \mathrm{N}_{s,a-1}(t)\exp\left(-M_{s,a-1} - F_{s,a-1}\right), \qquad \mathrm{a} = 2,\dots,A_s - 1 \\ \mathrm{N}_{s,A_s}(t+1) = & \mathrm{N}_{s,A_s-1}(t)\exp\left(-M_{s,A_s-1} - F_{s,A_s-1}\right) \\ & +\mathrm{N}_{s,A_s}(t)\exp\left(-M_{s,A_s} - F_{s,A_s}\right). \end{cases} \tag{3.1}$$

where $\mathrm{N}_{s,a}(t)$ stands for the abundance of the exploited species $s = 1, 2, 3$ (*Nephrops*, Hake and Sole, respectively) at age $a = 1, \dots, A_s$. Thus the state $\mathrm{N}_{s,a}(t+1)$ of the stock at time $t+1$ evolves according to both natural $M_{s,a}$ and total fishing $F_{s,a}$ mortality rates of the species s at age a. Total fishing mortality of species s at age a $F_{s,a}$ is derived from the sum of fishing mortality from all

17 sub-fleets in year t_0, $F_{s,a,f}(t_0)$ and from the fishing effort multipliers u as described in equation (3.2):

$$F_{s,a} = \sum_{f=1}^{17} u_f F_{s,a,f}(t_0). \tag{3.2}$$

with u_f that stands for the fishing effort multiplier of the sub-fleet f. Effort multipliers are defined as the ratio of sub-fleet fishing effort as compared to fishing effort in a reference year, and are introduced as control variables to define management strategies. In this study, effort multipliers are applied to the number of vessels per sub-fleet. The reference year is set at $t_0 = 2008$. Fishing mortalities depend both on the fishing effort by vessel (number of days at sea) and the number of vessel by sub-fleet (see the equation (3.7)). The biological parameters are described in the appendix in tables 3.3, 3.4 and 3.5 for *Nephrops*, Hake and Sole respectively. The estimated values of $F_{s,a,f}(t_0)$, the fishing mortality of the species s at age a induced by the sub-fleet f in 2008, include both landed and discarded fish, and are detailed in the appendix tables 3.6, 3.7 and 3.8 for *Nephrops*, Hake and Sole. The parameter values are derived from the ICES databases[2], reports of the Working Group on the Assessment of Southern Shelf Stocks of Hake, Monk and Megrim (WGHMM) (ICES, 2009) and the Ifremer, SIH, DPMA databases [3].

Introducing stochastic recruitment functions

Recruitment involves complex biological and environmental processes that vary over time. The recruits $\mathrm{N}_{s,1}(t+1)$ for each species are therefore assumed to be uncertain functions of the Spawning Stock Biomass at time t:

$$\mathrm{N}_{s,1}(t+1) = \varphi_s\Big(\mathrm{SSB}_s(t), \omega_s(t)\Big). \tag{3.3}$$

The Spawning Stock Biomass $\mathrm{SSB}_s(t)$ of the species s is given by:

$$\mathrm{SSB}_s(t) = \sum_{a=1}^{A_s} \gamma_{s,a} \upsilon_{s,a} \mathrm{N}_{s,a}(t), \tag{3.4}$$

with $(\gamma_{s,a})_{a=1,\dots,A_s}$ the proportions of mature individuals of species s at age a and $(\upsilon_{s,a})_{a=1,\dots,A_s}$ the weights of individuals of species s at age a. The function φ_s represents the specific stock-recruitment

[2] http://www.ices.dk/datacentre/StdGraphDB.asp

[3] DPMA stands for 'Direction des Pêches Maritimes et de l'Aquaculture' which corresponds to the Directorate for Sea Fisheries and Aquaculture at the French Ministry of Agriculture and Fisheries. SIH stands for 'Système d'Information Halieutique', the fisheries information system monitored by Ifremer, the French Research Institute for the Exploitation of the Sea (http://wwz.ifremer.fr/institut_eng).

relationship of each species s while $\omega_s(t)$ stands for uncertainties affecting the stock recruitment relationships through different possible scenarios Ω. In the present case-study, following STECF (2008) and the approach adopted by the working group WGHMM, the recruitment relationship of the species is set using an Ockham-Razor function as in O'Brien et al. (2002):

$$\varphi_s(\mathrm{SSB}_s, \omega_s) = \begin{cases} \omega_s \rightsquigarrow \mathcal{U}_s & \text{if } \mathrm{SSB}_s \geq B_s^{\mathrm{lim}}, \\ \mathrm{SSB}_s \dfrac{\overline{R}_s}{B_s^{\mathrm{lim}}} & \text{if } \mathrm{SSB}_s \leq B_s^{\mathrm{lim}}. \end{cases} \tag{3.5}$$

Here $\mathcal{U}_s$ stands for the uniform distribution relying on R_s^t, the historical time series of recruitment of species s[4] (ICES, 2009) . ICES limit reference biomass B_s^{lim} and the mean historical recruitment $\overline{R}_s$ values are specified in table 3.9 of the appendix. The three species have different biology and life cycles, therefore we assume that their recruitments are uncorrelated.

Catches and fishing mortality

For each period t, the exploitation of the three species is described by the catches $Y_{s,a,f}(t)$. These catches depend on initial fishing mortalities $F_{s,a,f}(t_0)$, effort multipliers u_f and abundances $\mathrm{N}_{s,a}(t)$ through the Baranov catch equation:

$$Y_{s,a,f}(t) = \mathrm{N}_{s,a}(t) u_f F_{s,a,f}(t_0) \frac{1 - \exp\left(-M_{s,a} - \displaystyle\sum_{f=1}^{N_f} u_f F_{s,a,f}(t_0)\right)}{M_{s,a} + \displaystyle\sum_{f=1}^{N_f} u_f F_{s,a,f}(t_0)}. \tag{3.6}$$

The initial fishing mortality $F_{s,a,f}(t_0)$ can be expressed as:

$$F_{s,a,f}(t_0) = q_{s,a,f} e_f(t_0) \mathrm{K}_f(t_0), \tag{3.7}$$

where $e_f(t_0)$ is the mean value of fishing effort by vessels of sub-fleet f expressed in number of days at sea and $\mathrm{K}_f(t_0)$ is the number of vessels by sub-fleet f, both for the baseline year 2008. Their values are given in table 3.13 in the appendix. The catchability $q_{s,a,f}$ corresponds to the fishing mortality of species s at age a associated with one unit of fishing effort from a vessel of sub-fleet f. Catchabilities are assumed constant over the simulation period.

[4] R_s^i is the sample i for the species s and $\mathbb{P}(\varphi_s = R_s^i) = \frac{1}{I}$ with I the number of possible values. Gaussian and continuous uniform distributions were also tested but did not significantly modify the results. Furthermore this 'historical data series' method is used by the scientists of the working group WGHMM of ICES.

Income

The gross income from catches of each sub-fleet $\mathrm{Inc}_f(t)$ is then estimated by introducing the market price of the species along with the estimates of discard rates, such that:

$$\mathrm{Inc}_f(t) = \sum_s \sum_{a=1}^{A_s} p_{s,a}(\widetilde{\omega}_s(t)) \upsilon_{s,a,f} Y_{s,a,f}(t)(1 - d_{s,a,f}). \tag{3.8}$$

where $\upsilon_{s,a}$ is the mean weight of landed individuals of species s at age a and $d_{s,a,f}$ represents the discard rate of individuals of age a by the sub-fleet f. Discard ratios were calibrated on the data available from the ICES working group WGHMM.[5] Price $p_{s,a}(\widetilde{\omega}_s(t))$ corresponds to the market value (euros by kg) of species s at age a for year t under the stochastic scenario $\widetilde{\omega}_s(t)$. Uncertainties on annual mean market price by species are introduced through a random mean price by species following a Gaussian law as:

$$p_s(\widetilde{\omega}_s) \rightsquigarrow \mathcal{N}(\mu_s^P, \sigma_s^P). \tag{3.9}$$

Gaussian laws are calibrated from ex-vessel prices for the three species for the 2000-2009 period, recorded in French harbours (data from Ifremer, SIH, DPMA). Prices by species $p_s(\widetilde{\omega}_s(t))$ are assumed to be independent by species and by year. Market price $p_{s,a}(\widetilde{\omega}_s(t))$ by age a are computed from the annual price by species $p_s(\widetilde{\omega}_s(t))$ as follows:

$$p_{s,a}(\widetilde{\omega}_s(t)) = p_s(\widetilde{\omega}_s(t))\Upsilon_{s,a}. \tag{3.10}$$

with $\Upsilon_{s,a}$, an age price coefficient calibrated from 2008 market prices for the three species and for different market categories (defined in terms of the size/age of fish) (Ifremer,SIH,DPMA). Parameters of the Gaussian law (μ_s^P, σ_s^P) for each species and age price coefficient $\Upsilon_{s,a}$ by species s and age a are displayed in the appendix in tables 3.3, 3.4 and 3.5.

Profits

The socio-economic viability per sub-fleet is assumed to be determined by their economic profitability, that is the difference between their gross income and their costs. The profit π_f of a sub-fleet

[5] A difference in discards between the *Nephrops* trawlers and the various fish trawlers was observed. The *Nephrops* trawlers appear to have a larger impacts on the first age class of Hake than the other trawlers. As discarding rates for Hake and Sole are not known per sub-fleet, we assume that discarding rates are the same for each sub-fleet, equal to the discarding rate of the whole fleet and assumed to be constant over the simulation period. For the same reason, fishing mortality is allocated between the sub-fleets according to their contribution to total landings (see in the appendix: tables 3.6, 3.7 and 3.8 for the values of fishing mortalities by sub-fleet for *Nephrops*, Hake and Sole and the tables 3.10, 3.11 and 3.12 for the estimated discards).

f is estimated as follows:

$$\pi_f(t) = \Big(\mathrm{Inc}_f(t) + \alpha_f u_f \mathrm{K}_f(t_0) e_f(t_0)\Big)(1-\tau_f) - \Big(\mathrm{V}_f^{fuel} p_{fuel}(t) e_f(t_0) + c_f^{var} e_f(t_0) + c_f^{fix}\Big) u_f \mathrm{K}_f(t_0). \quad (3.11)$$

Here the parameter α_f corresponds to the income per unit of effort of sub-fleet f derived from catches of species not explicitly modelled. Incomes from other species are including in an additive fashion as in Raveau et al. (2012). We assume that biomass and price of other species are constant, and that the impacts of modelled fleets on these species are relatively negligible. Rate τ_f is the landing cost by sub-fleet as a proportion of the gross income[6]. V_f^{fuel} corresponds to the volume of fuel (in litres) used by fishing effort unit (i.e. days at sea) for one vessel of sub-fleet f and $p_{fuel}(t)$ is the fuel price by litre of the year t that can be subjected to projection scenarios. The other variable cost c_f^{var} of a fishing effort unit by a vessel of sub-fleet f includes oil, supplies, ice, bait, gear and equipment costs while c_f^{fix} corresponds to the annual costs associated with vessel of the sub-fleet f, including maintenance, repair, management and crew costs, fishing firms, licenses, insurance premiums and producer organisation charges. Cost parameter values in the model are based on the economic data available for 2008 (Ifremer, SIH, DPMA) and are assumed to be constant over the simulation period. As mentioned in section 3.2.2, u_f corresponds to the fishing effort multiplier of sub-fleet f and is applied to the number of vessels per sub-fleet [7]. The full set of previous parameters used to estimate profits is displayed in the tables 3.13 and 3.14 in the appendix.

3.2.3 Fuel scenarios

As regards fuel price, two different scenarios are considered: a base case scenario BC where fuel price $p_{fuel}(t)$ is assumed to be steady over the simulation period and a most likely scenario ML where fuel price increases over time. Fuel prices under the most likely scenario are based on projections from the International Energy Agency (IEA, 2010; CAS, 2012). Table 3.1 summarizes both fuel scenarios.

[6] τ_f could potentially and theoretically be used as policy instrument through a landing tax. However, τ_f is here considered as a landing cost, a levy ad valorem which is paid for landing services. The differences between these landing costs by sub-fleet, displayed in table 3.14 , relate to the different locations of landings, and associated costs of landing services.

[7] It is most likely that in circumstances where a decrease in capacity was required to maintain long-term viability of the fishery, the fleets would first try to modulate their fishing effort (and associated variable costs) before vessels left the fishery. Therefore, we first tried to use fishing effort of the sub-fleets as a control. However, it turns out that economic issues in the fishery arise mainly from the annual, rather than variable, cost component of profits.

Table 3.1: Fuel price scenarios (in each row) considered in this study. Source:(IEA, 2010; CAS, 2012).

Scenarios	Description
BC	Base case scenario: constant fuel prices
	$p_{fuel}(t) = p_{fuel,ref} = 0.50$ €/L
ML	Most likely scenario: increase of fuel price
	$p_{fuel}(t) = p_{fuel,ref}$ for t=1
	$p_{fuel}(t) = p_{fuel}(t-1) + 0.03$ for t=2,. . . , 7
	$p_{fuel}(t) = p_{fuel}(t-1) + 0.0115$ for t=8,. . . , 12
	$p_{fuel}(t) = p_{fuel}(t-1) + 0.0135$ for t=13,. . . , 20

3.2.4 The co-viability diagnostic

The viability framework of analysis is used to describe trade-offs associated with alternative management approaches for the fishery. This requires the specification of constraints which capture the different objectives that may be pursued in managing the fishery. Given the stochastic nature of the model (i.e. uncertainties on recruitments and market prices), the performance of management strategies must be assessed in terms of the probability for these constraints to be met by the fishery at any point in time, under alternative scenarios (Doyen and De Lara, 2010). We consider two sets of constraints to define the viability of the fishery: the first set of constraints relates to the population viability of the three species; the second set of constraints relates to the socio-economic viability of each individual sub-fleet. Our set of biological constraints is related to Population Viability Analysis (PVA), well-known in biological conservation sciences (Morris and Doak, 2002). Population Viability Analysis (PVA) is a process of identifying the threats faced by a species and evaluating the likelihood that it will persist into the future. PVA is defined as the requirement that the Spawning Stock Biomass of each individual species is maintained above a threshold value. In this study, the thresholds correspond to B_s^{pa}, the biomass of precaution of the species s estimated by the International Council for the Exploration of the Sea. The constraint is specified as:

$$\mathrm{SSB}_s(t) \geq B_s^{\mathsf{pa}}, \qquad s = 1, 2, 3. \tag{3.12}$$

The biological performance of a management strategy, involving a particular vector of effort multipliers by sub-fleet u, can be assessed by the population viability probability PVA(u), as described by :

$$\mathrm{PVA}(u) = \mathbb{P}\Big(\text{constraints (3.12) are satisfied for } t = t_0, \ldots, T\Big). \tag{3.13}$$

We also consider the socio-economic objective of maintaining positive profits for the sub-fleets over time (socio-Economic Viability Analysis, EVA):

$$\pi_f(t) > 0, \qquad f = 1, \ldots, 16. \tag{3.14}$$

The socio-economic viability probability of the fishery related to a vector of effort multipliers EVA(u) is thus expressed by:

$$\mathrm{EVA}(u) = \mathbb{P}\Big(\text{constraints (3.14) are satisfied for } t = t_0, \ldots, T\Big). \tag{3.15}$$

In effect, this constraint aims to keep each segment of the fishery active (at positive profit levels), and the related social benefits of maintaining employment in each sub-fleet. Given that the sub-fleets are distributed across different coastal regions of the Bay of Biscay, this also ensures the maintenance of active commercial fishing operations and employment all along the coastline from which the fishery operates. The objective thus defined is in fact akin to a social constraint, as it essentially requires that levels of economic profitability achieved by sub-fleets allow these sub-fleets to continue participating in the fishery. In this sense, it is similar to the participation constraint defined by Péreau et al. (2012) in their bio-economic analysis of the effects of ITQ regulations on fisheries.

Co-Viability Analysis (CVA) of the fishery combines PVA and EVA and seeks to assess whether a management strategy allows for both sets of constraints to be observed simultaneously. The biological and socio-economic viability constraints characterize an acceptable sub-region of the phase space within which the fishery evolves. A particular trajectory followed by the fishery will be called viable if it remains in this region during the prescribed period of time, with a sufficiently high probability. Thus the bio-economic performance of a management strategy entailing a particular vector of effort multipliers u can be evaluated by the probability of co-viability of the fishery under this strategy, as defined by the equation (3.16) :

$$\mathrm{CVA}(u) = \mathbb{P}\Big(\text{constraints (3.12) and (3.14) are satisfied for } t = t_0, \ldots, T\Big). \tag{3.16}$$

Of particular interest are the vectors of effort multipliers by sub-fleet u such that the probability of co-viability CVA is high enough :

$$\mathrm{CVA}(u) \geq \beta,$$

where β stands for some confidence level (typically 90%, 95% or 100%).

3.2.5 Management strategies

We compare different management strategies relying on different combinations of effort multipliers u_f. In particular, we compare approaches which would focus on partial management of the fishery, centred on the bio-economic viability of harvesting a particular species, with strategies which attempt to manage the mix of species and fleets as a whole. The associated effort multipliers can differ between sub-fleets but it is assumed for sake of simplicity that they remain constant over time. Projections are computed over twenty years ($T = 2028$) starting from the initial stock abundances $\mathrm{N}(t_0)$ at year $t_0 = 2008$. The values of initial states are given by tables 3.3, 3.4 and 3.5 in the appendix. For each management strategy, viability probabilities are approximated by the proportion of viable trajectories among 1000 simulated trajectories. Each trajectory corresponds to different recruitment levels $\omega(.) = (\omega_1(.), \omega_2(.), \omega_3(.))$ and prices $\widetilde{\omega}(.) = (\widetilde{\omega}_1(.), \widetilde{\omega}_2(.), \widetilde{\omega}_3(.))$ for the three species, randomly selected every year according to equations (3.5) and (3.9). These ω_i and $\widetilde{\omega}_i$ are assumed to be independent and identically distributed.

In the following paragraphs we outline the specifications of each strategy.

The status quo (SQ) strategy simulates continued fishing mortalities at levels observed in the 2008 baseline year:

$$u_f^{\mathrm{SQ}} = 1, \qquad \forall f = 1, \ldots, 16.$$

The net present value strategy u^{NPV} is a conventional economic strategy where a central planner aims at maximizing the expected sum of discounted profits at the scale of the entire fishery. There is no guarantee that the profit of each sub-fleet will be positive due to the absence of constraints on these profits. The net present value is calculated as the aggregated value of discounted profits over all the sub-fleets:

$$\mathrm{NPV}(u) = \mathbb{E}\left[\sum_{t_0}^{T} \frac{1}{(1+\delta)^t} \sum_{f=1}^{16} \pi_f(t)\right], \tag{3.17}$$

where the discount rate is set to $\delta = 0.04$[8]. The combination of effort multipliers which define this

[8]This value of discount rate is used for the evaluation of public projects in France (Portney and Weyant, 1999; Lebègue, 2005).

strategy u^{NPV}, is such that:

$$\text{NPV}(u^{\text{NPV}}) = \max_u \text{NPV}(u). \tag{3.18}$$

The co-viability strategy u^{CVA} intends to guarantee both the conservation of Spawning Stock Biomass of all three species and the socio-economic viability of all the fishing sub-fleets. A central planner requires that both the biological and socio-economic constraints defined in (3.12) and (3.14) are satisfied. The associated combinations of effort multipliers u^{CVA} are identified such that they maximize the co-viability probability CVA(u) described in (3.16):

$$\text{CVA}(u^{\text{CVA}}) = \max_u \text{CVA}(u). \tag{3.19}$$

Capital inertia and the related difficulties in reducing excess capacity in fisheries are important issues which often plague fisheries management policies (Nøstbakken et al., 2011). To take this constraint into account, the selection of management strategies is carried out such that the distance between the values of u_f and status quo u_f^{SQ} is minimized, ensuring that the capacity adjustments identified as viable entail the least changes in fleet sizes possible. In other words,

$$|u^{\text{CVA}} - u^{\text{SQ}}| = \min\Big(|u - u^{\text{SQ}}|,\ \text{CVA}(u) = \max_z \text{CVA}(z)\Big) \tag{3.20}$$

The Sole (SOL) strategy investigates a mono-specific management strategy focused on the viability of the Sole fishery. In this sense it is a less cooperative strategy than CVA. The effort multipliers u^{SOL} only account for constraints on the Sole SSB$_3(t)$ and profitability goals $\pi_f(t)_{f=8,\ldots,11}$ for the Sole gill-netter sub-fleets ($f = 8, \ldots, 11$):

$$\begin{cases} \text{SSB}_3(t) \geq B_3^{\text{pa}}, \\ \pi_f(t) > 0 \text{ for } f = 8, \ldots, 11, \\ u_f = 1 \text{ for } f \neq 8, \ldots, 11. \end{cases} \tag{3.21}$$

The associated effort multipliers u^{SOL} are obtained by maximizing the probability that the Sole viability objectives will be met, as follows:

$$u^{\text{SOL}} \in \operatorname{Argmax}_u \mathbb{P}\Big(\text{constraints (3.21) are satisfied for } t = t_0, \ldots, T\Big). \tag{3.22}$$

Again, among the different solutions u^{SOL}, the one with minimal capacity change are selected as in (3.20).

The *Nephrops* (NEP) strategy investigates a mono-specific management strategy focused on the

Nephrops fishery. Similarly to the SOL strategy, the effort multipliers u^{NEP} are selected such that only the constraints related to the stock of *Nephrops* $\text{SSB}_1(t)$ and to profits $\pi_f(t)_{f=1,2,3}$ of the *Nephrops* trawlers ($f = 1, 2, 3$) are considered:

$$\begin{cases} \text{SSB}_1(t) \geq B_s^{\text{pa}}, \\ \pi_f(t) > 0 \text{ for } f = 1, 2, 3, \\ u_f = 1 \text{ for } f \neq 1, 2, 3. \end{cases} \tag{3.23}$$

Viable combinations of effort multipliers u^{NEP} are identified by maximizing the probability that these objectives are met, as follows:

$$u^{\text{NEP}} \in \underset{u}{\text{Argmax}}\, \mathbb{P}\Big(\text{constraints (3.23) are satisfied for } t = t_0, \ldots, T \Big). \tag{3.24}$$

Again, as in (3.20), among the different solutions u^{NEP}, the one with minimal capacity change are chosen.

The numerical implementations and computations of the model have been carried out with the scientific software SCILAB[9] 5.2.2. The nonlinear optimization problems (equations (3.18), (3.19), (3.22) and (3.24)) were solved numerically using the Scilab routine entitled 'optim_ga' which relies on a genetic algorithm [10].

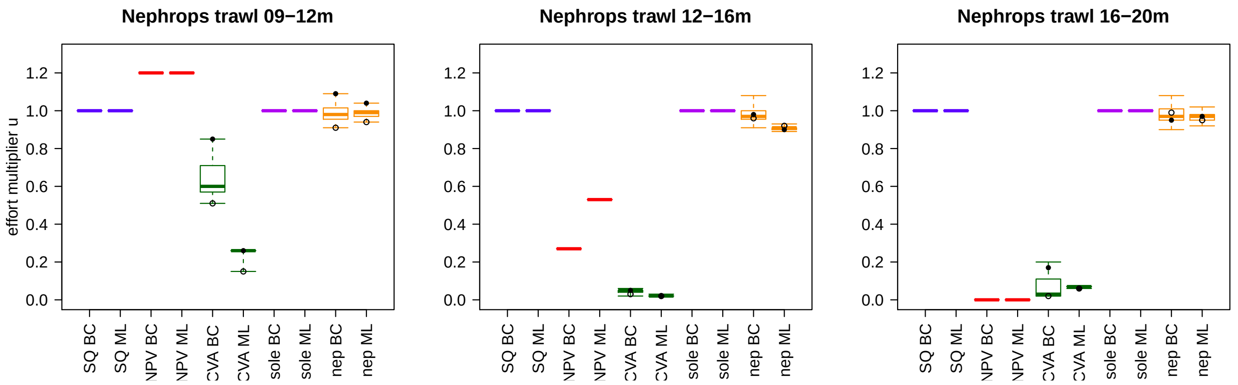
3.3 Results

Outcomes of the five strategies under both fuel scenarios are compared according to the combinations of effort multipliers which they entail, and to their biological and socio-economic performances (biological, socio-economic and co-viability probabilities, as well as net present values).

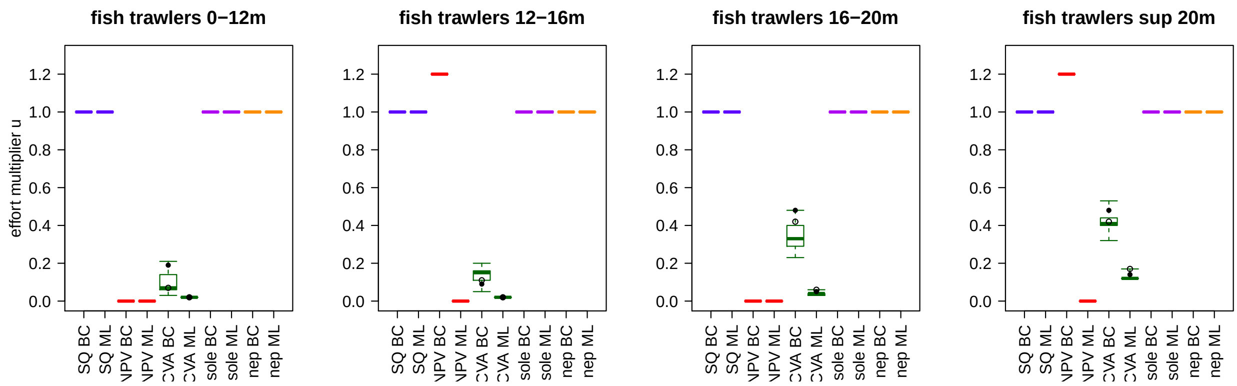
Figure 3.4 displays the effort multipliers u_f by sub-fleet which met the objectives for each strategy under both fuel scenarios. For some management strategies, multiple viable solutions (i.e. effort multiplier values by fleet) are found, therefore effort multipliers on figure 3.4 are represented

[9] SCILAB is a freeware http://www.scilab.org/ dedicated to engineering and scientific calculus. It is especially well-suited to deal with dynamic systems and control theory.

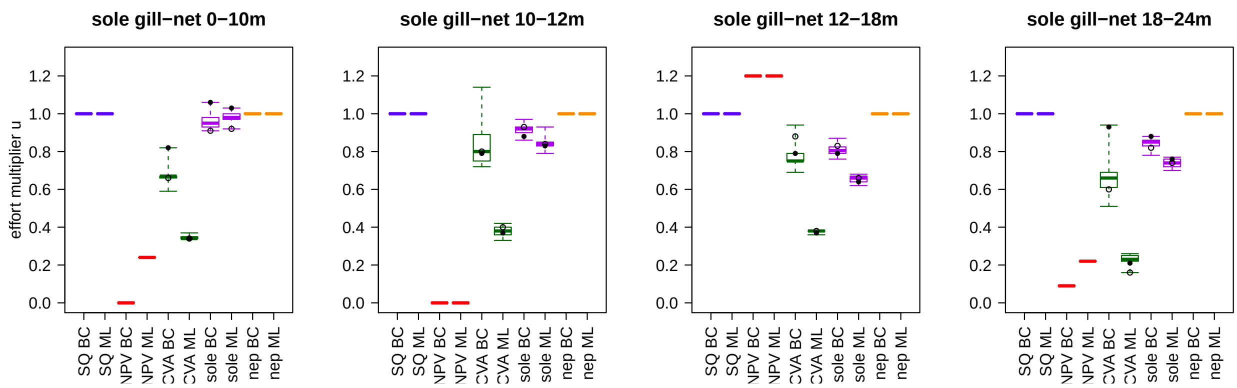
[10] See http://help.scilab.org/docs/5.3.3/en_US/optim_ga.html for details on 'optim_ga'. Genetic algorithms (GAs) are a search procedure based on Darwinian 'survival of the fittest' theory. GAs were developed to solve optimisation problems based on the mechanics of natural selection and genetics such as inheritance, mutation, selection and crossover. The artificial implementation of the natural selection and reproduction into genetic operations have been shown to optimize design problems (Fleming and Purshouse, 2002). GAs optimize by evolving or generating successive populations from an initial random population of individuals to improved populations. This type of numerical method has already been used for bio-economic purposes, for instance in Mardle and Pascoe (2000) and in Sathianandan and Jayasankar (2009).



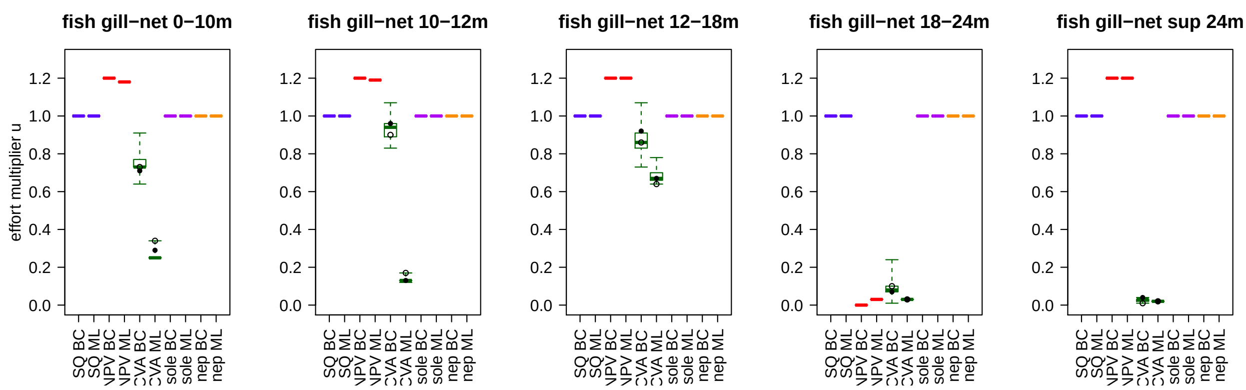
(a) Nephrops trawlers.



(b) Various fish trawlers.



(c) Sole gill-netters.



(d) Various fish gill-netters.

Figure 3.4: Range of the effort multipliers values u_f for the different sub-fleets f and the five strategies (SQ, NPV, CVA, SOL and NEP) under both fuel scenarios BC and ML. For the management strategies where different solution are possible (i.e. CVA, SOL and NEP) two different combinations of effort multipliers are displayed. Plain dots stand for one combination and empty dots for another. (a) Nephrops trawlers. (b) Various fish trawlers. (c) Sole gill-netters. (d) Various fish gill-netters.

by boxplots where range, median and quartiles are represented. To illustrate the fact that, despite the existence of multiple solutions, all combinations of the viable effort multiplier values identified for these sub-fleets are not possible, two particular sets of viable effort multipliers are shown. Table 3.2 gives the biological, socio-economic and co-viability probabilities and net present values associated to the five strategies under both scenarios. The viability probabilities and net present values are calculated using equations (3.13), (3.15), (3.16) and (3.17), respectively. Table 3.2 also provides the number of viable combinations of effort multipliers that were obtained by strategy. Figures 3.5 and 3.6 synthetize the bio-economic scores and trade-offs. Results are detailed in the following subsections.

Table 3.2: Range of the biological and socio-economic viability probabilities (PVA, EVA), co-viability probabilities (CVA) and net present value (NPV) of total fishery profits associated to combinations of effort multipliers obtained for each management strategy. Number of different optimal combination are also given.

Strategies	PVA (in %)	EVA (in %)	CVA (in %)	NPV (in millions of €)	Nb of different solutions
SQBC	98.5	31.1	30.5	685.1	1
SQML	98.5	0	0	525.8	1
NPVBC	100	0	0	1016.6	1
NPVML	100	0	0	934.3	1
CVABC	100	95 - 96.3	95 - 96.3	654.4 - 748.6	41
CVAML	100	94.4	94.4	396.3 - 429.6	53
SOLBC	100	31.5 - 31.7	31.5 - 31.7	709.4 - 711.2	52
SOLML	100	0	0	569.2 - 571.4	42
NEPBC	98.4- 98.7	28.8 - 33.8	.28.3 - 33.3	658.2 - 713.7	83
NEPML	98.7	0	0	555.3 - 562.3	45

3.3.1 Status quo strategy: not socio-economically viable

Figures 3.7 to 3.9 show the projections to 2028, under the status quo strategy, of the $\mathrm{SSB}_s(t)$ of each species and the profits $\pi_f(t)$ of each sub-fleet under base case and most likely fuel scenarios, respectively. Figure 3.7 first illustrates that this strategy is almost biologically viable in the sense that the population viability probability is close to one with PVA(u^{sq})= 98.5%. Only some trajectories for the Sole SSB violate the precautionary threshold B_3^{pa}. The other species fluctuate in safety zones despite the uncertainties affecting their recruitment. In other words, the biological risk is low. By contrast, figures 3.8 and 3.9 show that the socio-economic viability of the fishery appears threatened under this strategy. Indeed EVA(u^{sq})= 31.1% under a base case fuel scenario

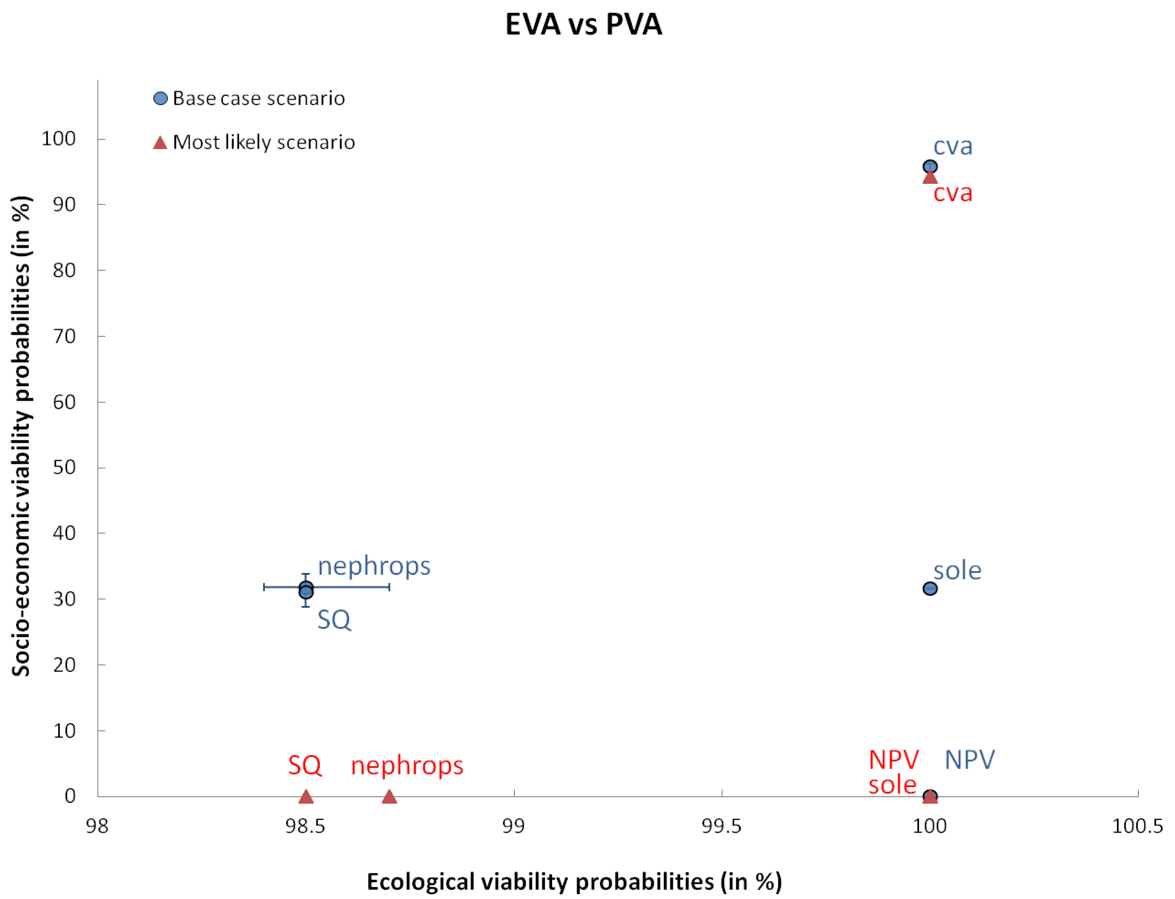


Figure 3.5: Socio-economic EVA(u) versus biological PVA(u) performances of each management strategy under both fuel scenarios. The blue dots represent the socio-economic and biological viabilities of each strategy under the base case scenario and the red triangles under the most likely scenario.

and EVA(u^{SQ})= 0 under a most likely scenario (i.e. the fuel price increase projection). The latter outcome implies that for every 1000 replicates of the strategy, at least one sub-fleet profit becomes negative during a period of time over the projection period.

3.3.2 NPV strategy : high total NPV but not socio-economically viable

As displayed by table 3.2 and figure 3.5, this strategy turns out to be biologically viable with a strong population viability probability PVA(u^{NPV})= 100% as shown in figure 3.5. Thus significant improvements in the status of stocks occur in the long run especially for *Nephrops* and Sole species. However, even though the global net present value of the fishery as a whole is higher than with other management strategies (c.f. figure 3.6), the NPV strategy is not socio-economically viable for some sub-fleets, the profitability of which vanishes. This leads to a collapse of the socio-economic

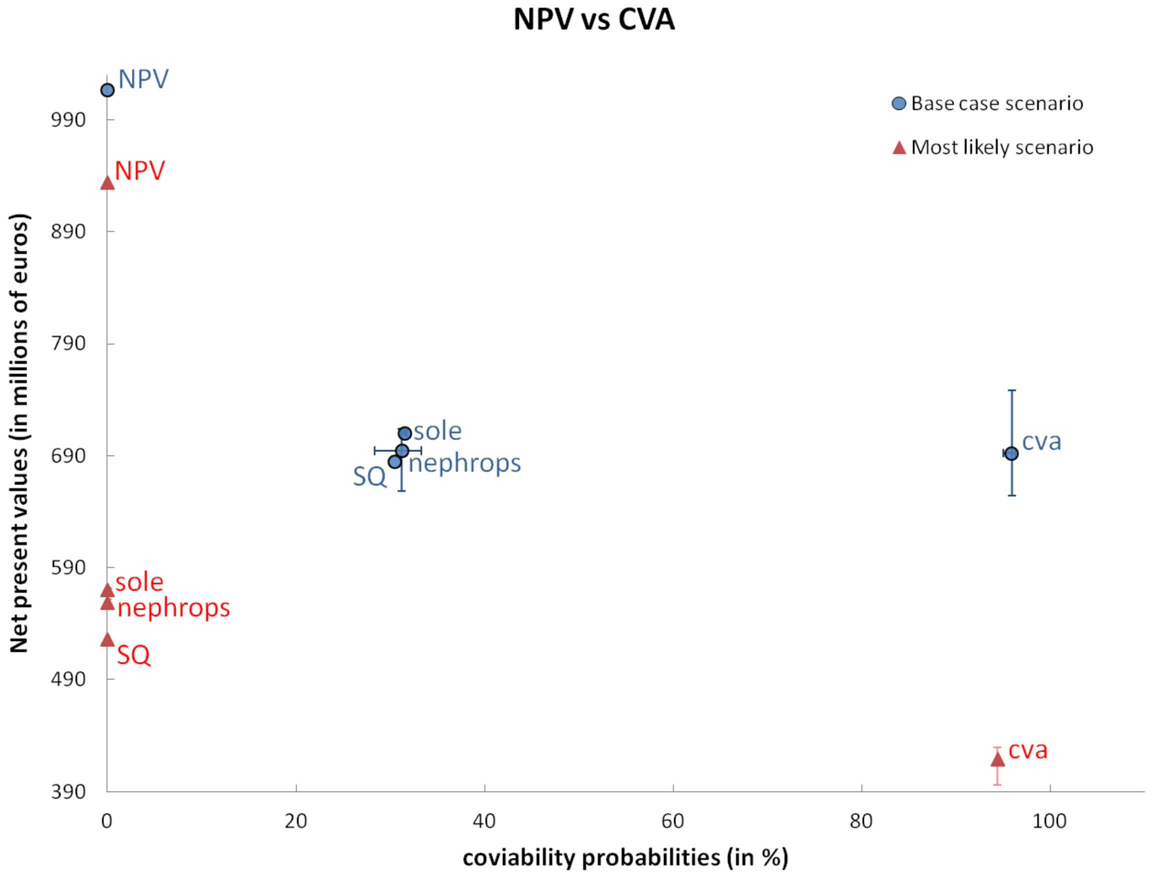


Figure 3.6: Mean net present values NPV(u) versus co-viability probabilities CVA(u) of each management strategy under both fuel scenarios. The blue dots stand for the strategies under the base case scenario and the red triangles under the most likely scenario.

viability probability, as defined in equation (3.15), with EVA(u^{NPV})= 0% under both scenarios (table 3.2 and figure 3.5), and results from the fact that the strategy requires these sub-fleets to become inactive (u_f= 0), e.g. the larger *Nephrops* trawlers or some of the various fish trawlers and Sole gill-netters as illustrated in figure 3.4. This result, based on the structure of interactions and socio-economic and technical parameter values used in these simulations, seems to indicate that these sub-fleets are relatively less efficient than other sub-fleets in the model; hence their contribution to catches and landings under a NPV strategy would be reduced to zero. In other words, the lack of viable outcomes under such a NPV strategy is due to intra-fleet heterogeneity, in terms of technical efficiency, costs and prices. The various fish trawlers of 12-16 meters and greater than 20 meters are active and see their capacity increase under a base case scenario; however these sub-fleets become inactive under a most likely scenario, due to their sensitivity to fuel prices as they use important quantities of fuel.

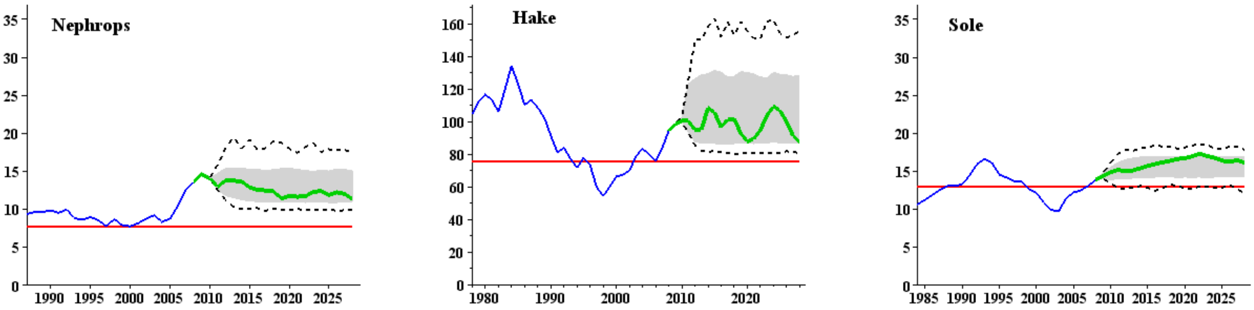


Figure 3.7: Trajectories of the spawning stock biomass $\mathrm{SSB}_s(t)$ of each species s (in thousands of tonnes) with a status quo strategy (i.e. with $u_f^{\mathrm{sq}}(t)$= 1 for all f). The viability thresholds are in red (i.e. Bpa reference points by species). The set of possibilities that includes all of the 1000 simulated trajectories is represented by the dark dotted lines and the grey field includes 95% of the trajectories. The green line is one particular trajectory among the 1000 trajectories associated to the same set of $\omega(.)$ and $\widetilde{\omega}(.)$ for each sub-figure of figures 3.7 to 3.9. The lines in blue represent the estimated historical SSB for each species: *Nephrops* (s = 1), Hake (s = 2) and Sole (s = 3).

3.3.3 CVA strategy: biologically and socio-economically viable

The effort multipliers combinations u^{CVA} associated to the CVA management strategy maximize the probability of co-viability, mixing biological and socio-economic constraints as defined in equations (3.12) and (3.14). As expected, this strategy is biologically viable with guaranteed population viability (PVA(u^{CVA})= 100%) as displayed in figure 3.5; i.e. biomass trajectories lie above the precautionary thresholds B_s^{pa} for every species. Moreover the socio-economic performance of this strategy is also high, with a socio-economic viability probability superior to 95% (EVA(u^{CVA}) $\geqslant$ 95%) as illustrated in the figure 3.5. In 95% of the cases, every sub-fleet exhibits strictly positive profit throughout time. As shown in figure 3.4, such bio-economic outcomes are obtained through redistributing fishing effort among the sub-fleets. While no sub-fleet is made inactive, the strategy leads to significant reductions in the capacity of some sub-fleets, mostly trawlers that have an important impact on modelled species. Moreover, to reduce socio-economic risk (i.e. to increase the probability of socio-economic viability), the sub-fleets with the most variability in their profit must decrease their capacity under this strategy. Figure 3.6 shows that while the profitability of each fleet is guaranteed and the socio-economic risk is reduced, the global socio-economic performance of the fishery (i.e. NPV) is smaller. Not surprisingly, the loss in NPV is stronger under a most likely scenario.

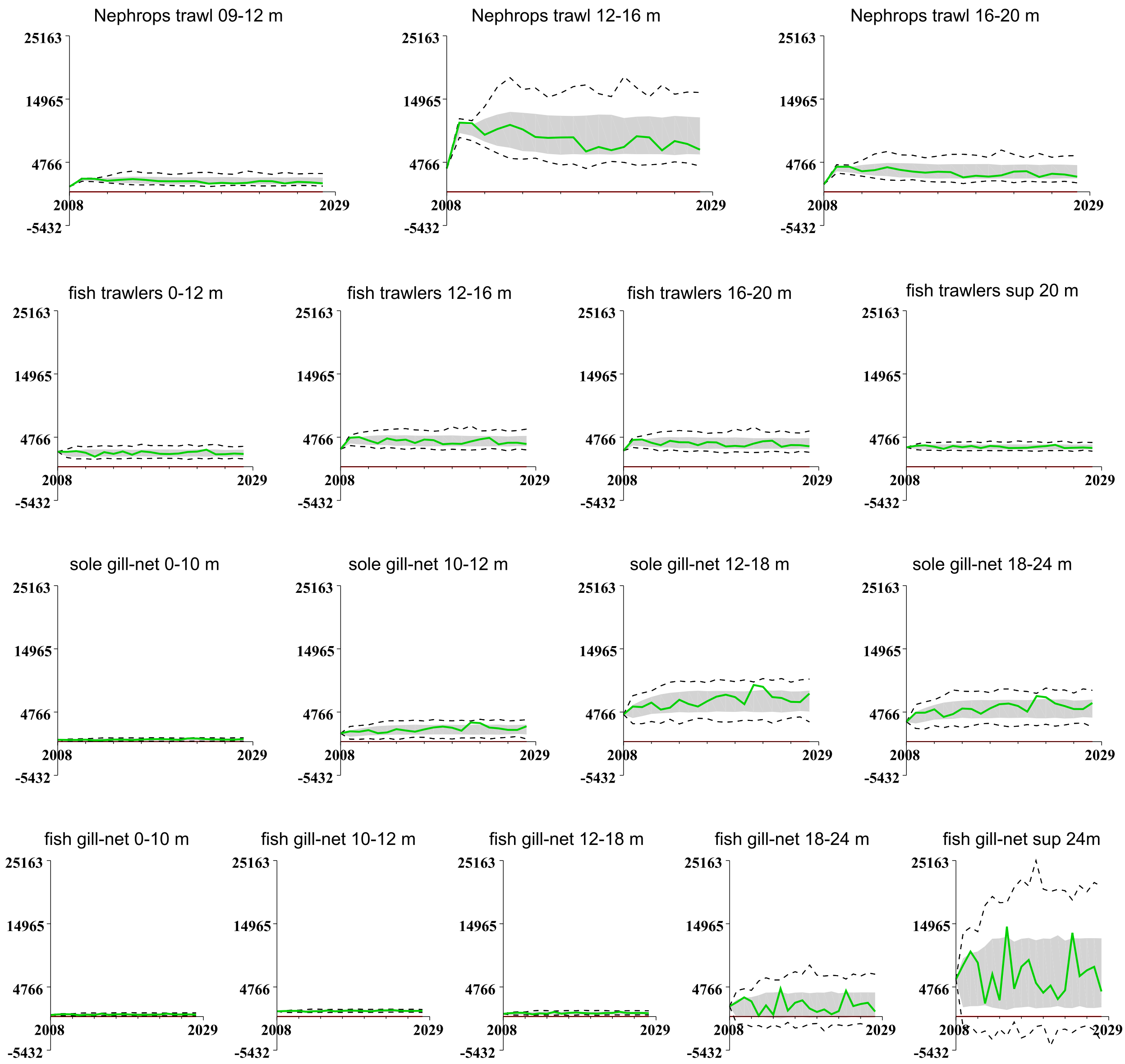


Figure 3.8: Trajectories of the profits $\pi_f(t)$ (in thousands of €) of each sub-fleet according to time t under a base case fuel scenario BC with a status quo strategy (i.e. with $u_f^{sq}(t)$= 1 for all f). The viability thresholds are in red (i.e. zero, strictly positive profits required). The set of possibilities that includes all of the 1000 simulated trajectories is represented by the dark dotted line and the grey field includes 95% of the trajectories. The green line is one particular trajectory among the 1000 trajectories associated to the same set of $\omega(.)$ and $\widetilde{\omega}(.)$ for each sub-figure of figures 3.7 to 3.9.

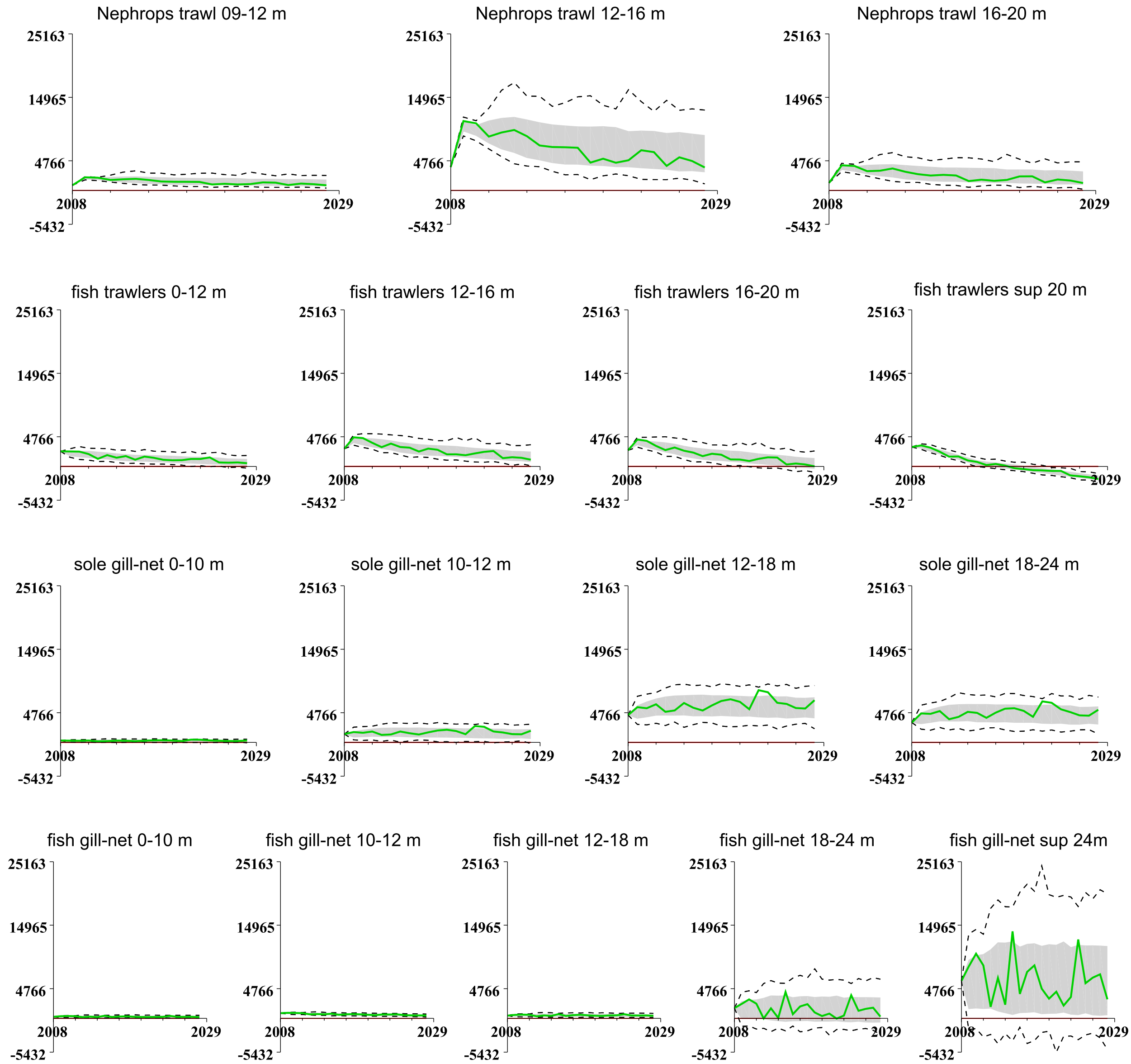


Figure 3.9: Trajectories of the profits $\pi_f(t)$ (in thousands of €) of each sub-fleet according to time t under a most likely fuel scenario ML with a status quo strategy (i.e. with $u_f^{SQ}(t)$= 1 for all f). The viability thresholds are in red (i.e. zero, strictly positive profits required). The set of possibilities that includes all of the 1000 simulated trajectories is represented by the dark dotted line and the grey field includes 95% of the trajectories. The green line is one particular trajectory among the 1000 trajectories associated to the same set of $\omega(.)$ and $\widetilde{\omega}(.)$ for each sub-figure of figures 3.7 to 3.9.

3.3.4 Sole strategy: not socio-economically viable

The SOL strategy involves mono-specific management targeting Sole as defined by equation (3.21). Hence only effort multipliers related to Sole gill-netters (f= 8,9,10,11) are affected as presented in figure 3.4. The results suggests that the sub-fleet of smaller Sole gill-netters (f= 8,9,10) should be favoured under such a strategy. This strategy is biologically viable as PVA(u^{SOL})= 100% (figure 3.5). Similarly, the socio-economic viability - in particular for the smallest Sole gill-netter sub-fleets - is slightly improved as shown by the figure 3.5 and table 3.2. However the strategy as a whole is not socio-economically sustainable as the socio-economic viability probability EVA(u^{SOL}) varies between 31.5 and 31.7% depending on the effort multipliers combinations for a base case fuel scenario and is equal to zero under a most likely scenario. Indeed, some sub-fleets, especially various fish gill-netters do not benefit from the strategy as their fishing effort remains fixed.

3.3.5 *Nephrops* strategy: not socio-economically viable

The NEP strategy is a mono-specific management strategy targeting the *Nephrops* fishery only, as defined by equation (3.23). Hence only effort multipliers u_f for *Nephrops* trawlers f= 1,2,3 are impacted by this strategy, as shown in figure 3.4. The results suggest that the sub-fleet of smaller vessels f= 1 should be favoured while the sub-fleet of moderate size trawlers should see its capacity slightly reduced. This especially occurs under a most likely scenario. This strategy appears to be biologically acceptable since it implies a high overall population viability probability (98.4 < PVA(u^{NEP}) < 98.7%), although some risks persist for Sole, like in the baseline SQ, as stressed by figure 3.5. As expected, the status of the *Nephrops* stock is improved, especially under a most likely scenario where larger *Nephrops* trawlers are subjected to more important capacity reduction. Similarly, the socio-economic viability for the *Nephrops* trawlers sub-fleets is maintained. However such a mono specific strategy is not co-viable as defined here, since the profitability of other sub-fleets, particularly for various fish gill-netters, is threatened, as in the status quo strategy. Note however that NEP and SOL strategies display higher net present value than the status quo strategy as shown in figure 3.6 under both fuel scenarios.

3.4 Discussion

3.4.1 Decision support for the Bay of Biscay mixed fishery

The modelling approach we propose to address trade-offs in managing the French Bay of Biscay demersal fishery allows us to directly compare the biological and socio-economic outcomes of single-stock management strategies with the outcomes of management strategies defined for the mix of species and fleets as a whole. More specifically, we can compare management strategies which would attempt to manage the *Nephrops* fishery or the Sole fishery separately - as has historically been the case until recent years - to two alternative fishery-wide management approaches aimed at (i) maximizing economic yield, or (ii) adjusting the capacity of fishing sub-fleets so that both the biological and the socio-economic viability constraints can be met for every single species and sub-fleet.

Based on our simulations and regarding the species and fleets modelled, it appears that the role of biological constraints in determining viable management strategies is weak as compared to socio-economic constraints. This is confirmed by assessing the impact of the biological constraints on the different strategies u, through calculation of the difference EVA(u) - CVA(u) between the probability of socio-economic viability and the probability of co-viability based on values of the table 3.2. The fact that for strategies SQ, NPV and CVA, this difference is zero, suggests that the marginal impact of the biological constraints is weak. In other words, given the bio-economic situation of the fishery as captured in our model for the late 2000s, sustainability risks in the fishery are rather socio-economic.

Based on the data used to calibrate the model and economic assumptions of the model (in particular assumptions on incomes from other species and on constant annual costs per vessel), simulation results show that the status quo strategy - consisting of maintaining fishing effort of all sub-fleets at their levels of the 2008 baseline - is not socio-economically viable. This outcome holds especially true under a most likely scenario where fuel increase projection are taken into account. The results indicate that, given the status of the species and fleets in 2008, and the nature of the technical interactions between fleets through bycatch and discards, there appears to be excess capacity in the fishery as a whole. Hence all alternative management strategies tested lead to some reduction in the capacity of the fleets. These results are not surprising as the Bay of Biscay demersal fisheries have suffered of chronic overcapacity (EC, 2009). These fisheries are currently managed through total allowable catch limits, and a limited entry system for *Nephrops* and Sole fisheries. However, management of the fisheries has historically been carried out per species, rather than in an integrated, more cooperative, approach. Therefore, the current situation in the fishery is probably relatively close to our simulated mono-specific management strategies, in which attempts are made to maximize the profits of different fleets individually, with no consideration for the global profit in the fishery as a whole. Under the new European management plan, future multi-species management definitions (EC, 2009) should improve the management of the Bay of Biscay demersal fisheries. The objective of such integrated management strategies could for example be to achieve Maximum Economic Yield (Grafton et al., 2010) at the scale of the fishery. However, our research highlights the existence of an alternative co-viability strategy which could be seen as more acceptable as it entails less drastic adjustments in capacity across fleets and regions, at least in a transition phase. The simulations allow to explore the trade-offs associated with alternative management approaches. Single-species management strategies improve the bio-economic status for the specific fishery they target, but do not achieve satisfactory results at the global scale. While the Sole management strategy achieves biological viability, it leads some sub-fleets other than the Sole gill-netters to become unprofitable. Depending on its effort multipliers combinations, the *Nephrops* management strategy produces better fishery-wide results as regards the socio-economic performance of the fleets, but still induces negative profits for some sub-fleets, as well as a moderate level of risk that the biological viability of the fishery will not be guaranteed. The two fishery-wide strategies we examine produce strongly contrasted outcomes. The net present value strategy achieves its objective of maximizing high level of socio-economic performance for the overall fishery. Despite the management systems in place, and a series of decommissioning plans which have been aimed at reducing the capacity of these fleets, the fishery seems far from realising its Maximum Economic Yield objective. Our approach provides an explanation for this, based on the observation that a strategy aimed at maximizing the net present value of profits derived from the fishery as a whole would actually lead to quite heterogeneous impacts on the fleets and sub-fleets. This management strategy can thus not be considered as socio-economically viable in the sense we have defined for the purpose of this analysis. This is because the strategy leads to stop certain sub-fleets entirely from fishing, in particular larger *Nephrops* trawlers, various fish trawlers and certain sub-fleets of Sole gill-netters. Such an outcome is a direct result of the differences in economic efficiency between sub-fleets and technical interactions between them that were assumed in the model. The fact that this makes the NPV strategy non-viable can be interpreted as capturing the resistance which may develop against such a strategy from the segments and regions of the fishery which would be negatively impacted.

The need to capture such constraints in bio-economic simulations of the potential benefits of alternative fisheries management strategies has been increasingly recognised (see Martinet et al. (2010) and Péreau et al. (2012) for recent applications).

Simulation results show that the co-viability strategy can be achieved by a number of combinations of capacity adjustments, which all allow the biological and socio-economic viability constraints to be met for all species and sub-fleets. The strategy however also points to the need for a global reduction of capacity even if reallocations in effort are not quite as drastic as the one suggested by the NPV strategy, and allow some activity to be maintained in all sub-fleets. This strategy might be expected to more easily achieve consensus among the multiple stakeholders involved in the fishery. Indeed, the NPV strategy does not provide much flexibility in the selection of capacity reduction across fleets, so leaves little room for negotiations. Circumstances under which such a strategy might be expected to be more easily adopted would be where the owners of vessels belonging to the sub-fleets which remain active and benefit from the adjustment could buy-out or compensate the owners of those vessels that are requested to leave (Clark et al., 2005; Martell et al., 2009). In this vein, Holland et al. (1999); Guyader et al. (2004); Squires (2010) give examples where buybacks of vessels, licences, gear, access, and other use and property rights can be considered as a useful transition policy tool to address overcapacity, overexploitation of fish stocks and distributional issues in fisheries.

The assumption of increasing fuel price, which is likely to occur, is detrimental to the all the bio-economic outcomes of simulations, as captured in figures 3.5 and 3.6. However, it does not change the nature of the qualitative outcomes and analysis.

3.4.2 CVA as step towards integrated management for mixed fisheries

It is increasingly recognised that a wide range of stakeholders are involved in fisheries and their management, including industrial, artisanal, subsistence and recreational fishermen, suppliers and workers in allied industries, managers, scientists, environmentalists, economists, public decision makers and the general public (Hilborn, 2007b). Each of these groups has an interest in particular outcomes from fisheries and the outcomes that are considered desirable by one stakeholder may be undesirable to another group. The consideration of this multi-dimensional nature of marine fisheries management appears as an unavoidable reality, which should influence the nature of decision-support tools used to assist in the decision-making processes associated with the selection of fisheries regulations. The bio-economic modelling framework we propose offers both formal recognition of the multi-objective nature of management strategies, and means to integrate this with current understanding of the dynamics of a mixed fisheries system, in assessing the trade-offs associated with alternative approaches to regulate such systems. The model illustrates the benefits of formally combining integrated bio-economic modelling with the multi-criteria evaluation underlying the viability framework of analysis. This allows management strategies to be assessed from a range of perspectives including the standard criteria of fish stock preservation and fishery-level economic efficiency, as well as other dimensions which have less frequently been included in formal bio-economic modelling approaches, such as concern for the maintenance of active and profitable fishing fleets. In addition, the viability approach allows characterizing management strategies in terms of their degree of flexibility, with some management strategies offering more options than others in terms of implementation. Since alternative options are bound to have different distributional impacts, it could be expected that strategies offering more alternatives may stand better chances of being adopted, as they provide greater 'bargaining space' for the stakeholders to reach consensus. In the simulation results obtained in this study, this is the case of the CVA strategy, which seems to provide greater adaptation options than the net present value strategy. Such characteristics of management strategies would appear particularly important in the context of mixed fisheries management, which application of the ecosystem approach requires to be managed as a whole, rather than in separate component fisheries. It is likely that models allowing alternative management strategies to be compared in this respect will have greater chances to be adopted as decision support tools in the future.

3.4.3 Perspectives

To go further, several authors (e.g. Mullon et al. (2004); Cury et al. (2005); Chapel et al. (2008)) have proposed the viability approach as a well-suited modelling framework for Ecosystem-Based Fishery Management (EBFM). EBFM must manage targeted species in the context of the overall state of the system, habitat, protected species, and non targeted species. The dynamics considered can potentially include complex mechanisms such as trophic interactions, competition, metapopulations dynamics or economic investment process to quote a few. Here the focus is on technical interactions through a multi-fleets and multi-species context, in particular the bycatch of hake by trawlers. For this specific case-study, the comparison of mono-specific approaches for Sole or *Nephrops* with the more integrated perspective of CVA stresses the importance of integrating management across the complex set of interactions that define these fisheries. Several expansions of this bio-economic model could be considered which could provide useful insights in support of an ecosystem approach to the management of the Bay of Biscay demersal fisheries. This could include the addition of other important commercial demersal species - for example Anglerfish (*Lophius piscatorius* and *L. budegassa*) which is another key species landed by some fleets. Moreover, many studies relating to trawling show that this fishing technique can also impact habitats, through re-suspension of the sediments, and impacts on the structure of benthic communities (Collie et al., 2000) that entail variations in the ecological production processes (Jennings et al., 2001). Therefore it could be important to also consider the ecological impacts of trawling in the evaluation of management strategies, which would mean including both the interactions between fishing levels and patterns across sub-fleets and the benthic habitats, and identifying levels of acceptable impacts on the basis of which to set additional viability constraints. On the human side of the analysis, it could also be important to explicitly capture the Spanish and Belgian fleets, if the approach is to become relevant as a decision-support tool for joint management of the Bay of Biscay's fisheries. In addition, the assumption of constant effort over time should be relaxed in order to promote more adaptive strategies.

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3.5 Appendix

This appendix displays the values of the biological and economic parameters used to calibrate the bio-economic model presented in section 3.2.2. Tables 3.3 to 3.5 show the parameters values of stock dynamics and average market price by species and by age. Tables 3.6 to 3.8 give the estimated values of fishing mortalities by species, by age and by sub-fleet. The biological reference points B_s^{lim}, B_s^{pa} and average historical recruitment $\overline{R}_s$ for every species are displayed in table 3.9. Tables 3.10 to 3.12 show the estimated values of discard rates by species, by age and by sub-fleet. And tables 3.13 and 3.14 give the values of the economic parameters and fishing effort in 2008 for every sub-fleet.

Table 3.3: *Nephrops* parameters ($s = 1$), $t_0 = 2008$. *Source: ICES; Ifremer, SIH, DPMA* .

Age a	1	2	3	4	5	6	7	8	9
Initial abund. $N_{1,a}(t_0)$ ($*10^3$ indv)	642616	650008	328988	180528	65279	23173	8304	4257	4679
Maturity $\gamma_{1,a}$	0	0	0,75	1	1	1	1	1	1
Mean weight(kg\ indv) $\upsilon_{1,a}$	0,004	0,009	0,016	0,027	0,037	0,046	0,058	0,068	0,091
Natural mortality $M_{1,a}$	0,3	0,3	0,25	0,25	0,25	0,25	0,25	0,25	0,25
Commercial category coefficient $\Upsilon_{1,a}$	0.97	0.97	0.87	0.87	0.87	0.87	1.4	1.4	1.68
Mean price(€\ kg) μ_1^P	10.46								
Standard deviation price(€\ kg) σ_1^P	0.453								

Table 3.4: Hake parameters ($s = 2$), $t_0 = 2008$. *Source: ICES; Ifremer, SIH, DPMA*.

Age a	0	1	2	3	4	5	6	7	8+
Initial abund. $N_{2,a}(t_0)$ ($*10^3$ indv)	236062	132608	61571	25195	5219	1606	497	162	45
Maturity $\gamma_{2,a}$	0	0,11	0,73	0,93	0,99	1	1	1	1
Mean weight(kg\ indv) $\upsilon_{2,a}$	0,029	0,25	0,716	1,572	2,503	3,452	4,393	5,773	6,747
Natural mortality $M_{2,a}$	0,4	0,4	0,4	0,4	0,4	0,4	0,4	0,4	0,4
Commercial category coefficient $\Upsilon_{2,a}$	0.54	0.54	0.79	1.11	1.49	1.87	1.87	1.87	1.87
Mean price(€\ kg) μ_2^P	3.69								
Standard deviation price(€\ kg) σ_2^P	0.78								

Table 3.5: Sole parameters ($s = 3$), $t_0 = 2008$. *Source: ICES; Ifremer, SIH, DPMA*.

Age a	2	3	4	5	6	7	8+
Initial abund. $N_{3,a}(t_0)$ ($*10^3$ indv)	23191	17416	10707	4864	3425	2627	2590
Maturity $\gamma_{3,a}$	0,32	0,83	0,97	1	1	1	1
Mean weight(kg\ indv) $\upsilon_{3,a}$	0,189	0,241	0,297	0,352	0,423	0,449	0,599
Natural mortality $M_{3,a}$	0,1	0,1	0,1	0,1	0,1	0,1	0,1
Commercial category coefficient $\Upsilon_{3,a}$	0.69	0.82	0.99	1.14	1.14	1.14	1.12
Mean price(€\ kg) μ_3^P	12.41						
Standard deviation price(€\ kg) σ_3^P	0.8						

Table 3.6: The values of fishing mortality on *Nephrops* ($s = 1$): $F_{1,a,f}(t_0)$. *Source: ICES; Ifremer, SIH, 2008.*

Fleets	Sub-fleets	Age a								
		1	2	3	4	5	6	7	8	9
Nephrops	09-12 m	0.002	0.027	0.039	0.039	0.034	0.035	0.035	0.035	0.035
trawlers	12-16 m	0.01	0.161	0.233	0.232	0.205	0.206	0.207	0.209	0.209
	16-20 m	0.004	0.057	0.083	0.083	0.073	0.073	0.074	0.074	0.074
Various fish	0-12 m	0	0.003	0.004	0.004	0.004	0.004	0.004	0.004	0.004
trawlers	12-16 m	0.002	0.03	0.044	0.043	0.038	0.039	0.039	0.039	0.039
	16-20 m	0.002	0.035	0.051	0.051	0.045	0.045	0.045	0.046	0.046
	sup 20 m	0	0.008	0.011	0.011	0.01	0.01	0.01	0.01	0.01
Sole	0-10 m	0	0	0	0	0	0	0	0	0
gill-netters	10-12 m	0	0	0	0	0	0	0	0	0
	12-18 m	0	0	0	0	0	0	0	0	0
	18-24 m	0	0	0	0	0	0	0	0	0
Various fish	0-10 m	0	0	0	0	0	0	0	0	0
gill-netters	10-12 m	0	0	0	0	0	0	0	0	0
	12-18 m	0	0	0	0	0	0	0	0	0
	18-24 m	0	0	0	0	0	0	0	0	0
	sup 24 m	0	0	0	0	0	0	0	0	0
Other fleets		0.001	0.015	0.021	0.021	0.019	0.019	0.019	0.019	0.019

Table 3.7: The values of fishing mortality on Hake ($s = 2$): $F_{2,a,f}(t_0)$. *Source: ICES; Ifremer, SIH, 2008.*

Fleets	Sub-fleets	Age a								
		0	1	2	3	4	5	6	7	8+
Nephrops	09-12 m	0.009	0.004	0.001	0	0	0	0	0	0
trawlers	12-16 m	0.051	0.026	0.006	0.001	0.001	0.001	0.001	0.001	0.002
	16-20 m	0.033	0.016	0.004	0	0.001	0.001	0.001	0.001	0.001
Various fish	0-12 m	0.016	0.013	0.006	0.002	0.002	0.001	0	0	0
trawlers	12-16 m	0.018	0.015	0.007	0.002	0.003	0.001	0	0	0
	16-20 m	0.016	0.013	0.006	0.002	0.002	0.001	0	0	0
	sup 20 m	0.011	0.009	0.004	0.001	0.002	0	0	0	0
Sole	0-10 m	0	0	0	0	0.001	0	0	0	0
gill-netters	10-12 m	0	0	0	0.001	0.002	0.001	0	0	0
	12-18 m	0	0	0	0.002	0.004	0.002	0.001	0	0
	18-24 m	0	0	0.001	0.005	0.008	0.004	0.001	0.001	0
Various fish	0-10 m	0	0	0	0.001	0.002	0.001	0	0	0
gill-netters	10-12 m	0	0	0	0.001	0.002	0.001	0	0	0
	12-18 m	0	0	0	0.002	0.004	0.002	0.001	0	0
	18-24 m	0	0	0.005	0.025	0.044	0.023	0.008	0.003	0.002
	sup 24 m	0	0.001	0.013	0.067	0.119	0.062	0.022	0.009	0.005
Other fleets		0.022	0.253	0.444	0.734	0.764	0.843	0.728	0.875	0.88

Table 3.8: The values of fishing mortality on Sole ($s = 3$): $F_{3,a,f}(t_0)$. *Source: ICES; Ifremer, SIH, 2008.*

Fleets	Sub-fleets	Age a						
		2	3	4	5	6	7	8+
Nephrops trawlers	09-12 m	0.002	0.002	0.001	0.001	0.001	0.001	0.001
	12-16 m	0.009	0.011	0.009	0.008	0.008	0.008	0.008
	16-20 m	0.005	0.006	0.005	0.004	0.004	0.004	0.004
Various fish trawlers	0-12 m	0.014	0.017	0.013	0.01	0.007	0.007	0.007
	12-16 m	0.014	0.018	0.014	0.012	0.013	0.013	0.013
	16-20 m	0.017	0.021	0.016	0.014	0.015	0.015	0.015
	sup 20 m	0.007	0.009	0.007	0.006	0.007	0.006	0.006
Sole gill-netters	0-10 m	0.002	0.005	0.008	0.008	0.01	0.009	0.011
	10-12 m	0.011	0.028	0.042	0.045	0.053	0.052	0.059
	12-18 m	0.018	0.065	0.087	0.094	0.148	0.145	0.138
	18-24 m	0.015	0.054	0.072	0.078	0.123	0.121	0.115
Various fish gill-netters	0-10 m	0	0.001	0.002	0.002	0.002	0.002	0.002
	10-12 m	0.001	0.003	0.005	0.005	0.006	0.006	0.007
	12-18 m	0.001	0.003	0.004	0.004	0.006	0.006	0.006
	18-24 m	0	0	0	0	0	0	0
	sup 24 m	0	0	0	0	0	0	0
Other fleets		0.062	0.113	0.072	0.072	0.09	0.079	0.083

Table 3.9: Biological reference points B_s^{lim}, B_s^{pa} and mean recruitment $\overline{R}_s$ for every species. This last one is computed over 1987-2006 for the *Nephrops*, 1992-2006 for the Hake and 1993-2006 for the Sole. *Source: ICES; Ifremer, SIH.*

	Nephrops	Hake	Sole
B_s^{lim} (tonnes)	7733	54521	9706
B_s^{pa} (tonnes)	7733	75784	13000
$\overline{R}_s$ (10^3 individuals)	699387	241776	23414

Table 3.10: Estimated discard in percentage for *Nephrops* ($s = 1$): $d_{1,a,f}$. *Source: ICES; Ifremer, SIH, 2008.*

Main fleets	Age a								
	1	2	3	4	5	6	7	8	9
Nephrops trawlers	0.999	0.972	0.344	0.063	0.023	0.013	0.014	0.017	0.01
Various fish trawlers	0.999	0.972	0.344	0.063	0.023	0.013	0.014	0.017	0.01
Sole gill-netters	0.999	0.972	0.344	0.063	0.023	0.013	0.014	0.017	0.01
Various fish gill-netters	0.999	0.972	0.344	0.063	0.023	0.013	0.014	0.017	0.01

Table 3.11: Estimated discard in percentage for Hake ($s = 2$): $d_{2,a,f}$. *Source: ICES; Ifremer, SIH, 2008.*

Main fleets	Age a								
	0	1	2	3	4	5	6	7	8+
Nephrops trawlers	0.999	0.374	0.	0.	0.	0.	0.	0.	0.
Various fish trawlers	0.998	0.237	0.	0.	0.	0.	0.	0.	0.
Sole gill-netters	0.	0.	0.	0.	0.	0.	0.	0.	0.
Various fish gill-netters	0.	0.	0.	0.	0.	0.	0.	0.	0.

Table 3.12: Estimated discard in percentage for Sole ($s = 3$): $d_{3,a,f}$. *Source: ICES; Ifremer, SIH, 2008.*

Main fleets	Age a						
	2	3	4	5	6	7	8+
Nephrops trawlers	0.15	0.01	0.	0.	0.	0.	0.
Various fish trawlers	0.15	0.01	0.	0.	0.	0.	0.
Sole gill-netters	0.15	0.01	0.	0.	0.	0.	0.
Various fish gill-netters	0.15	0.01	0.	0.	0.	0.	0.

Table 3.13: Initial number of vessels $K_f(t_0)$, effort by vessel $e_f(t_0)$ and rate of extra fishing income α_f of the sixteen sub-fleets. *Source: Ifremer, SIH, DPMA, 2008.*

Fleets	Length(m)	Nb vessel : $K_f(t_0)$	Fishing effort/vessel (nb day at sea): $e_f(t_0)$	Income from other species (in €/ effort unit): α_f
Nephrops trawlers	09-12 m	19	170.3	297
$f = 1, 2, 3$	12-16 m	75	183.4	429
	16 -20 m	22	177.	716
Various fish trawlers	0-12 m	110	157.7	622
$f = 4, 5, 6, 7$	12-16 m	45	192.7	1375
	16-20 m	49	180.3	1751
	sup 20 m	37	197.1	3597
Sole gill-netters	0-10 m	28	139.	311
$f = 8, 9, 10, 11$	10-12 m	42	145.5	503
	12-18 m	40	202.9.	765
	18-24 m	23	201.7	1150
Various fish gill-netters	0-10 m	32	153.8	303
$f = 12, 13, 14, 15, 16$	10-12 m	30	178.8	847
	12-18 m	6	145.	1466
	18-24 m	9	210.3	1500
	sup 24 m	10	260.6	1141

Table 3.14: Mean reference costs of the sixteen sub-fleets. *Source: Ifremer, SIH, DPMA, 2008*

Fleets	Length(m)	Landing cost τ_f	Volume of fuel by fishing effort unit (in L), V_f^{fuel}	Variable cost by fishing effort unit by vessel (in €), c_f^{var}	Annual costs by vessel (in €), c_f^{fix}
Nephrops trawlers	09-12 m	0.04	482	58	101837
$f = 1, 2, 3$	12-16 m	0.05	653	81	174104
	16-20m	0.07	925	160	234836
Various fish trawlers	0-12 m	0.05	257	44	77779
$f = 4, 5, 6, 7$	12-16 m	0.05	863	108	218506
	16-20 m	0.07	1076	188	245285
	sup 20 m	0.07	1999	308	388951
Sole gill-netters	0-10 m	0.06	78	70	56601
$f = 8, 9, 10, 11$	10-12 m	0.05	290	140	132326
	12-18 m	0.08	348	213	256373
	18-24 m	0.07	622	453	378872
Various fish gill-netters	0-10 m	0.05	59	28	42874
$f = 12, 13, 14, 15, 16$	10-12 m	0.05	248	69	111911
	12-18 m	0.06	396	230	223622
	18-24 m	0.07	811	595	513353
	sup 24 m	0.03	1099	556	913096

Chapter 4. Risk versus economic performance in a mixed fishery: the case of the Northern Prawn Fishery in Australia

In preparation for *Ecological Economics*:

Gourguet S., Thébaud O., Dichmont C., Jennings S., Little L.R., Pascoe S., Deng R.A., Doyen L.

The interest of the co-viability approach for the sustainable management of the French Bay of Biscay demersal mixed fishery has been investigated in chapters 2 and 3. To examine the general characteristic of this approach and its application to a different ecosystem with a contrasted historic and political context, another mixed fishery is studied in this thesis: the Australian Northern Prawn Fishery (NPF). The complexity of this fishery is analysed through a weekly multi-species, size-structured bio-economic model with dynamic allocation of fishing effort across multiple fishing strategies. This model is investigated in this chapter to examine the trade-offs between mean economic performance of the fishery and the variance of this performance, under a range of economic scenarios and strategies with respect to fleet capacity and effort allocation.

Abstract

Balancing bio-economic risks and high profit expectations is often a major concern in fisheries management. We examine this trade-off in the context of the Australian Northern Prawn Fishery (NPF), which is managed to achieve Maximum Economic Yield (MEY). The fishery derives its revenue from different prawn species with more or less uncertain dynamics and recruitment. A multi-species bio-economic and stochastic model is used to examine the trade-offs between mean economic performance of the fishery and the variance of this performance, under a range of economic scenarios and strategies with respect to fleet capacity and effort allocation. Simulation results show that the observed fishing strategy displayed by the fleet might be interpreted as seeking the best compromise between performance and risk. Increases in fleet size or in the annual fishing effort of vessels would only improve the expected economic performance of the fishery at the cost of increased variability of this performance. Under a likely economic scenario, adaptation of the fishery to maintain current levels of economic performance is likely to depend on the extent to which operators in the fishery are willing to accept higher levels of economic risk.

Keyword: Bio-economic modelling, uncertainty, risk-performance trade-offs, scenarios, fishing strategy, Northern Prawn Fishery.

4.1 Introduction

Globally, many capture fisheries do not achieve their full economic potential and are subject to excess capacity (Munro, 2010). For some fisheries, this may be due to failure in regulating the race to fish. Other fisheries may be managed to achieve Maximum Sustainable Yield (MSY), rather than Maximum Economic Yield (MEY). In some cases, social considerations may have dominated the management decision process leading to the approval of even higher levels of capacity. In other cases, differences between observed harvesting levels of individual species and the levels which would ensure MEY may be related to the fact that commercial fishers operate across a range of species, with varying ability to target these species separately, leading to difficulties in identifying optimal fishery-wide levels of fishing capacity and allocation of fishing effort. Moreover, revenues from fisheries may vary greatly from year to year owing to natural variation in fish stocks (Kasperski and Holland, 2013) that cannot be predicted with any reliability, leading to varying levels of economic risks for fishing operators (Sethi, 2010). In multi-species fisheries, the different fish stocks contributing to the overall catch may present different levels of natural variability, such that the choice of fishing strategies can be associated with trade-offs between mean and variance of the fishery's economic yield. While maximising economic yield is usually seen as a desirable objective for fisheries management, industry stakeholders usually also value stability over time. This may be due to risk aversion, but also to the need to maintain markets, avoid market saturation and guide investment decisions relating to non-malleable capital (Holland and Herrera, 2009).

This article focuses on the analysis of trade-offs between mean performance and performance variability of economic yield in a fishery, managed with the objective of achieving MEY, but in which the set of target species have different levels of environmentally-driven variability of recruitment. The analysis is based on a bio-economic modelling approach, and is applied to the case of the Northern Prawn Fishery (NPF) in Australia.

The NPF, which is located off Australia's northern coast (figure 4.1), is a multi-species trawl fishery based on several tropical prawn species.

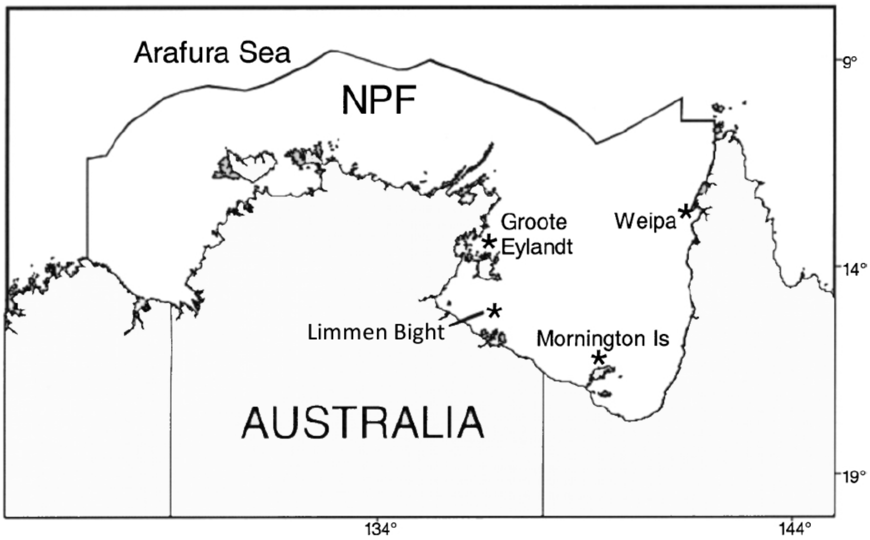


Figure 4.1: Map of northern Australia showing the extent of the Northern Prawn Fishery (Milton, 2001).

The NPF has a long history of collaborative management involving scientists, industry, and the fishery managers (Australian Fisheries Management Authority (AFMA)). It is one of Australia's most valuable federally managed commercial fisheries, and has regularly returned a positive profit

(Rose and Kompas, 2004) since its establishment in the late 1960s. However, in recent years the fishery has experienced a decline in value as a result of the increased supply of aquaculture-farmed prawns to both domestic and international markets, strong Australian currency and increasing fuel prices (Punt et al., 2011). Of the fifty species of prawns that inhabit Australia's tropical northern coastline, the trawl fishery targets only nine commercial species of prawns including tiger, banana and endeavour prawns. Revenue in the fishery is mostly obtained from the harvest of white banana prawns (*Penaeus merguiensis*), grooved tiger prawns (*Penaeus semisulcatus*) and brown tiger prawns (*Penaeus esculentus*), these three species accounting for 95% of the total annual landed catch value of the fishery (ABARE-BRS, 2010).

The NPF operates over two 'seasons' spanning the period April to November with a mid-season closure of variable length from June to August. Seasonal closures are in place to protect small prawns (closure from December to March), as well as spawning individuals (mid-season closure) (AFMA and CSIRO, 2012). The fishery effectively consists of two sub-fisheries that are (to a large degree) spatially and temporally separate. The 'banana prawn sub-fishery' is effectively a single species fishery based on the white banana prawn, while the 'tiger prawn sub-fishery' is a mixed species fishery targeting grooved and brown tiger prawns, as well as blue endeavour prawns (*Metapenaeus endeavouri*) which are caught as by product[1] (Woodhams et al., 2011).

The banana prawn sub-fishery operates mostly during the first season, which generally lasts between four and eight weeks depending on recruitment, while the tiger prawn sub-fishery occurs mostly during the second season (although in poor banana prawn years may start earlier). White banana prawns form dense aggregations ('boils') which are easily identified from spotter planes. White banana prawn stocks are strongly influenced by weather patterns, and the highest seasonal catches generally follow higher than average rainfall during the preceding summer (Vance et al., 1985). The variability of white banana prawn stocks makes it difficult to set catch or effort limits in a way that protects spawning stocks but also allows operators to profit from years in which prawns are abundant (Buckworth et al., 2013). The effort during the first season depends very little on tiger or endeavour prawn abundances.

[1] A third sub-fishery exists in the Joseph Bonaparte Gulf in the far western part of the fishery based on red-leg banana prawns. This sub-fishery is exploited by a relatively small number of vessels as it occurs at the same time as the (more valuable) tiger prawn sub-fishery, and is not included in the subsequent analysis.

In the second season, the fleet switches to the tiger prawn sub-fishery, for which catches per unit effort are lower but less variable. However, if banana prawns are still available in large enough numbers (catch rate for banana prawns above 500kg/boat per day), some vessels will continue to target them[2]. Key aspects of the biology, habitat requirements, catchability and value differ between the major target species of the tiger and banana prawn sub-fisheries in ways that have an important bearing on management. Tiger and blue endeavour prawn stocks are more stable and predictable than the white banana prawn stock. Moreover the former species are generally more dispersed relative to white banana prawns. Consequently, even though the same vessels are used in both sub-fisheries, the fishing gears and techniques differ. While banana prawns are caught in daytime trawls of relatively short duration (but lot of time searching), tiger prawns are taken at night (daytime trawling is banned during the tiger prawn season to reduce the capture of spawning tiger prawns, (AFMA and CSIRO, 2012)). The two tiger prawn species exhibit some spatial and temporal separation, with brown tiger prawns being dominant in the first part of the season and grooved tiger prawns being dominant in the second part. However, by-catch of the other tiger prawn species as well as endeavour species is prevalent over the whole season, providing an example of a classic mixed fishery in which it is impossible to target one species perfectly (Pascoe et al., 2010).

The NPF is currently managed using input controls in the form of limited entry, gear restrictions, as well as time and spatial closures. Management of the fishery has been supported by the development and application of a full Management Strategy Evaluation (MSE) approach (Dichmont et al., 2006, 2008; Venables et al., 2009). Following several industry and government funded buy-back schemes, the NPF now comprises 52 vessels, which is believed to be the number required to achieve Maximum Economic Yield (MEY) in the fishery (Barwick, 2011). By comparison, more than 120 vessels operated in the fishery a decade ago, and over 300 vessels in the 1970s and 1980s.

To date, bio-economic analysis of the fishery has been largely focused on the more predictable component of the fishery, namely the tiger prawn fishery (Dichmont et al., 2008, 2010; Punt et al., 2011). The analysis presented here uses a simplified representation of the bio-economic dynamics of the fishery, integrating the more variable banana prawn resource. Trade-offs between expected mean performance and risk associated in a selection of possible management strategies for the NPF,

[2]This happens relatively infrequently and in most years white banana prawn catches are less than two percent of the total catch in the tiger prawn sub-fishery.

taking into account the distribution of fishing effort across sub-fisheries, are assessed. The trade-offs are first examined under current management, and performances are then compared under a range of fishing strategies, including analysis of sensitivity to different assumptions regarding changes in fuel and prawn prices.

4.2 Material and Methods

The bio-economic model developed here synthesizes previous modelling work by Dichmont et al. (2003, 2008) and Punt et al. (2010, 2011) on the NPF, and extends it by explicitly modelling both tiger and banana prawn sub-fisheries. The model is based on recent developments in mixed fisheries bio-economic modelling (Gourguet et al., 2013). It includes white banana prawns, grooved tiger, brown tiger prawns and blue endeavour prawns. All of them are short lived species, but white banana prawns are with highly variable recruitments and endeavour prawns with less variable recruitments. Our analysis captures the major components and interactions that characterise the NPF (figure 4.2), as described in section 4.1.

4.2.1 A multi-species, stochastic and dynamic model

Population dynamics of tiger and blue endeavour prawns are based on a multi-species weekly time-step, sex-structured population model with Ricker stock-recruitment relationship and environmental uncertainties. The population dynamics model allows for week-specificity in recruitment, spawning, availability and fishing mortalities. Dynamics of grooved and brown tiger prawns are based on size-structured models, whereas the dynamics of blue endeavour is based on an aggregated population model. White banana prawns are represented without explicit density-dependence mechanisms, due to highly variable recruitment and absence of a defined stock-recruitment- relationship.

Tiger prawns: sex- and size-structured population dynamic models

Catches of grooved and brown tiger prawns are recorded and marketed together as 'tiger prawns'. However, since each species has a unique life cycle and occupies a different ecological niche, it is important that the population dynamics model be species-specific. The population dynamics of grooved and brown tiger prawns (s = 1, 2, respectively) are based on a sex- and size-structured

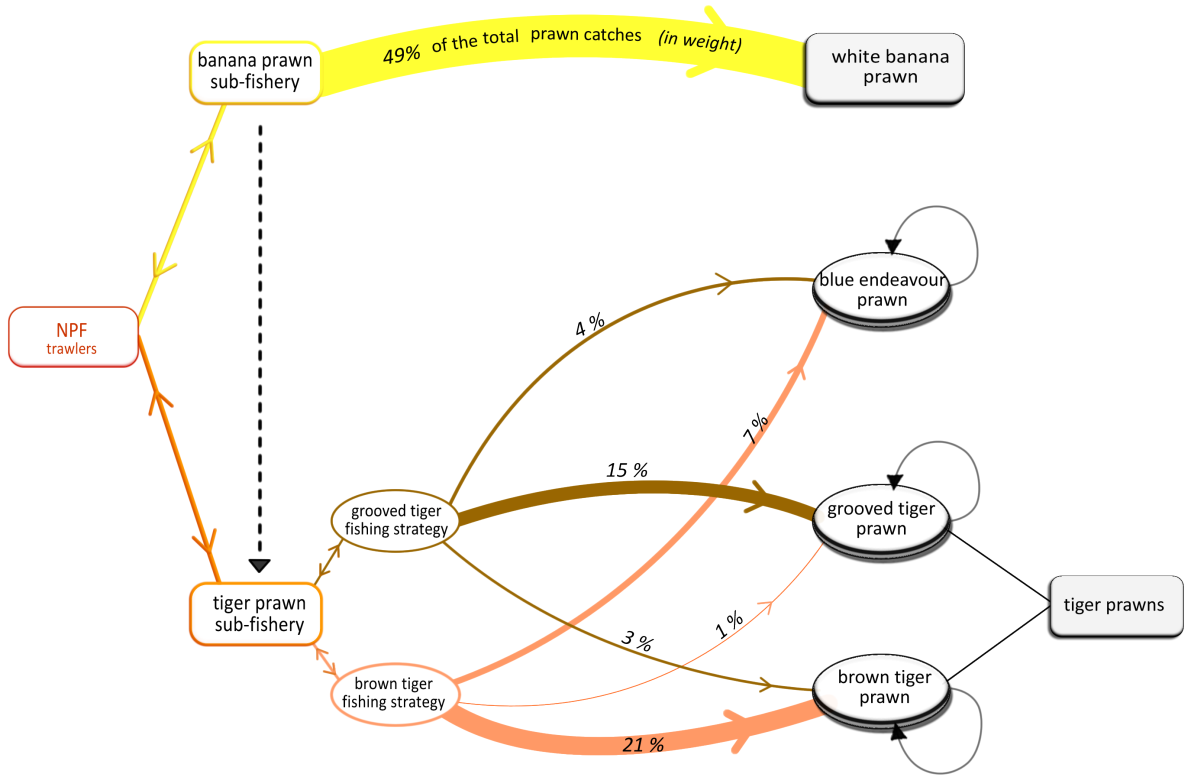


Figure 4.2: Stylized representation of the Northern Prawn Fishery used as a basis to develop the bio-economic model. The width of the arrows between the sub-fisheries and various prawns are proportional to the proportion of the catch by species by sub-fishery (tiger - with differentiation of grooved and brown tiger prawn fishing strategies - and banana prawn sub-fisheries) compared to the total catch of the fishery in 2010. The dashed arrow between tiger and banana prawn sub-fisheries illustrates the influence of the banana prawn season on the tiger prawn fishing effort.

model relying on a weekly time-step as presented in Punt et al. (2010) and given by the equation (4.1):

$$\vec{\mathrm{N}}_s(t+1) = f\Big(t, \vec{\mathrm{N}}_s(t), \vec{F}_s(t)\Big), \qquad s = 1, 2. \tag{4.1}$$

where $\vec{\mathrm{N}}_s(t)$ is the vector of abundance $\mathrm{N}_{s,x,l}(t)$ of prawns of species s of sex x (with $x = ♂$ for male and ♀ for female) in size-class l (1-mm size-classes between lengths of 15 to 55 mm) alive at the start of time t which corresponds to one time step (i.e. one week), and $\vec{F}_s(t)$ is the vector of fishing mortality $F_{s,l}(t)$ of animals of species s and size-class l at time t. Details on fishing mortality are given in section 4.2.2. The dynamic function f accounts for the species recruitment and mortality mechanisms as detailed in appendix A section 4.5.1.

Biological indicators are represented via spawning stock size index $\mathrm{S}_s(y(t))$ of the species s for the year $y(t)$ given by:

$$\mathrm{S}_s(y(t)) = \frac{1}{52} \sum_{t=52\left(y(t)-1\right)+1}^{52y(t)} \beta_s(t) \sum_l \gamma_{s,l} \frac{1-\exp\left(-Z_{s,l}(t)\right)}{Z_{s,l}(t)} \mathrm{N}_{s,♀,l}(t), \qquad s = 1, 2. \tag{4.2}$$

where $y(t)$ is the year corresponding to the time t. $\beta_s(t)$ measures the relative amount of spawning of species s during time t, and $\gamma_{s,l}$ corresponds to the proportion of females of species s in size-class l that are mature. $Z_{s,l}(t)$ stands for the total mortality of animals of species s in size-class l at time t and is defined by:

$$Z_{s,l}(t) = M_s + F_{s,l}(t). \tag{4.3}$$

with M_s the natural mortality of animal of species s.

Blue endeavour prawn: an aggregated population dynamic model

The population dynamics of blue endeavour prawn (species $s = 3$) is modelled as an aggregated process governed by equations (4.4) and (4.5):

$$\mathrm{N}_{3,x}(t+1) = \mathrm{N}_{3,x}(t) \exp\left(-Z_3(t)\right) + \alpha_3(t) \frac{R_3(\tilde{y}(t))}{2}. \tag{4.4}$$

where $\mathrm{N}_{3,x}(t)$ is the number of blue endeavour individuals of sex x at the start of time t, $Z_3(t)$ is the total mortality of blue endeavour prawns at time t, $\alpha_3(t)$ is the fraction of the annual recruitment of blue endeavour prawns during time t, and $R_3(\tilde{y}(t))$ is the recruitment[3] of blue endeavour prawns during the 'biological' year $\tilde{y}(t)$.

Recruits in the fishery $R_s(\tilde{y}(t)+1)$ for species $s = 1, 2, 3$ during a 'biological' year $(\tilde{y}(t) + 1)$ are assumed to be related to the spawning stock size index $\mathrm{S}_s(y(t))$ of species s for the year $y(t)$, according to a Ricker stock-recruitment relationship fitted assuming temporally correlated environmental variability and down-weighting recruitments, as described in appendix A section 4.5.2.

Weekly biomass $B_{3,x}(t)$ of blue endeavour of sex x at the start of time t is given by:

$$B_{3,x}(t) = \tilde{\upsilon}_{3,x} \mathrm{N}_{3,x}(t). \tag{4.5}$$

with $\tilde{\upsilon}_{3,x}$ the average mass of a blue endeavour prawn of sex x.

[3] the sex-ratio of the recruits is assumed to be 50:50 in the absence of data (Punt et al., 2011).

The spawning stock size index $S_3(y(t))$ of blue endeavour prawn for the year $y(t)$ is given by:

$$S_3(y(t)) = \frac{1}{52} \sum_{t=52\left(y(t)-1\right)+1}^{52y(t)} \beta_3(t) \frac{1-\exp\left(-Z_3(t)\right)}{Z_3(t)} N_{3,\female}(t). \tag{4.6}$$

Parameter $\beta_3(t)$ is defined as for the size-structured model in section 4.2.1.

White banana prawn: an uncertain resource

Abundance of white banana prawns (species $s = 4$) appears to be more heavily influenced by the environment than by fishing pressure (Die and Ellis, 1999; Venables et al., 2011) and its year to year availability is highly variable. More specifically, stocks are strongly influenced by weather patterns, generally peaking in years in which there has been high rainfall. In the present study, white banana prawn annual biomass is modelled as a uniform i.i.d. random variable:

$$B_4(y(t)) \rightsquigarrow \mathcal{U}(B_4^-, B_4^+), \tag{4.7}$$

with $B_4(y(t))$ the stochastic biomass of white banana prawn for the year $y(t)$, and B_4^- and B_4^+ the uniform law bounds.

4.2.2 Fishing mortality and catch

Fishing mortalities $F_{s,l,f}(t)$ due to sub-fishery f ($f = 1$ for grooved tiger prawn sub-fishery and $f = 2$ for brown tiger prawn sub-fishery) on animals of species $s = 1, 2$ (grooved and brown tiger prawns) in size-class l during time t and average fishing mortality $F_{3,f}(t)$ on animals of species $s = 3$ (blue endeavour prawn) from tiger prawn sub-fishery $f = 1, 2$, are given by:

$$\begin{cases} F_{s,l,f}(t) = A_s(t)\mathrm{Sel}_{s,l}q_{s,f}E_f(t), & s = 1,2 \quad \text{and} \quad f = 1,2 \\ F_{s,f}(t) = A_s(t)q_{s,f}E_f(t), & s = 3 \quad \text{and} \quad f = 1,2. \end{cases} \tag{4.8}$$

where $A_s(t)$ is the relative availability of animals of species s during time t and $E_f(t)$ is fishing effort (days at sea) associated with grooved or brown tiger prawn sub-fishery $f = 1, 2$ at time t. Catchability $q_{s,f}$ corresponds to the fishing mortality of species s associated with one unit of fishing effort of fishing strategy f and is assumed constant over the simulation period. $\mathrm{Sel}_{s,l}$ is the selectivity of the fishing gear on animals of species s in size-class l (assumed to be a logistic function of length, identical for both strategies[4], and constrained so that selectivity is < 1 for sizes $< l_\infty$ which is a von Bertalanffy growth curve parameter).

[4]This is assumed in the absence of catch length-frequency data by species and strategy.

Weekly catches $Y_{s,l,f}(t)$ of grooved ($s = 1$) and brown tiger prawns ($s = 2$), by size-class l and weekly catches $Y_{s,f}(t)$ of blue endeavour prawns ($s = 3$) by the grooved and brown tiger prawn fishing strategies ($f = 1, 2$); and annual catches $Y_{4,3}(y(t))$ of white banana prawns ($s = 4$) by the banana prawn sub-fishery ($f = 3$) are defined as in the system of equations 4.9:

$$\begin{cases} Y_{s,l,f}(t) = \sum_x \upsilon_{s,x,l} \mathrm{N}_{s,x,l}(t) F_{s,l,f}(t) \dfrac{1 - \exp\left(-M_s - \sum_{f=1,2} F_{s,l,f}(t)\right)}{M_s + \sum_{f=1,2} F_{s,l,f}(t)} & s = 1,2 \quad \text{and} \quad f = 1,2 \\ Y_{3,f}(t) = \sum_x \tilde{\upsilon}_{3,x} \mathrm{N}_s(t) F_{3,f}(t) \dfrac{1 - \exp\left(-M_3 - \sum_{f=1,2} F_{3,f}(t)\right)}{M_3 + \sum_{f=1,2} F_{3,f}(t)} & f = 1,2. \\ Y_{4,3}(y(t)) = q_{4,3} B_4(y(t)) \mathrm{E}_3(y(t)) \end{cases} \tag{4.9}$$

with $\upsilon_{s,x,l}$ the mass of an animal of species s ($s = 1, 2$) and sex x in size-class l.

4.2.3 Economic component

The economic component of the model estimates the flow of costs and revenues from fishing over time.

Fishing income

The annual gross income from catches by sub-fishery and fishing strategy is calculated from the market price p_s of the species s times the annual catch $Y_{s,f}(y(t))$ of this species s by the fishing strategy f as:

$$\begin{cases} \mathrm{Inc}_f(y(t)) = \displaystyle\sum_{t=52\left(y(t)-1\right)+1}^{52y(t)} \left(\sum_{s=1,2} \sum_l p_{tig,l}(y(t)) Y_{s,l,f}(t) + p_3(y(t)) Y_{3,f}(t) \right), & f = 1,2. \\ \mathrm{Inc}_3(y(t)) = p_4(y(t)) Y_{4,3}(y(t)), \end{cases} \tag{4.10}$$

Grooved and brown tiger prawns are marketed together as 'tiger prawns' under a common size- and time-dependent price, where $p_{tig,l}(y(t))$ is the average market price per kilogram for animals in size-class l (related to five market categories) during the year $y(t)$. The average price per kilogram of blue endeavour and white banana prawns is denoted $p_{s=3,4}(y(t))$ and is also time-, but not size-dependent.

The partial annual gross income of the whole fishery from catches of species s for year $y(t)$ is expressed by $\mathrm{Inc}_s(y(t))$ and calculated as in equation (4.11).

$$\begin{cases} \mathrm{Inc}_s(y(t)) = \displaystyle\sum_{t=52\left(y(t)-1\right)+1}^{52y(t)} \sum_{f=1,2} \sum_{l} p_{tig,l}(y(t)) Y_{s,l,f}(t), & f = 1,2 \quad \text{and} \quad s = 1,2 \\ \mathrm{Inc}_{s=3}(y(t)) = \displaystyle\sum_{t=52\left(y(t)-1\right)+1}^{52y(t)} \sum_{f=1,2} p_s(y(t)) Y_{s,f}(t), & f = 1,2 \quad \text{and} \quad s = 3 \\ \mathrm{Inc}_{s=4}(y(t)) = p_s(y(t)) Y_{4,3}(y(t)), & f = 3 \quad \text{and} \quad s = 4. \end{cases} \tag{4.11}$$

Fishing costs

Variable costs $c_f^{var}(t)$ for the sub-fishery f during time t, and annual fixed costs by vessel c_v^{fix} are detailed in equation (4.12):

$$\begin{cases} c_f^{var}(t) = \mathrm{c}^L \mathrm{Inc}_f(t) + \mathrm{c}^M \displaystyle\sum_{s=1}^{4} Y_{s,f}(t) + \left(\mathrm{c}_f^K + \mathrm{c}_f^F(y(t))\right) \mathrm{E}_f(t) \\ c_v^{fix} = W_v + (o + \varrho)\psi_v \end{cases} \tag{4.12}$$

where c^L is the share cost of labour (crew are paid a share of the income) and c^M is the cost of packaging and gear maintenance (assumed to be proportional to the fishery catch in weight). Unit costs c_f^K and $\mathrm{c}_f^F(y(t))$ are respectively the cost of repairs and maintenance and the cost of fuel and grease per unit of effort of sub-fishery f during the year $y(t)$. The values of these costs are assumed constant across grooved and brown tiger prawn fishing strategies but differ between tiger and banana prawn sub-fisheries. W_v are the annual vessel costs (i.e. those costs are not related to the level of fishing effort), o is the opportunity cost of capital and is assumed equal to the discount rate, set at 5 % following Punt et al. (2011), ϱ is the economic depreciation rate and ψ_v is the average value of capital by vessel.

Annual profit and net present value

The total annual profit $\pi(y(t))$ for the entire NPF for year $y(t)$ is given by:

$$\pi(y(t)) = \sum_{f=1}^{3} \left(\mathrm{Inc}_f(y(t)) - \sum_{t=52\left(y(t)-1\right)+1}^{52y(t)} c_f^{var}(t) \right) - c_v^{fix} \mathrm{K}(y(t)). \tag{4.13}$$

where $\mathrm{K}(y(t))$ is the number of vessels involved in the NPF during the year $y(t)$.

The net present value (NPV) of the flow of profits over simulation time is calculated as the aggregated value of discounted annual profits as in Punt et al. (2010) and is given by:

$$\mathrm{NPV} = \sum_{y(t)=1}^{T-1} \frac{\pi(y(t))}{(1+\delta)^{y(t)-1}} + \frac{[\pi(T)/\delta]}{(1+\delta)^{T-1}}. \tag{4.14}$$

where δ is the discount rate, and $\pi(T)$ is the level of profit during the terminal year of the simulation.

4.2.4 Parameter estimation

Dichmont et al. (2003) and Punt et al. (2010) describe the approaches used to estimate parameter values for the dynamic population models. The impact of parameter uncertainty was explored in Punt et al. (2010). A non-linear least-squares method was used for the estimation of the parameters (B_4^-,B_4^+ and $q_{4,3}$) related to white banana prawn by fitting observed data (c.f. 4.9) of white banana prawn catches (in weight) and annual banana fishing effort over 17 years (1994 to 2010). The values of cost parameters (c^L, c^M, c_f^K, c_f^F, W_v and ψ_v) are derived from an economic survey of the fishery during 2007-2008 (Perks and Vieira, 2010) and were adjusted for known changes in input prices to provide estimates of the costs in 2009-2010 values. The unit packaging and marketing cost parameter (c^M) was estimated by dividing the reported costs by the total catch to give a cost per kilogramme. Average repair and maintenance costs per unit of effort (i.e. per day) (c_f^K) were estimated by dividing the total reported costs by the annual effort (i.e. number of days fished the entire year). Fuel costs per day (c_f^F) were estimated in a similar manner to repair costs accounting for the different number of hours fished per day in the tiger and banana prawn sub-fisheries. The depreciation rate was set as in Punt et al. (2010). All these cost and price assumptions were discussed with, and validated by industry representatives who were members of the NPF Resource Assessment Group (RAG)[5]. Base case values of all biological and economic parameters are given in appendix B (section 4.6).

4.2.5 Effort allocation strategy $\mathrm{T}_{\mathrm{adapt}}$

The total annual fishing effort $\mathrm{E}(y(t))$, for the entire NPF, is calculated by:

$$\mathrm{E}(y(t)) = e(y(t))\mathrm{K}(y(t)). \tag{4.15}$$

[5]The NPF Resource Assessment Group has responsibility for assessing the dynamics and status of NPF species. The group comprises fishery scientists, industry members, fishery economists, and the AFMA manager responsible for the fishery.

where $e(y(t))$ is the annual average effort per vessel for the year $y(t)$ expressed in number of days at sea and $K(y(t))$ the number of vessels for that year. Exogenous technical constraints on $e(y(t))$ and $K(y(t))$ are included in the model and maximal nominal effort per week set at 7 days.

To capture what happens currently in the NPF, this total annual fishing effort is then allocated weekly between tiger and banana prawn fishing, but also between the two tiger prawn species through a simplified, three-step effort allocation model as shown in figure 4.3.

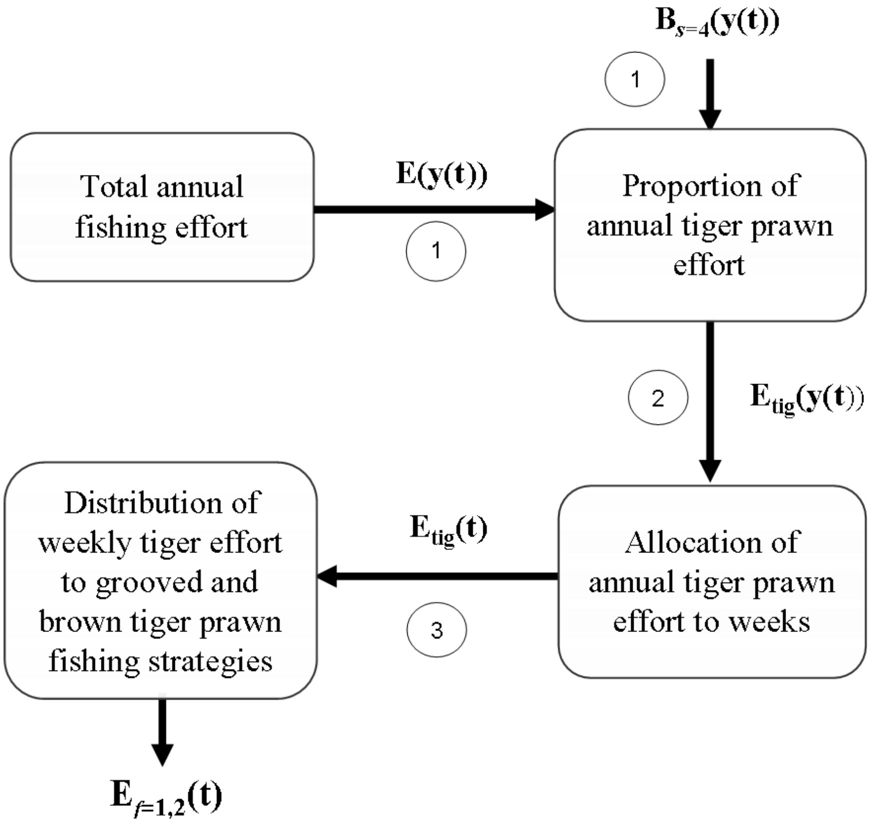


Figure 4.3: Flowchart of the algorithm used to determine the weekly effort (days at sea) during year $y(t)$ of grooved and brown tiger prawn fishing strategies ($f = 1, 2$), based on the total annual effort and white banana prawn annual biomass. The variables next to the arrows represent the output from one box and input into another box. The circles with numbers correspond to the three different steps of the algorithm.

- Step 1: Distribution of tiger and banana prawn annual effort

  An abundant banana prawn year will result in a decrease of the proportion of the annual effort directed to the tiger prawns fishing. The allocation of the total annual fishing effort between tiger and banana prawns fishing therefore depends on white banana prawn annual biomass $B_{s=4}(y(t))$. The arrows (1) in figure 4.3 illustrate this allocation.

A linear regression model between the annual banana prawns catch per unit effort (CPUE) and the annual proportion of tiger prawn effort ($f = 1 + 2$) using historical data from 1994 to 2010[6] was estimated and resulted in specifying the annual proportion of tiger prawn sub-fishery effort given banana prawn CPUE (CPUE_4) as:

$$\begin{cases} \mathrm{E}(y(t)) = \mathrm{E}_{1+2}(y(t)) + \mathrm{E}_3(y(t)), \\ \dfrac{\mathrm{E}_{1+2}(y(t))}{\mathrm{E}(y(t))} = a\mathrm{CPUE}_4(y(t)) + b, \qquad s = 4. \end{cases} \tag{4.16}$$

with $\mathrm{E}_{1+2}(y(t))$ the annual effort of tiger prawn sub-fishery for the year $y(t)$ and $\mathrm{E}_3(y(t))$ the banana prawn sub-fishery annual effort during the year $y(t)$. Details on CPUE are given in appendix A (4.5.3). Parameters a and b are estimated from the linear regression model which is displayed in figure 4.9 in the appendix C.

- Step 2: Weekly tiger prawn sub-fishery effort allocation

An empirical approach is taken to predict the weekly allocation of the tiger prawn sub-fishery effort for year $y(t)$. Because of the great variability of the various weekly effort patterns of the historical years, using a fixed weekly pattern is not relevant. It would assume weekly effort patterns that are not observed in the fishery. A solution could be to randomly select a year among the historical years 1994 to 2010. However, the season start dates and annual effort of banana prawn sub-fishery have an influence on the tiger prawn weekly effort patterns. Sensitivity analyses of weekly patterns would thus be necessary, but would increase the number of simulations to run increasing significantly the level of complexity of the model. For simplicity, for each year $y(t)$ and white banana annual biomass simulated, a historical weekly pattern is selected between 1994 and 2010 according to the proportion of annual effort dedicated to the tiger prawn sub-fishery for the year $y(t)$ (i.e. depending on the white banana annual biomass simulated). This step is illustrated by the arrow (2) in figure 4.3. Appendix D details the algorithm on which depend the selection of the historical weekly pattern.

- Step 3: Grooved and brown tiger prawns fishing strategies effort allocation

The final step of the effort distribution model (arrow (3) in figure 4.3) allocates the weekly tiger effort to the two species (grooved and brown tiger prawns). This is achieved using a

[6]Only historical data after 1994 are taken into account due to major changes in the fishery structure that occurred in that year.

fixed pattern[7] $\Upsilon^{strat}(t \text{ modulo } 52)$ or proportion of weekly tiger prawn effort directed towards grooved prawns ($f = 1$) at time (t modulo 52). The effort by week directed towards grooved ($f = 1$) and brown ($f = 2$) prawns is described by equation (4.17):

$$\begin{cases} \mathrm{E}_{1+2}(t) = \mathrm{E}_1(t) + \mathrm{E}_2(t), \\ \mathrm{E}_1(t) = \Upsilon^{strat}(t \text{ modulo } 52)\mathrm{E}_{1+2}(t). \end{cases} \tag{4.17}$$

4.2.6 Other effort allocation strategies

In this paper, the economic performance of the NPF is compared under various effort allocation strategies characterised by the proportion of effort allocated to the different sub-fisheries, namely more or less effort allocated to tiger or banana prawns. Consequently the strategies also contrast in terms of the resulting weekly effort patterns of tiger and banana prawn fishing (calculated as described in section 4.2.5). The seven strategies, detailed in table 4.1, include an 'adaptive' effort allocation strategy ($\mathrm{T}_{\mathrm{adapt}}$) which corresponds to the full effort allocation model described in section 4.2.5 and which currently reflects the situation in the NPF. The resulting proportion of total annual effort directed to the tiger prawns ranges between 60 and 76%. The six other strategies correspond to alternative 'specialisation effort allocation' strategies in which the annual proportion of total effort allocated to tiger prawns is pre-defined and hence no longer depends on banana prawn biomasses. Therefore under these six alternative effort allocation strategies, only the last two steps of the effort allocation model are applied. Three 'banana specialisation' effort allocation strategies (T_0, T_{10} and T_{20}) consist of setting the annual proportion of tiger prawn effort to 0%, 10% and 20% of total annual effort. Two 'tiger specialisation' effort allocation strategies (T_{90} and T_{100}) involve allocating 90% and 100% of the annual effort to the tiger prawn sub-fishery. Finally a 'balanced' effort allocation strategy (T_{50}) is analysed; in which total annual effort is split equally between the two sub-fisheries. Because of policy and technical constraints, the pattern of open and closed weeks is set to that which occurred in 2010 (i.e. with respect to the mid-season closure).

[7]The values of $\Upsilon^{strat}(t)$ correspond to the predicted proportion of tiger prawn effort directed towards the grooved prawns during week (t modulo 52) in 2010 derived from the CSIRO operating model.

Table 4.1: Effort allocation strategies (in each row) considered in this study. The strategies differ in the annual effort $E_{1+2}(y(t))$ allocated to tiger prawn sub-fishery.

Allocation strategies	Description	Annual effort of tiger prawn sub-fishery
T_0	annual proportion of tiger prawn sub-fishery effort sets to 0%.	$E_{1+2}(y(t)) = 0$
T_{10}	annual proportion of tiger prawn sub-fishery effort sets to 10%.	$E_{1+2}(y(t)) = 0.1E(y(t))$
T_{20}	annual proportion of tiger prawn sub-fishery effort sets to 20%.	$E_{1+2}(y(t)) = 0.2E(y(t))$
T_{50}	annual proportion of tiger prawn sub-fishery effort sets to 50%.	$E_{1+2}(y(t)) = 0.5E(y(t))$
T_{adapt}	'adaptive' effort allocation strategy.	see equations (4.16) and (4.17)
T_{90}	annual proportion of tiger prawn sub-fishery effort sets to 90%.	$E_{1+2}(y(t)) = 0.9E(y(t))$
T_{100}	annual proportion of tiger prawn sub-fishery effort sets to 100%.	$E_{1+2}(y(t)) = E(y(t))$

4.2.7 Fishing capacity management strategies

We assess the effects of changes in fishing capacity using four strategies regarding management of fishing capacity, defined in terms of the number of vessels $K(y(t))$ and days at sea per vessel $e(y(t))$. Table 4.2 summarizes the four fishing capacity strategies.

Table 4.2: Fishing capacity strategies (in each row).

Annual effort settings	Description
SQ	$K(y(t))$=52 vessels and $e(y(t))$=162 days at sea per vessel
e^+	$K(y(t))$=52 vessels and $e(y(t))$=196 days at sea per vessel
K^+	$K(y(t))$=78 vessels and $e(y(t))$=162 days at sea per vessel
K^-	$K(y(t))$=26 vessels and $e(y(t))$=162 days at sea per vessel

The status quo fishing capacity strategy SQ corresponds to an annual number of vessels and total annual effort equal to the values observed in 2010, i.e. $K(y(t))$=52 and $e(y(t))$=162. Under fishing capacity strategy e^+ the number of vessels remains at its 2010 level but annual effort per vessel is increased to that allowed by the maximum number of open weeks (28 weeks). Therefore the annual effort per vessel for the e^+ fishing capacity strategy is set to 196 days at sea. Fishing capacity strategy K^+ incorporates an increase in the annual number of vessel by 50% but leaves the average effort per vessel unchanged, i.e. $K(y(t))$=78 and $e(y(t))$=162. Similarly strategy K^- represents a 50% decrease in the annual number of vessels whereas no change in the average effort by vessel, i.e. $K(y(t))$=26 and $e(y(t))$=162.

4.2.8 Economic scenarios

The key economic outputs from the bio-economic model are the annual profit for the entire NPF and the associated net present value, all of which are sensitive to assumptions about the values of biological and economic parameters described in section 4.2.4. Sensitivity to economic parameters is explored through the analysis of scenarios incorporating different assumptions about changes in fuel and prawn prices. All other economic parameters were assumed to remain constant over the simulation period. We report results for only two economic scenarios[8], these being a 'base case' scenario (BC) and a 'most likely' scenario (ML) detailed in table 4.3.

Table 4.3: Economic scenarios (in each row) considered in this study.

Scenarios	Description
BC	Base case scenario: constant prawn and fuel prices
ML	Most likely scenario: prawn prices decrease by 3% per year and fuel price increases by 5% per year

The BC scenario assumes that prawn and fuel prices remain constant at their estimated 2010 levels over the simulation period. Variable and fixed costs are set to the average values estimated for the 2010-2012 period. The ML scenario represents a most likely evolution over the simulation period of key economic parameters for this fishery. Except for banana prawn[9], the main market for NPF prawns is Asia (especially Japan), and the price received is largely dependent on the Yen-AU$ exchange rate and the total supplies to this market (Punt et al., 2010). Therefore prawn prices are assumed to be independent of the landings of our model. Based on historical trends, the most likely scenario assumes a progressive prawn prices annual decrease of 3%. In this scenario fuel price is assumed to follow a progressive increase of 5% per year. Assumption of fuel price evolution is supported by a linear model from historical data given in figure 4.8 in the appendix C.

The biological and economic performances of the fishery for the seven effort allocation strategies, the four fishing capacity strategies and under the two economic scenarios are analysed accounting for the stochastic nature of the model (i.e. environmental variabilities applied to annual recruitments of tiger and blue endeavour prawns and to white banana prawn annual biomasses). For

[8]We tested different combination of scenarios including increase and decrease of prawn and fuel prices.

[9]Until recently the main market for banana prawns was also Asia, however most of banana prawns are sold now in domestic market.

every combination of effort allocation, fishing capacity strategies and economic scenarios, 1000 trajectories are simulated over a 10 year period from 2010. Each trajectory represents a possible state of nature for each year of the simulation, $\omega(.) = (\omega_1(.),\omega_2(.), \omega_3(.),\omega_4(.))$; which stands for the set of annual recruitments of tiger and blue endeavour prawns as detailed in Punt et al. (2011) and annual biomasses of white banana prawns in equation (4.7). The different $\omega_i(.)$ are assumed to be independent by species. Each combination of strategies and scenarios is simulated with the same set of $\omega(.)$. The numerical implementations and computations of the model have been carried out with the scientific software SCILAB[10].

4.3 Results

Economic performance of the fishery is studied through the mean value and variance of annual profit for the entire NPF and the NPV of the fishery.

4.3.1 Sensitivity of biological and economic performance indicators to economic scenarios

Figure 4.4 shows the evolution of the spawning stock size indices over the simulation period, for each of the three exploited tiger and blue endeavour prawns with the 'adaptive' effort allocation (T_{adapt}) strategy, reflecting the current situation of the fishery, and with the status quo (SQ) fishing capacity strategy. The most likely (ML) economic scenario being an economic scenario, there are no differences between the biological outputs under base case and most likely scenarios. Therefore a common output is displayed in figure 4.4. Figure 4.5 displays the evolution of the total annual profit over the simulation period with the T_{adapt} allocation strategy and SQ fishing capacity strategy, for the BC (figure 4.5(a)) and the ML (figure 4.5(b)) economic scenarios.

According to figure 4.4, the size of the spawning stocks remains relatively stable for both grooved tiger and blue endeavour prawns over the ten-year period to 2020 (with an average increase of 5% and 16%, respectively), while the evolution of the brown tiger prawn spawning stock size index shows an average decline of 32%.

[10]SCILAB is a free software http://www.scilab.org/ dedicated to engineering and scientific calculus. It is especially well-suited to deal with dynamic systems and control theory.

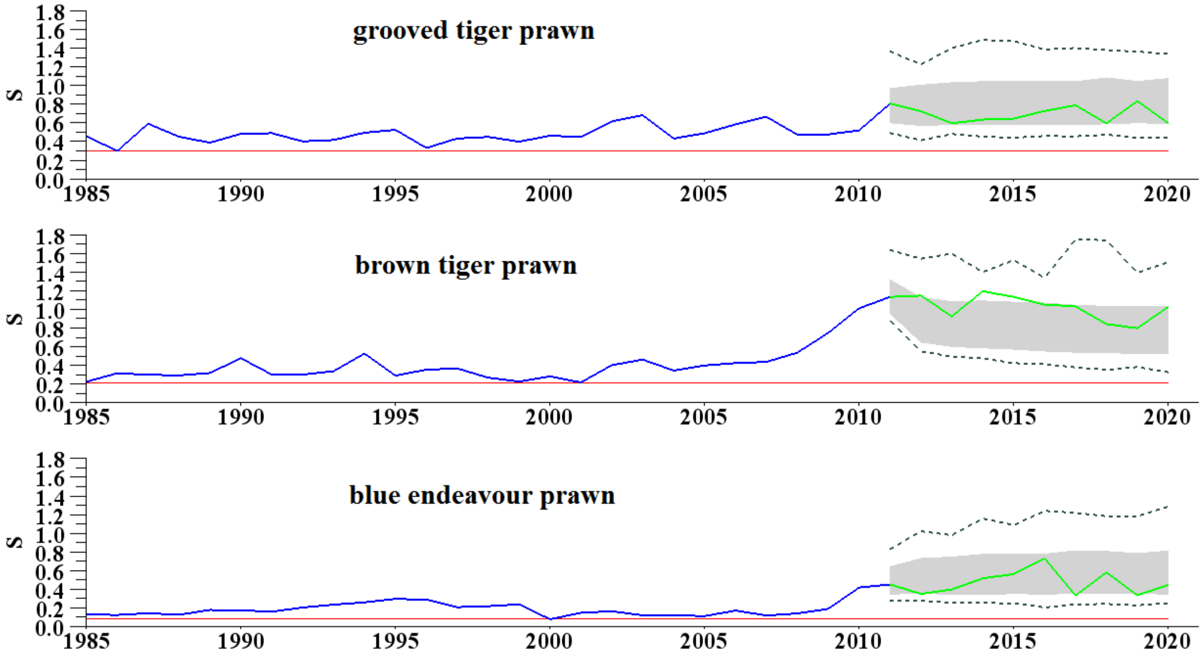
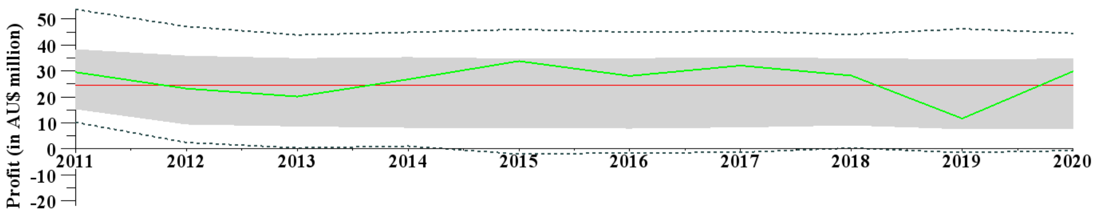
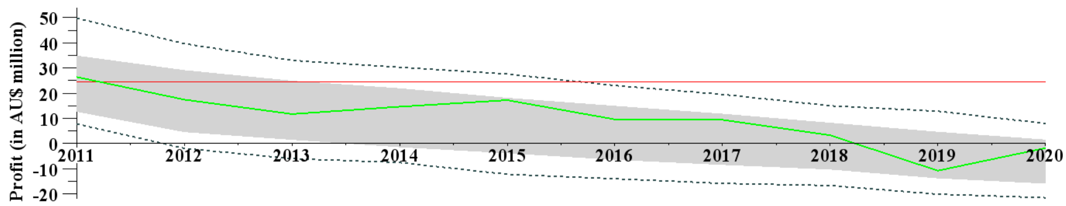


Figure 4.4: Trajectories over 10 years of the spawning stock size indices $S_s(y(t))$ of grooved, brown tiger and blue endeavour prawns with the T_{adapt} allocation strategy and SQ fishing capacity strategy. These outputs are similar under both BC and ML economic scenarios. In each sub-figure the blue line corresponds to the historical spawning stock size indices estimated for the past 25 years before the reference year 2010, the red line represents the historical minimal spawning stock size index by species, the dotted dark lines represent the field of possibilities that includes all of the 1000 simulated trajectories and the grey field includes 95% of these trajectories. The green line corresponds to a randomly selected trajectory among the 1000 trajectories associated to the same set of recruitments and banana biomasses $\omega(.)$ for each sub-plot of figures 4.4 and 4.5.

Under the BC scenario the model predicts sustained positive profit throughout the simulation period (figure 4.5(a)), except for 0.7% of the trajectories for which the annual profit is negative during at least one year of the simulation. Although positive profits are predicted, in 63.7% of trajectories, the annual profit will fall short of its 2010 reference level for at least one year of the simulation. Comparison of figures 4.5(a) and 4.5(b) shows that the economic performance of the fishery will deteriorate substantially under the ML economic scenario. Indeed given the projected decrease in market prices and increase in fuel price, 52.2% of the trajectories will have a negative annual profit for at least one year of the simulation. Moreover, there is 91.4% chance that the annual profit will be negative by the last year of the simulation compared to only 0.1% under the base case scenario. Furthermore there is a 100% chance that the profit will be below its 2010 reference level by 2020 (versus 68.3% probability with a base case scenario). The mean value of the reduction in the NPV of the NPF between the base case and most likely scenario is AU$ 463 million.



(a) $\pi(y(t))$ under a base case scenario BC.



(b) $\pi(y(t))$ under a most likely scenario ML.

Figure 4.5: Trajectories over 10 years of the total annual profits $\pi(y(t))$ with a T_{adapt} allocation strategy and a status quo fishing capacity strategy SQ for a base case scenario BC in (a) and a most likely scenario ML in (b). In each sub-figure the red line corresponds to the annual profit estimated for the reference year 2010, the dotted dark lines delimit the field of possibilities that includes all of the 1000 simulated trajectories and the grey field includes 95% of these trajectories. The green line corresponds to a randomly selected trajectory among the 1000 trajectories associated to the same set of recruitments and banana biomasses $\omega(.)$ for each sub-plot of the figures 4.4 and 4.5.

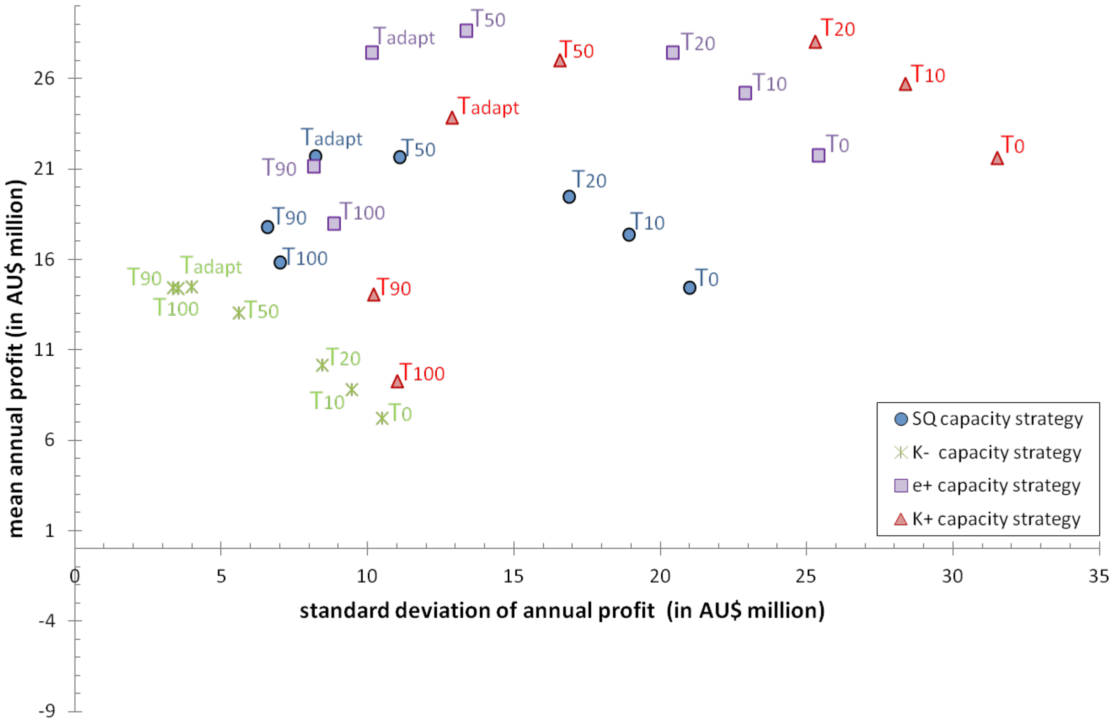
4.3.2 Mean-variance analyses

To study the trade-offs between the mean economic performance of the fishery and the variance of this performance under different economic and fishing management scenarios, mean-variance analyses are conducted.

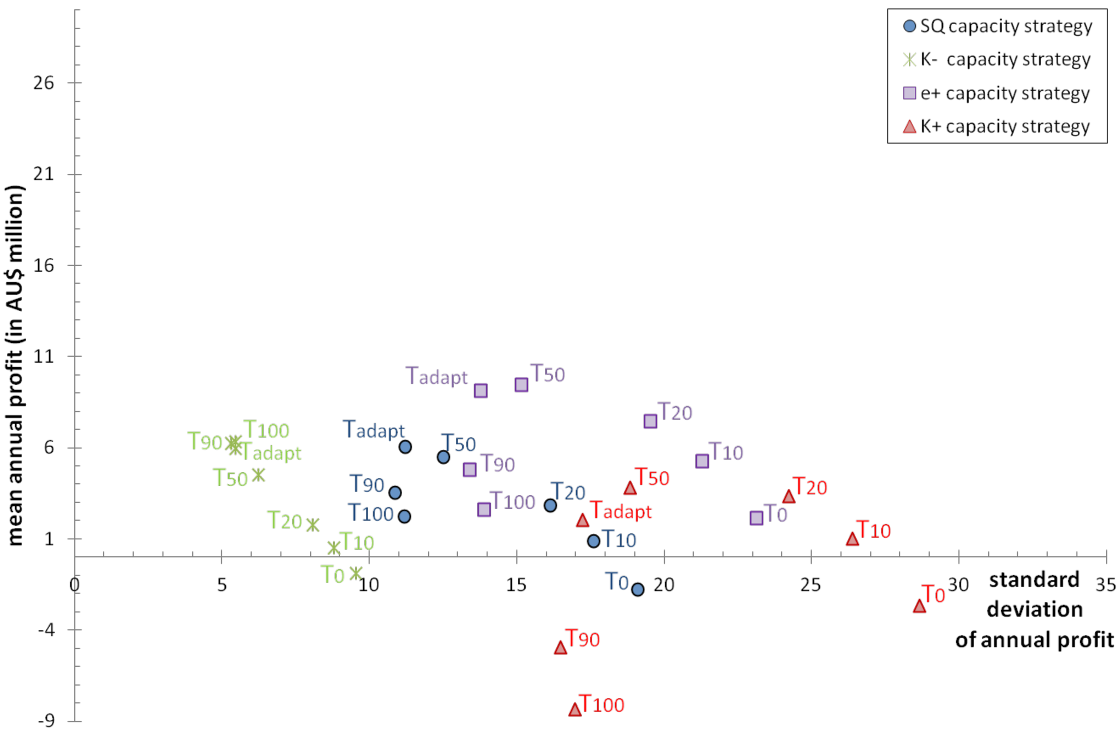
Figure 4.6 represents the average annual profit of the fishery versus its standard deviation under a BC in (a) and a ML economic scenario in (b).

Performance of various effort allocation strategies

Results for the SQ fishing capacity strategy (blue dots in figure 4.6) show that strategies involving higher levels of specialisation (banana or tiger prawn sub-fisheries) are marked with reduced average annual profit under both economic scenarios. The average NPV of the fishery is higher with the balanced T_{50} and the 'adaptive' T_{adapt} effort allocation strategies (AU$ 434 and AU$ 429 million for the base case scenario, and - AU$ 46 and - AU$ 34 million for the most likely scenario). Importantly, the banana specialisation strategies are also associated with higher economic variability, compared to the tiger specialisation strategies, for both economic scenarios.



(a) Base case scenario



(b) Most likely scenario

Figure 4.6: Economic mean-variance analysis: each dot represents the average annual total profit $\pi(y(t))$ over the years and the 1000 trajectories simulated versus the standard deviation associated. In each sub-plot the results are featured by effort allocation strategy (T_{adapt}, T_0, T_{10}, T_{20}, T_{50}, T_{90} and T_{100}) under different fishing capacity strategies. The blue circles correspond to a status quo fishing capacity strategy SQ, the purple cross to an increase in effort per vessel e^+, the red triangles to an increase in the number of vessels K^+ and the green square to a decrease in the number of vessels K^-. Effort allocation and fishing capacity strategies are considered under a base case economic scenario in (a) and under a most likely economic scenario in (b).

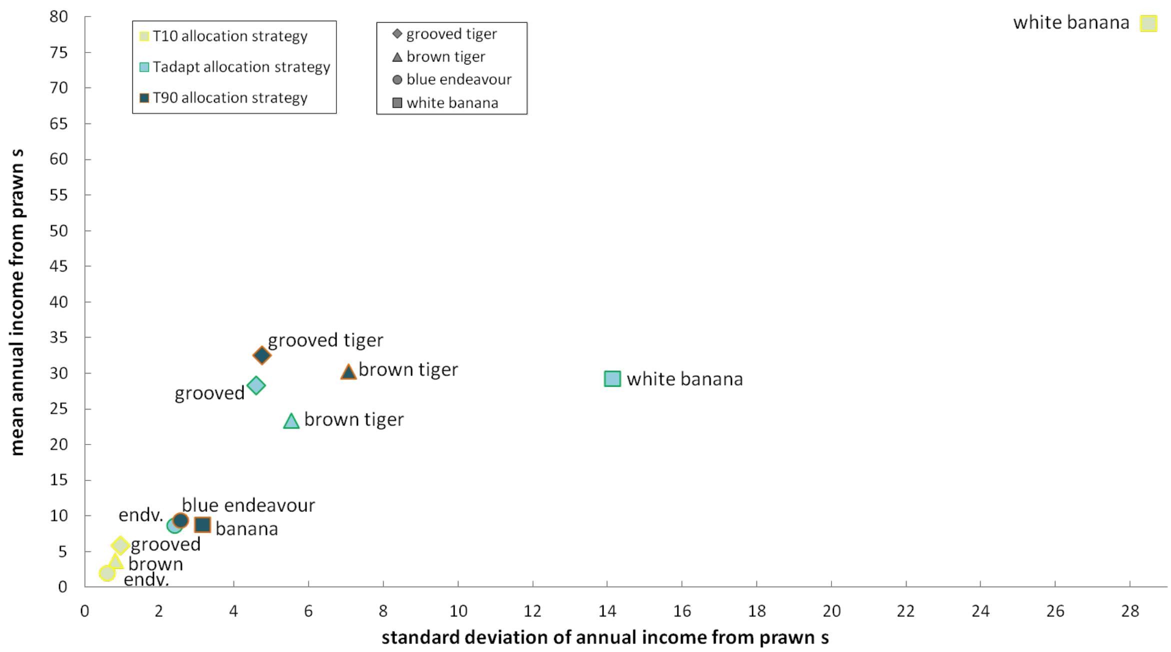


Figure 4.7: Mean-variance analysis of the partial total annual income coming from the four studied species: each dot represents the average over the mean annual income from species *s* (for all years and simulations) versus the standard deviation associated. Results are featured with the effort allocation strategy (T_{adapt}, T_{10} and T_{90}) in colours and the species are represented by different shapes of dots. The blue correspond to T_{adapt}, the red to T_{90} and the green to T_{10} allocation strategy. Whereas the diamonds represent the partial annual income coming from grooved tiger prawn catches, the triangles the ones coming from brown tiger prawn, the circles stand for blue endeavour prawn and the squares for white banana prawn. Effort allocation strategies are considered under a base case BC economic scenario and a status quo SQ fishing capacity strategy.

In Figure 4.7, the mean-variance analysis for the partial annual income coming from each of the four targeted species with the SQ fishing capacity and BC economic scenarios is illustrated for three contrasting effort allocation strategies: T_{10}, T_{adapt} and T_{90}. This figure allows us to distinguish the part of the economic variability resulting from the biological variability of each exploited species, and that for the three contrasted allocation strategies. This figure shows that the difference observed in the economic variability between tiger and banana ‘specialisation’ strategies is mainly due to the biological variability of white banana prawn catches. Biological variabilities of tiger and endeavour prawns play, therefore, only a minor role in the total economic variability. Furthermore, the economic variability attributed to brown tiger prawn catches is greater than that arising from either grooved tiger or blue endeavour prawns for both the ‘adaptive’ and tiger effort allocation strategies.

Performance of various fishing capacity strategies

Comparison of economic performance for effort allocation strategies across alternative fishing capacity strategies (figure 4.6) shows a positive relationship between fishing capacity and economic variability.

Table 4.4: Rank of the effort allocation strategies (in row) according their mean NPV values (1 being the strategy with the highest average NPV) for each level of fishing capacity and economic scenario (in column). Fishing capacity strategies are sorted in increasing order of fishing capacity.

		BC				ML			
		increase in fishing capacity ⟶				increase in fishing capacity ⟶			
	Strategies	K^-	SQ	e^+	K^+	K^-	SQ	e^+	K^+
incr. of pressure on tiger prawns ↓	T_0	7	7	5	5	7	7	7	5
	T_{10}	6	4	4	3	6	6	4	4
	T_{20}	5	3	2	1	5	4	3	2
	T_{50}	4	1	1	2	4	2	1	1
	T_{adapt}	1	2	3	4	3	1	2	3
	T_{90}	2	5	6	6	2	3	5	6
	T_{100}	3	6	7	7	1	5	6	7

The effort allocation strategies according to their mean NPV for each fishing capacity strategy and economic scenario are ranked in Table 4.4, with 1 indicating the most profitable in terms of fishery-wide NPV. Under the BC economic scenario the more fishing capacity increases, the better the banana specialisation strategies perform relative to the tiger prawn specialisation strategies (Table 4.4 and figure 4.6). Moreover, effort allocation strategies that involve higher tiger prawn specialisation perform relatively better, as compared to strategies where effort is directed more towards the banana prawn sub-fishery, when the ML economic scenario is considered. Furthermore, the highest mean NPV among all effort allocation strategies under the BC scenario occurs with a K^+ fishing capacity strategy and the T_{20} effort allocation strategy, while under the ML scenario, it is obtained with a K^- fishing capacity strategy and a T_{100} effort allocation strategy.

4.4 Discussion

4.4.1 The interest of an integrated bio-economic model

The simplified bio-economic model of the NPF presented in this article is based on the synthesis of a complex set of models developed in support of the Management Strategy Evaluation

(MSE) approach to managing this fishery (Dichmont et al., 2006, 2008; Venables et al., 2009). The model allows for the explicit representation of the tiger prawn sub-fishery targeting more predictable species, and the banana prawn sub-fishery targeting less predictable species. Prawns are short-lived species (i.e. 1-2 years life cycle) and their dynamics are expressed at a weekly time-step which allows representation of intra-annual and seasonal biological processes such as spawning and recruitment, and economic processes such as seasonal allocation of fishing effort, as is advocated by Anderson and Seijo (2010). While many features of the original models have been simplified, key aspects of model structure have been maintained where these were considered crucial to the understanding of the bio-economic system under study.

We use this model to compare the bio-economic performances of a range of fishing capacity strategies involving variations in number of vessels and annual effort per vessel, using mean variance analysis. Sensitivity of the results to variations in the economic conditions of the fishery, and to different effort allocation strategies incorporating varying degrees of specialisation is also explored. Based on the simulation results, the adaptation options available to the fishery can therefore be examined.

4.4.2 Trade-off between mean annual levels of profit and their variability

Previous studies contributed to setting the current number of vessels in the fishery to 52 vessels, by establishing this as the number required to achieve Maximum Economic Yield (MEY). Based on our simulations, it appears that the NPF currently operates with an effort allocation strategy allowing the best compromise between mean performance and variability in this performance, as is illustrated by the status quo fishing capacity simulation results for the 'adaptive' effort allocation strategy (where the fishery adapts its effort allocation to white banana prawn biomasses) (figure 4.6(a)). In addition, as illustrated in figure 4.6(a), the simulated fishery could achieve higher levels of average economic performance with this 'adaptive' effort allocation strategy, by adopting a fishing capacity strategy allowing a higher level of effort per vessel or a larger number of vessels. However, this would be obtained at the cost of increased economic risk, as illustrated by the higher inter-annual variability in profits under these strategies. When considering an increase of annual effort per vessel, the fishery could get even higher mean annual profit with a strategy allocating the same amount of effort towards both tiger and banana prawn sub-fisheries, but it would be at cost of about a 37% increase in the variance of simulated profit compared to an 'adaptive' effort allocation strategy. The difference in mean annual profit is even stronger with effort allocation strategies focusing effort more on banana prawns (T_{10}, T_{20} and T_{50}; see figure 4.6(a)) when status quo fishing capacity is compared with increased fishing capacity or annual effort per vessel. While these strategies all lead to average annual profits that are higher than with the status quo fishing capacity strategy and T_{adapt} allocation strategy, they also entail much higher levels of inter-annual variability in profits. On the other hand, if the fleet is targeting mainly tiger prawns (T_{100} and T_{90}), compared to alternative effort allocation strategies both the mean economic performance of the fishery and the inter-annual economic variability will decrease and there will also be a strong negative impact on mean spawning stock levels (as shown in the supplementary data). The difference in economic variability between tiger and banana specialisation allocation strategies derives mainly from differences in the biological variability of white banana prawn stocks, as exhibited in figure 4.7.

The model illustrates an important aspect of managing mixed fisheries for MEY, namely that given the biological variability of some of the target resources, increased average profits may be associated with increased variability in these profits. In cases where industry are risk averse, this may lead to management options with lower levels of performance, but with reduced economic risk, being preferred. Risk aversion of key stakeholders (fishers, industry, and more broadly, society) should therefore be included in the evaluation of management strategies. As Mistiaen and Strand (2000) pointed out, it is widely agreed that fisher's risk preference is a major determinant of their responses to various changes in fishing stock, market, and weather conditions. Therefore, it is important to integrate fisher's risk preference in modelling and analyse their decision-making behaviour (Nguyen and Leung, 2009). Results of the case study considered here, allow identification of an implicit risk aversion measure, which, for a given effort allocation strategy, can be estimated as the difference between the mean annual profits under the current fishing capacity strategy SQ and the fishing capacity strategy which maximises mean annual profits.

The NPF operates under a strong co-management structure. Nearly all of the industry is incorporated into a single company, which is represented in the management decision process. Industry have a direct role in determining annual effort targets (based on bio-economic advice), and was also involved in setting the number of vessels to achieve MEY. Management and industry objectives can be considered therefore relatively aligned (Pascoe et al., 2009). Thus the risk aversion of the fishery, can be estimated by the managers risk aversion which corresponds to difference between the mean annual profits under the current fishing capacity strategy SQ and the increase of alternative vessels strategies. This is approximately AU$ 2.17 million, or 10% of the average annual profits. Managers (or in this case industry) are willing to forgo AU$ 0.47 million in average annual profit to reduce the standard deviation of profit by one million. However, the choice of annual effort per vessel and allocation of fishing effort between prawn sub-fisheries relies mainly on fishers. Therefore, fishers' risk aversion can be estimated by the difference between the mean annual profits under the T_{adapt} allocation strategy with the current effort per vessel SQ and the combination of allocation strategy and level of effort per vessel which maximises mean annual profits (i.e. T_{50} allocation strategy and e^+ capacity strategy). This value is estimated at AU$ 6.97 million, or 32% of the average annual profits. This implies that fishers are therefore willing to forgo AU$ 1.35 million in average annual profit to reduce the standard deviation of profit by one million.

An intermediate adaptation option could increase the effort per vessel from 162 to 196 days at sea, while keeping an 'adaptive' effort allocation strategy with 52 vessels. Indeed this combination gives an AU$ 2.99 million of supplementary profit per million of supplementary standard deviation of profit, which could be considered as a good compromise between increase of the average economic performance and increase of economic variability. Moreover, with this management option, impacts on tiger and endeavour prawn species are reduced compared to tiger prawn specialisation strategies (c.f. supplementary data).

4.4.3 Expected effects of economic scenarios on the biological and economic performance of the fishery, and possible adaptation options

Analyses of economic performance of the fishery under different economic scenarios illustrate the importance of sensitivity analyses to key economic parameters in bio-economic assessments. Whereas it is difficult to predict the future evolution of prices and costs as they are influenced principally by external drivers, scenarios and projections based on the best available knowledge of these drivers show that the fishery is likely to encounter strong economic difficulties, with average annual profit levels expected to be low and even negative in some periods.

The simulation results under the most likely economic scenario (figure 4.6(b)) show that management strategies aiming at increasing the fleet size, given current costs per vessel, would fail to improve fishery performance, in terms of both mean levels of performance, and variability of this performance. Possible adaptation strategies involve maintaining 'adaptive' effort allocation strategies, and either reducing fleet size below its historical level, or maintaining the fleet size but increasing the fishing effort per vessel. The first option provides levels of overall annual profits comparable to those obtained under the status quo fishing capacity strategy, but with reduced variability. Given the reduced number of vessels, individual performance of vessels would remain relatively high under this strategy. The latter option allows an increase in the average economic performance of the fleet, but at the cost of increased variability in performance, or economic risk.

Increase in fuel prices leads to a relatively more important increase in variable costs for the banana prawn sub-fishery, compared to the tiger prawn sub-fishery, as the banana prawn sub-fishery uses a greater amount of fuel per effort unit. Therefore, the most likely economic scenario appears to favour tiger prawn specialisation strategies.

Possible adaptive responses from fishers to the increased economic vulnerability predicted under a most likely economic scenario involve combinations of changes in fishing capacity and effort allocation strategies. Our analyses show that to maximize the mean economic performance, the best combination would likely involve a 'balanced effort allocation' strategy (where the fishery allocates equally its annual effort between tiger and banana prawns) with 52 vessels, but with an increase in the effort per vessel from 162 to 196 days at sea. However, this begs the question as to why the industry did not increase effort when prices were higher and variable costs lower, in which case even greater profits would have been realised. If the objective is to minimize the economic variability of annual profit, a more realistic adaptation option could be to allocate a substantial proportion of the total annual effort to the tiger prawn sub-fishery and reduce the number of vessels.

4.4.4 Perspectives

While the results obtained are specific to the case study considered in this analysis, the methods proposed would apply to any mixed fishery where information allows calibration of a dynamic bio-economic model of fishing across a range of species presenting different levels of natural variability. It is likely that most of the mixed fisheries in the world would be subject to similar trade-offs between average economic performance and variability of this performance from year to year. If this is the case, we argue that the question of variability in returns of a fishery should also be considered when discussing the identification of management strategies aimed at MEY. This would of course pose the question of the degree of risk aversion of key stakeholders, including industry, the fishers, and more broadly, society.

Fisheries management increasingly acknowledges that fish population dynamics are complex and influenced by factors that are usually poorly understood. This is the case with the white banana prawn dynamics. It may be that the conclusions of our analysis would change if the patterns of variability in abundance of banana prawns changed, due for example to changes in the environmental drivers which determine its year-to-year fluctuations in abundance. In particular, rainfall and sea level rise have been identified by Hobday et al. (2008) as key impacts of climate change in the NPF region, which may have an impact on the dynamics of the different species targeted by the NPF, notably on white banana prawn abundance. Climate change projections for rainfall are highly uncertain; rainfall is projected to decrease across parts of northern Australia with some areas showing a slight increase which may have a positive impact on white banana prawn catches (Hobday et al., 2008). Climate change may also have an impact on seagrass beds and mangrove forests, which are important nursery grounds for tiger prawns and banana prawns, respectively (Sands, 2011). Coupling this model with projections derived from models relating climate change to the environmental drivers of prawn abundance could therefore allow a more informed evaluation of the likely trade-offs between mean performance and economic risk in this fishery.

Finally, while our analysis has focused exclusively on the bio-economic trade-offs associated with species with commercial value, another key dimension of mixed fisheries which may also need to be considered in evaluating options is the impacts of effort allocation strategies on bycatch species of low commercial value, as well as on threatened, endangered and protected species and on habitats (Woodhams et al., 2011). Different levels of fishing capacity and alternative effort allocation strategies, impacting differently on the surrounding ecosystem, will potentially lead to different outcomes in terms of the ecological impacts of fishing. This will be the focus of further research using the bio-economic model presented in this article.

4.5 Appendix A. Dynamics details of the bio-economic model

This appendix provides details on estimation and calculation of bio-economic model variables and parameters.

4.5.1 Tiger prawn abundance dynamics

The population dynamics of grooved and brown tiger prawns ($s = 1, 2$, respectively) are based on a sex and size-structured model relying on a weekly time-step and governed by the equation (4.18):

$$\vec{\mathrm{N}}_{s,x}(t+1) = \mathbf{X}_{s,x}\mathbf{Suv}_s(t)\vec{\mathrm{N}}_{s,x}(t) + \alpha_s(t+1)\frac{\vec{R}_s(\tilde{y}(t))}{2}, \qquad s = 1, 2. \tag{4.18}$$

where:

- Time t corresponds to one week,
- $\vec{\mathrm{N}}_{s,x}(t)$ is the vector corresponding to the abundance $\mathrm{N}_{s,x,l}(t)$ of prawns of species s of sex x ($x = ♀$ for female and ♂ for male) in size-class l alive at the start of time t.
- $\mathbf{X}_{s,x}$ is the size-transition matrix. It corresponds to the probability of an animal of species s and sex x in size-class i growing into size-class j during one time-step (i.e. week), and is assumed to be governed by a normal distribution as described in (Punt et al., 2010).
- $\mathbf{Suv}_s(t)$ is the diagonal survival matrix for the species s during the time t as:

$$\mathbf{Suv}_s(t) = \begin{bmatrix} \exp(-Z_{s,1}(t)) & \dots & 0 \\ \vdots & \ddots & \vdots \\ 0 & \dots & \exp(-Z_{s,L}(t)) \end{bmatrix}$$

  with $Z_{s,l}(t)$ the total mortality on animals of species s in size-class l during time t.
- $\alpha_s(t)$ is the fraction of the annual recruitment for the species s that occurs during the time t.

- $\vec{R}_s(\tilde{y}(t))$ is the vector of recruitment[11] of species s by size-class l during the 'biological' year $\tilde{y}(t)$:

$$R_{s,l}(\tilde{y}(t)) = \begin{cases} R_s(\tilde{y}(t)) & \text{if } l = 1, \\ 0 & \text{otherwise.} \end{cases} \tag{4.19}$$

  the size-class $l = 1$ corresponds to animals of 15 mm length. Equation (4.19) implies that recruitment contributes only to the first size-class in the model.

4.5.2 Recruitment estimation

The 'biological' year, $\tilde{y}(t)$, corresponding to time t is defined by:

$$\tilde{y}(t) = \begin{cases} y(t) & \text{for (t modulo 52)} < 40, \\ y(t) - 1 & \text{for (t modulo 52)} \geq 40. \end{cases} \tag{4.20}$$

where $y(t)$ stands for the year y corresponding to the time t.

Equation (4.20) implies that a 'biological' year $\tilde{y}(t)$ ranges from week 40 (roughly the start of October) of year $(y(t) - 1)$ to week 39 (roughly the end of September) of year $y(t)$. Recruitment involves complex biological and environmental processes that vary over time. Due to a large influence on recruitment events to the catch, it is important to take into account this variability. $R_s(\tilde{y}(t) + 1)$ stands for the recruits of species s in the fishery during a 'biological' year $(\tilde{y}(t) + 1)$ and is assumed to be related to the spawning stock size index $\mathrm{S}_s(y(t))$ of the year $y(t)$, according to a Ricker stock-recruitment relationship fitted assuming temporally correlated environmental variability and down-weighting recruitments, as in Punt et al. (2010, 2011), and given by:

$$R_s(\tilde{y}(t) + 1) = \alpha_s^{Rick} \mathrm{S}_s(y(t)) \exp\left(-\beta_s^{Rick} \mathrm{S}_s(y(t))\right) \exp\left(\eta_s(y(t) + 1)\right). \tag{4.21}$$

where α_s^{Rick} and β_s^{Rick} are the parameters of the Ricker stock-recruitment relationship for the species s and $\eta_s(y(t))$ represents the temporally correlated environmental variability term of the year $y(t)$ as:

$$\begin{cases} \eta_s(y(t)) = \rho_s \eta_s(y(t)) + \sqrt{1 - \rho_s^2} \xi_s(y(t)), \\ \xi_s(y(t)1) \rightsquigarrow \mathcal{N}(0, \sigma_s^2). \end{cases} \tag{4.22}$$

[11] the sex-ratio of the recruits is assumed to be 50:50 in the absence of data (Punt et al., 2011)

with ρ_s the environmentally driven temporal correlation in recruitment, and σ_s^2 is the environmental variability in recruitment.

4.5.3 CPUE

Annual average banana catch per unit effort (CPUE) are computed from white banana prawn annual biomass $B_{s=4}(y(t))$, as:

$$\text{CPUE}_4(y(t)) = q_{4,3}B_3(y(t)), \tag{4.23}$$

where $q_{4,3}$ is the catchability of the white banana prawn ($s = 4$) by the banana prawn sub-fishery ($f = 3$). Estimated values of $q_{4,3}$ are given in appendix B table 4.5.

4.6 Appendix B. Bio-economic parameter values

This appendix displays the values of the biological and economic parameters used to calibrate the bio-economic model presented in sections 4.2, 4.5.1 and 4.5.2. Table 4.5 displays the parameters related to the white banana prawn. Tables 4.6 and 4.7 summarize respectively the annual stock dynamics and catchabilities parameter values for the grooved and brown tiger and blue endeavour prawns. Tables 4.9 and 4.10 summarize the values of parameters involved in the profit equation. Table 4.11 exhibits the weekly proportion of tiger prawn sub-fishery effort directed towards grooved and brown tiger prawn fishing strategies used to split the tiger prawn sub-fishery effort into grooved and brown tiger prawn fishing strategies as described in section 4.2.5.

Table 4.5: Estimated parameters related to white banana prawn ($s = 4$ and $f = 3$).

	B_s^- (in thousand tonnes)	B_s^+ (in thousand tonnes)	catchability, $q_{s,f}$
white banana prawn	28.72	125.8	0.0000142

Table 4.6: Stock dynamic parameters by species s

Parameters	Values by species s		
	grooved tiger	brown tiger	blue endeavour
Natural mortality (week^{-1}). M_s	0.045	0.045	0.045
Ricker parameter. α_s^{Rick}	1182.51	1108.81	483.496
Ricker parameter. β_s^{Rick}	0.715945	0.581685	0.761467
Temporal correlation in recruitment. ρ_s	-0.379982	-0.322691	-0.339732
Variance in recruitment. σ_s^2	0.0682356	0.124784	0.0930268
Average mass (in kg/animal). $\tilde{v}_{s.sx}$			17.63 23.24

Table 4.7: Estimated values of catchabilities $q_{s,f}$ by species s and by tiger prawn fishing strategies $f = 1, 2$ for a fishing power of the fishery as in 2010.

prawn species	Tiger prawn sub-fishery	
	grooved tiger prawn fishing strategy $f = 1$	brown tiger prawn fishing strategy $f = 2$
grooved tiger	0.0001219	0.0000152
brown tiger	0.0000111	0.0001219
blue endeavour	0.0001149	0.0002839

Table 4.8: Weekly stock dynamic parameters by species s.

weeks	Relative weekly recruitment, $\alpha_s(t)$			Relative weekly spawning, $\beta_s(t)$			Relative weekly availability, $A_s(t)$		
	P.s.	*P.e.*	*M.e.*	*P.s.*	*P.e.*	*M.e.*	*P.s.*	*P.e.*	*M.e.*
1	0.0623	0.05	0.0217	0.0056	0.0065	0.0153	1	1	0.222
2	0.0724	0.0492	0.0234	0.0056	0.0065	0.0153	1	1	0.25
3	0.0753	0.046	0.0251	0.0056	0.0065	0.0153	1	1	0.278
4	0.0683	0.0395	0.0272	0.0056	0.0065	0.0153	1	1	0.285
5	0.0613	0.0331	0.0294	0.005	0.0067	0.0153	1	1	0.292
6	0.0544	0.0267	0.0315	0.0046	0.0068	0.0065	1	1	0.304
7	0.0475	0.0209	0.0336	0.0046	0.0068	0.0065	1	1	0.306
8	0.041	0.0191	0.0286	0.0046	0.0068	0.0065	1	1	0.333
9	0.0347	0.018	0.0212	0.0052	0.0129	0.0065	1	1	0.361
10	0.0283	0.0169	0.0187	0.0055	0.0175	0.0142	1	1	0.389
11	0.0225	0.016	0.0137	0.0055	0.0175	0.0142	1	1	0.417
12	0.0196	0.0166	0.0148	0.0055	0.0175	0.0142	1	1	0.406
13	0.0172	0.0173	0.0158	0.0058	0.018	0.0142	1	1	0.394
14	0.0148	0.0181	0.0168	0.0077	0.0213	0.0176	1	1	0.383
15	0.0124	0.0189	0.0179	0.0077	0.0213	0.0176	1	1	0.372
16	0.0106	0.0179	0.0189	0.0077	0.0213	0.0176	0.99	1	0.361
17	0.0093	0.0156	0.0177	0.0077	0.0213	0.0176	0.98	1	0.383
18	0.0079	0.0133	0.0165	0.0104	0.0294	0.0176	0.96	1	0.406
19	0.0066	0.011	0.0153	0.0108	0.0307	0.0214	0.95	1	0.428
20	0.0056	0.009	0.0141	0.0108	0.0307	0.0214	0.93	0.99	0.444

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	$\alpha_s(t)$			$\beta_s(t)$			$A_s(t)$		
weeks	*P.s.*	*P.e.*	*M.e.*	*P.s.*	*P.e.*	*M.e.*	*P.s.*	*P.e.*	*M.e.*
21	0.0053	0.0079	0.0148	0.0108	0.0307	0.0214	0.88	0.96	0.489
22	0.005	0.0068	0.0154	0.01	0.0255	0.0214	0.84	0.92	0.533
23	0.0047	0.0057	0.016	0.0088	0.0186	0.022	0.8	0.89	0.578
24	0.0045	0.0047	0.0166	0.0088	0.0186	0.022	0.75	0.85	0.622
25	0.0044	0.0047	0.0173	0.0088	0.0186	0.022	0.7	0.82	0.667
26	0.0044	0.005	0.0185	0.0104	0.0191	0.022	0.65	0.78	0.722
27	0.0044	0.0054	0.0198	0.0197	0.0218	0.026	0.59	0.74	0.778
28	0.0043	0.0057	0.021	0.0197	0.0218	0.026	0.53	0.7	0.833
29	0.0044	0.0056	0.0223	0.0197	0.0218	0.026	0.54	0.69	0.889
30	0.0046	0.0053	0.0236	0.0197	0.0218	0.026	0.59	0.72	0.917
31	0.0048	0.005	0.0249	0.0317	0.0301	0.026	0.64	0.74	0.944
32	0.0049	0.0047	0.0262	0.0365	0.0334	0.0305	0.69	0.77	0.972
33	0.0051	0.0044	0.0274	0.0365	0.0334	0.0305	0.74	0.79	1
34	0.0052	0.0042	0.0277	0.0365	0.0334	0.0305	0.8	0.82	0.994
35	0.0052	0.004	0.028	0.0369	0.0282	0.0305	0.86	0.84	0.989
36	0.0053	0.0038	0.0282	0.0379	0.0153	0.0287	0.92	0.87	0.983
37	0.0054	0.0036	0.0285	0.0379	0.0153	0.0287	0.98	0.89	0.978
38	0.0053	0.004	0.0288	0.0379	0.0153	0.0287	1	0.92	0.972
39	0.005	0.0047	0.0233	0.0379	0.0153	0.0287	1	0.94	0.939
40	0.0048	0.0054	0.0179	0.0491	0.0252	0.0199	1	0.96	0.906
41	0.0046	0.0061	0.0125	0.0491	0.0252	0.0199	1	0.99	0.872
42	0.0051	0.0098	0.007	0.0491	0.0252	0.0199	1	1	0.833
43	0.0065	0.0177	0.0072	0.0491	0.0252	0.0199	1	1	0.736
44	0.0078	0.0256	0.0074	0.0393	0.0212	0.0199	1	1	0.639
45	0.0092	0.0334	0.0076	0.032	0.0183	0.0185	1	1	0.542
46	0.0111	0.0406	0.0078	0.032	0.0183	0.0185	1	1	0.444
47	0.0159	0.0438	0.0094	0.032	0.0183	0.0185	1	1	0.394
48	0.0212	0.0464	0.0111	0.0261	0.017	0.0185	1	1	0.344
49	0.0265	0.0489	0.0128	0.0112	0.014	0.0097	1	1	0.294
50	0.0319	0.0514	0.0145	0.0112	0.014	0.0097	1	1	0.244
51	0.0405	0.0517	0.0161	0.0112	0.014	0.0097	1	1	0.194
52	0.0507	0.0509	0.0184	0.0112	0.014	0.0097	1	1	0.208

Table 4.9: Prawn prices $p_s(2010)$ (AU$ per kilogramme) by species group and size-class in 2010.

Species group	All sizes	<40 mm	40-45 mm	45-50 mm	50-55 mm	>55 mm
Tiger, $p_{tig,l}$	19.05	15.30	19.91	20.83	27.19	26.83
Endeavour, $p_{s=3}$	9.64					
Banana, $p_{s=4}$	9.5					

Table 4.10: Economic parameters.

(a) Variable costs		
	sub-fishery	
Parameters	tiger ($f = 1, 2$)	banana ($f = 3$)
Unit Cost of repairs and maintenance, c_f^K	332 (AU$/day)	529 (AU$/day)
Base unit cost of fuel and grease, $c_f^F(2010)$	1815 (AU$/day)	2236 (AU$/day)
Share cost of labor, c^L	0.24	0.24
Cost of packaging and gear maintenance, c^M	0.92 (AU$/kg)	0.92 (AU$/kg)

(b) Fixed costs and rates	
Parameters	Value
Annual vessel costs, W_v	296,847 (AU$/vessel)
Opportunity cost of capital, o	0.05
Economic depreciation rate, ϱ	0.037
Average value of capital, ψ_v	1,135,693 (AU$/vessel)

(c) NPF fishery status in 2010	
Variable	Value
Number of vessels, K(2010)	52
Annual average effort (days/vessel), $e(2010)$	162

Table 4.11: Pattern of weekly effort by tiger prawn fishing strategy (i.e. grooved or brown) set to 0 for closed weeks (predicted for the year 2010).

	Tiger prawn fishing strategies effort pattern	
weeks	proportion directed to grooved tiger prawn, $\Upsilon^{strat}(t)$	proportion directed to brown tiger prawn, $(1 - \Upsilon^{strat}(t))$
1	0	0
2	0	0
3	0	0
4	0	0
5	0	0
6	0	0
7	0	0
8	0	0
9	0	0
10	0	0
11	0	0
12	0	0
13	0	0
14	0.55052002	0.44947998
15	0.44202646	0.55797354
16	0.54208048	0.45791952
17	0.37494679	0.62505321
18	0.48131314	0.51868686
19	0.47449422	0.52550578
20	0.50102323	0.49897677
21	0.4236849	0.5763151

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weeks	proportion of effort towards grooved tiger prawn, $\Upsilon^{strat}(t)$	proportion of effort towards brown tiger prawn, $(1-\Upsilon^{strat}(t))$
22	0.46343995	0.53656005
23	0.46358818	0.53641182
24	0	0
25	0	0
26	0	0
27	0	0
28	0	0
29	0	0
30	0	0
31	0.14321391	0.85678609
32	0.17552077	0.82447923
33	0.21198361	0.78801639
34	0.29388628	0.70611372
35	0.45558605	0.54441395
36	0.57216275	0.42783725
37	0.67915149	0.32084851
38	0.73330751	0.26669249
39	0.77412768	0.22587232
40	0.7814252	0.2185748
41	0.82888647	0.17111353
42	0.80903904	0.19096096
43	0.82492339	0.17507661
44	0.83268046	0.16731954
45	0.83485189	0.16514811
46	0.80818265	0.19181735
47	0.79468166	0.20531834
48	0.72204043	0.27795957
49	0	0
50	0	0
51	0	0
52	0	0

4.7 Appendix C. Statistical analyses

This appendix displays the outputs of statistical analyses used to calibrate the bio-economic model and scenario projections. Figure 4.8 represents the linear regression used for the projection of the fuel prices and figure 4.9 the one used in the effort allocation model described in section 4.2.5

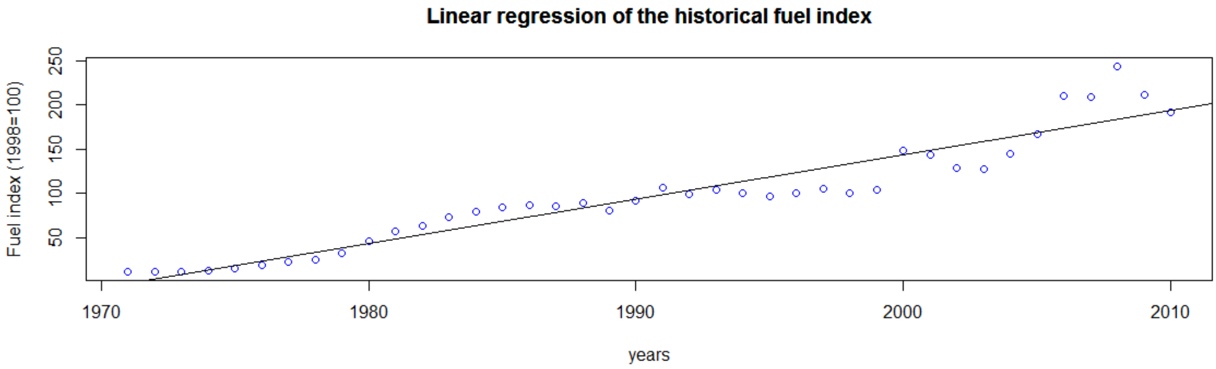


Figure 4.8: Linear regression of fuel price index relying on historical data from 1971 to 2010. $R^2 = 0.9033$ and Pvalue= $2.2 * 10^{-16}$ (<0.05, significant). The slope of the regression line is 4.9831 (meaning an increase of 5% per year). Data source: ABARES (2010).

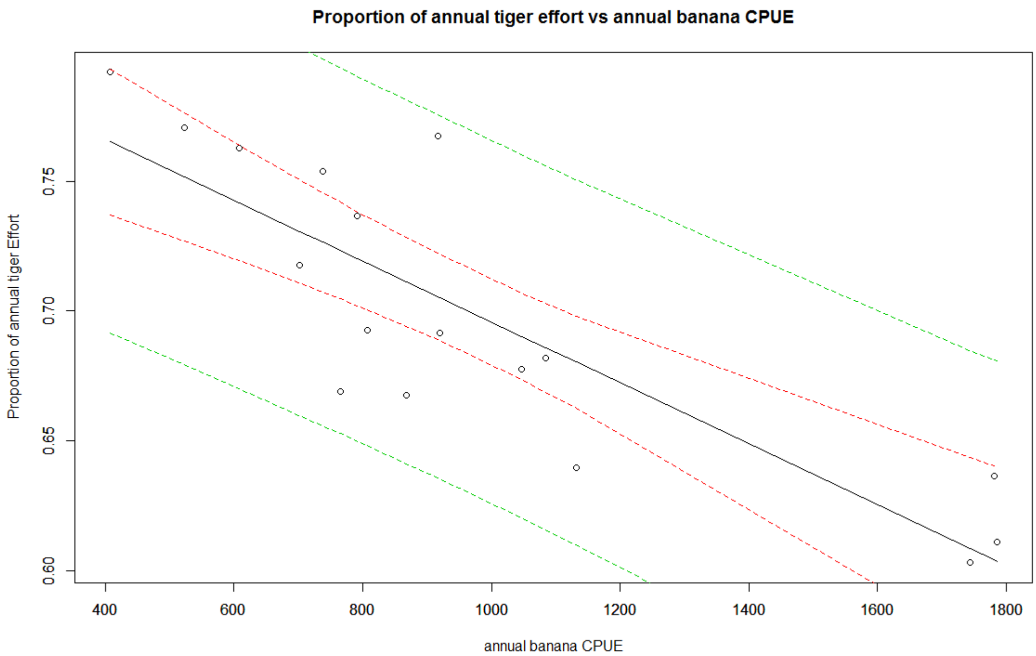


Figure 4.9: Linear regression ($\propto_{1+2}(y) = a\text{CPUE}_{s=4}(y) + b$) between the average annual banana catch per unit effort, $\text{CPUE}_{s=4}$ and the annual proportion of tiger prawn sub-fishery effort, $\propto_{1+2}$. Model relies on historical catches and effort data from 1994 to 2010. $R^2 = 0.7016$ and Pvalue= $1.657 * 10^{-5}$ (<0.05, significant). $a = -1.172 * 10^{-4}$ and $b = 0.813$.

4.8 Appendix D. Algorithm of weekly tiger effort allocation.

This appendix is a complement of the step 2 of effort allocation model described in section 4.2.5. Figure 4.10 described the algorithm used to allocate the annual effort of the tiger prawn sub-fishery to weeks according to the annual proportion of tiger effort associated.

The probability distribution on which depend the random selection of a year between 1994 et 2010 is such as the probability of selecting a historical year y_H is inversely proportional to the normalised value of the difference between the annual proportion of tiger prawn effort of year $y(t)$ and the annual proportion of tiger prawn effort of the year y_H. The historical weekly pattern of tiger prawn sub-fishery effort corresponding to the historical year y_H is then used to allocate the annual tiger prawn sub-fishery effort of the year $y(t)$ to weeks.

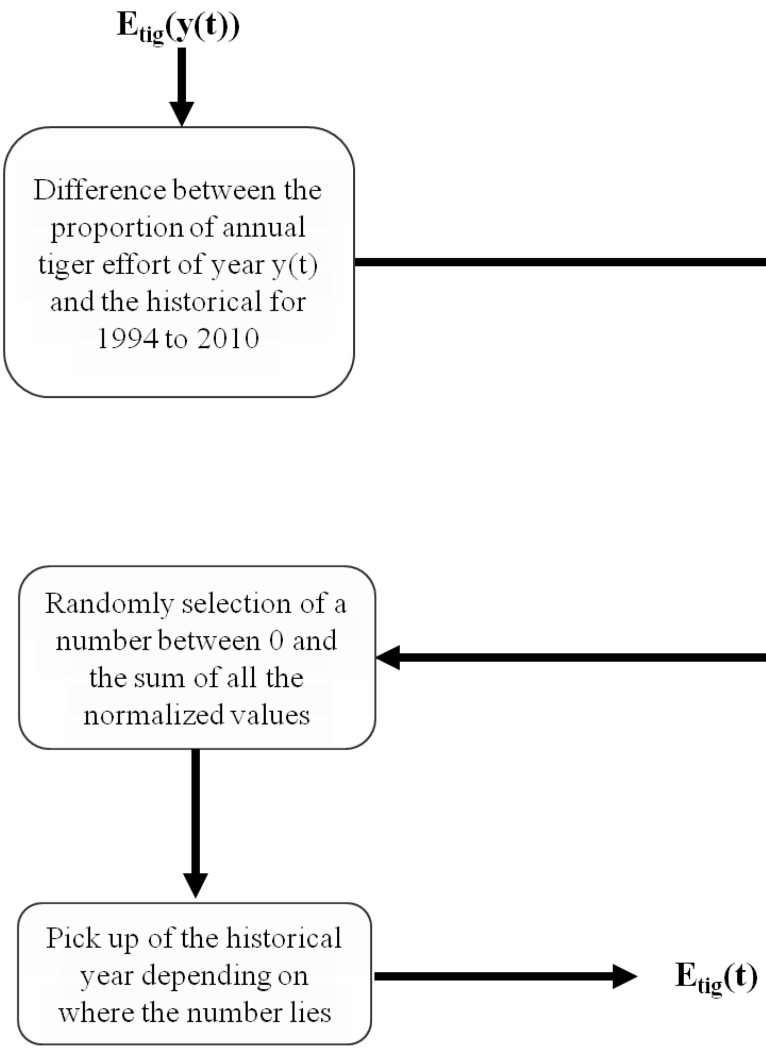


Figure 4.10: Flowchart of the algorithm used to determined the tiger prawn sub-fishery weekly effort pattern for the year $y(t)$.

4.9 Appendix E. To go further: Mean-variance analyses of S

Biological performance of the fishery is studied through the spawning stock size indices for grooved and brown tiger and blue endeavour prawns.

To study the trade-offs between the mean biological performance of the fishery and the variance of this performance under different economic assumptions and fishing management options, mean-variance analyses are setting up. Figures 4.11 shows the mean-variance analyses of the biological indicators, i.e. spawning stock size indices, of the NPF with seven effort allocation strategies and four fishing capacity strategies[12].

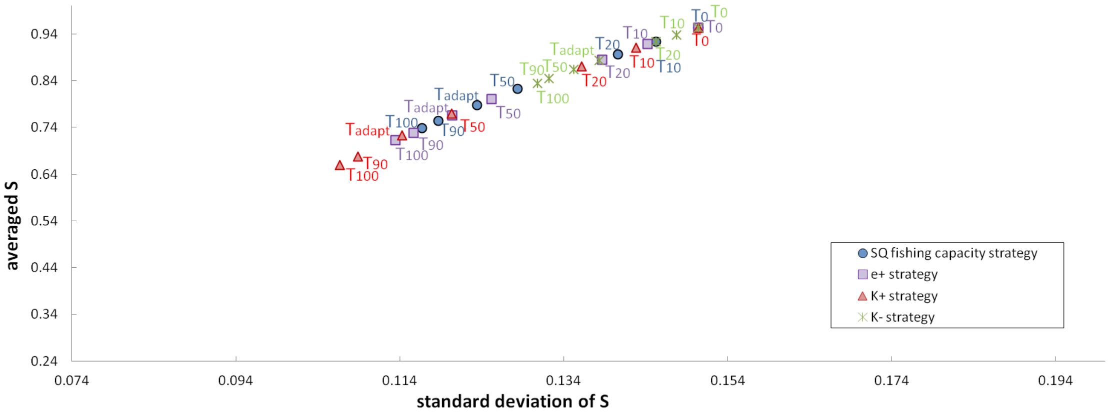
4.9.1 Performance of various effort allocation strategies

Results focusing on the SQ fishing capacity strategy outputs (blue dots in figure 4.11) show that the banana specialisation effort allocation strategies (T_0, T_{10} and T_{20}) resulting in a decrease in fishing pressure on species targeted by the tiger prawn sub-fishery, leads to the highest mean spawning stock size indices for each of the three species (figures 4.11(a) to 4.11(c)). This effect increases with the degree of banana prawn specialisation. Tiger prawn specialisation effort allocation strategies (T_{90} and T_{100}) have strong effects on the three species, greater tiger prawn specialisation entailing decreases in mean spawning stock size indices. Moreover, biological variability for grooved tiger prawn and blue endeavour prawn increases with the degree of banana prawn specialisation, while for brown tiger prawn it increases with the degree of tiger prawn specialisation.

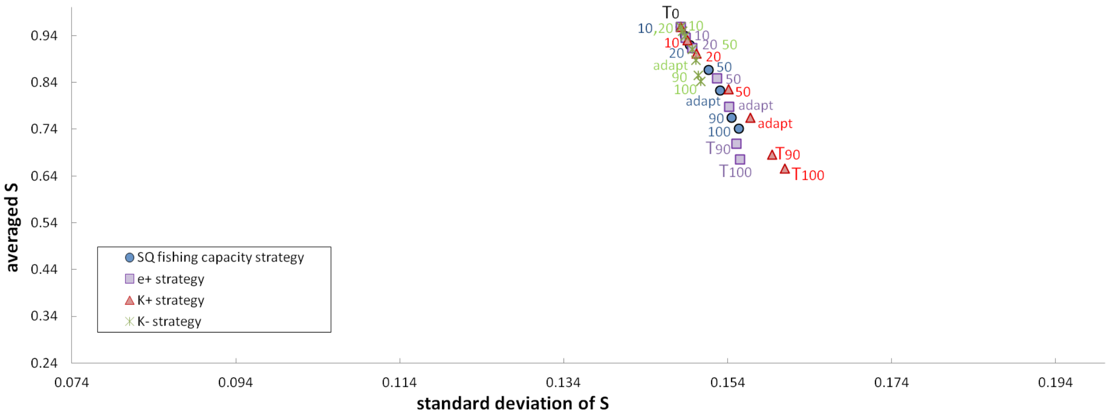
4.9.2 Performance of various fishing capacity strategies

Comparison of biological performance across the four fishing capacity strategies (figures 4.11 and 4.11(c)) shows that higher levels of fishing capacity decreases the spawning stock sizes indices. Biological variability of grooved tiger prawn and blue endeavour prawn also decreases when fishing capacity increases. However, variability of the brown tiger prawn slightly increases with fishing pressure (i.e. with increases in fishing capacity and tiger prawn specialisation strategies).

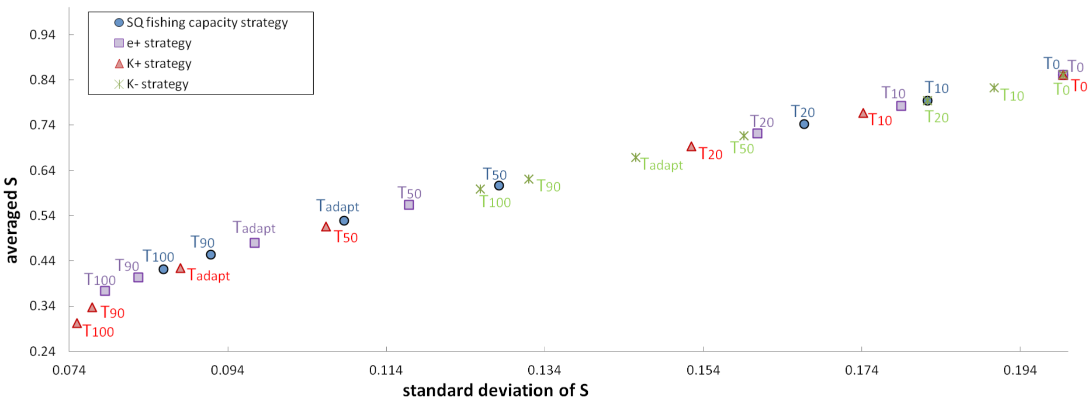
[12]Outputs under both base case and most likely economic scenarios are similar.



(a) Grooved tiger prawn, s = 1



(b) Brown tiger prawn, s = 2



(c) Blue endeavour prawn, s = 3

Figure 4.11: Biological mean-variance analysis of grooved tiger prawn in (a), brown tiger prawn in (b) and blue endeavour prawn in (c). Each dot represents the mean annual spawning stock size index $S_s(y(t))$ of species s versus the standard deviation associated. Results are presented by effort allocation strategy (T_{adapt}, T_0, T_{10}, T_{20}, T_{50}, T_{90} and T_{100}) under different capacity strategies. The blue circles correspond to a status quo (SQ), the purple cross to an increase in effort per vessel (e^+), the red triangles to an increase in the number of vessels (K^+) and the green square to a decrease in the number of vessels (K^-) fishing capacity strategy.

4.9.3 Discussion around mean-variance analyses of S

Analyses of alternative effort allocation strategies with various fishing capacity strategies shows that tiger prawn specialisation strategies have strong negative impact on mean spawning stock levels, while impacts on tiger and endeavour prawn species are reduced with banana prawn specialisation strategies.

Moreover, the biological variability determining economic risk in the fishery may arise from two different causes. On one hand, variability for grooved tiger and blue endeavour prawns increases with the degree of banana specialisation and the decrease in fishing capacity. This can be explained by the fact that with decreasing fishing pressure, and increases in stock biomasses, fluctuations due to stock-recruitment relationships become stronger. On the other hand, biological variability for brown tiger prawn slightly increases with fishing pressure (i.e. with the degree of tiger specialisation and level of fishing capacity). Brown tiger prawn is the species characterized by a decline in its spawning stock size over the projection. Therefore, above a certain level of fishing mortality, increases in fishing pressure tend to increase stock variability. This is because at small stock sizes, stock biomass may become strongly dependent on recruitment only. It could be a way to assess if a species is in danger of overexploitation or not. Stocks with small spawning stock sizes combined with high variability could be considered as not sustainably exploited.

Chapter 5. Co-viability analysis in the Northern Prawn Fishery

This chapter presents a stochastic co-viability analysis applied to the NPF and integrating a set of management objectives reflecting current issues in the fishery. This set includes biological, economic and biodiversity conservation management objectives. To assess the impacts of trawling on the broader biodiversity, an ecological dimension is integrated to the model presented in chapter 4. Trade-offs between biological, economic and biodiversity conservation management objective constraints in highly uncertain context are investigated, and viable management strategies are identified.

5.1 Introduction

Marine fisheries management is characterised by multiple, often conflicting objectives (Crutchfield, 1973; Charles, 1989), including ecological, economic and social viewpoints. There is growing evidence that fishing activities cause physical damage to habitats and affect not only the exploited stocks, but also populations of non-targeted species (Hall and Mainprize, 2005) because they use poorly selective gears, inducing catches of non-targeted fishes (i.e. by-catch and by-product) or unwanted length grades of the targeted species. By-catches from such fisheries consist mostly of small fish with no value to an industrial fishery. Most by-catch species are discarded and returned to the water with high mortality rate (Alverson et al., 1994). Discards represent a significant proportion of global marine catches and are generally considered to constitute waste, or suboptimal use of fishery resources (Kelleher, 2005). As a result, after the sustainability of the stocks themselves, the management and mitigation of by-catch is one of the most pressing issues facing the commercial fishing industry worldwide (Hall and Mainprize, 2005).

Demersal trawling, such as prawn trawling, can be particularly damaging to non-targeted species and habitats. Trawl nets used to catch prawns have small mesh and are towed along a biologically-diverse seabed. This results in large quantities of discarded by-catch, including impacts on endangered or vulnerable and often charismatic species, including turtles, sharks, rays, sea snakes, sawfish and seahorses. Alverson et al. (1994) estimated that around one-third of the world's discards are associated with prawn trawl fishing and Kelleher (2005) estimated that on average 62.3% of total prawn trawl catch in weight is discarded.

The Northern Prawn Fishery (NPF), located off Australia's northern coast and established in the late 1960s, is a multi-species trawl fishery which harvests several high-value prawn species, each with different dynamics. The fishery derives its revenue from an unpredictable naturally fluctuating resource, the white banana prawn (*Penaeus merguiensis*), and a more predictable resource comprising two tiger prawns species (grooved tiger prawn, *Penaeus semisulcatus* and brown tiger prawn, *Penaeus esculentus*). These three species account for 95% of the total annual landed catch value of the fishery (ABARES, 2010). The fishery operates over two 'seasons' spanning the period April to November with a mid-season closure of variable length from June to August. Seasonal closures are in place to protect small prawns (closure from December to March), as well as spawning individuals (mid-season closure) (AFMA and CSIRO, 2012). The fishery effectively consists of two sub-fisheries that are (to a large degree) spatially and temporally separate. The 'banana prawn sub-fishery' is a single species fishery based on the white banana prawn, while the 'tiger prawn sub-fishery' is a mixed species fishery targeting grooved and brown tiger prawns, as well as blue endeavour prawns (*Metapenaeus endeavouri*) which are caught as by product (Woodhams et al., 2011). The banana prawn sub-fishery operates mostly during the first season. The fleet then switches during the second season to the tiger prawn sub-fishery, for which catches per unit effort are lower than for white banana prawns, but less variable. However, if banana prawns are still available in large enough numbers, some vessels will continue to target them. Two different fishing strategies can also be identified within the tiger prawn sub-fishery, one associated with catching grooved tiger prawns (hereafter called the 'grooved tiger prawn fishing strategy') and the other associated with catching brown tiger prawns (hereafter called the 'brown tiger prawn fishing strategy'). Both tiger prawn fishing strategies result in by-catch of tiger and endeavour prawn species.

Environmental issues within the NPF include a high proportion of by-catch, interactions with protected species and potential impact of trawling on benthic communities (Woodhams et al., 2011). By-catch in the NPF consist of small fish, invertebrates, sponges, other megabenthos, rays, sawfish, sharks, sea snakes and turtles (Stobutzki et al., 2001). Many of these species are dead when discarded, or have a low survival rate (Hill and Wassenberg, 2000). The percentage of the by-catch in the total catches has been estimated to range between 89 and 95% depending on the fishing ground (Pender et al., 1992), which is higher than the average percentage of discard among prawn trawl fisheries worldwide (c.f. Kelleher, 2005). Demonstrating ecological sustainability is a legislative requirement for an increasing number of fisheries worldwide, particularly demersal trawl fisheries such as the NPF (Griffiths et al., 2006). The Australian Fisheries Management Act 1991 and the Environment Protection and Biodiversity Conservation Act 1999 require that negative effects on endangered species are avoided, catches of non-targeted species are reduced to a minimum, and the long-term sustainability of by-catch and by-product populations is demonstrated. Management of the NPF is aimed at achieving maximum economic yield (MEY), which implies both stock conservation and economic performance objectives. These must thus be balanced with the objective of limiting the impacts of the fishery on the broader ecosystem. The certification for sustainable fishing practices by the Marine Stewardship Council (MSC) in November 2012 acknowledged the efforts undertaken by the NPF to limit its impacts on ecosystem. The MSC is an international non-profit organisation set up to promote solutions to the problem of overfishing. The certification and eco-labelling program for wild-capture fisheries from MSC is consistent with the FAO "Guidelines for the Eco-labelling of Fish and Fishery Products from Marine Capture Fisheries" (FAO, 2009) which require that credible fishery certification and eco-labelling schemes include: (i) third-party fishery assessment utilising scientific evidence; (ii) transparent processes with built-in stakeholder consultation and objection procedures; and (iii) standards based on the sustainability of target species, ecosystems and management practices.

Few fisheries jurisdictions have adopted harvest control rules which explicitly account for multiple biological, ecological, economic, social and political objectives. In this context, viability modelling has been presented by several authors (Bene et al., 2001; Cury et al., 2005; Eisenack et al., 2006; Doyen et al., 2012; Péreau et al., 2012) as a potentially relevant bio- economic modelling framework. Viability theory - introduced mathematically by Aubin (1990) - aims at identifying decision rules such that a set of constraints, representing various objectives, is respected at any time. It can be useful in a multi-criteria context as this approach identifies a domain of possibilities, and trade-offs between potentially conflicting objectives or constraints (Baumgärtner and Quaas, 2009). It has also been recognized that wise use of fish resources over time should incorporate the inherent risk and uncertainty of fishery systems (Garcia, 1996; Hilborn and Peterman, 1996). By combining biological, economic and ecological goals from stochastic simulation models, the stochastic co-viability approach (Baumgärtner and Quaas, 2009; De Lara and Martinet, 2009; Doyen and De Lara, 2010), can be used to address important issues of vulnerability, risk, safety and precaution, and to determine the ability of a particular resource system to achieve specified sustainability objectives.

The main objective of this chapter is to propose a formal modelling approach allowing to assess trade-offs between biological, economic and biodiversity conservation management objectives within the NPF and identify viable management strategies. This is done by (i) applying a stochastic co-viability framework of analysis as proposed in Doyen et al. (2012) and Gourguet et al. (2013)

to the simplified bio-economic model of the NPF presented in the previous chapter; (ii) including in this CVA assessment a formal way of representing biodiversity conservation constraints; and (iii) based on this, assessing the trade-offs which the fishery may be facing with respect to alternative strategies in setting the fleet capacity and allocation effort (between tiger and banana prawn sub-fisheries) levels.

5.2 Material and Methods

5.2.1 A multi-species, multi fishing strategies, stochastic and dynamic bio-economic model

This study is based on the bio-economic model presented in chapter 4 which allows for the explicit modelling of the banana and tiger prawn sub-fisheries which target respectively an unpredictable naturally fluctuating resource and a more predictable one.

5.2.2 Biological component

Tiger and endeavour prawn dynamics

The bio-economic model includes explicit population dynamics of grooved and brown tiger and blue endeavour prawns based on a multi-species weekly time-step, sex-structured population model as described in chapter 4. Abundance dynamics of grooved ($s = 1$) and brown ($s = 2$) tiger prawns are based on size-structured models and the dynamics of blue endeavour prawn ($s = 3$) is based on an aggregated population model as expressed by:

$$\begin{cases} \vec{\mathrm{N}}_s(t+1) = f\Big(t, \vec{\mathrm{N}}_s(t), \vec{F}_s(t)\Big), & s = 1,2 \\ \mathrm{N}_3(t+1) = g\Big(t, \mathrm{N}_3(t), F_3(t)\Big). \end{cases} \tag{5.1}$$

where $\vec{\mathrm{N}}_s(t)$ is the vector of abundance $\mathrm{N}_{s,x,l}(t)$ of prawns of species $s = 1,2$ of sex x (with $x = ♂$ for male and ♀ for female) in size-class l (1-mm size-classes between lengths of 15 to 55 mm) alive at the start of time t which corresponds to one time step (i.e. one week), and $\mathrm{N}_{3,x}(t)$ is the abundance of blue endeavour prawn of sex x at time t. $\vec{F}_s(t)$ is the vector of fishing mortality $F_{s,l}(t)$ of animals of species $s = 1,2$ and size-class l, and $F_3(t)$ the fishing mortality of blue endeavour prawn at time t (details on fishing mortalities are given in section 4.2.2 of chapter 4). The dynamic functions f

and g accounts for the species recruitment and mortality mechanisms as detailed respectively in sections 4.5.1 and 4.2.1 of chapter 4.

Recruits are assumed to be related to the spawning stock size index S_s for species $s = 1, 2, 3$ according to a Ricker stock-recruitment relationship fitted assuming temporally correlated environmental variability and down-weighting recruitments, as described in chapter 4 section 4.5.2.

The annual spawning stock size indices of the grooved and brown tiger and blue endeavour prawns for the year $y(t)$ are given by:

$$\mathrm{S}_s(y(t)) = \frac{1}{52} \sum_{t=52\left(y(t)-1\right)+1}^{52y(t)} \beta_s(t) \sum_l \gamma_{s,l} \frac{1-\exp\left(-Z_{s,l}(t)\right)}{Z_{s,l}(t)} \mathrm{N}_{s,♀,l}(t), \qquad s = 1, 2. \tag{5.2}$$

$$\mathrm{S}_3(y(t)) = \frac{1}{52} \sum_{t=52\left(y(t)-1\right)+1}^{52y(t)} \beta_3(t) \frac{1-\exp\left(-Z_3(t)\right)}{Z_3(t)} \mathrm{N}_{3,♀}(t). \tag{5.3}$$

where $y(t)$ is the year corresponding to the time t, $\beta_s(t)$ measures the relative amount of spawning of species s during the time t, and $\gamma_{s,l}$ corresponds to the proportion of females of species s in size-class l that are mature. $Z_{s,l}(t)$ is the total mortality on animals of species s in size-class l during time t and is defined by:

$$Z_{s,l}(t) = M_s + F_{s,l}(t). \tag{5.4}$$

with M_s the natural mortality of animals of species s.

White banana prawn: an uncertain resource

Abundance of white banana prawns (species $s = 4$) appears to be more heavily influenced by the environment than by fishing pressure (Die and Ellis, 1999; Venables et al., 2011) and its year to year availability is highly variable. More specifically, stocks are strongly influenced by weather patterns, generally peaking in years in which there has been high rainfall. In the present study, white banana prawn annual biomass is modelled as a uniform i.i.d. random variable:

$$B_4(y(t)) \rightsquigarrow \mathcal{U}(B_4^-, B_4^+), \tag{5.5}$$

with $B_4(y(t))$ the stochastic biomass of white banana prawn for the year $y(t)$, and B_4^- and B_4^+ the uniform law bounds given in appendix 4.6 of chapter 4.

Sea snakes: impacted species

Assessing the performance of the NPF also requires that its impacts on marine biodiversity be considered. Dichmont et al. (2008) studied the effects of trawling in the NPF on benthic communities using a benthic impact model. The spatially explicit model estimates the primary effects of repeated trawling on the biomass of benthic organisms, but is focused on the tiger prawn sub-fishery. Their study demonstrates that effects of trawling on benthos are dependent on the aggregation of effort, suggesting that global effects are very small compared to local effects. Thus, the effects of trawling on benthos communities would only adequately capture the biodiversity impacts of trawling in a spatial analysis.

NPF operations also interact with several groups of threatened, endangered and protected (TEP) species including sea snakes, turtles, elasmobranchs (as sawfishes, sharks and ray), syngnathids (seahorses and pipe fishes) (AFMA, 2012). Interactions with these TEP species are required to be recorded on catch and effort logsheets and by crew member observers (CMO) and scientific observers. Reported interactions with sea snakes are generally an order of magnitude higher than reported interactions with other species (e.g., 4.1 sea snake interactions per day versus 0.056 turtle, 0.021 syngnathid and 0.6 sawfish interactions per day as reported by scientific observers during the tiger prawn season in 2010; c.f. Barwick, 2011). Many of these species are already managed under various plans of action (Recovery Plan for Marine Turtles in Australia, National Plan of Action and Conservation and Management of Sharks, and National Recovery Plan for Whales Sharks). Furthermore, the amount of by-catch species caught in prawn trawl nets has been significantly reduced since 2000 through the mandatory introduction of Turtle Excluder Devices (TEDs) and By-catch Reduction Devices (BRDs). Nets with TEDs are particularly effective at reducing catches of larger animals such as turtles (by 99%), large rays and sharks (> 1m wide and long, by 94% and 86%, respectively); in contrast, BRDs are more effective at excluding small fishes (Brewer et al., 2006). However, Brewer et al. (2006) estimate that nets with a combination of a TED and BRD reduced the catches of sea snakes (*Hydrophiidae*) by only 5%.

Sea snake catches appear to be significantly correlated to fishing effort in the fishery. Although the tiger and banana prawn sub-fisheries both use gear that can be broadly classified as demersal otter trawls, the method of gear deployment varies. In the tiger prawn sub-fishery, the trawl is generally lowered over suitable prawn habitat to fish as close as possible to the seabed, and is towed for three to four hours. In contrast, in the banana prawn sub-fishery the trawl gear is deployed for less than an hour on a prawn aggregation (or 'boil') in the water column identified using an echo sounder (Griffiths et al., 2007). The amount of by-catch thus varies by sub-fishery, making it important to consider their effects separately.

Linear regressions, between historical sea snake catches $\mathrm{Y}_{\mathrm{snake},f}(y(t))$ by sub-fishery $f = 1+2, 3$ (with $f = 1+2$ corresponding to the tiger prawn sub-fishery and $f = 3$ to the banana prawn sub-fishery) and the associated annual fishing effort $\mathrm{E}_f(y(t))$, are displayed in appendix B (figure 5.6). Table 5.1 displays the statistics of these regressions.

Table 5.1: Statistics of the linear regression between annual sea snake catches by tiger and banana prawn sub-fisheries and annual effort associated (intercept at 0).

	sub-fishery	
	tiger ($f = 1+2$)	banana ($f = 3$)
Adjusted R Square	0.785	0.778
Residual Variance σ_f^2	938.98	274.25
P-value	$8.843.10^{-6}$	$2.687.\ 10^{-5}$
Coefficient values a_f^{reg}	1.1883	0.5235

Estimation of total annual sea snake catches $\mathrm{Y}_{\mathrm{snake},f}(y(t))$ from tiger and banana prawn sub-fisheries are calculated separately:

$$\begin{cases} \mathrm{Y}_{\mathrm{snake},1+2}(y(t)) = a_{1+2}^{reg}\mathrm{E}_{1+2}(y(t)) + \xi_{1+2}(y(t)), \\ \mathrm{Y}_{\mathrm{snake},3}(y(t)) = a_3^{reg}\mathrm{E}_3(y(t)) + \xi_3(y(t)). \end{cases} \tag{5.6}$$

with

$$\begin{cases} \xi_{1+2}(y(t)) \rightsquigarrow \mathcal{N}(0, \sigma_{1+2}^2), \\ \xi_3(y(t)) \rightsquigarrow \mathcal{N}(0, \sigma_3^2). \end{cases} \tag{5.7}$$

where $\mathrm{E}_{1+2}(y(t))$ and $\mathrm{E}_3(y(t))$ are respectively the annual effort of tiger and banana prawn sub-fisheries during the year $y(t)$. a_{1+2}^{reg} and a_3^{reg} are the coefficient values from the linear regressions by sub-fishery $f = 1+2, 3$ given in table 5.1. $\xi_{1+2}(y(t))$ and $\xi_3(y(t))$ are the residual terms for the year $y(t)$ and are assumed to be independent normally distributed random variables with mean equal to zero and variance σ_{1+2} and σ_3, respectively.

Economic component

The economic component of the model estimates the flow of costs and revenues from fishing over time.

Gross income for grooved ($f = 1$) and brown ($f = 2$) tiger prawn fishing strategies are calculated from catches of tiger and blue endeavour prawns ($s = 1, 2, 3$) and gross income for banana prawn sub-fishery ($f = 3$) from catches of white banana prawn ($s = 4$), as described in chapter 4 section 4.2.3.

Total annual profit of the whole fishery $\pi(y(t))$ for year $y(t)$ is then expressed by:

$$\pi(y(t)) = \sum_{f=1}^{3} \sum_{t=52\left(y(t)-1\right)+1}^{52y(t)} \left(\mathrm{Inc}_f(t, \mathrm{E}_f(t)) - c_f^{var}\mathrm{E}_f(t)\right) - c_v^{fix}\mathrm{K}(y(t)). \tag{5.8}$$

where $\mathrm{Inc}_f(t, \mathrm{E}_f(t))$ is the annual gross income of fishing strategy f for the time t and related to $\mathrm{E}_f(t)$, the fishing effort (expressed in days at sea) of the fishing strategy f during time t, c_f^{var} corresponds to the variable cost for one unit of fishing effort of fishing strategy f, and c_v^{fix} is the annual fixed cost by vessel. Details on costs are given in section 4.2.3 in the chapter 4. $\mathrm{K}(y(t))$ is the number of vessels involved in the NPF during the year $y(t)$.

The net present value (NPV) of the flow of profits over the simulation time is calculated as in Punt et al. (2010):

$$\mathrm{NPV} = \sum_{y(t)=1}^{T-1} \frac{\pi(y(t))}{(1+\delta)^{y(t)-1}} + \frac{[\pi(T)/\delta]}{(1+\delta)^{T-1}}. \tag{5.9}$$

where δ is the discount rate and $\pi(T)$ is the level of profit during the terminal year of the simulation.

Major components and interactions that characterise the NPF and described earlier are represented in figure 5.1. This figure displays also the fact that weekly effort of tiger and banana prawn sub-fisheries are dependent, as explained in chapter 4 section 4.2.5.

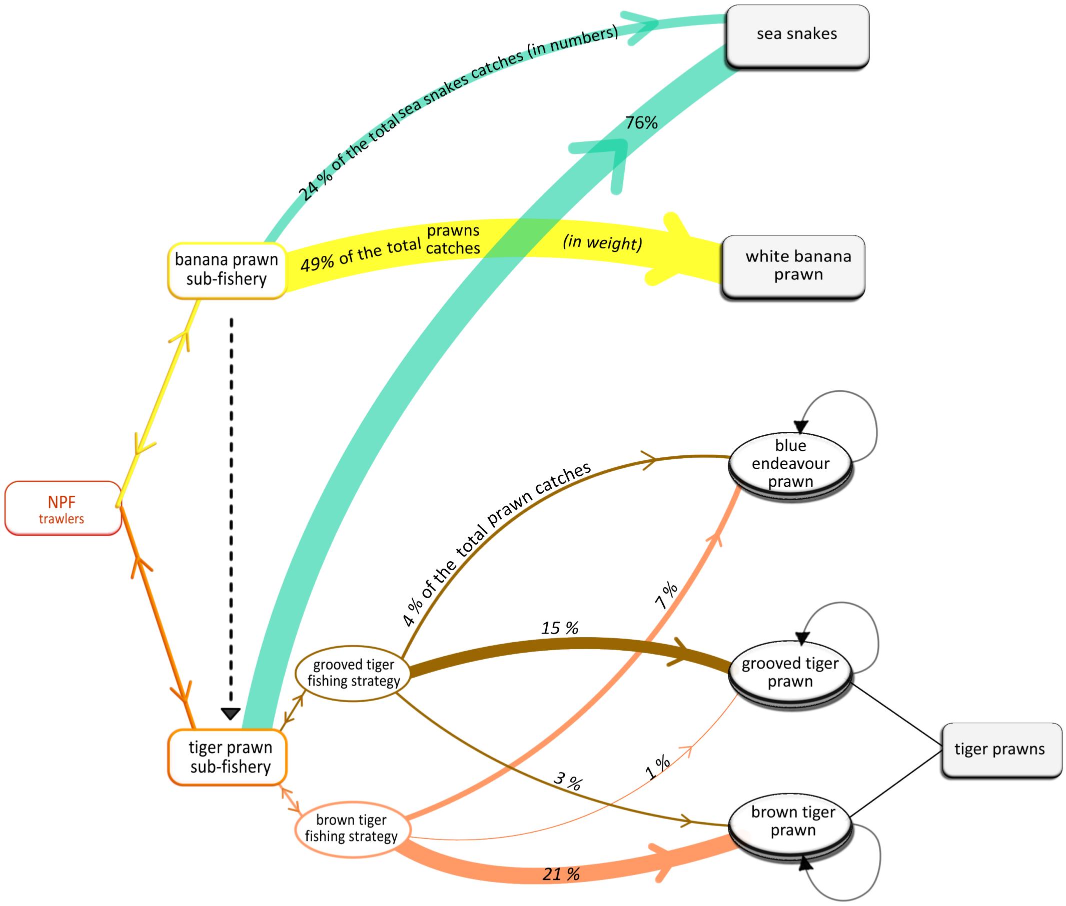


Figure 5.1: Stylized representation of the Northern Prawn Fishery used as a basis to develop the bio-economic model. The width of the arrows between the fishing strategies and prawn species are proportional to the proportion of catch by species by fishing strategies (grooved and brown tiger prawn and banana prawn sub-fisheries) compared to the total catch of the fishery in 2010. The width of the blue arrows towards the sea snake species are proportional to the sea snake catches by tiger and banana prawn sub-fisheries in 2010. The dashed arrow between tiger and banana prawn sub-fisheries illustrates the influence of the banana prawn season on the tiger prawn effort.

5.2.3 Effort allocation strategies

The biological, economic and biodiversity performances of the fishery are determined for seven effort allocation strategies which differ in the proportion of annual effort allocated to the tiger prawn sub-fishery[1]. The seven strategies also contrast in terms of the resulting weekly effort pattern of tiger and banana prawn sub-fisheries (estimated as described in the effort allocation model section 4.2.5 in chapter 4), and are detailed in table 5.2. They include an ‘adaptive’ effort allocation strat-

[1]more intermediate strategies were studied and analysed, however, for the sake of simplicity, only seven are displayed in this chapter.

egy (T_{adapt}) which currently best reflects the situation in the NPF. The resulting proportion of total annual effort directed to the tiger prawn sub-fishery ranges between 60 and 76%. Two 'banana specialisation' effort allocation strategies (T_0 and T_{10}) consist in setting the proportion of annual effort of the tiger prawn sub-fishery to 0% and 10% of total annual effort. Three 'tiger specialisation' effort allocation strategies (T_{80}, T_{90} and T_{100}) involve focusing the effort on the tiger prawn sub-fishery by allocating 80%, 90% and 100% of the annual effort to this sub-fishery. A 'balanced' effort allocation strategy (T_{50}), in which total effort is split equally between tiger and banana prawn sub-fisheries is also considered. The pattern of open and closed weeks is set to that which occurred in 2010, as a result of policy and technical constraints (e.g. mid-season closures).

Table 5.2: Effort allocation strategies (in each row) considered in this study. The strategies differ in the annual effort $E_{1+2}(y(t))$ allocated to tiger prawn sub-fishery.

Allocation strategies	Description	Annual effort of tiger prawn sub-fishery
T_0	annual proportion of tiger prawn sub-fishery effort sets to 0%.	$E_{1+2}(y(t)) = 0$
T_{10}	annual proportion of tiger prawn sub-fishery effort sets to 10%.	$E_{1+2}(y(t)) = 0.1E(y(t))$
T_{50}	annual proportion of tiger prawn sub-fishery effort sets to 50%.	$E_{1+2}(y(t)) = 0.5E(y(t))$
T_{adapt}	'adaptive' effort allocation strategy.	see equations (4.16) and (4.17) in chapter 4
T_{80}	annual proportion of tiger prawn sub-fishery effort sets to 80%.	$E_{1+2}(y(t)) = 0.8E(y(t))$
T_{90}	annual proportion of tiger prawn sub-fishery effort sets to 90%.	$E_{1+2}(y(t)) = 0.9E(y(t))$
T_{100}	annual proportion of tiger prawn sub-fishery effort sets to 100%.	$E_{1+2}(y(t)) = E(y(t))$

5.2.4 Fishing capacity strategies

We assess the effects of changes in fishing capacity using four strategies regarding management of fishing capacity, defined in terms of the number of vessels $K(y(t))$ and total annual days at sea per vessel $e(y(t))$. These include a status quo fishing capacity strategy SQ which stands for an annual number of vessels and total annual effort equal to the values observed in 2010, i.e. $K(y(t))=52$ and $e(y(t))=162$. Also considered are a capacity strategy, e^+ , where the number of vessels remains at its 2010 level while annual effort per vessel is increased to that the maximum number of days allowed by the open week pattern (including seasonal spawning and small prawns closures), corresponding to an annual effort per vessel of 196 days at sea, and a capacity strategy, K^-, which represents a 50% decrease in the annual number of vessels but no change in the average effort by vessel compared to 2010, i.e. $K(y(t))=26$ and $e(y(t))=162$. As shown in chapter 4, reducing the number of vessels and increasing the effort per vessel have contrasting effects on economic performance of the NPF, respectively decreasing the economic variability and increasing the average economic performance. A $K^{-}e^{+}$ fishing capacity strategy is therefore added to this study. Table 5.3 summarizes the four fishing capacity strategies taken into account in this study.

Table 5.3: Fishing capacity strategies (in each row). They differ in the total annual effort $E(y(t))$ and annual number of vessels $K(y(t))$.

Capacity strategies	Description
SQ	$K(y(t))$=52 vessels and $e(y(t))$=162 days at sea per vessel
e^{+}	$K(y(t))$=52 vessels and $e(y(t))$=196 days at sea per vessel
K^{-}	$K(y(t))$=26 vessels and $e(y(t))$=162 days at sea per vessel
$K^{-}e^{+}$	$K(y(t))$=26 vessels and $e(y(t))$=196 days at sea per vessel

5.2.5 Stochastic co-viability analysis

The stochastic co-viability framework is used to describe the trade-offs between biological, economic and biodiversity conservation management objectives under various management strategies. The method requires specifying constraints on the values of indicators associated with the biological, economic and biodiversity conservation objectives. Given the stochastic nature of the model (i.e. uncertainties in tiger and blue endeavour prawn recruitments, white banana prawn annual biomasses and annual sea snake catches), the performance of the fishery is assessed in terms of the probability of these constraints being met by the fishery at any point in time (Doyen and De Lara, 2010). The co-viability of the system is examined by simultaneously assessing the ability of the fishery to respect biological, economic and biodiversity conservation constraints at any time of the simulation and with sufficiently high probability.

In this study, the biological objective consists in ensuring the conservation of the prawn population by requiring that the spawning stock size index $S_s(y(t))$ of each individual species $s = 1, 2, 3$ is maintained above a threshold value as:

$$S_s(y(t)) \geq S_s^{\text{lim}}, \qquad s = 1, 2, 3. \tag{5.10}$$

with S^{lim} the limit spawning stock size index of species s defined as the minimal historically observed spawning stock size index value by species s over the 1970-2010 period (values in table 5.5 in appendix A).

NPF fishing management strategies are defined by three management variables: the annual number of vessels $\mathrm{K}(y(t))$, the total annual effort per vessel $e(y(t))$ and the annual proportion of effort $\propto_{1+2}(y(t))$ directed towards the tiger prawn sub-fishery. The biological viability is then assessed by the population viability probability (PVA), as described by:

$$\mathrm{PVA}\Big(\mathrm{K}, e, \propto_{1+2}\Big) = \mathbb{P}\Big(\text{constraints (5.10) are satisfied for } y(t) = y_0, \ldots, T\Big). \tag{5.11}$$

The economic objective in this study requires maintaining a minimum total annual profit for the NPF.

$$\pi(y(t)) \geq \pi^{\mathsf{min}} \tag{5.12}$$

with π^{min} the minimal profit set to 60% of the annual profit in 2010 (values in table 5.5 in appendix A).

The economic viability probability of the fishery (EVA) related to the annual number of vessels, the total annual effort per vessel and the annual proportion of effort directed to the tiger prawn sub-fishery is thus expressed as:

$$\mathrm{EVA}\Big(\mathrm{K}, e, \propto_{1+2}\Big) = \mathbb{P}\Big(\text{constraint (5.12) are satisfied for } y(t) = y_0, \ldots, T\Big). \tag{5.13}$$

A biodiversity conservation objective is also considered in this study, and viability on this domain requires maintaining the catch of sea snakes below a maximum 'allowed' level:

$$\mathrm{Y}_{\mathrm{snake}}(y(t)) \leq \mathrm{Y}_{\mathrm{snake}}(2010) \tag{5.14}$$

with $\mathrm{Y}_{\mathrm{snake}}(2010)$ the maximum allowed sea snake catch set to the value observed in 2010 (values in table 5.5 in appendix A).

The ecological or impact viability probability (IVA) of the NPF is then described by:

$$\mathrm{IVA}\Big(\mathrm{K}, e, \propto_{1+2}\Big) = \mathbb{P}\Big(\text{constraint (5.14) are satisfied for } y(t) = y_0, \ldots, T\Big). \tag{5.15}$$

Co-viability analysis requires that biological, economic and biodiversity constraints are jointly considered. These constraints characterize an acceptable sub-region of the phase space within which the fishery evolves. A particular trajectory followed by the fishery will be called viable if it remains in this region during the prescribed period of time, with a sufficiently high probability (set to 95% in this study). Thus, the bio-eco-diversity performance of the system is evaluated by the probability of co-viability (CVA) of the system in a stochastic context and given by:

$$\begin{aligned}\mathrm{CVA}\Big(\mathrm{K}, e, \propto_{1+2}\Big) = \mathbb{P}\Big(& \text{ constraints (5.10), (5.12) and (5.14)} \\ & \text{are satisfied for } y(t) = y_0, \ldots, T\Big).\end{aligned} \tag{5.16}$$

1000 trajectories of spawning stock size indices and annual total profits are simulated over a 10 year period from 2010. Furthermore, to account for the uncertainty in the estimation of sea snake catches, for each of these 1000 trajectories, 10 estimations of sea snake catches are made as described in equation (5.6). Each trajectory represents a possible state of nature for each year of the simulation, $\omega(.) = (\omega_1(.),\omega_2(.),\ \omega_3(.),\omega_4(.)\ ,\omega_5^i(.)_{i=1:10})$; which stands for the set of annual recruitments of tiger (grooved and brown) and blue endeavour prawn as detailed in chapter 4 and in Punt et al. (2010, 2011), of white banana prawn annual biomasses as in equation (14) in chapter 4, and of total annual sea snake catches. The different $\omega_i(.)$ are assumed to be independent by species. Each combination of effort allocation and fishing capacity strategies is simulated with the same set[2] of $\omega(.)$.

The numerical implementations and computations of the model have been carried out with the scientific software SCILAB[3].

5.3 Results

The biological, economic, ecological, and co-viability probabilities (PVA, EVA, IVA and CVA), and the overall economic performance of the whole fishery (i.e. tiger and banana prawn sub-fisheries), represented by the mean net present value (NPV) of the fishery, are analysed for various combination of effort allocation and fishing capacity strategies taking into account the stochastic nature of the model.

5.3.1 Management strategy performances

The values of the four viability probabilities and the associated mean NPV, for each combination of the seven effort allocation strategies and the four fishing capacity strategies described in sections 5.2.3 and 5.2.4 respectively, are given in table 5.4.

[2]The set of $\omega(.)$ used in this chapter is the same that the one used in chapter 4.

[3]SCILAB is a freeware http://www.scilab.org/ dedicated to engineering and scientific calculus. It is especially well-suited to deal with dynamic systems and control theory.

Table 5.4: Biological, economic, ecological and co-viability probabilities of seven effort allocation strategies with four fishing capacity strategies. Allocation strategies are displayed by fishing capacity strategy.

Capacity strategies	Allocation strategies		Viability probabilities				mean NPV (in AU$ million)
			PVA	EVA	IVA	CVA	
SQ	T_0	prop. tig. effort ⟶	100	0.4	99.98	0.4	283.43
	T_{10}		100	1.6	99.8	1.6	345.86
	T_{50}		100	35.8	31.81	11.79	434.26
	T_{adapt}		100	79	0.44	0.38	429.30
	T_{80}		100	76.6	0	0	374.39
	T_{90}		100	75	0	0	337.81
	T_{100}		100	57.5	0	0	291.64
e^+	T_0		100	2.2	99.14	2.19	430.81
	T_{10}		100	4.4	93.37	4.19	503.90
	T_{50}		100	66	0.01	0.01	572.23
	T_{adapt}		100	91.1	0	0	537.58
	T_{80}		100	87.4	0	0	453.95
	T_{90}		100	83.3	0	0	390.35
	T_{100}		99.8	62.8	0	0	315.88
K^-	T_0		100	0	100	0	141.71
	T_{10}		100	0.1	100	0.1	175.26
	T_{50}		100	15.9	100	15.9	264.69
	T_{adapt}		100	82.2	99.99	82.19	294.21
	T_{80}		100	86.1	99.97	86.07	287.58
	T_{90}		100	92.5	99.93	92.43	289.95
	T_{100}		100	92.3	99.78	92.09	287.94
K^-e^+	T_0		100	0.2	100	0.2	215.40
	T_{10}		100	1.1	100	1.1	255.48
	T_{50}		100	55.6	99.98	55.59	351.48
	T_{adapt}		100	97.3	99.77	97.07	375.08
	T_{80}		100	97.7	98.97	96.67	361.89
	T_{90}		100	98.5	97.13	95.63	355.76
	T_{100}		100	98.1	92.66	90.83	345.65

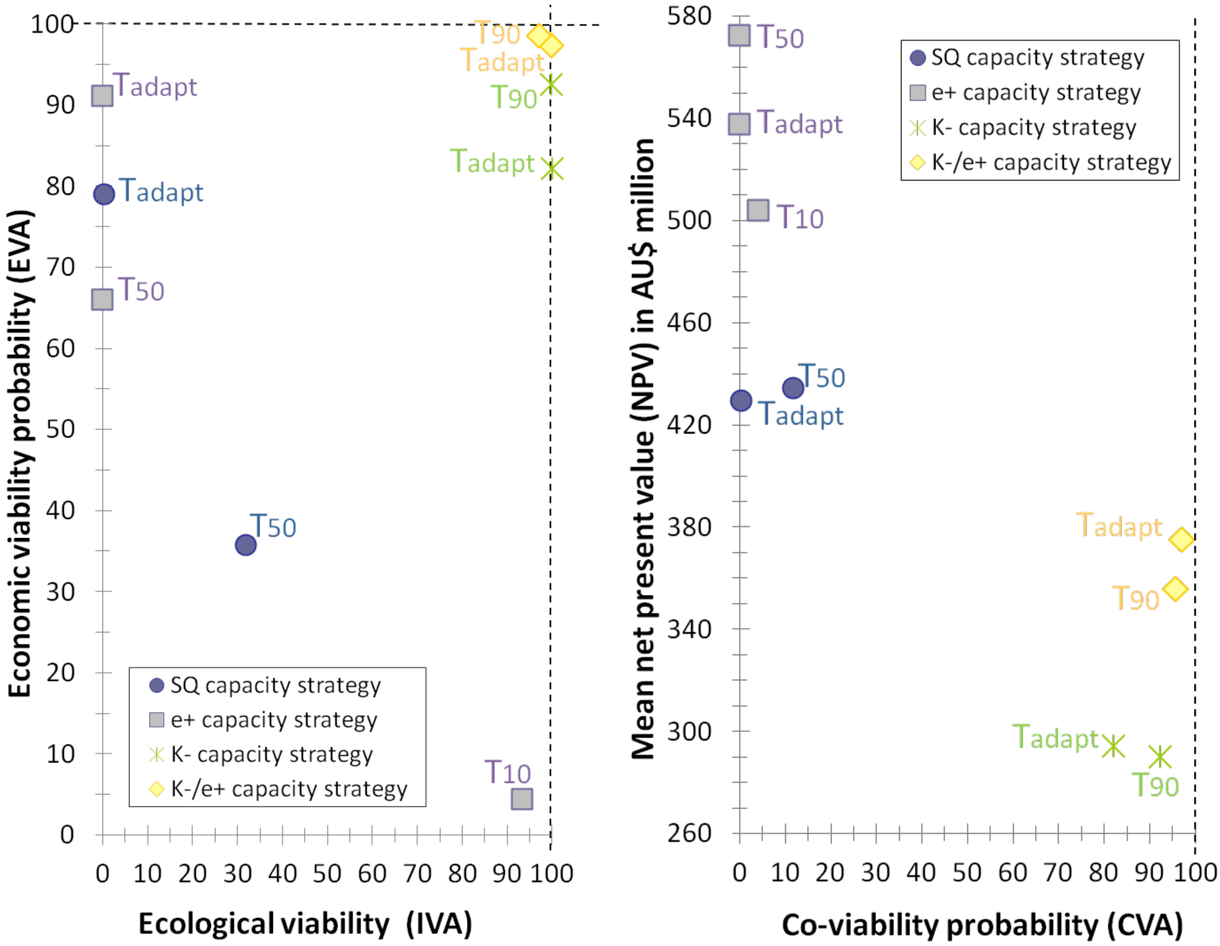
Table 5.4 shows that the biological constraints will be met in the fishery with a very high degree of certainty under all management strategy combinations. The economic and ecological constraints, however, are met with widely varying probabilities, and show the existence of trade-offs between management objectives. Results suggest that, under the modelling assumptions used, the current management of the fishery (i.e. T_{adapt} effort allocation strategy with SQ fishing capacity strategy) may not be viable, with its co-viability probability (CVA) equal to 0.38%. Less than 1% of the simulated trajectories remain within the biological, economic and biodiversity conservation constraints at all simulation times. More specifically this management strategy has a moderate economic risk with an economic viability probability (EVA) equal to 79% and a very low ecological viability:

$\mathrm{IVA}(\mathrm{SQ}, \mathrm{T}_{\mathrm{adapt}}) = 0.44\%$. Moreover, the fishery operates with 75% of the maximum NPV that could be achieved. The highest mean NPV would be obtained with a T_{50} 'balanced' effort allocation strategy combined with the e^{+} capacity strategy (NPV = AU\$ 572.23 million). However, this strategy would not be ecologically viable, with only a 0.01% probability of not exceeding the allowed level of sea snake catch for all years of the simulation. Moreover the minimal annual profit required by the economic objective is guaranteed for all simulation times in only 66% of the trajectories. The highest CVA would be obtained with a combined $\mathrm{T}_{\mathrm{adapt}}$ effort allocation strategy and a $\mathrm{K}^{-}\mathrm{e}^{+}$ capacity strategy: $\mathrm{CVA}(\mathrm{K}^{-}\mathrm{e}^{+}, \mathrm{T}_{\mathrm{adapt}}) = 97.07\%$. The associated NPV associated would be about 65% of the maximum achievable value. This economic loss can be interpreted as a 'cost of sustainability' associated with the objective to meet all the constraints imposed on the fishery, i.e. the opportunity cost of increasing CVA.

The effort allocation strategies with the highest CVA, EVA and NPV for each of the four fishing capacity strategies were selected for graphical comparison. Figure 5.2(a) displays the trade-off between the economic and ecological performances of these selected strategies, and figure 5.2(b) shows the trade-off between mean economic performance and the bio-eco-diversity performance (or co-viability probability), of the fishery.

Figure 5.2(a) shows that for both fishing capacity strategies involving 52 vessels (SQ and e^{+}), there is a trade-off between EVA and IVA across allocation strategies. More particularly, effort allocation strategies with the highest EVA are associated with IVA values equal or close to zero. However, for fishing capacity strategies involving a reduced number of vessels, there is a greater than 95% probability of not violating the IVA constraint. Figure 5.2(b) exhibits a strong trade-off between mean economic performance (mean NPV) and co-viability of the fishery.

The current effort allocation strategy $\mathrm{T}_{\mathrm{adapt}}$ is among the best performing of the selected strategies displayed in figure 5.2 regardless of the capacity strategy chosen. While the best performance in terms of the economic objective among all management strategy combinations is achieved with the T_{90} allocation strategy.

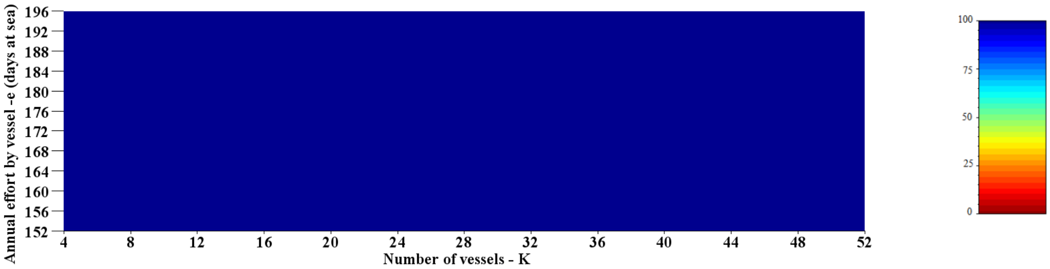


(a) EVA versus IVA (b) NPV versus CVA

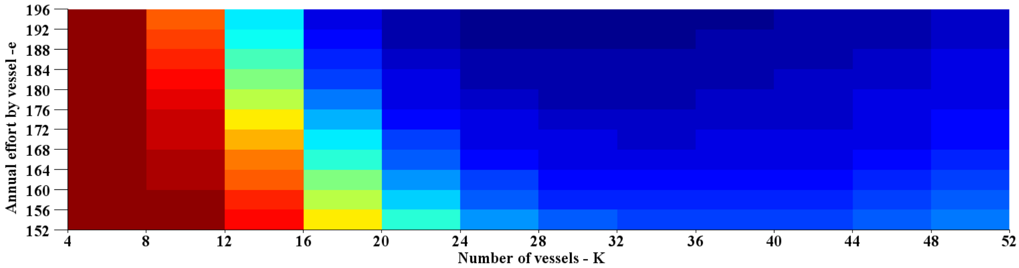
Figure 5.2: Economic viability probability (EVA) versus ecological viability probability (IVA) in (a) and mean net present value (NPV) versus co-viability probability (CVA) in (b) of selected management strategies. Among each of the four fishing capacity strategies, are selected the effort allocation strategies with the highest CVA, with the highest EVA and the highest NPV. Fishing capacity strategies are represented by colors and effort allocation strategies are written near the associated dot.

5.3.2 Exploratory approach for the best strategies

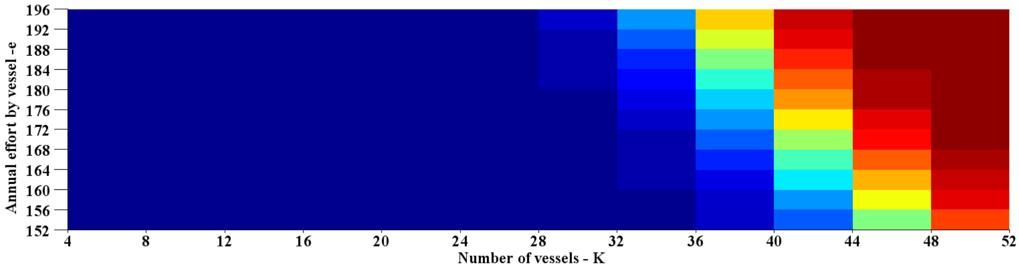
This section explores the performance of a set of more refined fishing capacity strategies which are combined with each of the two effort allocation strategies, the performances of which were highlighted in section 5.3.1: T_{adapt} and T_{90}. Figures 5.3 and 5.4 display the viability probabilities of T_{adapt} and T_{90} effort allocation strategies, respectively, as a function of a range of fishing capacity settings as defined by K and *e*. Based on the results presented above, viability probabilities are calculated for the number of vessels from 1 to 52 and the total annual effort per vessel from 152 to 196 days at sea (196 days being the maximum allowed by the seasonal closures).



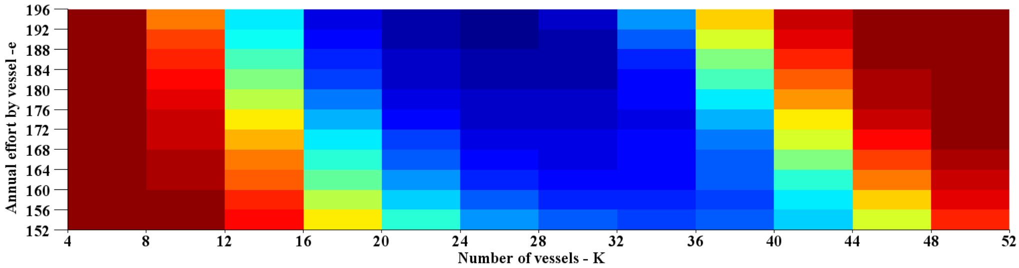
(a) Biological viability probability



(b) Economic viability probability

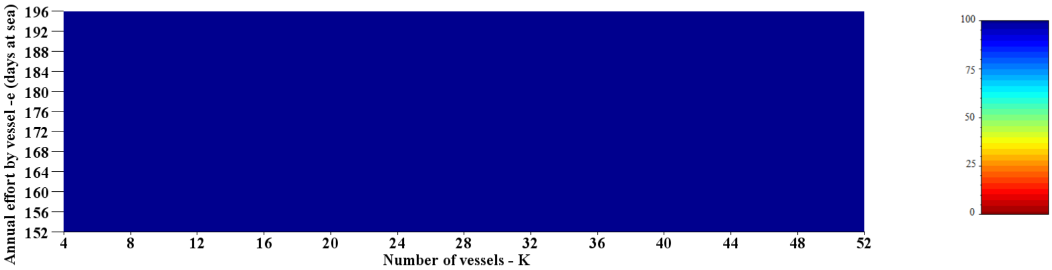


(c) Ecological viability probability

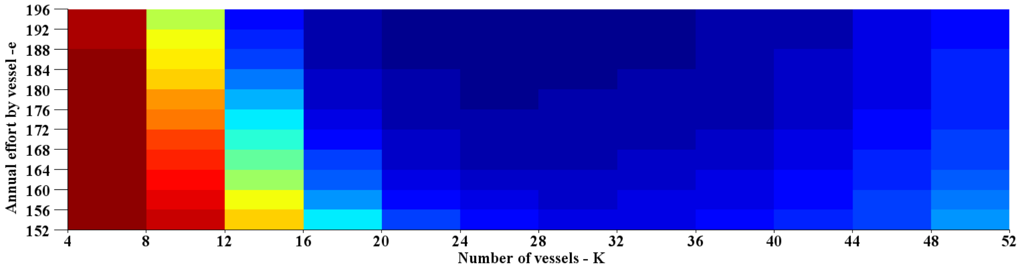


(d) Co-viability probability

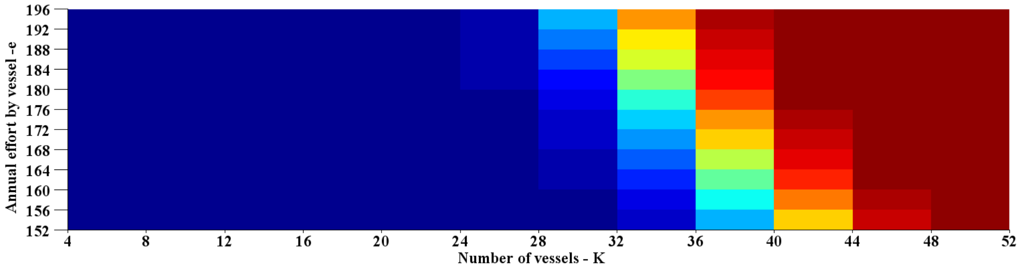
Figure 5.3: Viability probabilities of an adaptive effort allocation strategy T_{adapt} according to the number of vessels $K(y(t))$ (horizontal axis) and total annual effort per vessel $e(y(t))$ (vertical axis) for all years $y(t)$ of the simulation. Biological viability probability (PVA) is represented in (a), economic viability probability (EVA) in (b), ecological viability probability (IVA) in (c) and co-viability probability (CVA) in (d). Blue colors mean that the viability probability is closed to 100%, i.e. the constraints are respected for all time. Whereas reddish colors means that the viability probability is close or equal to 0%; i.e. for every trajectory at least one constraint is violated for at least one year.



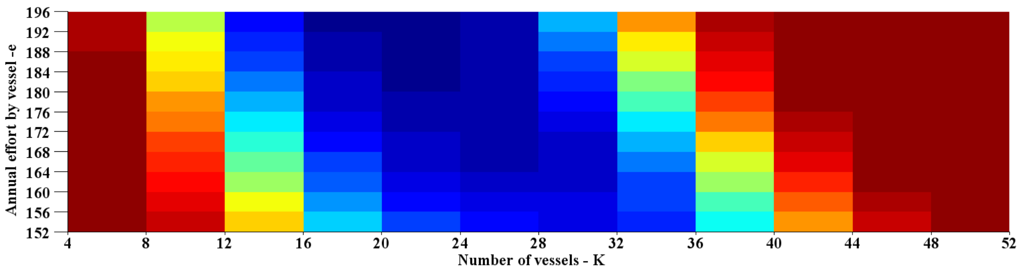
(a) Biological viability probability



(b) Economic viability probability



(c) Ecological viability probability



(d) Co-viability probability

Figure 5.4: Viability probabilities of a tiger specialisation effort allocation strategy T_{90} according to the number of vessels $K(y(t))$ (horizontal axis) and total annual effort per vessel $e(y(t))$ (vertical axis) for all years $y(t)$ of the simulation. Biological viability probability (PVA) is represented in (a), economic viability probability (EVA) in (b), ecological viability probability (IVA) in (c) and co-viability probability (CVA) in (d). Blue colors mean that the viability probability is closed to 100%, i.e. the constraints are respected for all time. Whereas reddish colors means that the viability probability is close or equal to 0%; i.e. for every trajectory at least one constraint is violated for at least one year.

Results confirm that meeting the biological constraint is not an immediate management issue for the fishery as represented in this model, as PVA is equal to 100% in all cases. Comparisons of figures 5.3(b) with 5.3(c) and 5.4(b) with 5.4(c) highlight the conflict between economic and biodiversity conservation constraints (i.e. between profits and sea snake catches). Indeed, while low levels of fishing capacity lead to high IVA values, they result in low probabilities of achieving the minimum annual profit in each year of the simulation.

The highest co-viability probability values are obtained with an annual effort per vessel of 196 days at sea for both the current adaptive and the tiger specialisation allocation strategies. Furthermore, under the T_{adapt} allocation strategy, the highest CVA are achieved when the number of vessels is slightly higher than the level required to maximise CVA under a T_{90} allocation strategy. This is consistent with the chapter 4 results, where effort allocation strategies involving higher tiger prawn specialisation performed relatively better, as compared to strategies where effort was directed more towards banana prawn sub-fishery, when reducing the number of vessels. This comes from the fact that the tiger prawn sub-fishery requires less fishing capacity than the banana prawn sub-fishery due to the nature of the resources.

5.3.3 Which number of vessels for a viable management?

We further explore the question of what is the optimal number of vessels in the fishery when both co-viability and mean NPV are important management or stakeholder considerations.

Results of section 5.3.2 exhibit that the best total annual effort per vessel, in terms of CVA and mean NPV, is 196 days at sea per year. Fishing capacity strategies combining an annual effort per vessel of 196 days and number of vessels with which the CVA is greater than or equal to 95% when associated with both T_{adapt} and T_{90} effort allocation strategies, are therefore selected for further analyses. With the T_{adapt} strategy, CVA greater than 95% are obtained with 21 to 29 vessels, while with T_{90} allocation strategy, this number ranges between 15 and 26 vessels. Figure 5.5 displays the average NPV versus the CVA calculated with the different annual number of vessels for T_{adapt} and T_{90} strategies with 196 days at sea per vessel.

Figure 5.5 shows that highest CVA are obtained with the T_{90} effort allocation strategy, while highest NPV are obtained with T_{adapt}. A trade-off exists between mean economic performance and

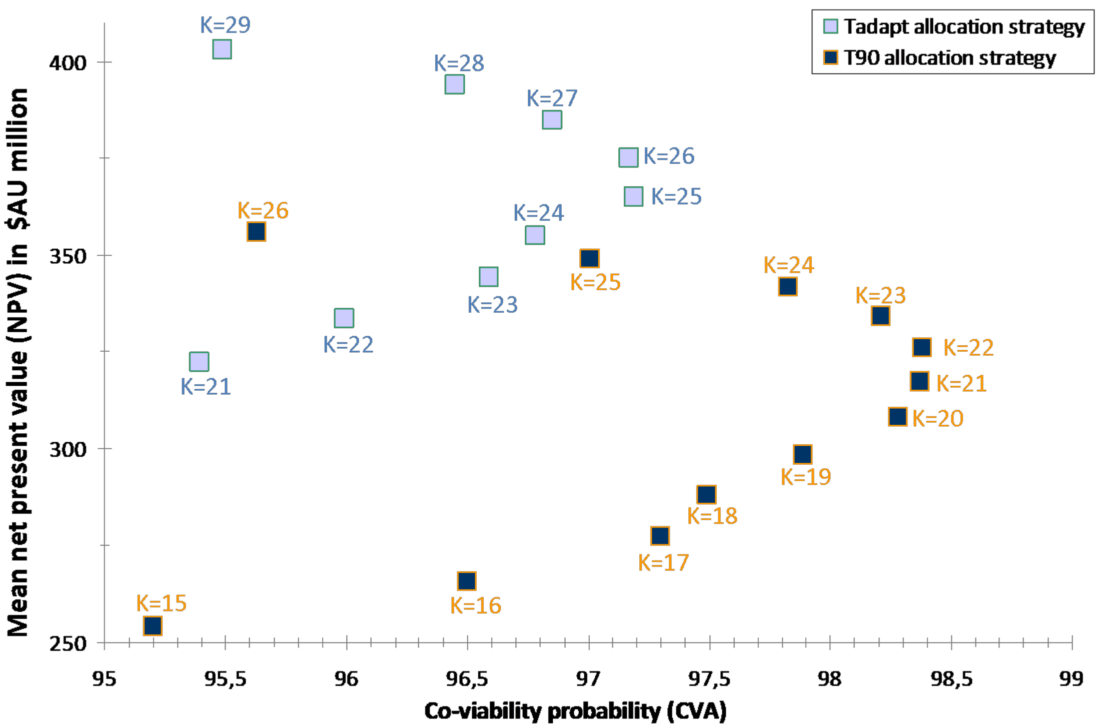


Figure 5.5: Mean net present value (NPV) of adaptive effort allocation strategy T_{adapt} (blue diamonds) and tiger specialisation allocation strategy T_{90} (red squares) with an annual effort per vessel of 196 days at sea and with various number of vessels (K) versus the co-viability probability (CVA) associated. Each dot is associated to a different annual number of vessel (written on the figure) with $K(y(t)) = 21$ to 29 vessels for T_{adapt} strategy and $K(y(t)) = 15$ to 26, for all years $y(t)$ of the simulation.

viability. For each effort allocation strategy, there is an optimal number of vessels to maximise CVA. However beyond this number, greater NPV can be achieved, but only at the cost of reducing CVA, i.e. referring to the 'cost of sustainability'. Furthermore, for a given number of vessels beyond 25, the 'adaptive' effort allocation strategy T_{adapt} dominates as will have higher CVA and NPV compared to the tiger specialisation allocation strategy T_{90}.

5.4 Discussion

Stochastic co-viability analysis is presented as a way to assess the trade-offs involved in managing mixed fishery bio-economic systems, in the presence of conflicting objectives. The modelling approach proposed here accounts for the interaction between tiger and banana prawn sub-fisheries, and allows assessing various management strategies for the Northern Prawn Fishery (NPF) as described in this simplified model. Co-viability probabilities are used as indicators of the sustainable performance of the fishery under uncertainty, taking into account the objectives of maintaining high levels of profit, preserving target stocks, and limiting the impacts of the fishery on marine biodiversity.

The study compares the co-viability probabilities (CVA) and a more classical economic performance measure, the average net present values (NPV), of the fishery, for different effort allocation strategies (relying on the proportion of effort allocated to the tiger prawn sub-fishery) and various fishing capacity strategies (depending on total annual effort per vessel and number of vessels). In this study, management strategies with a co-viability probability greater than 95% are considered viable.

5.4.1 Assessment of trade-offs associated with managing the fishery

Based on our simulations, and regarding the assessed prawn species taken into account in this study and the associated biological thresholds, it appears that the biological constraints have relatively less influence on the viability probability, as compared to the economic and biodiversity conservation constraints. Indeed the population viability probability (PVA) reaches 100% in almost all cases, which means that the biological objective is achieved at any time of the simulation and for any simulated state of nature (i.e. uncertainties on biological recruitment of grooved and brown tiger and blue endeavour prawns).

Based on the data used to calibrate the model and modelling assumptions (particularly, assumptions on stock-recruitment relationships, effort allocation model, sea snake catch estimations and economic assumptions), it appears that the status quo management approach may not be viable, when assessed against the economic and biodiversity constraints[4] defined in this analysis. Simulation results show that improving the viability status of the fishery would involve a reduction in the number of vessels (currently 52 as set from MEY objective analyses).

The analysis also reveals a trade-off between economic and biodiversity conservation objec-

[4]It is not surprising that the status quo management approach may not be ecologically viable, as defined in this study. Indeed the threshold for biodiversity conservation constraint is set to the 2010 sea snake catch and residual terms randomly distributed are integrated in sea snake catch estimations. Therefore sea snake catch estimations under status quo management approach will be above and below the 2010 sea snakes catch (the biodiversity conservation threshold to not exceed), and ecological viability probability is calculated regarding the respect of constraint for all time. This means that if the sea snake catches among one given trajectory are superior to this threshold for at least one year of the simulation, the trajectory will be considered as not ecologically viable. This reflects a willing to guarantee that 2010 sea snake catch is not exceeded for any state of nature.

tives. Some management strategies allow compromises leading to high co-viability probabilities. Analysis shows that the highest co-viability probabilities are associated with a reduction in number of vessels and an increase of their annual effort per vessel to the maximum allowed by seasonal closures, which corresponds to 196 days at sea. Moreover, analyses draw attention to two effort allocation strategies: an 'adaptive' effort allocation strategy (T_{adapt}, with a proportion of effort directed towards tiger prawn sub-fishery between 60 and 76%) and a 'tiger specialisation' effort allocation strategy allocating 90% of the annual effort towards tiger prawn sub-fishery (T_{90}). These allocation strategies when combined with a reduction of number of vessels and increase of their annual effort, allow reconciliation of the multiple objectives as defined in this study.

Figure 5.2(b) illustrates the strong trade-off between mean economic performance (mean NPV) and co-viability of the fishery[5]. A compromise between mean economic performance and viability must therefore be made, which we call the 'cost of sustainability'. This can be expressed as the opportunity cost of improving CVA in terms of reduced NPV. It has been demonstrated in chapter 4, that an implicit risk aversion level exists in the NPF industry. By its definition, high EVA may imply lower variability in combination with high mean profit. Therefore fishing industry looking to reduce economic risk, should seek high EVA, and consequently high CVA. Final decisions on management settings for sustainable NPF management are thus dependent on management objectives, but also on the risk aversion level of the fishing industry. For instance, if the fishing industry strives for strong economic performance among viable management strategies (i.e. highest NPV among CVA greater than or equal to 95%), reducing the fleet capacity to 29 vessels, while increasing the annual effort per vessel to 196 days at sea and keeping an 'adaptive' allocation effort strategy, would appear to be in order ($NPV(29,196,T_{adapt})$ =AU$ 403 million). However, if the fishing industry is seeking the highest possible co-viability probability (i.e. reducing at a minimum biological, economic and biodiversity conservation risks), the management options that perform best in our analysis involve a fleet capacity reduced to 22 vessels associated to an annual effort per vessel of 196 days at sea and an allocation of 90% of the total annual effort towards the tiger prawn sub-fishery ($CVA(22, 196, T_{90})$ =98.38%). The 'cost of sustainability' in this situation would be equal to AU$77 million, in terms of reduced expected NPV.

[5]Similar trade-off between mean economic performance (mean annual profit) and its variability was observed in chapter 4.

5.4.2 CVA as a step towards integrated management of mixed fisheries

The consideration of the multi-dimensional nature of marine fisheries management appears as an unavoidable reality. As part of this, consideration of the environmental impacts of fishing activities is a crucial concern, as these impacts can lead to changes in biodiversity and ultimately change the overall functionality of the ecosystem (Pauly et al., 1998; Dulvy et al., 2000). However, fishery scientists and managers often do not have the information required to properly assess fishery impacts on non-targeted species and communities, or to develop management measures to ensure the fishery operates in an ecologically sustainable manner (Zhou and Griffiths, 2008). In such cases, use of biodiversity indicators as proposed in this study can assist in explicitly addressing the impacts of fishing on biodiversity in assessments. The stochastic co-viability approach proposed here offers formal recognition of the multi-objective nature of management for the NPF, and means to integrate this with the current understanding of the dynamics of a mixed non-selective fishery system. The model illustrates the benefits of formally combining integrated bio-economic modelling with the multi-criteria evaluation underlying the co-viability framework analysis. This study demonstrates the value of the stochastic co-viability approach by providing a 'sustainability metric' through co-viability probabilities allowing to rank strategies and therefore help stakeholders to choose the appropriate fishing management settings according to their contextual management objectives. This approach allows for identification of a wide range of possibilities, according to various external sources of pressure on management, notably the risk aversion of the fishing industry, but also environmental 'pressures' from environmental lobbies and government policies. It provides a 'bargaining space' for stakeholders to identify potential compromises. The co-viability probabilities can therefore 'bring peace' in fisheries management as it does not favour any of the objectives over another. As such, management decisions may be more likely to be accepted by the various stakeholders.

5.4.3 Perspectives

The viability approach has been proposed by several authors (e.g. Mullon et al., 2004; Cury et al., 2005; Chapel et al., 2008), as a well-suited modelling framework for Ecosystem-Based Fishery Management (EBFM). EBFM must manage targeted species in the context of the overall state of the system, habitat, protected species, and non-targeted species. This study is a first step in this direction for the NPF. However, extensions of the biodiversity indicator could be considered to assess the differences in impacts from tiger and banana prawn sub-fisheries. Following the study of Bustamante et al. (2010), impacts from tiger and banana prawn sub-fisheries could be assessed more accurately with the integration of benthos species in our model. Aggregation of these species in two groups, as sessile and mobile benthos, can be relevant to take into account the contrasted impacts of tiger and banana prawn sub-fisheries on the ecosystem. For instance, tiger prawn sub-fishery have greater impacts on sessile benthos than banana prawn sub-fishery. Moreover, prawns and some mobile benthos species being predators of certain sessile benthos species, competition relationships exist between prawns and some mobile benthos species. The integration of such trophic interactions (prey-predator and indirect competition) can thus also reinforce this study. The work presented in Bustamante et al. (2010), which employs a trophic mass-balance model (using Ecopath with Ecosim software) to explore the ecological effects of demersal trawling in the NPF, could be adapted for our study.

Adding a spatial dimension to the model could also be relevant as tiger and banana prawn sub-fisheries do not fish in the same locations, and this could allow the incorporation of a broader ecological indicator accounting for effects of trawling on habitats. Indeed many studies relating to trawling and dredging have shown a depletion of the habitat, a re-suspension of the sediments and damage to the structure of the benthic communities (Collie et al., 2000) that entail variations of the production processes (Jennings et al., 2001). Therefore it is important to take these impacts into considerations when managing mixed trawl fisheries.

Several other expansions of this modelling could also be considered like the modelling of dynamic control variables through a dynamic annual number of vessels. A more social objective could also be added to the study through a social constraint, for instance via a minimal production of prawns to guarantee.

5.5 Appendix A. Bio-economic parameter values

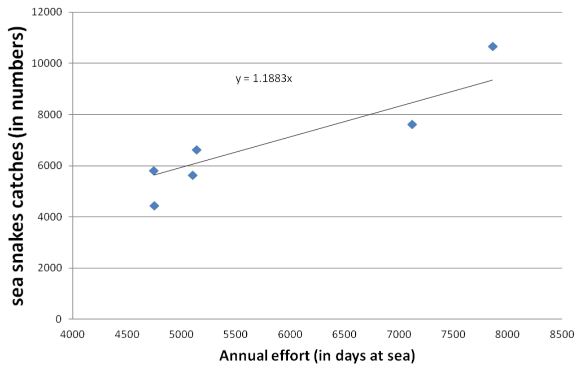
This appendix displays the values of the biological, economic and ecological parameters used in the definition of the constraints described in section 5.2.5.

Table 5.5: Threshold used in co-viability approach.

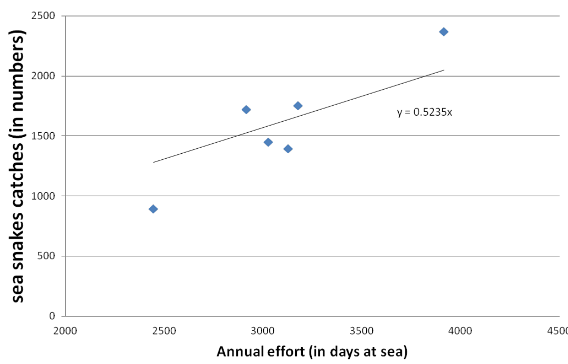
Threshold		Value
Biological S_s^{lim}	grooved tiger prawn, $s = 1$	0.293539
	brown tiger prawn, $s = 2$	0.234883
	blue endeavour prawn, $s = 3$	0.128637
Economic, π^{min}	Profit of reference, $0.6\pi(2010)$	7,140,000 (AU$)
Ecological, $Y_{\mathrm{snake}}(2010)$	Sea snake catch estimated in 2010	8430

5.6 Appendix B. Statistics

This appendix displays the linear regressions between historical sea snake catches $Y_{\mathrm{snake},f}(y(t))$ by sub-fishery $f = 1+2, 3$ and the associated annual fishing effort $E_f(y(t))$ of the prawn sub-fishery $f = 1 + 2, 3$ in figure 5.6.



(a) Tiger prawn sub-fishery



(b) Banana prawn sub-fishery

Figure 5.6: Linear regression between historical annual sea snake catches by sub-fishery and annual effort associated. Regression for the tiger prawn sub-fishery is represented in (a) and banana prawn sub-fishery in (b).

Chapter 6. General discussion

This thesis focused on the bio-economic risk and viability analysis in the management of mixed fisheries. The research was based on two case studies; the French Bay of Biscay demersal mixed fishery (BoB) and the Australian Northern Prawn Fishery (NPF). Both are industrial, high economic value, fisheries characterized by high uncertainties and using non selective trawl technology inducing important by-catches and discards. Although the two fisheries differ in their ecosystems, exploitation histories, management, and in the level of knowledge and complexity of their ecological and economic dynamics, the models developed, and evaluations carried out, for these two cases were similar. Bio-economic modelling, risk and stochastic viability analyses of these fisheries were used to identify trade-offs among multiple management objectives, and to investigate alternative management options that might ensure the bio-economic sustainability of two contextually different fisheries.

This chapter first discusses and compares the two bio-economic models developed. It then discusses the viability analyses results obtained, the viable management strategies identified and the viability drivers. Finally, perspectives for future research are provided.

6.1 Important common features of the bio-economic models

The first objective of the thesis was to develop bio-economic models for the BoB fishery and the NPF which capture the key biological and economic processes governing these fisheries. The models developed were to (i) integrate ecological and economic drivers of changes in the fisheries and ecosystems; (ii) include the complex dynamic dimensions of these drivers; and (iii) deal with biological and economic sources of uncertainty.

6.1.1 Complexity of the integrated models

Modelling of both BoB and NPF fishery systems aimed at grouping various components of the fisheries, which are not usually brought together, into a single model. In confronting the classic trade-off between increasing modelling complexity and maintaining the ability to derive general and robust conclusions about the systems considered, many features of the original BoB and NPF models have been simplified. However, key aspects of model structure have been maintained where these were considered crucial to the understanding of the bio-economic system under study.

Numerous features of both models (summarized in box 5) were similar. First both models are dynamic in order to capture the complex mechanisms, processes and drivers at play. In particular, the multi-species and multi-fleet (or fishing strategies) dimensions of mixed fisheries are accounted for in both the BoB and the NPF models, which was required in order to address the issue of technical interactions. Regarding the biological processes, both models were structured in age (BoB) or in size (NPF) according to the biology of the key modelled species. Age- (or size-) structured bio-economic models provide a more accurate evaluation of the nature of technical interactions as well as of the gross incomes from landings (which are usually affected by the size structure of the landed catch). Another common complexity related to the non-linearities underlying the growth of species. Recruitment was assumed to be related to spawner stocks, however due to differences in assessments, data availability and biology, different assumptions were adopted in the two case studies. Recruitment of BoB species was modelled assuming a segmented function where recruitment decreases for spawning stock biomasses below a certain level (this function is either called hockey stick or Ockham Razor function (Barrowman and Myers, 2000; Mesnil and Rochet, 2010)). This assumption is supported by empirical evidence, across taxonomic groups, that recruitment tends to be poorest when spawner abundance is low (Myers and Barrowman, 1996). In contrast, Ricker stock recruitment relationships were assumed in the NPF model as in Dichmont et al. (2003) and Punt et al. (2010).

The decisional and management viewpoints adopted in the thesis, and the associated papers, focussed on fishing effort controls and constituted another common ingredient of the models and another aspect of their complexity as multi-fleet input controls were taken into account in both cases. In this same vein, the use of projections and scenarios over rather large temporal horizon (20-30 years), as compared to the unit-scales (1 year, 1 week) of the dynamics, brought another element of complexity to the modelling, and highlights the multi-scale (temporal) perspective of the work. Economic features were also incorporated in both models through production, price and cost functions, which enabled the evaluation of economic indicators relating to fisher, industry and societal concerns, and further contributed to the common complexity of the models.

The analyses in this thesis also captured processes that were specific to each fishery (as summarized in box 5). In particular, the bio-economic models developed used different time-steps reflecting the different biological characteristics of the two marine systems. Prawns are a short-lived species (i.e. 1-2 years life cycle), and the NPF model was expressed on a weekly basis. At a weekly time step, intra-annual and seasonal biological processes could be represented, including spawning and recruitment, and economic processes such as seasonal allocation of effort, as advocated in Anderson and Seijo (2010). The BoB model structure and parameters were developed on an annual basis as in Macher et al. (2008) based on the ICES yearly[1] stock assessments. The life cycles of the species in this fishery are longer than those of tropical prawns and a shorter time step would have added unnecessary complexity to the model. An adaptive and rather complex effort allocation process was also included in the NPF model to represent the specific current fishing strategy between the predictable and less predictable prawn resources. This representation of the temporal effort allocation process between fishing strategies, allowed account to be taken of relevant effort allocation patterns at the scale at which the analysis was being performed. Analyses in the BoB fishery and the NPF included constraints representing the aim to conserve target species spawning stocks and to maintain economic profits. However, while the economic effects of by-catch and other targeted species were only accounted for indirectly in the BoB case study through the gross revenue from other species catches, sea snake catches related to fishing effort were explicitly included in the NPF model. Among all the threatened, endangered and protected (TEP) species caught by the NPF, sea snakes were chosen as a proxy to assess impacts of trawling on broader biodiversity, sea snake catches appearing to be significantly correlated to the effort of tiger and banana prawn sub-fisheries.

There is increasing awareness of the need to more fully take the complexity of resource management problems into account (Pahl-Wostl, 2007). This thesis showed that this can be done using integrated modelling, in a way that includes only the necessary levels of complexity, while also being based on a close representation of reality (as displayed in the calibration graphs in appendix B). This approach of modelling only the necessary complexity is related to the work of Plagányi

[1] Knowledge on BoB species intra-annual biological dynamic is poor, although a quarterly model in length has recently been developed for Hake.

et al. (2012) which described and reviewed 'Models of Intermediate Complexity for Ecosystem assessments' (MICE) that focused on management actions on short timescales. MICE are context- and question-driven and limit complexity by accounting for only the components of the ecosystem needed to address the main effects of the management question under consideration (Plagányi et al., 2012).

6.1.2 Uncertainties

The bio-economic models examined in this thesis account for uncertainties underlying both ecological and economic processes. More generally, the management of fisheries is exposed to a wide range of uncertainties Garcia (1996); Hilborn and Peterman (1996); Francis and Shotton (1997). Among these are uncertainty in the size, composition, and spatial distribution of stocks; uncertainty in stock dynamics, especially in recruitment; and variations in economic parameters, such as costs and prices (Francis and Shotton, 1997; Jensen, 2008; Fulton et al., 2011).

Inter-annual recruitment variability is an important form of natural variability in fisheries (Francis and Shotton, 1997). Two different sources of recruitment uncertainty can be identified: demographic and environmental stochasticities. These correspond to unpredictable environmental factors that affect the fecundity of some (in demographic stochasticity) and sometimes all (in environmental stochasticity) individuals in a population (Lande, 1993). Recruitment uncertainty of *Nephrops*, Hake and Sole were simulated in the BoB case study with an uniform distribution relying on historical time series of recruitment. This is a simplified way to integrate uncertainty in recruitment estimations when refined data on recruitment processes are not available. Alternatively, when enough data are available, it is possible to simulate the environmental influence on recruitment through environmentally driven temporal correlation as was modelled in the NPF for tiger and blue endeavour prawn recruitments.

In addition to environmental stochasticity, fisheries are subject to economic uncertainty with strong influences from both internal and external factors. This is particularly the case for fuel price. The fuel price scenario assumed in the BoB case relied on projections from the International Energy Agency (IEA, 2010; CAS, 2012), while the fuel price scenario in the NPF case was based on historical data regression. A common increasing trend in fuel price is assumed in both cases, however the scenarios are different, as the case studies relate to different contexts in terms of factors such as national policies and taxes, and are not affected in the same way by worldwide influences. Market prices for fish assumed in this study were based on historical data for both fisheries. However, uncertainties were introduced in a different way, as market prices in the two fisheries are not impacted in the same way by external and internal factors. Uncertainties on annual mean market price in the BoB fishery were introduced through a random mean price by species following a Gaussian distribution calibrated from ex-vessel prices for the 2000-2009 period. The mean prices were assumed to be independent by species and by year, but also independent of the landings. Experts in the fishery have attempted to establish econometric relationships that take into account different drivers influencing prices (such as landings structure and quantity, national landings and imports). However up until now, robust models of such correlations have proved difficult to establish, therefore it seemed preferable in this analysis to simply introduce price uncertainty, with no particular assumptions as to the drivers of price variability. On the other hand, except for banana prawns, the main market for NPF prawns is Asia, and the price is largely dependent on the Yen-AU$ exchange rate and the total international supplies to the market (Punt et al., 2010). The assumed trends affecting prawn prices in the NPF have been observed in other wild prawn or shrimp fisheries around the world (Gillett, 2008). A fall of shrimp prices has been observed in Australia, and in the United States, Indonesia and Nigeria to cite some (Gillett, 2008). Some of the downward pressure on prices on captured shrimps comes from the increasing amount of farmed shrimp on the world market, especially whiteleg shrimp (*Litopenaeus vannamei* from China (Gillett, 2008). These trends are out of the control of local industry and management. Prawn prices were thus assumed to be independent of the NPF landings, and scenarios were based on projections from historical trends; they did not include price variability.

Results showed that the biological variability integrated in both models had an important impact on the economic outcomes. In the BoB fishery, this is especially true for the sub-fleets that are strongly dependent on the *Nephrops* (*Nephrops* trawlers) and on the Hake (large various fish gill-netters). The variability in annual profits in the BoB fishery results from the combined effects of the biological and market prices uncertainties. Regarding the NPF, the variability in annual profits stems from the biological variability; in this case and when considering the large range of possibilities for annual profits, it is clear that biological variability has a crucial role for bio-economic risk assessments. Furthermore based on the most likely economic scenarios identified in this work, it has been possible to highlight, in both case studies, the potential of future economic risks for both fisheries.

Box 5: Comparison of the BoB and NPF bio-economic models.

The table 6.1 summarizes the main features of the bio-economic models developed in this thesis for the BoB and the NPF fisheries. When components differed between chapters, only the most detailed representation in each of the two case studies is represented in the table.

Table 6.1: Features of BoB and NPF bio-economic models.

		Bay of Biscay demersal fishery	**Northern Prawn Fishery**
Common complexity	Multi-species	3 with population dynamics	3 with population dynamics, 2 without population dynamics
	Dynamic structure	Age	Size
	Stock-recruitment relationship	Hockey stick function (or also called Ockham Razor, i.e. double linear recruitment curve)	Ricker
	Multi-fleet	4 main fleets split into 16 sub-fleets according to vessel length classes + one 'other fleet'	2 sub-fisheries with one of the sub-fishery separated in two fishing strategies
	Input control	Number of vessels by sub-fleets	Number of vessels and annual effort per vessel
Specific complexity	Time-step	Annual	Weekly
	Effort process		Adaptive effort allocation process
	Other species	Indirect economic effects of other species catches	Direct impact of fishing on by-catch
Uncertainty	Environmental	Uniform distribution relying on historical time series of recruitment	Environmentally driven temporal correlation
	Economic	Stochasticity on market prices	Prawn prices scenario based on historical trends
		Fuel price scenario based on International Energy Agency projections	Fuel price scenario based on historical trends

6.1.3 The role of data in integrated models

The integrated modelling undertaken in this thesis required a wide range of data spanning biological, ecological and economic domains. Due to their economic importance, both fisheries have been studied for a number of years and have been subject to the collection of economic and biological information through targeted surveys.

The parameterization of the BoB model was based mainly upon the assumptions made by the Working Group on the Assessment of Southern Shelf Stocks of Hake, Monk and Megrim (WGHMM) of ICES which can be considered as the best source of knowledge about the dynamics of *Nephrops*, Hake and Sole stocks in the Bay of Biscay. Fleet economic data relied on information from the French Ministry of Fisheries and Aquaculture and from the Fisheries Information System (SIH) of Ifremer. The bio-economic model parameter values, recruitment and costs functions in the NPF relied on estimations and assumptions made by the Commonwealth Scientific and Industrial Research Organisation (CSIRO) and the Australian Bureau of Agricultural and Resource Economics and Sciences (ABARES), which can be considered as the best source of knowledge concerning the fishery. The modelling work in this thesis was only possible thanks to the availability of this wide range of biological and economic data in both contexts.

Such availability of biological and economic data (including time series data for four decades in the NPF case) is not often encountered in marine fisheries. Where it is not, developing the type of integrated modelling proposed in this thesis will often imply a need for the reinforcement and integration of marine information systems related to the marine fisheries for which the models are being developed.

6.2 Viability analyses and bio-economic risk

Stochastic co-viability analyses (CVA) were applied to both case studies to investigate how the fisheries can operate within a set of constraints relating to multiple management objectives in uncertain contexts. The issue of threshold and confidence level definitions in viability analyses were found to be important, especially in light of the outcomes from the mean-variance analyses (chapter 4) investigating the balance between expected economic performance and economic risk in mixed fisheries management.

6.2.1 Multiple objectives and potential conflicts

Fisheries management is often characterised by multiple and sometimes conflicting objectives (Crutchfield, 1973; Charles, 1989). Viability analysis provides a formal approach to examine this aspect of fisheries management (Bene et al., 2001; Eisenack et al., 2006; Doyen et al., 2007; Martinet et al., 2007, 2010). The multiple criteria analysed in this thesis related to biological, socio-economic and biodiversity conservation (chapter 5) management objectives (as summarized in box 6).

In the analyses presented, an important objective related to economic viability (EVA). The definition given to this objective in the applications to both the BoB fishery and the NPF was to maintain a minimum level of annual profits of the fleets. A more standard economic objective could have been that of reaching the dynamic MEY at the fishery level, which could involve trading current losses/gains for future gains/losses. Defining the economic constraints based on annual profits meant that an equal value was placed on profit levels in all years, implying that losses in any particular year cannot be compensated by gains in another. This specification captures some of the social and political dimensions involved in determining acceptable management strategies in practice. In addition to this inter-annual equity, another social dimension was integrated in the BoB analyses through the objective of maintaining the activity for all sub-fleets, even the less efficient ones. The sub-fleets are spatially distributed along the coast of the Bay of Biscay. This objective implied the maintenance of the geographical distribution of fishing activity and the associated territorial impacts. Such an objective was not seen as relevant in the NPF, which is considered as a single industry: NPF Industry Pty Ltd, a collective of trawler operators, processors and marketers acting together as a single voice for the industry in the Northern Prawn Fishery. Moreover, given that the recent levels of profit in the NPF were relatively close to those that have been estimated to generate dynamic MEY (Kompas et al., 2010), the objective of maintaining recent annual profit levels was considered to be closely aligned with that of MEY in this fishery.

Simulation results showed that biological and socio-economic constraints are not in conflict in the Bay of Biscay demersal mixed fishery analyses. This is due to the fact that to increase the socio-economic viability, reductions in fishing capacity are needed, which increases biological viability. In the NPF, economic and target stock conservation constraints were also not in conflict, but the biodiversity conservation constraint was in conflict with the economic constraint over most of the control space. This reflects the fact that to enhance the economic viability, some management strategies led to increases in fishing effort, and especially increases in tiger prawn sub-fishery fishing effort which conflicts with respecting the biodiversity conservation constraint as it was defined.

6.2.2 Sustainability and bio-economic risk

Stochastic co-viability analysis (Baumgärtner and Quaas, 2009; De Lara and Martinet, 2009) was used in this thesis to address issues of risk and precaution and to determine the ability of the BoB fishery and the NPF to achieve specified sets of sustainability objectives, with a sufficiently high probability.

Revenues from fisheries may vary greatly from year to year owing to natural variation in fish stocks (Kasperski and Holland, 2013) that cannot be predicted with any reliability, leading to varying levels of economic risks for fishing operators (Sethi, 2010). The mean-variance analyses for the NPF (chapter 4) indicated a strong trade-off between mean expected economic performance and the associated inter-annual variance. These analyses have also shown that the balance between expected economic performance and economic risk is an issue for management. Fishery risk aversion reflects the fact that industry stakeholders usually value stability over time. Therefore management strategies with reduced inter-annual bio-economic risks may be favoured. But this implies loss in potential performance, as certain strategies that could entail higher levels of economic performance (but with greater variability) would then be excluded as viable options.

The question of which threshold and confidence level to choose remains a crucial issue in viability analyses, especially in contexts where environmental and economic uncertainty coexist. For instance under a most likely economic scenario, adaptations of the NPF fishery to maintain current levels of economic performance are likely to depend on the extent to which operators in the fishery accept higher levels of economic risks. This could be re-interpreted in terms of economic thresholds: a lower economic threshold would lead to some trajectories being economically viable when they would not have been with a higher threshold. It could also be interpreted in terms of risk management through the definition of the confidence level that is required for a management strategy to be considered viable. Indeed a lesser level of confidence (for instance, 90% instead of 95% or 100%) would lead to the conclusion that the fishery is (socio-) economically viable when this would not have been the case with a higher confidence level.

Similarly, biological viability assessments depend on biological threshold definitions (summarized in box 6). In the BoB fishery, while biological thresholds were set to biomasses of precaution estimated by ICES working groups for Hake and Sole, the threshold used in this study for the *Nephrops* was the minimal historically observed spawning stock biomass. Similarly, in the absence of biomass of precaution assessments, the biological thresholds for the NPF were set to their minimal historical levels. These levels can be considered like limit reference points, more than precautionary reference points. Even if a more precautionary approach might lead to select a higher level for these thresholds, historical records show that the stocks did recover from their lowest levels in the past. Therefore maintaining their SSB above this minimal historical level could be expected to allow avoiding stock collapses, as long as the broader ecological conditions in which the stocks occur are maintained. NPF simulations showed that similar difficulties may exist with respect to setting the thresholds for ecological impacts, given the variability of these impacts. In the analysis presented, the choice of the 2010 sea snake catch level as ecological threshold combined with variability in sea snake catches, could be considered 'extreme', as it implies that any simulated sea snake catch would have to remain below the 2010 catch.

Box 6: CVA applied to sustainable management of the BoB fishery and the NPF.

The table 6.2 summarizes the different components governing the co-viability analyses applied to the BoB and NPF case studies. For the sake of simplicity, only CVA components from chapter 3 (BoB) and 5 (NPF) are summarized here.

Table 6.2: Components of the co-viability analyses applied to the Bay of Biscay demersal mixed fishery and the Northern Prawn Fishery.

		Bay of Biscay demersal fishery	**Northern Prawn Fishery**
Management objectives	Biological	Preserving SSB of *Nephrops*, Hake, Sole	Preserving S of tiger and blue endeavour prawns
	(Socio-) economic	Maintaining positive profits for each sub-fleet	Guaranteeing minimum level of annual profit for the whole fishery
	Ecological		Reducing sea snakes catches
Thresholds	Biological	Biomasses of precaution for Hake and Sole, and minimum historical SSB for *Nephrops*	Minimal historical S
	(Socio-) economic	Zero	60 % of the estimated profit in 2010
	Ecological		Estimated sea snakes catches for 2010 (not to exceed)

The difficulty in setting viability thresholds in our analyses emphasizes the need for better ecological information on the basis of which appropriate thresholds could be identified.

6.3 Fishing strategies and co-viability drivers

The modelling approach proposed in this thesis allowed direct comparison and assessment of biological, socio-economic and ecological outcomes and of the global sustainability performance of various fishing strategies, including status quo strategies (i.e. maintenance of fishing management settings of a reference year), single-stock management (BoB) or catch specialisation (NPF) strategies, but also catch diversification strategies (NPF) or management strategies defined for the mix of species and sub-fleets as a whole (BoB), including strategies aimed at maximizing the net present value of the fisheries.

6.3.1 Status quo: socio-economic viability at stake

Current management for both the BoB fishery and the NPF was analysed to identify if these mixed fisheries might be sustainable, given the biological, economic and biodiversity sustainability criteria identified in the analysis.

Based on the model calibrations and assumptions, BoB simulations showed that a status quo strategy – consisting of maintaining fishing effort of all sub-fleets at their 2008 baseline levels – may not be socio-economically viable given the set of parameter values and viability constraints adopted, which required guaranteed positive profits for each sub-fleet. While most of the sub-fleet profits were positive, profits of some sub-fleets (e.g. large various fish gill-netters) were negative for at least one year of the simulation for each recruitment and price scenario; and in a most likely economic scenario, the annual profits of (mainly) trawlers dropped dramatically due to their fuel dependency. Furthermore, based on the simulations and assumptions of the BoB model, results indicated that, given the status of the species and fleets in 2008, and the nature of the technical interactions between fleets through by-catch and discards, there appears to be excess capacity in the fishery as a whole, and especially in the *Nephrops* trawler fleets.

NPF simulations showed that the status quo strategy – consisting of maintaining fishing capacity and allocation of effort between sub-fisheries at their 2010 baseline levels – may also not be economically viable because of the variability that characterizes the interactions of the fishery with the ecosystem. Results emphasized important inter-annual economic variability due to biological variability. The viability assessment results arise therefore mainly from the objective to maintain inter-annual equity. Furthermore, under a most likely economic scenario, the maintenance of the current levels of profit might be strongly compromised. The NPF results showed that a reduction in the number of active vessels might help the fishery to reduce its economic risk while maintaining its current level of profit, especially under a most likely economic scenario.

6.3.2 Mono-specific and specialized strategies: bio-economic risks

Strategies relying on standard management approaches were investigated to identify and compare alternative options for the fisheries. While a set of mono-specific management strategies were explicitly investigated for the BoB fishery (chapter 3) as this has historically been the approach taken in practice, in the NPF analyses, we considered the possibility that the fleet specializes in the catch of tiger prawns, banana prawns, or a combination of both.

In the BoB fishery, Hake is more widely distributed than *Nephrops* and Sole and should therefore be managed at a larger scale. Only two mono-specific management strategies, one for *Nephrops* and one for Sole, were investigated in the BoB study. Based on our simulations, it appeared that mono-specific management strategies improved the bio-economic status for the specific fishery and species they target, but did not achieve satisfactory results at the global scale. While the Sole management strategy achieved biological viability, it led some sub-fleets other than the Sole gill-netters to become unprofitable. The *Nephrops* management strategy produced better fishery-wide results as regards to the socio-economic performance of the sub-fleets, but still induced negative profits for some sub-fleets, as well as a moderate level of biological risk (concerning Hake and Sole viabilities). More generally, according to our criteria for sustainability, mono-specific strategies did not perform well in the BoB case, as the performance of all species or sub-fleets were not improved. In other words, mono-specific strategies did not allow meeting all the constraints defined in this study, and did not entail economic efficiency.

The NPF analyses described the risk management aspects of specialised versus diversified effort allocation strategies. Based on our simulations, it appeared that specialisation strategies (which distribute the total annual effort more towards tiger or banana prawn sub-fisheries) performed worse than strategies allocating effort between tiger and banana prawn sub-fisheries equally, or adaptively (i.e. depending on white banana prawn biomasses), when the fishing capacity of the fishery was similar to or greater than the current situation. More specifically, effort allocation strategies that specialised in banana prawns were associated with the greatest economic variability as they target a more fluctuating and unpredictable resource; and tiger specialisation strategies were associated with greater biological impacts, ecological risks and decreases in mean economic performances. However, in a context where a decrease in the number of vessels would occur, tiger specialisation strategies might perform better in terms of economic viability than other strategies. This is because a reduced fleet seems more favourable to strategies focusing on the less fluctuating and more predictable resource.

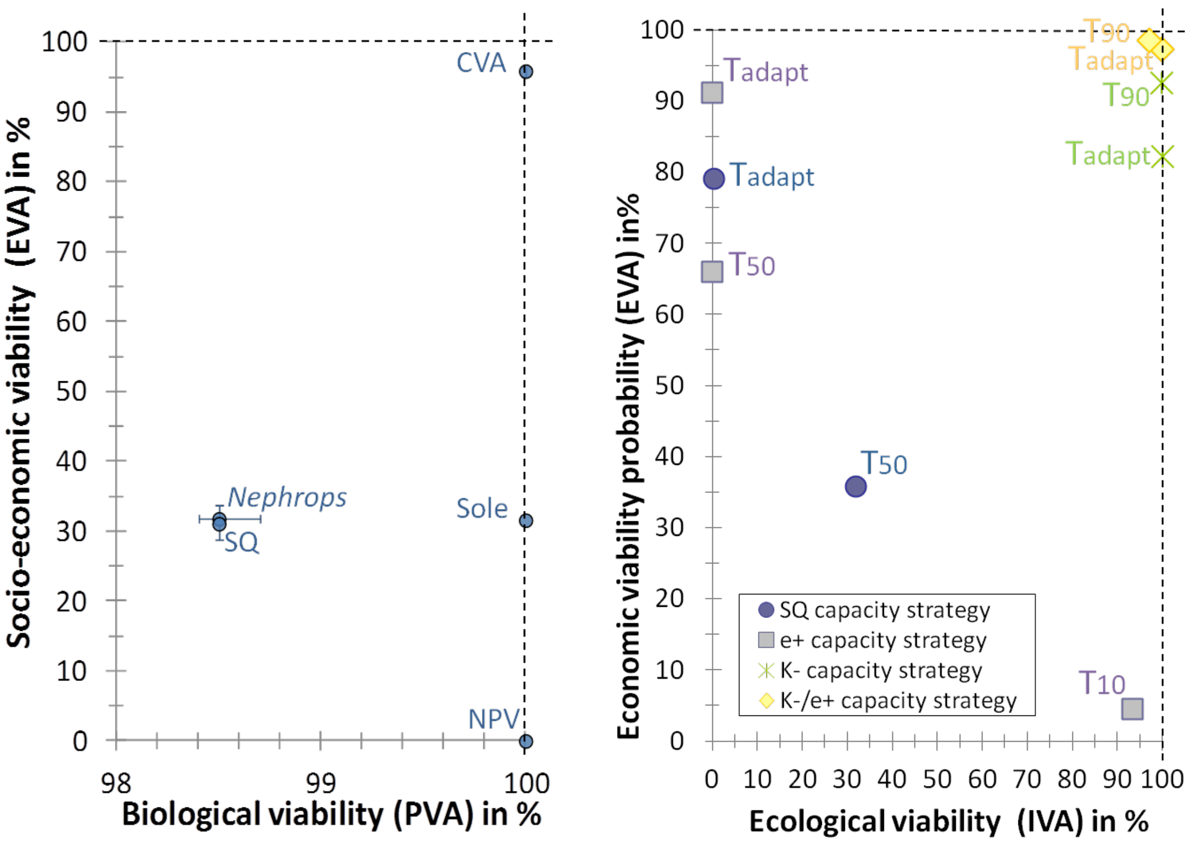
6.3.3 CVA strategies: win-win strategies exist

It appeared that for both fisheries, viable management strategies can be exhibited where all biological, socio-economic and ecological (for the NPF case) viability constraints are met with sufficiently high probabilities (superior to 95%).

Chapters 2 and 3 on the BoB fishery showed that CVA management strategies (those including multiple species and multiple fleets management) led to 'win-win' situations, where for a number of combinations of capacity adjustments among sub-fleets, both biological and socio-economic viability constraints were respected for each species and sub-fleet (figure 6.1(a)). According to these analyses, multi-species management is therefore required to guarantee viable management of the BoB fishery as a whole. This issue is being addressed under the new (European) Common Fisheries Policy in which multi-species management plans must be defined (EC, 2009). However, analyses showed that, even if managing at the fishery-level improves the fishery performances, there are distributional impacts in optimizing the system as a whole, as some fleets may need to reduce their fishing activity. Strategies such as CVA management strategies may provide a means to address these trade-offs in practice.

As demonstrated in chapter 5, it is also possible to identify a small 'bargaining space' for the NPF, in terms of fishing capacity and allocation of fishing effort, where all the biological, economic and biodiversity conservation constraints could be respected with the highest probabilities (figure

6.1(b)). This consensus is associated with a decrease in the number of vessels, as in the BoB fishery.



(a) EVA versus PVA in the BoB fishery (b) EVA versus IVA in the NPF

Figure 6.1: Economic viability probability (EVA) versus biological viability probability (PVA) in the Bay of Biscay mixed demersal fishery in (a) and versus ecological viability probability (IVA) in the Northern Prawn Fishery in (b). Sub-figure (a) is adapted from figure 3.5 of chapter 3. Sub-figure (b) corresponds to the figure 5.2(a) of chapter 5.

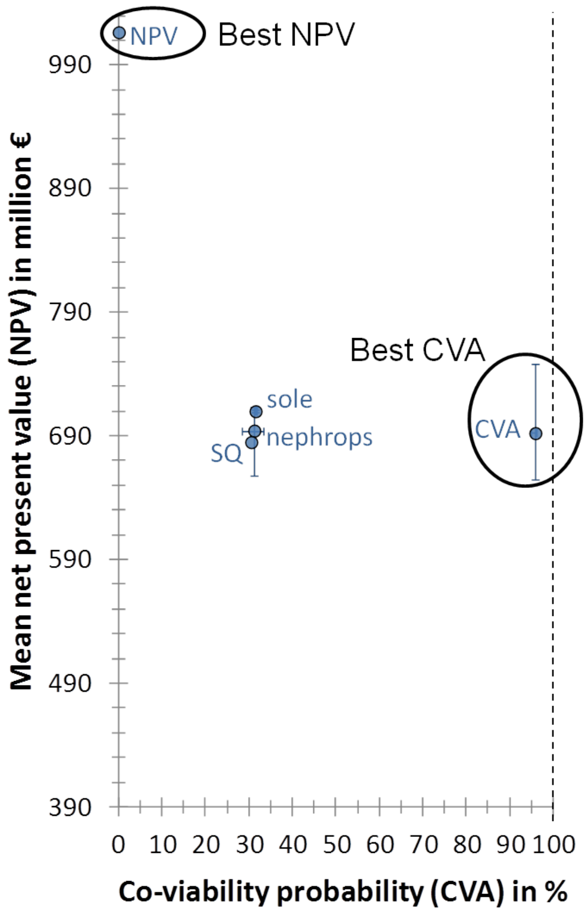
CVA management strategies provided flexibility in management settings for both case studies. CVA strategies might therefore be expected to more easily achieve consensus among multiple stakeholders involved in the fishery compared to standard optimisation management approaches which offer only a limited number of management solutions.

6.3.4 Cost of sustainability

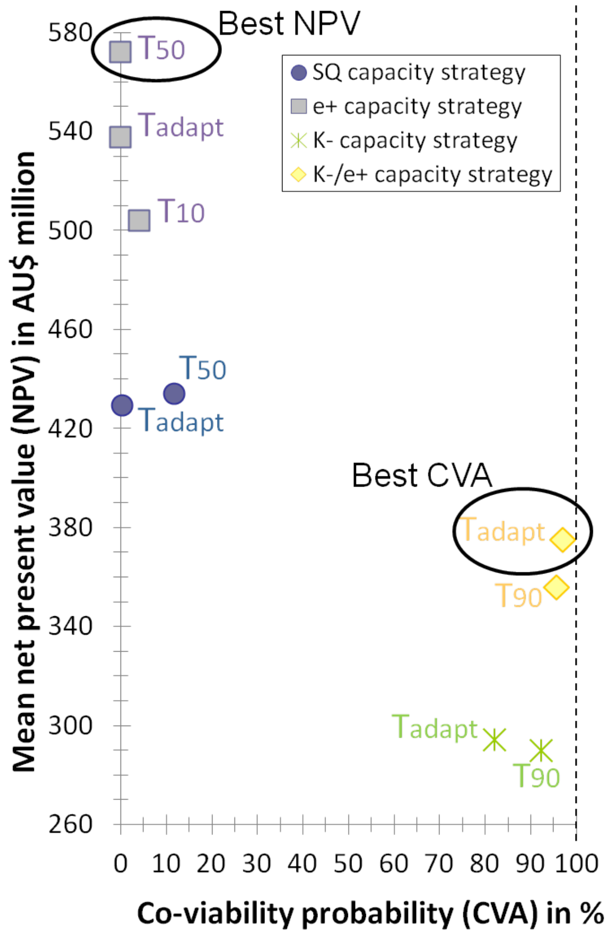
A more conventional economic strategy that maximised the NPV of the fishery was also investigated for comparisons with CVA management strategies, in terms of sustainability and mean economic performances.

For the BoB fishery, under the NPV strategy, a central planner aimed at maximizing the expected sum of discounted profits at the scale of the entire fishery, and so there was no guarantee 'a priori' that the profit of each sub-fleet would be positive due to the absence of constraints on these profits. This strategy did improve the global economic performance of the fishery, but led to heterogeneous impacts on the sub-fleets, including stopping some from fishing. Resistance may develop to such a management strategy from the segments and regions of the fishery which would be negatively impacted. Based on these results, it thus appears that strategies that maximised the NPV entailed distributional impacts which may lead the strategies to be perceived as unacceptable in practice. In a general way, management strategies leading to the highest NPV, were associated with zero co-viability probabilities (due to the socio-economic constraint); while CVA strategies led to the highest co-viability probabilities, but with lower NPV (as shown in figure 6.2(a)). A trade-off in fishery management performance based on mean NPV and co-viability criteria was also observed for the NPF (figure 6.2(b)). In the NPF, all fishing capacity management and effort allocation strategies leading to increased NPV were associated with strongly reduced ecological and economic viability as defined in this study. The decrease in economic viability under a high NPV strategy reflects increased inter-annual variability and violation of the inter-annual-equity objective.

Overall, analyses presented in this thesis highlighted that high co-viability probabilities can be achieved, but only at the cost of reducing the economic yield compared to a strategy maximizing the NPV. This economic loss can be interpreted as a 'cost of sustainability' associated with the objective to meet all the constraints imposed on the fisheries, i.e. the opportunity cost of increasing co-viability probabilities. For the BoB fishery, an estimation of this cost can be made by calculating the difference between the mean NPV value with CVA strategies and that with a strategy maximizing NPV. Based on the assumptions in this thesis, this corresponds, to 315 million € (or 31% of the highest NPV obtained in our analyses) over 20 years. For the NPF this cost can be estimated from the difference between the highest NPV value obtained in chapter 5 (which is associated with a balanced effort allocation stragegy T_{50} combined to an increase of the effort per vessel following a e^+ fishing capacity strategy, c.f. figure 6.2(b)) and that with the management strategy leading to the highest CVA (i.e. adaptive effort allocation strategy T_{adapt} with K^-e^+ fishing capacity scenario corresponding to a decrease of number of vessels of half and maximum allowed increase of effort



(a) NPV versus CVA in the BoB fishery



(b) NPV versus CVA in the NPF

Figure 6.2: Mean net present value (NPV) versus co-viability probability (CVA) in the Bay of Biscay mixed demersal fishery in (a) and in the Northern Prawn Fishery in (b). Sub-figure (a) is adapted from figure 3.6 of chapter 3. Sub-figure (b) corresponds to the figure 5.2(b) of chapter 5.

per vessel, c.f. figure 6.2(b)). The ‘cost of sustainability’ for the NPF is thus estimated in this thesis to be around AU$ 197 million (or 35% of the highest NPV obtained in our analyses) over 10 years. These ‘ costs of sustainability’ cannot be directly compared between the two fisheries, as NPV were not calculated in the same way, for the same period of time, with the same discount rate values and for the same objective thresholds. However, it is interesting to consider the percentage of the highest NPV values - which could be obtained - that they represent. Based on the assumptions and constraints defined in this thesis for the BoB fishery and the NPF, it appears that respecting all constraints considered in this study may be relatively comparable in terms of costs for both fisheries, even if the biodiversity conservation objective in the NPF study added a constraint which the fishery must meet to be assessed as viable.

Box 7: Conclusions from the BoB fishery and NPF management strategies evaluated.

Table 6.3 summarizes the results from bio-economic modelling and stochastic co-viability analyses for the Bay of Biscay demersal mixed fishery and the Northern Prawn Fishery sustainable management.

Table 6.3: Results for the BoB fishery and the NPF management.

Trade-offs among objectives	Synergy between biological and socio-economic constraints	Economic and biodiversity constraints in conflict
Current fishing settings (SQ)	Biologically viable, not socio-economically viable	Biologically viable, while not economically and ecologically viable
Alternative management strategies	Mono-specific management strategies: around 30% of EVA	In a general way: CVA of diversification effort allocation strategies > CVA of specialized strategies
	CVA strategies: more than 95 % of EVA	Highest CVA: PVA=100%, EVA=98.5% and IVA 99.88%
	NPV strategy: 0% of EVA (closure of some sub-fleets not consistent with social constraints)	Highest NPV: 66% of EVA and 0.01% of IVA

6.4 Perspectives

6.4.1 Implications of other species and repartition in stocks

The models developed for this thesis account for a limited number of the species occurring in each of the case study fisheries. In both case studies, data availability was the main determinant of which species were integrated in the analyses. For instance, Anglerfish (*Lophius piscatorius* and *L. budegassa*) is an important species for the BoB fishery in terms of landing value (13% of the French national landing value in 2008) and, while initially integrated in the chapter 3 study, data reliability issues[2] resulted in this species removal from the study. However, new assessments of this species will soon be available and the integration of Anglerfish in the BoB analyses will allow a more comprehensive representation of the fishery. Furthermore, the explicit integration of Spanish and Belgian fleets in the analyses would also allow a more detailed analysis and discussion about the distributional aspects of management in the BoB.

In the case of the NPF, other species targeted by the fishery are increasing in economic im-

[2]In 2007, the ICES working group WGHMM rejected the eXtended Survivor Analysis (XSA) age based assessments of Anglerfishes because of data quality (increased discards not incorporated) and ageing estimation problems were clearly identified.

portance and it may become more important to consider them in future modelling, as the value of the prawn catches decreases (under a most likely economic scenario). For instance, accounting for commercial by-products such as bugs (*Thenus sp.*) could become an important aspect of developing alternative strategies for the fishery. Moreover Dichmont et al. (2008) distinguished multiple prawn stocks distributed across different regions in their operating model. Five stocks are therefore identified with different biological and catchability parameters. Tiger and blue endeavour species occur in three of the five regions, whereas two of the easternmost regions contain only three species, with grooved tiger prawn absent from one area and brown tiger prawn from the other. White banana prawn is also not spread homogeneously. Repartitioning stocks accounting for their spatial distribution could lead to a more accurate analysis and may be especially important for predicting economic consequences where fishing costs (especially those related to fuel cost) are variably impacted. Stock repartitioning may also affect predictions of biological and ecological consequences through its effect on the aggregation of effort and associated impacts on stocks, benthos and habitats.

6.4.2 Toward Ecosystem-Based Fishery Management

As result of the World Summit on Sustainable Development (Johannesburg, 2002), the pressure on governments to introduce a form of fisheries management that takes into consideration the effects of fishing not only on the targeted species but on the whole ecosystem further increased. In this context, Ecosystem-Based Fishery Management (EBFM), which aims to sustain healthy marine ecosystems and the fisheries they support, is a new direction for fishery management, essentially reversing the order of management priorities so that management starts with the ecosystem (including state of the system, habitat, protected species, and non targeted species) rather than one target species (FAO, 2003).

The analyses in this thesis were carried out as first steps towards an EBFM approach to the bio-economic modelling of the two fisheries. The impacts of trawling on the broader biodiversity in the NPF were accounted for through catches of sea snakes. However, other impacts are induced by trawling, including impacts on benthos communities and habitats. In addition, due to variations in gear deployment methods between tiger and banana prawn sub-fisheries, the amount of by-catch also varies considerably by sub-fishery. One way to assess more accurately the separate effects from tiger and banana prawn sub-fisheries would be to incorporate benthos species modelled as two distinct groups: sessile and mobile benthos. However, this model extension would be relevant only if a spatial dimension was also been added to the model, indeed Dichmont et al. (2008) and Bustamante et al. (2010) demonstrated that effects of trawling on benthos are dependent on the aggregation of effort, suggesting that global effects at the scale of the fishery are very small compared to local effects. Accounting for the spatial aggregation of effort would make meeting the ecological constraint more difficult, especially for the banana prawn sub-fishery which has a more important aggregation level than tiger prawn sub-fishery. Were the trade-off between the amount of fishing effort and the aggregation of this effort accounted for in the analyses, the identification of viable management strategies would be improved.

Extending the NPF analysis to incorporate benthos communities would also enable exploration of the trophic interactions between prawns and some mobile benthos species that are themselves predators of certain sessile benthos species. Mullon et al. (2004); Cury et al. (2005); Chapel et al. (2008) and Doyen et al. (2007) showed how the viability approach can be associated with the integration of ecosystem considerations, as trophic interactions. An applied co-viability study of the management of the small-scale coastal fishery in French Guiana (Cissé et al., 2013) accounted for Lotka-Volterra trophic interactions among 13 species on which this commercial fishery relies. This study is included as appendix (appendix C). The study shows the value of taking trophic relationships into account in bio-economic modelling, especially when management strategies are assessed through CVA. Indeed when a fleet targets a species which is a predator to another targeted species, this could have an indirect positive effect on the prey species, therefore conclusions regarding viable management may be different with and without the consideration of trophic interactions.

Integrating trophic interactions and incorporating a spatial dimension in the analyses would thus better serve the needs of ecosystem-based approaches to the sustainable harvesting of marine biodiversity in fisheries such as the NPF.

6.4.3 Climate change and its implications for NPF and BoB modelling

As seen in this thesis, white banana prawn dynamics are highly unpredictable and mainly driven by environmental factors. Furthermore this high variability has been shown to have economic consequences and thus is of relevance to management. The ability to accurately model banana prawn biomasses is challenging but will be crucial for NPF management. Given the importance of environmental drivers, potential impacts of climate change should be accounted for in future extensions of these analyses. The frequency and intensity of cyclones are predicted to increase under climate change scenarios. Climate change projections for rainfall are highly uncertain; however, rainfall is projected to decrease across parts of northern Australia with some areas showing a slight increase. These increases may have a positive impact on white banana prawn catches (Hobday et al., 2008). Integration of 'regime switch' scenarios, where the frequency of high rain events (i.e. modelled through higher probabilities to have important white banana prawn biomasses) varies quite radically, would be a valuable extension of the work presented in this thesis. Climate change may also have a negative impact on seagrass beds and mangrove forests, which are important nursery grounds for tiger and banana prawns, respectively (Sands, 2011). Furthermore, as pointed out by (Thébaud and Blanchard, 2011), changes in the physical environment of fish stocks, in particular in the context of global warming, can impact population distribution and dynamics (recruitment, growth, reproduction, and mortality), hence the structure of fish assemblages and of landings derived from their exploitation. Modifications of the components of marine ecosystems exploited by fishing fleets, due to both direct and indirect effects of fishing, are re-enforced by climate change, and can lead to different biological and socio-economic conclusions regarding viability of fisheries. Therefore including climate change scenarios in the Bay of Biscay case study would also provide insights into sustainable management of the BoB demersal mixed fishery under different possible climate futures.

6.4.4 Towards output control variables

One of the major issues confronting Bay of Biscay fisheries management is how to reach MSY for the individual species within a constant Total Allowable Catch (TAC) (STECF, 2011). Therefore analyses with quota entitlements as the control variables, instead of fishing capacity, would be relevant. In this perspective, the co-viability approach would be used to generate a set of quotas by sub-fleet (instead of effort multipliers) that are consistent with MSY and socio-economic objectives or constraints, and which take into account the reallocation of effort abilities by sub-fleet. Similarly while the NPF has historically been controlled by input regulation measures, its management is moving toward Individual Transferable Quota (ITQ), therefore co-viability analyses involving quotas as control variables (Péreau et al., 2012) could be used to support future management assessments for the NPF.

6.4.5 Setting maximum admissible variabilities

Risk aversion is an important dimension of the evaluation of management strategies. The analysis presented in this thesis demonstrated that a degree of risk aversion seems to characterize the NPF. Furthermore, there are current constraints on maximum inter-annual variation of TACs of 15% in the Bay of Biscay (STECF, 2011), which is advantageous to fishers in planning future strategies and investments. Even though economic risk aversion was not explicitly accounted for in the BoB and NPF viability analyses, the viability approach, with biological and economic variability, provides a way of incorporating risk attitudes towards the resulting economic outcomes. More specifically, an assumption of risk aversion would lead to selecting high thresholds for economic indicators.

One way to integrate explicitly this aversion to risk in viability analysis would be to add a constraint on the inter-annual variability of annual profits (or indirectly, of the control variables). For instance, for a system to be socio-economically viable, one could require that the profit trajectories should be maintained above a pre-defined threshold, and the associated inter-annual variability should be maintained below a minimum level. This could be a way to explicitly integrate a societal concern for economic stability over years.

6.4.6 Capturing fleet dynamics

The inclusion of human behaviour in models of marine resource use is increasingly considered as a key challenge for modellers to address (Fulton et al., 2011). Fisher's risk preference is, for instance, a major determinant of their responses to various changes in fish stocks, markets or weather conditions (Mistiaen and Strand, 2000). Therefore, it is important to integrate fisher's risk preference by modelling their decision-making behaviour (Nguyen and Leung, 2009). The behaviour of fishers in the Bay of Biscay in the short-term ,as they switch between metiers during the year, can have a subsequent impact on stocks. An interesting extension of the BoB analyses would be to integrate such short-term behaviour; for instance by bringing together multi-agent models, game theory and viability approach as in Bendor et al. (2009) and Doyen and Péreau (2012).

Furthermore, 'human response uncertainty' includes a dimension which is often overlooked that relates to the implementation of management measures (Fulton et al., 2011). Economic, social and cultural drivers can cause fishers to act in unanticipated ways, which can in turn undermine the intent of management actions. The uncertainty generated by unexpected behaviour is critical as it has unplanned consequences and leads to unintended management outcomes. This means that even if viable management decisions are taken, their implementation can be uncertain (Hennessey and Healey, 2000). Accounting for implementation uncertainties will be therefore crucial for fisheries management (Angel et al., 1994). Extending this work to make direct recommendations to fishery managers requires that models be built and incorporated within the viability framework such that viability probabilities account for implementation uncertainties, as it is generally accounted for in Management Strategy Evaluation (MSE). MSE is indeed explicitly designed to identify fishery rebuilding strategies and ongoing harvest strategies that are robust to uncertainty and natural variation (Holland, 2010).

Stochastic co-viability analyses as studied in this thesis identified a 'bargaining space' for stakeholders to reach some consensus, rather than single choices. Therefore implementation uncertainties might be reduced within a stochastic co-viability management framework. This can be very important practically, and particularly in an EBFM context which involves multiple stakeholders and multiple objectives that need to be simultaneously examined if the decision-support tools are to be considered useful.

List of Acronyms

ABARES	Australian Bureau of Agricultural and Resource Economics and Science
AFMA	Australian Fisheries Management Authority
BoB	Bay of Biscay demersal fisheries
BRD	By-catch Reduction Device
CMO	Crew member observers
CSIRO	Commonwealth Scientific and Industrial Research Organisation
CVA	Co-viability analysis
DCF	Data Collection Framework
DPMA	Direction des Pêches Maritimes et de l'Aquaculture
EBFM	Ecosystem-Based Fishery Management
EC	European Commission
EVA	Economic viability analysis
FAO	Food and Agriculture Organization of the United Nations
ICES	International Council for the Exploration of the Sea
IEA	International Energy Agency
Ifremer	French Research Institute for the Exploitation of the Sea
IPBES	Intergovernmental Platform on Biodiversity and Ecosystem Services
ITQ	Individual Transferable Quota
IVA	Biodiversity impact viability analysis, i.e. ecological viability
MEY	Maximum Economic Yield
MLS	Minimum landing size
MSC	Marine Stewardship Council
MSE	Management Strategy Evaluation
MSY	Maximum Sustainable Yield
NPF	Northern Prawn Fishery
NPV	Net present value
PVA	Population viability analysis, i.e. biological viability
SIH	Système d'Information Halieutique
SSB	Spawning stock biomass
TAC	Total Allowable Catch
TED	Turtle Excluder Device
TEP	Threatened, endangered and protected

UN United Nations

WGHMM Working Group on the Assessment of Southern Shelf Stocks of Hake, Monk and Megrim

XSA eXtended Survivor Analysis

List of Variables

In common for all chapters:

B	Biomass
N	Abundance
q	Catchability
F	Fishing mortality
E	Effort per sub-fishery (in days at sea)
e	Effort per vessel (in days at sea)
K	Number of vessels
Y	Catch (i.e. production)
p	Market price
Inc	Income
c	Cost
π	Profit
δ	Discount rate
f	Dynamic function
γ	Proportion of mature individuals (in NPF case, it concerns only the mature females)
$\upsilon_{s,a}$	weight or mass of an individual of species s of age a in BoB case
$\upsilon_{s,l}$	weight or mass of an individual of species s in size-class l in NPF case
β	confidence rate
PVA	Biological viability probability
EVA	Economic viability probability
IVA	Ecological viability probability
CVA	Co-viability probability
NPV	Mean net present value

Specific to chapter 1:

x	System: either abundance or biomass
u	control or decision

Specific to the Bay of Biscay mixed demersal fishery case study:

$d_{s,a}$	discard rate of individuals of species s of age a
φ	Stock-recruitment relationship function

SSB	Spawning stock biomass
B^{pa}	Biomass of precaution
B^{lim}	Biomass limit
u	effort multiplier (i.e. control)
V^{fuel}	Volume of fuel (in litres) used by fishing effort unit (i.e. days at sea)
p_{fuel}	fuel price by litre
τ	Landing cost

Specific to the Northern Prawn Fishery case study:

$\mathbf{X}$	Size-transition matrix
$\mathbf{Suv}$	Diagonal survival matrix
$\tilde{y}(y)$	Biological year corresponding to the year y
R	Annual recruitment
$\alpha(t)$	Fraction of the annual recruitment that occurs during the time t
α^{Rick}	Parameter of Ricker stock-recruitment relationship
β^{Rick}	Parameter of Ricker stock-recruitment relationship
$\tilde{\upsilon}_s$	average weight or mass of an individual of species s
S	Spawning stock size index
$\beta(t)$	Relative amount of spawning during time t
$\mathrm{A}_s(t)$	Relative availability of animals of species s during time t
$\mathrm{Sel}_{s,l}$	Selectivity of the fishing gear on animals of species s in size-class l
CPUE	Catch per unit effort
W	Annual vessel costs
o	Opportunity cost of capital
ϱ	Depreciation rate
ψ_v	Average value of capital by vessel
$\propto_{1+2}$	annual proportion of annual effort directed towards the tiger prawn sub-fishery

Strategy and scenario names:

SQ	Status quo fishing capacity strategy
NEP	Mono-specific management strategy focused on the *Nephrops* fishery
SOL	mono-specific management strategy focused on the Sole fishery
CVA	fishing strategy aiming at maximising the co-viability probability
NPV	fishing strategy aiming at maximising the net present value of the whole fishery

K^+	Increase of vessels fishing capacity strategy
K^-	Decrease of vessels fishing capacity strategy
e^+	Increase of effort per vessel fishing capacity strategy
K^-e^+	Decrease of number of vessels and increase of effort per vessel fishing capacity strategy
T_{adapt}	Adaptive effort allocation strategy
T_0	Extreme banana specialisation effort allocation strategy (annual proportion of tiger prawn sub-fishery effort sets to 0%)
T_{10}	Banana specialisation effort allocation strategy (annual proportion of tiger prawn sub-fishery effort sets to 10%)
T_{20}	Banana specialisation effort allocation strategy (annual proportion of tiger prawn sub-fishery effort sets to 20%)
T_{50}	Balanced effort allocation strategy (annual proportion of tiger prawn sub-fishery effort sets to 50%)
T_{80}	Tiger specialisation effort allocation strategy (annual proportion of tiger prawn sub-fishery effort sets to 80%)
T_{90}	Tiger specialisation effort allocation strategy (annual proportion of tiger prawn sub-fishery effort sets to 90%)
T_{100}	Extreme tiger specialisation effort allocation strategy (annual proportion of prawn sub-tiger fishery effort sets to 100%)
BC	Base Case economic scenario
ML	Most likely economic scenario

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APPENDIX

Appendix A. Genetic algorithm for optimisation

The nonlinear optimization problems raised in Chapter 3 were solved numerically using an adapted version of the SCILAB[1] 5.2.2. routine entitled 'optim_ga' which relies on a genetic algorithm[2]. Genetic algorithms (GAs) are a search procedure based on Darwinian 'survival of the fittest' theory. GAs were developed to solve optimisation problems based on the mechanics of natural selection and genetics such as inheritance, mutation, selection and crossover. The artificial implementation of the natural selection and reproduction into genetic operations have been shown to optimize design problems (Coello, 2000; Van Veldhuizen and Lamont, 2000; Fleming and Purshouse, 2002). GAs optimize by evolving or generating successive populations from an initial random population of individuals to improved populations. This type of numerical method has already been used for bio-economic purposes in Mardle and Pascoe (2000) and in Sathianandan and Jayasankar (2009) for instance.

The 'optim_ga' routine has been adapted to our problems and the figure A.1 shows the different steps of the modified algorithm.

[1] SCILAB is a freeware dedicated to engineering and scientific calculus. It is especially well-suited to deal with dynamic systems and control theory.
[2] See http://help.scilab.org/docs/5.3.3/en_US/optim_ga.html for details on 'optim_ga'.

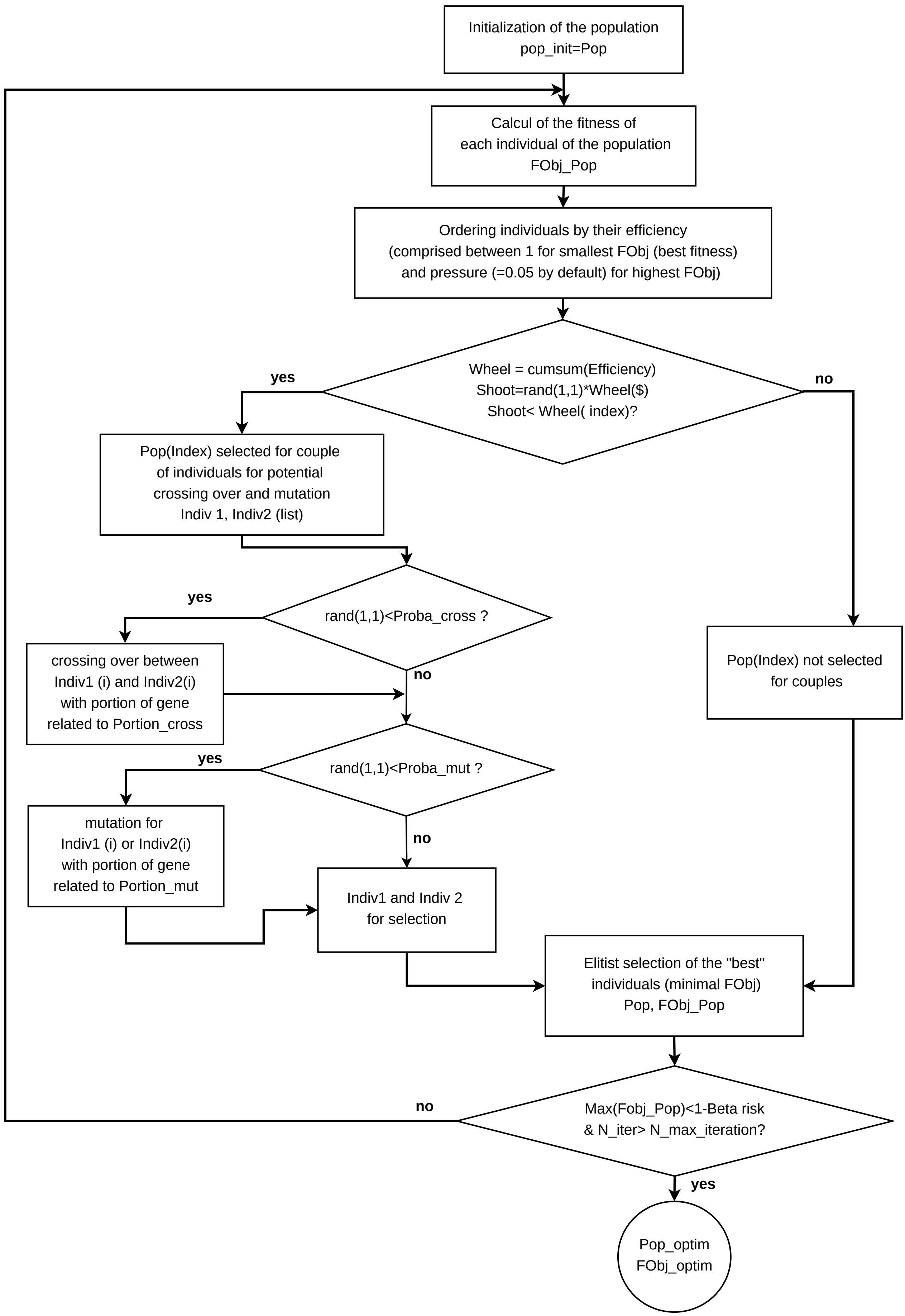

Figure A.1: Flowchart of the genetic algorithm used in chapter 3.

Appendix B. Calibration

This appendix displays come calibration graphs regarding the BoB fishery and the NPF bio-economic models.

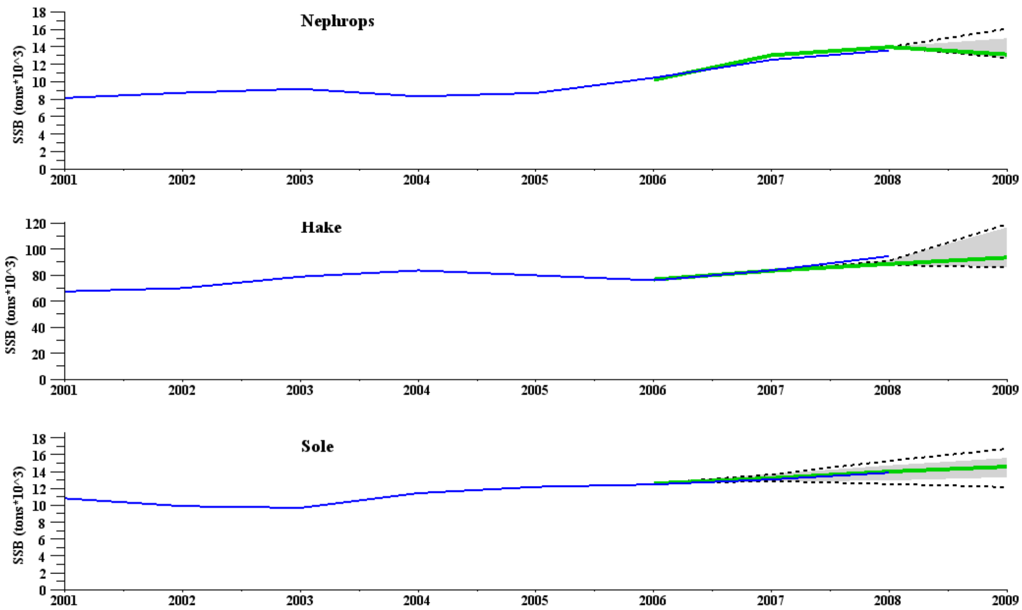

Figure B.1: Spawning stock biomasses of *Nephrops*, Hake and Sole (BoB) simulated by the bio-economic model developed in this thesis (with a status quo management strategy) and the one historically estimated. The set of possibilities simulated in the BoB bio-economic model that includes all 1000 simulated trajectories is represented by the dark dotted lines and the grey field includes 95% of the trajectories. The green line corresponds to a randomly selected trajectory among the 1000 trajectories associated to the same set of recruitments $\omega(.)$ for figures B.1 and B.2. The lines in blue represent the estimated historical SSB for each species: *Nephrops*, Hake and Sole.

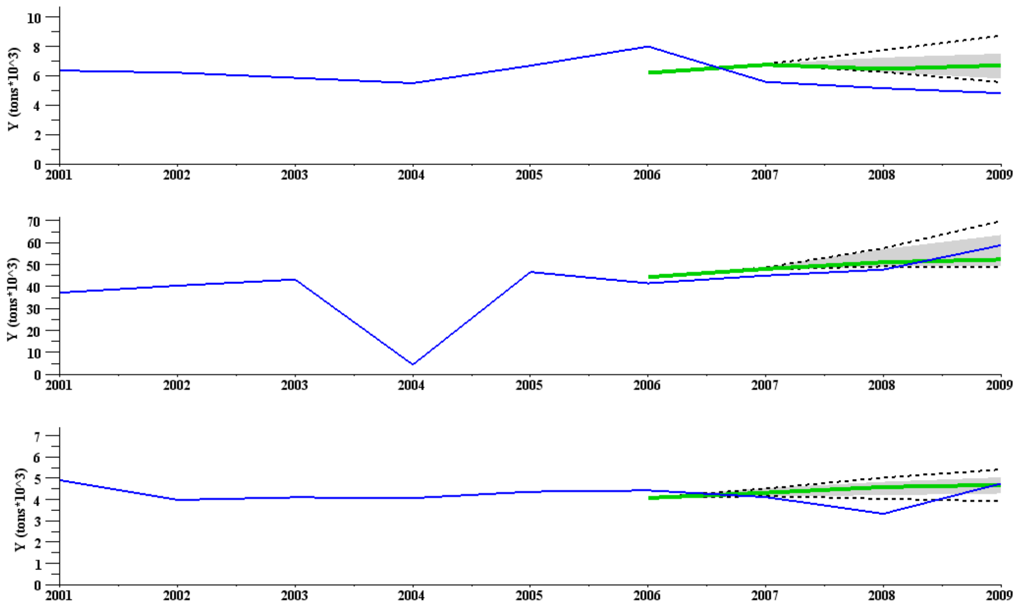

Figure B.2: Total annual catches of *Nephrops*, Hake and Sole (BoB) simulated by the bio-economic model developed in this thesis (with a status quo management strategy) and the one historically observed. The set of possibilities simulated in the BoB bio-economic model that includes all 1000 simulated trajectories is represented by the dark dotted lines and the grey field includes 95% of the trajectories. The green line corresponds to a randomly selected trajectory among the 1000 trajectories associated to the same set of recruitments $\omega(.)$ for figures B.1 and B.2. The lines in blue represent the estimated historical catches for each species: *Nephrops*, Hake and Sole.

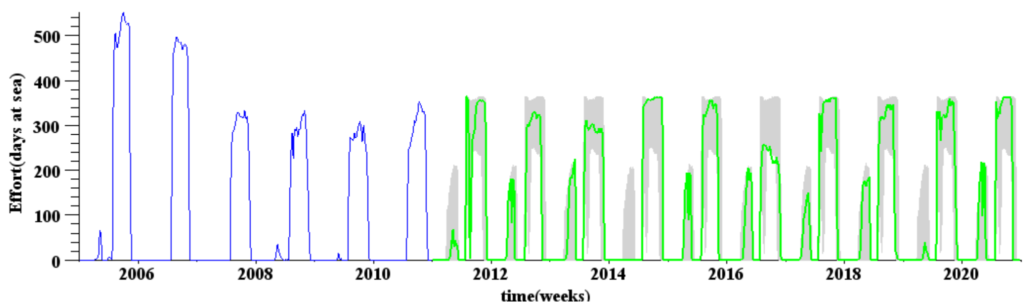

Figure B.3: Weekly tiger prawn sub-fishery effort simulated by the bio-economic model developed in this thesis (with the T_{adapt} allocation strategy and SQ fishing capacity strategy) and the one historically observed. The grey field includes 95% of the 1000 trajectories simulated by the NPF bio-economic model. The green line corresponds to a randomly selected trajectory among the 1000 trajectories associated to the same set of tiger recruitments and banana biomasses $\omega(.)$ for figures B.4, B.5 and B.3. The lines in blue represent the weekly historically observed tiger prawn sub-fishery effort.

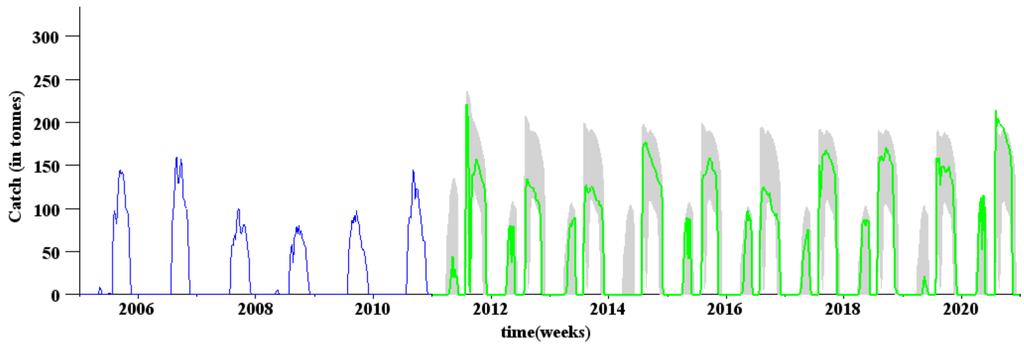

Figure B.4: Total weekly catches of tiger prawns by the tiger prawn sub-fishery simulated by the bio-economic model developed in this thesis (with the T_{adapt} allocation strategy and SQ fishing capacity strategy) and the one historically observed. The grey field includes 95% of the 1000 trajectories simulated by the NPF bio-economic model. The green line corresponds to a randomly selected trajectory among the 1000 trajectories associated to the same set of tiger recruitments and banana biomasses $\omega(.)$ for figures B.4, B.5 and B.3. The lines in blue represent the weekly historically observed tiger prawn catches.

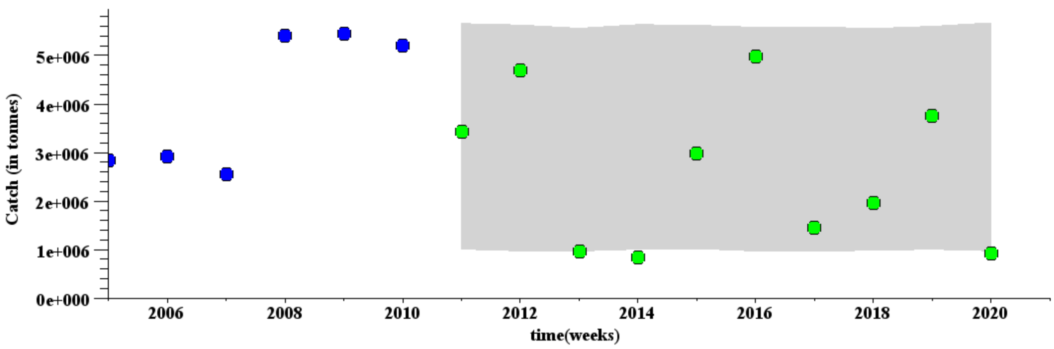

Figure B.5: Total annual catches of white banana prawns by the banana prawn sub-fishery simulated by the bio-economic model developed in this thesis (with the T_{adapt} allocation strategy and SQ fishing capacity strategy) and the one historically observed. The grey field includes 95% of the 1000 trajectories simulated by the NPF bio-economic model. The green dots set corresponds to a randomly selected trajectory among the 1000 trajectories associated to banana biomasses $\omega(.)$ for figures B.4, B.5 and B.3. The dots in blue represent the annual catches historically observed.

Appendix C. Trophic interactions and co-viability analysis

This appendix displays a published work based on the coastal fishery in French Guiana management. This study is similar to the work presented in the different chapters of the thesis, but includes trophic relationship between 13 species taken into account in the modelling.

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A bio-economic model for the ecosystem-based management of the coastal fishery in French Guiana

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ABSTRACT. This paper offers a theoretical and empirical model of ecosystem-based fishery management. A multi-species and multi-fleet model integrating Lotka–Volterra trophic dynamics as well as production and profit assessments is developed and applied to the coastal fishery of French Guiana. This small-scale fishery constitutes a challenging example with high fish biodiversity, several non-selective fleets and a potentially increasing local food demand due to demographic growth. The dynamic model is calibrated with 13 species and four fleets using monthly catch and effort data from 2006 to 2009. Several contrasted fishing scenarios including status quo, total closure, economic and viable

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strategies are then simulated. They are compared from the viewpoints of both biodiversity preservation and socioeconomic performance, assuming fixed landing prices and fixed costs. We show that fishing outputs, including food supply and fleet profitability, can be sustained on average but a loss of species cannot be avoided.

1. Introduction

Marine fishery resources are under extreme pressure worldwide. According to recent studies (Garcia and Grainger, 2005; FAO, 2010), three-quarters of fish stocks are maximally exploited or over-exploited. Moreover, the proportion of marine fish stocks which are intensively exploited is growing. Hence, sustainability is nowadays a major concern raised by international agreements and guidelines to fisheries management. Standard approaches to the sustainable management of fisheries such as MSY (maximum sustainable yield), MEY (maximum economic yield) or ICES[1] precautionary approaches usually address each exploited species separately (Grafton *et al.*, 2007). These management approaches have not succeeded in avoiding biodiversity loss, over-exploitation and fishing overcapacity worldwide (Hall and Mainprize, 2004). The ecosystem approach for fisheries (EAF) or ecosystem-based fisheries management (EBFM) advocate an integrated management of marine resources to promote sustainability (FAO, 2003). Such a management policy requires first to account for the complexity of ecological mechanisms that encompass community dynamics, trophic webs, geographical processes and environmental uncertainties (habitat, climate). Furthermore, by putting emphasis on sustainability, this type of approach strives to balance ecological, economic and social objectives for present and future generations and to handle a large range of goods and services provided by marine ecosystems (Jennings, 2005), including both monetary and non-monetary values.

However, operationalizing the EBFM approach remains unclear and challenging. It requires models, indicators, reference points and adaptive management strategies. Plaganyi (2007) provides an overview of the main types of modeling approaches and analyzes their relative merits for fisheries assessment in an ecosystem context. Modeling approaches and metrics useful for planning implementing and evaluating EBFM are also discussed in Marasco *et al.* (2007), with particular emphasis on management strategy evaluation. The use of ecosystem indicators is analyzed by Rice (2000) and Cury and Christensen (2005). In particular, Link (2005) emphasizes the need for a multi-criteria approach to achieve ecological, economic and social objectives.

This article discusses the sustainable management of a multi-species and multi-fleet fishery from an ecosystem-based perspective for the small-scale fishery of French Guiana. Taking an EBFM approach to this case study was challenging. The fishery is characterized by various complex features including a high equatorial fish biodiversity impacted by several

[1] International Council for the Exploration of the Sea, see http://www.ices.dk.

non-selective fleets and demographic growth which could potentially affect local food demand and consequently the production of this fishery.

2. Case study

The continental shelf of French Guiana is a tropical ecosystem under the influence of the Amazon estuary, as is the entire North Brazil Shelf Large Marine Ecosystem (LME) which contains a high biodiversity (Leopold, 2004). With 350 km of coastline, French Guiana benefits from a 130,000 km^2 exclusive economic zone (EEZ) including 50,000 km^2 of continental shelf. The coastal fishery operates 16 km offshore at depths of 0–20 m. Several landing points are spread along the coastline, and this fishery currently involves about 200 wooden boats locally named *pirogues* (P), *canots créoles* (CC), *canots créoles améliorés* (CCA) and *tapouilles* (T). *Pirogues* are canoes equipped with an outboard engine, which fish for periods of a few hours essentially in estuaries using ice stored in an old refrigerator. Compared to *pirogues*, *Canots créoles* are more adapted to sea navigation. *Canots créoles améliorés* have cabins and ice tanks which make it possible to fish for several days. *Tapouilles* are wider boats with a cabin and an inboard diesel engine. The gears used are drift or fixed nets, with mesh sizes between 40 and 100 mm. The type of fleet, the length of gill nets, the number of days spent at sea and the location of fishing activities all have an influence on the quantity of fish landed and on the species composition of the total harvest. Of the numerous coastal species, 30 are exploited and about 15 species, including weakfish, catfish and sharks, represent more than 90 per cent of the production. Annual landings have been estimated at approximately 2,700 tonnes for the past few years, as reported in the Ifremer[2] Information System (http://www.ifremer.fr/guyane/Chiffres-cles).

The coastal fishery plays an important socioeconomic role for all the small towns along the coastline where more than 90 per cent of the population is located. However, assessment of this fishery only began in 2006 with data collection monitored by Ifremer. Production and fishing effort values are collected on a daily basis at the main landing points by observers from local communities. An exhaustive sampling is performed due to the small number of boats (approximately 200). Seventy five per cent of the fishing activity is observed on a daily basis from January to December. Each year, some 3,600 landings are recorded. For each landing, the production by species is estimated or weighed by the observers or reported by the fishermen. Other information is also collected, such as trip duration, net length and fishing area. Since the boats are under 12 m in length, fishermen are not obliged to provide this information. The data collection system depends significantly on the fishermen's collaboration. Economic assessment started in 2009 with a survey on production costs and selling prices carried out in the field. The coastal fishery in French Guiana remains largely informal despite: (1) the founding of the French Guiana fishers' cooperative (CODEPEG) in 1982; (2) the implementation of a system of professional

[2] French Research Institute for Exploration of the Sea.

licenses in territorial waters by the regional fisheries committee in 1995; and (3) the progressive application of national and European regulations (role of crew, safety inspections of boats, etc.). There is no quota for catches, and no limitation concerning exploited species and their size.

This coastal fishery provides an interesting case study from the perspective of EBF management. The current state of this fishery is usually postulated as safe, and the biodiversity associated with this resource does not seem to be threatened by fishing activity. Nevertheless, the sustainability of the fishery could be threatened by increasing local demand for fish linked to the demographic projections suggesting a 100 per cent increase in the local population over the next 20 years. Consequently, this increasing demand for local fish will affect fishing pressure. The question arises whether both the marine ecosystem and the fishing sector can cope with such changes and contribute to food security.

To examine these issues, this paper proposes a theoretical and empirical modeling of EBFM, using a multi-species and multi-fleet model integrating Lotka–Volterra trophic dynamics and profit functions. The dynamic model is calibrated on a monthly basis with 13 species and four fleets (P, CC, CCA and T) using catch and effort data from 2006 to 2009 derived from the Ifremer fishery information system. Ecological and economic performance of contrasting fishing scenarios including status quo, total closure, economic and viable strategies are examined and compared.

The main contribution of this work is two-fold. First, it proposes for the first time decision support tools for the management of the French Guiana coastal fishery by providing a bio-economic model, analysis and scenarios using time series on catch and fishing effort together with economic parameters. In the broader context of small-scale fisheries, such a bio-economic work relying on a perennial database is new to the best of our knowledge. It is acknowledged that small-scale fisheries are poorly managed due to a lack of tools and data adapted to their complexity, while these fisheries are crucial to sustaining many communities especially in developing or underdeveloped countries (Garcia *et al.*, 2008). The second contribution of this study is to advocate the use of co-viability approaches as a fruitful modeling framework for EBFM and sustainability issues. By accounting for complex and non-linear dynamics in a trophic and multi-fleet context and by addressing biodiversity issues, the paper shows that viability modeling (Bene *et al.*, 2001) can be applied to high-dimensional environmental systems. Moreover, this work points out that, by balancing ecological and economic goals with production and food security objectives over the next 40 years, the viability approach is well suited to coping with sustainability due to its multi-criteria perspective and the fact that it takes intergenerational equity into account, as in Péreau *et al.* (2012).

The paper is structured as follows. Section 3 is devoted to the description of the ecosystem-based model together with bio-economic indicators and scenarios. Section 4 provides the calibration results and the outputs of the different fishing scenarios with respect to biodiversity and socioeconomic indicators. Results are discussed in terms of sustainability, EBFM and management tools in section 5. The final section provides a conclusion.

Table 1. *The 13 selected species representing about 90% of the catches of the fishery*

Common name	*Scientific name*	*Trophic level T_i (Fishbase)*
Acoupa weakfish	*Cynoscion acoupa*	4.05
Crucifix sea catfish	*Hexanematichtys proops*	4.35
Green weakfish	*Cynoscion virescens*	4.03
Common snooks	*Centropomus parallelus, Centropomus undecimalus*	4.2
Sharks	*Sphyrna lewini, Carcharhinus limbatus, Mustelus higmani*	4.5
Smalltooth weakfish	*Cynoscion steindachneri*	3.25
South American silver croaker	*Plagioscion squamosissimus*	4.35
Tripletail	*Lobotes surinamensis*	4.04
Gillbacker sea catfish	*Arius parkeri*	4.11
Bressou sea catfish	*Aspistor quadriscutis*	3.5
Goliath grouper	*Epinephelus itajara*	4.09
Flathead grey mullet	*Mugil cephalus*	2.13
Parassi mullet	*Mugil incilis*	2.01

3. Methods

The numerical implementations of the model are carried out with the scientific software SCILAB 5.2.2.[3]

3.1. *The ecosystem-based model*

Among the 30 exploited species, 13 were selected for the model as shown in table 1. These species represent 88 per cent of the total landing from 2006 to 2009. A virtual 14th species which stands for all the other marine producers was added. A potential trophic web (see figure a in the online appendix, available at http://journals.cambridge.org/EDE) was built with these selected species, according to their diet (Leopold, 2004) and their trophic level (table 1).

The ecosystem-based model is a multi-species, multi-fleet dynamic model described in discrete time with a monthly step. The states of the species in the ecosystem-based model are supposed to be governed by a complex dynamic system based on Lotka–Volterra trophic interactions and fishing efforts from the different fleets which play the role of controls in the system. Thus, at each step t, the biomass $B_i(t+1)$ (kg) of species i at time $t+1$ depends on other stocks $B_j(t)$ and fishing efforts $e_k(t)$ of fleet k (time spent at sea, in hours) through the relation:

$$B_i(t+1) = B_i(t)\left(1 + r_i + \sum_{species\ j=1}^{14} s_{i,j} B_j(t) - \sum_{fleets\ k=1}^{4} q_{i,k} e_k(t)\right). \quad (1)$$

[3] SCILAB (http://www.scilab.org) is an open-source software dedicated to scientific calculus and well suited to the simulation of dynamic systems.

Here r_i stands for the intrinsic growth rate of the population i and $s_{i,j}$ the trophic effect of species j on species i (positive if j is a prey of i and negative if j is a predator of i). The parameter $q_{i,k}$ measures the catchability of species i by fleet k. It corresponds to the probability of a biomass unit of species i being caught by a boat of fleet k during one fishing effort unit. The number of the fleet k from $k = 1$ to $k = 4$ corresponds respectively to CC, CCA, P and T.[4]

The catches $H_{i,k}$ of species i by fleet k at time t are thus given by the Schaefer production function:

$$H_{i,k}(t) = q_{i,k}e_k(t)B_i(t). \tag{2}$$

3.2. Model and calibration inputs

Values used to define the model parameters came from different sources. Daily observations (catches and fishing efforts) from the landing points all along the coast are available from January 2006 to December 2009. Every month during this 48-month period, for each of the four fleets, fishing effort and catches were identified for the 13 species, for a total of 2,688 observations. The literature (Leopold, 2004) and Fishbase[5] provided qualitative trophic interactions concerning the sign of the relationship between species and intrinsic growth rates to start the calibration. In particular, only prey–predator and mutual competition relationships are considered in the Lotka–Volterra model, and not symbiotic relationships between species. Initial stocks, catchabilities, trophic intensities and refined intrinsic growth rate values of this ecosystem were estimated through a least square method. This method consisted of minimizing the mean square error between the monthly observed catches $H_{i,k}^{\text{data}}$ and the catches $H_{i,k}$ simulated by the model, as defined by equations (1) and (2):

$$\min_{B_0;\, s;\, q;\, r} \sum_{t=\text{January 2006}}^{\text{December 2009}} \sum_{i=1}^{13} \sum_{k}^{4} \left(H_{i,k}^{\text{data}}(t) - H_{i,k}(t)\right)^2. \tag{3}$$

Here $(B_0;\ s;\ q;\ r)$ is the set of parameters to identify. $B_0 = B(t_0)$ is the vector (14×1) of initial stocks ($t_0 =$ December 2005), s the matrix (14×14) of trophic interactions, q the matrix (14×4) of catchabilities, and r a vector (14×1) of intrinsic growth rates.

Several simple biological and productive constraints on parameters were taken into account for the optimization process (equation 3). In particular, several intra-specific interaction coefficients were set to zero (typically *B.catfish*, *F.mullet* and *P.mullet*, $i = 10, 12, 13$), prey–predator relationships (A. weakfish serve as prey for sharks $s_{5,1} > 0$ and sharks are predators of A. weakfish $s_{1,5} < 0$), common prey relationships (A. weakfish also serve as prey for G. groupers $s_{11,1} > 0$) and mutual competition (the predators shark and G. grouper prey on each other, $s_{5,11} < 0$ and $s_{11,5} < 0$) were considered

[4] Between 2006 and 2009, there were 71 CC, 60 CCA, 45 P and 10 T.
[5] See http://www.fishbase.org.

(table a, online appendix). Some catchability parameters $q_{i,k}$ were also set at zero since some species are not caught by fleets, typically fleet T (table b, online appendix). The non-linear optimization problem (equation 3) was solved numerically using the Scilab routine entitled 'optim_ga' which relies on an evolutionary (or genetic) algorithm.[6]

3.3. Model outputs: ecological indicators

After calibration, ecological and economic indicators were computed to assess the performance of both the ecosystem and the fishery. We first focused on biodiversity indices. Although the choice of a biodiversity metric remains controversial as pointed out in Magurran (2007), we selected the species richness, Simpson and marine trophic indicators provided by equations (4), (5) and (6).

3.3.1. Species richness

Species richness $\mathrm{SR}(t)$ indicates the estimated number of species represented in the ecosystem. It is measured by an indicator function based on abundances $N_i(t)$, computed as the ratio between the biomass $B_i(t)$ and the common weight w_i of each species, derived from the Fishbase information system:

$$\mathrm{SR}(t) = \sum_i \mathbf{1}_{\{]0,+\infty[\}}(N_i(t)), \quad \text{with } N_i(t) = \frac{B_i(t)}{w_i}, \tag{4}$$

where the function $\mathbf{1}_{\{]0,+\infty[\}}$ corresponds to the characteristic function[7] of positive reals. Thus, it is assumed that a species disappears whenever its abundance falls to zero (Worm *et al*., 2006). It should be noted that rare species have a relatively huge impact on the species richness index.

3.3.2. Simpson's diversity

The Simpson index $\mathrm{SI}(t)$ is expressed as:

$$\mathrm{SI}(t) = 1 - \sum_i f_i^2(t), \quad \text{with } f_i(t) = \frac{N_i(t)}{N(t)}, \tag{5}$$

where $N(t) = \sum_i N_i(t)$. The index SI estimates the probability of two individuals belonging to the same species. The index varies between 0 and 1.

[6] See http://help.scilab.org/docs/5.3.3/en_US/optim_ga.html for details on 'optim_ga'. A genetic algorithm is a search heuristic that mimics the process of natural evolution. This heuristic is routinely used to generate solutions to non-linear optimization. Genetic algorithms belong to the larger class of evolutionary algorithms which use techniques inspired by natural evolution, such as inheritance, mutation, selection and crossover. In our case, the genetic algorithm ended up performing better than the usual optimization or calibration algorithms. This type of numerical method has already been used for bio-economic purposes in Mardle and Pascoe (2000), for instance, and for other tropical fisheries in Sathianandan and Jayasankar (2009).

[7] $\mathbf{1}_{\{]0,+\infty[\}}(x) = 1$ if $x > 0$; 0 otherwise.

A perfectly homogeneous community would have a Simpson diversity index score of 1. Such a metric gives more weight to the more abundant species. The addition of rare species causes only small changes in the value.

3.3.3. Marine trophic index
The trophic level indicates the location of a species in a food web, starting with producers (e.g., phytoplankton, plants) at level 0, and moving through primary consumers that eat primary producers (level 1) and secondary consumers that eat primary consumers (level 2), and so on. In marine fishes, the trophic levels vary from two to five (top predators). The marine trophic index MTI(t) of the ecosystem (Pauly and Watson, 2005) is computed from the trophic level of each species T_i (table 1) and their relative abundances f_i (see equation 5):

$$\mathrm{MTI}(t) = \sum_{i=1} f_i(t) T_i. \tag{6}$$

3.4. Model outputs: economic indicators
We now turn to the assessment of the fishing sector through production and profitability values of the fishery provided by equations (7) and (8).

3.4.1. Food supply
We first considered the total catches $H(t)$ within the fishery which play the role of food supply:

$$H(t) = \sum_k \sum_i H_{i,k}(t). \tag{7}$$

This supply must be compared with local food demand, which is expected to increase at an exogenous rate provided by demographic scenarios and projections over the next 20 years.

3.4.2. Profits
The profit $\pi_k(t)$ of each fleet k was derived from the landings of each species $H_{i,k}$, the landing prices $p_{i,k}$, fixed costs c_k^f, variable costs c_k^v and the crew share earnings β_k as follows:

$$\pi_k(t) = (1 - \beta_k) \left(\sum_i p_{i,k} H_{i,k}(t) - c_k^v e_k(t) \right) - c_k^f. \tag{8}$$

Prices, variable costs and fixed costs are those collected for 2008 (table c, online appendix). They were assumed to remain unchanged throughout the simulations. Share contract β is the salary system commonly used in this fishery for the CCA fleet ($k = 2$) and T fleet ($k = 4$). Crews are remunerated with a share of the landing value minus the variable costs. CC fleet ($k = 1$) and P fleet ($k = 3$) crews are mostly made up of boat owners, occasionally assisted by a family member. If there is a pay system for these fleets, it differs from one owner to another. Hence, to simplify, we set $\beta_k = 0$ for CC and P fleets and $\beta_k = 0.5$ for CCA and T fleets. Variable costs c_k^v include fuel consumption, ice, food and lubricants. Equipment depreciation, maintenance and repairs are incorporated in the fixed costs c_k^f.

The total profit $\pi(t)$ is the sum of profits over all fleets:

$$\pi(t) = \sum_k \pi_k(t). \tag{9}$$

3.5. *Fishing scenarios*

From the calibrated model, scenarios were simulated according to different fishing efforts over 40 years. We distinguished four scenarios: *closure* (CL), *status quo* (SQ), *economic* (PV) and *co-viability* (CVA). The set of ecological and economic indicators introduced previously were evaluated for these four scenarios.

3.5.1. The closure scenario (CL)

The CL scenario corresponds to the implementation of a no fishing zone over the whole French Guiana coastal area:

$$e_k(t) = 0, \quad \forall\, k = 1, \ldots, 4 \quad \forall\, t = t_1, \ldots, t_f$$

where t_1 corresponds to January 2010 and t_f to December 2050.

3.5.2. The status quo scenario (SQ)

The SQ scenario simulates a steady fishing effort based on the mean pattern of the efforts between 2006 and 2009:

$$e_k(t) = \bar{e}_k, \quad \forall k = 1, \ldots, 4 \quad \forall t = t_1, \ldots, t_f$$

with $\bar{e}_k$ representing the mean efforts between 2006 and 2009 for the fleet k as follows:

$$\bar{e}_k = \frac{1}{t_1 - 1} \sum_{t=t_0}^{t_1-1} e_k(t), \tag{10}$$

where t_0 and $t_1 - 1$ correspond to January 2006 and December 2009, respectively.

3.5.3. The economic scenario (PV)

The PV scenario maximizes the present value of all the future profits aggregated among the fleets $\pi(t)$ defined by equation (9). The present value depends on fishing effort patterns as follows:

$$\mathrm{NPV}(e(.)) = \sum_{t=t_1}^{t_f} (1+\gamma)^{-t}\pi(t),$$

where γ is the discount rate set at $\gamma = 3$ per cent. The optimal program underlying the PV scenario is defined by

$$\max_{e_k(t)} \mathrm{NPV}(e(.)). \tag{11}$$

In this scenario, it is assumed that the fishing efforts $e_k(t)$ rely on a control strategy that can be adapted every 5 years.[8] In other words, eight decisions $(e_k(t_1), e_k(t_2), \ldots, e_k(t_8))$ are available for each fleet k as follows:

$$e_k(t) = \begin{cases} e_k(t_1) & \text{for } t = t_1, \ldots, t_1 + 60 \\ e_k(t_2) & \text{for } t = t_2, \ldots, t_2 + 60 \\ \vdots & \\ e_k(t_8) & \text{for } t = t_8, \ldots, t_8 + 60 \end{cases} \tag{12}$$

where t_1 and $t_n = t_{n-1} + 60$, for $n = 2$ to 8, are decisive months.

The optimal effort $e_k(t)$ solutions of the intertemporal program (equation 11) were approximated numerically by again using an evolutionary algorithm, in particular the routine entitled 'optim_ga' in Scilab.

3.5.4. The co-viability scenario (CVA)

The purpose of the CVA scenario is to provide a satisfactory balance over time between fleet profitability, biodiversity and local food demand. Thus, viable levels of fishing effort aim at complying with the bio-economic constraints below:

- A profitability constraint: $\pi_k(t) \geq 0,\ \forall\, t = t_1, \ldots, t_f,\ \forall\, k = 1, \ldots, 4$
- A species richness constraint: $\mathrm{SR}(t) \geq 11,\ \forall\, t = t_1, \ldots, t_f$
- A food security constraint: $H(t) \geq H(2009) \cdot (1+d)^t,\ \forall t = t_1, \ldots, t_f,$

where d stands for the growth rate of the population. The profitability constraint holds for each fleet separately and not for the aggregated rent as

[8] A refined time decomposition for fishing intensities (for instance, a one-year time step) would have improved the analysis by capturing a broader intertemporal flexibility in fishing strategy. However, it would have required very demanding computation times. Steady efforts over 5 years as imposed here capture rigidity and inertia mechanisms in behaviors which may occur in reality. We plan to expand the time step for decisions in future models.

in the PV scenario. Concerning the biodiversity constraint, no co-viability path maintaining the whole set of 13 species was exhibited. This explains why the species richness required was relaxed to only 11 species. Finally, the food security constraint assumed an increase in the local fish demand at the annual rate of $d = 3$ per cent, according to the demographic scenario which predicts a doubling of French Guiana's population by 2030 (INSEE, 2011). Moreover, it was assumed that fish species can be substituted, in the sense that a drop in the consumption of one species can be compensated for by a rise in the consumption of other species.

Following DeLara and Doyen (2008) and Doyen and De Lara (2010), viable efforts for the CVA scenario were obtained by maximizing the following criterion:

$$\max_{e_k(t)} \prod_{t=t_1}^{t_f} \mathbf{1}_{\{]0,+\infty[\}}(\pi_k(t))\mathbf{1}_{\{]0,+\infty[\}}(\mathrm{SR}(t)-11)\mathbf{1}_{\{]0,+\infty[\}} \cdot (H(t) - H(2009)\cdot(1+d)^t) \tag{13}$$

where again, efforts $e_k(t)$ are meant to be control strategies that can change every 5 years as in equation (12), and $\mathbf{1}_{\{]0,+\infty[\}}$ represents the characteristic function on positive reals. The numerical method again relies on the evolutionary optimization routine.

3.6. Sensitivity analysis and uncertainty margins

A sensitivity analysis was carried out to evaluate the role played in the bio-economic outputs by the different calibrated parameters (tables a and b, online appendix). To achieve this, we ran additional simulations based on the SQ scenario. Given the large number of parameters, we limited the sensitivity analysis by simultaneously perturbing all the parameters of the same group, i.e., initial stocks B_0, catchabilities q, trophic intensities s and intrinsic growth rates r. For each group of estimated biological parameters, a noise ranging from -10 per cent to $+10$ per cent of the calibrated values was added to the parameters. The relative differences in bio-economic outputs including average catches per annum $\overline{H} = \frac{12}{t_f - t_1}\sum_{t=t_1}^{t_f} H(t)$, net present value (NPV) and specific richness $\mathrm{SR}(t_f)$ were computed. Sensitivity analysis was also carried out to examine the impact of the choice of time horizon on the outputs. Therefore, other simulations with the SQ scenario were performed, increasing the simulation length t_f from December 2060 to December 2100. The corresponding bio-economic results were compared with those obtained with $t_f =$ December 2050.

In line with this, in order to assess the reliability of the outputs for each effort scenario, simulations were replicated 400 times by introducing uncertainties in the estimated parameters (r, s, q, B_0). For each simulation, a noise ranging from -10 per cent to $+10$ per cent of the calibrated values was again randomly added to the parameters.

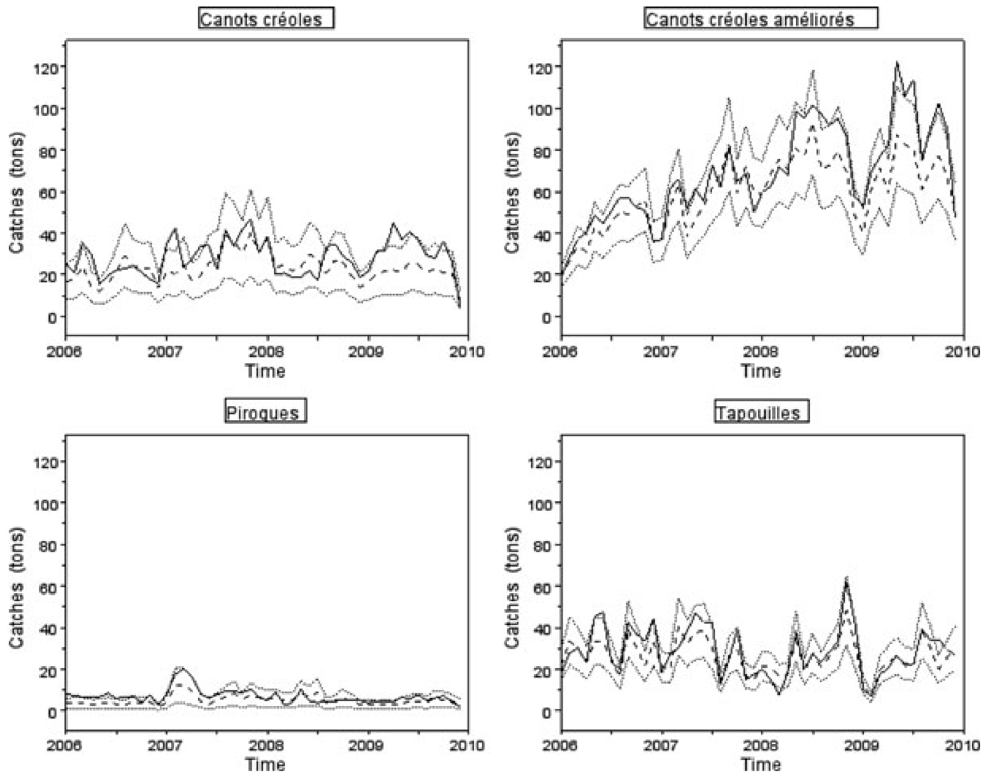

Figure 1. *Comparison by fleet k between historical catches* $\sum_{\text{species } i} H^{\text{data}}_{\text{species } i,k}(t)$ *(solid lines) and simulated catches* $\sum_i H_{i,k}(t)$ *(dashed lines), with the confidence intervals at 95% (dotted lines)*

4. Results

4.1. Calibration and sensitivity results

Figure 1 presents the historical and simulated catches by fleet, with 95 per cent confidence intervals. For each fleet k, confidence intervals[9] were computed from the mean relative errors Δ_k between observed and simulated catches from January 2006 to December 2009,

$$\Delta_k = \frac{1}{48} \sum_{t=t_0}^{t_1-1} \left| \frac{H_k^{data}(t) - H_k(t)}{H_k(t)} \right|, \tag{14}$$

where $H_k(t) = \sum_i H_{i,k}(t)$ stands for catches by fleet k at time t over the whole 13 species i. The mean relative errors equal[10] $\Delta_1 = 0.259$ for CC, $\Delta_2 = 0.13$ for CCA, $\Delta_3 = 0.354$ for P and $\Delta_4 = 0.176$ for T.

[9] For each month t, 95 per cent confidence intervals are $[1 - 1.96 * \Delta_k, 1 + 1.96 * \Delta_k] * H_k^{data}(t)$.

[10] The relative errors for the Euclidean or quadratic norm, $\Delta_k^* = \sqrt{\frac{1}{48} \sum_{t=t_0}^{t_1-1} \left(\frac{(H_k^{data}(t) - H_k(t))}{H_k(t)} \right)^2}$, yields: $\Delta_1^* = 0.308$ for CC, $\Delta_2^* = 0.151$ for CCA, $\Delta_3^* = 0.414$ for P and $\Delta_4^* = 0.257$ for T.

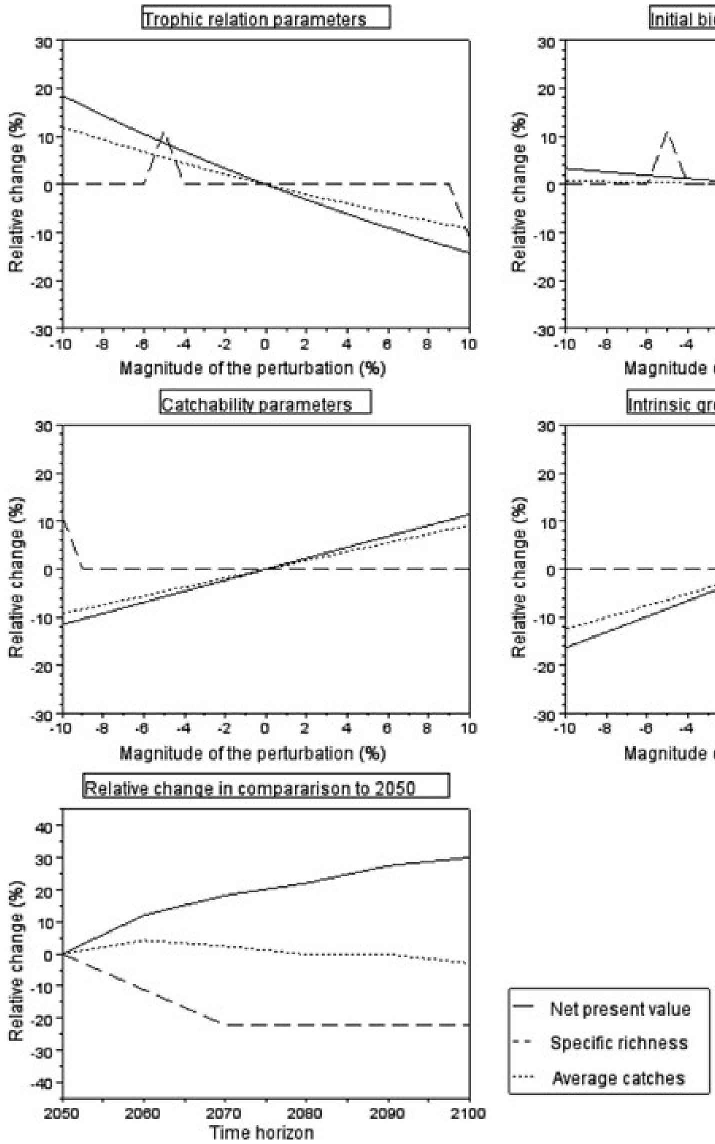

Figure 2. *Relative changes of NPV (solid line), average annual catches $\overline{H}$ (dotted line), species richness $SR(t_f)$ (dashed line), according to variations in input parameters by 1% increments from −10% to +10% (a, b, c and d), and time horizon (e). The baseline is status quo scenario SQ*

Figure 2 displays the sensitivity results. They stress the fact that the parameters with the greatest impact were intrinsic growth rates r_i and trophic interactions s_{ij}. The relative changes in NPV and average catch outputs appear to be approximately linear functions of the perturbations with slopes between 0 and 1.8 highlighting bounds for the marginal effects of the parameters. In particular, the impact of initial biomasses was small since the relative changes were less than the perturbation magnitude for these biomasses. Trophic intensities and intrinsic growth rates were the inputs for which a perturbation entailed larger relative changes in the outputs. The non-linear nature of the species richness index is captured by the staircase shape of the relative change as well as the peaks observed. Moreover, the relative changes in bio-economic outputs in comparison to the 2050 time horizon show a reduced impact of the temporal target in the results. In particular, the NPV is not affected by a change of horizon mainly because of the discount involved. Of interest is the fact that species richness is stabilized after 2070. The average annual catches continue to rise with the time horizon, which emphasizes the fact that overall fishery production does not collapse after year 2050 and could even be enhanced.

4.2. Scenarios, effort levels

Figure 3 displays the effort multipliers $\frac{e_k(t)}{\bar{e}_k}$ by fleet for each fishing scenario. These effort multipliers are based on the comparison between effort $e(t)$ and the mean pattern of efforts $\bar{e}_k$ between 2006 and 2009 defined in equation (10). The SQ effort multiplier is equal to one, as expected. It turns out that the PV scenario induces the largest decrease in fishing efforts to maximize the present value of aggregated rent. In particular, the PV scenario implies stopping fishing activity for the CC and CCA fleets during the entire simulation period. With regard to the T fleet, fishing effort is increased in the first two decades of the simulation and stopped in the last decade. By contrast, the fishing effort of the P fleet follows an opposite pattern. Effort is nil during the first two decades of the simulation and is increased after 2030. The multiplier for the T fleet reaches 2.4 in the first part of the simulation, while for the P fleet, multipliers range from 2.2

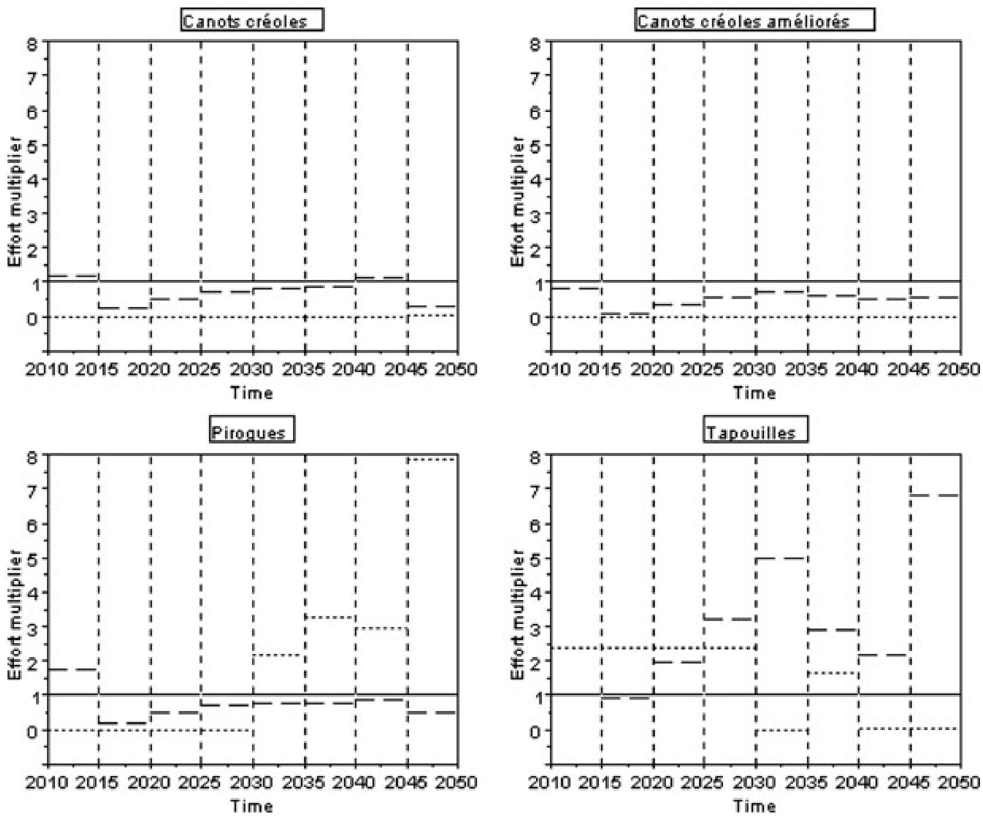

Figure 3. *Fishing effort multiplier $u_k(t) = \frac{e_k(t)}{\bar{e}_k}$ by fleet and scenario: SQ (solid line), economic PV (dotted line), co-viability CVA (dashed line)*

to 7.8 for the second part of the simulation. In contrast, the CVA scenario guarantees an activity for every fleet throughout time. On average, its effort level is lower than the baseline SQ except for the T fleet, which exhibits an effort multiplier ranging from 0.9 to 6.8. The average multiplier of the viable strategy is 0.7 for CC, 0.51 for CCA, 0.75 for P and 3.0 for T.

4.3. Ecological results

Trends in the evolution of species richness according to the scenarios are plotted in figure 4 (marine trophic and Simpson diversity evolutions are available in figures b and c in the online appendix). The 'mean' trajectories induced by the calibrated values are plotted together with margin errors of 400 simulations derived from the perturbation of the parameters selected randomly in [−10 per cent; +10 per cent]. First it appears that a loss of species occurs for every scenario, as species richness decreases in every case except in the CL scenario, as expected (at least when the parameters are not perturbed). In other words, implementing a no fishing zone should maintain species diversity. By contrast, the baseline SQ scenario leads to the worst result in terms of diversity loss. Species richness ranges from 11 to 8 at the end of the simulation period. The mean simulation provides nine species at the end and species like *Crucifix catfish*, *Common snook*, *Silver croaker* and *Bressou catfish* disappear. With the PV scenario, both *Crucifix catfish* and *Bressou catfish* collapse. The final state of species richness with the CVA scenario is qualitatively identical to the PV scenario since 11 species remain at the end while the same species disappear. From mean estimated

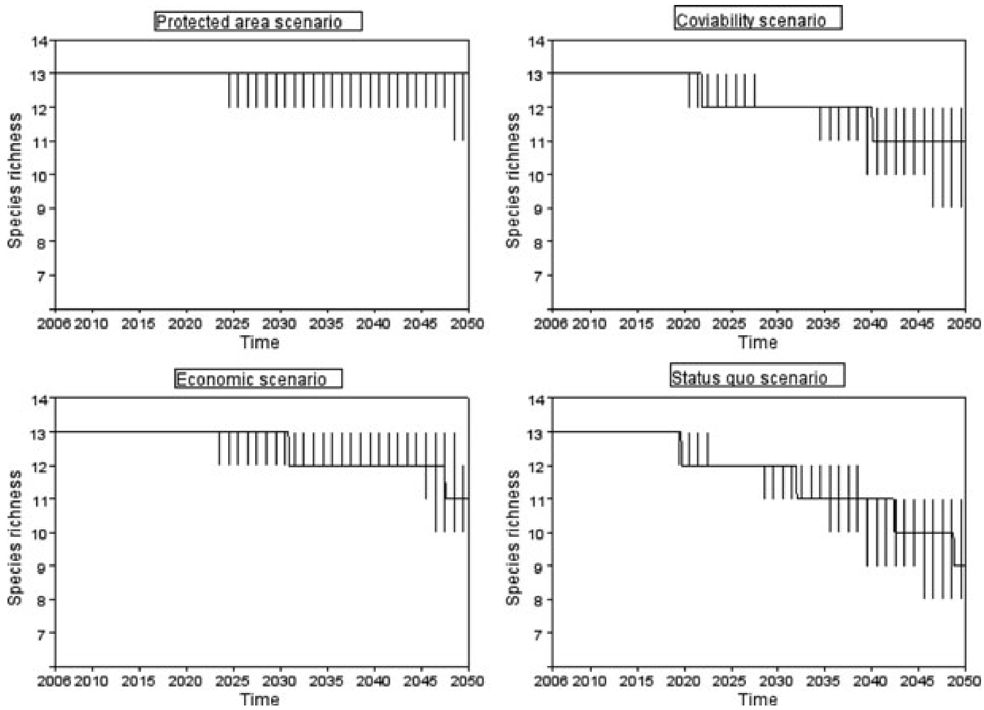

Figure 4. *Species richness $SR(t)$ evolution by scenario (solid lines), with uncertainties (vertical lines)*

parameters, two species (*Crucifix catfish* and *Bressou catfish*) become extinct in the SQ, CVA and PV scenarios, but the extinction periods are not identical: species extinctions are delayed in proportion to the reductions in effort level. Extinction periods of these two species correspond to years 2020–2032 for the SQ scenario, 2022–2040 for the CVA scenario and 2031–2047 for the PV scenarios respectively.

The trajectories of the two other biodiversity indices are more complex and difficult to interpret. The species abundances change considerably in the simulation period. In particular, a major change occurs around 2015 for all ecological indicators when certain species start to decline. This decrease is illustrated by the decline in catches between 2015 and 2020 for the SQ scenario (figure 5). At the start of the mean simulation, the total biomass is not equally distributed among the species with SI $= 0.5$, and the marine ecosystem is dominated by species with a low trophic level, MTI $= 2.5$. At the end of the mean simulation, for all scenarios, diversity indices are better than those at the beginning (SI ranges from 0.61 to 0.77, MTI from 2.79 to 3.08, according to the scenario).

The impact of uncertainties is significant, as the ecological indices appear volatile in particular for the last years. This indicates that the results should be considered with caution.

4.4. Economic results

Catches and profits for the SQ, PV and CVA scenarios are plotted in figures 5–8. The main biomass changes in years 2015–2020 also affect the catches and profits. The SQ scenario seems economically viable in terms of profitability, as annual profits are positive during almost the entire period for all fleets. However, exceptions occur for the CC and CCA fleets in the first years of the simulation and for the P fleet in the 2010–2011 and 2026–2034 periods. Not surprisingly, the PV scenario yields the highest cumulative discounted profit, between €1.125 and 2.399 billion vs. €123.2–203.3 million vs. for the SQ scenario and €84.7–239.9 million for the CVA scenario. The greatest fishing activity occurs in the second part of the simulation for the P fleet. One explanation can be found in the high value of the selling prices for this fleet (table c, online appendix). On average, the CVA scenario provides positive annual profits for each fleet throughout the simulation despite the fact that the CVA fishing effort is lower than the SQ effort. However, as the CVA scenario effort levels were computed from the mean estimated parameters, the uncertainties may alter the profitability in certain years.

Comparison of the fish demand curve with the supply curves by scenario (figure 5) shows that yield levels may differ broadly from local fish demand projections. In particular, for a period of several years, the mean production is lower than the fish demand[11] except for the mean CVA scenario, as expected. In the same vein, the mean cumulative supply over 40 years of the CVA scenario with $H = \sum_t H(t) \approx 262$ Ktons is the

[11] It should be pointed out that prices are fixed and then do not clear the market. This assumption could be relaxed in future work.

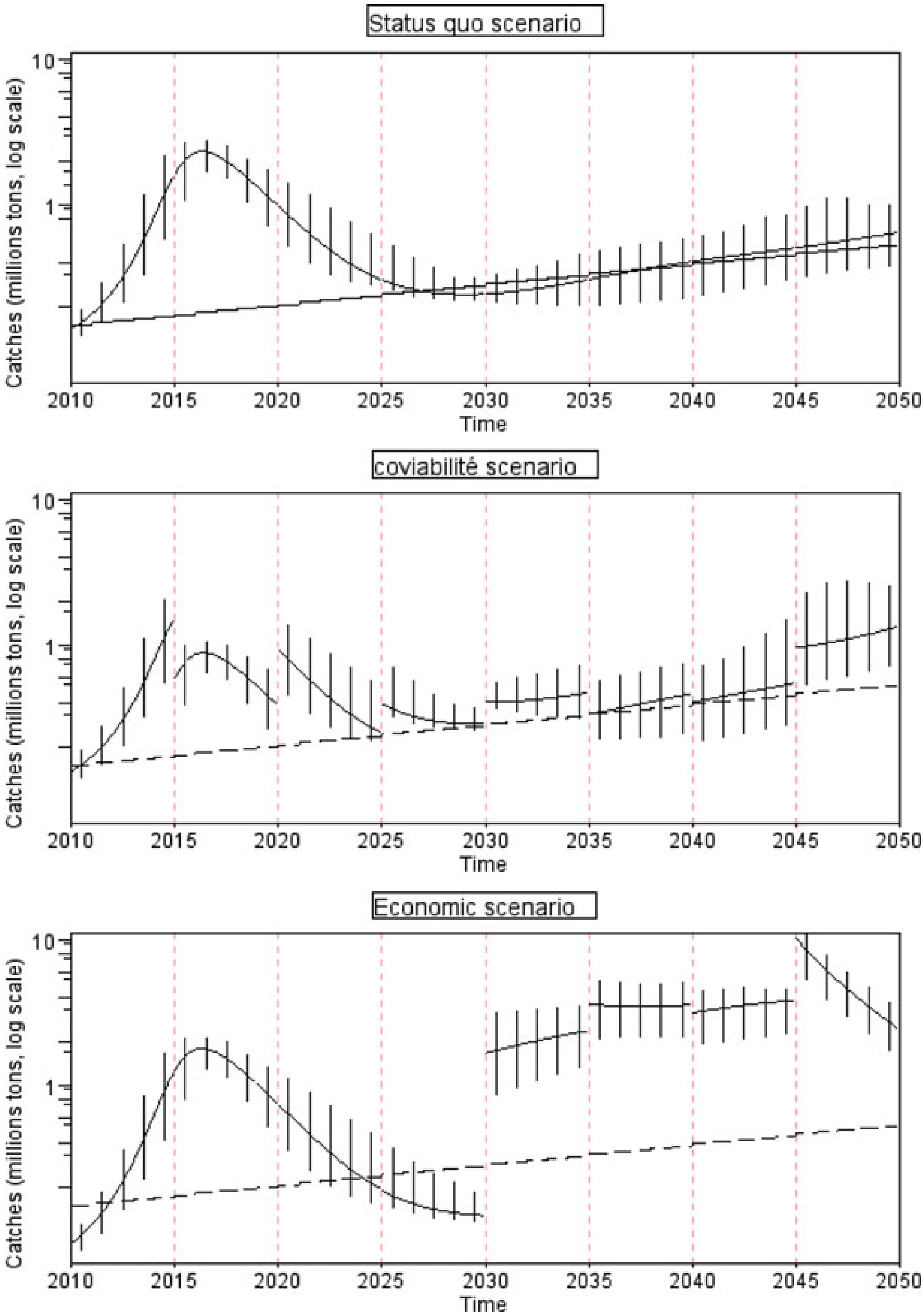

Figure 5. *Total catches $H(t)$ by scenarios (solid lines) vs. local fish demand (dashed line), with uncertainties (vertical lines)*

closest to the cumulative fish demand of 144 Ktons as compared to the SQ and PV scenarios with $H = 284$ and $H = 986$ Ktons respectively. However, it also appears that the food security constraint of the CVA scenario may be violated during some years when uncertainties are taken into account.

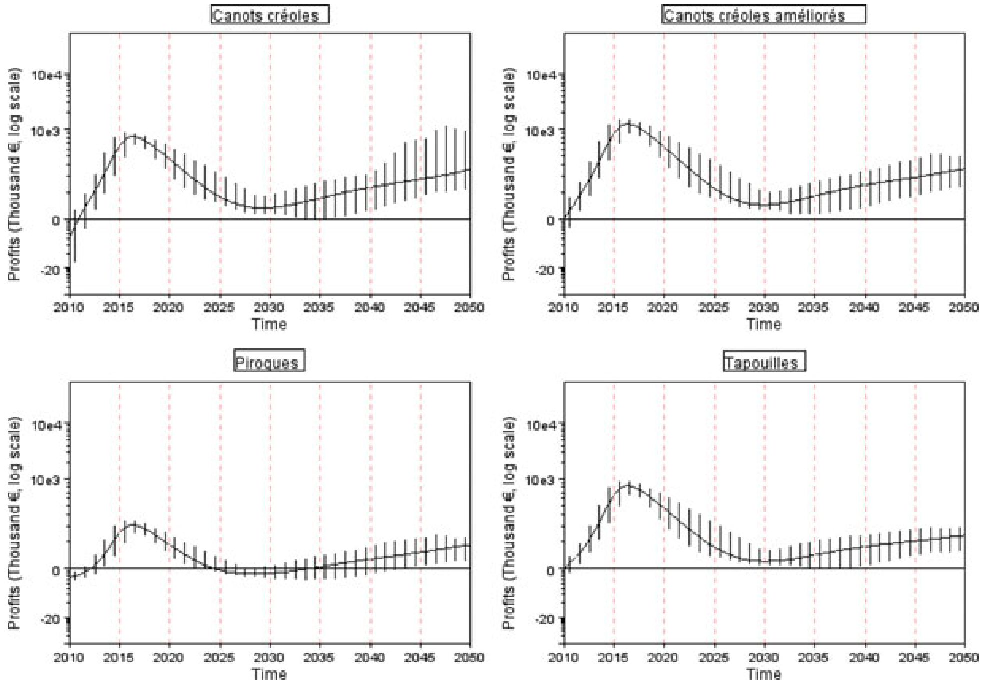

Figure 6. *Profit $\pi_k(t)$ by fleet for the SQ scenario (solid lines), with uncertainties (vertical lines). The dotted line stands for profitability threshold*

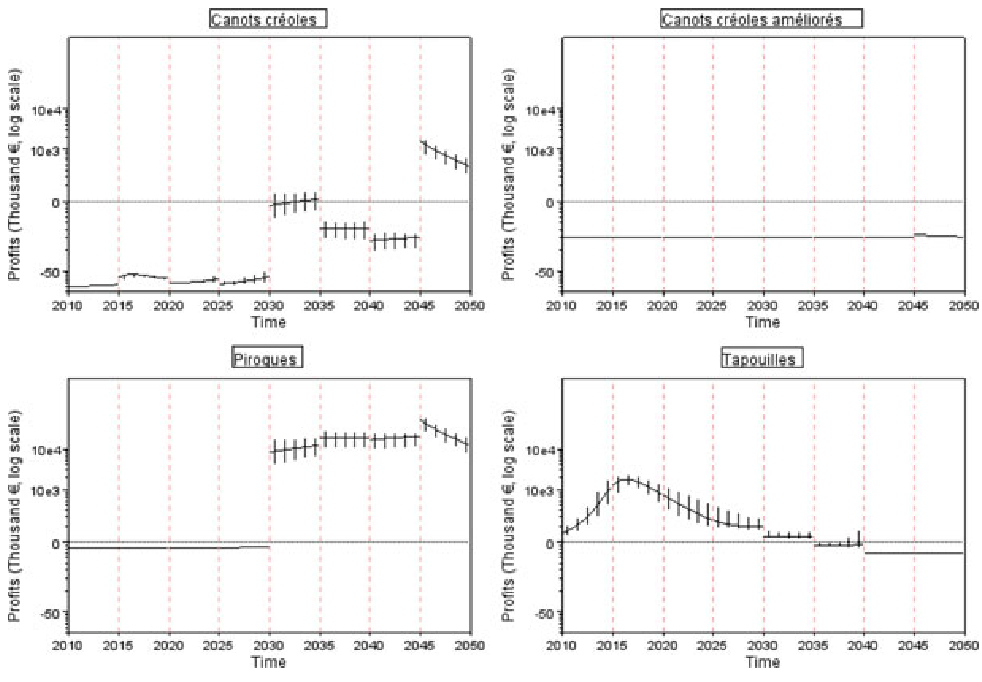

Figure 7. *Profit $\pi_k(t)$ by fleet for the PV scenario (solid lines), with uncertainties (vertical lines). The dotted line stands for profitability threshold*

5. Discussion

5.1. Co-viability as a step towards sustainability

Let us first analyze our results in terms of sustainability. Obviously, a total fishery closure is not a satisfactory solution either economically or socially

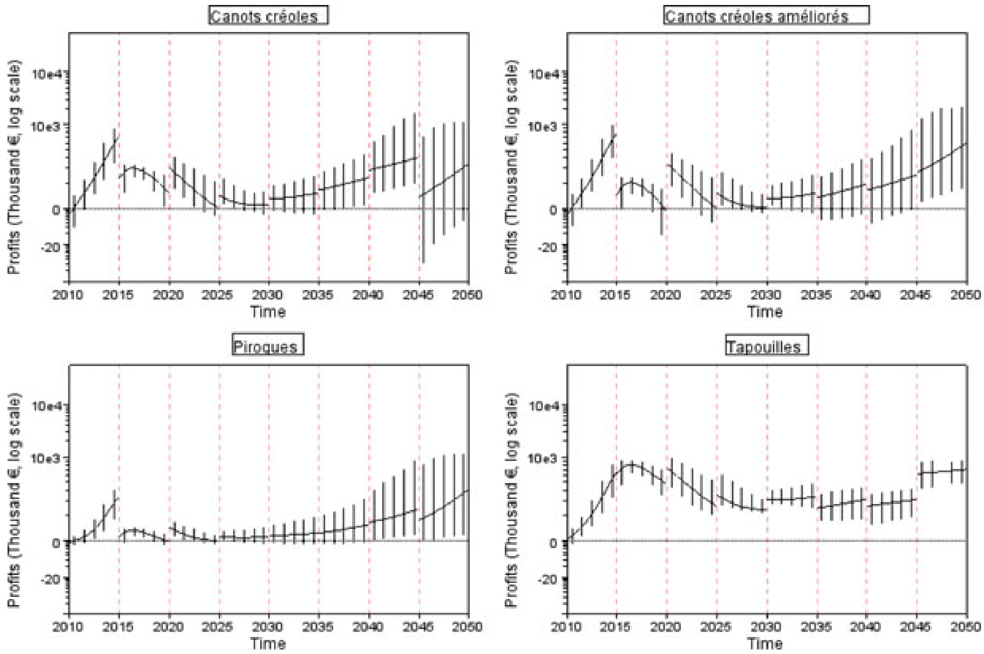

Figure 8. *Profit $\pi_k(t)$ by fleet for the CVA scenario (solid lines), with uncertainties (vertical lines). The dotted line stands for profitability threshold*

in terms of jobs, income and food consequences. It turns out that maintaining constant efforts through the SQ scenario is also not a suitable and sustainable strategy. In fact, aside from the fact that the CC and P fleets do not realize any profit in the first years, the SQ scenario does not satisfy the constraint of local consumption from years 2028–2038 in the mean regime and provides the worst performance for species richness. The calibration context can partially explain the negative profits of these fleets at the beginning of the simulation. Indeed, economic data are based on year 2008 which was unusual: fuel prices reached a record and thus production costs rose considerably. More generally, the low prices at first sale and the production costs did not allow every vessel to generate profits. Not surprisingly, the largest cumulative discounted profit and the most important fish supply are obtained with the PV scenario. However, this scenario may not be socially acceptable since profits are not evenly distributed between fleets over time. This happens because this scenario imposes that the CC and CCA fleets cease their activities, inducing negative profits for these fleets due to fixed costs (figure 7). That some fleets exhibit negative profits is consistent from the social planner's point of view underlying the PV approach, since aggregated profits are optimized by favoring the most efficient fleets. A better balance between biodiversity and socioeconomic performance can be reached with the CVA scenario, at least on average. Although two species disappear, this scenario appears to be the best compromise: it allows annual positive mean profits for every fleet and satisfies local consumption during the 40 years of simulation. However, the variability of outputs due to noise in parameters suggests that a stochastic or robust approach would be fruitful to guarantee this viability in an uncertain context.

In addition to analysis on the case study, this work advocates an integrated and multi-criteria approach. A wide range of stakeholders are involved in fisheries, including: industrial, artisanal, subsistence and recreational fishermen; suppliers and workers in allied industries; managers, environmentalists, biologists, economists; public decision makers and the general public. Each of these groups has an interest in particular outcomes from fisheries, and the outcomes that are considered desirable by one stakeholder may be undesirable for another group (Hilborn, 2007). Considering this multi-dimensional nature of marine fisheries management is a way to guarantee the reasonable exploitation of aquatic resources, allowing the creation of conditions for sustainability from economic, environmental and social viewpoints. The present work is fully in line with these considerations. First, of interest is the use of bio-economic models and assessments articulating ecological and socioeconomic processes and goals as in Bene *et al.* (2001); Doyen *et al.* (2012); Péreau *et al.* (2012). Moreover, by focusing on sustainability and viability, the present model exhibits management strategies and scenarios that account for intergenerational equity. As emphasized in Martinet and Doyen (2007) and DeLara and Doyen (2008), viability is closely related to the maximin (Rawlsian) approach with respect to intergenerational equity. In this respect, the CVA strategy turns out to be a promising approach.

5.2. Co-viability as a step towards EBFM

Several authors have proposed the viability approach as a new, innovative and well-suited modeling framework for EBFM (Cury *et al.*, 2005; Doyen *et al.*, 2012). They argue that the viability approach, especially co-viability, is relevant in handling EBFM issues because it may simultaneously account for dynamic complexities, bio-economic risks and sustainability objectives balancing ecological, economic and social dimensions for fisheries. In particular, Cury *et al.* (2005) and Doyen *et al.* (2007) show how the approach can potentially be useful for integrating ecosystem considerations for fisheries management. Mullon *et al.* (2004), Bene and Doyen (2008) and Chapel *et al.* (2008) emphasize the ability to address complex dynamics in this framework. The computational and mathematical modeling methods proposed in this paper through the CVA strategy are motivated by a similar prospect. One major advantage of the co-viability approach is the fact that the viability framework is dynamic and thus makes it possible to capture the interactions and co-evolution of marine biodiversity and fishing. The dynamics can potentially include complex mechanisms such as trophic interactions, competition, metapopulation dynamics or economic investment processes. Here the focus is both on trophic and technical interactions through a multi-fleet and multi-species context as in Doyen *et al.* (2012).

Projections over 40 years for different fishing scenarios highlight the complexity of mechanisms at play, particularly their non-linearity. With regard to this point, the trajectories of ecological indicators are representative and should not be interpreted separately. The species richness for the CL scenario can be sustained, meaning that all species are present at the end of the mean simulation. However, the Simpson and marine trophic indices reveal that species abundances change over the simulation period, even more when uncertainties on estimated parameters are considered. Diversity index (SI, MTI) values at the end of the mean simulation lead to the following findings: (1) total biomass is better distributed among species and (2) the species with a high trophic level are better represented. Thus, the effects of fishing on the species can be deduced: fishing leads to ecosystem specialization.

5.3. Decision support for the French Guiana small-scale fishery

Small-scale fisheries remain poorly managed because of their heterogeneity, difficulties in getting consistent and perennial data and the lack of regulation tools. The problem is more acute in a tropical context with a high-level informal activity and high biodiversity with low stock biomass (this is typically valid for reef ecosystems). In French Guiana, waters are very turbid and productive due to the proximity of the Amazon river. There are no reefs, but biodiversity is high, as is biomass. The bio-economic database monitored from 2006 with the help of local communities who collected time series data offers the opportunity to go a step further towards building management tools. Since the decline of the French Guiana industrial shrimp fishery (Chaboud *et al.*, 2008), the coastal fishery has become a sector with a high potential for development. In 2008, coastal fishery production was higher than shrimp and red snapper landings. However, as previously stated, there is no quota for catches, and no limitation concerning exploited species and their size. Regulation tools are derived from commonly used national and European fisheries management systems. These standards concern the gear selectivity (mesh size) and the global size of the fleet through total engine power and total vessel capacity. However, due to the lack of studies on the stock status for the main exploited species, rules relating to overall fleet size have not been adapted to the changing level of fish stocks. The only aim of the current management strategy is to prevent fishing activity by unauthorized boats. The present bio-economic study should contribute to the design of more scientific and relevant assessments and regulations for both the marine ecosystem and this small-scale fishery. At this stage, we would like to point out the methodological interest of sustaining the fishery information system to achieve such goals.

Fishing scenario outputs show that fishing performance, including food supply and profitability of fleets, can be increased or sustained. In particular, this suggests that the marine ecosystem and the fishing sector could cope with food demand and contribute to food security. This could have positive consequences for the development of French Guiana, since the coastal fishery plays an important socioeconomic role for the small towns along the coastline where more than 90 per cent of the population is located. However, there is a risk of losing fish biodiversity due to fishing pressure. This loss of biodiversity could potentially alter some ecosystem services (not taken into account in the current model) and the outcomes of the fishery itself in the long run. Thus, some fish stocks should be evaluated more specifically in order to anticipate their depletion (*Crucifix catfish*, *Bressou catfish*). Depending on the endangered stocks, conservation measures for the productive and reproductive capacities of these stocks should be taken. This could be achieved by banning fishing in nursery areas or providing incentives for using more selective fishing techniques. In this way, the co-viability approach could enable long-term management of the French Guiana coastal fishery. The CVA scenario suggests that such a multi-functional sustainability would be maintained with a small increase in the T fleet's effort and a relative reduction for the other fleets (CC, CCA or P). This management strategy entails implementing limitations on fishing effort. Nevertheless, this scenario may remain attractive for the different stakeholders involved since the profitability constraint for each fleet, the species richness constraint and the food security constraint are all satisfied. In this sense, the CVA strategy could be potentially operationalized with the fishermen's cooperation.

6. Conclusion

This work provides a bio-economic model and analysis for the coastal fishery in French Guiana. It relies on a multi-species and multi-fleet model integrating Lotka–Volterra trophic dynamics and profit functions. The dynamic model is calibrated using data from the Ifremer fishery information system. Ecological and economic performance of contrasting fishing scenarios including status quo, total closure, economic and viable strategies are compared. The major contribution of the paper is two-fold. First, it proposes for the first time decision support tools for the management of the small-scale fishery in French Guiana. Small-scale fisheries are poorly managed due to a lack of tools and data, although these fisheries are crucial to sustaining many communities especially in developing or underdeveloped countries (Garcia *et al.*, 2008). The present work emphasizes the interest of bio-economic models which rely on a perennial database in this context of small-scale fisheries. The second contribution of this study is to advocate the use of viability approaches as a relevant modeling framework for EBFM and sustainability issues. Such sustainability is known to be difficult to achieve because economic, social and ecological goals can contradict each other (Pitcher, 2001). The paper points out that, by balancing ecological and economic goals with production and food security objectives over several decades, the viability approach is well suited to address sustainability. By accounting for complex and non-linear dynamics and by addressing biodiversity issues, the paper also shows how viability modeling can be applied to high-dimensional environmental systems. More generally, the present work suggests that adopting the viability method would enable other objectives of the EBFM approach to be taken into account. For instance, fisheries are urged to transform their practices progressively, to favor eco-friendly technologies, to reinforce the quality and reliability of products and services and to create jobs. New management policies integrating all these dimensions in accordance with public goals need to be defined, especially in this kind of small-scale coastal fishery (Blanchard and Maneschy, 2010).

Due to the uncertainties underlying the calibrated parameters, the results of this paper should be interpreted with caution. The reliability of some parameters needs to be reinforced to obtain a more accurate model. Up to now, only shrimp and red snapper fisheries have been widely studied in French Guiana. It turns out that certain parameters are estimated from Fishbase or from the literature. Consequently, it would be fruitful to integrate more values from local field studies dedicated to this ecosystem (for instance, intrinsic growth rates and trophic levels). Stomach content data analysis would also improve trophic interaction evaluations. Similarly, as landings are computed from catchabilities and initial stocks, it would be important to obtain a refined estimation of these parameters. These uncertainties suggest that a more robust approach based on stochastic viability methods should be used (Doyen and De Lara, 2010; Doyen *et al.*, 2012). Doing so would significantly strengthen the robustness of the outcomes and assertions of this dynamic complex model. At this stage, we would like to point out the advantage of sustaining the Fishery Information System with the help of local communities.

Furthermore, the ecosystem-based model is based on simplified dynamics. In fact, species in French Guiana's coastal ecosystem present different trophic levels (from 2.01 to 4.35), leading us to consider predator–prey relationships between the 13 species selected in the model. We used a basic Lotka–Volerra model because of the high number of species considered and the lack of biological data. Indeed, other models such as an individual-based model would have required us to calibrate even more biological parameters. In future work, we plan to refine the Lotka–Volterra model by adding a predator saturation effect, such as the Holling functional response (Holling, 1959), when preys are abundant.

Many other issues could be addressed in future work. From an economic and social viewpoint, taking into account the demand mechanism and endogenous prices is necessary to improve the predictions of the model. A next step would be to integrate social indicators such as employment level and job satisfaction to evaluate the scenarios with regard to social performance (Blanchard and Maneschy, 2010). From an ecological perspective, it would be interesting to extend the number of species in order to include the effects of fishing activities on the dynamics of other species (such as mammals, turtles or birds) and on plankton dynamics. In line with this, comparisons with the Ecopath (EwE) approach could be informative. Another interesting goal would be to include the effects of climatic changes, for instance sea surface temperatures (Thébaud and Blanchard, 2011). Finally, a spatial extension of this model could also be considered to integrate, for instance, the effects of protected areas.

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Appendix D. Résumé long

Cette thèse a pour principal objectif de modéliser les principaux processus biologiques et économiques régissant des pêcheries multi-espèces et multi-flottilles afin de proposer des stratégies viables pour la gestion durable de ces pêcheries mixtes, dans un contexte stochastique et multi-objectifs. Plus spécifiquement, cette thèse utilise des analyses de co-viabilité stochastique pour étudier les arbitrages entre des objectifs contradictoires de gestion (conservation, et viabilité économique et sociale) des pêcheries mixtes. Cette thèse est structurée en six chapitres. Le premier chapitre présente une introduction générale sur le bio-économie des pêcheries et l'approche de viabilité, et expose ensuite les objectifs de la thèse et son contenu. Les quatre chapitres suivants contiennent des contributions originales portant sur la gestion durable de pêcheries mixtes. Les chapitres 2 et 3 analysent l'application de l'approche de co-viabilité aux pêcheries mixtes démersales du golfe de Gascogne en France, tandis que les chapitres 4 et 5 se focalisent sur l'étude de la pêcherie crevettière australienne du Nord. Le chapitre final résume et discute les conclusions générales de la thèse.

D.1 Introduction générale

D.1.1 Contexte général

Ces dernières décennies ont été caractérisées par une dégradation accélérée de populations marines (Hutchings and Reynolds, 2004) avec des conséquences encore inconnues. Bien que la gestion durable des ressources soit un objectif important dans la plupart des pêcheries (FAO, 1999; Heino and Enberg, 2008), la gestion des pêches par des mesures traditionnelles de conservation n'a pas réussi à maintenir les stocks de poissons à des niveaux durables. Worm et al. (2009) ont en effet estimé que 63% des populations mondiales de poissons évaluées requièrent une reconstitution de leurs stocks ; tandis que seulement 22% des pêcheries mondiales sont considérées durables (UN, 2008). Dans ses évaluations les plus récentes, la FAO confirme les tendances observées depuis plusieurs décennies sur la surexploitation et la diminution des ressources halieutiques (FAO, 2012).

En 2009, 57,4% des stocks mondiaux étaient pleinement exploités, autrement dit ces stocks ont atteint le niveau de production maximale à l'équilibre (Maximum Sustainable Yield, MSY), 29,9% des stocks étaient surexploités, et seulement 12,7% étaient non pleinement exploités.

Les pêcheries sont sujettes à « la tragédie des communs » (Gordon, 1954), ce qui est une des raisons expliquant la surexploitation. Ceci résulte de la nature d'accès libre des ressources halieutiques (Gordon, 1954; Hardin, 1968). Les externalités négatives créées par chaque pêcheur sur le reste de la flottille incitent à être le premier à capturer le poisson. Chaque individu tend alors à maximiser sa part prise sur la ressource commune. Une des conséquences de cette « course au poisson » est la surcapacité, où les flottilles de pêche deviennent trop grandes par rapport au potentiel de production des stocks de poissons et la surexploitation des stocks (Beddington et al., 2007). Surcapacités et surexploitations ont pour conséquence une perte économique pour la société. Cette perte a été estimée par Arnason, Kelleher et Willman (Bank and FAO, 2009) comme la différence entre le bénéfice actuel et le bénéfice potentiel de l'exploitation des ressources marines, et selon leurs estimations elle varierait entre 24 et 72 milliards US$ par année (avec une moyenne de 50 milliards US$) à l'échelle mondiale. Cette perte pourrait toutefois être évitée si les stocks étaient reconstruits à des niveaux plus élevés, les capacités de captures réduites et les subventions perverses arrêtées.

De nombreuses parties prenantes sont généralement impliquées dans la pêche : pêche industrielle, artisanale, de subsistance et récréative, les fournisseurs et les travailleurs dans les industries connexes, les gestionnaires, les scientifiques, les environnementalistes, les économistes, les décideurs publics ou le grand public (Hilborn, 2007b). Chacun de ces groupes a un intérêt particulier, et ces divers intérêts peuvent entrer en conflit (Hilborn, 2008). Il est alors de plus en plus largement accepté que des approches de gestion plus intégrées sont nécessaires en comparaison aux gestions traditionnelles centrées sur un secteur et une espèce unique. Ces approches intégrées visent à prendre en compte les interactions entre les pêcheries et leur environnement au sens large, englobant la préservation de la biodiversité marine, mais aussi des objectifs économiques et sociaux, ceci à travers un large éventail d'intérêts (FAO, 2003; Pikitch et al., 2004; Nomura, 2008; Kempf, 2010). Dans cette optique, de nombreux scientifiques et intervenants préconisent une approche écosystémique des pêches (AEP). La FAO (2003) définie l'AEP comme une approche « s'efforçant d'équilibrer des objectifs sociétaux divers, en tenant compte des connaissances et des incertitudes sur les composantes biotiques, abiotiques et humaines des écosystèmes et de leurs interactions, et en appliquant une approche intégrée de la pêche dans des limites écologiques significatives ».

La mise en œuvre de la gestion écosystémique nécessite des outils d'aide à la décision pour mettre en évidence les arbitrages au sein d'options de gestion, en évaluant leurs impacts biologiques, sociaux et économiques. Compte tenu de la complexité des pêcheries, ces outils d'aide à la décision sont de plus en plus basés sur des modèles de simulation informatique.

D.1.2 Modèles bio-économiques

L'interdépendance entre la ressource halieutique et les utilisateurs explique l'apparition de modèles mathématiques combinant à la fois la biologie et l'économie. Les modèles bio-économiques (combinant des considérations biologiques et économiques) ont été développés dans la pêche depuis le milieu des années 1950 (Gordon, 1954; Schaefer, 1954). Ces modèles, dont le but est d'intégrer les activités humaines avec des modèles écologiques, peuvent montrer les implications de décisions de gestion (Anderson and Seijo, 2010) et sont de plus en plus utilisés comme un outil d'aide à la décision pour la gestion des pêches.

Comme l'ont souligné Mardle et al. (2002), pour élaborer des politiques de gestion de pêches, les cibles à atteindre doivent être définies. Toutefois, les cibles de capture pour la gestion des pêcheries varient à travers le monde. La production maximale équilibrée (Maximum Sustainable Yield, MSY) - c'est-à-dire le niveau de stock pour lequel sa croissance naturelle est au maximum - est un point de repère essentiel pour la gestion des pêcheries. Toutefois la production économique maximale (Maximum Economic Yield, MEY) est également considérée comme une norme pour les stratégies de reconstitution des stocks. L'avantage du MEY par rapport au MSY est à la fois économique et biologique. Le MEY est sensible aux changements dans les conditions économiques telles que les prix et les coûts de production, et correspond par définition à une maximisation du profit global associé à l'exploitation. De plus, le MEY peut être associé à une plus grande taille du stock à l'équilibre que le MSY. Toutefois, la gestion des pêches au MEY reste controversée et l'opérationnalisation du MEY nécessite un engagement fort de l'industrie (Dichmont et al., 2010).

Il a également été reconnu que l'utilisation rationnelle des ressources halieutiques dans le temps devrait intégrer le risque et l'incertitude qui caractérisent les pêcheries (Garcia, 1996; Hilborn and Peterman, 1996). La mise en œuvre de la gestion du risque est donc cruciale pour réduire le risque d'effondrement des ressources marines et de communautés de pêcheurs. Le principe de précaution a alors été suggéré. Il implique l'application d'une vision prudente en tenant compte des incertitudes et la nécessité d'agir avec une connaissance incomplète (Garcia, 1996). L'approche est utilisée par le Conseil International pour l'Exploration de la Mer (CIEM) pour la fixation de quotas de pêche. L'utilisation de points de référence est également préconisée comme un guide pour la gestion des pêches (Caddy and Mahon, 1995).

La gestion des pêches est généralement caractérisée par de multiples objectifs, dont certains peuvent être contradictoires (Crutchfield, 1973; Charles, 1989). Bien que les objectifs de gestion ne soient souvent pas clairement établis, Charles (1989) résume certains des objectifs les plus souvent mis en avant : (i) la conservation des ressources, (ii) le maintien de la viabilité des communautés de pêche, (iii) la production alimentaire, (iv) la production de richesse économique, (v) la génération de revenus raisonnables pour les pêcheurs, et (vi) le maintien de l'emploi pour les pêcheurs.

Une question importante est donc de déterminer les procédures de gestion qui donnent des résultats acceptables par rapport à cette diversité d'objectifs associés au développement durable, tout en étant robustes aux incertitudes (De Lara and Martinet, 2009).

D.1.3 Approche de viabilité

L'approche de viabilite a été proposée en tant qu'approche alternative pouvant aider à satisfaire des objectifs multiples sous incertitude. La théorie de la viabilité, développée en mathématique à partir des travaux de Jean-Pierre Aubin Aubin (1990), vise à contrôler les systèmes dynamiques (en particulier avec des problèmes de contrôle non linéaires) dans le but de les maintenir à l'intérieur d'un ensemble donné d'états admissibles, appelé ensemble des contraintes de viabilité. Bien que les problèmes d'optimisation dynamique soient habituellement formulés sous contraintes, le rôle joué par ces contraintes pose des problèmes techniques difficiles et n'est généralement pas abordé comme un problème spécifique (De Lara and Doyen, 2008). En outre, la procédure d'optimisation réduit la diversité des formes possibles de l'évolution en sélectionnant, en général, une seule trajectoire (De Lara and Doyen, 2008). L'analyse de viabilité, au lieu de maximiser une fonction objectif unique, se concentre sur le rôle des contraintes et sur la caractérisation des chemins et des décisions viables. L'ensemble des états viables forme ce qui est appelé le noyau de viabilité : à l'extérieur de ce noyau de viabilité, il n'y a pas d'évolution possible qui puisse empêcher l'effondrement du système. Le point essentiel réside dans l'identification de la frontière du noyau de viabilité et des contrôles qui doivent être appliqués lorsque le système se déplace vers sa limite (Bene et al., 2001).

Comme la gestion de la pêche se caractérise par de multiples objectifs de gestion, l'analyse de co-viabilité, combinant diverses contraintes (telles que biologique et économique), montre la capacité des mesures de gestion à maintenir les stocks de capitaux naturels et économiques au-dessus de certains niveaux minimaux. De plus, afin de prendre en compte le risque et l'incertitude constituant les principaux problèmes en matière de gestion, les contraintes de co-viabilité doivent être articulées avec l'incertitude, dans un sens probabiliste ou stochastique.

Seuls quelques cas appliqués (Béné and Doyen, 2000; De Lara et al., 2007; Doyen et al., 2007; Martinet et al., 2007; De Lara and Martinet, 2009; Martinet et al., 2010; De Lara et al., 2011; Péreau et al., 2012) ont fait usage de l'approche de co-viabilité pour intégrer des objectifs économiques et écologiques pour la gestion des pêches. Parmi ces études, seuls Doyen et al. (2007) et De Lara and Martinet (2009) intègrent l'incertitude affectant les dynamiques biologiques. Bien que ces études intègrent plusieurs espèces dans leur analyse, la diversité au sein de l'industrie de la pêche n'est toutefois pas prise en compte.

D.1.4 Objectifs de la thèse

Les objectifs de cette thèse visent à contribuer à l'amélioration des outils de gestion pour une gestion durable de pêcheries mixtes. Cette thèse étudie plus spécifiquement l'utilisation de l'analyse de co-viabilité stochastique comme un outil décisionnel pour aider à l'évaluation des stratégies de gestion durable pour la réglementation de pêcheries mixtes, et ceci dans différents contextes. Cette analyse est développée pour deux cas d'études (un français et un australien), dont chacun affiche des caractéristiques différentes, permettant ainsi l'évaluation de l'approche pour différents écosystèmes. Conformément à la question de la durabilité, la modélisation dynamique intègre des composantes biologiques et économiques des systèmes avec incertitudes ainsi que des objectifs de gestion multiples.

Cette thèse a pour objectifs (i) le développement de modèles bio-économiques captant les processus biologiques et économiques clés qui régissent ces pêcheries, (ii) la comparaison de stratégies de pêche en termes de viabilité et de risques bio-économiques pour ces deux cas d'études, (iii) l'identification de stratégies de pêche viables, et (iv) l'identification des principaux moteurs de la viabilité écologique et économique concernant ces pêcheries.

D.1.5 Cas d'études

Deux pêcheries mixtes sont analysées dans cette thèse : la pêcherie française mixte démersale du golfe de Gascogne (BoB) et la pêcherie crevettière australienne du Nord (NPF). Ces deux pêcheries sont multi-espèces et utilisent des stratégies multiples de pêche, induisant des impacts directs et indirects sur les écosystèmes. Les deux pêcheries utilisent en effet une technologie non sélective (chalut) induisant d'importantes captures accessoires et rejets. En outre, elles sont d'un grand intérêt commercial et industriel. Par conséquent, leur gestion durable est une préoccupation sociétale majeure. Tout en s'appuyant sur une méthodologie commune, cette thèse examine les deux cas, d'une manière qui reconnait leurs différentes histoires, leurs contextes sociaux et politiques, et les différences dans les stratégies et objectifs de gestion.

Pêcherie mixte démersale du golfe de Gascogne (BoB)

La pêcherie mixte démersale du golfe de Gascogne (BoB) opère au niveau des divisions VIIIa et b de la grille du Conseil International pour l'Exploration de la Mer (CIEM) et comprend des flottilles de pêche françaises, espagnoles et belges. Cette thèse porte sur les flottilles françaises. Les principaux engins de pêche utilisés sont le chalut, le filet maillant et la palangre, qui induisent des niveaux variables d'impact sur un large éventail d'espèces. Parmi les 200 espèces pêchées dans le golfe de Gascogne, trois des plus importantes en pourcentage de la valeur des débarquements nationaux français totaux sont la langoustine (*Nephrops norvegicus*, 6%), le merlu (*Merlucius Merlucius*, 7%) et la sole (*Solea solea*, 11%). La gestion de cette pêcherie repose principalement sur des mesures de conservation : totaux autorisés de captures (TAC) révisés chaque année, tailles minimales de débarquement (MLS), et tailles minimales des mailles des chaluts. En conformité avec la réforme de la politique commune de la pêche (EC, 2009), un plan de gestion multi-spécifique pour le golfe de Gascogne devrait remplacer les anciens plans de gestion mono-spécifiques pour la sole et le merlu. Les analyses mises en œuvre dans cette thèse fournissent des indications importantes sur l'efficacité de stratégies de gestion, pouvant éclairer l'élaboration du nouveau plan de gestion (EC, 2009).

Pêcherie crevettière du Nord australien (NPF)

La pêcherie crevettière australienne du Nord (NPF) est située au large de la côte nord de l'Australie, et est une pêche au chalut ciblant plusieurs espèces de crevettes tropicales, chacune avec une biologie différente. Le principal revenu de la pêche (95% de la valeur totale annuelle des débarquements) provient d'une ressource naturellement fluctuante et imprévisible, la crevette banane blanche (*Penaeus merguiensis*), et d'une ressource plus prévisible comprenant deux espèces de crevettes tigrées (*Peaneus semisulcatus* et *Penaeus esculentus*). La NPF a été la première pêcherie en Australie à adopter pour objectif de gestion le niveau de rendement économique maximal (Maximum Economic Yield, MEY) (Woodhams et al., 2011). Cette pêche est principalement gérée par des contrôles d'entrée, sous la forme de permis échangeables d'effort (en quantités restreintes) et de fermetures saisonnières. Depuis sa création à la fin des années 1960, cette pêche a régulièrement produit des profits positifs (Rose and Kompas, 2004). Toutefois, elle a connu ces dernières années une baisse de valeur en raison de l'offre accrue de crevettes d'élevage aquacole, d'une monnaie australienne forte sur le marché des changes et de l'augmentation des prix du carburant (Punt et al., 2011). Le modèle bio-économique et l'analyse de co-viabilité développés dans cette thèse permettent l'exploration de nouvelles stratégies de gestion, et ceci pour différents scénarios économiques.

D.2 Chapitre 2

Le cas français de la pêcherie mixte démersale du golfe de Gascogne est présenté dans le chapitre 2 avec une focalisation sur les pêcheries de langoustines et de merlus à travers un modèle bio-économique multi espèces, multi flottilles, structuré en âge avec un pas de temps annuel, et intégrant de l'incertitude au niveau du recrutement. Les interactions entre les deux flottilles ont été prises en compte par les captures accessoires de jeunes merlus par la flottille langoustinière. L'intégration d'une telle complexité constitue une première étape vers le développement d'une gestion écosystémique des pêches (AEP) pour les pêcheries mixtes. Le modèle bio-économique a été utilisé afin d'étudier, à travers une analyse de co-viabilité stochastique (CVA), les performances bio-économiques de la pêcherie au sein d'un ensemble de contraintes liées au maintien de la biomasse du stock reproducteur (SSB) de merlu au-dessus d'une biomasse de précaution, et à la garantie de profits strictement positifs pour chacune des deux flottilles de pêche merlu et langoustine.

Etant donné l'ensemble des valeurs des paramètres et les contraintes de viabilité adoptées dans cette analyse, il est montré que la stratégie du statu quo, consistant à maintenir l'intensité de pêche au niveau de référence de 2008, peut être considérée comme non économiquement viable. En effet, tandis que les profits de la flottille de pêche merlu sont pour la plupart du temps positifs, les profits de la flottille langoustinière sont négatifs pour au moins une année de chaque scénario de recrutement. En outre, même si la viabilité biologique est plus élevée que la viabilité économique, les trajectoires des biomasses reproductrices de merlu passent souvent sous le seuil de précaution. Les analyses de viabilité ont néanmoins révélé l'existence d'espaces de contrôle viable sans risque bio-économique, où la probabilité de maintenir la co-viabilité du système est proche de 100%. Les stratégies viables de gestion, basées sur le modèle présenté dans ce chapitre, montrent qu'une réduction sévère de l'effort de pêche des chalutiers langoustiniers pourrait être nécessaire afin de garantir la co-viabilité du système.

La pêcherie mixte démersale du golfe de Gascogne est toutefois composée de plusieurs sous-flottilles, avec des effets biologiques et des structures de coûts différents, qui ne sont pas représentées dans ce chapitre. Par conséquent, cette analyse ne permet pas d'étudier les effets distributifs de stratégies alternatives de gestion lorsque plusieurs sous-flottilles et plus d'espèces sont comptabilisées.

D.3 Chapitre 3

Le chapitre 3 étend l'analyse du chapitre 2 en ajoutant une autre espèce à l'étude, la sole, et en incluant d'avantage de flottilles dans le modèle. Dans cette étude, les navires de la flotte française sont décrits de façon plus précise que dans le chapitre 2. Les différents navires sont en effet répartis en seize groupes et ceci en fonction de l'utilisation de leur engin principal, de la structure de leurs débarquements, de leurs coûts et de leur classe de longueur. De plus, comme l'ont souligné Garcia (1996) et Hilborn and Peterman (1996), les pêcheries évoluent dans des contextes très incertains. L'incertitude sur l'abondance des ressources, prise en compte dans le chapitre 2, fait partie de différentes sources d'incertitude identifiées. Cependant, d'autres sources d'incertitude existent, notamment l'incertitude sur les futures conditions économiques, incluant la variabilité économique des prix du marché et des coûts. Par conséquent, il était important d'intégrer des scénarios économiques dans l'analyse de cette pêcherie pour permettre une étude plus approfondie des performances bio-économiques de stratégies de gestion alternatives. Ce chapitre inclut donc un scénario relatif aux prix du carburant et introduit de la variabilité au niveau des prix de vente des débarquements.

Les probabilités de viabilité du système sont comparées entre des stratégies centrées sur la gestion séparée de la pêche langoustinière et du merlu (comme cela est historiquement le cas), une stratégie visant à maximiser le rendement économique de l'ensemble de la pêcherie (stratégie NPV) et des stratégies de gestion multi-spécifiques (stratégies CVA). Les stratégies de gestion multi-spécifiques visent à ajuster la capacité de pêche par sous-flottille afin que les contraintes de viabilité biologique et économique puissent être satisfaites pour chacune des trois espèces et des seize sous-flottilles. Les simulations permettent, par rapport au chapitre 2, de comparer les arbitrages associés aux différentes allocations d'effort de pêche au sein des seize sous-flottilles. D'après les simulations, il apparaît que le rôle des contraintes biologiques est faible par rapport aux contraintes économiques. Les stratégies de gestion mono-spécifiques améliorent le statut bio-économique de la pêche spécifique qu'elles visaient, mais ne permettent pas d'obtenir des résultats satisfaisants à l'échelle de l'ensemble de la pêcherie, puisque certaines sous-flottilles présentent des profits négatifs. De plus, la stratégie visant à maximiser la valeur actuelle nette (NPV) de l'ensemble de la pêcherie conduit à des impacts hétérogènes au sein des sous-flottilles, impliquant même des fermetures pour certaines d'entre elles. Une certaine résistance, de la part des segments et des régions de la pêcherie qui seraient touchés négativement, est donc à attendre face à une telle stratégie de gestion. En revanche, même si les stratégies de gestion multi-spécifiques CVA mettent en évidence une nécessité de réduire la capacité de pêche globale de la pêcherie, les combinaisons d'ajustements de taille des sous-flottilles fournies par ces stratégies sont moins drastiques que celles associées à la stratégie de gestion NPV et permettent de satisfaire les contraintes biologiques et socio-économiques pour chaque espèce et sous-flottille.

D.4 Chapitre 4

Afin d'explorer les aspects généraux de la gestion de pêcheries mixtes et d'évaluer l'aspect transposable de l'approche de co-viabilité stochastique, une autre pêcherie mixte, la pêcherie crevettière australienne du Nord (NPF), a été étudiée dans cette thèse. Le chapitre 4 présente ce cas d'étude.

Les revenus provenant de la pêche peuvent varier considérablement d'année en année en raison de la variation naturelle des stocks de poissons (Kasperski and Holland, 2013), conduisant à des niveaux de risques économiques variables pour les opérateurs de pêche (Sethi, 2010), ce qui constitue une question importante pour la gestion des pêcheries mixtes. Un point important pour la gestion de la NPF vient du fait que ses revenus proviennent d'une ressource naturellement fluctuante et imprévisible, la crevette banane blanche, et d'une ressource plus prévisible comprenant deux crevettes tigrées (« grooved » et « brown »). La pêcherie fonctionne avec deux principales stratégies de pêche ciblant soit les crevettes bananes, soit les crevettes tigrées. Alors que les articles publiés portent essentiellement sur les méthodes de pêche ciblant les crevettes tigrées (Dichmont et al., 2008; Punt et al., 2011), cette thèse porte sur l'ensemble de la pêcherie. Un modèle bio-économique, hebdomadaire, multi-espèces, structuré en taille et avec allocation dynamique de l'effort entre stratégies de pêche est développé. Ce modèle est décrit dans le chapitre 4. Il y est utilisé afin d'examiner les compromis entre performance économique moyenne de la pêche et variance de cette performance, au sein d'une gamme de stratégies de gestion concernant la capacité de la flotte et la répartition de l'effort entre différentes stratégies de pêche, et ceci pour différents scénarios économiques. Comme pour le cas du golfe de Gascogne dans le chapitre 3, des scénarios relatifs aux prix des crevettes et du carburant ont été intégrés dans ces analyses pour tenir compte de l'incertitude économique. Selon les simulations présentées dans cette thèse, il apparaît que la stratégie de pêche actuelle de la NPF permet un bon compromis entre performance économique moyenne et la variabilité de cette performance. Il apparaît également que l'augmentation de la taille de la flottille ou de l'effort de pêche annuel des navires n'améliorerait la performance économique de la pêche qu'au prix de l'augmentation de la variabilité de cette performance. Ce chapitre étudie la prise en compte de l'aversion au risque dans les stratégies de pêche pour la gestion de pêcheries mixtes, ce qui n'est pas analysé dans les deux chapitres précédents. Il apparait que dans le cas de la NPF, l'aversion au risque de l'industrie de pêche et des pêcheurs pourrait être déterminante dans la définition d'options de gestion. Dans le scénario économique le plus probable, les adaptations de la pêcherie pour maintenir des niveaux élevés de performance économique sont donc susceptibles de dépendre de la volonté des opérateurs de pêche d'accepter des niveaux élevés de risques économiques.

D.5 Chapitre 5

Le chapitre 5 étend le modèle bio-économique présenté dans le chapitre 4 avec l'application d'une analyse de co-viabilité stochastique afin d'évaluer les compromis dans la gestion d'une pêche au chalut, et d'identifier des stratégies viables de gestion. Les compromis entre des objectifs biologiques, économiques et de gestion de la biodiversité dans un contexte très incertain ont été plus particulièrement étudiés. La prise en compte des impacts environnementaux des activités de pêche est une préoccupation cruciale, car ces effets peuvent conduire à des changements dans la biodiversité et, éventuellement changer la fonctionnalité globale de l'écosystème (Pauly et al., 1998; Dulvy et al., 2000). Une dimension écologique supplémentaire, via la simulation de captures de serpents de mer, est donc intégrée dans ce chapitre afin d'évaluer les impacts du chalutage sur la biodiversité en général. D'après les espèces de crevettes prises en compte dans cette étude et les seuils biologiques associés, les résultats montrent que la probabilité de co-viabilité du système est plus influencée par les contraintes économiques et de réduction des impacts sur la biodiversité que par les contraintes biologiques. Par ailleurs, compte tenu des hypothèses de modélisation, l'approche de gestion basée sur le statu quo parait non viable, par rapport aux contraintes économique et de conservation de la biodiversité. L'addition d'une contrainte de conservation de la biodiversité (limitation de mortalité sur les serpents de mer) ajoute une pression supplémentaire qui conduit à une accentuation des trade-offs entre les objectifs de gestion.

Les décisions finales concernant les paramètres de gestion pour la gestion durable de la NPF dépendent des objectifs de gestion (comprenant la maximisation du rendement économique), mais aussi du niveau de l'aversion au risque de l'industrie de pêche et des pêcheurs (comme illustré dans le chapitre 4). Les trade-offs entre la performance économique moyenne de la pêcherie (via la valeur actuelle nette) et la probabilité de co-viabilité sont également étudiés dans ce chapitre afin d'identifier des stratégies de gestion permettant les meilleurs compromis entre rendements économiques élevés et faibles risques bio-économiques. Les analyses montrent que les adaptations de la pêche pour la gestion viable impliqueraient une augmentation de l'effort annuel total par navire combinée à une réduction de la capacité de la flottille.

D.6 Discussion

Ce chapitre compare les deux modèles bio-économiques développés pour les cas d'étude français et australien. Il discute ensuite des résultats obtenus, ainsi que des stratégies viables de gestion et des facteurs de viabilité identifiés. Enfin, il fournit quelques perspectives pour de futures recherches.

D.6.1 Points communs clés des modèles bio-économiques

Pour les deux applications BoB et NPF, les configurations des modèles sont différentes tout en ayant des points communs. D'une manière générale, la modélisation des deux systèmes de pêche vise à regrouper diverses composantes de la pêche qui ne sont habituellement pas réunies en un seul modèle. Les modèles BoB et NPF intègrent des dimensions multi-espèces et multi-flottilles (ou multi stratégies de pêche), ce qui est nécessaire pour aborder la question des interactions techniques. Les différentes flottilles de pêche interagissent en effet techniquement : une flottille ciblant une espèce va capturer d'autres espèces en tant que captures accessoires.

Les deux modèles sont structurés en âge (BoB) ou en taille (NPF) en fonction de la biologie des espèces clés modélisées. Les recrutements sont supposés être liés aux stocks de reproducteurs, mais en raison de différences dans les évaluations, la disponibilité des données et la biologie, différentes hypothèses ont été adoptées dans les deux cas d'études (une fonction appelée bâton de hockey ou rasoir d'Ockham (Barrowman and Myers, 2000; Mesnil and Rochet, 2010) a été utilisée pour BoB et des relations de stock-recrutement de Ricker ont été supposées dans le modèle NPF comme dans Dichmont et al. (2003) et Punt et al. (2010)). Des dimensions économiques ont également été intégrées dans les deux modèles grâce à des fonctions de production, de prix et de coûts, ce qui permet l'évaluation d'indicateurs économiques relatifs aux pêcheurs, à l'industrie dans son ensemble et aux préoccupations sociétales. Les analyses des pêcheries BoB et NPF incluent des contraintes représentant l'objectif de conserver les stocks reproducteurs d'espèces cibles et de maintenir des profits économiques positifs ou supérieurs à un certain niveau. Les effets économiques des prises accessoires sont indirectement comptabilisés dans le cas d'étude BoB via des revenus brut provenant d'autres espèces capturées. D'un autre côté, les captures de serpents de mer reliées à l'effort de pêche sont explicitement incluses dans le modèle NPF, comme une variable sur laquelle une contrainte peut s'exercer et qui reflète les objectifs de conservation. Cette thèse montre qu'il est possible de prendre en compte plus pleinement la complexité des problèmes de gestion des ressources (Pahl-Wostl, 2007; Plagányi et al., 2012), en utilisant une modélisation intégrée comprenant seulement les niveaux nécessaires de complexité, tout en étant fondée sur une représentation proche de la réalité.

Les modèles bio-économiques incluent des incertitudes sous-jacentes aux processus écologiques et économiques considérés. La variabilité interannuelle du recrutement étant une forme importante de variabilité naturelle de la pêche (Francis and Shotton, 1997), elle a été prise en compte à la fois dans le modèle BoB via une distribution uniforme des recrutements de merlu, sole et langoustine et dans le modèle NPF via des corrélations temporelles avec stochasticité environnementale pour les crevettes tigrées et « endeavour ». En plus de la stochasticité environnementale, les pêcheries sont sujettes à l'incertitude économique avec de fortes influences de facteurs à la fois internes et externes. Ceci est particulièrement le cas pour les prix du carburant. Le scénario de prix du carburant supposé dans le cas BoB s'appuie sur des projections de l'agence internationale de l'énergie (IEA, 2010; CAS, 2012), tandis que le scénario de prix du carburant dans le cas NPF est basé sur une régression de données historiques. Une tendance commune à la hausse des prix du carburant est postulée dans les deux cas, cependant, les scénarios sont différents, puisque les cas d'étude se rapportent à des contextes différents en termes de politiques et de taxes nationales par exemple, et ne sont pas affectés de la même manière par les influences mondiales. Les prix de marché pour les poissons de ces études sont basés sur des données historiques. Des incertitudes sur ces prix ont été introduites d'une manière différente selon le cas d'étude, les marchés des deux pêcheries n'étant pas touchés de la même façon par les facteurs externes et internes.

Les résultats montrent que la variabilité biologique intégrée dans les deux modèles a un impact important sur les résultats économiques. La variabilité des profits annuels du cas BoB provient des effets combinés des incertitudes biologiques et sur les prix de marché. Dans le cas de la NPF, la variabilité des profits annuels découle de la variabilité biologique, et si l'on considère la vaste gamme de profits annuels possibles, il apparait que la variabilité biologique a un rôle crucial pour l'évaluation des risques bio-économiques.

D.6.2 Analyses de viabilité et risques bio-économiques

Des analyses de co-viabilité stochastiques (CVA) ont été appliquées aux deux cas d'études afin d'étudier comment les pêcheries peuvent fonctionner dans un ensemble de contraintes liées à plusieurs objectifs de gestion, et ceci dans des contextes incertains.

La gestion des pêches est souvent caractérisée par de multiples objectifs qui peuvent être parfois contradictoires (Crutchfield, 1973; Charles, 1989). L'analyse de co-viabilité permet une approche formelle pour examiner cet aspect de la gestion des pêches (Bene et al., 2001; Eisenack et al., 2006; Doyen et al., 2007; Martinet et al., 2007, 2010). Les critères multiples analysés dans cette thèse sont liés à des objectifs de gestion biologiques, socio-économiques et de préservation de la biodiversité. Dans les analyses présentées, un objectif important est lié à la viabilité économique (EVA). De plus, l'analyse de co-viabilité stochastique (Baumgärtner and Quaas, 2009; De Lara and Martinet, 2009) a été utilisé dans cette thèse pour aborder les questions de risque et de précaution et afin de déterminer la capacité des pêcheries BoB et NPF d'atteindre des ensembles spécifiques d'objectifs de développement durable, avec une probabilité suffisamment élevée.

Les revenus provenant de la pêche peuvent varier considérablement d'année en année en raison de la variation naturelle des stocks de poissons (Kasperski and Holland, 2013). Cette variation ne peut pas être prédite avec fiabilité, ce qui conduit à différents niveaux de risques économiques pour les opérateurs de pêche (Sethi, 2010). Les analyses de moyenne-variance pour la NPF (chapitre 4) décrivent un important compromis entre la performance économique moyenne attendue et la variance interannuelle associée (risque économique). L'aversion au risque de la pêcherie reflète le fait que les intervenants de l'industrie apprécient généralement une certaine stabilité des performances économiques dans le temps. C'est pourquoi les stratégies de gestion avec des risques bio-économiques réduits peuvent être favorisées. Mais cela peut impliquer une perte économique, puisque certaines stratégies qui pourraient entraîner des niveaux élevés de performance économique moyenne (au prix d'une plus grande variabilité) seraient alors exclues en tant qu'options viables.

La question de savoir quels seuils et quel niveau de confiance choisir demeure cruciale dans les analyses de viabilité, surtout dans des contextes où incertitudes environnementales et économiques coexistent. Ceci est mis en lumière par les résultats des analyses moyenne-variance du chapitre 4. Par exemple, dans le scénario économique le plus probable, les adaptations de la NPF pour maintenir les niveaux actuels de performance économique sont susceptibles de dépendre de la mesure selon laquelle les opérateurs de pêche acceptent des niveaux de risques économiques plus ou moins élevés. De plus la difficulté de fixer les seuils de viabilité biologique, en particulier dans les analyses sur le cas d'étude australien, souligne la nécessité d'une meilleure information écologique afin d'identifier des seuils appropriés.

D.6.3 Stratégies de pêche et facteurs de co-viabilité

L'approche de modélisation proposée dans cette thèse permet d'évaluer les résultats biologiques, socio-économiques et écologiques, ainsi que les performances en terme de viabilité globale de différentes stratégies de pêche et de les comparer directement. Les stratégies de pêche évaluées incluent une stratégie de statu quo (c'est-à-dire le maintien des paramètres de gestion des pêcheries d'une année de référence), des stratégies de gestion mono-spécifiques (BoB) ou de spécialisation de l'effort de pêche sur une espèce (NPF), mais aussi des stratégies de diversification (NPF), ou des stratégies de gestion définies pour l'ensemble des espèces et des sous-flottilles (BoB), comprenant des stratégies visant à maximiser la rente actuelle nette des pêcheries.

Les gestions actuelles des pêcheries BoB et NPF ont été analysées afin d'identifier si ces pêcheries mixtes pourraient être durables, compte tenu des critères de durabilité biologique, économique et de biodiversité définis dans l'analyse. Sur la base des simulations et des hypothèses du modèle BoB, il semble y avoir une surcapacité de la flotte dans son ensemble, et en particulier pour les chalutiers langoustiniers. Les résultats pour la NPF soulignent une variabilité économique interannuelle importante en raison de la variabilité biologique. Les simulations NPF montrent également que la réduction du nombre de navires actifs pourrait aider la pêcherie à réduire son risque économique tout en maintenant son niveau actuel de profit, en particulier dans le scénario économique futur le plus probable.

Des stratégies reposant sur des approches de gestion standard ont été étudiées afin d'identifier et de comparer des options alternatives pour la pêche. Un ensemble de stratégies de gestion mono-spécifiques a été étudié pour la pêcherie BoB (chapitre 3), qui correspond à l'approche adoptée en pratique jusqu'à une période récente. La stratégie ciblant seulement la gestion de la langoustine produit des résultats meilleurs à l'échelle de la pêcherie (par rapport à une stratégie ciblant seulement la gestion de la sole, et à une stratégie statu quo) en ce qui concerne la performance socio-économique des sous-flottilles, mais entraine tout de même des profits négatifs pour certaines sous-flottilles, ainsi qu'un niveau modéré de risque biologique (concernant le merlu et la sole). Les analyses NPF décrivent les niveaux de risques associés à des stratégies d'allocation d'effort dites de « spécialisation » ou de « diversification ». D'après les simulations, il apparait que les stratégies de « spécialisation crevettes tigrées » (qui distribuent l'effort total annuel plus vers les crevettes tigrées que vers les crevettes bananes) donnent de moins bons résultats (en termes de performance économique moyenne et de probabilité de co-viabilité), que les stratégies de répartition équitable de l'effort entre crevettes tigrées et bananes, ou adaptative, lorsque la capacité de pêche est semblable ou supérieure à celle actuellement observée. Cependant, dans un contexte de diminution du nombre de navires, des stratégies de « spécialisation crevettes tigrées » auraient une meilleure performance en termes de viabilité économique que d'autres stratégies. Ceci découle du fait qu'une flotte réduite semble plus compatible avec des stratégies axées sur une ressource moins fluctuante et plus prévisible (c'est à dire les crevettes tigrées).

Il semble que pour les deux cas d'étude, des stratégies viables de gestion, où toutes les contraintes de viabilité définies sont satisfaites avec des probabilités assez élevées (supérieures à 95%), peuvent être identifiées. Les chapitres 2 et 3 sur la pêcherie BoB montrent que les stratégies CVA (c'est-à-dire de gestion multi-espèces et multi-flottilles) conduisent à des situations « gagnant-gagnant » où, pour un certain nombre de combinaisons d'ajustements de capacité des sous-flottilles, les contraintes de viabilité biologiques et socio-économiques sont respectées pour chaque espèce et sous-flottille. Selon ces analyses, une gestion multi-espèces est donc nécessaire pour garantir une gestion viable de la pêcherie française démersale mixte du golfe de Gascogne. Il est également possible d'identifier pour la NPF un petit « espace de négociation », en termes de capacité de pêche et de répartition de l'effort de pêche, où toutes les contraintes biologique, économique et de conservation de la biodiversité peuvent être respectées avec des probabilités élevées. Ce consensus est associé à une diminution du nombre de navires, comme c'est le cas dans la pêcherie BoB.

Une stratégie économique plus conventionnelle qui maximise la valeur actuelle nette (NPV) des pêcheries a également été étudiée pour comparaison avec les stratégies de gestion CVA, en termes de viabilité et de performances économiques moyennes. Dans le cas du golfe de Gascogne, cette stratégie économique classique améliore la performance économique globale de la pêcherie, mais conduit à des impacts hétérogènes sur les sous-flottilles, incluant l'arrêt de la pêche pour certaines d'entre elles. D'une manière générale, les stratégies de gestion menant à la plus haute NPV, sont associées à des probabilités de co-viabilité nulles , tandis que les stratégies CVA conduisent à une probabilité de co-viabilité plus élevée, mais sont liées à une réduction de la valeur actuelle nette de la pêcherie. Un compromis entre performance économique moyenne en termes de NPV et probabilités de co-viabilité a également été observé pour la NPF. Les gestions de capacité de pêche et les stratégies de répartition de l'effort conduisant à une augmentation du NPV sont associées à une réduction importante des viabilités écologique et économique (telles que définies dans cette étude). La diminution de la viabilité économique dans le cadre d'une stratégie maximisant la NPV reflète une plus grande variabilité interannuelle et des violations de l'objectif d'équité interannuelle.

Les analyses présentées dans cette thèse soulignent que de fortes probabilités de co-viabilité peuvent être atteintes, mais seulement au prix d'une réduction du rendement économique par rapport à une stratégie maximisant la valeur actuelle nette de la pêcherie. Cette perte économique peut être interprétée comme un « coût de la durabilité » associé à l'objectif de satisfaire toutes les contraintes imposées à la pêcherie, à savoir le coût d'opportunité, en termes de rendements économiques globaux, d'augmenter les probabilités de co-viabilité.

D.6.4 Perspectives

Les modèles développés dans cette thèse prennent en compte un nombre limité d'espèces dans chacun des cas d'étude. De plus, l'intégration explicite des flottilles espagnoles et belges dans les analyses portant sur le golfe de Gascogne permettrait une analyse plus détaillée et une discussion plus poussée des aspects distributifs de la gestion de cette pêcherie. Dans le cas de la NPF, d'autres espèces ciblées par la pêche ont de plus en plus d'importance économiquement, il peut alors devenir intéressant de les considérer dans de futures modélisations, étant donné que la valeur des captures des crevettes diminuerait (dans le scénario économique le plus probable).

Les dynamiques de crevettes bananes blanches sont fortement imprévisibles et principalement influencées par des facteurs environnementaux. Il a été montré dans cette thèse que cette grande variabilité a des conséquences économiques et est donc importante pour la gestion. La capacité à modéliser avec précision les biomasses de crevettes bananes est actuellement limitée, mais reste cruciale pour la gestion de la NPF. Compte tenu de l'importance des facteurs environnementaux, les impacts potentiels du changement climatique pourraient être pris en compte dans de futures extensions de ces analyses.

L'aversion au risque est une dimension importante dans l'évaluation de stratégies de gestion. L'analyse présentée dans cette thèse démontre qu'un certain degré d'aversion au risque semble caractériser la NPF. De plus, il y a actuellement, dans le golfe de Gascogne, des contraintes sur la variation interannuelle maximale des totaux admissibles de captures (TAC) qui ne doit pas excéder 15% (STECF, 2011) ; ce qui est avantageux pour les pêcheurs pour la planification de stratégies et d'investissements futurs. Même si l'aversion au risque économique n'a pas été explicitement introduite dans les analyses de viabilité des deux cas d'études, l'approche de viabilité peut être un moyen d'intégrer les attitudes face au risque dans les résultats économiques qui en résultent, de par les définitions des seuils définissant les contraintes de viabilité biologiques et socio-économiques.

L'inclusion du comportement humain dans les modèles d'utilisation des ressources marines est de plus en plus considéré comme un défi majeur à résoudre pour les modélisateurs (Fulton et al., 2011). L'incertitude des réponses humaines comporte une dimension, souvent négligée, qui se rapporte à la mise en œuvre des mesures de gestion (Fulton et al., 2011). En effet même si des décisions viables de gestion sont prises, leur mise en œuvre peut être incertaine (Hennessey and Healey, 2000), et selon Angel et al. (1994), la prise en compte de ces incertitudes serait cruciale pour la gestion des pêches. Les analyses de co-viabilité stochastique, étudiées dans cette thèse, identifient un « espace de négociation » pour les parties prenantes permettant de parvenir à un consensus. Par conséquent les incertitudes de mise en œuvre pourraient être réduites dans un cadre de gestion de co-viabilité stochastique. Cela peut être très important dans la pratique, et en particulier dans un contexte de gestion écosystémique des pêches (AEP) impliquant de multiples acteurs et objectifs devant être examinés simultanément, si les outils décisionnels doivent être considérés comme utiles pour l'aide à la décision.

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