Université Paris I - Panthéon Sorbonne
U.F.R. de sciences économiques

Année 2014

Numéro attribué par la bibliothèque

|X|X|X|X|X|X|X|X|X|X|X|X|

THÈSE

Pour l'obtention du grade de
Docteur de l'Université de Paris *I*
Discipline : Sciences Economiques

Présentée et soutenue publiquement par

Esther Regnier

le 20 Juin 2014

Interactions between aquaculture and fisheries, and the viability approach to risk management in harvested ecosystems

---

Directeurs de thèse : Katheline Schubert et Michel De Lara

JURY :

Christopher Costello	professeur à l'Université de Californie, Santa Barbara
Hippolyte d'Albis	Professeur à l'Université Paris 1 Panthéon Sorbonne
Vincent Martinet	Chargé de recherche à l'INRA-Économie publique
Martin Quaas	Professeur à l'Université de Kiel (rapporteur)
Olivier Thébaud	Economiste à l'IFREMER (rapporteur)

L'université Paris I Panthéon-Sorbonne n'entend donner aucune approbation ni improbation aux opinions émises dans cette thèse. Ces opinions doivent être considérées comme propres à leur auteur.

À tous ceux qui ont rendu possible cette thèse

À ma famille, à mes amis

Contents

Acknowledgments **12**

Introduction **15**

The challenges to the sustainable management of fisheries 15

Existing management options 17

The role of governance 20

Aquaculture: a source of food security? 24

The issues faced by the aquaculture industry 24

An analysis of the impacts of aquaculture production on food fish supply and demand 27

The complexity of fisheries management in practice 34

Defining ecosystem sustainable yields under uncertainty 34

The viability approach to risk management in harvested ecosystems 36

Bibliography 40

1 Is aquaculture really an option? **47**

1.1 Introduction 47

1.2 The demand side 52

1.3 Biological interactions 53

1.4 The baseline situation: capture fishery alone 55

1.5 Introducing aquaculture 57

1.5.1 The aquaculture sector and feed fishery 58

1.5.2 The coupling 60

1.5.3 Comparison with the baseline situation 65
1.5.4 Improving the efficiency of aquaculture 67
1.6 Extension: Endogenous consumer tastes 69
1.7 Conclusion 71

Appendix A **78**
A.1 Proof of Proposition 1, (i) and (iii) 78
A.2 Stability 82
A.2.1 No fishing 82
A.2.2 Capture fishery alone 82
A.2.3 Capture fishery and aquaculture: proof of Proposition 1, (ii) 83
A.3 Proof of Proposition 2 85
A.4 Proof of Propositions 3 and 4 86
A.4.1 Proof of Proposition 3 86
A.4.2 Proof of Proposition 4 88

2 Competition between farmed and wild fish: the French sea bass and sea bream markets **89**
2.1 Introduction 89
2.2 The sea bass and sea bream French markets 95
2.3 The data set 97
2.3.1 Descriptive statistics 99
2.4 The theoretical framework 101
2.5 Empirical methodology 102
2.5.1 The integration order of a time series 103
2.5.2 The concept of cointegration as a market delineation approach 104
2.5.3 The Error Correction Model representation 107
2.5.4 The Johansen multivariate cointegration test 107
2.5.5 The weak exogeneity test 108
2.6 Empirical results 109
2.6.1 Unit root tests 109

2.6.2 Competition between wild and farmed fish within the sea bream and sea bass markets . . . 111
Testing for weak exogeneity of the wild and farmed sea bream 112
2.6.3 Competition between wild and farmed fish across different species . . 113
2.6.4 Assessing the effect of product form on market integration 114
2.7 Discussion . . . 115
2.8 Conclusion . . . 121
Bibliography . . . 122

Appendix B **129**
B.1 Pairwise price comparisons . . . 129
B.2 Robustness checks . . . 132

3 Managing harvested ecosystems under uncertainty: the viability approach 136
3.1 A review of different approaches to risk management in fisheries 136
3.1.1 The economics approach . . . 137
Fisheries resources economics . . . 140
3.1.2 The operational approach . . . 144
3.1.3 The viability approach . . . 145
3.2 The stochastic viability approach . . . 150
3.2.1 A generic ecosystem model with uncertainty and the associated sustainability constraints . . . 151
3.2.2 Viable scenarios . . . 152
3.2.3 Viability probability and ecosystem viable yields under risk 153
3.2.4 Application to the anchovy–hake Couple in the Peruvian Upwelling Ecosystem (1971–1981) . . . 155
3.2.5 Lotka-Volterra dynamical model with uncertainties 155
3.2.6 Uncertainty sets and probability distributions of uncertainty scenarios 156
3.2.7 Maximum viability probability of satisfying production and preservation requirements in the Peruvian anchovy–hake fishery 160
3.3 The robust viability approach . . . 167

3.3.1 The robust viability kernel . . . 168
3.3.2 The deterministic viability kernel . . . 169
3.3.3 Application to the anchovy-hake Couple in the Peruvian Upwelling Ecosystem (1971–1981) . . . 170
3.3.4 Choice of uncertainty sets . . . 170
Uncertainty sets reduced to extreme values . . . 170
3.3.5 Discussion on the viability kernels . . . 171
Deterministic viability kernel, robust viability kernel and empirical uncertainties . . . 171
Robust viability kernel and extreme uncertainties . . . 176
3.3.6 Robust harvesting strategies . . . 178
3.4 Conclusion . . . 181
Bibliography . . . 183

Appendix C **190**
C.1 Numerical computation of maximal viability probabilities . . . 190
C.2 The deterministic viability kernel . . . 191
C.3 Numerical computation of robust viability kernels . . . 193
C.3.1 Dynamic programming equation . . . 194
C.3.2 Numerical resolution of the dynamic programming equation . . . 195

4 Conclusion **196**
Bibliography . . . 199

List of Figures

1.1 Existence, uniqueness and stability of an interior steady state (shaded zone: instability) 63

1.2 Introducing aquaculture may reduce the long run wild edible fish catch, total catch and welfare (dashed curves: absent aquaculture; plain curves: with aquaculture) 68

1.3 Long run effects of an improvement of aquaculture's efficiency, absent biological interactions (dashed curves: productivity effect alone; plain curves: productivity and preference effects) 71

A.1 Existence and uniqueness of an interior solution when $d_1 < \overline{d}_1$ 79

A.2 Existence and uniqueness of an interior solution when $d_1 > \overline{d}_1$ 80

A.6 Interior solutions 81

B.1 Prices of whole sea bass and sea bream in logarithms: within species comparison (2007-2012) 130

B.2 Prices of whole sea bass and sea bream in logarithms: between species comparison (2007-2012) 130

B.3 Prices of sea bass and sea bream in logarithms: between fish form comparison (2007-2012) 131

B.4 Dispersion of the eigenvalues: wild/farmed sea bream & sea bass VECMs . . 135

3.1 Estimated and simulated biomasses over 1971–1981 158

3.2 Empirical uncertainties $(\bar{\varepsilon}_y(t), \bar{\varepsilon}_z(t))_{t=t_0,\ldots,T-1}$ characterized in (3.11) 159

3.3 Uncertainty sets $\mathbb{S}_E$ (diamonds) and $\mathbb{S}_{ER}$ (grid) 159

3.4 Density dependent probability distribution $\mathbb{P}_{dd}$ of uncertainties in $\mathbb{S}_E$ 160
3.5 Maximal viability probability w.r.t. various minimal catch levels of anchovy and hake, and uncertainty sets $\mathbb{S}_E$ and $\mathbb{S}_{ER}$ equipped with the uniform probability distribution $\mathbb{P}_u$ 163
3.6 Iso-probability curves w.r.t. $\mathbb{S}_E$, equipped with the uniform probability distribution 164
3.7 Maximal viability probability w.r.t. various minimal catch levels of anchovy and hake, the uncertainty set $\mathbb{S}_E$ and probability distributions $\mathbb{P}_{dd}$ 166
3.8 Maximal viability probability w.r.t. minimal biomass levels of anchovy and minimal catch levels of hake, the uncertainty set $\mathbb{S}_E$ and the uniform probability distribution 167
3.9 Deterministic and robust viability kernels for the uncertainty sets $\mathbb{S}_E$ and $\mathbb{S}_{ER}$ 172
3.10 Deterministic, stochastic and robust viability kernels for the uncertainty set $\mathbb{S}_E$ 175
3.11 Comparing the deterministic and robust viable kernels for uncertainty sets $\mathbb{S}_M$ and $\mathbb{S}_H$ 178
3.12 A robust effort strategy $(v_y, v_z) = \gamma_t(y, z)$ for $t = 0$, ecosystem dynamics in (3.10) and uncertainty set $\mathbb{S}_E$ 180
3.13 Possible robust effort strategies w.r.t. biomass couples in $\mathbb{V}\text{iab}_E^R(t_0)$ 181

List of Tables

1.1 Calibration 65
1.2 Comparison with the baseline 66

2.1 French national consumption of sea bass in tons 96
2.2 French national consumption of sea bream in tons 96
2.3 Mean yearly consumption of fresh sea bass and sea bream of Kantar WorldPanel sample of households in France: January 2007–September 2012 99
2.4 Mean price of fresh sea bass and sea bream in France, Kantar WorldPanel household purchases: January 2007–September 2012 100
2.5 ADF tests for unit roots in price series of fish 110
2.6 KPSS tests of stationarity over price series of fish 110
2.7 Within species bivariate Johansen tests between wild and farmed fish 111
2.8 Sea bream model: the estimated cointegration vector 112
2.9 Sea bream model: adjustment parameters statistical significance 113
2.10 Cross species bivariate Johansen tests between farmed and wild fish 113
2.11 Farmed sea bass versus wild sea bream: the estimated cointegration vector . 114
2.12 Bivariate Johansen test between fish of different form 115
2.13 Market integration outcomes 116

B.1 Mean-comparison test: wild versus farmed sea bass 129
B.2 Mean-comparison test: wild versus farmed sea bream 129
B.3 Lagrange-multiplier test: wild/farmed sea bream VECM 132
B.4 Lagrange-multiplier test: wild/farmed sea bass VECM 132
B.5 Jarque−Bera test: wild/farmed sea bream VECM 133

B.6 Jarque−Bera test: wild/farmed sea bass VECM 133
B.7 Eigenvalue stability condition: wild/farmed sea bream VECM 134
B.8 Eigenvalue stability condition: wild/farmed sea bass VECM 134

3.1 Parameters of the Lotka-Volterra model (3.10) 156

Acknowledgments

Foremost, I am very grateful to Katheline Schubert, for accepting to supervise me, and for the reliability of her supervision along these PhD years. Due to her interest and openness to the research project I proposed, she gave me the chance to carry this fulfilling professional experience. Working with her was a prime opportunity to gain expertise in resource economics, technical skills, and to understand the conduct of academic research. Beside, she always managed to make herself available, with kindness, despite the continuous requests to which she is prone.

I thank Michel De Lara for his supervision. Especially, for initiating me to computer programming and to the viability theory. I consider my training in viability theory as a valuable knowledge; I view this theoretical framework as a powerful analytical tool to address dynamic control problems. Initially, I thought Michel's working methods were demanding but they turned out to be efficient. I am grateful for the analytical and technical skills that he enabled me to acquire. Having Michel as a supervisor also gave me the opportunity to spend time in a engineer lab and learned about their research fields, as well as to meet new colleagues. I thank Jean-Christophe, Pauline, Thomas and Vincent for being so friendly to me and for giving me a place in their group of doctorate students.

I thank Basak Bayramoglu for our shared work experience, as co-authors. I found this collaboration efficient and very pleasant. I hope we have the opportunity to renew it in the future.

I am grateful to the five members of my PhD committee, Christopher Costello, Hippolyte d'Albis, Vincent Martinet, Martin Quaas and Olivier Thébaud for their time and involvement in my defense. They are among the researchers that inspire me by their work. I thank them for their many comments and suggestions, that helped improve my dissertation.

I thank Viviane Makougni, our lab administrator, for her logistical and material support, but also for her care and her listening. Her enthusiasm and unwavering good mood made the lab a place where one feels good. I thank Elda Andre, Loic Sorel, the IT service, Stephane, Joel and all members of the front desk for contributing to make the MSE a pleasant place to work.

A thousand thanks to all the doctoral students that accompany me along the way. In particular those of the environment and macroeconomic team, my colleagues and friends: Ekrame, Djamel, Stefanija, Anastasia, Lorenzo, Bertrand, Claire, Diane, Alexandra, Francesco, Axelle, Guillaume, Baris, Thais, Kristel, Yvan, Medhi, Elliot and most particularly Antoine and Vincent with whom I shared my office during three years. Their every day presence, support, professional and personal exchanges made my stay at the MSE happy and humanly rich. Sharing this experience with many others was encouraging and made me feel more confidant. I do not know all the details of the rest of my journey, fortunately, but I am sure I will miss this place. I also wish to thank Kenet, my PhD colleague and friend from the PSE campus, with who I always enjoy to share and discus ideas.

Of course, I thank my friends for being there. Marion, Manu, Elise, Veronique, Adrien and Thais, I feel lucky to have you. Your support, your understanding and your constancy means much to me. I thank my beloved Caledonian team, Charlotte, Marion, Tara, Marie and Sophie who stand by me since many years now, and on who I can always count.
Last but not least, I thank my exceptional family for their unflagging support, for encouraging and always having confidence in my choices.

Introduction

The world of fisheries is complex, dynamic and contested. At the core of fishery management lie technical challenges but also fundamental socioeconomic issues such as valuation and ownership. So far, as for other natural resources, fisheries resources have been largely over exploited and alarms from the scientific community on rapidly declining stocks and species disappearance have been repeatedly published. In what follows, I first strive to explain the repeated failures to manage fisheries in a sustainable way. In this perspective, I overview existing management instruments and the weaknesses of fisheries governance.

Next, I raise the main challenges faced by the aquaculture industry to take over on capture fisheries, as a sustainable source of food security. This leads us to an introduction to the two first chapters of this dissertation, which focuses on the economic and ecological impacts of the expansion of aquaculture, and its implications for the capture fishery sector.

Finally, I expose the complexity of managing harvested ecosystems under uncertainty, in practice. Thus, I present the third chapter of this dissertation which puts forward a theoretical management framework grounded in viability theory to deals with risk, ecosystem dynamics and conflicting sustainability objectives. In particular, I examine the different analytical possibilities provided by this framework to handle uncertain dynamics.

The challenges to the sustainable management of fisheries

Foremost thing, what is a fishery? The United Nations Food and Agricultural Organization (FAO) defines a fishery as the "people involved, species or type of fish, area of water or

seabed, method of fishing, class of boats, purpose of the activities or a combination of the foregoing features". This definition allows for a large variety of bio-economic systems where a harvesting infrastructure targets specific fish species.

The FAO centralizes worldwide catch figures through collaboration with national governments and fish stock status estimations provided by the scientific community. So far, there is an ongoing evidence of overfishing, despite the numerous management measures implemented worldwide. The FAO's latest publication on the state of world fisheries (FAO [20]) reports that in 2009, about 57.4% of world marine fish stocks are estimated as fully exploited while 29.9% are overexploited. These figures are based on stocks of assessed species, which account for 20% of global catch. Several studies have attempted to appraise the status of the remaining stocks through innovative methodologies and tend to report even more alarming conclusions (Costello et al. [14]).

In all events, an increasing trend in the percentage of overexploited, depleted and recovering stocks is observed since the mid-1970s. In the same way, since the early 1990s, landings are marked by a small decline, but seems to have stabilized lately. In 2006 world marine landings were worth 80.2 million tons against 78.9 million tons in 2011. This late stabilisation can be explain by the fact that (1) the composition of landings has changed — fishermen turn to other resources as targeted species become scarce — (2) technological progress allows to reach out new fishing grounds (i.e. deeper and further from coasts).

Many factors can explain unsuccessful attempts to manage fisheries sustainably. A fishery is common-pool resource. That is, a non-excludable and rival good. As public economics made the evidence, common property resources produce negative externalities. Precisely, rivalry for the resource. Even though they might be concerned about the long term sustainability of the resource, fishermen are trapped in a race to catch their share of the fish stock before someone else does. This phenomenon is the so called tragedy of the commons described by Hardin [27]: "Each man is locked into a system that compels him to increase his herd without limit, in a world that is limited".

According to Clark [11] exhaustion of a species can also be explain by the fact that in some cases it is economically optimal. Considering fisheries resource as natural capital assets, when the rate of return of a common risk class of assets is greater than the expected return

of a fish stock, it is more profitable to exhaust the resource and invest revenues rather than to harvest at growth rate.

Nowadays, the world population growth, and most of all, the increase in standards of living in developing countries, result in a growing demand for animal protein (Hall et al. [25]). To keep pace with such demand, wild fisheries are subject to an increasing pressure.

For these reasons, alike for other renewable resources, fisheries resources require to be regulated to insure a sustainable exploitation. A sustainable use of natural resources is defined by as an exploitation that "meets the needs of the present without compromising the ability of future generations to meet their own needs" (WCED [58]).

Existing management options

At date, two types of instruments to manage fisheries stand out: regulatory or legal measures and incentive-based instruments. Regulatory measures, usually advocated by biologists, are technical measures, also called command-and-control tools. Among them are restrictions on fishing gear, areas, seasons or on time at sea, minimum legal fish size, harvesting quotas, limitations on fleet capacity or limited access through licensing. These types of instruments tend to confer less freedom to fishermen, as they do not leave them the possibility to select the less costly way of meeting policy goals. The most prevalent regulatory instrument is the quota or total allowable catch (TAC) tool which sets upper limits for the total amount of a fish species that can be landed from particular areas.

Incentive-based instruments tackle the ownership and valuation issues inherent to fisheries resources, either through right-based approaches or by influencing pricing mechanisms through taxes/subsidies. Right-based approaches fix the property right problem by allocating fishing quotas/efforts, or by assigning fishermen exclusive use of an area. When transferability of rights is admitted right-based systems rely on market mechanisms as well.

As an example, individual transferable quotas (ITQs) systems gives a fisher, vessels and/or producers an exclusive right to catch a specified portion of the Total Allowable Catch (TAC) set on an individual stock. The TAC is set by a regulatory agency and then divided into units that can be bought, sold or leased among participants in the fishery.

ITQs systems or variants, are increasingly employed fisheries management systems. In

2008, approximately 10% of the marine harvest was managed by ITQs (Chu [10]). That year, 148 major fisheries around the world were identified as having adopted some variant of this approach (Costello et al. [13]). ITQs systems are reckoned to end the "race for fish", improve fleet efficiency, reduce over-exploitation and increase ex-vessel prices (Chu [10], Grafton [24], McCay [37], Shotton [49]). Through individual guarantees on harvest quantities, ITQs allow fishermen to spread catches over the fishing seasons in a way that minimize operational costs. It also allows them to conduct safer fishing strategies (Hilborn et al. [29]). As a result, the relaxation of fishing competition improves the quality of catches and increases prices. The tradable feature of ITQs is intended to make quotas flow to the hands of those that most value them i.e. those which generate highest profits. Theoretically, a decrease in fleet capacity occurs as economic rationality leads least efficient vessels to sell out their quotas to the more efficient and withdraw from the fishery. Concerning conservation objectives, the exclusiveness nature of ITQs coupled to the fact that individual quotas constitute assets, which value lies on the ecological viability of the stock perceived by the market, are assumed to ensure the respect of TAC. What is more, the ITQs approach tends to establish the collection of fees for funding management costs and scientific research on users of the resource rather than on taxpayers (Grafton [24], Hilborn et al. [29]). Cost-recovery can improve economic efficiency and conservation of the resource, in the sense that those who pay or are payed for management services influence the choice of expenditures and performance of a fishery (Sutinen and Johnston [51]).

However, it is well-established that without an effective monitoring and enforcement system, ITQs do not prevent fishermen from bycatch or high grading (Beddington et al. [7], Ostrom [42]). This observation runs counter the argument that owning a quota gives incentives to enhance their asset value. Cheating behaviours aiming at maximizing present rents can be explained when the use of the fishing right presents limited duration or when they are leased. Right-based management strategies actually embody a trade-off between the more sustainable harvesting practice induced by permanent rights and greater social equity allowed by time limited fishing rights.

Indeed, ITQs management strategies are much criticized because they present political

challenges. The initial allocation mechanisms of access rights is a highly controversial stake [1]. By granting private property rights a wealth transfer takes place, which creates distributional conflicts in certain cases (Anderson [4], Grafton [24], Péreau et al. [44], Turris [55]). In addition, the tradable feature of ITQs can cause concentration of rights leading to monopoly configurations. Since running facilities for landings in several harbors is costly, the abandon of harbors by the industrials that concentrates fishing rights has been observed in Iceland (Report IGF-CGAAER, 2012). This raises the question of whether the total catch share per shareholder should be capped. What is more, the requirement of citizens to exercise sovereignty on national resources makes undesirable the holding of large shares of TAC by foreigners. Thus, some ITQs management schemes impose a limitation on foreign quota holding.

What is more, these policy instruments are not explicitly designed to manage the ecosystem effects of fishing (Gibbs [23]). Regulatory instruments often supplement incentive-based management systems as these approaches are not always fully effective in preventing ecological impacts of fishing.

Besides management instruments, an adaptive process in designing management schemes is of importance. Ostrom [42] insists on the fact that "successful institutional arrangements may emerge only by working with the users of a common-pool resource over time to develop a system that is well matched to the ecological system as well as to the practices, norms and long-term economic welfare of the participants". She highly fosters experience and adaptation in opposition to "one-size-fits all" solutions and recommends the implication of all stakeholder in the design process of management schemes, including industrials. Decentralized management is a mean of diversifying management rules and increasing compliance with regulations.

Regarding ITQ programmes, an efficient set of institutions rather developed in well functioning democracies is required. ITQs are not politically feasible or rational for several developing economies. Robinson et al. [45] claims "the focus ought to be trying to understand what improvements are possible, given the political forces at work in Africa". Beyond institutional quality, the author has identified two other prerequisite for ITQs systems to take

[1] Commonly, the initial allocation of TAC shares is carried in proportion to vessels historical fishing records.

place. ITQs systems have emerged in countries where fishing represents a larger proportion of GDP than it is typical. Cases of successful reform seem to be driven by the government rather than by the industry.

Self-management or self-governance refers to fisheries management situations where governance decisions are taken by fishery participants themselves (Townsend et al. [54]). According to Uchida et al. [56], three conditions are required to transform resources into club goods: "First, fishing ground boundaries need to be defined in accordance to the ecology of the targeted fish so that only members have exclusive use rights to the fish. Second, group membership needs to be well-defined and controlled. Finally, and most importantly [...], the groups need to be 'privileged'; that is, forming a group needs to bring higher present value of benefits to each member than nonmembers and the status quo". The authors put forwards that self-management alternatives are said to have advantages over command-and-control in parts of the world where the government's capacity is weak in enforcement and monitoring, or where it lacks institutional capacity to implement a market-based tool.

The role of governance

The governance of fisheries has important responsibilities in the failure to implement sustainable fisheries management schemes. Taking the case of the European Union (EU)'s Common Fisheries Policy (CFP), the commission provides subsidies to fuel purchase, modernize fleets or sustain fisheries that are not cost-efficient. Instead, these pubic funds could be used to mitigate overcapacity or invested in research to enhance the sustainability of fisheries. Such types of transfers to fisheries support environmentally harmful fishing methods such as deep-sea trawling. By scraping the seabed up to 2000 meters, trawl nets destroy valuable habitats and catch species displaying low growth and reproductive rates. Indeed, the more tough life conditions in deep waters produce more vulnerable ecosystems.

In the case of France, Bloom [2], a non-profit organization, denounced the consequent waste of public money that this activity represents. Less than ten vessels are involved in deep-sea trawling in France, and turn out unprofitable despite the subsidies they perceive. In 2013, the European Commission proposed to vote a ban on deep-sea trawling. However,

[2]Bloom: http://www.bloomassociation.org/

the amendment was rejected. France was the most hostile country to the prohibition of bottom trawling. The compromise adopted aims at freezing the currently trawled areas (not fishing beyond), and better defining acceptable trawling zones within this space. Following the strong public mobilisation generated by Bloom activism, in particular on social networks, several restaurants in Paris engaged in withdrawing deep water species from their menus (hoki, cutlass fish and blue ling). This initiative illustrates the strong role that consumers can play in modifying environmentally damaging production habits.

The legal measures used by the CFP to regulate harvesting within its jurisdiction include: TACs, gear regulations, closed seasons, closed areas, minimum allowable sizes for individual species and structural measures that controls fleet capacity.

The International Council for the Exploration of the Seas (ICES) produces scientific advice for the European Commission. The commission then forms a proposal in light of this evidence and discussions with various relevant departments and committees, including the scientists. Proposals are then sent to the Council of Ministers, made up of national ministers from member states, which has the final authority to negotiate and formulate fishery regulations.

In facts, it is well documented that TACs are frequently set higher than the levels recommended by scientists (Karagiannakos [32]). According to OCEAN2012 [3], in the last years, the catch limits agreed were on average 46% higher than scientific advice. In addition legal measures are generally much less severe than advised by scientists. Daw and Gray [17] explain the failure to translate scientific discovery into practical policies comes, among other things, from deficiencies of the political system. Electoral politics of fishery ministries lack of incentive to comply with scientific catch recommendations as measures involved are often likely to cause hardship and unemployment. The EU's CFP has failed to reduce overcapacity, which create political pressure to set quotas higher enough to keep all vessels working. Furthermore, pay-offs from conservation measures are uncertain and may not occur before the end of the ministry's term. Such unpopular measures are generally left aside when planning to run the next elections. Besides, Hoffmann and Quaas [30] demonstrate theoretically that inefficiently high TAC levels are a consequence of the uncertainty about future majorities and

[3]OCEAN2012: http://ocean2012.eu/pages/3-the-issue

decision making in the council. According the authors, more sustainable fishery management requires binding long term commitments instead of annual votes.

The failure to convert scientific recommendations into practice also originates in the poor enforcement of regulations. Discarding at sea, illegal landings and the mis-allocation of catches to fishing grounds contribute to outpace exploitation rates recommended by scientists. A new CFP is effective since January 2014. It focuses, *inter alia*, on banning discards. All fish caught and not covered by individual quotas should be landed and counted against the species' quotas. An overall by-catch quota could be established on a fishery base, as a reserve. To ensure full reporting of fishing activities, vessels need to be equipped with electronic tracking technologies. Funding for such monitoring devices is scheduled.

Illegal, unreported and unregulated fishing (IUU) seriously undermines the attempts to exploit fisheries in a sustainable way. Illegal fishing refers to activities conducted in the exclusive economic zone (EEZ) of a coastal State by national and foreign vessels, without permission, or in violation of its laws and regulations. This is often the case where monitoring capacity lacks or is weak. As in international waters, where vessels flying the flag of member states to a regional agreement, practice illegal fishing in violation of international law or of management measures set by regional fisheries management organizations (RFMOs). Unreported fishing refers to fishing activities which have not been reported, or have been misreported, to the relevant national authority or RFMO. Unregulated fishing refers to fishing activities conducted in areas where there is no applicable conservation or management measures, or activities conducted by vessels without nationality, or by those flying the flag of a State that is not party to the relevant RFMO, in a manner that is not consistent with or contravenes the conservation and management measures of that organization.

Beyond the fact that IUU fishing dilutes the effect of conservation management and policy measures, it undermines labour standards, harms markets for legally harvested fish, encourages corruption and reduces prospects for food security, economic growth and stability, especially in developing coastal nations (UNODC [57]). In 2009, illegal fishing was estimated to cause losses to the legal economy in the range of 10 to 23 billion USD annually (Agnew et al. [1]).

Species of fish that are in short of supply and are of high value particularly constitute

strong economic incentives for IUU fishing. Overall, the profitability of practices such as fiscal dumping also gives incentive to bypass law. There are several reasons to the lack of monitoring and control of fishing activities. It is costly and in several countries, specially the developing ones, capacity lacks or it is not a priority. Too little penalties also fuel IUU fishing as revenues generated by fishermen exceed costs while the contrary holds for regulatory authorities. Penalties paid within the European community averaged between 1.0 and 2.5 percent of the value of IUU landings (OECD [41]). Furthermore, the heterogeneity in governance systems, capacity and political will makes it difficult for States to collaborate effectively in combating IUU fishing. It is hard to know exactly how much IUU fishing is taking place. We do know that, for some important fisheries, IUU fishing accounts for a large percentage of total catches.

Expensive, easily transported seafood products have become one of the currencies traded in broader illegal commodity transactions involving drugs, arms and human trafficking (HTSF [31]). In this context, INTERPOL is taking a transnational approach to share information and monitoring capacity. In a first stage, by developing a data base interface. Exchanging information among nations is one of the most challenging issues to combat IUU fishing activities. The project also aims at approaching IUU fishing activity from a crime perspective, as it results in tax evasion. Joint intervention in other related areas such as money laundering, drug trafficking or people smuggling could contribute to provide more interests in cooperating (Leroy [36]).

Bluefin tuna is the example of an endangered fish species which illegal catches feed forecasts of commercial extinction. Bluefin tuna counts among world's most lucrative commercial fisheries. Each year, the International Commission for the Conservation of Atlantic Tunas (ICCAT) sets an overall fishing quota for the eastern Atlantic and Mediterranean populations of bluefin tuna. In 2009, the adjusted quota set by ICCAT was 21,780 tons[4], while its own scientists advised no more than 10,000 tons. Official trade records indicate the total amount of bluefin tuna traded in 2009 and 2010 was 70,646 tons. This is a combined overage of 35,306 tons (Bregazzi [8]). When including black market, conservationists suspect the

[4]The 2010 quota was worth 13,306 tons. It was the first time that the quota was within the range of scientific advice.

actual catch is 60,000 tons per year. The ICCAT affirmed in 2009 that Atlantic bluefin tuna stocks have declined dramatically over the last 40 years, by 72% in the Eastern Atlantic, and by 82% in the Western Atlantic. If current fishing levels continue, ICCAT scientists have determined that the population has less than a 24% chance of rebuilding by the 2022 target set by ICCAT members.

The case of bluefine tuna illustrates our current inability to coordinate and adopt appropriate measures in a situation where a resource is endangered. The bluefine tuna belongs to the class of large predator species. Its status of threatened species comes in line with the gradual transformation of food webs that fishing is inducing worldwide. As exposed by Pauly et al. [43], landings are transiting from long-lived, high trophic level, piscivorous bottom fish towards more short-lived, low trophic level invertebrates and planktivorous pelagic fish.

Aquaculture: a source of food security?

The issues faced by the aquaculture industry

Since the 1970s, global demand for fish has kept increasing, particularly in the developing countries where population and income growth constitute the main drivers of the recent rise in world fish consumption (Delgado et al. [19]). While output from capture fisheries stagnates, total production of fish continues to rise due to aquaculture. Aquaculture is the farming of freshwater and saltwater organisms such as fish, mollusks, crustaceans and aquatic plants. In 2010, world aquaculture reached 60 million tons (excluding aquatic plants and non-food products) contributing up to 47% of world food fish production against 9% in 1980. An important share of world aquaculture production is located in Asia, which output reached 89% by volume in 2010 (FAO [20]).

According to the FAO forecasts' "to maintain the current level of per-capita consumption, by 2030 the world will require at least another 23 million tonnes of aquatic animal food - which aquaculture will have to provide". World demand for fish is outpacing the ability of the world fisheries to supply it. The aquaculture industry is expected to eclipse the production of wild fish, providing food security to many developing countries, for which fisheries resources are a vital source of food.

However there are concerns that the expansion and intensification of this sector, coupled with its ecological and social impacts, jeopardize the sustainability of aquaculture. The practice of intensive farming of finfish and shellfish is characterized by inputs of high quality resources and energy. Even though the aquaculture industry still has an important growth potential, it increasingly faces problems of scarcity of space and feed.

A significant part of the aquaculture expansion is expected to occur in coastal areas, where it directly affects resource systems already experiencing large pressure from human activities. There is a risk that the anticipated benefits from aquaculture come at the expense of increased pressure on coastal ecosystem services, thus compromising the food security of coastal communities and reducing biodiversity.

Moreover, open-water aquaculture is a polluting industry, that releases effluents into the wild. Farmed fish feed is loaded in nitrogen. The pollution may come from a variety of sources, including feeding and medical treatment. Nitrogen contributes to daily amounts of protein, helping to promote optimal fish growth and health. However, fish dejections loaded in nitrogen are released in rivers and coastal areas, polluting natural environments. Besides, cultures are subject to parasites, in particular to sea lice which is known for being much harmful to farmed fish. Green Warriors[5], an environmental organisation, reported that Norwegian salmonids farms make use of diflubenzuron, a highly toxic chemical, to eliminate sea lice from cultures, though this product is banned from use in the European market. The use of such pesticide is detrimental to ecosystems.

Regarding fish feed, the aquaculture industry essentially relies on fish meal and fish oil. Fish meals corresponds to fish flour made from low value species while fish oil is usually derived from the tissues of these same organisms. The dependency of the aquaculture industry on the availability of fish meal and fish oil raises concerns among environmental groups about potentially negative effects on wild fish stocks (Natale et al. [38], Naylor et al. [39]). As the supply of these inputs is limited, this dependency also has implications for the future growth of the aquaculture industry (Shamshak and Anderson [48]).

In 2008 fed-aquaculture corresponds to 60% of world farmed aquatic animal production. In fact, because animal protein support faster growth in farmed species, the industry tends to

[5]Green Warriors: http://www.nmf.no

administer compound feeds[6] to herbivorous species, besides the carnivorous and omnivorous ones[7]. Among the different animal husbandry industries, aquaculture is the largest user of fish meal (63%) and fish oil (81%).

The expansion of the aquaculture sector urges to find substitutes to these inputs so as to sustain its growth rate. Especially, as tighter quotas and additional controls on unregulated fishing are expected to decrease the supply of compound feeds. So far, the aquaculture sector has succeeded in maintaining a high growth rate (6.3% in recent years), despite the stagnant supply of compound feeds, due to a more cost-effective inclusion of fish meal and fish oil in fish diets, and to the decrease in demand for other use. Yet, it is reckon that for certain species, the intake of specific fatty acids is required to conserve the dietary properties of the fish. The well known FIFO ratio (fish in-fish out), gives the number of tons of wild fish necessary to produce one ton of farmed fish (including fish oil and fish meal requirements). The FIFO ratio varies quite a lot between surveys. Tacon and Metian (2008) gives an overall FIFO ratio of 0.7 in 2006, which means on average 0.7 kg of wild fish were required to produce 1 kg of flesh. This ratio includes all bred species: crustaceans, carnivorous, omnivorous and herbivorous. At the carnivorous species-group level, the study reports a salmon FIFO ratio of 4.9. Naylor et al. [40] supports the figures conveyed by Tacon and Metian [53]. On the other hand, IFFO[8] finds an overall ratio of 0.66 using the same data base, which is fairly close. However, the FIFO salmon ratio is quite lower, with a 2.2 value. Both studies attest of substantial decrease in FIFO ratio since the 90's. Nonetheless, IFFO ratios reflect greater achievements in terms of feed efficiencies. Plant nutrients can be source of proteins, but not that necessary to conserve flesh quality. To fill the gap, research is focusing on the culture of microbial ingredients to product proteins. At date, such feed alternative has not proved sufficiently cost-effective.

The small oily fish used to produce aquaculture feeds generally belong to low trophic levels

[6]Compound feeds used by the aquaculture sector are pellets made of fish meal and/or fish oil, and fertilizers.

[7]FAO [20] reports that just 8 species or species groups account for 62.2% of the total compound feed used: grass carp, common carp, Nile tilapia, Indian major carps, whiteleg shrimp, crucian carp, Atlantic salmon, and pangasiid catfishes.

[8]International Fishmeal and Fish Oil Organisation: http://www.iffo.net

(essentially sardine and anchovy). The fisheries serving this purpose are called *reduction* fisheries. According to Smith et al. [50], fishing low trophic level species has widespread effects on marine ecosystems. Despite the fact that the removal of these species may impair food webs, thereby, reducing the abundance of predators higher in the food chain, they also represent a source of animal protein for direct human consumption, especially in developing countries (Tacon and Marc [52]). Thus, although aquaculture contributes to food fish supply and to people's livelihood in certain areas, it may affect food security through these two channels, in others.

An analysis of the impacts of aquaculture production on food fish supply and demand

The two first chapters of this dissertation address the interactions between the aquaculture and the capture fishery sectors, and analyze the implications of the expansion of this food fish production process in several respects. Especially, on supply and demand of fish food, on fish price, wild stocks and consumer welfare. The aim is to give insights on the long term perspectives of both food fish production sectors, and on the potential evolutions in the fish market structure that are likely to occur.

The aquaculture and capture fishery sectors interact in two respects: at the biological and market level. Biological interactions stem from the dependence of aquaculture on wild fish stocks for the feeding of breed species. The withdraw of small oily fish — prey species — can affect the abundance of predator species that are targeted by the capture fishery sector for direct human consumption. This is a possibility but not necessarily the case that the ecosystem effect of reduction fisheries affect other fisheries. On the other hand, market interactions refer to the price dynamics induced by the presence of farmed and wild fish products on stalls. The competition between farmed and wild fish depends on consumer preferences for both types of products.

Chapter 1

The first chapter of this dissertation is co-authored with Katheline Schubert and untitled *Is aquaculture really an option?* It introduces a theoretical framework to investigate the impact of the introduction of aquaculture on fish consumption, welfare and on the wild fish stocks, taking into account its dependence on wild feed fish and consumer preferences. The framework retained to investigate these issues is that of a competitive aquaculture sector and open access fisheries. The model consists of the demand side and three sectors: an edible fish fishery, a reduction fishery and an aquaculture sector producing farmed fish.

A biological interaction takes place between the feed fish, harvested to grow farmed fish, and the wild edible fish, which feeds on this same reduction stock. This ecosystem effect of fishing is described by a Lotka-Volterra model, while the harvesting technology is modeled by the classical Schaefer [47] production function. Regarding the aquaculture production technology, the conversion rate of feed fish is given. Feed fish is the only variable input considered, with a decreasing marginal productivity. Consumers may choose to consume wild or farmed fish, which are strong substitutes. The model is solved in a partial equilibrium framework i.e. income is exogenous.

A few papers have investigated the market interactions between aquaculture and capture fisheries. Anderson [2] supposes that the wild and the farmed species are the same, and consequently have the same market price. He shows that in the case of a fishery in open access exploited beyond the maximum sustainable yield, the entry of competitive aquaculturists increases total fish supply, thereby, reducing consumer prices and increasing natural fish stock. Indeed, profit loss mechanically reduces fishing effort. Ye and Beddington [59] assume both goods are imperfect substitutes with positive cross-price elasticities. Similarly, the authors find positive consumer benefits of aquaculture via increased fish supply and reduced prices. Yet, the imperfect substitutability between farmed and wild products limits the extent to which aquaculture production impacts the capture fishery activity.

Our study comes closer to Hannesson [26], which calls for a simple modelling of both, market and biological interactions. Two consumption goods are available: a farmed and wild edible fish, which are perfect substitutes. As in our setting, the biological interaction is embodied in a prey-predator dynamics between the harvested wild edible fish stock and

the feed fish stock, harvested by the aquaculture sector to grow farmed fish. Unlike us, the model does not account for the demand side. Relying on numerical simulations, the author finds that in open access, the total food fish production is slightly higher than without aquaculture. However, this increase in supply is carried at the expense of the wild edible fish stock, which is severely diminished compared to the situation without aquaculture. The author concludes feed fishing should walk by global optimization, which takes into account the biological interaction between both production process. This outcome is driven by key assumptions regarding the relative cost-effectiveness of both fish production sectors, with a special emphasize on the efficiency in converting feed fish into edible flesh in the wild compared to the performance of the aquaculture sector.

Our long run results regarding the impact of aquaculture entry are conditional on the degree of dependence of the wild edible species on the feed fish stock and on income level. When this interaction is moderate and under a maximum income level, the model admits a stable interior steady state. The introduction of aquaculture is beneficial to consumers and its consequences are conform to what is found in the literature, in absence of biological interactions (Anderson [2], Ye and Beddington [59]). By increasing global fish supply, aquaculture decreases the price of the wild product, thus, fishing effort decreases allowing the edible stock to recover despite the fact that aquaculture exploits the prey species as a production input. In fact, there even exists a range of income levels for which the introduction of aquaculture prevents the wild edible species of collapsing. On the other hand, the feed fish stock is always lower as one can expect. Otherwise, it is not obvious analytically whether total wild fish consumption increases in all events, yet, this low interactions scenario benefits to consumers whose utility is always increased.

In the case where biological interactions are high, aquaculture leads to a decline in the feed fish stock and the wild edible fish stock, a decrease in wild edible fish supply and an increase in its price. Indeed, when assuming a strong dependence of the predator species on the feed fish population, aquaculture threatens the edible species by altering its food web. For stronger biological interactions, the harvesting of the predator species by the edible fishery reduces the competition for the feed fish stock, allowing the aquaculture to pursue its activity at higher levels of expenditure on food fish. Yet, for high levels of income, aquaculture actually

provokes the collapse of the wild edible fishery though it would have remained alone. The competition between the predator species and the aquaculture sector for feed fish becomes too high as food fish demand rises. The net effect of farming on total fish consumption and welfare is ambiguous. We show through numerical simulations that the introduction of aquaculture may decrease utility. Such result can be explained by the fact that the fisheries are supposed to be in open access, meaning that the exploitation of resources is economically inefficient. In this situation, the introduction of aquaculture adds an extra inefficiency which may lead to a decreased utility, in spite of the fact that more consumption options are offered to consumers.

Besides, we further investigate the influence of consumer preferences on these outcomes by introducing additional assumption on their consumption behaviour. We endogeneize consumer preferences, assuming that they are carnivorous species-biased. We chose to focus on a demand arising from rather wealthiest populations. The wild fish is a high value carnivorous species. The taste of consumers for the farmed fish depends on its diet: the more carnivorous the species is, the more consumers like it. On the other hand, the more carnivorous the farmed species is, the more inefficient and costly is its breeding, in the sense that the production of 1 kg of flesh requires more than 1 kg of wild feed fish. It turns out that as the efficiency of the aquaculture sector improves, consumers tend to dislike the farmed fish. The improvement in the aquaculture conversion ratio of feed into flesh can be interpreted as a shift towards the breeding of less carnivorous species. At given species, it can also be viewed as efficiency improvements of feeding formulas, resulting in a modification of flesh properties that consumers penalize. In any event, as both product types become less substitutable, consumer transfer part of their demand on the wild edible species. Thus, the beneficial impact of aquaculture on the wild edible stock and consumer welfare is mitigated.

According to the OECD a potential answer to the limited supply of feed would be to split the salmon market for instance into a cheap market fed with reduced fish oil and an expensive one fed with real fish oil. Differentiating supply through two lines of quality could better match consumers willingness to pay for food fish and better ensure the prosperity of the industry. Another option to enhance the sustainability of aquaculture would be to direct consumer preferences towards less carnivorous species by means of marketing strategies or

educative campaigns.

Chapter 2

The second chapter of this dissertation is co-authored with Basak Bayramoglu and untitled *Competition between farmed and wild fish: the French sea bass and sea bream markets*. It consists in an empirical analysis of the behavior of consumers towards farmed and wild fish products. The objective is to sort out to what extent both product types are treated as substitutes in facts. Evidence on the intensity of market competition between the aquaculture and capture fishery production processes gives insights on the extent to which the supply of farmed fish is likely to affect food fish price dynamics. This outcome depends on how consumers perceive wild versus farmed products, but also on the relative cost, competitiveness and production quantities of each sector.

Substitutability between wild and farmed fish is of importance as it is likely to condition the evolution and guidance of each sector. If the farmed fish product has a lower market value relatively to its wild counterpart, the fish market may prove segmented with respect to this product attribute. As emphasized in chapter 1 and in the theoretical literature (Ye and Beddington [59]), when both types of goods are imperfect substitutes, the extent the increase in food fish supply diminishes wild fish prices and alleviate pressure on wild edible stocks is lower. The fishery sector then benefits from a price premium and occupy the higher segment of the fish market. Yet, without proper management of the resource, the wild fish supply may become scarce, and the fishery's rent dissipated. On the other hand, if both product types are perfect substitutes, the market share and economic viability of each sector will depend on their relative cost-effectiveness in producing food fish.

Anderson [3] argues that the prevailing difference between aquaculture and traditional fisheries lies in the degree of control on production parameters. The aquaculture production technique decides of fish feed, seeds quality, light, temperature, water quality and also the production timing and space. This level of control over the production process is perceived as a strength, since it allows to determine the color, size, texture, fat content or other parameters of the delivered product, in accordance with consumers preferences (Asche et al. [6]). What is more, the reliability and consistency of aquaculture supply all year long has an

addictive effect on consumers (Knapp et al. [33]). On the other hand, as emphasized in §, the increasing awareness of consumers that aquaculture is a polluting industry tends to be reflected in prices. As an example, France is the first importer of Norwegian farmed salmon. Yet, French consumers recently turned away from their favorite fish since the release of several scientific studies on the presence of pollutants in the fish flesh. Sales fell by more than 25% between November and April 2013 (Lemaire [35]). In Norway, near Bergen, the subject is also at stake. In addition, consumers tend to be cautious about farmed fish feeding and the potential consequences of changes in species' natural diet. Thereby, consumer's behavior towards farmed fish is likely to direct research efforts and management choices adopted by the aquaculture sector to foster the industry's market insertion.

In this study, we test for market interactions between farmed and wild products in France, focusing on the case of sea bass and sea bream species, respectively. Our empirical approach relies on the Law of One price (LOP) and on the concept of cointegration which consists in testing for price parity between products, wild versus farmed fish prices in our case. In a competitive sector, the duration of a constant relative price relation between goods shows that the matching of supply and demand has reached a stable long-term balance. Since the structure of production costs between the aquaculture and capture fishery industry are quite different, we posit that the existence of a long run relationship between both fish products stems from the arbitrage behavior of consumers between goods, rather than from supply factors.

The cointegration approach to market delineation has been extensively used in the literature to analyze market integration between different fish species, geographic area, or to test whether consumer distinguish between fish products origin, form, or production process. Many studies concern salmon, in particular, Asche et al. [5] undertake cointegration test for market delineation and also estimate a dynamic system of demand equations on a same data set concerning fresh, frozen salmon and crustacean in the EU market. Authors find that both approaches provide compatible and complementary results.

This cointegration approach is convenient as only price series are required. Our data set consists of domestic (monthly) price series, provided by Kantar WorldPanel[9], for the sea bass

[9]Kantar WorldPanel: www.kantarworldpanel.com.

and sea bream species, purchased by households from 2007 to 2012 in France.

The importance of the sea bass and sea bream markets at European level, makes these species study subjects of interest. In Europe, sea bass and sea bream taken together represent the largest production of farmed species, after Atlantic salmon (FEAP [21]). Aquaculture is in fact the main production method of these two species. France is the third intra-EU importer of farmed fish, while the extent of domestic catches enables national consumption of wild and farmed fish to be sustained for both species. Indeed, France undertakes 60% of world sea bass capture and 32% of the world sea bream captures As detailed further on, in recent years, about 50% of French consumption of sea bass was farmed while that of sea bream was around 70%.

Little empirical analysis on sea bass and sea bream has been conducted in the literature (Asche et al. [6]), despite of the considerable trade flow they generate in the EU.

What emerges from our analysis is that the degree of market interaction between wild and farmed fish differs from one species to another. While wild and farmed sea bream markets are found to be partially substitutes, wild and farmed sea bass markets appear disconnected. For both species, wild fish is always more expensive than farmed fish. Furthermore, based on the observation that wild sea bass is more expensive than wild sea bream, and displays a greater price differential with its farmed counterpart than does sea bream, we conclude that consumers may be sensitive to the seafood production process when it comes to higher-value species. In the case of sea bass, this attribute overrides the species type attribute.

On the other hand, below a certain price level, when testing for price parity between fish products differing by several attributes (species, form and production process), price closeness seems to be the main criteria intervening in consumers' arbitrage. More economically constrained consumers, or those less willing to pay for fresh fish appear less sensitive to product attributes. These outcomes suggest farmed fish may better succeed in lower-value segments of the fish market.

The price competitiveness of farmed sea bass and sea bream supply in the French fish market has much to do with the fact that supply is essentially imported from countries with lower labor costs. Because the French labor force displays other advantages — high technical skills — the farming of sea bass and sea bream has developed into less intensive and more

costly farming techniques. This quality industry image cancels the price penalty incurred by aquaculture and should support an insertion into top-end fish market segments.

The aquaculture sector faces a quality/quantity trade-off in its development strategy which must be sorted out according to local comparative advantages (space availability, labor force characteristics etc.). Foremost, it is important to identify whether and where a potential demand for a supply strategy exists, and if so, of what scale. Thus, the importance of the substitutability issue between wild and farmed fish, which is likely to differ according to species types, regional consumption habits or wealth.

The complexity of fisheries management in practice

Defining ecosystem sustainable yields under uncertainty

Practically, the sustainable management of fisheries is a complex task. First of all, the purpose of sustainability implies dealing with conflicting objectives. Management decisions must balance the risk of resource collapse due to excessive exploitation versus the risk of forgone economic benefits if the harvests are lower than necessary.

In the same way, the level of participation in a fishing sector must be balanced against the economic profitability of the sector. Else, open access to a resource mechanically leads to the bionomic equilibrium (Clark [11]), where fishing has reduced the fish stock to a level such that catch rates barely cover fishing costs.

According to Cochrane [12], sustainability objectives encompass economic profitability, ecological viability and overall stability of the bioeconomic system. When one of the ecological, economic or social objectives is not met, fisheries face an unsustainable situation.

Secondly, fisheries management schemes must take into account the complexity of ecosystem dynamics. Roughly, a species' dynamics can be described by its mortality, recruitment and catch rate at each point in time. However, targeted species belong to integrated biological systems, with numerous linkages between inherent species. The fishing of a particular species and the subsequent bycatch affects the community structure by diminishing its abundance relatively to interacting species. In fisheries science, bycatch refers to species caught by a fishery intended to target another species, as well as reproductively-immature juveniles

of the target species. Applying the following definition of bycatch — catch that is either unused or unmanaged — to global marine fisheries data gives a bycatch rate representing 40.4% of global marine catches (Davies et al. [16]).

Hence, to be sustainable, management strategies must account for biological and technical interactions (i.e. bycath) between species. There is a growing demand for moving from single species management schemes to an ecosystem approach of fisheries management (Garcia et al. [22]). It is reckon that single species approach can be misleading for the determination of long term yields and ecosystems' health. The World Summit on Sustainable Development (Johannesburg, 2002) encouraged the application of an ecosystem approach by 2010.

However, the ecosystem approach of fisheries faces many issues, ranging from the high cost of the science required (developing data collection, analytical tools, and models) to the practical difficulties of changing the governance system and processes (Cury et al. [15], Sainsbury et al. [46]). Indeed, the dynamics of ecosystems are complex and poorly understood. Besides, fisheries modeling requires estimations of stock status and of total withdrawal from stocks. Such information remains imprecise and error prone. Quantifying resources is an issue in fishery management, rooted in the fact that, in contrast to other living species or natural resources, fish stocks and fish are not easily observable.

Thirdly, uncertainty inherent to fisheries is recognized to play an important role in the failure of management regimes. As mentioned, it can concern stock status, the structure or dynamics of ecosystems. Otherwise, uncertain climatic hazards or technical progress are likely to affect fisheries productivity. Some claim that fishing decreases the resilience of fish populations, rendering them more vulnerable to environmental change (Lauck et al. [34]), and that not accounting for uncertainty can lead to excessive harvest of a resource (Hilborn and Walters [28]).

The standard criteria to address dynamic issues under uncertainty in economics is the discounted expected utility. In fishery economics, optimality is obtained by maximizing the expected value of the sum of discounted revenue from harvesting. Meanwhile, the discounted expected utility framework is controversial. Its application to intergenerational environmental issues raises equity concerns with regards to the standard of living of future generations. Furthermore, this approach is rarely used in practice of environmental management.

In practice, fisheries management strategies are evaluated in a multicriteria framework with no clear axiomatic foundations. The key scientific concept used to provide regulatory recommendations in practical decision making literature is a "management procedure" (MP). This concept was developed by the International Whaling Commission in the late 1980's. MPs are defined by Butterworth and Punt [9] as a set of rules which describes recommendations for management actions.

The Management strategy evaluation (MSE) is a method elaborated for comparing management procedures, which takes into account uncertainty, conflicting objectives and time horizon. As detailed in Sainsbury et al. [46], the MSE approach consists in defining an operational set of management objectives, and evaluating the performance of various alternative MPs with respect to the specified objectives, taking into account uncertainty in the modelling processes. The method consists in testing a particular MP in a great number of simulations over a given time period, each simulation representing a plausible "state of nature" (scenario), and in computing statistics over the simulation results to summarize the performance of the particular MP.

The weakness of this method is that it is not able to prescribe an optimal bioeconomic strategy. That is, once the performance statistics are computed for each MP, comparison of the likely distance of corresponding management actions to the conflicting management objectives requires making a trade-off choice. This approach thus provides a clear description of the consequences of management procedures but with no common currency between the different objectives and risks to sum up the results and rank the alternative management procedures. Hence, decision bodies in charge of examining management actions are left with several indicators providing information on risk and benefits of alternative options, but lack of a measure embracing a plurality of criteria.

The viability approach is an alternative theoretical management framework to address dynamic control problems under constraints, and that can deal with uncertainty. In its stochastic version, it provides a common value to assess and rank alternative management options. In this respect, this approach can be viewed as a scientific tool to support multi-criteria decision making. The following section introduces the analytical possibilities provided by this method, and its relevance for addressing fisheries management issues under uncertainty.

The viability approach to risk management in harvested ecosystems

The stochastic viability theory (De Lara and Doyen [18]) aims at analysing if there exists states for which dynamics of a system, under uncertainty, and constraints describing given objectives, are compatible. It allows identifying control strategies such that the set of constraints is respected with maximal probability at each date over a given time span. This probability is called maximal viability probability and constitutes our common value to assess management actions. The higher it is, the lower is the risk of violating constraints.

Constraints often come in the form of binding thresholds, supposed to maintain the existence and good health of the system. They are generally set constant over time, implying that all generations are subject to the same constraints. The (stochastic) viability approach is a positive rather than normative approach. It does not fall under the rational of welfare maximization. The approach concentrates on describing the window of opportunities with respect to policy goals. That is, all the evolutions of a dynamical system consistent with specified objectives. In fact, in the stochastic approach, optimization focuses on the likeliness of satisfying all constraints at all date.

The robust viability approach (De Lara and Doyen [18]) is an extreme case of the stochastic one. It seeks all the evolutions of a dynamical system that satisfy, at all times, given objectives, despite of uncertainty. This means that constraints must be satisfied at each date with probability of one. This handling of uncertainty contrasts with the stochastic approach, which admits low levels of constraints violation.

Adapting the viability framework to bioeconomic systems allows to seek consistency between fisheries dynamics under uncertainty, and conflicting economic and conservation objectives. As in the economics or marine science literature, state variables generally correspond to biomasses and controls to harvesting efforts, while uncertainty can take various forms depending on its source. It is worthwhile noting that while in the stochastic version of the viability approach, the set of scenarios is equipped with a probability distribution, the robust framework does not require any probabilistic assumptions, as failure or success with respect to scenarios are the only options.

Besides, given that wildlife populations often display wide fluctuations in an unpredictable way, fisheries management goals and schemes should be updated regularly, in accordance to

the new data on stock assessments. Hence, in the frame of short time horizons, considering constant economic or ecological objectives appears reasonable.

The third chapter of this dissertation is co-authored with Michel De Lara and untitled *Managing harvested ecosystems under uncertainty: the viability approach*. This chapter explores how the viability analysis deals with uncertainty in its stochastic and robust approach, and the sensitivity of fisheries management implications to the way uncertainty is accounted for. We carry this analysis in the situation where little information on biological dynamics is available, preventing to identify a probability density of uncertainty scenarios. Indeed, when dealing with uncertainty here, we do not address how to specify the probability distribution of uncertainty scenarios, neither which regulatory instrument is most suited, given the nature of uncertainty. Our question of interest is what is the risk minimizing management approach that provides the best compromise among conflicting objectives.

We proceed to a numerical application of the stochastic and robust viability analysis to the anchovy-hake couple in the Peruvian upwelling ecosystem. In fact, the peruvian anchovy stock is the world's largest fishery resource, with annual landings of 5 to 10 million tons. It generates up to one-third of the world's fish meal supply. On the other hand, hake is targeted for direct human consumption. Both species are related by a prey-predator relationship. Thus, this application enables to put forward the management challenges involved by the biological interactions between the aquaculture and capture fishery sectors, in practice.

The anchovy-hake couple in the Peruvian upwelling ecosystem is described by a discrete-time two-species dynamical model, where states are biomasses, and where two controls act as harvesting efforts of each species. Uncertainties take the form of environmental disturbances affecting each species growth factors, and are assumed to take their values in a given set. Time sequences of uncertainties are called scenarios. These disturbances may account for environmental randomness or a misspecification of population dynamics. Constraints are imposed for each species: a minimum safe biomass level, usually identified by biologists, and a minimum required harvesting level assumed to ensure economic needs.

In a first stage we apply the stochastic viability approach to exhibit the trade-offs between various levels of production requirements on each species, respectively. Starting from an initial biomass couple, we provide the maximum probability of satisfying various levels

of minimal catch thresholds at each date over a time span, given fixed minimal biomass thresholds.

The data that is available on the Peruvian hake-anchovy upwelling ecosystem is provided by *El Instituto del Mar del Perú* (IMARPE). Biomass time series are limited to ten points per species. Thereby, the two probability distributions of uncertainties that we assume in our application are arbitrary. In addition, we also consider different uncertainty sets. The idea is to appraise the sensitivity of our results to these hypotheses.

We find that, for a same uncertainty set, the weight affected to uncertainties considerably changes the set of policy objectives that can be envisaged with a high probability. This outcome demonstrates the fickleness of management recommendations to assumptions on the distribution of uncertainty scenarios. Due to the poor information available on the ecosystem dynamics, the soundness of the hypotheses on the distribution of uncertainties that can be derived is poor. In this case, we consider the uniform distribution as more cautious, because it puts all empirical uncertainties on the same footing.

On the other hand, we observe an important gap between objectives that can be envisaged when admitting a low level of risk and when strictly avoiding risk taking. Low risk taking reveals an important and sustainable harvesting potential of the Peruvian anchovy-hake fishery, relatively to what is targeted in practice. In this case study, suppressing risk unwarrantably reduce the leeway of the decision maker with respect to pursued policy goals.

What is more, we stress the operational scope of the stochastic viability framework. The maximum production levels that can be guaranteed with a high probability over a time span can provide support in defining sustainable harvesting quota levels and strategies for each species. Such regulatory measures would account for ecosystem dynamics, uncertainty and ensure the ecological viability of the system.

In a second stage, we conduct a robust viability analysis of this same ecosystem model under risk and constraints, focusing on the concept of robust viability kernel. That is, the set of initial biomasses for which there exists at least one effort strategy such that all constraints are satisfied at all times, whatever the uncertainty scenario. The robust viable kernel is an insightful mean to display the impact of uncertainty on the possibility of a sustainable management. Wherever a fishery stands, the set of robust states enables to predict whether

economic and ecological objectives can be guaranteed over a time span, despite of uncertainty.

Reducing uncertainties to zero amounts to dressing the problem as deterministic. We proceed to a comparison of deterministic and robust viable kernels to shade light on the distance between the outcomes of these two extreme approaches: ignoring uncertainty vs. hedge against any risk. We do not advocate the robust viability approach as a suitable decision tool for fishery management, since it involves economic costs for society, that are not justified when no catastrophic or irreversible events are expected, or when their likeliness is low. Our aim is to emphasize the impact of accounting for uncertainty on management possibilities that arise from a same methodology.

We find that accounting for uncertainty sensibly shrinks the deterministic viability kernel (without uncertainties). In addition, we have been able to shed light on the uncertainties that really matter for a precautionary approach, and that they correspond to extreme cases. Assessing which uncertainties truly impact the robust viability kernel can help the decision-maker to focus on those uncertainties that are relevant for sustainable management.

We show that not only the absolute value of extreme uncertainties delimit the set of robust viable states, but also the possible arbitrary evolutions of scenarios, switching from one extreme to another between time periods. We label such scenarios worst-case scenarios. However, because the distance between extreme uncertainties can always be amplified, the notion of worst-case is contingent on the level of caution that one chooses to adopt.

Lastly, we examine effort strategies arising from the robust framework.

This chapter is an opportunity to emphasize the different types of analysis and the wide range of information that can be derived from the viability framework to support decision making in the sustainable management of fisheries.

Bibliography

[1] D. J. Agnew, J. Pearce, G. Pramod, T. Peatman, R. Watson R, J. R. Benddington, and T. J. Pitcher. Estimating the Worldwide Extent of Illegal Fishing. *PLoS ONE*, 4(2): e4570, 2009.

[2] J. L. Anderson. Market interactions between aquaculture and the common-property commercial fishery. *Marine Resource Economics*, 2:1–24, 1985.

[3] J. L. Anderson. Aquaculture and the future: Why fisheries economists should care. *Marine Resource Economics*, 17(2):133–151, 2002.

[4] L. G. Anderson. A note on market power in ITQ fisheries. *Journal of Environmental Economics and Management*, 21(3):291 – 296, 1991.

[5] F. Asche, K. G. Salvanes, and F. Steen. Market delineation and demand structure. *American Journal of Agricultural Economics*, 79(1):139–150, 1997.

[6] F. Asche, T. Bjørndal, and J. A. Young. Market interactions for aquaculture products. *Aquaculture Economics & Management*, 5(5-6):303–318, 2001.

[7] J. R. Beddington, D. J. Agnew, and C. W. Clark. Current problems in the management of marine fisheries. *Science*, 316(5832):1713–1716, 2007.

[8] R. M. Bregazzi. Mind the gap. Technical report, The Pew Environment Group, Washington, D.C., 2011.

[9] D. S. Butterworth and A. E. Punt. Experiences in the evaluation and implementation of management procedures. *ICES J. Mar. Sci.*, 56(6):985–998, 1999.

[10] C. Chu. Thirty years later: the global growth of ITQs and their influence on stock status in marine fisheries. *Fish and Fisheries*, 10(12):217–230, 2009.

[11] C. W. Clark. *Mathematical Bioeconomics*. Wiley, New York, second edition, 1990.

[12] K. L. Cochrane. Reconciling sustainability, economic efficiency and equity in fisheries: the one that got away? *Fish and Fisheries*, 1:3–21, 2000.

[13] C. Costello, J. Lynham, S. E. Lester, and S. D. Gaines. Economic incentives and global fisheries sustainability. *Annual Review of Resource Economics*, 2(1):299–318, 2010.

[14] C. Costello, D. Ovando, R. Hilborn, S. D. Gaines, O. Deschenes, and S. E. Lester. Status and solutions for the world's unassessed fisheries. *Science*, 338(6106):517–520, 2012.

[15] P. Cury, C. Mullon, S. Garcia, and L. J. Shannon. Viability theory for an ecosystem approach to fisheries. *ICES J. Mar. Sci.*, 62(3):577–584, 2005.

[16] R. W. D. Davies, S. J. Cripps, A. Nickson, and G. Porter. Defining and estimating global marine fisheries bycatch. *Marine Policy*, 33(4):661 – 672, 2009.

[17] T. Daw and T. Gray. Fisheries science and sustainability in international policy: a study of failure in the European Union's Common Fisheries Policy. *Marine Policy*, 29(3):189 – 197, 2005.

[18] M. De Lara and L. Doyen. *Sustainable Management of Natural Resources. Mathematical Models and Methods*. Springer-Verlag, Berlin, 2008.

[19] C. L. Delgado, N. Wada, M. W. Rosegrant, S. Meijer, and M. Ahmed. Outlook for fish to 2020. Technical report, The WorldFish Center, Food policy Reports no.15, Penang, Malaysia, 2003.

[20] FAO. The State of World Fisheries and Aquaculture. Technical report, Fisheries and Aquaculture Departement, Rome, 2012.

[21] FEAP. European agriculture production report 2003-2012. Technical report, The Federation of European Aquaculture Producers, 2014.

[22] S. Garcia, A. Zerbi, C. Aliaume, T. Do Chi, and G. Lasserre. The ecosystem approach to fisheries. Issues, terminology, principles, institutional foundations, implementation and outlook. *FAO Fisheries Technical Paper*, 443(71), 2003.

[23] M. T. Gibbs. Why ITQs on target species are inefficient at achieving ecosystem based fisheries management outcomes. *Marine Policy*, 34(3):708–709, 2010.

[24] R. Q. Grafton. Individual transferable quotas: theory and practice. *Reviews in Fish Biology and Fisheries*, 6(1):5–20, 1996. ISSN 0960-3166.

[25] S.J. Hall, A. Delaporte, M. J. Phillips, M. Beveridge, and M. O'Keefe. Blue Frontiers: Managing the environmental costs of aquaculture. Technical report, The WorldFish Center, Penang, Malaysia, 2011.

[26] R. Hannesson. Aquaculture and fisheries. *Marine Policy*, 27(2):169–178, 2003.

[27] G. Hardin. The tragedy of the commons. *Science*, (3859), 1968.

[28] R. Hilborn and C. F. Walters. *Quantitative fisheries stock assessment. Choice, dynamics and uncertainty*. Chapman and Hall, New York, London, 1992.

[29] R. Hilborn, J. Parrish, and K. Litle. Fishing rights or fishing wrongs? *Reviews in Fish Biology and Fisheries*, 15(3):191–199, 2005.

[30] J. Hoffmann and M. F. Quaas. Fisheries management and the common pool problem. *Department of Economics, University of Kiel*, 2013.

[31] HTSF. Closing the net: Stopping illegal fishing on the high seas. Technical report, The High Sea Task Force: Fisheries ministers of Australia, Canada, Chile, Namibia, New Zealand and the United Kingdom, WWF, IUCN and the Earth Institute at Columbia University, London, UK, 2006.

[32] A. Karagiannakos. Total allowable catch (TAC) and quota management system in the European Union. *Marine Policy*, 20(3):235 – 248, 1996.

[33] G. Knapp, C. A. Roheim, and J. L. Anderson. *The great salmon run: competition between wild and farmed salmon*. TRAFFIC North America, Washington D.C.: World Wildlife Fund, 2007.

[34] T. Lauck, C. W. Clark, M. Mangel, and G. R. Munro. Implementing the precautionary principle in fisheries management through marine reserves. *Ecological Applications*, 81, Special Issue:72–78, 1998.

[35] M. Lemaire. Le saumon de Norvège en eaux troubles. *Le plus France Info*, 2014.

[36] A. Leroy. Policy coherence against IUU fishing. Technical report, OECD, Paris, 2013.

[37] B. J. McCay. Social and ecological implications of itqs: an overview. *Ocean & Coastal Management*, 28(3):3–22, 1995. ISSN 0964-5691.

[38] F. Natale, J. Hofherr, G. Fiore, and J. Virtanen. Interactions between aquaculture and fisheries. *Marine Policy*, 38:205–213, 2013.

[39] R. L. Naylor, R. J. Goldburg, J. H. Primavera, N. Kautsky, M. C. M. Beveridge, J. Clay, C. Folke, J. Lubchenco, H. Mooney, and M. Troell. Effects of aquaculture on world fish supplies. *Nature*, 405:1017–1024, 2000.

[40] R. L. Naylor, R. W. Hardy, D. P. Bureau, A. Chiu, M. Elliott, A. P., I. Foster, D. M. Gatlin, R. J. Goldburg, K. Hua, and P. D. Nichols. Feeding aquaculture in an era of finite resources. *PNAS*, 106:15103–15110, 2009.

[41] OECD. Why fish piracy persists: The economics of illegal, unreported and unregulated fishing. Technical report, Paris, 2005.

[42] E. Ostrom. Institutions and the environment. *Economic Affairs*, 28(3):24–31, 2008.

[43] D. Pauly, V. Christensen, J. Dalsgaard, R. Froese, and F. Torres. Fishing down marine food webs. *Science*, 279(5352):860–863, 1998.

[44] J.-C. Péreau, L. Doyen, L.R. Little, and O. Thébaud. The triple bottom line: Meeting ecological, economic and social goals with individual transferable quotas. *Journal of Environmental Economics and Management*, 63(3):419–434, 2012.

[45] J. A. Robinson, R. Torvik, and T. Verdier. Political foundations of the resource curse. *Journal of Development Economics*, 79(2):447–468, 2006.

[46] K. J. Sainsbury, A. E. Punt, and A. D. M. Smith. Design of operational management strategies for achieving fishery ecosystem objectives. *ICES Jounal of Marine Science*, 57:731–741, 2000.

[47] M. B. Schaefer. Some aspects of the dynamics of populations important to the management of commercial marine fisheries. *Bulletin of the Inter-American tropical tuna commission*, 1:25–56, 1954.

[48] G. Shamshak and J. Anderson. Offshore aquaculture in the United States: Economic considerations, implications and opportunities. NOAA. Chapter 4 :73-96, 2003.

[49] R. Shotton. Use of property rights in fisheries management. Paper 401/1, FAO, Rome, 2000.

[50] A. D. M. Smith, C. J. Brown, C. M. Bulman, E. A. Fulton, P. Johnson, I. C. Kaplan, H. Lozano-Montes S. Mackinson, M. Marzloff L. J. Shannon, Y.-J Shin, and J. Tam. Impacts of fishing low–trophic level species on marine ecosystems. *Science*, 333:1147–1150, 2010.

[51] J. G. Sutinen and R. J. Johnston. Angling management organizations: integrating the recreational sector into fishery management. *Marine Policy*, 27(6):471–487, 2003.

[52] A. G. J. Tacon and M. Marc. Fishing for feed or fishing for food: Increasing global competition for small pelagic forage fish. *AMBIO: A Journal of the Human Environment*, 2009.

[53] A. G. J. Tacon and M. Metian. Global overview on the use of fish meal and fish oil in industrially compounded aquafeeds: Trends and future prospects. *Aquaculture*, 285: 146–158, 2008.

[54] R. Townsend, R. Shotton, and H. Uchida. Case studies in fisheries self-governance. Technical report, FAO Paper. No. 504., Rome, 2008.

[55] B. R. Turris. A rejoinder to e. pinkerton et al., the elephant in the room: The hidden costs of leasing individual transferable fishing quotas. *Marine Policy*, 34(3):431–436, 2010.

[56] H. Uchida, E. Uchida, J-S. Lee, J-G. Ryu, and D-Y. Kim. Does self-management in fisheries enhance profitability?: Examination of Korea's coastal fisheries. *Marine Resource Economics*, 25(1):37–59, 2010.

[57] UNODC. Transnational organized crime in the fishing industry. Technical report, United Nations Office on Drugs and Crime, 2011.

[58] WCED. Our common future. Technical report, World Commission on Environment and Development, United Nations, New York, 1987.

[59] Y. Ye and J. R. Beddington. Bioeconomic interactions between the capture fishery and aquaculture. *Marine Resource Economics*, 11(2):105–123, 1996.

Chapter 1

Is aquaculture really an option?

1.1 Introduction

While breeding of terrestrial animals was implemented about 8 000 years ago and substituted to hunting quite rapidly, it took us a very long time to repeat the experience with fish. Aquaculture exists in many parts of the world since the Middle Ages but did not replace fishing until now. However, the increasing needs in food fish make things change rapidly.

The world population growth and the increase in standards of living in developing countries result in a growing demand for animal protein. To keep pace with such demand, wild fisheries are subject to high pressure. According to FAO (2012), in 2009, about 57.4% of world marine fish stocks are estimated as fully exploited and 29.9% as overexploited. An increasing trend in the percentage of overexploited, depleted and recovering stocks is observed since the mid-1970s. In the same way, since the early 1990s, overall landings are marked by a small decline. Many agree that the maximum capture fishery potential from world's oceans has been reached.

In the last three decades aquaculture has been the fastest growing food industry, with an annual average growth rate of 8.8% (FAO, 2012). Focusing on fish production for human consumption, aquaculture has nearly doubled this quantity in recent years. This sector managed to provide 47% of world food fish production in 2010 (FAO 2012). In fact, aquaculture is increasingly viewed as a source of food security. According to the FAO's projections, in order to maintain the current level of per capita consumption of fish protein, global aquaculture

production will need to increase by 60% by 2050.

However, the production methods of aquaculture do present certain limitations in terms of environmental sustainability. Aquaculture essential inputs are: land, water, labor, feed and fingerlings. The degree of use of these inputs depends on the characteristics of the production process (whether it is extensive, semi-intensive or intensive), and on the species bred. In any event, inland and costal farms cause the destruction of natural habitats and erode biodiversity. In addition, the release of untreated water, food and faeces damages wild ecosystems, in particular through pathogene invasions. The use of fertilizers in fish diets produces wastes loaded in nitrogen, phosphorus and other substances inducing eutrophication[1]. Regarding fingerlings, they are mainly sourced from the wild rather than derived from hatcheries, occasioning disastrous effects on natural populations (Naylor *et al.*, 2000, FAO, 2012). Finally, aquaculture depends on wild fish stocks for feeding carnivorous and omnivorous species. Fish meal and fish oil, which are key ingredients in aquaculture feed, are made from small oily fish belonging to low trophic levels[2] (LTL) for about 80% and wastes from processed fish for 20% (Fishmeal Information Network, 2011). Fisheries specialised in catches of prey fish, also termed forage fish, are called reduction fisheries.

The demand for fish meal and fish oil participates to the fishing pressure drilled on wild stocks. It consumes 19.4% of global marine landings (FAO, 2012). At date, reduction fisheries are described as fully exploited or over-exploited (Grainger and Garcia, 1996; Alder *et al.*, 2008). What is more, Smith *et al.* (2011) emphasize widespread effects of fishing LTL species on marine ecosystems. The study puts forward the trade-off between protecting biodiversity versus contributing to food security, through direct human consumption of LTL species and by providing feed to the agribusiness. The authors support the conclusion that lower exploitation rates of LTL species are required to mitigate ecosystem impacts of reduction fisheries.

Aquaculture is the world's largest user of fish meal and fish oil: in 2009 it consumed 53% of fish meal and 81% of fish oil world production (IFFO, 2011). The sector has succeeded in maintaining a high growth rate in spite of non-increasing landings of feed fish, thanks

[1]Eutrophication corresponds to a great increase of phytoplankton, due to the abnormal presence of artificial or natural substances in waters, resulting in the depletion of oxygen in the water, which induces reductions in specific fish and other animal populations.

[2]Among the species intended to fish meal production there are anchovy, jack mackerel and sardines.

to important progress in terms of rationalization of fish meal inputs (Asche and Bjorndal, 2011; Shamshak and Anderson, 2008). However, a large increase in aquaculture production is expected, making essential further efficiency improvements in the formulation of fish diets. Especially as reduction fisheries are increasingly targeted for more profitable purposes such as pharmaceutical uses.

Several studies ask about the degree of substitutability between fish meal and plant-based food. Soya meal emerges as a great candidate. It possesses most of the characteristics allowing high flesh quality. However, Kristofersson and Anderson (2006) demonstrate empirically that since the late 90s both types of protein have become poor substitutes, breaking with the strong historical price relationship displayed until then. According to Shamshak and Anderson (2008), beyond some degree of replacement of fish meal by plant-based food, some farmed species are subject to declines in health, growth rate and omega 3 levels due to the lower protein quality and content. The aquaculture industry has recently undertaken the production of a genetically modified salmon species (AquaAdvantage), which growth is enhanced despite very low protein intake. The counterpart of such innovation is that the nutritive properties of the flesh are not conserved. Single cell proteins or zooplankton are considered as potential substitutes to fish meal proteins. Yet, their production costs remain too high to be used in significant amounts in aquaculture feed (Olsen and Hasan, 2012). At date, there does not seem to exist a protein source displaying required properties and profitable at the same time.

Our aim in this chapter is to analyze the impact of the introduction of aquaculture on fish consumption, welfare and on the wild fish stocks, taking into account its dependence on wild feed fish and consumer preferences. The framework retained to investigate these issues is that of a competitive aquaculture sector and open access fisheries.

Motivations for this latter hypothesis are the following. Although the general trend is to tighten regulations, resource management schemes are far from being implemented or efficient worldwide. Furthermore, referring to Costello et al (2012), it appears that countries with several unassessed fisheries, which likely goes along with poor management schemes, have often expanded aquaculture[3]. The depletion of fishery resources and the subsequent

[3]Personal remark of Christopher Costello to the authors.

loss in food production potential may have fostered the development of other fish production techniques. In any event, this observation further supports our assumption that farming co-exists with open access fisheries. Overall, it is necessary to analyze mechanisms at work in absence of intervention, especially in order to properly direct management initiatives.

Our model is highly stylized. It consists of the demand side and three sectors: an edible fish fishery, a reduction fishery and an aquaculture sector producing farmed fish.

The characteristics of the technologies at stake for fishing and farming, as well as two types of interactions, namely biological interactions and economic interactions on the marketplace, are essential.

First, consumers can obtain fish by two means: fish can be either fished or farmed. One important question is the relative efficiency of these two technologies in producing edible fish. Another important point is of course the relative cost of the two production methods.

Secondly, we account for biological interactions between fish stocks to reflect the ecosystem effects of fishing. While the aquaculture sector harvests feed fish to grow farmed fish, the wild edible fish feeds on this same stock. Thus, we investigate how critical is the removal of feed fish for the biomass of the wild edible species and for its supply.

Thirdly, market interactions between wild edible fish and farmed fish arise. Consumers may choose to consume wild or farmed fish, which are strong substitutes, depending on the prices of the two types of edible fish and their preferences.

We derive steady state outcomes from our model as well as the pattern of the trajectories of fish prices, consumption and stock levels to appraise the dynamics resulting of these interactions. Overall, what emerges from our analysis is that when biological interactions are moderate, that is when the wild edible species displays a low degree of food dependence on the feed fish stock, aquaculture increases welfare via a higher total fish supply, and alleviates the pressure on the edible stock, which increases in spite of the fact that the prey species is less abundant. Otherwise, when biological interactions are strong, the harvest of the prey species for the aquaculture industry results in a lower stock and supply of edible fish, and a higher wild fish price. The introduction of the aquaculture food production may even result in lower global fish supply and utility.

In addition, we explore the consequences of an improvement of aquaculture productivity

in the case where biological interactions are absent, and show that when the wild species are heavily exploited, this improvement is beneficial.

As we suspect that such an improvement is at least partly achieved by farming less carnivorous species, we finally endogeneize consumer preferences by assuming that they are carnivorous species-biased. Precisely, we study an extension of the model where preferences vary according to the characteristics of the farmed fish, which we link to the productivity of aquaculture technology. Our assumption is the following. On the one hand, the more carnivorous the farmed species is, the more inefficient is its breeding, in the sense that the production of 1kg of flesh requires a lot more than 1kg of wild feed fish. On the other hand, the taste of consumers for the farmed fish depends on its diet: the more carnivorous the farmed species is, the more consumers like it[4], or the more substitutable to the wild edible fish they perceive it is. It turns out that the improvement of the productivity of aquaculture has a negative side effect: aquaculture farms fish that consumers like less, or find less substitutable to wild fish. We conjecture that in these circumstances there may exist a utility-maximizing farmed species.

A few papers have investigated the market interactions between aquaculture and capture fisheries. Anderson (1985) supposes that the wild and the farmed species are the same, and consequently have the same market price. He shows that in the case of a fishery in open access exploited beyond the maximum sustainable yield, the entry of competitive aquaculturists increases total fish supply, thereby, reducing consumer price and increasing natural fish stock. Indeed, profit loss mechanically reduces fishing effort. Ye and Beddington (1996) assume both goods are imperfect substitutes with positive cross-price elasticities. Similarly, the authors find positive consumer benefits of aquaculture via increased fish supply and reduced prices. Yet, the imperfect substitutability between farmed and wild products limits the extent to which aquaculture production impacts the capture fishery activity. Hannesson (2002) considers both market and biological interactions. Like Anderson (1985), he assumes that the edible fish and the farmed fish are the same species. This species feeds on a prey in the wild, which is also the input of the aquaculture activity. The argument of the paper

[4]Worldwide, carnivorous species such as grouper, cod-fish, halibut, sole etc. display higher economic values that omnivorous ones (FranceAgriMer, 2012; Alaska Departement of Fish &Game, 2010).

revolves around the comparison between the costs of fishing and farming on the one hand, and the efficiency of the transformation of one unit of feed fish into edible fish in the wild and within the aquaculture technology on the other hand. The author relies on numerical simulations. In open access, total food fish production is found to be slightly higher than without aquaculture, but the wild edible fish stock severely drops relatively to the situation absent aquaculture.

Hannesson (2002) is probably the paper that comes closest to ours. Yet, in Hannesson (2002) the farmed and wild edible fish are the same, and have the same price, i.e. are perfect substitutes. One of our main contributions is that we account for the demand side. This allows us to analyze how preferences for wild versus farmed fish intervene in price and market equilibria. Our theoretical setting also allows us to examine the consequences of different degrees of biological interactions on these equilibria.

The remaining of the chapter is as follows. Section 2 presents the demand side features. Section 3 describes the two-species biological model we consider. Section 4 derives the short run dynamics and the long run state of the edible fishery in open access, in absence of aquaculture. This constitutes our baseline situation for appraising the impact of the aquaculture activity. In Section 5 we introduce aquaculture and proceed to the coupling of the different sectors. We analyze the resulting steady state, its stability, and compare it to that of the baseline situation. We also examine the consequences of an improvement of aquaculture's efficiency. We finally endogeneize consumer tastes in Section 6. Section 7 concludes.

1.2 The demand side

Consumers purchase two types of goods: wild fish and farmed fish. The utility function of the representative consumer at each date t is given by:

$$U(Y_{1t}, Y_{2t}) = \left[(1-\alpha)Y_{1t}^{1-\frac{1}{\sigma}} + \alpha Y_{2t}^{1-\frac{1}{\sigma}}\right]^{\frac{1}{1-\frac{1}{\sigma}}}, \quad \sigma > 1, \quad \alpha \in]0,1[\tag{1.1}$$

with Y_1 the wild fish and Y_2 the farmed one.

It is reasonable to assume that the two fish products are strong substitutes. Hence, the elasticity of substitution, σ, is greater than 1.

The budget constraint of the representative consumer is:

$$P_{1t}Y_{1t} + P_{2t}Y_{2t} = I \tag{1.2}$$

where I represents total expenditures on fish consumption, exogenous and supposed to be stationary, and P_{1t} and P_{2t} are respectively the market price of wild and farmed fish.

When maximizing the utility function with respect to the budget constraint we obtain the following demand functions for the two types of fish:

$$Y_{1t}^d = \frac{I}{P_{1t}\left[1 + \left(\frac{\alpha}{1-\alpha}\right)^{\sigma}\left(\frac{P_{1t}}{P_{2t}}\right)^{\sigma-1}\right]} \tag{1.3}$$

$$Y_{2t}^d = \frac{I}{P_{2t}\left[1 + \left(\frac{1-\alpha}{\alpha}\right)^{\sigma}\left(\frac{P_{2t}}{P_{1t}}\right)^{\sigma-1}\right]} \tag{1.4}$$

As it is well known when preferences are represented by a CES utility function, the response of Y_1^d (resp. Y_2^d) to a variation of P_2 (resp. P_1) depends on the value of the elasticity of substitution. Here, the two goods are strongly substitutable ($\sigma > 1$). Therefore Y_1^d (resp. Y_2^d) is increasing in P_2 (resp. P_1).

1.3 Biological interactions

Now, we introduce the possibility of biological interactions between the two wild species. This is only a possibility, and not necessarily the general rule: it may be the case that no biological interactions exist, because both fish stocks belong to totally different geographical areas. Indeed, the Peruvian anchovies constitute the world largest fishery, landings reaching 10% of global fish catches in peak years. It is the most important input in the fish meal and oil industry. By relying on this industry, salmon farming in Norway or pangasius farming in Vietnam do increase pressure on fish resources worldwide but do not impact reduction fisheries at the local level.

Our assumption is that when biological interactions exist between the two wild species, they are of the predator-prey type. Species 1 a high-value species harvested for human consumption — the predator —- while species 3 is a low-value non-edible pelagic species — the prey.

We characterize biological interaction by specific functional forms to be able to derive analytical results. The Lotka-Volterra model is commonly used to describe interspecies dynamics in the literature (Sivert and Smith, 1977; Hannesson, 1983, 2002; Hofbauer and Sigmund, 1998). A two-species, non-linear density dependent version of the Lotka-Volterra model is considered:

$$F_1(X_{1t}, X_{3t}) = a_1 X_{1t} - b_1 X_{1t}^2 + d_1 X_{1t} X_{3t} \tag{1.5}$$

$$F_3(X_{1t}, X_{3t}) = a_3 X_{3t} - b_3 X_{3t}^2 - d_3 X_{1t} X_{3t} \tag{1.6}$$

where X_1 and X_3 stand for the stock of species 1 and 3. Parameters d_1 and d_3 define the ecological interdependence between the two species, of the predator-prey type: $d_1 \geq 0, d_3 \geq 0$. Parameters $b_1 \geq 0$, $b_3 \geq 0$ traduce the fact that the maintenance needs of the biomass grow faster than the ability to acquire food as the biomass increases (Hannesson, 1983). The rate of growth of each species in absence of the other is given by $a_1 \geq 0$, $a_3 > 0$. If $a_1 > 0$, species 1 can survive without species 3. If $a_1 = 0$, species 3 is necessary to the survival of species 1.

Without any human intervention, the system composed of the two fish populations evolves according to:

$$\dot{X}_{1t} = F_1(X_{1t}, X_{3t}) \tag{1.7}$$

$$\dot{X}_{3t} = F_3(X_{1t}, X_{3t}) \tag{1.8}$$

It immediately appears that there exists four steady states satisfying $\dot{X}_{1t} = \dot{X}_{3t} = 0$:

1. a steady state where both populations disappear: $\widetilde{X}_1 = \widetilde{X}_3 = 0$;
2. a steady state where population 1 goes extinct but not population 3: $\widetilde{X}_1 = 0$, $\widetilde{X}_3 = a_3/b_3$;
3. a steady state where it is the contrary: $\widetilde{X}_1 = a_1/b_1$, $\widetilde{X}_3 = 0$;
4. an interior steady state where both populations coexist:

$$\widetilde{X}_1 = \frac{b_3}{b_1 b_3 + d_1 d_3}\left(a_1 + \frac{a_3 d_1}{b_3}\right)$$
$$\widetilde{X}_3 = -\frac{d_3}{b_1 b_3 + d_1 d_3}\left(a_1 - \frac{a_3 b_1}{d_3}\right)$$

The condition of existence of this steady state is:

$$\frac{a_1}{b_1} < \frac{a_3}{d_3} \tag{1.9}$$

Notice that it is always satisfied if $a_1 = 0$; that is, if species 1 cannot survive without species 3.

We show in Appendix B.1 that when parameters satisfy condition (1.9) , the stable steady state that prevails in the long run is steady state 4, where both fish populations coexist; when it is not satisfied, the stable steady state is steady state 3, where the prey population is extinct. In the remaining of the chapter we make the assumption that absent any human intervention the equilibrium that prevails is the one where both fish populations coexist. Therefore, condition (1.9) is supposed to be satisfied.

1.4 The baseline situation: capture fishery alone

We first study the biological and economic features of the capture fishery in absence of aquaculture. This will be useful to appraise the impact of aquaculture activity. In this baseline situation, utility is linear in the quantity of wild fish consumed, and the demand function reduces to:

$$Y_{1t}^d = \frac{I}{P_{1t}} \tag{1.10}$$

The dynamics of the capture fishery reads:

$$\dot{X}_{1t} = F_1(X_{1t}, X_{3t}) - Y_{1t} \tag{1.11}$$

$$Y_{1t} = q_1 E_{1t} X_{1t} \tag{1.12}$$

and the evolution of the feed stock is given by equation (1.8). The production function of the fishery is the classical Schaefer (1957) production function, where $q_1 > 0$ is the catchability coefficient of the species and E_1 the effort devoted to fishing.

Fishermen profit is given by:

$$\pi_{1t} = P_{1t} Y_{1t} - c E_{1t} \tag{1.13}$$

where c stands for the unit cost of effort.

We make the assumption that the wild resource is in open access. Consequently, fishermen enter the sector until dissipation of the rent (Gordon 1954):

$$\dot{E}_{1t} = \beta\pi_{1t} = \beta(q_1 P_{1t} X_{1t} - c)E_{1t}, \quad \beta > 0 \tag{1.14}$$

We add to the previous dynamic system the equilibrium of the wild fish market at each date $Y_{1t} = Y_1^d(P_{1t})$, where $Y_1^d(P_{1t})$ is given by (1.10). Eliminating P_1 and Y_1 yields the following three-dimensional dynamic system in X_{1t}, X_{3t} and E_{1t} :

$$\begin{cases} \dot{X}_{1t} = F_1(X_{1t}, X_{3t}) - q_1 E_{1t} X_{1t} \\ \dot{X}_{3t} = F_3(X_{1t}, X_{3t}) \\ \dot{E}_{1t} = \beta\left(I - cE_{1t}\right) \end{cases} \tag{1.15}$$

It immediately appears that there exists four steady states satisfying $\dot{X}_{1t} = \dot{X}_{3t} = \dot{E}_{1t} = 0$:

1. a steady state where both populations disappear: $X_1^* = X_3^* = 0$, $E_1^* = 0$;

2. a steady state where population 1 goes extinct but not population 3: $X_1^* = 0$, $X_3^* = a_3/b_3$, $E_1^* = 0$;

3. a steady state where it is the contrary: $X_1^* = \frac{1}{b_1}\left(a_1 - \frac{q_1 I}{c}\right)$, $X_3^* = 0$, $E_1^* = \frac{I}{c}$; the condition of existence of this steady state is $I < ca_1/q_1$. Notice that this situation cannot characterize the long term of this economy under condition (1.9). Indeed, as this condition ensures that both species coexist when there is no human intervention, fishing of the predator (species 1), cannot worsen the long term stock of the prey (species 3).

4. an interior steady state where both populations coexist:

$$X_1^* = \frac{b_3}{b_1 b_3 + d_1 d_3}\left(a_1 + \frac{a_3 d_1}{b_3} - \frac{q_1 I}{c}\right) \tag{1.16}$$

$$X_3^* = -\frac{d_3}{b_1 b_3 + d_1 d_3}\left(a_1 - \frac{a_3 b_1}{d_3} - \frac{q_1 I}{c}\right) \tag{1.17}$$

$$E_1^* = \frac{I}{c} \tag{1.18}$$

The conditions of existence of this steady state in the case of predator-prey interactions (d_1, $d_3 > 0$) are: $\frac{q_1 I}{c} < a_1 + \frac{a_3 d_1}{b_3}$ and $\frac{q_1 I}{c} > a_1 - \frac{a_3 b_1}{d_3}$. Under condition (1.9), the second one is always satisfied, while the first one may be binding.

Therefore, condition

$$I < I_w(d_1) := \frac{c}{q_1}\left(a_1 + \frac{a_3 d_1}{b_3}\right) \tag{1.19}$$

gives the maximum revenue that consumers can spend on fish without inducing the extinction of the edible species. This revenue I_w is an increasing function of d_1, the parameter characterizing the strength of the predator effect. The higher is d_1, the higher is the surplus growth of species 1 subsequent to a unit intake of species 3. Hence, the more edible fish can be caught.

Notice that at the interior steady state $X_1^* < \widetilde{X}_1$ and $X_3^* > \widetilde{X}_3$: fishing of population 1 alleviates the predator effect on population 3.

We show in Appendix B.2 that under condition (1.19), the steady state that prevails is the interior steady state where both wild fish populations coexist in the long run. It is globally stable: for any initial value of the effort below a certain level[5], the dynamic paths followed by the stock and effort converge to the steady state, which is a stable node or a stable focus, depending on the parameters. When condition (1.19) is not satisfied, i.e. when the revenue consumers spend on fish is too high, the wild edible species collapses and the steady state that prevails is steady state 2.

1.5 Introducing aquaculture

We now introduce the aquaculture sector, which exploits the stock of pelagic fish as an input, and study the long run outcomes derived from the coupling of the demand side and all three productive sectors. We also identify the nature of the equilibrium of this system. Next, we compare these steady state outcomes to that of the baseline situation. Lastly, we look at the

[5]Suppose that the initial effort is $E_{10} \geq 1/q_1$. Then, according to the specification of the catch function, the initial catch is $Y_{10} \geq X_{10}$: the entire stock is harvested at once, extinction occurs immediately. Hence the initial effort must be $E_{10} < 1/q_1$.

influence of the efficiency of the aquaculture technology on the long run status of both wild fish stocks and welfare.

1.5.1 The aquaculture sector and feed fishery

Farmers purchase fish meal and fish oil in the form of compounded feed, which are pellets providing nutrients and different supplements to farmed fish. These pellets are produced by a specialized industry. Here, we consider for simplicity that farmers buy feed fish directly to the reduction fishery. It is actually their unique variable input in this model. Other inputs, mainly capital and labor, are supposed to be fixed and normalized to 1. Feed fish is harvested from the prey stock, X_3, distinct from the predator fish stock exploited for human consumption X_1. Feed fish harvesting takes place in open access. Its price is set by the equalization of fishermen supply and the demand from aquaculture. Regarding farmers, they are in competition on the farmed fish market. They decide at each date of the feed quantity that maximizes their profit.

The production function of the representative farmer reads:

$$Y_2 = kY_3^{\gamma} \tag{1.20}$$

with Y_2 the farmed fish production, Y_3 the input of feed fish, $\gamma \in]0,1[$ the share of feeds in the production technology of farmed fish. It is set below one to account for the decreasing marginal productivity of feed fish. The parameter $k > 0$ is the efficiency of the aquaculture sector in converting feed fish into farmed fish flesh. It may be interpreted either as the diet of the farmed species or as technical progress. In the first case, a high k means that the aquaculture sector has chosen to farm a rather omnivorous fish species, which does not require too much feed fish to grow. In the second case, a high k implies that a given species can be grown with relatively few feed, more exactly, low animal protein intake. In the remaining of the document we comment our results on the basis of the first interpretation of k — k reflects a specific species diet — in order to ease understanding.

Notice that k may be related to the well known FIFO (fish in-fish out) ratio, which gives the number of tons of wild fish necessary to produce one ton of farmed fish (including fish oil and fish meal requirements). The FIFO ratio varies quite a lot between surveys. Tacon

and Metian (2008) reports an overall FIFO ratio of 0.7. This ratio includes all bred species: crustaceans, carnivorous, omnivorous and herbivorous. At the carnivorous species-group level, the study reports a salmon FIFO ratio of 4.9. Naylor *et al.* (2009) finds fairly close figures to those conveyed in Tacon and Metian (2008). On the other hand, IFFO (2012) finds an overall FIFO ratio of 0.3 and a salmon FIFO ratio of 1.4. In any event, both studies attest of substantial decrease in FIFO ratio since the 90's.

Maximizing their profit, $\pi_{2t} = P_{2t}Y_{2t} - P_{3t}Y_{3t}$, farmers buy feed to produce farmed fish up to the point where the gain provided to the farming industry by a marginal increase in feed input is equal to its cost (i.e. P_{3t}). Given our specification of the production function of aquaculture (1.20), this yields:

$$\frac{P_{3t}Y_{3t}}{P_{2t}Y_{2t}} = \gamma. \tag{1.21}$$

Since the feed fishery is also in open access, fishermen enter the sector until dissipation of the rent, and we have the equivalent of equation (1.14). The unit cost of fishing c is supposed to be the same in the two fishing sectors, as well as the speed of adjustment β.

The fact that revenues from the aquaculture activity are directly proportional to the revenues of the feed industry (equation (1.21)) allows us to aggregate the aquaculture sector and the feed sector and to write the dynamic system representing the supply of farmed fish as:

$$\begin{cases} \dot{X}_{3t} = F_3(X_{1t}, X_{3t}) - \left(\frac{Y^s_{2t}}{k}\right)^{\frac{1}{\gamma}} \\ \dot{E}_{3t} = \beta\left(\gamma P_{2t} Y^s_{2t} - cE_{3t}\right) \\ Y^s_{2t} = k\left(q_3 E_{3t} X_{3t}\right)^{\gamma} \end{cases} \tag{1.22}$$

This dynamic system represents the evolutions of the feed fish stock and the effort devoted to fishing the feed species as functions of the price of farmed fish and the aquaculture's technology characteristics only. It can be directly compared to the corresponding system for wild fish:

$$\begin{cases} \dot{X}_{1t} = F_1(X_{1t}, X_{3t}) - Y^s_{1t} \\ \dot{E}_{1t} = \beta\left(P_{1t} Y^s_{1t} - cE_{1t}\right) \\ Y^s_{1t} = q_1 E_{1t} X_{1t} \end{cases} \tag{1.23}$$

where the evolutions of the edible fish stock and the effort depend on the price of wild fish only.

Interactions between the two systems will come from the demands for wild and farmed fish, depending on both prices, and the market equilibria, as we are going to show.

1.5.2 The coupling

We now suppose that at a given date, let's say $t = 0$, where the wild edible fishery is at the steady state, the aquaculture activity is introduced. The initial effort is supposed to satisfy: $E_{30} < 1/q_3$, so that the entire feed fish stock is not harvested at once. Hence at $t = 0$ wild fishing and aquaculture coexist. In what follows we study the evolution of these two activities over time and the long run equilibria to which the system may converge. A priori, these equilibria may be of four types: one interior equilibrium where wild fishing and aquaculture still coexist, an equilibrium where aquaculture has disappeared, an equilibrium where the wild edible fishery has collapsed, and lastly an equilibrium where both fisheries and aquaculture have collapsed. But according to the specification of the utility function (1.1), when one of the arguments Y_1 or Y_2 tends to zero, its marginal utility becomes infinite. Market forces are not going to allow this situation to happen, which implies that the only relevant long term equilibria will be the interior one (coexistence of both fisheries) and the one where both fisheries collapse simultaneously.

Starting from systems (1.23) and (1.22), we introduce demands for both types of fish and the equilibria of the two fish markets. Define

$$A_t = \frac{1}{1 + \left(\frac{1-\alpha}{\alpha}\right)^{\sigma} \left(\frac{P_{2t}}{P_{1t}}\right)^{\sigma-1}} \tag{1.24}$$

From (1.3) and (1.4), the two demand functions can be written as:

$$P_{1t} Y_{1t}^d = (1 - A_t) I \tag{1.25}$$

$$P_{2t} Y_{2t}^d = A_t I \tag{1.26}$$

Hence $A_t \in [0, 1]$ represents the share of consumer expenditures allocated to buying farmed fish, and $1 - A_t$ the share allocated to buying wild fish. A_t characterizes the market interactions between the fish populations, stemming from consumer preferences, as opposed to the biological interactions studied above. When $A_t = 0$, consumers consume only wild fish. On

the contrary, when $A_t = 1$, only farmed fish is consumed. According to the remark we made above about the consequences of the specification of the utility function, we are going to focus on the case where both wild fish populations exist at the steady state, and determine the conditions allowing this co-existence as well as the dynamic properties of this equilibrium. Hence we suppose that $A_t \in]0, 1[\ \forall t$.

The equilibria on the fish markets read:

$$P_{1t} q_1 E_{1t} X_{1t} = (1 - A_t) I \tag{1.27}$$

$$P_{2t} k \left(q_3 E_{3t} X_{3t}\right)^{\gamma} = A_t I \tag{1.28}$$

The ratio of equations (1.28) and (1.27) yields:

$$\frac{A_t}{1 - A_t} = \frac{P_{2t}}{P_{1t}} \frac{k \left(q_3 E_{3t} X_{3t}\right)^{\gamma}}{q_1 E_{1t} X_{1t}}$$

Replacing the price ratio by its expression as a function of A given by (1.24) allows us to obtain:

$$\frac{A_t}{1 - A_t} = \frac{\alpha}{1 - \alpha} \left(\frac{k \left(q_3 E_{3t} X_{3t}\right)^{\gamma}}{q_1 E_{1t} X_{1t}} \right)^{\frac{\sigma - 1}{\sigma}} \tag{1.29}$$

The final dynamic system describes the evolutions of the two wild fish stocks and the two efforts exerted. It is obtained by putting together systems (1.22) and (1.23), and using (1.25) and (1.26) to eliminate $P_{1t} Y_{1t}$ and $P_{2t} Y_{2t}$:

$$\begin{cases} \dot{X}_{1t} = F_1(X_{1t}, X_{3t}) - q_1 E_{1t} X_{1t} \\ \dot{E}_{1t} = \beta \left[(1 - A_t) I - c E_{1t} \right] \\ \dot{X}_{3t} = F_3(X_{1t}, X_{3t}) - q_3 E_{3t} X_{3t} \\ \dot{E}_{3t} = \beta \left[\gamma A_t I - c E_{3t} \right] \end{cases} \tag{1.30}$$

where A_t is given by (1.29).

The interior steady state associated to this system is characterized by the following equations, giving the two stationary stocks and efforts as functions of A, which is itself a function

of these same variables:

$$\widehat{X}_1 = \frac{b_3}{b_1 b_3 + d_1 d_3}\left(x_1 + y_1 \widehat{A}\right) \text{ with } x_1 = a_1 + \frac{a_3 d_1}{b_3} - \frac{q_1 I}{c} \text{ and } y_1 = \left(q_1 - \frac{d_1}{b_3}\gamma q_3\right)\frac{I}{c} \tag{1.31}$$

$$\widehat{E}_1 = \frac{I}{c}(1 - \widehat{A}) \tag{1.32}$$

$$\widehat{X}_3 = -\frac{d_3}{b_1 b_3 + d_1 d_3}\left(x_3 + y_3 \widehat{A}\right) \text{ with } x_3 = a_1 - \frac{a_3 b_1}{d_3} - \frac{q_1 I}{c} \text{ and } y_3 = \left(q_1 + \frac{b_1}{d_3}\gamma q_3\right)\frac{I}{c} \tag{1.33}$$

$$\widehat{E}_3 = \gamma \frac{I}{c}\widehat{A} \tag{1.34}$$

$$\frac{\widehat{A}}{1 - \widehat{A}} = \frac{\alpha}{1 - \alpha}\left(\frac{k\left(q_3 \widehat{E}_3 \widehat{X}_3\right)^{\gamma}}{q_1 \widehat{E}_1 \widehat{X}_1}\right)^{\frac{\sigma - 1}{\sigma}} \tag{1.35}$$

Proposition 1 contains our results concerning the study of the interior steady state. Figure 1.1 provides a simple representation of these results.

Proposition 1 *(i) A sufficient condition of existence of an interior steady state where wild fishing and aquaculture coexist is:*

$$I < \overline{I} := c\left(\frac{a_1}{q_1} + \frac{a_3}{\gamma q_3}\right) \tag{1.36}$$

Under condition (1.36), the interior steady state is unique.

(ii) Absent biological interactions ($d_1 = d_3 = 0$), the unique interior steady state, when it exists, is globally stable; this remains true when biological interactions are moderate (sufficient conditions for stability are: $d_1 \leq b_3 \frac{q_1}{q_3}$, $d_3 \leq \frac{b_1}{\gamma}$). Besides, whatever the level of biological interactions, if the revenue spent on fish I is sufficiently small, the unique steady state is globally stable.

(iii) If $I \geq \overline{I}$, when $d_1 \leq \overline{d}_1 := b_3 \frac{q_1}{\gamma q_3}$, there is no interior steady state; but when $d_1 > \overline{d}_1$ there may exist up to 2 interior steady states.

Proof. See Appendices A and B.3. □

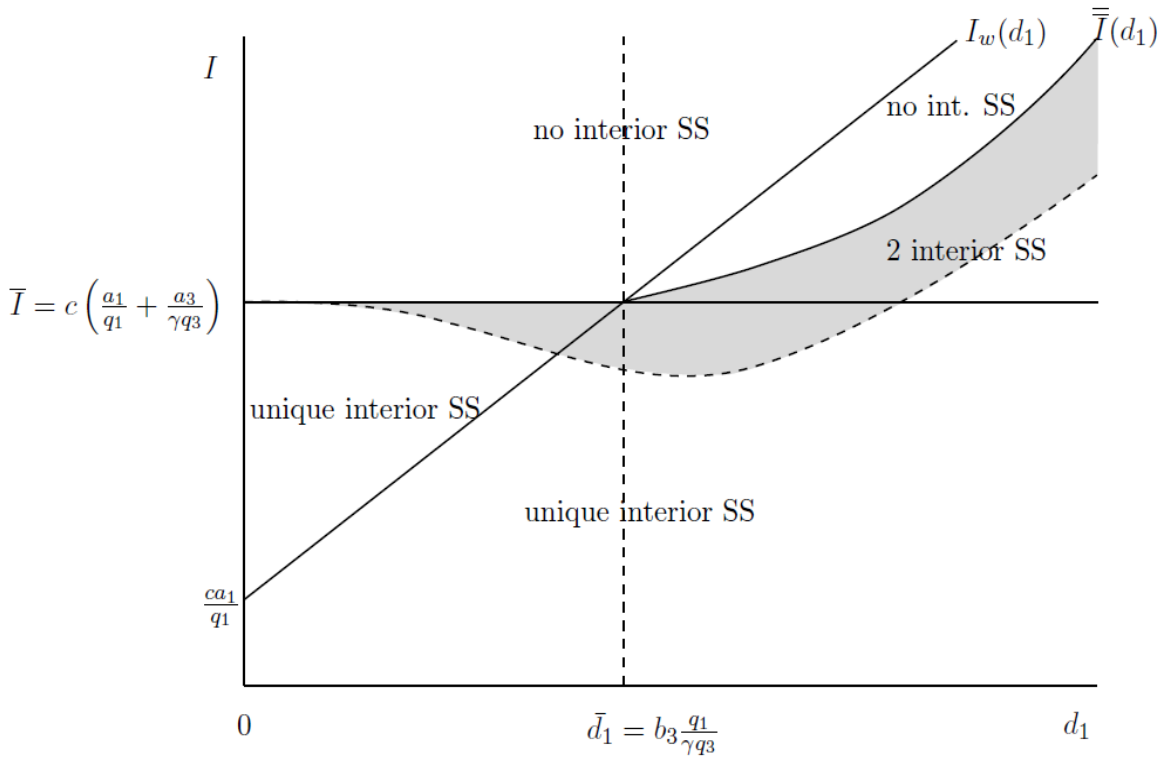


Figure 1.1: Existence, uniqueness and stability of an interior steady state (shaded zone: instability)

Absent biological interactions, the interior steady state, provided that it exists i.e. that condition (1.36) is satisfied, is unique and globally stable. As one would expect, the threshold $\overline{I}$ is an increasing function of the unit cost of effort and the intrinsic growth rate of each species, and a decreasing function of the catchability coefficient of both fisheries. It also depends on γ, the elasticity of farmed fish production to feed input. The higher γ, the lower $\overline{I}$, highlighting the fact that the dependence of aquaculture on feed fish must be mitigated for fish production to cope with population growth and increasing demand. Notice that $\overline{I}$ does not depend on k, the parameter characterizing the productivity of the aquaculture technology. When the revenues spent on fish consumption tend towards the threshold $\overline{I}$ both wild fish stocks tend to 0. When I reaches the threshold, we assist to the simultaneous collapse of the edible wild fish stock and the feed one. The relevant steady state is then $\widehat{X}_1 = \widehat{X}_3 = 0$.

When biological interactions are moderate these results still hold. But stronger biolog-

ical interactions coupled to a relatively high revenue spent on fish consumption (although smaller than $\overline{I}$) may provoke instability (see the shaded area in Figure 1.1). Then the system converges to the steady state where both populations collapse.

For high biological interactions, namely when the parameter d_1 characterizing the predator effect is above the threshold $\overline{d}_1$, the revenue that the system can bear is increased above $\overline{I}$ and there may exist two, one or no steady states. More precisely, there exist two steady states for $I < \overline{\overline{I}}(d_1)$ and no steady state for $I > \overline{\overline{I}}(d_1)$, $I = \overline{\overline{I}}(d_1)$ being the limit case (see Figure 1.1). We were unable to compute analytically the value of $\overline{\overline{I}}(d_1)$, nor to obtain analytical results for the stability of the steady state(s), so we resort to numerical simulations.

We use the values proposed in Hannesson (2002) for biological parameters, while economic and technological parameters are chosen such that steady state outcomes sketch the state of world resources, market prices and quantities, in relative values. The parameter k is set such that aquaculture is very inefficient, meaning that the farmed species is a high value carnivorous species with a high FIFO. The parameter γ gives feed costs equal to 50% of the aquaculture production value. Such value is likely, though belonging to the upper range values of γ reported by Asche and Bjorndal (2011) for the salmon industry. Parameter α, which weights farmed fish in preferences, is set to 0.4, implying that consumers weight wild fish more than farmed fish. This assumption is grounded on the following argument. First, there exists large empirical evidence that for a given species the price of the wild product is higher than the price of the farmed one (FranceAgriMer, 2012). Furthermore, aquaculture does not actually have the capacity to produce all species existing in the wild. At date, the food fish supply of capture fisheries is much more diversified, which is something consumers value (Quaas and Requate, 2013). Lastly, Nielsen *et al.* (2007) establishes that wild products are perceived to be healthier[6]. Table 1.1 gives our reference calibration, where we set $\overline{d}_1 = 0.005$ and $\overline{I} = 11$.

We make the assumption that the ratio d_3/d_1 remains the one of the reference calibration

[6]When it comes to farmed salmon, for which market integration has been extensively studied, facts and literature support the idea that consumers are indifferent between wild or farmed products. Going even further, Knapp *et al.* (2007) argues that for a same salmon species, consumers tend to prefer the farmed product for its consistent quality, the reliability of its supply and its more appealing aspect. But farmed salmon is an exception, which experienced a rather unique market story.

Table 1.1: Calibration

a_1	b_1	d_1	q_1	a_3	b_3	d_3	q_3	σ	c	γ	k	β	α
0.01	0.05	0.02	0.01	0.09	0.01	0.05	0.04	2	2	0.5	0.04	0.05	0.4

above when d_1 varies (i.e. $d_3/d_1 = 2.5$), and compute numerically the region where the steady state is unstable by making I and d_1 vary (see Figure 1.1).

The simulations also allow us to see that in the region where there exist two steady states, either both are unstable or one of them is unstable and the other stable; in this last case, the stable steady state is the one corresponding to the smaller $\widehat{A}$.

1.5.3 Comparison with the baseline situation

Intuition would lead us to think that the introduction of aquaculture is always beneficial for consumers, whatever their preferences and the efficiency of the aquaculture technology, because more options are offered to them. We are going to show that this is not the case in our model. Such a result can be explained by the fact that both fisheries are in open access, which leads fishermen to take inefficient decisions. Moreover, intuition does not have much to offer about the effects of the introduction of aquaculture on biological populations, on fish price, on the quantities consumed or on the effort devoted to fishing in presence of biological interactions. The following proposition compares the steady state outcomes obtained in the baseline situation where both fish populations coexist (equations (1.16) to (1.18)) to those obtained when the wild edible fishery and aquaculture coexist (equations (1.31)-(1.35)).

Proposition 2 *Introducing aquaculture leads in the long run to:*

(i) a smaller total effort devoted to fishing;

(ii) a higher stock of edible wild fish and a lower price iff $d_1 < \overline{d}_1$*, and vice versa, and a lower feed fish stock in all events;*

(iii) an ambiguous effect on wild fish consumption when $d_1 < \overline{d}_1$, a decrease of wild fish consumption when $d_1 > \overline{d}_1$, and an ambiguous effect on total fish consumption in all events;

(iv) a higher utility when $d_1 \leq \overline{d}_1$, but a possibly negative effect on utility when $d_1 > \overline{d}_1$.

Proof. See Appendix C. □

Table 1.2: Comparison with the baseline

Moderate biological interactions: $d_1 < \overline{d}_1$	**Strong biological interactions:** $d_1 > \overline{d}_1$
$\widehat{E}_1 + \widehat{E}_3 < E_1^*$	$\widehat{E}_1 + \widehat{E}_3 < E_1^*$
$\widehat{X}_1 > X_1^*$	$\widehat{X}_1 < X_1^*$
$\widehat{P}_1 < P_1^*$	$\widehat{P}_1 > P_1^*$
$\widehat{Y}_1 \begin{cases} > Y_1^* = 0 \text{ when } I_w(d_1) < I < \overline{I} \\ \gtreqless Y_1^* \text{ otherwise} \end{cases}$	$\widehat{Y}_1 < Y_1^*$
$\widehat{U} > U^*$	$\widehat{U} \gtreqless U^*$

Proposition 2, which results are summarized in Table 1.2, calls for the following comments.

The total effective long run level of fishing effort is of course $\widehat{E}_1 + \widehat{E}_3$. We show in Appendix C that there also exists a virtual total level of effort I/c, constant, which must be splitted into an effective effort $\widehat{E}_1$ devoted to catch the edible wild species, and a virtual effort $\widehat{E}_3/\gamma > \widehat{E}_3$ devoted not only to catch the feed species but also to transform it into edible farmed fish. Total effective fishing effort is smaller with aquaculture than without, whatever the initial state of the edible wild fish stock.

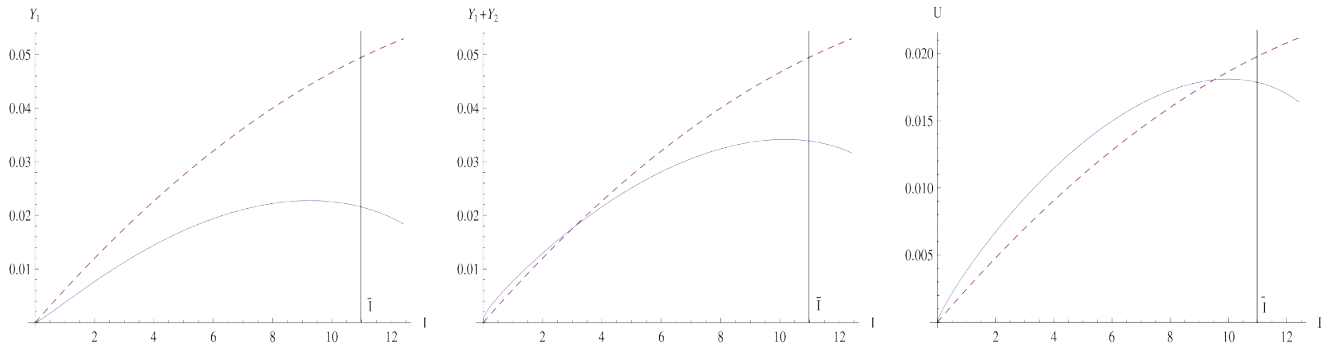
When the predator effect is moderate ($d_1 < \overline{d}_1$), the long run consequences of the introduction of aquaculture are conform to what is expected, and to the results found in the literature (Anderson, 1985; Ye and Beddington, 1996). That is, the effects of market interactions dominate the effects of biological interactions. The introduction of aquaculture does alleviate the pressure on the wild edible fish stock, in the sense that this stock is higher in the long run with aquaculture than without. It may also lead to a higher *total* fish consumption, which is not surprising and, less intuitively, to a higher *wild* fish consumption. In particular, when $I_w(d_1) < I < \overline{I}$ (see Figure 1.1), the introduction of aquaculture prevents the collapse of the wild fishery. Finally, the introduction of aquaculture is always beneficial to consumers, whatever their preferences and the efficiency of the aquaculture technology. When biological interactions are moderate, aquaculture is really an option to increase food security.

When the predator effect is strong ($d_1 > \overline{d}_1$) the results are quite different. In this case, the effects of biological interactions dominate those of market interactions. Aquaculture worsens the pressure on the wild edible fish stock and leads to a decrease of total wild fish stocks in the long run. Indeed, as the introduction of aquaculture reduces the stock of feed fish in the long run, less food is left for the predatory species which growth rate decreases. Then, even if species 1 is less harvested, its long run stock decreases because of the shortage of its prey. For the same reason, the introduction of aquaculture, requiring the exploitation of low value fisheries that were not exploited before, has ambiguous effects on total fish consumption and utility. In particular, when $\overline{I} < I < I_w(d_1)$ (see Figure 1.1), introducing aquaculture may lead to a decrease in welfare, and even to the instability of the system and the collapse of both fish stocks, which would have survived absent aquaculture.

We illustrate the previous results in the case of a strong predatory effect by numerical simulations, performed with the value of parameters in Table 1.1, except that $d_1 = 0.02 > \overline{d}_1$. Figure 1.2 shows the catch of edible fish, the total catch and utility as functions of the revenue I.

1.5.4 Improving the efficiency of aquaculture

The efficiency of the aquaculture technology is traduced by parameter $k > 0$, which intervenes in the production function of the aquaculture sector, as it conveys the sector efficiency



Wild edible catch | Total catch | Consumer utility

Figure 1.2: Introducing aquaculture may reduce the long run wild edible fish catch, total catch and welfare (dashed curves: absent aquaculture; plain curves: with aquaculture)

in transforming low-value feed fish into high-value edible fish (see (1.20)). As explained explained earlier, k may be interpreted as the farmed species diet. When k is high the quantity of feed fish required to produce farmed fish is low. This implies that the species farmed is rather omnivorous or herbivorous. Conversely, when k is low the farmed species is a carnivorous one. According to this interpretation, the species of the farmed fish varies with k.

We study the influence of k on the steady state variables, when the wild fishery and aquaculture coexist. To do so, we perform a comparative statics exercise using system (1.31)–(1.35), in the neighborhood of the interior steady state. As mentioned earlier, the hypothesis of biological interactions between edible and feed fish stocks is not always effective. Here, we only consider the case where biological interactions are absent (which we can extend to the case where they are moderate), in order to focus on the role of technology.

Proposition 3 *Absent biological interactions,*

(i) Long term stocks, efforts and prices in the edible fish sector and the feed fish sector evolve in opposite directions according to k. As for catches, their evolution depends on the initial state of the fisheries (heavily exploited or not).

(ii) When the wild fish stocks are heavily exploited in the initial steady state, the edible fish stock and catch rise with k at the expense of the feed fish stock and catch, while the effort and the price decrease in the first sector and increase in the second one. The production of farmed fish increases, and its price decreases. As the consumptions of the two edible fish

increase, consumer utility increases as well.

Proof. See Appendix D1. □

The first part of the proposition states that a technological shock in the aquaculture sector will have opposite effects on the two wild fisheries, as far as stocks, efforts and prices are concerned. Intuitively, we could expect that as the aquaculture sector becomes more efficient, the production of farmed fish would increase, its price decrease, and that a substitution effect would induce a decrease of the demand for wild fish, as farmed and wild fish are strong substitutes. But wild fisheries are in open access, and thus not managed on a profit-maximizing basis. Moreover, these fisheries may be heavily exploited, in the sense that stocks are under half their carrying capacity. So the previous intuition, valid for standard man-made goods, may prove very wrong in our case. In fact, we cannot even be sure that farmed fish production will increase and wild fish production decrease in response to a technological shock to aquaculture efficiency.

In the second part of the proposition, we consider an initial steady state where both stocks are heavily exploited, in the sense that they are . This assumption is convenient since it allows us to obtain clear analytical results. It is also quite relevant, given the state of world fisheries. In this case, the improvement of the aquaculture efficiency is favourable to the edible fishery at the expense of the feed fishery. Effort decrease in the edible fishery while it increases in the feed fishery. Indeed, fishing feed fish becomes more attractive, as a same quantity may be transformed into more farmed fish. Production increases in the edible fishery due to the smaller effort exerted by fishermen in a situation of initial biological inefficiency. Aquaculture's production also increases, in spite of the decrease of the catch of feed fish, because of the improvement of the aquaculture efficiency. Utility increases as well.

1.6 Extension: Endogenous consumer tastes

We now endogeneize consumer tastes. We consider that the wild fish product is a highly valued carnivorous species, while the farmed fish can be of any type between a carnivorous species analogous to the wild one to an omnivorous species. Our assumption is that consumer

preferences depend on this attribute. A change in k may affect either the weight of the farmed species in the utility function, or the elasticity of substitution between wild and farmed fish, or both.

In equation (1.1), α is the weight of farmed fish in preferences, and σ is the elasticity of substitution between the wild and the farmed species. We suppose that either α or σ may be a function of k. In the first case, $\alpha(k) \in]0, 0.5]$, meaning that consumers never weight farmed fish more than wild fish, and $\alpha'(k) < 0$, meaning that among farmed species consumers prefer the carnivorous ones. In the second case, we suppose that the lower k the higher σ, meaning that consumers perceive as highly substitutable wild and farmed fish having the same carnivorous diet, but that as the properties of the flesh differs, wild and farmed fish become less substitutable.

The following proposition sums up the consequences of these assumptions.

Proposition 4 *Absent biological interactions, and when consumer preferences depend on k, the effects of an improvement in aquaculture efficiency stated in Proposition 3 are completely reversed, if the weight affected to farmed fish or the elasticity of substitution between wild and farmed fish becomes sufficiently low as the farmed fish becomes less carnivorous.*

Proof. See Appendix D2. □

Absent any effect of k on consumer preferences, the weight affected by consumers to each product –wild and farmed– in utility is invariant, and the elasticity of substitution between wild and farmed fish as well. The effect of k is simply a productivity effect: the higher k, the more efficient the aquaculture technology. Now, when consumer preferences depend on k (through the weight α or the elasticity of substitution σ), a preference effect adds to the productivity effect: increasing k means not only having a more efficient aquaculture technology but also breeding fish that consumers like less, or that are less substitutable to wild fish. As k increases, the preference effect may progressively dominate the productivity effect. Thereby, when the farmed species is very carnivorous and the wild fish stocks are heavily exploited, increasing k i.e. choosing to breed a less carnivorous species benefits to the edible fish stock, at the expense of the other stock. Indeed, increasing the substitute

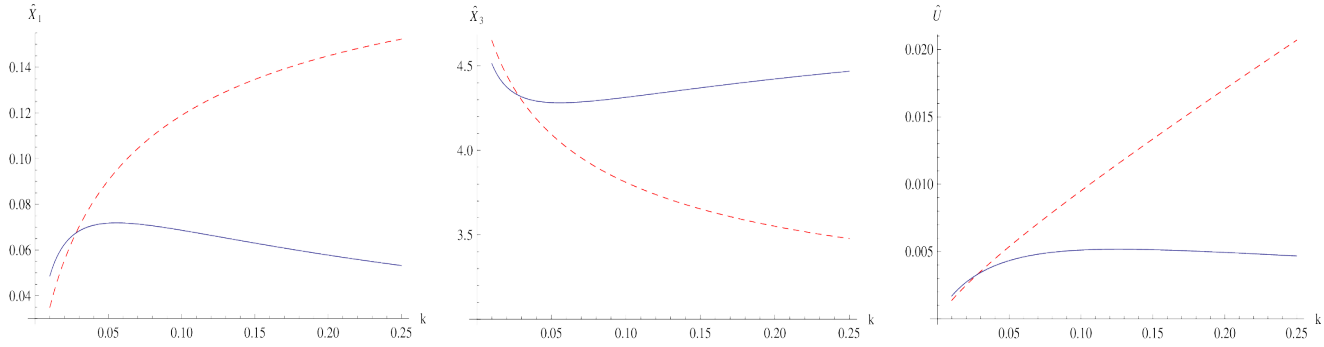
availability releases pressure on the wild edible fish stock. But increasing k too much reverses the process.

As a consequence, our conjecture is that there exists a utility-maximizing farmed species type. Consumers being sensitive to the properties of the flesh consumed, it is no use for the aquaculture sector of producing a less carnivorous species, else consumers will be trapped between a highly valued wild product, whose consumption is limited, and a cheap farmed fish they dislike.

We verify numerically that this situation may actually happen, in the case where the weight affected to the farmed fish in the utility function depends on k. We use the following specification:

$$\alpha(k) = \frac{0.5\alpha_{\min}}{\alpha_{\min} + (0.5 - \alpha_{\min})k}, \quad 0 < \alpha_{\min} < 0.5 \tag{1.37}$$

with $\alpha_{\min}$ the minimum weight affected to the farmed fish. Numerical simulations are performed with the same parameters as in Table 1.1, except that $d_1 = d_3 = 0$ and $\alpha_{\min} < 0.05$. Figure 1.3 shows the two wild fish stocks and utility as functions of k.



Edible fish stock | Feed fish stock | Consumer utility

Figure 1.3: Long run effects of an improvement of aquaculture's efficiency, absent biological interactions (dashed curves: productivity effect alone; plain curves: productivity and preference effects)

1.7 Conclusion

Many hopes are placed on aquaculture. This production technology is expected to bring more food security by increasing or at least maintaining the current per capita level of fish protein

given population growth, and to alleviate fishing pressure on wild edible fish stocks. This article analyzes the impact of aquaculture on wild fish stocks and on consumer welfare. By means of a two-species Lotka-Volterra model for biological interactions and a simple modeling of the aquaculture technology, we provide some answers to these issues. We find that under the condition of coexistence of aquaculture and the edible fishery, which relates to income, the coupling of all three sectors yields a unique steady state. Nonetheless, while in absence of biological interactions the equilibrium is always stable, it is not necessarily the case when introducing species interdependencies. Indeed, stability is conditional on the intensity of biological interactions and on the income level.

Actually, most of our results regarding the impact of aquaculture entry are conditional on the degree of dependence of the wild edible species on the feed fish stock. When this interaction is moderate, by increasing global fish supply, aquaculture decreases the price of the wild product, thus, fishing effort decreases allowing the edible stock to recover despite the fact that aquaculture exploits the prey species as a production input. In fact, there even exists a range of income levels for which the introduction of aquaculture prevents the wild edible species of collapsing. On the other hand, the feed fish stock is always lower as one can expect. Otherwise, it is not obvious analytically whether total wild fish consumption increases in all event, yet, this low interactions scenario benefits to consumers whose utility is always increased.

In the case where biological interactions are high, aquaculture leads to a decline in the feed fish stock and the wild edible fish stock, a decrease in wild edible fish supply and an increase in its price. Indeed, when assuming a strong dependence of the predator species on the feed fish population, aquaculture threatens the edible species by altering its food web. We find that for high levels of income, aquaculture actually provokes the collapse of the wild edible fishery though it would have remain alone. The net effect of farming on total fish consumption and welfare is ambiguous. We show through numerical simulations that the introduction of aquaculture may decrease utility. Such result can be explained by the fact that the fisheries are supposed to be in open access, meaning that the exploitation of resources is economically inefficient. In this situation, the introduction of aquaculture adds an extra inefficiency which may lead to a decreased utility, in spite of the fact that more

consumption options are offered to consumers.

Aquaculture is often criticized on the basis that it is a very inefficient production process. FIFO ratios remain high, even if they have dramatically decreased over the last 15 years, and it seems desirable to lower them further to produce more from a limited input. We study the effects of efficiency gains in the aquaculture sector, and show that when wild fish stocks are initially heavily exploited, they lead to an increase in the production of wild and farmed fish and, consequently, to an increase in utility. Nevertheless, IFFO believes that aquaculture can continue to decrease its use of fish oil but that there are physiological limits preventing from going below a certain limit depending on the species. According to the OECD a potential answer to the limited supply of feed would be to split the salmon market for instance into a cheap market fed with reduced fish oil and an expensive one fed with real Omega 3 fish oil. Differentiating supply through two lines of quality could better match consumers willingness to pay for food fish and better ensure the prosperity of the industry. Another option to enhance the sustainability of aquaculture would be to direct consumer preferences towards less carnivorous species by means of marketing strategies or educative campaigns.

Lastly, we emphasize the influence of consumer preferences. Following empirical evidence, we suppose that preferences are carnivorous species-biased, and we link this characteristics of preferences to the efficiency of aquaculture: the more efficient aquaculture is, the less carnivorous is farmed fish and the less consumers like it, or the less substituable to wild fish it is. This leads to a trade-off in the choice of the farmed species, and we suspect that there will exist a farmed species diet that maximizes utility. Indeed, it realizes the optimal balance between quantity available and expected flesh properties.

Beyond the limited supply of feed, other factors are expected to slow aquaculture growth such as land scarcity, stricter regulations or consumer awareness of the sector ecological impacts. Consumer concern for environment or health may affect their behaviour towards farmed products. More evidence on how consumers perceive farmed fish could shade light on the perspectives of food fish production processes.

Bibliography

[1] Alaska Departement of Fish & Game. 2010. Alaska Fishery Exvessel Prices.

[2] Alder, J. Campbell, B. Karpouzi, V. Kaschner, K. Pauly, D. 2008. Forage fish: From ecosystems to markets. *Annual Review of Environment and Resources*, 53, 153–166.

[3] Anderson, J. L. 1985. Market interactions between aquaculture and the common-property commercial fishery. *Marine Resource Economics*, 2(1), 1–24.

[4] Asche, F., Bjorndal, T., Young, J. 2001. Market interactions for aquaculture products. *Aquaculture Economics & Management*, 5(5-6), 303–318.

[5] Asche, F. and Bjorndal, T. 2011. *The Economics of Salmon Aquaculture*. Wiley-Blackwell, second edition.

[6] Costello, C., Ovando, D., Hilborn, R., Gaines, S. D., Deschenes, O. and Lester, S. E. 2012. Status and Solutions for the World's Unassessed Fisheries. *Science* 338, 517–520.

[7] FAO Fisheries and Aquaculture Departement. 2012. *The State of World Fisheries and Aquaculture*.

[8] FAO. 2011. *Aquaculture development. 6. Use of wild fishery resources for capture based aquaculture*. FAO Technical Guidelines for Responsible Fisheries. No. 5, Suppl. 6.

[9] Fernández-Polanco, J. 2010. Consumers knowledge and preferences about fish farmed species. The Fifteenth Biennial Conference of the International Institute of Fisheries Economics & Trade (IIFET), Markets and marketing of seafood products, MA01.

[10] Fishmeal Information Network (FIN). 2011. *Fishmeal and fish oil facts and figures.*

[11] FranceAgriMer. 2012. *Consommation des produits de la pêche et de l'aquaculture−Données statistiques 2011.*

[12] Gantmacher, F.R. 1959. *The Theory of Matrices*. Chelsea Publishing Company, New York.

[13] Grainger, R. J. R. and Garcia, S. M. 1996. Chronicles of marine fishery landings: Trend analysis and fisheries potential. FAO Fisheries Technical Paper nÂ°359.

[14] Gordon, H. S. 1954. Economic theory of a common-property resource: the fishery. *Journal of Political Economy*, 62, 124–142.

[15] Hannesson, R. 1983. Optimal harvesting of ecologically interdependent fish species. *Journal of Environmental Economics and Management*, 10(4), 329–345.

[16] Hannesson, R. 2002. Aquaculture and fisheries. *Marine Policy*, 27, 169–178.

[17] Hofbauer, J. and Sigmund, K. 1998. Evolutionary Games and Population Dynamics. *Cambridge University Press*.

[18] International Fishmeal and Fish Oil Organisation (IFFO). 2011. The Facts, Trends and IFFO's Responsible Supply Standard.

[19] IFFO Position Statement, Version 1. 2012. How many kilos of feed fish does it take to produce one kilo of farmed fish, via fishmeal and fish oil in feed?

[20] Kasulo, V., Perrings, C. 2006. Fishing down the value chain: Biodiversity and access regimes in freshwater sheries –the case of Malawi. *Ecological Economics*, 59(1), 106–114.

[21] Knapp, G., Roheim, C. A., and Anderson, J. L. 2007. The Great Salmon Run: Competition Between Wild and Farmed Salmon. TRAFFIC North America. Washington D.C.: World Wildlife Fund.

[22] Kristofersson, D. and Anderson, J.L. 2006. Is there a relationship between fisheries and farming? Interdependence of fisheries, animal production and aquaculture. *Marine Policy*, 30(6), 721–725.

[23] Natale, F., Hofherr, J., Gianluca, F. and Virtanen, J. 2013. Interactions between aquaculture and fisheries. *Marine Policy*, 38, 205–213.

[24] Naylor, R. L., Goldburg, R. J., Primavera, J. H., Kautsky, N., Beveridge, M.C.M., Clay, J., Folke, C., Lubchenco, J., Mooney, H., Troell, M. 2000. Effect of aquaculture on world fish supplies. *Nature*, 405, 1017–1024.

[25] Naylor, R. L., Hardy, R. W., Bureau, D. P., Chiu, A., Elliott, M., Farrell, A. P., Foster, I., Gatlin, D. M., Goldburg, R. J., Hua, K. and Nichols, P. D. 2009. Feeding aquaculture in an era of finite resources. *PNAS*, 106, 15103–15110.

[26] Nielsen, M., Setala, J., Laitinen, J., Saarni, K., Virtanen J. and Honkanen, A. 2007. Market integration of farmed trout in Germany. *Marine Resource Economics*, 22, 195–213.

[27] Olsen, L. R. and Hasan M. R. 2012. A limited supply of fishmeal: Impact on future increases in global aquaculture production. *Trends in Food Science & Technology,* 1–9.

[28] Quaas, M. F. and Requate, T. 2013. Sushi or fish fingers? Seafood diversity, collapsing fish stocks, and multispecies fishery management. *Journal of Environmental Economics and Management*, 115(2), 381–422.

[29] Schaefer, M. B., 1957. Some considerations of population dynamics and economics in relation to the management of the commercial marine fisheries. *Journal de l'Office des recherches sur les pêcheries du Canada*, 1957, 14(5), 669–681.

[30] Shamshak, G. and Anderson, J. 2008. Offshore aquaculture in the United States: Economic considerations, implications and opportunities. NOAA (Pre-Publication Copy), July, chapter 4: 73–96.

[31] Sivert, W. and Smith, W. R. 1977. Optimal exploitation of multispecies community. *Mathematical Biosciences*, 33: 121–134.

[32] Smith, A.D.M., Brown, C.J., Bulman, C.M., Fulton, E.A., Johnson, P., Kaplan, I.C., Lozano-Montes, H., Mackinson, S., Marzloff, M., Shannon, L.J., Shin, Y.-J. and Tam, J. 2011. Impacts of Fishing Low–Trophic Level Species on Marine Ecosystems. *Science* 333, 1147–1150.

[33] Tacon A. G.J. and Metian, M. 2008. Global overview on the use of fish meal and fish oil in industrially compounded aquafeeds: Trends and future prospects. *Aquaculture*, 285: 146–158.

[34] Ye, Y. and Beddington, J. R. 1996. Bioeconomic interactions between the capture fishery and aquaculture. *Marine Resource Economics*, 11, 105–123.

Appendix A

A.1 Proof of Proposition 1, (i) and (iii)

Plugging the expressions of stationary stocks and efforts given by (1.31)–(1.34) into (1.35) yields:

$$\frac{q_1\left(\gamma q_3\right)^{-\gamma}}{k\left(\frac{\alpha}{1-\alpha}\right)^{\frac{\sigma}{\sigma-1}}}\left(\frac{I}{c}\right)^{1-\gamma}\left(\frac{\widehat{A}}{1-\widehat{A}}\right)^{\frac{1}{\sigma-1}}\widehat{A}\frac{b_3}{b_1b_3+d_1d_3}\left(x_1+y_1\widehat{A}\right)=\left(-\widehat{A}\frac{d_3}{b_1b_3+d_1d_3}\left(x_3+y_3\widehat{A}\right)\right)^{\gamma} \tag{A.1}$$

Figures A.1 and A.2 portray the two members of this equation, in the different cases that may occur, depending on the value of the parameters. The left-hand side member is denoted $f(\widehat{A})$ and the right-hand side member $g(\widehat{A})$.

Since $d_3 > 0$, $x_3 < 0$ and $y_3 > 0$ (see (1.31) and (1.33)), the condition of existence of $g(\widehat{A})$ is $x_3 + y_3\widehat{A} \leq 0$ i.e. $\widehat{A} \leq -\frac{x_3}{y_3}$. $g(\widehat{A})$ is then a positive inverted U-shaped function, with $g(0) = g\left(-\frac{x_3}{y_3}\right) = 0$, and, since $\gamma < 1$ and $-d_3x_3 > 0$, $g'(0) = +\infty$.

As for the $f(.)$ function, we have

$$f'(\widehat{A}) = \frac{q_1\left(\gamma q_3\right)^{-\gamma}}{k\left(\frac{\alpha}{1-\alpha}\right)^{\frac{\sigma}{\sigma-1}}}\left(\frac{I}{c}\right)^{1-\gamma}\left(\frac{\widehat{A}}{1-\widehat{A}}\right)^{\frac{1}{\sigma-1}}\frac{b_3}{b_1b_3+d_1d_3}\left[\frac{1}{\sigma-1}\frac{1}{1-\widehat{A}}\left(x_1+y_1\widehat{A}\right)+x_1+2y_1\widehat{A}\right]$$

hence

$$\lim_{\widehat{A}\to 0} f'(\widehat{A}) = 0^+ \text{ if } x_1 > 0,\ 0^- \text{ if } x_1 < 0$$

$$\lim_{\widehat{A}\to 1} f'(\widehat{A}) = +\infty \text{ if } x_1 + y_1 > 0,\ -\infty \text{ if } x_1 + y_1 < 0$$

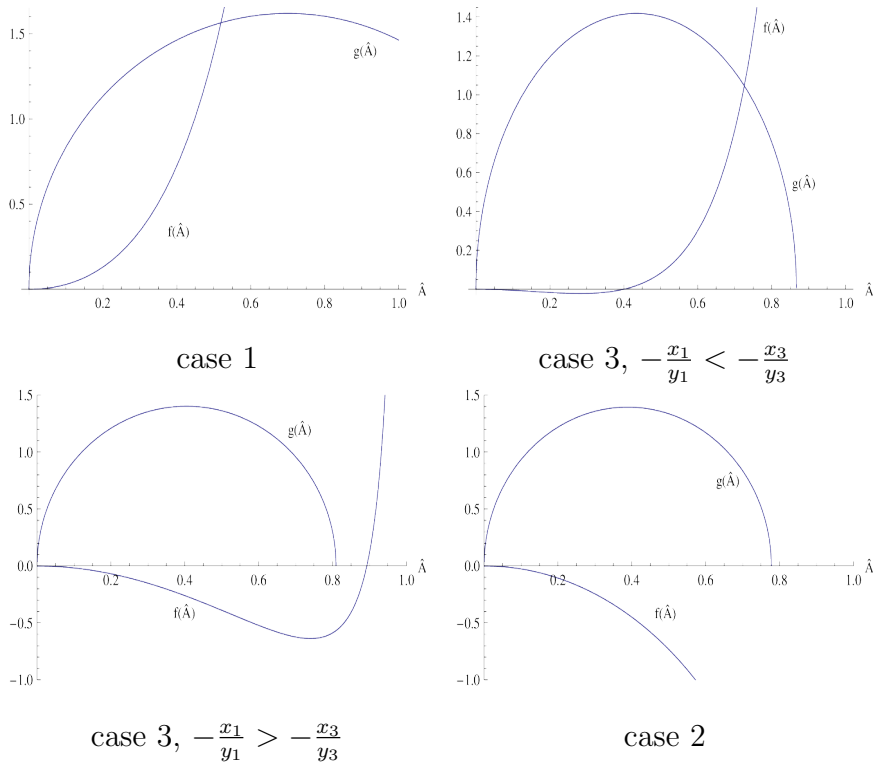


Figure A.1: Existence and uniqueness of an interior solution when $d_1 < \overline{d}_1$

- *Case 1.* When $x_1 > 0$ and $x_1 + y_1 > 0$, $f(\widehat{A})$ is a positive function, increasing from 0 to $+\infty$ when $\widehat{A}$ increases from 0 to 1. The solution to equation $f(\widehat{A}) = g(\widehat{A})$ exists and is unique.

- *Case 2.* When $x_1 < 0$ and $x_1 + y_1 < 0$, $f(\widehat{A})$ is a negative function, decreasing from 0 to $-\infty$ when $\widehat{A}$ increases from 0 to 1. There exists no solution to equation $f(\widehat{A}) = g(\widehat{A})$.

- *Case 3.* When $x_1 < 0$ and $x_1 + y_1 > 0$, which requires $y_1 > 0$, $f(\widehat{A})$ is first decreasing and then increasing, has two roots 0 and $-\frac{x_1}{y_1} > 0$, and tends to $+\infty$ when $\widehat{A}$ tends to 1. In this case, the solution to equation $f(\widehat{A}) = g(\widehat{A})$ exists and is unique iff $-\frac{x_1}{y_1} \leq -\frac{x_3}{y_3}$, and there is no solution if $-\frac{x_1}{y_1} > -\frac{x_3}{y_3}$. Notice that when $-\frac{x_1}{y_1} \nearrow -\frac{x_3}{y_3}$, $\widehat{A} \nearrow -\frac{x_3}{y_3}$ and $\widehat{X}_1, \widehat{X}_3 \to 0$.

- *Case 4.* When $x_1 > 0$ and $x_1 + y_1 < 0$, which requires $y_1 < 0$, $f(\widehat{A})$ is first increasing and then decreasing, has two roots 0 and $-\frac{x_1}{y_1} > 0$, and tends to $-\infty$ when $\widehat{A}$ tends to 1. In this case, there may exist two, one or no solution to equation $f(\widehat{A}) = g(\widehat{A})$. When

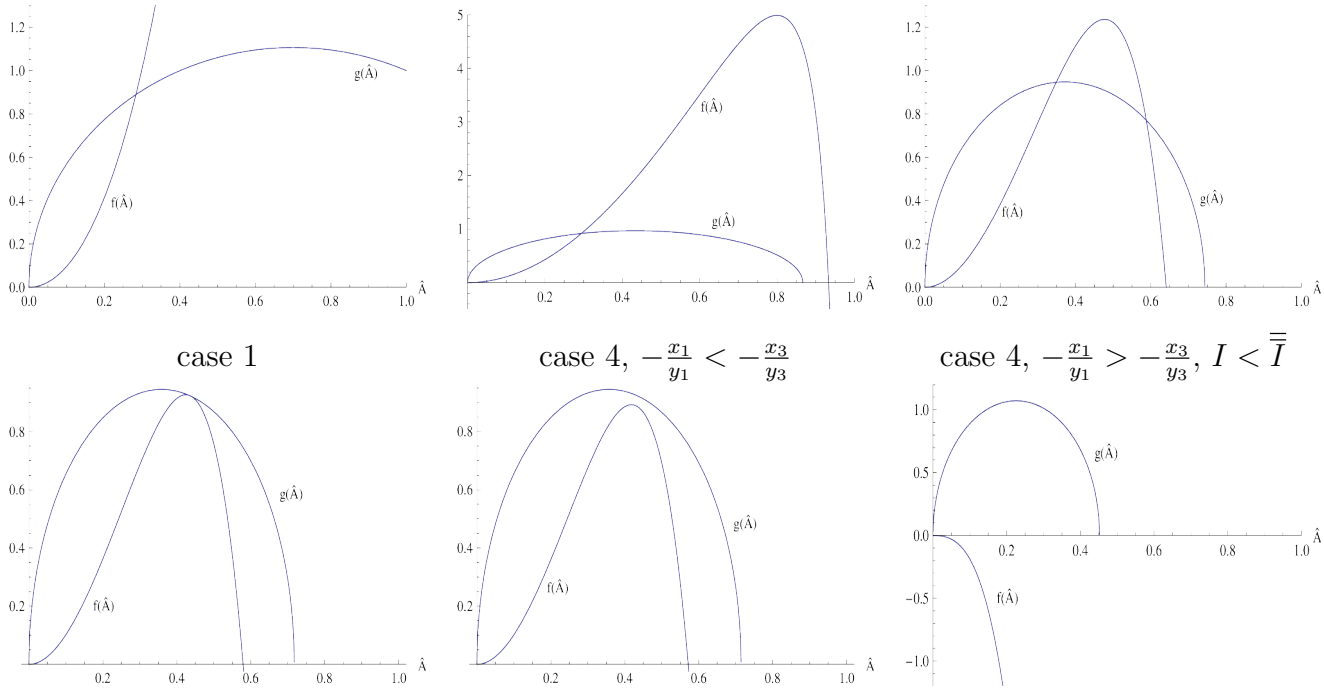


Figure A.2: Existence and uniqueness of an interior solution when $d_1 > \overline{d}_1$

$-\frac{x_1}{y_1} > -\frac{x_3}{y_3}$ the solution is unique. When $-\frac{x_1}{y_1} \searrow -\frac{x_3}{y_3}$ there are 2 solutions, a strictly positive one $(\widehat{X}_1, \widehat{X}_3 > 0)$ and a second one characterized by $\widehat{X}_1, \widehat{X}_3 \to 0$.

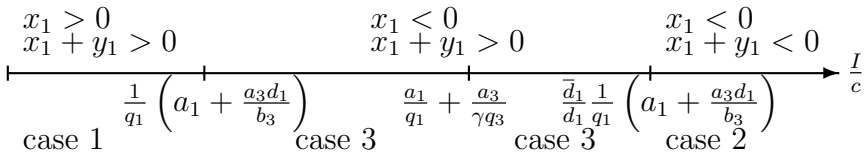
We have

$$x_1 = a_1 + \frac{a_3 d_1}{b_3} - q_1 \frac{I}{c}$$
$$x_1 + y_1 = a_1 + \frac{a_3 d_1}{b_3} - \frac{d_1}{b_3} \gamma q_3 \frac{I}{c}$$
$$\frac{x_1}{y_1} - \frac{x_3}{y_3} = \frac{b_1 b_3 + d_1 d_3}{b_3 d_3 y_1 y_3} q_1 \gamma q_3 \frac{I}{c} \left(\frac{a_1}{q_1} + \frac{a_3}{\gamma q_3} - \frac{I}{c} \right)$$

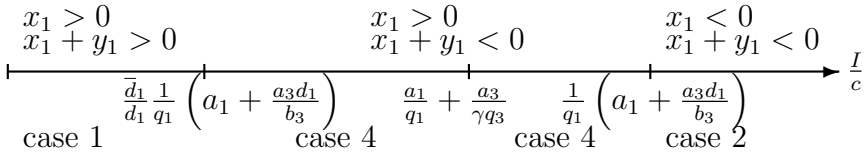
Notice that $\frac{x_1}{y_1} - \frac{x_3}{y_3}$ has the same sign as $\frac{1}{y_1} \left(\frac{a_1}{q_1} + \frac{a_3}{\gamma q_3} - \frac{I}{c} \right)$.

Define $\overline{d}_1 = b_3 \frac{q_1}{\gamma q_3}$. Simple computations based on the previous observations allow us to obtain the results summarized below and portrayed on Figure A.6:

Case $d_1 < \overline{d}_1$



Case $d_1 > \overline{d}_1$



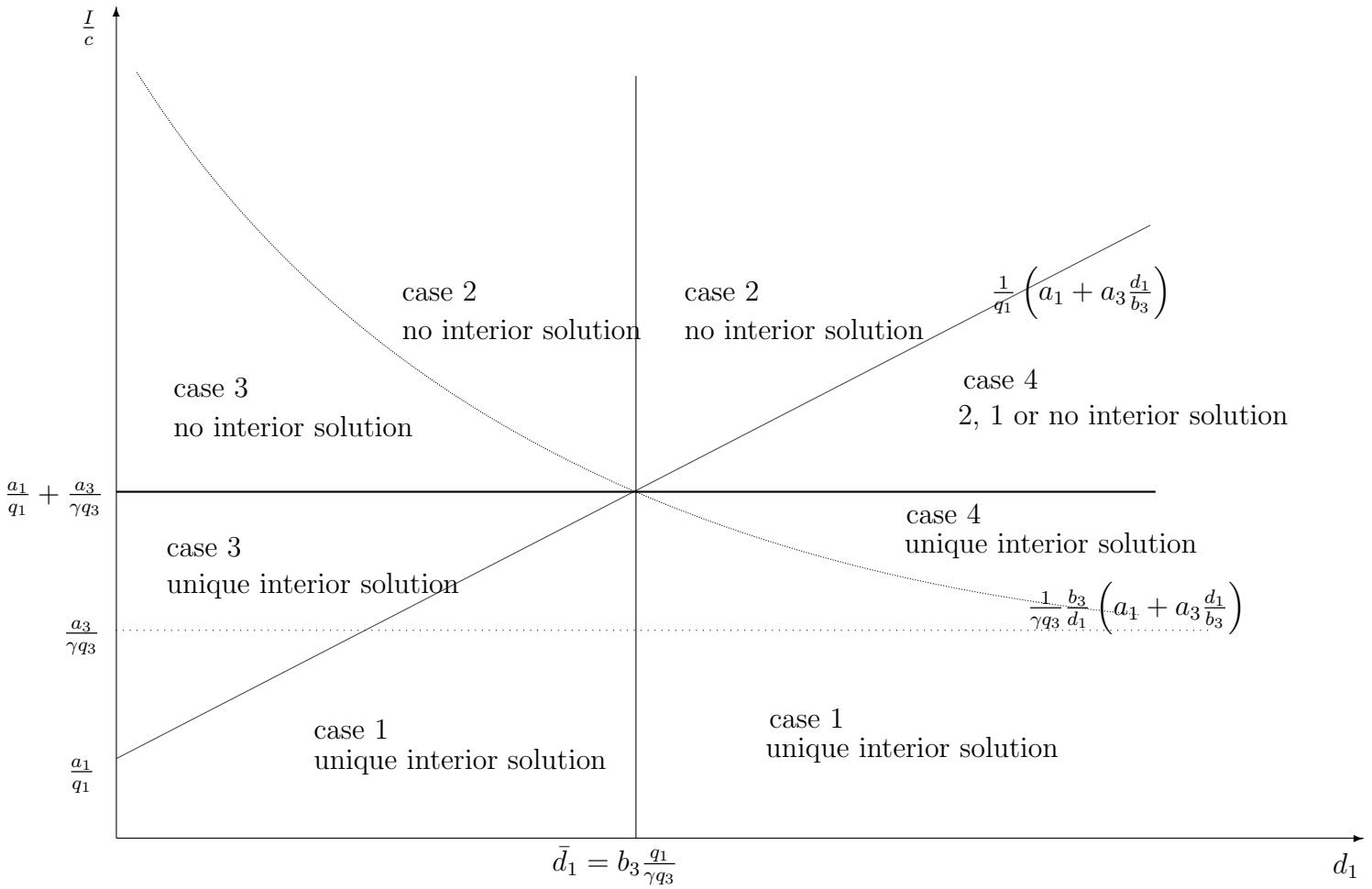


Figure A.6: Interior solutions

This proves points (i) and (iii) of Proposition 1.

A.2 Stability

A.2.1 No fishing

The linearization of the dynamic system (1.7)–(1.8) around a steady state yields the following Jacobian matrix:

$$\widetilde{J} = \begin{pmatrix} a_1 - 2b_1\widetilde{X}_1 + d_1\widetilde{X}_3 & d_1\widetilde{X}_1 \\ -d_3\widetilde{X}_3 & a_3 - 2b_3\widetilde{X}_3 - d_3\widetilde{X}_1 \end{pmatrix}$$

It immediately appears that steady states 1 and 2 are unstable: the eigenvalues evaluated at these steady states are respectively $a_1 > 0$, $a_3 > 0$ and $a_1 + \frac{a_3 d_1}{b_3} > 0$, $-a_3 < 0$. As for steady state 3, the eigenvalues are $-a_1 < 0$, $a_3 - \frac{a_1 d_3}{b_1}$. Hence steady state 3 is asymptotically stable iff $a_3 - \frac{a_1 d_3}{b_1} < 0 \iff \frac{a_1}{b_1} > \frac{a_3}{d_3}$. Notice that this condition is the opposite of condition (1.9) of existence of steady state 4. Finally, for steady state 4 we have:

$$\det \widetilde{J} = b_1 b_3 + d_1 d_3 > 0$$
$$\mathrm{tr}\widetilde{J} = -b_1 - b_3 < 0$$

Therefore the two roots of the characteristic equation are either real and negative or complex with a negative real part, depending on the sign of the discriminant, that reads: $(b_1 - b_3)^2 - 4d_1 d_3$. Steady state 4 is a stable node in the first case, a stable focus in the second one. Notice that the first case occurs when biological interactions are mild ($d_1 d_3$ small), and vice versa.

A.2.2 Capture fishery alone

The Jacobian matrix of the dynamic system (1.15) linearized around a steady state is:

$$J^* = \begin{pmatrix} \frac{\partial F_1(X_1,X_3)}{\partial X_1}\Big|_{X_1^*,X_3^*} - q_1 E_1^* & -q_1 X_1^* & \frac{\partial F_1(X_1,X_3)}{\partial X_3}\Big|_{X_1^*,X_3^*} \\ 0 & -\beta c & 0 \\ \frac{\partial F_3(X_1,X_3)}{\partial X_1}\Big|_{X_1^*,X_3^*} & 0 & \frac{\partial F_3(X_1,X_3)}{\partial X_3}\Big|_{X_1^*,X_3^*} \end{pmatrix}$$
$$= \begin{pmatrix} a_1 - 2b_1 X_1^* + d_1 X_3^* - q_1 E_1^* & -q_1 X_1^* & d_1 X_1^* \\ 0 & -\beta c & 0 \\ -d_3 X_3^* & 0 & a_3 - 2b_3 X_3^* - d_3 X_1^* \end{pmatrix}$$

As it is the case without human intervention, steady states 1 and 2, characterized by the extinction of both wild species or the extinction of species 1, the predatory edible wild fish, are unstable: the eigenvalues evaluated at these steady states are respectively $-\beta c < 0$, $a_1 > 0$, $a_3 > 0$ (steady state 1 is globally unstable) and $-\beta c < 0$, $a_1 + \frac{a_3 d_1}{b_3} > 0$, $-a_3 < 0$ (steady state 2 is a saddle point). As for steady state 3, characterized by the extinction of species 3, the eigenvalues are $-\beta c < 0$, $-a_1 + q_1 \frac{I}{c} < 0$, $a_3 - \frac{d_3}{b_1}\left(a_1 - q_1 \frac{I}{c}\right) > 0$ according to condition (1.9). Hence steady state 3 is also unstable. Finally, for steady state 4 the characteristic equation reads:

$$(-\beta c - \lambda)\left(\lambda^2 + (b_1 X_1^* + b_3 X_3^*)\lambda + (b_1 b_3 + d_1 d_3) X_1^* X_3^*\right) = 0$$

It admits 3 roots: $-\beta c < 0$ and are 2 other roots, either real and negative or complex with a negative real part, depending on the sign of the discriminant, that reads: $(b_1 X_1^* - b_3 X_3^*)^2 - 4 d_1 d_3 X_1^* X_3^*$. Hence steady state 4 is asymptotically stable, and is a stable node in the first case, a stable focus in the second one.

A.2.3 Capture fishery and aquaculture: proof of Proposition 1, (ii)

The linearization of the dynamic system (1.30) in the neighborhood of the steady state yields the following Jacobian matrix:

$$J = \begin{pmatrix} \frac{\partial F_1(X_1,X_3)}{\partial X_1}\Big|_{\widehat{X}_1,\widehat{X}_3} - q_1 \widehat{E}_1 & -q_1 \widehat{X}_1 & \frac{\partial F_1(X_1,X_3)}{\partial X_3}\Big|_{\widehat{X}_1,\widehat{X}_3} & 0 \\ a_{21} & a_{22} - \beta c & -a_{23} & -a_{24} \\ \frac{\partial F_3(X_1,X_3)}{\partial X_1}\Big|_{\widehat{X}_1,\widehat{X}_3} & 0 & \frac{\partial F_3(X_1,X_3)}{\partial X_3}\Big|_{\widehat{X}_1,\widehat{X}_3} - q_3 \widehat{E}_3 & -q_3 \widehat{X}_3 \\ -a_{21} & -a_{22} & a_{23} & a_{24} - \beta c \end{pmatrix}$$

with

$$\begin{cases} a_{21} = \beta I \frac{\sigma-1}{\sigma} \frac{\widehat{A}(1-\widehat{A})}{\widehat{X}_1} \\ a_{22} = \beta I \frac{\sigma-1}{\sigma} \frac{\widehat{A}(1-\widehat{A})}{\widehat{E}_1} = \beta c \frac{\sigma-1}{\sigma} \widehat{A} \\ a_{23} = \beta I \frac{\sigma-1}{\sigma} \gamma \frac{\widehat{A}(1-\widehat{A})}{\widehat{X}_3} \\ a_{24} = \beta I \frac{\sigma-1}{\sigma} \gamma \frac{\widehat{A}(1-\widehat{A})}{\widehat{E}_3} = \beta c \frac{\sigma-1}{\sigma} (1 - \widehat{A}) \end{cases}$$

and

$$\begin{aligned}
\left.\frac{\partial F_1(X_1,X_3)}{\partial X_1}\right|_{\widehat{X}_1,\widehat{X}_3} - q_1\widehat{E}_1 &= \left.\frac{\partial F_1(X_1,X_3)}{\partial X_1}\right|_{\widehat{X}_1,\widehat{X}_3} - \frac{F_1(\widehat{X}_1,\widehat{X}_3)}{\widehat{X}_1} = -b_1\widehat{X}_1 \\
\left.\frac{\partial F_1(X_1,X_3)}{\partial X_3}\right|_{\widehat{X}_1,\widehat{X}_3} &= d_1\widehat{X}_1 \\
\left.\frac{\partial F_3(X_1,X_3)}{\partial X_1}\right|_{\widehat{X}_1,\widehat{X}_3} &= -d_3\widehat{X}_3 \\
\left.\frac{\partial F_3(X_1,X_3)}{\partial X_3}\right|_{\widehat{X}_1,\widehat{X}_3} - q_3\widehat{E}_3 &= \left.\frac{\partial F_3(X_1,X_3)}{\partial X_3}\right|_{\widehat{X}_1,\widehat{X}_3} - \frac{F_3(\widehat{X}_1,\widehat{X}_3)}{\widehat{X}_3} = -b_3\widehat{X}_3
\end{aligned}$$

Tedious computations show that the characteristic polynomial reads:

$$P(\lambda) = (\beta c + \lambda)\, Q(\lambda)$$

with

$$Q(\lambda) = \mu_3\lambda^3 + \mu_2\lambda^2 + \mu_1\lambda + \mu_0$$

and

$$\begin{cases}
\mu_3 = 1 \\
\mu_2 = \frac{\beta c}{\sigma} + b_1\widehat{X}_1 + b_3\widehat{X}_3 \\
\mu_1 = \frac{\beta c}{\sigma}\left(b_1\widehat{X}_1 + b_3\widehat{X}_3\right) + (b_1b_3 + d_1d_3)\,\widehat{X}_1\widehat{X}_3 + \frac{\sigma-1}{\sigma}\beta I\widehat{A}(1-\widehat{A})\,(q_1 + \gamma q_3) \\
\mu_0 = \frac{\beta c}{\sigma}\,(b_1b_3 + d_1d_3)\,\widehat{X}_1\widehat{X}_3 + \frac{\sigma-1}{\sigma}\beta I\widehat{A}(1-\widehat{A})\left(b_1\gamma q_3\widehat{X}_1 + b_3q_1\widehat{X}_3 + d_3\gamma q_1\widehat{X}_1 - d_1q_3\widehat{X}_3\right)
\end{cases}$$

$P(\lambda)$ admits one negative real root equal to $-\beta c$, plus the 3 roots of $Q(\lambda)$. We apply the Routh-Hurwitz criterion to $Q(\lambda)$. Clearly, $\mu_3 > 0$, $\mu_2 > 0$ and $\mu_1 > 0$. The sign of μ_0 is ambiguous. Besides, the sign of $\mu_2\mu_1 - \mu_3\mu_0$ is also ambiguous:

$$\begin{aligned}
\mu_2\mu_1 - \mu_3\mu_0 &= \left(\frac{\beta c}{\sigma} + b_1\widehat{X}_1 + b_3\widehat{X}_3\right)\left(\frac{\beta c}{\sigma}\left(b_1\widehat{X}_1 + b_3\widehat{X}_3\right) + (b_1b_3 + d_1d_3)\,\widehat{X}_1\widehat{X}_3 + \frac{\sigma-1}{\sigma}\beta I\widehat{A}(1-\widehat{A})\,(q_1 + \gamma\right. \\
&\quad \left. - \frac{\beta c}{\sigma}\,(b_1b_3 + d_1d_3)\,\widehat{X}_1\widehat{X}_3 - \frac{\sigma-1}{\sigma}\beta I\widehat{A}(1-\widehat{A})\left(b_1\gamma q_3\widehat{X}_1 + b_3q_1\widehat{X}_3 + d_3\gamma q_1\widehat{X}_1 - d_1q_3\widehat{X}_3\right)\right. \\
&= \left(b_1\widehat{X}_1 + b_3\widehat{X}_3\right)\left[\left(\frac{\beta c}{\sigma}\right)^2 + \frac{\beta c}{\sigma}\left(b_1\widehat{X}_1 + b_3\widehat{X}_3\right) + (b_1b_3 + d_1d_3)\,\widehat{X}_1\widehat{X}_3\right] \\
&\quad + \frac{\sigma-1}{\sigma}\beta I\widehat{A}(1-\widehat{A})\left[\frac{\beta c}{\sigma}\,(q_1 + \gamma q_3) + q_1\,(b_1 - \gamma d_3)\,\widehat{X}_1 + q_3(d_1 + \gamma b_3)\widehat{X}_3\right]
\end{aligned}$$

Nevertheless, we can obtain the following results.

- Absent biological interactions ($d_1 = d_3 = 0$) we have $\mu_0 > 0$ and $\mu_2\mu_1 - \mu_3\mu_0 > 0$. In this case, the linearized dynamic system is stable (see Gantmacher, 1959). This remains true as long as biological interactions are not too strong. More precisely, a sufficient condition for $\mu_0 > 0$ is $d_1 \leq b_3\frac{q_1}{q_3}$, and a sufficient condition for $\mu_2\mu_1 - \mu_3\mu_0 > 0$ is $d_3 \leq \frac{b_1}{\gamma}$.

- When the revenue spent on fish I tends to 0, $\widehat{X}_1 \to \widetilde{X}_1$, $\widehat{X}_3 \to \widetilde{X}_3$, $\mu_0 \to \frac{\beta c}{\sigma}\left(b_1b_3 + d_1d_3\right)\widetilde{X}_1\widetilde{X}_3 > 0$ and $\mu_2\mu_1 - \mu_3\mu_0 \to \left(b_1\widetilde{X}_1 + b_3\widetilde{X}_3\right)\left[\left(\frac{\beta c}{\sigma}\right)^2 + \frac{\beta c}{\sigma}\left(b_1\widetilde{X}_1 + b_3\widetilde{X}_3\right) + \left(b_1b_3 + d_1d_3\right)\widetilde{X}_1\widetilde{X}_3\right] > 0$ and the system is stable.

A.3 Proof of Proposition 2

(i) Eliminating $\widehat{A}$ between equations (1.32) and (1.34) yields a relationship between the two long run effort levels:

$$\widehat{E}_1 + \frac{\widehat{E}_3}{\gamma} = \frac{I}{c}$$

Remember that absent aquaculture the optimal level of effort in the capture fishery is $E_1^* = I/c$. Then obviously $\widehat{E}_1 + \widehat{E}_3 < E_1^*$.

(ii) Now, comparing $\widehat{X}_1$ (equation (1.31)) to the stock of the baseline case without aquaculture X_1^* (equation (1.16)), we get:

$$\widehat{X}_1 = X_1^* + \frac{b_3}{b_1b_3 + d_1d_3}y_1\widehat{A}$$

Hence

$$\widehat{X}_1 > X_1^* \Longleftrightarrow y_1 > 0 \Longleftrightarrow d_1 < \overline{d}_1 = b_3\frac{q_1}{\gamma q_3}$$

The steady state expressions of the wild edible fish price are $P_1^* = c/(q_1X_1^*)$ and $\widehat{P}_1 = c/(q_1\widehat{X}_1)$, hence the result. Concerning the wild feed fish stock, comparing $\widehat{X}_3$ (equation (1.33)) and X_3^* (equation (1.17)) yields:

$$\widehat{X}_3 = X_3^* - \frac{d_3}{b_1b_3 + d_1d_3}y_3\widehat{A} < X_3^* \quad \text{since } d_3 > 0 \text{ and } y_3 > 0$$

(iii) When aquaculture lowers the wild edible fish stock ($d_1 > \overline{d}_1$), as $\widehat{E}_1 < E_1^*$, the supply of wild edible fish is necessarily lower: $\widehat{Y}_1 < Y_1^*$. It may even be the case that the introduction

of aquaculture causes the collapse of the wild fish stock in the long run: when $\overline{I} < I < I_w(d_1)$ and no interior steady state exists or there exist two unstable interior steady state, the wild fishery alone would have been sustainable.

Now, when aquaculture leads to an increased wild edible stock ($d_1 < \overline{d}_1$), its impact on wild fish supply is ambiguous, except in the particular case where the introduction of aquaculture prevents the edible fish stock from collapsing, that is when $I_w(d_1) < I < \overline{I}$. Likewise, when aquaculture increases wild fish supply, it obviously increases also total fish supply, whereas when aquaculture decreases wild fish supply the net effect of aquaculture on total fish supply is ambiguous.

(iv) Turning to the comparison of utilities, we obtain:

$$\left(\frac{U(\widehat{Y}_1,\widehat{Y}_2)}{U(Y_1^*,0)}\right)^{1-\frac{1}{\sigma}} = \left(\frac{\widehat{Y}_1}{Y_1^*}\right)^{1-\frac{1}{\sigma}}\left(1+\frac{\alpha(k)}{1-\alpha(k)}\left(\frac{\widehat{Y}_2}{\widehat{Y}_1}\right)^{1-\frac{1}{\sigma}}\right)$$

$$= \left(\frac{\widehat{Y}_1}{Y_1^*}\right)^{1-\frac{1}{\sigma}}\frac{1}{1-\widehat{A}} = \left(\frac{\widehat{E}_1\widehat{X}_1}{E_1^*X_1^*}\right)^{1-\frac{1}{\sigma}}\frac{1}{1-\widehat{A}} = \left(\frac{\widehat{X}_1}{X_1^*}\right)^{1-\frac{1}{\sigma}}\left(1-\widehat{A}\right)^{-\frac{1}{\sigma}}$$

$$U(\widehat{Y}_1,\widehat{Y}_2) > U(Y_1^*,0) \Longleftrightarrow \left(\frac{\widehat{X}_1}{X_1^*}\right)^{1-\frac{1}{\sigma}} > \left(1-\widehat{A}\right)^{\frac{1}{\sigma}} \Longleftrightarrow \frac{\widehat{X}_1}{X_1^*} > \left(1-\widehat{A}\right)^{\frac{1}{\sigma-1}}$$

This condition is always satisfied when $\widehat{X}_1 \geq X_1^*$ i.e. when $d_1 \leq \overline{d}_1$, which is a sufficient condition for aquaculture to increase welfare. We exhibit numerically a case where the introduction of aquaculture leads to a decrease of utility.

A.4 Proof of Propositions 3 and 4

A.4.1 Proof of Proposition 3

From system (1.31)–(1.34) (with $d_1 = d_3 = 0$) we get for stocks and efforts:

$$d\widehat{X}_1 = \tfrac{q_1}{b_1}\tfrac{I}{c}d\widehat{A}, \qquad d\widehat{E}_1 = -\tfrac{I}{c}d\widehat{A}$$
$$d\widehat{X}_3 = -\tfrac{\gamma q_3}{b_3}\tfrac{I}{c}d\widehat{A}, \quad d\widehat{E}_3 = \gamma\tfrac{I}{c}d\widehat{A}$$

Hence $d\widehat{X}_1$ and $d\widehat{X}_3$ are always of opposite signs, as well as $d\widehat{E}_1$ and $d\widehat{E}_3$.

As for catches and prices, we obtain:

$$\begin{array}{ll}
\frac{d\widehat{Y}_1}{\widehat{Y}_1} = \frac{d\widehat{E}_1}{\widehat{E}_1} + \frac{d\widehat{X}_1}{\widehat{X}_1} = \left(\frac{a_1/b_1}{\widehat{X}_1} - 2\right)\frac{d\widehat{A}}{1-\widehat{A}}, & \frac{d\widehat{P}_1}{\widehat{P}_1} = -\frac{d\widehat{A}}{1-\widehat{A}} - \frac{d\widehat{Y}_1}{\widehat{Y}_1} = \left(1 - \frac{a_1/b_1}{\widehat{X}_1}\right)\frac{d\widehat{A}}{1-\widehat{A}} \\
\frac{d\widehat{Y}_3}{\widehat{Y}_3} = \frac{d\widehat{E}_3}{\widehat{E}_3} + \frac{d\widehat{X}_3}{\widehat{X}_3} = \left(2 - \frac{a_3/b_3}{\widehat{X}_3}\right)\frac{d\widehat{A}}{\widehat{A}}, & \frac{d\widehat{P}_3}{\widehat{P}_3} = \frac{d\widehat{A}}{\widehat{A}} - \frac{d\widehat{Y}_3}{\widehat{Y}_3} = \left(\frac{a_3/b_3}{\widehat{X}_3} - 1\right)\frac{d\widehat{A}}{\widehat{A}} \\
\frac{d\widehat{Y}_2}{\widehat{Y}_2} = \frac{dk}{k} + \gamma\frac{d\widehat{Y}_3}{\widehat{Y}_3}, & \frac{d\widehat{P}_2}{\widehat{P}_2} = \frac{d\widehat{A}}{\widehat{A}} - \frac{d\widehat{Y}_2}{\widehat{Y}_2}
\end{array}$$

Absent biological interactions, a_1/b_1 (resp. a_3/b_3) is the carrying capacity of species 1 (resp. species 3). We thus have $1 - \frac{a_1/b_1}{\widehat{X}_1} < 0$ and $\frac{a_3/b_3}{\widehat{X}_3} - 1 > 0 : d\widehat{P}_1$ and $d\widehat{P}_3$ are always of opposite signs. For catches, things depend on the initial value of the stock with respect to half its carrying capacity. This proves (i).

We have expressed so far how our variables evolve according to a variation of $\widehat{A}$, the market interaction variable. We must now determine how $\widehat{A}$ itself evolves according to a variation of k, the efficiency of aquaculture.

Equation (1.35) defining $\widehat{A}$ can be also written as:

$$\frac{\widehat{A}}{1-\widehat{A}} = \frac{\alpha}{1-\alpha}\left(\frac{\widehat{Y}_2}{\widehat{Y}_1}\right)^{\frac{\sigma-1}{\sigma}}$$

Totally differentiating this equation, we obtain:

$$\frac{d\widehat{A}}{\widehat{A}\left(1-\widehat{A}\right)} = \frac{\sigma-1}{\sigma}\left[\frac{d\widehat{Y}_2}{\widehat{Y}_2} - \frac{d\widehat{Y}_1}{\widehat{Y}_1}\right]$$

from which we deduce:

$$\frac{d\widehat{Y}_2}{\widehat{Y}_2} = \frac{d\widehat{A}}{\widehat{A}\left(1-\widehat{A}\right)} + \frac{\sigma}{\sigma-1}\frac{d\widehat{Y}_1}{\widehat{Y}_1}$$

As we have shown above that for $\widehat{X}_1 \leq \frac{a_1/b_1}{2}$, $d\widehat{Y}_1$ and $d\widehat{A}$ have the same sign, this equation shows that it is also the case for $d\widehat{Y}_2$.

Finally, we can deduce from this equation, by replacing $d\widehat{Y}_1$ and $d\widehat{Y}_2$ $d\widehat{A}$ by their expression as a function of $d\widehat{A}$, that:

$$\left[\frac{1}{1-\widehat{A}} - \frac{\sigma-1}{\sigma}\left(\gamma\left(2 - \frac{a_3/b_3}{\widehat{X}_3}\right) + \frac{\widehat{A}}{1-\widehat{A}}\left(2 - \frac{a_1/b_1}{\widehat{X}_1}\right)\right)\right]\frac{d\widehat{A}}{\widehat{A}} = \frac{\sigma-1}{\sigma}\frac{dk}{k}$$

For $\widehat{X}_1 \leq \frac{a_1/b_1}{2}$ and $\widehat{X}_3 \leq \frac{a_3/b_3}{2}$, the term between brackets on the left-hand side of this

equation is unambiguously positive. Then $d\widehat{A}/dk > 0$. It immediately follows that:

$$\begin{array}{llll} \frac{d\widehat{X}_1}{dk} > 0, & \frac{d\widehat{E}_1}{dk} < 0, & \frac{d\widehat{X}_3}{dk} < 0, & \frac{d\widehat{E}_3}{dk} > 0 \\ \frac{d\widehat{Y}_1}{dk} > 0, & \frac{d\widehat{P}_1}{dk} < 0, & \frac{d\widehat{Y}_3}{dk} < 0, & \frac{d\widehat{P}_3}{dk} > 0 \\ \frac{d\widehat{Y}_2}{dk} > 0, & \frac{d\widehat{P}_2}{dk} < 0 & & \end{array}$$

This proves (ii).

A.4.2 Proof of Proposition 4

When preferences depend on k, either through the weight α of farmed fish in utility or through the elasticity of substitution σ between wild and farmed fish, the previous equation becomes:

$$\left[\frac{1}{1-\widehat{A}} - \frac{\sigma-1}{\sigma}\left(\gamma\left(2-\frac{a_3/b_3}{\widehat{X}_3}\right)+\frac{\widehat{A}}{1-\widehat{A}}\left(2-\frac{a_1/b_1}{\widehat{X}_1}\right)\right)\right]\frac{d\widehat{A}}{\widehat{A}} = \left[1-\frac{1}{\sigma(k)}\left(1-\frac{k\sigma'(k)}{\sigma(k)}\right)+\frac{k\alpha'(k)}{\alpha(k)(1-\alpha(k)}\right.$$

As $\alpha'(k) < 0$ and $\sigma'(k) < 0$, it may be the case that the right-hand side of the above equation is negative. More precisely, there may exist a threshold for the parameter k above which $d\widehat{A}/dk$ changes sign.

Chapter 2

Competition between farmed and wild fish: the French sea bass and sea bream markets

2.1 Introduction

Worldwide, an important share of marine fish stocks are under threat. Worm et al. (2009) report that 63% of assessed fish stocks worldwide require rebuilding. FAO (2012) reports that about 57.4% of world marine fish stocks are estimated as fully exploited and 29.9% as overexploited. While marine fish production is marked by a small decline since the early 1990s, aquaculture has been the fastest growing food industry since the early 1980s, with an annual average growth of 8.8% (FAO, 2012). In 2010, aquaculture provided nearly 50% of global seafood[1] production. "With the increasing contribution from aquaculture to seafood supply the interactions between fisheries and aquaculture are expected to become even more important and therefore deserve exhaustive investigations from socio-economic and biological perspectives"" (Natale *et al.,* 2013, p.205). These interactions are especially important at the food market where the competition between wild and farmed fish species affects fish

[1]To be understood as finfish and shellfish production.

price dynamics.[2] The international trade in fish products also plays a significant role in the evolution of fish prices, given that fish and fishery products are among the most traded food commodities worldwide (World Trade Organization, WTO[3]). Among agricultural, food, animal and fish products, fish has been one of the fastest growing category between 1990 and 2000, namely an increase of 52% in trade value (Anderson, 2003). These observations led us to investigate whether the expansion of the global supply of farmed fish has affected the price of certain wild species consumed in France.

The information on market interactions of farmed and wild fish give insights on the extent to which farmed fish prices are likely to affect wild fish prices. These outcomes depend on how consumers perceive wild versus farmed products, but also on the relative cost competitiveness and production quantities of each sector. For instance, farmed fish species regarded as low-value products relatively to their wild counterpart might lead to segmented markets, where wild products occupy the high-value segment. This option entails a maintained pressure on wild edible resources. On the other hand, if a farmed fish is at least as much valued than its wild counterpart, the prosperity of the capture fishery sector will rely on its relative price competitiveness.

At the same time, the valuation of farmed fish is likely to influence the technological evolution of the aquaculture sector and affect the manner in which wild fish resources are managed. Indeed, the aquaculture sector strives to find solutions to the limited availability fish meal and fish oil[4], which are required inputs to sustain the flesh properties of several farmed species. At date, it does not seem to exist a protein source displaying required properties and profitable at the same time. However, consumers depreciating the modification of fish diet and the subsequent change in flesh nutritive quality, gives incentives to the aquaculture industry to work out a suitable substitute to natural fish populations. The

[2]The interactions between fisheries and aquaculture also exist at the biological level as marine fisheries provide inputs to aquaculture production, that is fish meal and fish oil. The study of this type of interactions is out of the scope of our study.

[3]http://www.wto.org/english/res_e/reser_e/ersd201003_e.htm.

[4]Fish meal and fish oil are made from small oily fish belonging to low trophic levels. Farmers purchase fish meal and fish oil in the form of compounded feed, which are pellets providing nutrients and different supplements to farmed fish.

aquaculture supply may also split into two branch: a high quality fish supply fed, with fish meal and fish oil, and a lower one, relying on cheaper and nutritionally poorer feed. Besides, if the willingness of consumers to pay for quality farmed fish is high, the management of fisheries resources used for feed, and that of species biologically depending on them, might be enhanced or revised worldwide. Lastly, the public response to farming practices and their environmental impact may also orientate the development of this production sector.

In short, the substitutability between farmed and wild fish is likely play a role in the future development of the structure of fish markets and on the production technologies of seafood. It this respect, more evidence on how consumers perceive fish from aquaculture is required, as facts and literature are not unanimous about the behavior of consumers regarding wild versus farmed products. Since the aquaculture takes place in confined area (ponds, pen nets, cages, raceways, ranching, recirculating systems, etc.), this production technology displays a high degree of control over the attributes of the grown fish. Consumer preferences may determine exact specification for color, size, texture, fat content or other parameters making the delivered product more convenient (Asche *et al.*, 2001). Along the same lines, Knapp *et al.* (2007) argue that farming has changed the timing of the supply which is no longer necessarily seasonal and that the increased availability of farmed salmon has lead to increased demand. Sustainability concerns may represent another reason for a shift of preference from wild fish to farmed fish. On the other hand, wild products are perceived as healthier, at times, more tasty, and aquaculture raises environmental concerns which provide some competitive advantages to wild products (Natale *et al.*, 2013).

A consumer perception study conducted in 2007 by the firm Via Aqua (for the French interprofessional office of marine and aquaculture products (Ofimer) and the French interprofessional comity of aquaculture products (Cipa)) joins some of these conclusions. While 5% of consumers favor farmed products, 50% are rather in favor of wild products, and 45% report they are indifferent. Positive aspects of farmed products mentioned by respondents are resource preservation, products freshness, availability and aspect. By contrast, consumers are rather sceptical regarding the healthiness of feeds, the species well-being and genuineness (FranceAgriMer, 2011).

Our aim is to examine if the production process of fish is a significant attribute in con-

sumers fish consumption choice, compared to the species type or other product attributes. If consumers operate a vertical differentiation between farmed and wild products, we question what is the ranking. To do so, we test for market integration between farmed and wild products in France, focusing on the case of sea bass and sea bream species, respectively. Market integration is an indicator that explains how much different markets are related to each other. Our empirical approach relies on the Law of One price (LOP) and the concept of cointegration which consists in testing for price parity between wild and farmed fish products. Our data set consists of domestic (monthly) price series, provided by Kantar WorldPanel[5], for a number of fish species consumed by households from 2007 to 2012 in France.

The importance of the sea bass and sea bream markets at European level, makes these species study subjects of interest. In Europe, sea bass and sea bream taken together represent the largest production of farmed species, after Atlantic salmon (FEAP, 2012). Aquaculture is in fact the main production method of these two species. The EU provides two thirds of their world aquaculture production, respectively, followed by Egypt in the case of farmed sea bass, and by Turkey in the case of sea bream. For these two species, intra-EU trade is substantial, Greece being the major exporter of sea bass towards Italy, the United Kingdom and France, and of sea bream towards Italy, Portugal and France. Except for sea bream imports from Turkey, trade between the EU and third countries remains limited.

Eventhough most sea bass comes from aquaculture, capture fisheries account for more than 10 % of the total sea bass production worldwide. The EU has a major role in that it provides 83% of the world captures of sea bass — France undertaking 60% of this capture — and 32% of the world captures of sea bream — France and Spain together undertaking 74% of this capture (EC, Maritime Affairs and Fisheries, 2012).

France is of particular interest to analyse market interactions between farmed and wild sea bass and sea bream products, respectively. As mentioned above, for both species, it is the third intra-EU importer of farmed fish, while the extent of domestic catches enables national consumption of wild and farmed fish to be sustained for both species. As detailed further on, in recent years, about 50% of French consumption of sea bass was farmed while that of sea bream was around 70%.

[5]Kantar WorldPanel: www.kantarworldpanel.com.

Market interactions between fisheries and aquaculture have been investigated from a theoretical and empirical point of view. The findings of this research indicate that the effects of aquaculture development on fish stock and welfare depend on the degree of substitution between wild and farmed species. Anderson (1985) shows that given a fishery under open access and exploited beyond the maximum sustainable yield, the entry of aquaculture increases total fish supply, and thus, reduces pressure on the wild fish stock through a lower fish consumer price. Similarly, Ye and Beddington (1996) find positive social benefits of aquaculture entry, but to a lesser extent when considering wild and farmed fish as imperfect substitutes with positive cross-price elasticities. Valderrama and Anderson (2010) show how a limited-entry fishery regulation successfully enables rents extraction relatively to open access, but then the decrease in fish price resulting from the increased supply of farmed products dissipates these rents.

There is a considerable amount of fish-specific and country-specific studies that question which fish attributes influences consumers' choice. Many concern salmon, and often use the cointegration concept to analyze market integration between different species, different geographic area, or to test whether consumer distinguish between fish products origin, form, or production process (Gordon et al., 1993; Asche et al., 1999; Clayton and Gordon, 1999; Asche, 2000; Jaffry *et al*). Focusing on the production process attribute, several studies report a highly integrated market for wild and farmed salmon products (Asche *et al.*, 2001; Asche *et al.*, 2005; Knapp *et al.*, 2007). Regarding salmon origin, Asche and Sebulonsen, (1998) find that that it does not seem to influence consumers choices to any extent in the in France and the UK salmon markets. Regarding product from, Asche *et al.* (1997) use an almost ideal demand system to estimate the demand for fresh Atlantic salmon, frozen Atlantic salmon and frozen Pacific salmon in the European Union. All goods are found to be substitutes, indicating one market for all three product forms.

Other studies focus on white fish[6] markets. Nielsen (2005) finds evidence of a partially-integrated European white fish market and a perfectly spatially integrated cod market. The

[6]White fishes are demersal fish with fins. Unlike oily fish, white fish contains oils only in their liver, rather than in their gut, and can therefore be gutted as soon as they are caught, on board the ship. White fish have dry and white flesh.

empirical results provided by Asche *et al.* (2002) also confirm the existence of an integrated EU white fish market. The existence of an integrated market was also demonstrated for some other fish species (Nielsen *et al.*, 2007; Norman-Lopez, 2009). Salmon does not in general compete, however, with big volume whitefish species. The evidence on market integration seems to indicate that "farmed species competes mainly with the same wild species (and other species in the same segment), but not with other species"(Asche *et al.*, 2001, p.311).

As concerns sea bass and sea bream, very little empirical analysis has been conducted in the literature (Asche *et al.*, 2001), despite of the considerable trade flow they generate in the EU. To the best of our knowledge, there are only two studies which investigate market integration between wild and farmed sea bass and sea bream, respectively. Asche and Steen (1998) find that sea bass and sea bream "may compete with portion trout and several white fish species" within the EU" (Asche *et al.*, 2001, p. 311). Brigante and Lem (2001) found no evidence of price link between farmed and wild species of sea bass and sea bream in the Italian market. Unlike them, we find that in France, a partial market integration exists between fresh whole wild and farmed sea bream, while this is not the case fresh for whole sea bass.

The present chapter contributes to the literature on the measure of market interactions between wild and farmed products on the sea bass and sea bream markets, respectively. We examine the market interactions between wild and farmed products on the sea bass and sea bream French markets, respectively. We also question how farmed and wild fish interact across these species. Finally, we check whether the form of the product (whole versus cut) influences on the outcomes of the market integration analysis. We carry out our empirical analysis by means of a bivariate cointegration framework. We also conduct the test of LOP for these fish species. The market integration of farmed and wild products should give insights on the extent to which farmed product prices are likely to affect wild product prices, and in turn, marine stock status of concerned wild species and fishermen activity. These outcomes depend on how consumers perceive wild versus farmed products, but also on the relative cost competitiveness and production quantities of each sector.

The remaining of the chapter is organized as follows. In the following section, we describe the main characteristics of the French sea bass and sea bream markets. Section 3 presents

the data set and descriptive statistics. Section 4 exposes our econometric methodology. Section 5 provides the estimation results. Section 6 discuses the outcomes of our empirical analysis. Finally, section 6 offers concluding remarks on the potential competition between fresh farmed and wild products in the sea bass and sea bream French markets.

2.2 The sea bass and sea bream French markets

Sea bass is a demersal white fish which essentially originates from Eastern Atlantic, Mediterranean Sea, and Black Sea in European markets. France, UK, Italy, Turkey and Egypt have significant sea bass fisheries. Sea bream is also a demersal white fish found in the Atlantic and Mediterranean, but commercial harvests are small.

Table 2.1 and 2.2 infers from trade flows, the yearly French consumption of sea bass and sea bream, in volume, between 2008 and 2011. The figures displayed account for all type/from of fish product (frozen, canned, filleted, whole, ect.) and all varieties of sea bass and sea bream species. Over this time period, the average total French consumption of sea bass is worth 10,157.6 tons against 13,017 tons for sea bream. It appears that French catches of sea bass represented about 50.5% of national consumption, against 36.4% for sea bream.

In France aquaculture production is not expected to increase due to land ownership and legal issues restricting availability of sites. Both species are increasingly supplied by aquaculture operations, mostly in the Mediterranean Sea. The rapid expansion of the production of these species in the early 1990's is driven by the rise of aquaculture in Turkey and Greece. Juveniles are produced on an increasingly large scale in hatcheries, and availability is no longer a constraint on the industry.

Although France is a much smaller market for bass and bream than Spain and Italy, as domestic production is limited, any consumption rise has to be covered by imports (FAO Globefish, 2009).

Within the fresh finfish *household* consumption — excluding the catering sector — in 2011, sea bass is the 7^{th} most consumed species in value (63,547 k€[7]) and the 6^{th} in volume

[7]Value of total household purchases of sea bass in France in 2011

(4,563 tons[8]), while sea bream is 9^{th} in value (51,145 k€) and 7^{th} in volume (4,534 tons), the two leading species in value and volume being salmon and cod. These latest figures combine wild and farmed fish consumption.

Sea bass (*in tons*)	**2008**	**2009**	**2010**	**2011**
Wild production	3279	5566	6129	5535
Farmed production	4163	2877	2337	2452
Imports	4525.3	4906.5	5507.1	5626.2
Exports	-3869.7	-2974.3	-2831.1	-2597.5
Consumption	8097.6	10375.2	11142	11015.7
Data source: Eurostat.				

Table 2.1: French national consumption of sea bass in tons

Sea bream (*in tons*)	**2008**	**2009**	**2010**	**2011**
Wild production	2795	5285	5725	5126
Farmed production	1182	1278	1239	1412
Imports	8340.2	9474.5	9055.4	8461.9
Exports	-1784	-1973.6	-1925.8	-1622.2
Consumption	10533.2	14063.9	14093.6	13377.7
Data source: Eurostat.				

Table 2.2: French national consumption of sea bream in tons

According to these figures and to FranceAgriMer (2011), about[9] 50% of French consumption of sea bass is farmed while that of sea bream is around 70%. There actually exists several varieties of sea bass and sea bream species. The scientific name of the only farmed sea bass species is *Dicentrarchus labrax*, while that for the main farmed sea bream species is *Sparus aurata*. Otherwise, figures for wild production in Table 2.1 and 2.2 account for all other harvested varieties labeled as *sea bass* or *sea bream* in the fish market.

[8]Volume of total household purchases of sea bass in France in 2011

[9]It was not possible to obtain the share of wild and farmed fish within the national imports and export figures of each species, though we presume they essentially consist of farmed fish.

2.3 The data set

Our data set is provided by Kantar WorldPanel[10]. It reports the aggregated monthly home consumption of fresh fish by a sample of French households, so as aggregated prices. The time series data cover the period January 2007 to September 2012 (69 points). The appointment *fresh products* includes products sold in large retailers, supermarkets, fishmongers, marketplaces and groceries. We do not study other product forms (i.e. transformed products, frozen or canned fish consumption) as Kantar WorldPanel cannot provide the distinction farmed/wild for these product types.

Fresh products constitute 33.3% of household seafood expenditures (FranceAgriMer, 2011). Since farmed fish are mostly sold in fresh products forms (EC, Maritime Affairs and Fisheries, 2012), this data set represents a good base for investigating market integration between farmed and wild fish species. Moreover, commercial catches of sea bass and sea bream, which are the focus species of our study, are mainly sold fresh as well.

For both species, sea bass and sea bream, we have information on fish-specific quantity consumed (in kg), fish-specific price (euros/kg), production mode (farmed/wild) and form (whole/cut).

To be more specific, the scientific names of the sea bass and sea bream varieties included in our study are given below, so as their main capture zones (source: FishBase). The sea bass species type we address are *Dicentrarchus labrax* (Eastern Atlantic, Mediterranean and Black Sea) and *Anarhichas lupus* (Northeast Atlantic, Northwest Atlantic, Baltic Sea and Northwestern Mediterranean). Sea bream price series concern: *Sparus aurata* (Northeast Atlantic, northern Mediterranean and Black Sea), *Spondyliosoma cantharus*(Eastern Atlantic, Mediterranean and the Black Sea), *Pagellus bogaraveo* (Eastern Atlantic and Western Mediterranean), *Coryphaena hippurus* (Atlantic, Indian and Pacific), *Sebastes mentella* (Western and Eastern Atlantic), *Sebastes marinus* (Western and Eastern Atlantic) and *Lithognathus mormyrus* (Eastern Atlantic, Mediterranean, Western Indian Ocean). Price figures are aggregate within each species, to form a unique sea bass and sea bream price time series.

[10]Kantar WorldPanel: www.kantarworldpanel.com.

The French household sample surveyed by Kantar WorldPanel is weekly queried and comprises 20,000 households assumed to be representative of the French population (according to demographic criteria defined by INSEE). Within the sample, the yearly replacement ratio of respondent is worth 25%. About 12,000 of the surveyed households declare purchases of products without bar codes; that is, fresh fish products. It is this data subset on which our study focuses. Notice that only species for which at least 2% of households are buyers are considered as significant and transmitted by Kantar WorldPanel. Kantar surveys are based on voluntary declarations of households. Therefore, Kantar WorldPanel is not exhaustive in measuring aquatic products purchases by all French households. All fish and fish products taken together, this panel is estimated to relate 80% of the total French seafood consumption (FranceAgriMer, 2011).

The data set relates that on average, over the period January 2007 to September 2012, 32.6% of fresh sea bass and 42% of fresh sea bream consumed by households in France is farmed. Between 2007 and 2012, we observe a shift in farmed sea bass consumption share from 27.13% to 36.5%, while for sea bream this share has remained steady around 40%. Since available time series concern household consumption of fresh sea bass and sea bream, it is normal that these figures do not match with those of total national consumption i.e. 50% for farmed sea bass and 70% for farmed sea bream. Yet, the difference is not negligible. Furthermore, as Kantar WorldPanel data set is established on the basis of households consumption recollection, for 20% of the fresh sea bass consumption reported, the distinction farmed/wild is unfilled against 30% for fresh sea bream. This does not represent, however, a limit to our empirical analysis as the methodology we apply requires only price information on the seafood products to test for market integration.

Table 2.3 reports detailed consumption statistics for Kantar WorldPanel sample of households in France.

We focus our analysis on the whole product form, rather than on the cut one, as our data set relates that 70.3% for fresh sea bass and 83.5% for fresh sea bream. These figures corroborate the 2004 report for the European Commission (Roth and Ukendt, 2004) on sea bass and sea bream markets which states that both species are almost universally sold as whole fish. We choose to compare products of identical form to ensure that the outcome of

	Sea bass		Sea bream	
	framed	**wild**	**farmed**	**wild**
whole	1012.6	1609.3	1410.7	1731.7
(%)	(83.2)	(64.1)	(89.7)	(79)
cut	204.6	900.7	161.3	458.4
(%)	(16.8)	(35.9)	(10.3)	()
Total	1217.2	2510	1572	2190.1
(%)	(100)	(100)	(100)	(21)

Quantities in tons.

Data source: Kantar WorldPanel.

Table 2.3: Mean yearly consumption of fresh sea bass and sea bream of Kantar WorldPanel sample of households in France: January 2007–September 2012

integration tests are not biased by this product attribute.

2.3.1 Descriptive statistics

Table 2.4 gives the average price of fresh sea bass and sea bream, by product form and production process, consumed between January 2007 and September 2012 by the French households included in our data set. To conduct a market delineation approach, price series have been deflated using the monthly consumer price index for food of the OECD (base year 2005). The overall price of sea bass is 12.55€/kg and 9.93€/kg for sea bream over our observation period. Hence, sea bass is a higher-valued species relatively to sea bream. Cut products are always more expensive as the price is increased by the cost of labor. An additional explanation is that the price per kg of whole fish is discounted by the weight of wastes (fishbone, skin, etc.).

Besides, wild products always display a higher economic value. Focusing on whole product form, the mean prices of wild sea bass is statistically higher than that of farmed sea bass, at the 5% level. The same holds for sea bream. In the case of sea bass, the price difference between wild and farmed products reaches 54% against 24% for sea bream. Both mean equality tests are reported in Appendix B.1, in Table B.1 and B.2. We also plotted the

	Sea bass		Sea bream	
	farmed	**wild**	**farmed**	**wild**
whole	9.28	14.24	8.29	10.3
	(0.13)	(0.18)	(0.12)	(0.15)
cut	13.18	14.29	11.26	13.9
	(0.33)	(0.22)	(0.31)	(0.19)
average	10.6	14.26	8.59	11.02

Prices in €/kg. Standard errors are shown in parentheses.

Data source: Kantar WorldPanel.

Table 2.4: Mean price of fresh sea bass and sea bream in France, Kantar WorldPanel household purchases: January 2007–September 2012

pairwise price series for which we test for market integration, in Figures B.1, B.2 and B.3.

In fact, except for salmon, who experienced a rather unique market story, empirical evidence is in favour of higher prices for wild rather than farmed products, at given species (FranceAgriMer, 2012). There are several explanations to the uncommon price dynamics of farmed/wild salmon. It is the aquaculture species produced at the largest scale worldwide. The strong market development of farmed salmon was made possible, inter alia, by a preexisting demand in several countries. Indeed, salmon was part of the consumption habits of many countries, specially wealthy ones. Thereafter, economies of scales realized in its production enabled to make it economically accessible to a larger population share. Lastly, the large fillets it delivers allows for salmon product differentiation and innovation, thus, enlarging the whole of consumers reached by this species. At date, the fact that at a given salmon species, farmed products may be more expensive than wild ones, reflects the argument by Knapp *et al.* (2007), that consumers get use to and favor a reliable and abundant supply, among other positive aspects of farmed products.

Regarding sea bass and sea bream, both species are less convenient it that their smaller fillets make marketing innovation less obvious. However, as mentioned earlier, both species supply coming to market have much increased in recent years. What is surprising, though, is that market prices are reflecting this large increase in supply very differently. Sea bream prices

have been falling drastically whereas sea bass prices have been quite stable (FAO, Globefish, 2009). Thus, some producers suffers from considerably reduced margins. According to FAO, over time, markets will certainly be able to absorb larger production volumes, but more stable equilibrium prices require more product innovation and the development of additional markets in the long run.

2.4 The theoretical framework

The empirical evidence that different species of fish are imperfect substitutes is interpreted in Quaas and Requate, (2013), as consumers having " preferences for diversity". This means that the utility of agents increase with the number of varieties consumed. The authors thus model stronger preferences for variety by a lower elasticity of substitution between different fish species.

This demand behaviour is found under monopolistic competition, where the horizontal/vertical differentiation of commodities provides market power to producing firms i.e. a firm may increase its price, all other prices kept constant, without loosing entire demand (Chamberlin, 1933; Robinson, 1933; Dixit-Stiglitz, 1977).

We question here how the production process attribute of fish — wild versus farmed — is received by consumers. As capture fisheries have long existed in absence of aquaculture, wild and farmed fish cannot be complementary goods. Yet, are they substitutes? If yes, are they imperfect or perfect substitutes ? If imperfect, what type of differentiation does the aquaculture and fisheries production technologies produce? Does not exists a natural ranking of consumer preferences between farmed and wild fish ?

Regarding price figures in 2.3.1, and more generally, there exists empirical evidence that for a given species the price of the wild product is higher than the price of the farmed one (FranceAgriMer, 2012). This observation rather suggests that the production process produces a vertical differentiation. That is, that at equal price, consumers unanimously prefer the same product. The mention farmed/wild would result in a quality distinction, at the benefit of wild products.

In any event, product differentiation results in consumers with a limited willingness to

substitute one for another. Microeconomic theory predicts that at the long-run equilibria, if a commodity price is kept constant, the introduction of a substitute leads either to the entry of new clients, or previous ones proceed to a re-allocation of their expenditure. Consequently, in the presence of competition, albeit unperfect, the price of the initial commodity should be decreased to maintain ist market share. If wild and farmed products are substitutes, capture fisheries will loose market share at constant price. Evidence on the market interactions between aquaculture and capture fisheries can help to improve the management of these sectors, i.e. enhance their economic viability and seafood supply.

To examine whether farmed and wild fish are substitutes, and to what extent, we apply the concept of cointegration as a market delineation method and test for the (relative) Law of One Price. The rationale for our empirical approach and the motivation for adopting it is exposed in the following section.

2.5 Empirical methodology

Stigler (1969) defines the market for a good as "the area within which the price of a good tends to uniformity, allowance being made for transportation costs". When the Law of One Price (LOP) holds, that is, when there exists no arbitrage opportunities between identical goods, markets are said to be perfectly integrated. If the price of identical goods differ by more than transportation costs in the long run, it is a sign of inefficient markets.

The literature has provided a number of contributions which apply the LOP test to assess whether prices significantly differ between distant areas or when crossing an international border. For instance, Chen and Knez (1995) develop two notions of integrated markets in the finance literature. First, perfectly integrated markets are consistent with the LOP: portfolios with similar payoffs should be assigned similar prices. Second, markets cannot be integrated if there are cross-market arbitrage opportunities. Broda and Weinstein (2008) compare the price of a vast number a product sharing a common barcode system between Canada and the USA. One of their main finding is that, using micro level data reveals much smaller deviation in the LOP, in response to borders and distance, than does aggregate data, as in Engel and Rogers (1996). Hence, the degree of market segmentation across the border

appears similar to that within borders.

When price are in logarithms, Broad and Weinstein (2008) refer to the relative LOP, by opposition to the absolute LOP defined by Stigler (1969). In this release, the LOP tests whether a shift in one good's price results in the same percentage variation of that of another, whereby the relative price is constant. Under the condition that variables are stationary, the relative LOP can be tested by running a simple OLS regression:

$$ln(P_{1t}) = B + Aln(P_{2t}) + \varepsilon_t, \tag{2.1}$$

where p_{it} is the price of good $i = 1, 2$, at time t. The coefficient A corresponds to a price elasticity between both goods, and B is a constant term. If $A = 0$, then no relationship exists between these two goods. If $A = 1$, then the relative LOP holds. In the reminder of the document, the acronym LOP will refer to this relative definition.

However, time series displaying stochastic evolutions may be incidentally correlated, leading to significant coefficients though prices of goods under scrutiny are not related. This phenomenon produces what is commonly called a *spurious regression*. Here below, we present different stationarity tests of time-series.

2.5.1 The integration order of a time series

Several tests exist to identify the order of integration of a time series, including the Augmented Dickey-Fuller (ADF) unit root test[11] and the Kwiatkowski-Phillips-Schmidt-Shin (KPSS) test[12], which we rely on in this study. The inferences from the KPSS test are complementary to those derived from the Dickey-Fuller distribution.

Consider two price variables, p_{1t} and p_{2t}, where lower caption letters refer to natural logarithms. The underlying model to the Augmented Dickey-Fuller test is:

$$\Delta p_t = \phi_1 p_{t-1} - \sum_{i=2}^{l} \phi_i \Delta p_{t-i+1} + \varepsilon_t \tag{2.2}$$

where a constant term c, and/or a time trend intercept t can be added to improve the explanatory power of the specification.

[11] The appropriate critical values applying to the Augmented Dickey-Fuller test are tabulated in Engle and Yoo (1987) and McKinnon (1991).

[12] Critical values for the KPSS test are taken from Kwiatkowski et al. (1992).

The null hypothesis is H0: $|\phi_1| = 0$. If it is accepted for one of these three models, the process is non-stationary. The parameter ϕ_1 is estimated applying the OLS method. The lagged terms $\sum_{i=2}^{l} \Delta\phi_i p_{t-i+1}$ aim at controlling for autocorrelation in the error term. To identify the order of integration of a time series, it must be differentiated as much as necessary for the null hypothesis to be accepted.[13]

The KPSS test is a Lagrange multiplier test of the null hypothesis of trend or level stationarity of the series. The LM statistic must be larger than the critical value for the null hypothesis to be rejected.[14]

2.5.2 The concept of cointegration as a market delineation approach

In the event that variables are non-stationary, a cointegration analysis must be used to investigate the relationship between time-series. The concept of cointegration enables to identify whether a stable long run relationship exists between prices, revealing the existence of a price parity condition. In which case, we can infer that markets display a level of integration.

Market integration refers to prices among different locations or related goods following similar patterns over a long period of time. Here, the concept of market is broader than a place where homogenous goods are traded. In a competitive market, the duration of a constant relative price relation shows that the matching of supply and demand have reached a stable long-term balance in examined subsectors. In which case, agents arbitrage between goods should resolve the short run price disparities. If a shock with lasting price effects occurs, such as a change in consumer preferences or in production costs of one sector, than one will detect a break in the cointegration relation. In this context, the evidence of a price parity condition between goods may reflects a degree of substitutability between them. It can also stem from a supply factor that has a spillover effect on several markets.

Our aim is to test for market integration between farmed and wild fish at the food market.

[13]The appropriate critical values to apply for the Augmented Dickey-Fuller test are those tabulated by Engle and Yoo (1987) or by McKinnon (1991).

[14]Critical values for the KPSS test are taken from Kwiatkowski, Phillips, Schmidt, and Shin (1992).

Since the structure of production costs between the aquaculture and capture fishery industry are quite different, we posit that the market integration between both fish products would stem from the demand behavior of consumers.

The notion of market integration can apply to any type of related markets and all levels of market interdependencies may be observed. As an example, in financial literature, Chen and Knez (1995) develop two notion of integrated market. Perfectly integrated markets are consistent with the LOP, i.e. portfolios with similar payoffs should be assigned close prices. Second, markets cannot be integrated if there are cross-market arbitrage opportunities.

The more usual method to assess the market relationship between different products is to estimate a demand equation or a system of demand equations and measure cross-price elasticities. However, its implementation is often limited by the unavailability of detailed income data, and quantities of all consumption forms of a commodity, required to define the structure of the demand system.

The cointegration approach to market delineation is convenient in the sense that only price series are required. The drawback is that this method relates less precise information on the relationship between markets. It may reveal whether two goods display constant relative price in the long run, but not the degree of substitutability between them. However, Asche *et al.* (1997) undertake cointegration test for market delineation and an estimation of a dynamic system of demand equation on the same data set, and find both approach provide compatible and complementary results.

In fisheries, appropriate quantity data are usually lacking (Asche *et al.*, 2001). This is why the literature has much used the information on price differentials between products to analyze market interactions between seafood goods. As mentioned in Section 2.3, since we do not dispose of reliable information on quantities nor on household attributes either, we adopt the cointegration approach to analyze market interactions between wild and farmed fish.

In the two-variables case, cointegrated time series display the same order of integration. Consider two price variables, p_{1t} and p_{2t}, where lower caption letters refer to natural logarithms. Consider variables p_{1t} and p_{2t}, which are $I(d)$ processes, with d the number of differentiation to operate for the variables to be stationary. Then, p_{1t} and p_{2t} are cointegrated

iff their linear combination:

$$z_t = p_{1t} - \beta p_{2t} \tag{2.3}$$

is $I(d-b)$, with $d \geq b \geq 0$. An order of integration of z_t inferior to that of the time series, means that the variables p_{1t} and p_{2t} display a common stationary dynamic. Their movements may diverge in the short run, but their long run relation is stable. The residual z_t measures the equilibrium error.[15]

Assuming p_{1t} and p_{2t} are $I(1)$, then the vector $\mathbf{p_t} = \begin{pmatrix} p_{1t} \\ p_{2t} \end{pmatrix}$ is said to be cointegrated if there exists a unique nonzero (2×1) vector $\boldsymbol{\beta}$, such that:

$$z_t = \boldsymbol{\beta\prime}\mathbf{p}_t \rightarrow I(0), \tag{2.4}$$

with $\boldsymbol{\beta\prime} = [1, -\beta]$ the cointegration vector.

In this bivariate context, testing for the LOP comes down to controlling whether the coefficients of the cointegration vector sum to zero, that is $\boldsymbol{\beta\prime} = [1, -1]$. This may be carried out using a likelihood-ratio test[16] of restrictions on parameters in $\boldsymbol{\beta}$. While cointegration allows to identify market boundaries, the LOP specifies market inter-dependencies (Nielsen, 2005). If a pairwise cointegration test reveals a common stationary trend and the test for the LOP is validated, then the goods relative price is constant and markets are perfectly integrated. When a pairwise cointegration test reveals a common stationary trend, but the test for the LOP is rejected, then markets are partially-integrated. Lastly, if no common stationary trend is found, goods do not belong to the same market.

[15] Stigler (1969) provides several potential reasons to short-run price deviations, among which, variations in the relative quality of goods or stochastic shocks to supply/demand. Stochastic shocks are generally limited in size and time by the intertemporal substitution behavior of sellers and buyers if goods belong to the same market.

[16] A likelihood-ratio test is used to compare the fit of two models, one of which is nested within the other. It is a general method of testing model assumptions.

2.5.3 The Error Correction Model representation

The Granger representation theorem (1981) has proven that non-stationary time series may be modeled by an Error Correction Model (ECM):

$$\Delta p_{1t} = \alpha \hat{z}_{t-1} + \sum_i \delta_i^2 \Delta p_{2t-i} - \sum_j \delta_j^1 \Delta p_{1t-j} + \varepsilon_t \tag{2.5}$$

where ε_t is a white noise, α is called the adjustment parameter, and subscripts i and j stand for time lags. The parameter α is estimated over the lagged residual of the linear combination between p_{1t} and p_{2t}: $\hat{z}_{t-1} = p_{1t-1} - \hat{\beta} p_{2t-1}$. It must be significative and negative to ensure that the system returns toward the long run equilibrium. Hence, ECM enables to combine short-run variation around the equilibrium via the differentiated variables, and long run evolution conveyed by variables in level through $\hat{z}_{t-1}$ (Lardic and Mignon, 2002).

2.5.4 The Johansen multivariate cointegration test

The concept of cointegration can be generalized to n variables, as long as they are $I(d)$ processes, with $d > 0$. As in the bivariate case, the existence of cointegration vectors requires that the linear combination of the n variables is $I(d-b)$:

$$z_t = p_{1t} - \beta_2 p_{2t} - \cdots - \beta_n p_{nt} \tag{2.6}$$

though all variables must not display necessarily the same degree of integration d for $n > 2$. The multivariate case is more complex than the bivariate case due to the possible existence of several cointegration combinations.

The ECM representation impart a dependent-explanatory relation between studied variables, though they may be endogenously defined. Therefore, Johansen (1988) developed a multivariate approach of cointegration based on the maximum likelihood estimator to identify the number of cointegration relations between n variables simultaneously studied. This approach appeals to a vector autoregressive (VAR) system in ECM form (Bourbonnais, 2009):

Consider a VAR(l) model in matrix form:

$$\mathbf{p}_t = A_0 + A_1 \mathbf{p}_{t-1} + A_2 \mathbf{p}_{t-2} + \cdots + A_l \mathbf{p}_{t-l} + \boldsymbol{\varepsilon}_t \tag{2.7}$$

with:

$\mathbf{p}_t$: vector of dimension $(n \times 1)$,

A_0: a vector of dimension $(n \times 1)$,

A_i: a matrix of dimension $(n \times n)$, with $i = 1, \ldots, l$.

This model can be reformulated in first-differences as follows:

$$\Delta \mathbf{p}_t = A_0 + B_1 \Delta \mathbf{p}_{t-1} + B_2 \Delta \mathbf{p}_{t-2} + \cdots + B_{l-1} \Delta \mathbf{p}_{t-l+1} + \pi \mathbf{p}_{t-1} + \boldsymbol{\varepsilon}_t \tag{2.8}$$

where the residual series $\boldsymbol{\varepsilon}_t \sim i.i.d$ and they are normally distributed with mean 0, and matrix B_i are functions of matrix A_i and $\pi = \big(\sum_{i=1}^{l} A_i - I \big)$. The matrix π can be written as $\pi = \boldsymbol{\alpha}\boldsymbol{\beta}\prime$, where $\boldsymbol{\alpha}$ is the vector of adjustment parameters and $\boldsymbol{\beta}$ is a vector containing the coefficients of the long run relation between variables. Also, similarly to the ECM representation, notice that $\boldsymbol{\beta}\prime \mathbf{p}_{t-1} = \mathbf{z}_{t-1}$, thus $\pi \mathbf{p}_{t-1}$ writes $\boldsymbol{\alpha} \mathbf{z}_{t-1}$.

Each independent linear combination in π corresponds to a cointegration vector. Thus, if the rank of matrix π is comprised between 1 and n, then there exists r cointegration relations. A rank of π equal to n implies that all variables are $I(0)$, a rank equal to 0 implies that none of the linear combinations between time series are stationary. Johansen (1995) proposes two types of tests to identify the rank of π : the trace test and the maximal eigenvalue test. Both tests work by successively excluding hypothesis on the value of r. At first, the null hypothesis $H_0 : r = 0$ is tested against $H_0 : r > 0$. If H_0 is rejected, the same alternative hypothesis is tested for $r + 1$, and so on until H_0 is accepted.[17]

To conduct a bivariate cointegration test between prices of wild and farmed fish, here follows an explicit expression of the VECM we estimate:

$$\begin{aligned}
\Delta p_{1t} &= a_0^1 + b_1^1 \Delta p_{1t-1} + b_2^1 \Delta p_{2t-1} + \cdots + b_{2l}^1 \Delta p_{2t-l} + \alpha^1 (p_{1t-1} - \beta p_{2t-1}) + \varepsilon_t^1 \\
\Delta p_{2t} &= a_0^2 + b_1^2 \Delta p_{1t-1} + b_2^2 \Delta p_{2t-1} + \cdots + b_{2l}^2 \Delta p_{2t-l} + \alpha^2 (p_{1t-1} - \beta p_{2t-1}) + \varepsilon_t^2.
\end{aligned} \tag{2.9}$$

2.5.5 The weak exogeneity test

The weak exogeneity test tackles the hypothesis that a variable has influenced the long-run stochastic path of the other variable of the system, but is not been influenced by them. It

[17] Critical values for these tests can be found in Johansen and Juselius (1990).

tests whether the price of one good p_{1t}, conditions that of the another, p_{2t}, or vice versa. In our case study, the interest is to identify if a cointegration relation is lead by one of the commodities under scrutiny.

The weak exogeneity test relates to the vector of adjustment parameters, $\boldsymbol{\alpha}$, of a VECM. It consists in testing the null hypothesis $H_0 : \boldsymbol{\alpha} = 0$. Recall that these parameters ensure the return of the system variables towards the long run equilibrium. They weight the cointegration relation in each equation of the model. If H_0 is accepted for one of the model equations, the associated dependent variable is said to be weakly exogenous.

2.6 Empirical results

2.6.1 Unit root tests

As mentioned, the first step in tackling market integration between goods is to examine whether price series are non-stationary in level. Table 2.5 reports results of the ADF test for the price series of interest to our analysis. In each case, the test statistic figuring in column one is obtained after selecting the most appropriate specification among the three models underlying the ADF test, including lags number. For all price series, test statistics are greater than the critical values reported in the second column, implying that their exists a unit root, i.e. the null hypothesis of non-stationarity is not rejected at the 5% level. In the third column, we note that for all price series in first-differences, the null hypothesis is rejected at the 5% level.

Table 2.5: ADF tests for unit roots in price series of fish

Prices in log	*Test stat. level*	*Critical value (5%)*	*Test stat. first-diff.*	*Critical value (5%)*
Whole farmed sea bass	0.185	-1.950	-6.046*	-1.950
Whole wild sea bass	-2.786	-3.492	-4.171*	-1.950
Whole farmed sea bream	-2.691	-2.917	-5.855*	-1.950
Whole wild sea bream	-0.270	-1.950	-5.125*	-1.950
Cut wild sea bass	0.186	-1.950	-5.678*	-1.950
Cut wild sea bream	-2.649	-2.918	-9.017*	-1.950

*Statistically significant at the 5% level.

Table 2.6 reports the results of the KPSS test. For almost all price series, the null hypothesis of stationarity in trend is rejected at the 5% level. For whole farmed sea bream and cut wild sea bream, it is rejected at the 10% level. However, both tests tend to converge. Evidence of non-stationarity in level and stationarity in first-difference allows us to conduct cointegration tests over these variables.

Table 2.6: KPSS tests of stationarity over price series of fish

Prices in log	*Test stat. level*	*Test stat. first-diff.*
Whole farmed sea bass	0.207*	.067
Whole wild sea bass	0.164*	.036
Whole farmed sea bream	0.24*	.056
Whole wild sea bream	0.143**	.023
Cut wild sea bass	0.179*	.044
Cut wild sea bream	0.123**	.037

Note: *Statistically significant at the 5% level.

**Statistically significant at the 10% level.

Critical value at the 5% level: 0.146.

Critical value at the 10% level: 0.119.

2.6.2 Competition between wild and farmed fish within the sea bream and sea bass markets

The results of pairwise cointegration tests between whole wild and farmed sea bass, and whole wild and farmed sea bream, respectively, are displayed in Table 2.7. Robustness checks for both VECM are reported in Appendix B.2. The first two columns provide the value of the trace and maximum-eigenvalue statistics for testing the null hypothesis of no cointegration vector, while columns three and four report these statistics for the null hypothesis of one cointegration vector. We rely on the Akaike Information Criterion (AIC) and the Bayesian Information Criterion (BIC) to select the number of lags in explanatory varibales.

Table 2.7: Within species bivariate Johansen tests between wild and farmed fish

	Null hypotheses				*LOP test*	
	Rank=0		*Rank=1*			
Prices in log (Whole fish)	*Trace statistic*	*Max-eigen. statistic*	*Trace statistic*	*Max-eigen. statistic*	*LR*	*p-value*
Sea bass: farmed/wild	33.608	19.779**	13.829	13.883		
Critical value at the 1% level	(24.60)	(20.20)	(12.97)	(12.97)		
Sea bream: farmed/wild	21.345	21.345	6.996*	6.996*	4.59	0.0322
Critical value at the 5% level	(19.96)	(15.67)	(9.42)	(9.24)		

Note: *Statistically significant at the 5% level. **Statistically significant at the 1% level.

In the case of sea bass, the trace test does not accept the null hypothesis of one cointegration between price series, while the maximum-eigenvalue test accepts the null hypothesis of no cointegration vector at the 1% level. This results indicates the absence of a long term price parity condition between wild and farmed sea bass. In order words, markets for these two fish products are not integrated.

We proceed to the Gregory-Hansen residual-based test (1996) for cointegration in the presence of a regime shift, between the prices series of wild and farmed sea bass, to control wether the cointegration relation between both prices is marked by a break. Yet, we find no evidence of cointegration neither. We conclude that both goods are not treated as substitutes

by consumers, though we do not rule out the possibility that characteristics of the supply side might actually account for this outcome.

Regarding sea bream, both the trace and maximum-eigenvalue tests accept the null hypothesis of one cointegration vector at the 5% level. Hence, whole wild and farmed sea bream display a long run price parity condition. This result implies that the production process is not determinant in the purchase decisions of this species. Nevertheless, in the last column we report results for the LOP test and find that it does not hold. The null hypothesis imposing restrictions, $H_0 : [\beta_{wild}, \beta_{farmed}] = [1, -1]$, on the long-run parameters is rejected at the 5% level. This means that wild and farmed sea bream markets are partially integrated (i.e. goods are imperfect substitutes).

Table 2.8 reports the estimate of the long-run relation between wild and farmed sea bream, normalized with respect to the price of wild sea bream.

Table 2.8: Sea bream model: the estimated cointegration vector

	Coefficient
β_{wild}	1
β_{farmed}	-0.232
constant	-1.834

Testing for weak exogeneity of the wild and farmed sea bream

Table 2.9 reports the estimates of adjustment coefficients for the VECM between farmed and wild sea bream. The adjustment parameter labeled α_{wild} (resp. α_{farmed}) intervenes in the equation where the dependent variable is wild sea bream (resp. farmed sea bream). We observe that α_{wild} is significant at the 5% level while not α_{farmed}. These outcomes imply that the price of farmed sea bream is weakly exogenous. In other words, farmed sea bream is the price leading production sector in the French whole fresh sea bream market.

Table 2.9: Sea bream model: adjustment parameters statistical significance

	Coefficient	*Std. Err.*	*p-value*
α_{wild}	-3.79 *	0.211	0.000
α_{farmed}	-0.81	0.169	0.417

*Statistically significant at the 5% level.

2.6.3 Competition between wild and farmed fish across different species

Table 2.10 reports pairwise cointegration tests between wild and farmed fish across different species. Precisely, we test for cointegration and the LOP between farmed sea bass and wild sea bream, as well as between farmed sea bream and wild sea bass. The idea is to appraise how farmed sea bass and se bream interact with some wild white fish of different species in the French fresh fish market. The third and fourth columns report the trace and maximum-

Table 2.10: Cross species bivariate Johansen tests between farmed and wild fish

	Null hypotheses				*LOP test*	
	Rank=0		*Rank=1*			
Prices in log (Whole fish)	*Trace statistic*	*Max-eigen. statistic*	*Trace statistic*	*Max-eigen. statistic*	*LR*	*p-value*
Farmed sea bream/Wild sea bass	76.428	57.213	19.215	19.215		
Farmed sea bass/Wild sea bream	35.69	26.573	9.118*	9.118*	0.03	0.8580
Critical value at the 5% level	(19.96)	(15.67)	(9.42)	(9.24)		

Note: *Statistically significant at the 5% level.

eigenvalue statistics for the null hypothesis of one cointegration vector. Not surprisingly, as farmed sea bass price is not cointegrated with its wild counterpart, farmed sea bream, is not treated as a substitute for wild sea bass neither, specially as sea bass is a higher value species [18].

[18]We also apply the Gregory-Hansen test (1996) for cointegration with regime shifts between the prices series of farmed sea bream and wild sea bass, but we find no evidence of cointegration neither.

On the other hand, our test results indicate that farmed sea bass and wild sea bream do compete in the French fresh fish market. Both tests, the trace and maximum-eigenvalue tests, accept the null hypothesis of one cointegration vector at the 5% level. Furthermore, the LOP holds between these two fish.

Table 2.11 reports the estimate of the long-run relation between wild sea bream and farmed sea bass, normalized with respect to the price of wild sea bream. As it appears, β_{farmed_bass} is quasi equal to 1, thus the satisfaction of the LOP.

Table 2.11: Farmed sea bass versus wild sea bream: the estimated cointegration vector

	Coefficient
β_{wild_bream}	1
β_{farmed_bass}	-1.064
constant	0.033

2.6.4 Assessing the effect of product form on market integration

Finally, we proceed to additional bivariate Johansen tests in order to assess to what extent does the form of products affect consumers' behavior. Indeed, intuition tells us that those purchasing whole products have an economic constraint more binding than their time constraint, contrary to those purchasing cut fish. Or, that they may be more familiar with seafood products. Thus, they may not be reluctant to proceed to the cutting exercise themselves.

The two first line in Table 2.12 report the same two within species cointegration tests as in Table 2.7, but with whole wild sea bass (resp. bream) prices replaced by cut wild bass (resp. bream). Fresh cut wild and whole farmed sea bass prices are not cointegrated. The null hypothesis of one cointegration vector is not accepted by the maximum-eigenvalue and trace test, at the 5% level. The product form does not affect the outcome of the cointegration test between wild and farmed sea bass in Table 2.7.

Alongside, we also find that the maximum-eigenvalue and trace statistics for the bivariate Johansen test between cut wild sea bream and whole farmed sea bream do not valid the null hypothesis of one cointegration vector at the 5% level. This result deviates from that between

whole wild sea bream and whole farmed sea bream in Table 2.10. The third cointegration test concerns cut wild bream and whole farmed bass. The null hypothesis of no cointegration vector is significative at the 5% level according to the trace test while the maximum-eigenvalue test does not accept the null hypothesis of one cointegration vector. This finding also contrast with the results of the bivariate Johansen test between both fish in their whole form in Table 2.10[19].

Hence, the cut form of fish apparently has an impact on the outcomes of our market delineation analysis.

Table 2.12: Bivariate Johansen test between fish of different form

	Null	*hypotheses*		
	Rank=0		*Rank=1*	
Prices in log	*Trace statistic*	*Max-eigen. statistic*	*Trace statistic*	*Max-eigen. statistic*
Whole wild bass/cut wild bass	65.85	33.75	32.1	32.1
Whole farmed bream/cut wild bream	67.9	20.68	47.22	20.68
Whole farmed bass/cut wild bream	15.17*	11.66	3.5	3.5

Note: *Statistically significant at the 5% level.

2.7 Discussion

Table 2.13 summarizes the results of our market delineation analysis of sea bass and sea bream species in France.

[19]We find no evidence of cointegration vector between the price of cut wild sea bass and whole farmed sea bream. We do not report this bivariate Johansen test because the outcome abounds the same conclusions than the other tests in Table 2.12 .

Table 2.13: Market integration outcomes

	Whole farmed sea bream	Whole farmed sea bass
Whole wild sea bass	not integrated	not integrated
Whole wild sea bream	partially-integrated	perfectly-intergated
Cut wild sea bass		not integrated
Cut wild sea bream	not integrated	not integrated

The way in which we interpret the outcomes of these bivariate Johansen tests is the following. The existence of a cointegration relation between the price series of a wild and farmed fish testifies to a certain degree of substitutability between them. Indeed, since the structure of production costs of the aquaculture and capture fishery industry are quite different, we find it reasonable to assume that market integration between both fish products stems from the demand behaviour of consumers rather than from spillover effects of common supply factors.

In the case of sea bass and sea bream farming, feed, juveniles and labour account for around 70% of production costs[20], of which about 35% for fish feed (Roth, and Ukendt, 2004). Feed cost is mostly affected by factors such as growth in import demand and soybean price (FAO, Globefish, 2013). In the case of fisheries, capital investment and operation costs rather constitutes the main costs. The latter can be divided in labour costs, running costs[21] and vessel costs. In France, 80-90% of sea bass and sea bream catches come from trawling. Running costs represent about 30% of demersal fish trawlers' total costs (FAO, 2005).

In the absence of cointegration between two price series, we do not rule out the possibility that demand for each product interact. We consider that characteristics of the supply side may account for this outcome.

Precisely, in a competitive market, if consumers consider wild and farmed fish of a same

[20]Fuel and energy consumption represents about 1% of total costs in Greece against 6% in France. Typically, this expense is for running seawater pumps, oxygenation and other machinery, vehicles, boats and refrigeration plant.

[21]Running costs are principally composed of fuel, lubricants, cost of selling fish, harbour dues, cost of ice, food and supplies for the crew.

species as perfect substitutes, then both product prices should equalize (allowance being made for transportation costs), else the less competitive sector will be ousted. If prices do not equalize, this can mean that both commodities are imperfect substitutes, which does not preclude the possibility of a price parity condition. If no price parity condition is found: either the farmed and wild fish are not be substitutes at all, or the markets of wild and farmed sea bass may not be efficient.

Asche *et al.* (2001) argue that the relative market share of each type of product is of importance in prices adjustment mechanisms. That is, the availability of each product must be widespread to make consumer arbitrage possible between both product types.

Lastly, data deficiencies can also be the cause of no cointegration between times series.

The fresh whole sea bream market

We find that the whole farmed and wild sea bream markets are partially integrated, and that their price relationship is lead by farmed sea bream (Table 2.7).

Content of the differences in the structure of costs faced by the aquaculture and capture fishery industry, we conclude that these products display a certain degree of substitutability. Partially integrated markets means that whole wild and farmed sea bream are imperfect substitutes. The wild fish benefits from a price premium (survey average price: 10.3€/kg against 8.29€/kg), indicating a relatively higher willingness to pay for this product. However, the fact that whole farmed sea bream is price leader suggests that whole wild sea bream may incur a downward price pressure. This influence of farmed sea bream supply on the price dynamics of its wild counterpart, is in line with what microeconomic theory predicts (see 2.4), regarding the introduction of a substitute in a given market. Whether consumer perception of farmed products evolves positively or negatively will determine the price competitiveness required from the fishery sector, and in turn, its economic viability. The positive consequence of a downward pressure on wild sea bream price is that it may reduce fishing pressure on sea bream stocks, allowing supply to increase. It may also push fisheries to proceed to an economic rationalisation of the industry. In these respects, and as it is generally the case, market competition benefits to the consumer.

Yet, fishermen rarely target a single species. A lower profitability of sea bream due to

harsh price competition from the aquaculture sector may lead fishermen to report fishing pressure on other fish species within their bundle of catches. If aquaculture production may alleviate pressure on a given wild species, it is not possible to qualify and quantify the net effect of farming on ecosystems'sustainability owing to the numerous linkages between inherent species.

The fresh whole sea bass market

Our empirical results in Table 2.7 indicate that wild and farmed sea bass markets are not integrated. Markets for fresh sea bass and sea bream differ in that: (1) wild sea bass is a higher valued species than wild sea bream on average; (2) the price differential is more important between wild and farmed sea bass, than between wild and farmed sea bream; (3) the market shares of wild versus farmed fish is better balanced in the case of sea bream.

It is not obvious whether the absence of cointegration between farmed and wild sea bass relates to consumers' sensitivity to the fish production process when it comes to higher-value species, or to supply features which may prevent market interactions between both fish. Here bellow, we consider several scenarios to explain this result.

Considering the latter explanation, we saw in the data section 2.3, that the average share of farmed sea bass, within French households consumption, between 2007 and 2012, is 31.3% (all forms of product included). Supposing our quantity figures are fully faithful, we posit that this share is reasonable enough to consider that market interaction between fresh farmed and wild sea bass is effective. Yet, the share of farmed fish in household consumption of whole sea bass evolves increasingly over our survey period, meaning that the balance between farmed and wild sea bass has improved recently. The monthly price series covering January 2007 to September 2012 might be to short to reveal market integration between these two fish products. What is more, the supply of farmed and wild sea bream, which price series are cointegrated, is more balanced (i.e. 40.5% of fresh sea bream consumed by households in France is farmed over our observation period).

In the event where market inefficiencies, such as a lake of trade-off opportunities at finer regional scale, accounts for the absence of market integration between wild and farmed sea bass, the high price difference between whole wild and farmed sea bass likely reflects, *inter*

alia, the higher production costs of the capture fishery sector compared to the aquaculture ones. This makes sense content of the lower labor costs faced by Greece to produce the imported farmed sea bass, relatively to those faced by the French fishery sector.

On the other hand, supposing the supply of fresh whole farmed and wild sea bass are efficient, the absence of market integration between both fish means that the *production process* attribute prevails on the *species type* attribute, in consumers purchasing decision. Actually, the production process attribute produces a separation in the whole French sea bass market, with the wild fish occupying the high-value segment. Indeed, the important price differential between wild and farmed fish, either in the case of sea bass or sea bream, can reflect the negative perception of aquaculture practices by consumers relatively to fisheries. This distinction in the willingness to pay for wild fish relatively to famed fish should theoretically be repeated in the case of other species. It can also relate to the more scare supply of wild fish. In this sense, the fish supply from aquaculture can be seen as a necessary complement to fisheries supply i.e. a lower quality but more abundant fish product, that compensates for the shortfall of the production of fisheries.

In any case, from our empirical analysis, the higher-value attributed sea bass compared to sea bream (survey average price, including all fish form and production process: 12.55€/kg against 9.93€/kg) does not produce the same price relation between wild and farmed fish, for each of these species (i.e. no market integration versus partial-market integration). Because our data set and methodology do not able to derive price elasticities, we cannot reliably cast wild sea bass in the terms of microeconomic typology of goods (normal versus luxury good). Nonetheless, this result difference can be explained in that consumers purchasing lower value fish are less regarding about flesh properties. If this is the case, the outcomes of our analysis are in line with Natale et al. (2013) who argue that interactions between wild and farmed products are likely to become more common, particularly in lower price aquaculture segments, as the production of newer aquaculture species expand to a level capable of conditioning the market.

Cross species market integration

We teste how farmed sea bass and sea bream interact with wild sea bream and sea bass, respectively, in Table 2.10. These two species are white fish species and have a similar flesh appearance. Sea bass meat is pinkish when raw and cooks up opaque white. Sea bream meat is also rosy-colored when raw and turns white when cooked (seafood handbook). Also, both fish have the same average size. As we are considering whole products, this attribute matters. The average sea bass length is worth 50 cm; the average sea bream length is worth 30-45 cm[22] (FishBase).

Not surprisingly, our empirical results in Table 2.10 infer that fresh whole farmed sea bream and wild sea bass price series are not cointegrated. These two fish display even higher price differences than whole wild and farmed sea bass. The price differential is not an impediment to substitutability as quantity versus quality trade-off can be carried. Yet, as it is not the case between wild and farmed sea bass in Table 2.7; it is even less likely between wild bass and farmed bream who differ by the specie type attribute in addition to the production process attribute.

Alongside, whole farmed sea bass and wild sea bream do compete in the French fresh whole fish market. Precisely, we find that the markets for these two products are perfectly integrated. As this result stem for the test for the *relative* LOP, it means that goods are close substitutes; not perfect substitutes. This result is quite surprising as wild and farmed whole sea bream markets are found to be only partially-integrated. However, referring to Table 2.4, over our survey period the average price of whole wild sea bream (10.3 €/kg) is closer to that of whole farmed sea bass (9.28 €/kg) than to that of whole farmed sea bream (8.29 €/kg). According to this cross species result, the argument that seems to prevail in consumers arbitrage is the price they are willing to pay to consume fresh fish rather than the production process or the species type when it comes to the lower-value fresh fish market segment.

Notice that perfectly integrated markets for whole farmed sea bass and wild sea bream rather runs against the hypothesis that the farmed sea bass market presents inefficiencies, as discussed previously.

[22]Notice that these figures correspond to the wild fish length; the only we dispose of.

The product form attribute

Lastly, in Table 2.12, we control wether the fish form (cut/whole) is a determinant product attribute in consumption decision. As when considering both fish in their whole form (Table 2.7), we find that the market for cut wild and whole farmed sea bass is not integrated. This finding is not surprising given that the price differential between cut wild sea bass and whole farmed sea bass is slightly greater than that between both fish in their whole form (average wild price exceed: 54% against 53%).

In contrast, market for cut wild bream and whole framed bream are not found to be integrated, so as that of cut wild bream and whole framed bass, although both pairwise cointegration tests yield opposite result when considering fish in their whole form in Table 2.7 and Table 2.10. Here again, the price differential between cut wild sea bream and whole farmed sea bream is greater than that between both fish in their whole from (average wild price exceed: 67.7% against 24.3%). Same remarque for cut wild bream and whole framed bass (average wild price exceed: 49.8% against 11%).

Thereby, the product from attribute alters market integration outcomes, at least for the species considered in our analysis, in their fresh form. The price premium and the gain in preparation time entailed to the cut fish form apparently overrides other products attributes.

Besides, we have pointed out in the case of whole sea bream, that the production process attribute does not constitute an impediment to substitutability between fish, although imperfect substitutes. In the case of whole wild sea bream and whole farmed bass, neither the species type, nor the production process seem to be a binding criteria to substitutability between these fish products. These cointegration relationships have in common that they concern fish in a same price range, below 10 €/kg on average.

Hence, the expenditure involved appears as a overriding decision criterion when purchasing fish in lower value market segments of the French fresh white fish market.

2.8 Conclusion

In this study, we teste for price parity between farmed and wild fish, within the sea bass and sea bream French fresh fish markets. We conduct our empirical analysis applying a

bivariate cointegration framework over fish price series. Price series are monthly and concern the period 2007–2012.

What emerges from our analysis is that market interactions between wild and farmed fish of a same form and species, differ from one species to another. Whole wild and farmed sea bream markets are found to be partially integrated while whole wild and farmed sea bass markets are not integrated. Based on the observation that wild sea bass is more expensive than wild sea bream, and displays a greater price differential with its farmed counterpart than does sea bream, we conclude that consumers may be sensitive to the seafood production process when it comes to higher-value species. In the case of sea bass, the production process attribute overrides the species type attribute in consumers purchasing decision.

On the other hand, when testing for price parity between fish products belonging to a lower price range, price closeness seems to be the main surrogate endpoint, at least within the French fresh withe fish market. This suggests that farmed fish is more likely to incorporate lower-value segments of the fish market.

Either in the case of bass or bream, wild fish displays a higher price. This price premium reflects either that French fisheries production is too scare, more costly than farmed fish imports from Greece and other commercial partners, or/and that the aquaculture industry may bear the brunt of a negative image. In particular, consumers tend to be cautious about farmed fish feeding and the potential consequences of changes in species' natural diet.

In France, farmed production of sea bass and sea bream strives to meet a sustainability and quality image, which has enabled the industry to develop export despite the tough competition (FNSEA, 2006). Yet supply is low due to space regulatory constraints, the feed supply issue and high labor costs, hindering production competitiveness. The expansion of the aquaculture industry is obviously subjected to a trade-off between quantity and quality. The need to enhance food security versus consumers' requirements in terms of sustainability and food nutritive quality portends a split of the aquaculture supply into branches of different standing.

Acknowledgments. The authors acknowledge the support of the French National Research Agency (ANR) under the CLEANER project (ANR-NT09-505778).

Bibliography

[1] **Anderson, J. L.**, (1985). Market interactions between aquaculture and the common-property commercial fishery*, Marine Resource Economics*, Vol. 2(1), 1-24.

[2] **Anderson, J. L.**, (2003). The International Seafood Trade, Cambridge: Woodhead Publishing.

[3] **Andrew, M. and Keithly, G. L.**, (2011). Source-Based Preferences and U.S. Salmon Imports, *Marine Resource Economics*, Vol. 26, 191-209.

[4] **Asche, F., Salvanes, K. G. and Steen, F.**, (1997). Market delineation and demand structure, *American Journal of Agricultural Economics*, Vol. 79(1), 139-150.

[5] **Asche, F. and Sebulonsen, T.**, (1998). Salmon prices in France and the UK: Does origin or market place matter? *Aquaculture Economics & Management,* Vol. 2(1), 23-30.

[6] **Asche, F. and Steen, F.**, (1998). The EU - one or several fish markets: an aggregated market delineation study of the EU fish market, *SNF-Report 61/98*.

[7] **Asche, F., Bremnes, H. and Wessells, C. R.**, (1999). Product aggregation, market Integration, and relationships between prices: An application to world salmon markets, *American Journal of Agricultural Economics*, Vol. 81(3), 568-581.

[8] **Asche, F.**, (2001). Testing the effect of an anti-dumping duty: the US salmon market, *Empirical Economics*, Vol. 26(2), 343-355.

[9] **Asche, F., Bjorndal, T. and Young, J.**, (2001): Market interactions for aquaculture products, *Aquaculture Economics & Management,* Vol. 5(5-6), 303–318.

[10] **Asche, F., Gordon, D. V. and Hannesson, R.**, (2002): Searching for price parity in the European whitefish market, *Applied Economics*, Vol. 34(8), 1017-1024(8).

[11] **Asche, F., Guttormsen, A. G., Sebulonsen, T. and Sissener, E. H.**, (2005). Competition between farmed and wild salmon: the Japanese salmon market, *Agricultural Economics,* Vol. 33, 333-340.

[12] **Chen, P. and Knez, P.J.**, (1995) Measurement of market integration and arbitrage, *The Review of Financial Studies*, Vol. 8(2), 287-325.

[13] **Broda, C. and Weinstein D. E.**,(2008) Understanding International Price Differences Using Barcode Data, NBER Working Paper No. 14017.

[14] **Bourbonnais, R.** *Econométrie*, 7^e édition Dunod, Paris, 2009.

[15] **Brigante, R. and Lem, A.**, (2001). Price interaction between aquaculture and fishery, *Working paper, XIII EAFE Conference, Salerno,* April 2001.

[16] **Chamberlin, E.**, (1933). *Theory of Monopolistic competition*, Harvard University Press, 213 p.

[17] **Clayton, P. L. and Gordon, D. V.**, (1999). From Atlantic to Pacific: price links in the US wild and farmed salmon market, *Aquaculture Economics & Management*, 3(2), 93-104.

[18] **Dixit, A. K. and Stiglitz, J. E.**, (1977). Monopolistic Competition and Optimum Product Diversity, *American Economic Review*, 67(3), 297-308.

[19] [**European Commission, Maritime Affairs and Fisheries**], (2012). *Fisheries and Aquaculture in Europe*, N°56.

[20] **Engle, C. and Rogers, J. H.**, (1996). How Wide Is the Border? *American Economic Review*, Vol. 86(5), 1112-1125.

[21] **Engle, R. F. and Yoo, S.**, (1987). Forecasting and Testing in Cointegrated Systems, *Journal of Econometrics*, Vol. 35 (1), 143-159.

[22] [**FAO**], (2005). *Economic Performance and Fishing Efficiency of Marine Capture Fisheries*, FAO Fisheries Technical paper 482.

[23] [**FAO**], (2010). *The State of World Fisheries and Aquaculture*, Fisheries and Aquaculture Department.

[24] [**FAO**], (2011). *Review of the State of World Marine Fishery Resources*, FAO Fisheries and Aquaculture Technical Paper 569, Rome.

[25] [**FAO GLOBEFISH**], (2009). *Seabass and Seabream*, European Price report: http://www.globefish.org/seabass-and-seabream-market-reports.html.

[26] [**FAO GLOBEFISH**], (2013). *Fish oil and Fish meal*, European Price report: http://www.globefish.org/fish-oil-and-fishmeal-january-2013.html.

[27] [**La Fédération Nationale des Syndicats d'Exploitants Agricoles (FNSEA)**], (2006).

[28] [**FranceAgriMer**], (2011). *Consommation des Produits de la Pêche et de l'Aquaculture*. Les donnéeséstatistiques de FranceAgriMer 2012 / PECHE ET AQUACULTURE.

[29] [**FranceAgriMer**], (2011). *La Commercialisation et la Consommation des Produits Aquacoles, Focus sur le bar et la dorade royale*. LES STNTHESES de FranceAgriMer 2011 / PECHE ET AQUACULTURE.

[30] [**FranceAgriMer**], (2012). *Les produits de la Pêche et de la Pisciculture en France*, Dossier de Presse.

[31] [**The Federation of European Aquaculture Producers (FEAP)**], (2014). *European agriculture production report 2003-2012*, http://www.feap.info.

[32] **Granger, C. W. J.**, (1981) Some properties of Time Series Data and their Use in Econometric Model Specification. *Journal of Econometrics*, Vol. 16(1), 121-130.

[33] **Gordon, D. V.**, (1993) A Fish Is a Fish Is a Fish? Testing for Market Linkages on the Paris Fish Market. *Marine Resource Economics*, Vol. 8, 331-343.

[34] **Ivanov, V., Kilian, L.**, (2001). A practitioner's guide to lag-order selection for vector autoregressions, *CEPR Discussion Paper,* No. 2685. London, Centre for Economic Policy Research.

[35] **Jaffry, S., Pascoe, S., Taylor, G., and Zabala, U.**, (2000). Price interactions between salmon and wild caught fish species on the Spanish market, *Aquaculture Economics & Management*, Vol. 4(3/4), 157-167.

[36] **Johansen, S.**, (1988). Statistical Analysis of Cointegration Vectors, *Journal of Economic Dynamics and Control*, Vol. 12, 231-254.

[37] **Johansen, S. and Juselius, K.**, (1990). Maximum Likelihood Estimator and Inferences on Cointegration with Application to the Demand for Money, *Oxford Bulletin of Economics and Statistics*, Vol. 52(2), 169-201.

[38] **Johansen, S.**, (1995). *Likelihood-based inference in cointegrated vector autoregressive models*, Oxford university press.

[39] **Knapp, G., Roheim, C. A., and Anderson, J. L.**, (2007). The great salmon run: competition between wild and farmed salmon, TRAFFIC North America, Washington D.C.: World Wildlife Fund.

[40] **Kwiatkowski, D., Phillips, P. C. B., Schmidt, P., Shin, Y**. (1992). Testing the null hypothesis of stationarity against the alternative of a unit root, *Journal of Econometrics,* Vol. 54 (1–3), 159–178.

[41] **Lardic, S. and Mignon, V.** *Économétrie Des Séries Temporelles Macroéconomiques et Financières*, Economica, 2002.

[42] **McKinnon, J. G.**, (1991). Critical Values for Cointegration Tests, in Engle R. F. and Granger, C. W. J. (eds), *Long-run Econometrics and Statistics*, Oxford University Press, 267-276.

[43] **Natale, F., Hofherr, J., Gianluca, F., Virtanen, J.** (2013). Interactions between aquaculture and fisheries, *Marine Policy*, Vol. 38, 205-213.

[44] **Naylor, R. L., Hardy, R. W., Bureau, D. P., Chiu, A., Elliott, M., Farrell, A. P., Forster, I., Gatlin, D. M., R. J. Goldburg, Hua, K. and Nichols, P. D.**, (2009). Feeding aquaculture in an era of finite resources, *PNAS*, Vol. 106 (36), 15103-15110.

[45] **Nielsen, M.**, (2005). Price formation and market integration on the European first-hand market for whitefish, *Marine Resource Economics*, Vol. 20, 185-202.

[46] **Nielsen, M., Setala, J., Laitinen, J., Saarni, K., Virtanen, J. and Honkanen, A.**, (2007). Market integration of farmed trout in Germany, *Marine Resource Economics*, Vol. 22, 195-213.

[47] **Norman-Lopez, A.**, (2009). Competition between different farmed and wild species: the US Tilapia market, *Marine Resource Economics,* Vol. 24, 237-251.

[48] [**OECD**], (2010). Advancing the Aquaculture Agenda, Workshop Proceedings.

[49] **Robinson, J.**, (1933). *The Economics of Imperfect Competition*, London: Macmillan, 352 p.

[50] **Stigler, G.J.** (1969). The Theory of Price, London, England: MacMillan.

[51] **Tacon, A. G. J. and Metian, M.**, (2008). Global overview on the use of fish meal and fish oil in industrially compounded aquafeeds: Trends and future prospects, *Aquaculture*, Vol. 285, 146-158.

[52] **Roth, E. and Ukendt, M.**, (2004). *Study of the market for aquaculture produced seabass and seabream species*. The European Commission DG Fisheries, Final Report 23rd April 2004. Department of Marketing.

[53] **Valderrama, D. and Anderson, J.**, (2010). Market interactions between aquaculture and common-property fisheries: Recent evidence from the Bristol Bay sockeye salmon fishery in Alaska, *Journal of Environmental Economics and Management*, Vol. 59, 115-128.

[54] **Ye, Y. and Beddington, J. R.**, (1996). Bioeconomic interactions between the capture fishery and aquaculture, *Marine Resource Economics*, Vol. 11, 105-123.

[55] **Worm, B., Hilborn, R., Baum, J. K., Branch, T. A., Collie, J. S., Costello, C., Fogarty, M. J., Fulton, E. A., Hutchings, J. A., Jennings, S., Jensen, O. P., Lotze, H. K., Mace, P. M., McClanahan, T. R., Minto, C., Palumbi,**

S. R., Parma, A. M., Ricard, D., Rosenberg, A. A., Watson, R., Zeller, D., (2009). Rebuilding Global Fisheries, *Science*, Vol. 325, 578-585.

Appendix B

B.1 Pairwise price comparisons

Variable	Mean	Std. Err.	Std. Dev.	[95% Conf. Interval]
Wild sea bass	14.24	0.18	1.51	[13.87; 14.60]
Farmed sea bass	9.28	0.13	1.10	[9.03; 9.54]
diff	4.96	0.23	1.88	[4.51; 5.42]
H1: mean(diff) < 0		t = 21.92		Pr(T < t) = 1.0000

Number of observations: 69.

Student test degrees of freedom: 68.

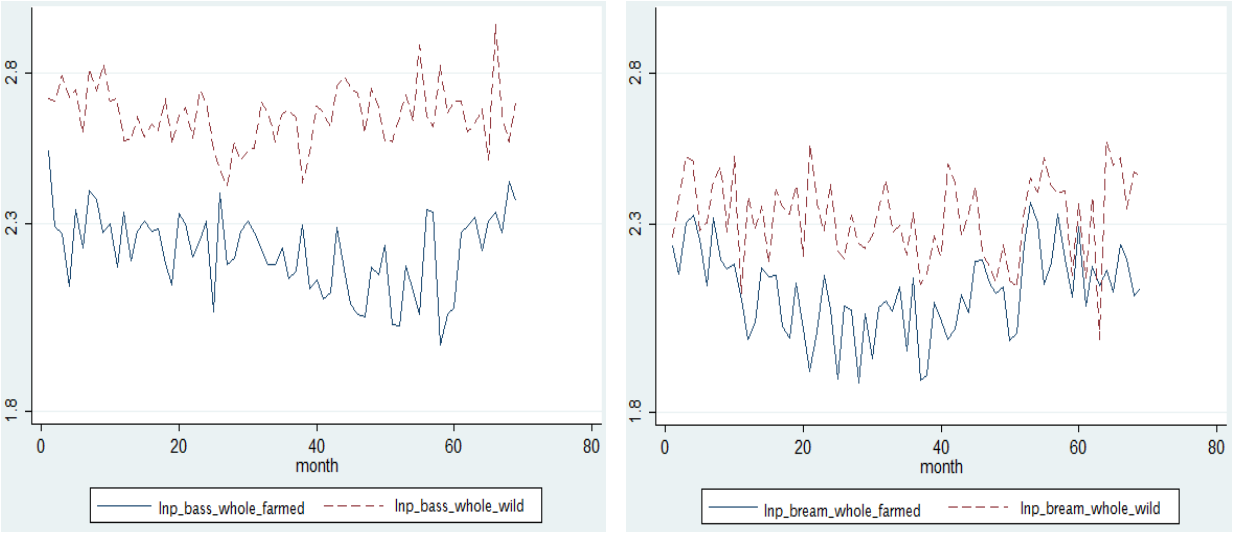
Table B.1: Mean-comparison test: wild versus farmed sea bass

Variable	Mean	Std. Err.	Std. Dev.	[95% Conf. Interval]
Wild sea bream	10.30	0.15	1.28	[9.99 ; 10.61]
Farmed sea bream	8.29	0.12	0.97	[8.06 ; 8.52]
diff	2.01	0.15	1.28	[1.70; 2.32]
H1: mean(diff) < 0		t = 12.9902		Pr(T < t) = 1.0000

Number of observations: 69.

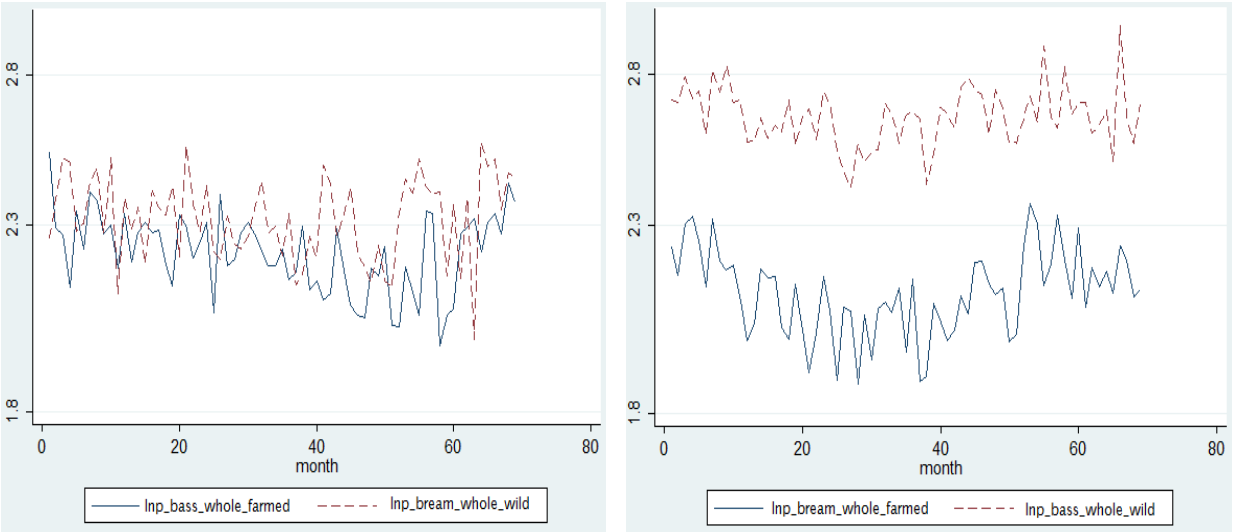
Student test degrees of freedom: 68.

Table B.2: Mean-comparison test: wild versus farmed sea bream



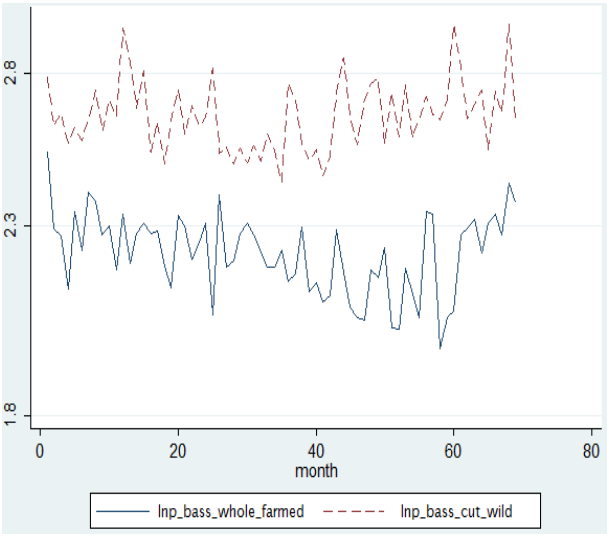
(a) bass (b) bream

Figure B.1: Prices of whole sea bass and sea bream in logarithms: within species comparison (2007-2012)

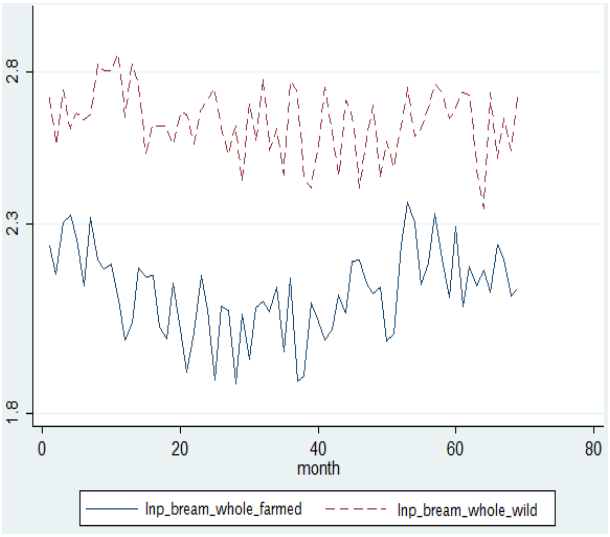


(a) farmed bass vs. wild bream (b) farmed bream vs. wild bass

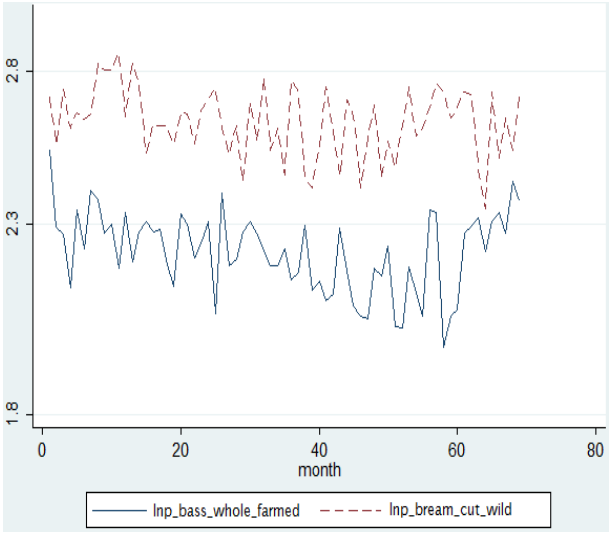
Figure B.2: Prices of whole sea bass and sea bream in logarithms: between species comparison (2007-2012)



(a) whole farmed vs. wild cut bass



(b) whole farmed vs. wild cut bream



(c) whole farmed bream vs. cut wild bass

Figure B.3: Prices of sea bass and sea bream in logarithms: between fish form comparison (2007-2012)

Prices are displayed in logarithms. Farmed and wild sea bream prices seem relatively correlated over the time period, specially since 2010 (around month 36 in Figure B.1). This is less the case for farmed and wild sea bass, especially since 2009 (around month 24 Figure B.2). Figure B.2(a) suggests that farmed sea bass and wild sea bream have a common price trend, while it is hard to conclude regarding farmed sea bream and wild sea bass prices

(Figure B.2(b)). Figure B.3 concerns pairwise price comparison between products of different form.

B.2 Robustness checks

We report below robustness checks for the two VECMs models of particular interest in this analysis. That is, the wild/farmed sea bass and sea bream VECMs depicted in Table 2.7.

We carry-out the Lagrange-multiplier (LM) test which controls for autocorrelation in the residuals of a VECM. It is based on the Chi-square test statistic. The null hypothesis is that there is no autocorrelation in the residuals for any of the lag orders tested. It appears from Table B.3 and Table B.4 that in both cases the null hypothesis is accepted at the 5% level for all tested lag orders.

Table B.3: Lagrange-multiplier test: wild/farmed sea bream VECM

lag	χ^2_a	Prob
1	6.8125*	0.14613
2	11.5575*	0.02096
3	5.6003*	0.23105
4	4.7071 *	0.31869

Note: *Statistically significant at the 5% level.

[a] Test statistic of degrees of freedom 4.

Table B.4: Lagrange-multiplier test: wild/farmed sea bass VECM

lag	χ^2_a	Prob
1	5.7705*	0.21695
2	6.8071*	0.14644
3	2.9689*	0.56304
4	6.3802*	0.17250

Note: *Statistically significant at the 5% level.

[a]Test statistic of degrees of freedom 4.

In addition, we report in Table B.5 and B.6 the Jarque-Bera test statistic for each equation and all equations jointly of our two VECMs, which are computed against the null hypothesis that disturbances are normally distributed. The Jarque-Bera statistic tests skewness and kurtosis jointly. In both tables, the single equations and overall Jarque-Bera statistics do not reject the null hypothesis of normality at the 5% level.

These outcomes further support our VECM specifications as the multivariate approach to cointegration developed by Johansen (1995) is based on the maximum likelihood estimator which assumes that errors are independently and normally distributed.

Table B.5: Jarque−Bera test: wild/farmed sea bream VECM

Dependent Var.	χ^2	Prob
Whole wild sea bream	0.838*[a]	0.65764
Whole farmed sea bream	0.362*[a]	0.83451
All equations	1.200 *[b]	0.87810

Note: *Statistically significant at the 5% level.

[a] Test statistic of degrees of freedom 2.

[b] Test statistic of degrees of freedom 4.

Table B.6: Jarque−Bera test: wild/farmed sea bass VECM

Dependent Var.	χ^2	Prob
Whole wild sea bass	7.070*[a]	0.02916
Whole farmed sea bass	1.913*[a]	0.38418
All equations	8.983*[b]	0.06152

Note: *Statistically significant at the 5% level.

[a]Test statistic of degrees of freedom 2.

[b] Test statistic of degrees of freedom 4.

Lastly, we check whether the eigenvalue stability condition holds. This condition provides indicators of whether the number of cointegrating equations is misspecified or whether the

cointegrating equations, which are assumed to be stationary, are not stationary. It uses the coefficient estimates from the previously fitted VECM to back out estimates of the coefficients of the corresponding VAR and then compute the eigenvalues of the companion matrix. Roots strictly inferior to 1 calls for stability of the specification, though there is no distribution theory to measure how far a root should be from 1. The roots for the wild/farmed sea bream and sea bass VECMs are reported in Table B.7 and B.8 plotted in Figure B.4. The modulus of each eigenvalue is strictly less than 1; graphically all the eigenvalues lie inside the unit circles. These outcomes show evidence that the cointegration equations are stationary.

Table B.7: Eigenvalue stability condition: wild/farmed sea bream VECM

Eigenvalues	Modulus
1	1
-.57861	.57861
.251345	.251345
.03782637	.037826

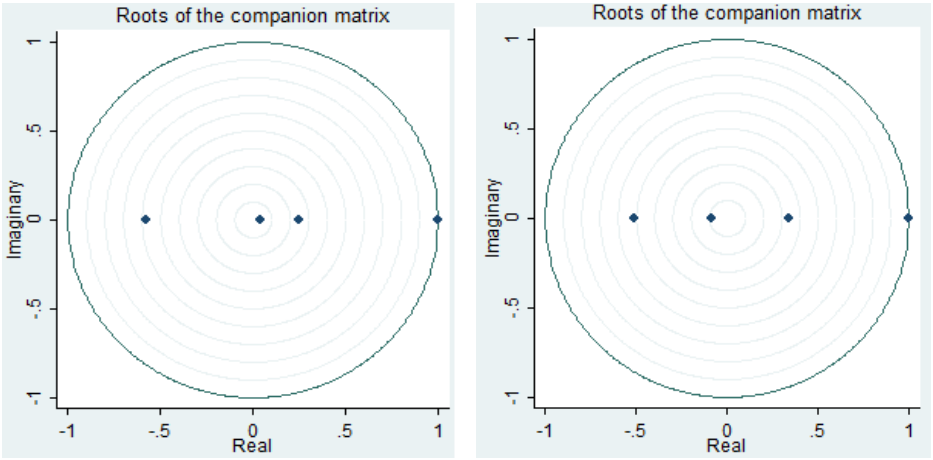
Note: The VECM specification imposes a unit modulus.

Table B.8: Eigenvalue stability condition: wild/farmed sea bass VECM

Eigenvalue	Modulus
1	1
-.5154871	.515487
.3401161	.340116
-.08615876	.086159

Note: The VECM specification imposes a unit modulus.

Figure B.4: Dispersion of the eigenvalues: wild/farmed sea bream & sea bass VECMs



(a) sea bream

(b) sea bass

Chapter 3

Managing harvested ecosystems under uncertainty: the viability approach

3.1 A review of different approaches to risk management in fisheries

There is a growing demand for moving from single species management schemes to an ecosystem approach of fisheries management (Garcia et al. [31]). The World Summit on Sustainable Development (Johannesburg, 2002) encouraged the adoption of an ecosystem approach by 2010. The ecosystem approach of fisheries faces many issues, ranging from the high cost of the science required (developing data collection, analytical tools, and models) to the practical difficulties of changing the governance system and processes (Cury et al. [19], Sainsbury et al. [56]).

Moreover, uncertainty inherent to fisheries leads to risk in decision making (Hilborn et al. [34]), and is recognized to play an important role in the failure of management regimes. Fisheries modeling requires estimations of stock status, total withdrawal from stock and a comprehension of ecosystems' response to harvesting. Because fish stocks and fish are not easily observable, in contrast to other living species, such information remains imprecise and error prone. Indeed, fisheries data exhibit such large year-to-year fluctuations which Ludwig and Walters [40] consider for a large part as measurement errors. Furthermore, the structure

and dynamics of ecosystems are often poorly known. At last, uncertain climatic hazards, technical progress or other factors inherent to wildlife are likely to affect fisheries productivity in an unpredictable way. Some claim that fishing decreases the resilience of fish populations, rendering them more vulnerable to environmental change (Lauck et al. [38]) and that not accounting for uncertainty can lead to excessive harvest of a resource (Hilborn and Walters [33]).

In this section, we review different existing approaches to the sustainable management of fisheries under uncertainty. More precisely, in each case, we review how the approach addresses policy goals in a dynamic setting with uncertainty. That is, whether the approach fits into a normative or positive aim, and the degree of precaution which is adopted with respect to risk. We start with the economic field. Next, we overview how decision making is undertaken in practice. Lastly, we propose the viability approach as a suitable theoretical framework to design management strategies of harvested ecosystems under uncertainty as it deals jointly with i) ecosystem dynamics, ii) conflicting issues of production and preservation and iii) dynamics uncertainties.

3.1.1 The economics approach

In economics, the standard criterion to address intertemporal allocation issues of consumption under uncertainty is the discounted utility (DU). According to the utilitarianism theory, the morally right action is to maximize the happiness of the greatest number, even if it is at the expense of some individual (Bentham [9]). Society's overall welfare is obtained by summing the utilities of each individual, and generation, in an intertemporal framework. This conception of social welfare has been criticized for being unequitable because it overlooks distributional concerns, in the sense that, any additional unit of utility has the same value, regardless of the initial endowment of the recipient.

The introduction of a discount rate on future generations' utility level in the welfare maximizing programme is also controversial. The discount rate stands for the societal preference for present. The higher it is, the less society cares for the future generations' welfare. Hence, it gives primacy to the present generation on the use of the productive capacity, at the expense of future ones. Chichilnisky [15] submits that "an appropriate discounting of future

utilities need not be the dictatorship of the present". Chichilnisky [14] also raises that utility is not appropriate for comparison between populations or generation as preferences may differ. However, economists justify the discounted utility criterion by the fact that it imputes to society a capital accumulation rate that individuals are willing to undertake. Otherwise, the specification of intertemporal utility functions generally includes a parameter for relative risk aversion, which is difficultly disentangle from the intertemporal elasticity of substitution. A high risk aversion tends to soften the influence of the discount factor by smoothing consumption between periods: a consumer decides to save more immediately given his low expected returns on what he saves.

Environmental degradation and resources depletion have raised concern for the state of nature and subsequent production potential bequeath to future generations. The exploitation of exhaustible resources is a key issue in the the debate of intergenerational equity. The bequeaths depends on the willingness of current generations to sacrifice current consumption for achieving environmental objectives to the benefit of future generations. Only those who reside in the present may decide, as conveyed by the discounted utility criterion. On the other hand, several thinkers view access to a minimal standard of living as a right that should be guaranteed to all generations. Rawls theory of justice goes further; he assumes that each person should have equal rights regardless of the underlying growth and welfare implications.

Considering the global warming stake, the challenge is roughly to define how much emission reduction should the present generation achieve to limit future damages, while the damage function and the abatement-cost function are uncertain. Within this trade-off, the value of the discount factor is one of the core element on which Stern [62] and Nordhaus [50] disagree. Nordhaus sets the discount rate[1] to be consistent with observable economic outcomes like the real return on capital. His argument is pragmatic: if economic data indicates that people have a strong preference for the present, policy propositions must be designed in adequacy for action to take place. In this respect, the efficient policy strategy is to concentrate abatement efforts in the future to diminish costs. Stern has a more normative approach: all generations have the same importance and should benefit from the maximum level of consumption possible at their time. The discount parameter he chooses is quasi null.

[1]His calibration of the discount rate is based on the Ramsey equation.

Consequently, the optimizing policy will more equitably share depollution efforts between generations.

In this line, at the other extreme of the discounted utility, is the max-min criterion. It is fed into the debate on intergenerational equity by Rawls [54], who attacks the utilitarian approach to social choice. It assumes welfare is maximized only when the utility of those society members that have the least is improved. Going further, Solow [60] argues that intergenerational equity reports to identifying what each generation can demand of its ancestors and should leave to its descent. The max-min criterion is well suited in this respect. It advocates a constant utility between generations[2]. Furthermore, it satisfies the Hartwick's rule definition of sustainability[3]. This investment rule states that if a resource is finite, earlier generations should off-set extractions by reinvesting resource rents into reproducible capital, thus keeping the value of net investments equal to zero. Thereby, the standard of living of society does not decline in the future. However, the max-min criterion is criticized for passing over the utility of generations that are not the poorest. Moreover, their is no concern for growth (Long and Martinet [39]).

Alternative normative approaches for ranking social options have been developed, lying in between the DU and the max-min criteria. Chichilnisky [13] proposes a welfare function that weights the discounted sum of utilities, and a second term, the utility withdrawn by the limiting generation in an infinite time horizon. Alvarez-Cuadrado and Long [1] propose a weighted sum of the discounted sum of utilities and the utility of the generation that is worst-off over time. This last criteria is labeled the "Mixed Bentham-Rawls" criterion. Both initiatives tend to preserve a minimum utility level to each generation, which is endogenously defined through the different weights affected to them. Long and Martinet [39] develop an indicator called "Rights and Welfare Indicator". It combines a welfare index, based on the discounted sum of utilities, and an index of rights to minimum needs, which are endogenously defined. In contrast, Martinet [42] focuses on the set of minimal rights that can be guaranteed to all generations, without considering welfare. There are no possible trade-offs between

[2]Though, the initial capita stock must be sufficiently high not to perpetuate poverty.

[3]Provided that the economy is not distorted and, as raised by Solow [60], that the natural resource and produced capital goods/inputs display a degree of substituability.

generations. The aim is to highlight the trade-offs embodied in the different objectives pursued and determine sustainable thresholds.

Fisheries resources economics

Fisheries resources are classified as renewable resources although commercial depletion may occur. This situation corresponds to the case where a stock is so low that it is unprofitable to fish for it. In fisheries economics, optimality is generally obtained by maximizing the expected discounted sum of net revenues from the harvest of a resource under uncertainty. In this field, the discounted expected utility criterion, classically applied in economic theory, is not consistently retained. In fact, as exposed in §3.1.2, the expected discounted utility criterion is not a reference criterion in the practice of fisheries management neither.

Fisheries resources are described by dynamical models resting upon biological foundations. Using a logistic framework to describe population dynamics, the Schaefer [57] model is one of the first model to illuminate the connections among fishing, stock dynamics, and various potential long-run equilibria. The work of Schaefer [57] gave birth to the key notion of maximum sustainable yield (MSY) [4] (Gordon [32]). Today, this notion is criticized for several reasons, in particular because it promotes a single-species management approach.

Since then, fisheries dynamic models attempt to capture the ecosystem dimension of the exploited system rather than the single demographic structure of the exploited stock, although this is already a challenging exercise. Of course, various degrees of simplifications are introduced on the basis of biologists work to reduce complexity of the analysis.

Thus, the standard fishery management problem comes down to maximizing the expected discounted net sum of fishing revenues with respect to harvesting effort, given a fishery dynamic model, and may be subject to different sources of uncertainty. The production structure can be fixed, or not, depending on the fishery management issue that is analyzed (i.e. the optimal investment in fleet capacity, the optimal harvesting time, the minimization of monitoring costs, etc.).

Under uncertainty, if the distribution of random variables is assumed to be fully known,

[4]The largest yield that can be taken from a species' stock over an indefinite period, without decreasing the population size.

such a setting is solved by means of a stochastic dynamic programming equation, where state variables generally correspond to biomasses, and controls to harvesting efforts. Optimality is reached when the value function has converged to a stable outcome. The optimal control policy is then directly expressed as a function of the current state variables (Miranda and Fackler [48]).

However, the application of optimal control theory to fisheries management suffers from some criticisms. The economic rationale of discounting future resource rents leads an optimal fish stock to depend on the discount factor, the distribution of uncertainty and different economic/biological parameters of the problem. The higher is the discount factor the closer will be the stationary stock from zero. Clark et al. [18] allege that, in some cases, the exhaustion of a species can be economically optimal. Casting fisheries resources into market assets, the authors explain that "the social rate of discount is to be seen as reflecting the opportunity cost of investing in the resources". Thereby, when the rate of return of a common class of assets is greater than the expected return of the fish stock at stake, it is more profitable to exhaust the resource and invest subsequent revenues rather than to harvest it at growth rate. This approach to fishery management is clearly anti-egalitarian if the rents from resource exhaustion are not invested into reproducible and substitutable productive capacity.

Furthermore, though maximizing a criteria has the advantage to always[5] propose an answer to the problem, it stresses a unique decision rule or control policy. No alternative/sub-optimal policies are explored. Though, the transition towards the optimal states dictated by the decision rule may cause economic hardship, which a society is not willing undertake.

Risk management in fisheries economics

A part of fisheries economics literature tackles the consequences of biological uncertainty on the optimal control policy, the optimal regulatory setting or instrument, assuming that the probability density of uncertain variables are known, i.i.d. process. Most often, the regulator is assumed to set policy instruments before uncertainty is realized, within an exercise period. These random variables convey different sources of uncertainty, ranging from imperfect infor-

[5]Given that some general conditions on the ingredients/functions of the optimization programme are satisfied (C. P. Simon [11]).

mation regarding the size of fish stocks, uncertain population dynamics or stock-recruitment relationships, to environmental stochasticity.

Let us overview the findings of a some of these studies. Mirman and Spulber [49] analyzes how markets deal with the presence of stochastic biological growth to allocate renewable resources over time. A competitive allocation of property rights achieves the optimal expected rents level, given endogenous prices and a complete set of contingent futures markets. The open access property arrangement with constant price is also examined. It this setting, it is shown that the average harvesting decisions, based on current stock levels may be very misleading since a stock which may survive "on average" may be extinct in a finite number of years with probability one.

Anderson [2] questions how uncertainty affects the choice of instrument in a dynamic optimization setting. A stochastic model of a dynamic fishery is used to compare the relative performance of a tax on landings and a harvesting quota in this paper. He finds that the instrument that performs most efficiently depends on the specific fishery being regulated. Addressing the same question, Weitzman [64] considers a stochastic stock-recruitment relation, and shows that the first-best solution can be reached only by a tax system. The dilemma between price and quantity controls under uncertainty has actually been much discussed in different fields of the economic literature (Anderson and Young [3], Dasgupta and Stiglitz [20], Fishelson and Flatters [30], Spence [61], Weitzman [63]).

Because managing fisheries is a interdisciplinary topic, part of the literature that addresses the impact of risk on the exploitation of renewable resources actually straddles biosciences and economics issues. As an example, (Reed [55]) shows a constant-escapement feedback policy is optimal in maximizing expected discounted net revenue from an animal resource whose dynamics are described by a stochastic stock-recruitment model, provided that unit harvesting costs satisfy certain conditions.

Based on a logistic growth curve, Benddington and May [6], Doubleday [25], Sissenwine [58] argue that for a given level of environmental randomness, population numbers and yield exhibit greater fluctuations as harvesting effort increases, specially as the point of maximum sustainable yield (MSY) is approached or exceeded. May et al. [46] investigates whether this evidence holds for other growth curves, focusing density-independent noise. The authors

find that this result remains broadly true for other recruitment models, and concludes that important trade-offs need to be considered between the high levels of fluctuation and unpredictability generated by the quest of highest yield on the one hand, and on the other the lower variance and greater stability accompanying lower yields.

Mendelssohn [47] compares the outcomes derived from two extreme policy strategies for stochastic harvesting models. One approach is to maximize the long-run expected returns, given the density probability of environmental randomness. The other is to identify the decision rule that minimizes the probability for the natural population to violate a minimum threshold size. The rent optimizing strategy gives a higher total expected discounted harvest but the risk minimizing strategy performs better in reducing the risk of being in the low population size. The authors show that, for many problems, the risk minimizing strategy either does not exist, or else it is an "extreme" policy that is equally undesirable.

One may object that this conclusion depends on the likeliness of highly adverse events. Overall, this study nicely illustrates the core dilemma faced by management authorities: how to set quotas on a randomly varying population such as to avoid severe depletion of the stock level when it has been overestimated, but also, so as not to undermine catches below what could have been caught safely in case of underestimation? Depending on the reliability of available information, should policy instruments adapt to the average risk, the extreme one? These questions constitute a common challenge in the economics field.

Indeed, let us recall that the studies presented above rest upon the assumption that the probability density of stochastic variables is fully known. Hence, the questions that arise are how are the models of these random processes defined? What strategy should be adopted when too little information on biological dynamics is available?

A frequent method for specifying the probability distribution of uncertainty scenarios in marine science is the Bayesian approach (Clark [16], Clark and Kirkwood [17], Ludwig and Walters [41], Punt and Hilborn [52]). This method is advocated by the FAO for stock assessment in fisheries (Punt and Hilborn [53]). The Bayesian approach consists in building a prior density distribution on the basis of historical estimates, as well as, through inferences from other species, and updating this prior with current data to define a posterior.

3.1.2 The operational approach

In practice, management or effort strategies are evaluated in "multicriteria" frameworks with no clear axiomatic foundations. The key scientific concept used to provide regulatory mechanisms recommendations in practical decision making literature is a "management procedure" (MP). This concept was developed by the International Whaling Commission (IWC) in the late 1980's. MPs are defined by Butterworth and Punt [10] as a set of rules which describes recommendations for management actions. For instance, a constant harvest rate rule is a management strategy. It has already been experimented on the Alaskan Pacific halibut, and on other fisheries. A MP is analogous to a control policy or decision rule in the economics literature. Ideally, to select a MP, performance of different MPs should be evaluated and ranked with respect to each decision criteria. In case of interactions between two species a "joint management procedures" is developed.

In spite of many criticisms (Larkin [37]), the maximum sustainable yield (MSY) is a management benchmark which has been, and is still, extensively targeted by management authorities, among other criteria. The IWC classifies as sustained management stocks, those for which the management strategy harvests 90% of the estimated MSY if the biomass exceeds the corresponding MSY stock level (MSYL). The quota is reduced by a additional 10% for every 1% that the stock falls below the MSYL.

The Management strategy evaluation (MSE) is a method elaborated for comparing management procedures, which takes into account uncertainty, conflicting objectives and time horizon. As detailed in Sainsbury et al. [56], the MSE approach consists in defining an operational set of management objectives, and evaluating the performance of various alternative MPs with respect to the specified objectives, taking into account uncertainty in the modelling processes. The method consists in testing a particular MP in a great number of simulations over a given time period, each simulation representing a plausible "state of nature" (scenario), and in computing statistics over the simulation results to summarize the performance of the particular MP. Generally, this is done by ways of mathematical expectations or of weighted summations of the different scenarios postulated.

The weakness of this method is that it is not able to prescribe an optimal bioeconomic strategy. That is, once the performance statistics are computed for each MP, comparison

of the likely distance of corresponding management actions to the conflicting management objectives requires to make a trade-off choice. This approach thus provides a clear description of the consequences of management procedures but with no common currency between the different objectives and risks to sum up the results and rank the alternative management procedures. To complement the analysis, one can plot MPs in a two dimensional space (each quantifying an objective), and draw a trade-off curve like do Oliveira and Butterworth [51], but the decision method remains visual. Consequently, the decision maker is left with a multi-criteria decision problem and is free to apply his own weighting and risk preferences to alternative objectives.

3.1.3 The viability approach

The stochastic viability theory (De Lara and Doyen [21]) aims at analysing if there exists states for which dynamics of a system, under uncertainty, and constraints describing given objectives, are compatible. These constraints are supposed to maintain the existence and good health of the system. We set forward the stochastic viability approach to address fishery management issues as it accounts for risk, several criteria — here conflicting sustainability objectives — and provides a common value to assess and rank alternative management procedures. In this respect, this approach can be viewed as a scientific tool to support multi-criteria decision making. Indeed, decision bodies in charge of examining management actions are generally faced with several indicators providing information on risk and benefits of alternative options, but lack of a measure embracing a plurality of criteria.

Basically, the stochastic viability framework consists of a control dynamic model subject to uncertainties, under constraints. Scenarios are sequences of uncertainties over the planning horizon. The set of scenarios is equipped with a probability distribution. The stochastic viability approach allows to identify control strategies — also called feedbacks[6] — such that a set of constraints, representing various objectives, is respected with maximal probability. This probability is called maximal viability probability and constitutes our common value to assess management actions. The higher it is, the lower is the risk of violating constraints.

[6] A feedback strategy is a rule mapping states towards controls. It corresponds to the decision rule in the economists' dialect, or to the management strategy in the operational approach.

Technically, control strategies are closed loop control systems: the control is dependent on the state. They can be computed by means of a stochastic dynamic programming equation under proper assumptions.

The theory concentrates on initial states as follows. Starting from a so-called *stochastic viable state*, it seeks a control strategy guaranteeing constraints for all dates of a time span, whatever the uncertainty scenario, with a maximal probability. The set of stochastic viable states is called the *stochastic viability kernel*. Several strategies may achieve the same viability probability level for a given initial state. The concept of stochastic viability kernel is convenient as it allows to delineate the bundle of initial states attached to any level of guarantee with respect to constraints. Thus, it conveys a great deal of information on the feasibility of pursued objectives. It also shades light on the set of states to be avoided. Furthermore, it is possible to observe the sensitivity of the stochastic viable states with respect to constraints, by varying their level.

Adapting this framework to bioeconomic systems allows to seek consistency between a fisheries dynamics and conflicting economic and conservation objectives. As in the economics or marine science literature, state variables generally correspond to biomasses and controls to harvesting efforts, while uncertainty can take various forms depending on its source.

A set of binding thresholds describes the sustainability of the system. The respect of these thresholds ensures that ecosystems are likely to persist as source of goods and services. Obviously, the (stochastic) viability approach does not fit into the economic rational of maximizing the resource rent. It is positive rather than normative. Actually, it is closer to what is undertaken in practice (see §3.1.2). Because it is backed on safety thresholds, the viability approach is particularly suited to the management of fisheries, which is increasingly governed by biological reference points constituting bottom line for stock depletion (Smith et al. [59]). Economic thresholds are commonly assumed to be provided by policymakers rather than derived from a fishery production structure. However, it is certainly possible to introduce such modelling component in the viability theoretical framework.

What is more, thresholds are generally set constant over time, implying that all generations are subject to the same constraints. This formalization of the problem is in line with the egalitarian vision of resource exploitation advocated by Rawls [54], Solow [60]. In fact, Doyen

et al. [28] demonstrate that the viability framework allows to characterize the maximin path as a particular viable trajectory. Going further, the authors explain that "whenever the solution of a given optimization problem can be formulated in terms of a viability kernel, the solution inherits the properties of the kernel". This result follows from the fact that the viability approach aims at describing all the evolutions of a dynamical system, that satisfy, at every times, given objectives, with a maximum probability in the stochastic setting. It permits to explore alternative management options, which offers flexibility to the decision maker.

As exposed in §3.1.2, part of the economists would object that addressing management goals without accounting for society's time preference will result in unadapted policy recommendations i.e. which imposes sacrifices in terms of welfare, to individuals that are not willing to bear them. Nonetheless, given that wildlife populations often display wide fluctuations in an unpredictable way, fisheries management goals and schemes should be updated regularly, in accordance to the new data on stock assessments. Hence, if the time frame of a management exercise does not exceed a couple of years, keeping sustainability constraints unchanged appears sensible in view of the lifetime of one generation.

In the absence of uncertainty, problems of dynamic control under constraints can be addressed applying the viability theory (Aubin [4]). An essential feature of the deterministic viability theory is that all constraints must be satisfied at all date. The deterministic viability theory does not allow for trade-off between objectives or generations which guarantees intergenerational equity when constraints are static.

Several studies have applied the deterministic viable control method to the management of natural resources (Martinet and Doyen [43]) and, in particular, to fisheries management (Béné et al. [8], Chapel et al. [12], De Lara et al. [23], Eisenack et al. [29], Martinet and Doyen [43], Martinet et al. [44, 45]) . Béné et al. [8] illustrate the application of such tool to a fishery, in a deterministic framework, and determine irreversible over exploitation configuration of the renewable resource. Martinet and Doyen [43] exhibit the sustainable technological configurations and policy options required to obtain a perennial production−consumption system based on the use of an exhaustible natural resource with the viability approach. De Lara et al. [23] also uses the viability control approach to define under which condition a

management strategy based on spawning stock biomass is sustainable. These studies show that, in deterministic cases, the viability approach seeks for a path belonging to acceptable corridors.

It is true that the maximization of a criterion results in smoother responses with respect to stocks and controls, than the binary sorting produced by minimal thresholds. Yet, boundaries are generally set on states and controls values as well, in the formulation of an optimization problem. Moreover, the stochastic viability theory allows to soften the deterministic approach by accepting constraint violations in few scenarios, and evaluates the probability of those that satisfy the requirements at all time. Thanks to the common value provided by this method —the viability probability— modifying threshold levels enables to evaluate trade-offs between objectives in their own units, given an initial state. As an example, it is possible to analyze how the maximum viability probability varies with respect to the different objectives a policymaker considers. Supposing a two-species setting, requiring a high yield over both species will result in a low viability probability, whereas a low catch thresholds should lead to a higher viability probability. Likewise, at a given viability probability level, setting a high minimal yield for one species will require lowering or increasing the harvest threshold of the other species, depending on the relationship linking them.

Studies applying the stochastic viable framework are less numerous (De Lara and Martinet [22], Doyen et al. [27, 28]). De Lara and Martinet [22] applies the stochastic viability approach to deal with a management issue in which two species exhibit technical interaction. In the Bay of Biscay Nephrops, trawlers fishery targets nephrops and gets juvenile hakes as a bycatch. As a result catched hakes will never become neither biologically mature nor economically available for the Hake fishery. Using the stochastic viability approach, the authors of this article succeed in defining an optimal management rule which addresses conflicting objectives and accounts for risk and resource dynamics.

The robust viability approach (De Lara and Doyen [21]) is an extreme case of the stochastic one. It seeks all the evolutions of a dynamical system that satisfy, at all times, given objectives, despite of uncertainty. This means constraints must be guaranteed with probability of one. Applied to the context of fisheries resources, species production and preservation requirements should be guaranteed whatever the scenarios of uncertainties affecting the dy-

namics. In contrast to the stochastic framework, the robust framework does not require to assign probabilistic assumptions to uncertainty scenarios, as failure or success with respect to scenarios are the only options. Thus, as in the deterministic approach, the robust viability theory allows no trade-offs between pursued objectives or time periods: all constraints must be satisfied for all times, whatever the uncertainties.

Applying such processing of uncertainty in the management of fisheries — a complete elimination of risk — may lead to pointless economic losses for society if the probability of catastrophic events is low. Yet, it can be adapted in common situations where very little information is available on the distribution of uncertainty. When a prior distribution cannot be derived as no historical biomass time series are available, the robust approach can be viewed as convenient.

Very few studies have undertaken a robust approach to these issues (Béné and Doyen [7]).

In what follows, we apply the stochastic viability theory to a discrete-time two-species dynamical model. Uncertainties take the form of environmental disturbances affecting each species growth factors, and are assumed to take their values in a given set. These disturbances may account for environmental randomness or a misspecification of population dynamics. Constraints are imposed for each species: a minimum safe biomass level, usually identified by biologists, and a minimum required harvesting level assumed to ensure economic needs. Our aim is to explore the consistency of various production levels with fixed preservation objectives and fisheries dynamics subject to uncertainty. In this perspective, we proceed to a numerical application of the stochastic viability approach to the Peruvian anchovy-hake upwelling ecosystem between 1971 and 1981, which dynamics is subject to uncertainty. Starting from an initial biomass couple, we provide the maximum viability probability of satisfying various levels of minimal catch thresholds, given fixed minimal biomass thresholds. After exposing our assumptions on the uncertainty sets and on the probability distribution of uncertainty scenarios (we consider different uncertainty sets and probability densities to appraise the sensitivity of our results to these hypothesis), we exhibit the trade-offs between various levels of production requirements of each species, respectively.

The data that is available on the the Peruvian hake-anchovy upwelling ecosystem is provided by *El Instituto del Mar del PerÃº* (IMARPE). Biomass time series are limited to

ten points per species. Thereby, it is not possible to accurately specify stochastic processes that affect this ecosystem. Thus, the probability distribution of uncertainties that we assume in our application is arbitrary. Our stochastic viability analysis of the harvested Peruvian anchovy-hake upwelling ecosystem is undertaken to illustrate how this approach deals with ecosystem management issues.

Content of the few data with which we must deal, we conduct, in a second stage, a robust viability analysis of this same ecosystem model under risk and constraints, focusing on the concept of robust viability kernel. That is, the set of initial biomasses for which their exists at least one effort strategy such that all constraints are satisfied at all times, whatever the uncertainty scenario. Thus, starting from a robust viable biomass couple, it is possible to drive the system on a sustainable path along which catches and biomasses stand above production and biological minimums, despite uncertainties. Reducing uncertainties to zero amounts to dressing the problem as deterministic. We proceed to a comparison of deterministic and robust viable kernels to shade light on the distance between the outcomes of these two extreme approaches: ignoring uncertainty vs. hedge against any risk.

We do not advocate the robust viability approach as a suitable decision tool for fishery management, since it involves economic costs for society, that are not justified when no catastrophic or irreversible events are expected, or when their likeliness is low. Our aim is to emphasize the impact of accounting for uncertainty on management possibilities that arise from a same methodology. It is also an opportunity to emphasize the different analysis and the wide range of information that can be derived from the viability framework to support decision making in the sustainable management of fisheries.

3.2 The stochastic viability approach

In this section, we first present a class of generic harvested nonlinear ecosystem models with uncertainty and the associated stainability constraints. Next, we expose how the stochastic viability theory provides the maximum viability probability of satisfying a set of constraints, given a dynamic system under uncertainty. We define the concept of *ecosystem viable yields under risk*. At last, we proceed to a numerical application of the stochastic viability analysis

to the Peruvian anchovy–hake fisheries between 1971 and 1981. Within this application, we illustrate the trade-offs between the different management objectives pursued. We also emphasize the influence of the assumption on the probability distribution of uncertainties, on the likeliness of achieving policy goals.

3.2.1 A generic ecosystem model with uncertainty and the associated sustainability constraints

To avoid technical difficulties and ease conceptual understanding, we consider a discrete-time dynamic model, with two species, each targeted by a specific fleet.[7] Each species is described by its biomass: the two-dimensional state vector (y, z) represents the biomass of both species. The two-dimensional control vector (v_y, v_z) comprises the harvesting effort for each species, respectively, each lying in $[0, 1]$. Two terms ε_y and ε_z correspond to uncertainties affecting each species, respectively. The discrete-time control dynamical system we consider is given by

$$\left\{ \begin{array}{rcl} y(t+1) & = & y(t)\mathcal{R}_y\big(y(t), z(t), \varepsilon_y(t)\big)\big(1 - v_y(t)\big) \ , \\ z(t+1) & = & z(t)\mathcal{R}_z\big(y(t), z(t), \varepsilon_z(t)\big)\big(1 - v_z(t)\big) \ , \end{array} \right. \tag{3.1}$$

where t stands for time (typically, periods are years), and ranges from the initial time t_0 to the time horizon T. The two functions $\mathcal{R}_y : \mathbb{R}^3 \to \mathbb{R}$ and $\mathcal{R}_z : \mathbb{R}^3 \to \mathbb{R}$ represent biological *growth factors*, and are supposed to be continuous.

The property that the growth factor $\mathcal{R}_y(y, z, \varepsilon_y)$ of species y depends on the other species biomass z (and vice versa) captures ecosystem features of species interactions. Furthermore, these interactions are complicated by uncertainties ε_y and ε_z. After two periods, $\varepsilon_y(t)$ indirectly impacts $z(t+2)$ through $y(t+1)$, so that both disturbances affect both species. According to the nature of the interaction between y and z, uncertainties affecting one of the species will constitute lagged positive or negative externalities for the other species. Catches are given by $v_y y \mathcal{R}_y\big(y, z, \varepsilon_y\big)$ and $v_z z \mathcal{R}_z\big(y, z, \varepsilon_z\big)$ (measured in biomass).

This model is generic in that no explicit or analytic assumptions are made on how the growth factors $\mathcal{R}_y$ and $\mathcal{R}_z$ indeed depend upon both biomasses (y, z) and upon the uncertainties $\big(\varepsilon_y, \varepsilon_z\big)$, except continuity.

[7] This approach can be easily extended to more than two species in interaction.

Uncertainties $(\varepsilon_y(t), \varepsilon_z(t))$ in (3.1) are assumed to take their values in a known two-dimensional set:

$$(\varepsilon_y(t), \varepsilon_z(t)) \in \mathbb{S}(t) \subset \mathbb{R}^2. \tag{3.2}$$

An uncertainty *scenario* is defined as a sequence of length $T - t_0$ of uncertainty couples:

$$\big(\varepsilon_y(\cdot), \varepsilon_z(\cdot)\big) = ((\varepsilon_y(t_0), \varepsilon_z(t_0)), \ldots, (\varepsilon_y(T-1), \varepsilon_z(T-1))) \in \prod_{t=t_0}^{T-1} \mathbb{S}(t). \tag{3.3}$$

Now, we propose to define *sustainability* as the ability to respect preservation and production minimal levels for all times, building upon the original approach of Béné et al. [8]. For this purpose, we consider:

- on the one hand, *minimal biomass levels* $y^\flat \geq 0$, $z^\flat \geq 0$, one for each species,
- on the other hand, *minimal catch levels* $Y^\flat \geq 0$, $Z^\flat \geq 0$, one for each species.

One will have noticed that our economic constraints — minimal catch levels — do not account for the production costs faced by each fishery, nor for the ex-vessel price of the targeted species. The fisheries' productive structure and a demand model could be developed to complete our theoretical framework. Yet, our analysis fits into the scenario where a policymaker knows what are the required production levels that meet social and economic needs of the community he heads, and comes to us in order to identify the feasibility of these management objectives.

3.2.2 Viable scenarios

The set of scenarios is denoted by

$$\Omega = \prod_{t=t_0}^{T-1} \mathbb{S}(t)\ . \tag{3.4}$$

We suppose that Ω is equipped with a probability distribution $\mathbb{P}$ such that $(\varepsilon_y(t), \varepsilon_z(t))$ are independently distributed w.r.t. time t. This restrictive assumption is a key ingredient to obtain a dynamic programming equation with state $\big(y, z\big)$ in §3.2.3.

An effort *strategy* γ is a sequence of mappings from biomasses towards efforts as follows

$$\gamma = \{\gamma_t\}_{t=t_0,\ldots,T-1}, \quad \text{with} \quad \gamma_t : \mathbb{R}^2 \to [0,1]^2\ . \tag{3.5}$$

Given $\big(y_0, z_0\big)$, an effort strategy γ as in (3.5) and the dynamic model (3.1) jointly produce state paths by the initial state $\big(y(t_0), z(t_0)\big) = \big(y_0, z_0\big)$ and the closed-loop dynamics

$$\left\{ \begin{array}{rcl} y(t+1) & = & y(t)\mathcal{R}_y\big(y(t), z(t), \varepsilon_y(t)\big)\big(1 - \gamma_t(y(t), z(t))\big) \ , \\ z(t+1) & = & z(t)\mathcal{R}_z\big(y(t), z(t), \varepsilon_z(t)\big)\big(1 - \gamma_t(y(t), z(t))\big) \ , \end{array} \right. \tag{3.6}$$

and control paths by

$$(v_y(t), v_z(t)) = \gamma_t(y(t), z(t)) \quad t = t_0, \ldots, T-1 \ . \tag{3.7}$$

Notice that, as in (3.7), controls $(v_y(t), v_z(t))$ are determined by constantly adapting to the state $(y(t), z(t))$ of the system, itself affected by uncertainties $(\varepsilon_y(t-1), \varepsilon_z(t-1))$.

We denote $\Omega_{\gamma, t_0, (y_0, z_0)}$ the subset of scenarios $\big(\varepsilon_y(\cdot), \varepsilon_z(\cdot)\big)$ associated to $\big(y(t_0), z(t_0)\big) = \big(y_0, z_0\big)$, for which

- preservation (minimal biomass levels): $\forall t = t_0, \ldots, T$,

$$y(t) \geq y^\flat \ , \quad z(t) \geq z^\flat \ , \tag{3.8}$$

- and production requirements (minimal catch levels): $\forall t = t_0, \ldots, T-1$,

$$v_y(t) y(t) \mathcal{R}_y\big(y(t), z(t), \varepsilon_y(t)\big) \geq Y^\flat \ , \quad v_z(t) z(t) \mathcal{R}_z\big(y(t), z(t), \varepsilon_z(t)\big) \geq Z^\flat \ , \tag{3.9}$$

hold true, and where $\big(y(t), z(t)\big)$ and $\big(v_y(t), v_z(t)\big)$ are given by (3.6) and (3.7). Hence, $\Omega_{\gamma, t_0, (y_0, z_0)}$ is the set of *viable scenarios* with respect to γ. That is, scenarios for which the biomasses and efforts trajectories defined by strategy γ are consistent with targeted ecological and economic goals.

3.2.3 Viability probability and ecosystem viable yields under risk

Given an initial state (y_0, z_0), initial time t_0 and horizon T, the *viability probability* is $\mathbb{P}[\Omega_{\gamma, t_0, (y_0, z_0)}]$, where $\mathbb{P}$ is the probability attached to the set Ω of scenarios in (3.4). It reflects the level of guarantee provided by the strategy γ over the specified objectives (3.8)–(3.9). The *maximal viability probability* is the supremum — $\sup_\gamma \mathbb{P}[\Omega_{\gamma, t_0, (y_0, z_0)}]$ — over all effort strategies γ. This quantity measures the best that can be achieved in terms of probability

to guarantee preservation and production objectives (3.8) and (3.9). There may exist several strategies able of achieving the maximal viability probability w.r.t. targeted objectives over the time horizon T.

Until now, objectives (3.8) and (3.9) have been treated as equally important. In what follows, we let production thresholds $Y^\flat$ and $Z^\flat$ in (3.9) vary and observe how the maximum viability probability evolves w.r.t. $Y^\flat$ and $Z^\flat$, while treating preservation objectives $y^\flat$ and $z^\flat$ in (3.8) as a binding reference. We do so for different assumptions on the probability distribution attached to the set Ω of scenarios in (3.4)

Freezing conservation rather than economic constraints is justified as follows: violating a minimal catch threshold in period t may cause hardship to fishermen in that period, whereas violating a minimal biomass threshold in period t can have longer term consequences on both types of objectives, as biological thresholds generally correspond to the biomass below which recruitment becomes substantially reduced (Beddington et al. [5]). The "overfished" stock may take several periods to recover, and catches will remain undermined alongside. Nonetheless, on an experimental basis, we also report the outcome of a trade-off analysis between preservation and production objectives.

Starting from a biomass couple (y_0, z_0), with $y_0 > y^\flat$, $z_0 > z^\flat$, we depict the maximum viability probability of satisfying various levels of catch thresholds $Y^\flat$ and $Z^\flat$, at given biomass minimal safety levels $y^\flat$, $z^\flat$. In this respect, the maximum viability probability $\sup_\gamma \mathbb{P}[\Omega_{\gamma,t_0,(y_0,z_0)}]$ can be expressed as a function of minimal catch levels $Y^\flat$ and $Z^\flat$, that we denote $\phi(Y^\flat, Z^\flat)$. Notice that the strategies γ that maximize the viability probability will most likely differ according to the catch thresholds at stake.

Ecosystem viable yields under risk at confidence level β, are the species minimal catch levels — here couples $(Y^\flat, Z^\flat)$ — that can be guaranteed with a probability at least equal to β: $\phi(Y^\flat, Z^\flat) \geq \beta$. By accepting a low degree of constraint violation, the stochastic viability approach allows for trade-offs between production constraints, and between time periods.

In § 3.2.2, we assumed that $(\varepsilon_y(t), \varepsilon_z(t))_{t=t_0,\dots,T-1}$ are independently distributed over time, under a probability distribution $\mathbb{P}$. Therefore, effort strategies maximizing the viability probability can be computed by means of a dynamic programming equation with state $\big(y, z\big)$ (De Lara and Doyen [21], Doyen and De Lara [26]), associated to dynamics (3.1) preser-

vation (3.8) and production (3.9) minimal thresholds. Because we investigate several levels of catch thresholds, and because the number of constraint dimensions we consider is high (i.e. two per species), resolution requires a powerful computational tool. To carry the numerical resolution, we discretized over state, control variables and catch thresholds (see §C.1).

3.2.4 Application to the anchovy–hake Couple in the Peruvian Upwelling Ecosystem (1971–1981)

We now apply the above stochastic viability framework to an analysis of the Peruvian anchovy–hake fisheries between 1971 and 1981. For this, we extend the model in De Lara et al. [24] to the stochastic case. We first develop our assumptions on the uncertainty set $\mathbb{S}(t)$ in (3.2) and on the probability distribution $\mathbb{P}$ that equips the set Ω of scenarios in (3.4). We then display how the maximum viability probability $\phi(Y^\flat, Z^\flat)$ defined in § 3.2.3 varies with respect to a range of hake-anchovy minimum catch thresholds.

3.2.5 Lotka-Volterra dynamical model with uncertainties

The Peruvian anchovy-hake system is modeled as a prey-predator system, where the anchovy growth rate is decreasing in the hake population. We describe this interaction by the following discrete-time Lotka-Volterra dynamical system

$$\begin{array}{rcl} y(t+1) & = & y(t) \overbrace{\left(\varepsilon_y(t) + R - \dfrac{R}{\kappa} y(t) - \alpha z(t)\right)}^{\mathcal{R}_y\left(y(t), z(t), \varepsilon_y(t)\right)} \left(1 - v_y(t)\right) \\ z(t+1) & = & z(t) \underbrace{\left(\varepsilon_z(t) + L + \beta y(t)\right)}_{\mathcal{R}_z\left(y(t), z(t), \varepsilon_z(t)\right)} \left(1 - v_z(t)\right), \end{array} \tag{3.10}$$

where $R > 1$, $0 < L < 1$, $\alpha > 0$, $\beta > 0$ and $\kappa = \frac{R}{R-1} K$, with $K > 0$ the carrying capacity for the prey. The variable y stands for anchovy biomass and z for hake biomass. The model (3.10) is a *decision model* the purpose of which is not to provide detailed biological "knowledge" about the Peruvian upwelling ecosystem, but rather to capture the essential features of the system in what concerns decision making.

Managing prey-predator interactions complicates the achievement of sustainability objectives as an "ever increasing biomass" of one species does not reflect an ecological improve-

ment with respect to the ecosystem. That is, beyond the trade-off entailed by production and preservation objectives, prey-predator interactions introduce a trade-off between fish stocks levels in the sense that the enhancement of a biomass necessarily takes place at the expense of the other.

The five parameters of the deterministic version of the Lotka-Volterra model (that is, with $\varepsilon_y(t) = 0$ and $\varepsilon_z(t) = 0$ in the dynamical system (3.10)) have been estimated in De Lara et al. [24], based on 11 yearly estimations of the Peruvian anchovy-hake biomasses and catches over the time period 1971–1981. Their values are given in Table 3.1.

Parameters	**Estimates**
R	2.25 $year^{-1}$
L	0.945 $year^{-1}$
κ	67113 10^3 $tons$
K	37285 10^3 $tons$
α	1.220 10^{-6} $tons^{-1}$
β	4.845 10^{-8} $tons^{-1}$

Table 3.1: Parameters of the Lotka-Volterra model (3.10)

El Instituto del Mar del PerÃº (IMARPE) has a precise knowledge of the local fishermen catch requirements and the biological dynamics of the Peruvian anchovy–hake ecosystem. We follow the thresholds recommended by IMARPE in IMARPE [35, 36] for the 1971–1981 time period:

- minimal biomasses: $y^\flat = 7,000$ ktons and $z^\flat = 200$ ktons in (3.8),
- minimal catches: $Y^\flat = 2,000$ ktons and $Z^\flat = 5$, ktons in (3.9).

3.2.6 Uncertainty sets and probability distributions of uncertainty scenarios

Here below, we specify the uncertainty sets $\mathbb{S}(t)$ in (3.2), in which the uncertainties $\varepsilon_y(t)$ and $\varepsilon_z(t)$ in (3.10) take their values. For the sake of simplicity, we consider finite, stationary

uncertainty sets $\mathbb{S} = \mathbb{S}(t)$, though this feature is not required for a dynamic programming equation to hold true. We compute the maximum viability probability of satisfying a range of anchovy–hake minimal yields, given safety biomass levels, and two different uncertainty sets $\mathbb{S}$. First, we form an uncertainty set $\mathbb{S}_E$ with empirical values. Second, we refine this set.

We equip the two retained sets $\mathbb{S}$ with two arbitrary probability distributions to appraise the sensitivity of our results to the assumption on the stochastic process inherent to the Peruvian anchovy–hake ecosystem. First, we test for the uniform distribution, and the set Ω of scenarios with the product probability, hence uniform too. The second probability distribution we consider accounts for the density of empirical uncertainties in $\mathbb{S}_E$.

Empirical uncertainties set and a refinement

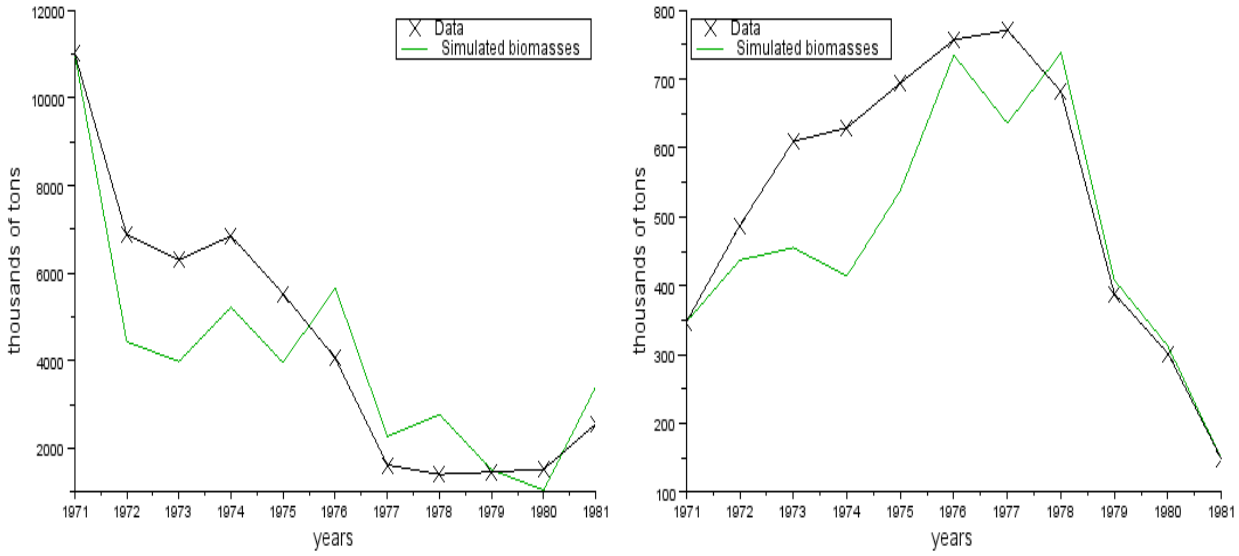
Figure 3.1 depicts the estimated biomasses of Peruvian anchovy and hake over the years 1971–1981 and the simulated biomasses with the deterministic version of the Lotka-Volterra model (that is, with $\varepsilon_y(t) = 0$ and $\varepsilon_z(t) = 0$ in the dynamical system (3.10)), given the harvesting efforts estimated over years 1971–1981 [8].

The time period 1971–1981 is denoted by $t = t_0, \ldots, T$, with $t_0 = 0$, and $T = 10$. Let $(\bar{y}(t), \bar{z}(t))_{t=t_0,\ldots,T}$ and $(\bar{v}_y(t), \bar{v}_z(t))_{t=t_0,\ldots,T-1}$ denote the estimated biomass and effort trajectories. We set $\bar{\varepsilon}_y(t)$ and $\bar{\varepsilon}_z(t)$ implicitly defined by

$$\left\{ \begin{array}{rcl} \bar{y}(t+1) & = & \bar{y}(t)\big(\bar{\varepsilon}_y(t) + R - \frac{R}{\kappa}\bar{y}(t) - \alpha\bar{z}(t)\big)\big(1 - \bar{v}_y(t)\big) \\ \bar{z}(t+1) & = & \bar{z}(t)\big(\bar{\varepsilon}_z(t) + L + \beta\bar{y}(t)\big)\big(1 - \bar{v}_z(t)\big), \end{array} \right. \tag{3.11}$$

so that (3.10) is satisfied. Figure 3.2 displays the points $\{(\bar{\varepsilon}_y(t), \bar{\varepsilon}_z(t)) | t = t_0, \ldots, T-1\}$, (there are 10 points as 1971 observations are used as starting points for simulating biomasses). We denote $\bar{\varepsilon}_y^{min} = \min_t \bar{\varepsilon}_y(t) = -0.25$, $\bar{\varepsilon}_y^{max} = \max_t \bar{\varepsilon}_y(t) = 1.54$, $\bar{\varepsilon}_z^{min} = \min_t \bar{\varepsilon}_z(t) = -0.38$ and $\bar{\varepsilon}_z^{max} = \max_t \bar{\varepsilon}_z(t) = 0.088$.

[8] Precisely, the biomass couple estimated in 1971 constitutes our starting state for simulating species biomasses. We plug this initial estimate of the anchovy–hake biomass couple and the 1971 catch values of each species in the deterministic version of the Lotka-Volterra model described in (3.10). This allows us to simulate the value of both biomasses in the following period. We renew this operation for each date until 1981, except that the current biomass couple we plug in the model is henceforth, the simulated one, while still considering the estimated catch couple of the current date.



(c) Anchovy

(d) Hake

Figure 3.1: Estimated and simulated biomasses over 1971–1981

We choose to base our analysis on the two following uncertainty sets:

- The *empirical uncertainties set*

$$\mathbb{S}_E = \{(\bar{\varepsilon}_y(t), \bar{\varepsilon}_z(t)) | t = t_0, \ldots, T-1\} \cup \{(0,0)\}, \tag{3.12}$$

  which is made of the ten empirical uncertainty couples (see diamonds in Figure 3.2) and the uncertainty couple $(\varepsilon_y, \varepsilon_z) = (0,0)$ (corresponding to the deterministic case).

- The *refined empirical uncertainties set*, $\mathbb{S}_{ER}$, made of 900 uncertainty couples produced by a 30×30 grid over the surface $[\bar{\varepsilon}_y^{min}, \bar{\varepsilon}_y^{max}] \times [\bar{\varepsilon}_z^{min}, \bar{\varepsilon}_z^{max}]$, including all the uncertainty couples of $\mathbb{S}_E$ (see the grid in Figure 3.3).

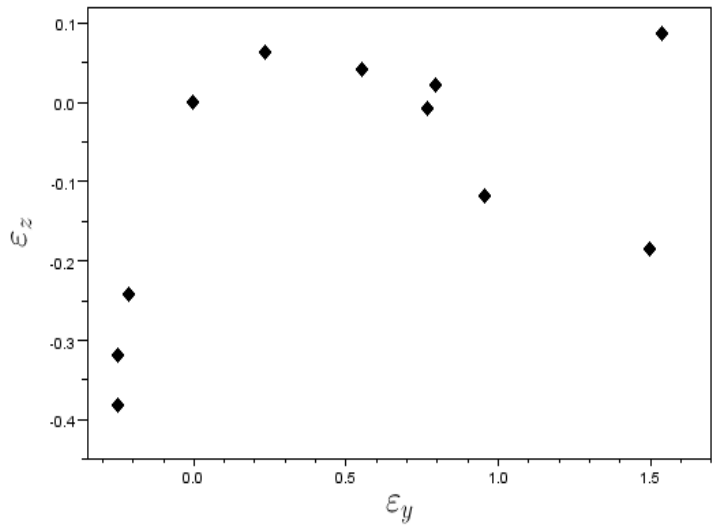


Figure 3.2: Empirical uncertainties $(\bar{\varepsilon}_y(t), \bar{\varepsilon}_z(t))_{t=t_0,\ldots,T-1}$ characterized in (3.11)

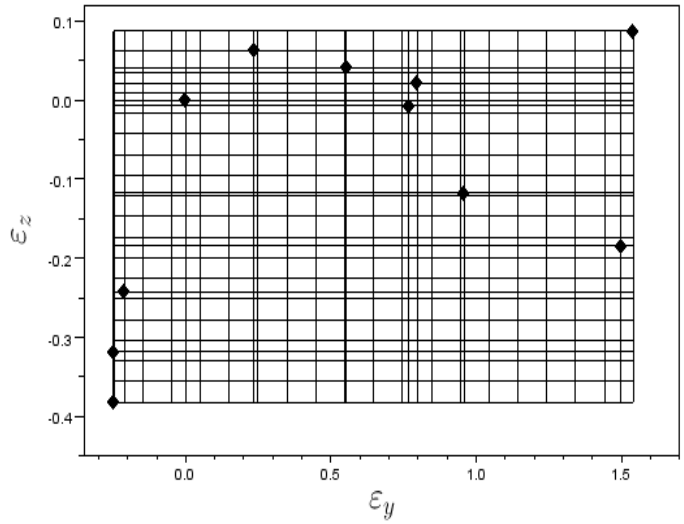


Figure 3.3: Uncertainty sets $\mathbb{S}_E$ (diamonds) and $\mathbb{S}_{ER}$ (grid)

The two probability distributions of uncertainties in $\mathbb{S}$ we test for are:

- We equip $\mathbb{S}_E$ and $\mathbb{S}_{ER}$ with the uniform distribution law, denoted $\mathbb{P}_u$.
- We equip $\mathbb{S}_E$ with a distribution $\mathbb{P}_{dd}$ (Figure 3.4), emulating the density of uncertainty couples in $\mathbb{S}_E$:

  $\mathbb{P}_{dd} = [0.005, 0.02, 0.02, 0.02, 0.23125, 0.005, 0.005, 0.23125, 0.23125, 0.23125]$.

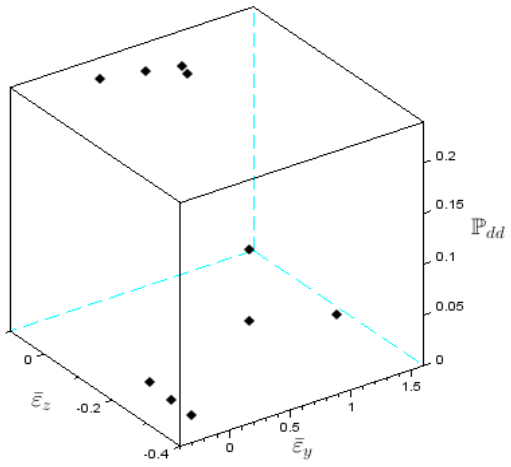


Figure 3.4: Density dependent probability distribution $\mathbb{P}_{dd}$ of uncertainties in $\mathbb{S}_E$

Both assumptions are arbitrary and considered for experimental purpose.

3.2.7 Maximum viability probability of satisfying production and preservation requirements in the Peruvian anchovy–hake fishery

We report how the maximum viability probability of satisfying a range of anchovy-hake minimal yields varies, for different uncertainty sets $\mathbb{S}$ in (3.2) and different probability distributions attached to set Ω of scenarios in (3.4). This allows us to observe how a change in the production requirement of one species affects that of the other species, at a given confidence level β. In complement, we report the maximum viability probabilities w.r.t. variations in one species' preservation constraint and the other species' production constraint, all else being equal.

These numerical analyzes are intended to shed light on the trade-offs between management objectives involved by the anchovy–hake fisheries in the Peruvian Upwelling ecosystem. The aim is also to examine how assumptions on uncertainty influences our outcomes. Note that, for all results displayed in this section, the time horizon chosen for simulations is T=10.

Ecosystem viable yields under risk with the uniform distribution of uncertainties

Figure 3.5 displays the maximum viability probability as functions of the minimal catch levels, $\phi_E(Y^\flat, Z^\flat)$ and $\phi_{ER}(Y^\flat, Z^\flat)$, obtained given uncertainty sets $\mathbb{S}_E$ and $\mathbb{S}_{ER}$, both equipped with the uniform distribution, $\mathbb{P}_u$. Alongside, minimal biomass thresholds are kept constant, and set equal to IMARPE's recommendations for the anchovy–hake couple in the Peruvian upwelling ecosystem: $y^\flat = 7,000$ kt and $z^\flat = 200$ kt (see §3.2.5).

We consider 15 equidistant hake minimal catch thresholds, $Z^\flat$, ranging from 2 to 250 kt, against 15 equidistant anchovy minimal catch thresholds, $Y^\flat$, ranging from 1,000 to 10,000 kt. For each minimal yield couple, $(Y^\flat, Z^\flat)$, the maximum viability probability is computed given the initial biomass couple estimated in 1971, namely (y_0, z_0) =(11,019 kt, 347 kt). This initial state of the fishery complies with IMPARPE's conservation requirements.

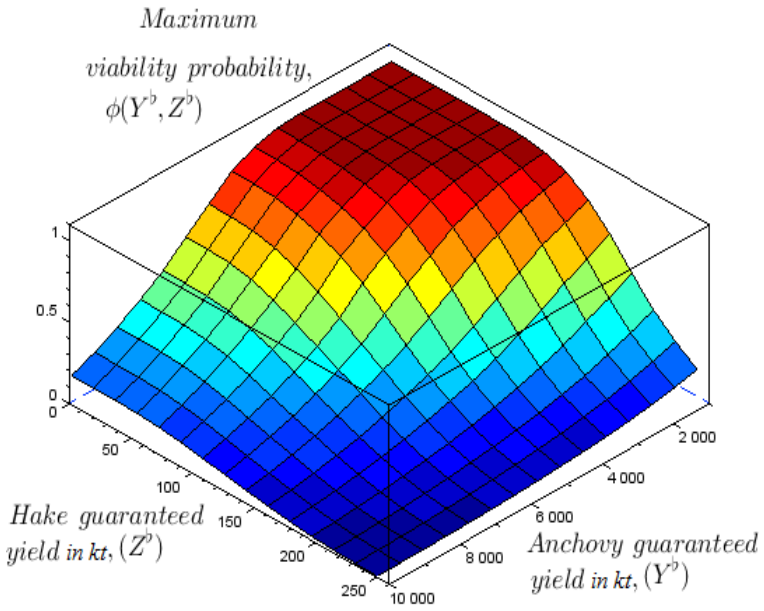
According to Figure 3.5, wether the dynamic system is exposed to $Card(\Omega_E) = (11)^{10}$ or $Card(\Omega_{ER}) = (900)^{10}$ scenarios, the maximum viability probability of guaranteeing various catch couples, $(Y^\flat, Z^\flat)$, at given minimal biomass levels, $(y^\flat, z^\flat)$, is sensibly the same. As we assume that scenarios are equipped with a uniform probability distribution, the smaller is the set of scenarios tested, the higher is the probability attached to a single scenario, and vise versa. This breakdown of the weight attached to each scenario seems to make the maximum probability of satisfying targeted objectives little sensitive to the number of disturbances included in $\mathbb{S}_E$ and $\mathbb{S}_{ER}$.

The maximum viability probability varies quite smoothly w.r.t. minimal yield requirements. The lower are the minimal catch thresholds, the higher it is. Indeed, poor objectives are easy to hold. One could have thought that the higher is the minimal yield requirement of hake, $Z^\flat$, the higher can be the anchovy requirement, $Y^\flat$, due to the decrease in predator population. Yet, this intuition does not work as high hake yields go along with a high stock level.

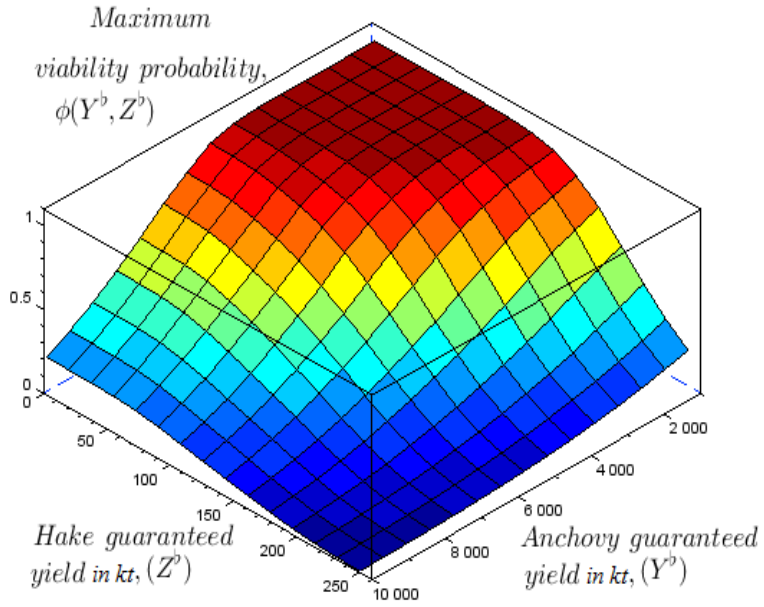
The couple of ecosystem viable yields for which both catch requirement are the highest at the 90% level is worth $(Y^\flat, Z^\flat)$ = (4214 kt, 108 kt). For any minimal yield couple within this set, there exists an effort strategy such that it can be harvested with 90% chance at each date over a period of length $T = 10$. Recall that the minimal catch thresholds recommended by IMARPE for the anchovy–hake couple in the Peruvian upwelling ecosystem are worth

$Y^\flat = 2,000$ ktons and $Z^\flat = 5$ ktons in (3.9). Thereby, on the basis of our assumptions on the set of uncertainties and their probability distribution, IMARPE's production requirements appear modest.

However, in facts, IMARPE's minimal biomass threshold for anchovy was violated early as 1972. Thus, well before 1981, IMARPE's anchovy minimal catch threshold could no longer be satisfied. This observation reinforces the need to implement caps on catches and to monitoring them. Ideally, the quota regulatory tool should lead to the full exploitation of the production potential of a resource. At the least, quotas can provide benchmarks that help prevent the violation of biological reference points, essential to the survival of a bioeconomic system.



(a) $\mathbb{S}_E$



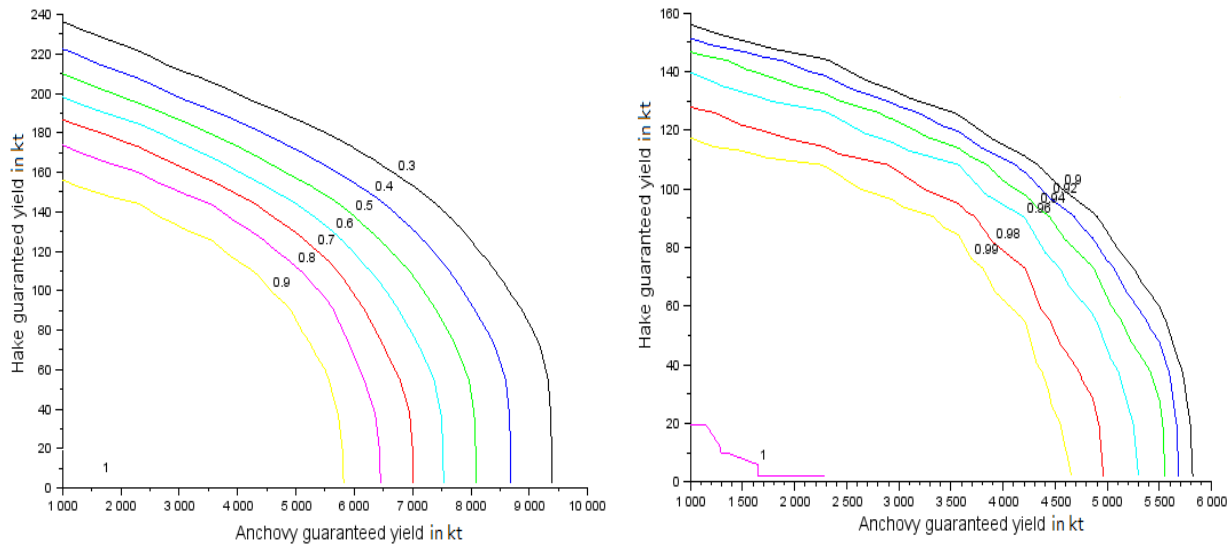
(b) $\mathbb{S}_{ER}$

Figure 3.5: Maximal viability probability w.r.t. various minimal catch levels of anchovy and hake, and uncertainty sets $\mathbb{S}_E$ and $\mathbb{S}_{ER}$ equipped with the uniform probability distribution $\mathbb{P}_u$

The iso-curves in 3.6 are obtained by operating a transversal cut in Figure 3.5(a). They depict the trade-off between both species catch entailed by the prey-predator relationship:

the increase in a species minimal yield can only take place at the expense of the other species' minimal catch threshold. Starting from 90% level in Figure 3.6(a), the maximum viability probability decreases by 10% steps quite evenly w.r.t. minimal catch thresholds. In Figure 3.6(b), the decline in minimal catches, between the 90% and 99% level, is in proportion more substantial. What is most surprising is the drop in minimal production requirements between the 99% and the 100% level of guarantee. The production goals that can be contemplated considerably changes when admitting a 1% risk or not at all.

Given this outcome, the decision maker has much interest to take at least a 1% risk. Indeed, it is a low risk taking which provides him with much flexibility regarding the minimum production goals that he can target. Supposing our assumptions on uncertainty are sound, avoid taking any risk would probability inflict unnecessary economic losses to society.



(a) $0.3 \leq \phi(Y^\flat, Z^\flat) \leq 1$

(b) $0.9 \leq \phi(Y^\flat, Z^\flat) \leq 1$

Figure 3.6: Iso-probability curves w.r.t. $\mathbb{S}_E$, equipped with the uniform probability distribution

Ecosystem viable yields under risk with a density dependent distribution of uncertainties

Figure 3.7 displays the maximum viability probability, $\phi_E(Y^\flat, Z^\flat)$ as a function of minimal catch levels, obtained with the uncertainty set $\mathbb{S}_E$, equipped with the density dependent

probability distribution, $\mathbb{P}_{dd}$. The maximal viability probability is computed for the same initial state and range of minimal hake and anchovy catch thresholds as in Figure 3.5. Here again, minimal biomass thresholds are kept equal to IMARPE's biological recommendations.

The highest ecosystem viable yield couple at the 90% level is worth $(Y^\flat, Z^\flat)$ = (6,820 kt, 131 kt). In comparison with Figure 3.5 the distribution $\mathbb{P}_{dd}$ of uncertainties supports greater risk-taking. Supposing the regulator knows that (y_0, z_0) =(11,019 kt, 347 kt) and sets the total allowable catch of each species in accordance with the highest catch couple that can be guaranteed at the 90% level, than harvesting effort would increase compared to the exploitation rates supported by the uniform distribution, $\mathbb{P}_u$. To fit into a more operational approach, for each acceptable initial biomass couple, we could seek for the maximum catch requirement that can be guaranteed with a high probability over horizon T. Such mapping would provide support to define harvesting quota levels for each species, that secure minimal biomass levels. In fact, on this basis, a constant harvesting strategy could be recommended, where at each date, each species' quota level is equal to the highest ecosystem viable yield couple that can be guaranteed at the chosen confidence level β.

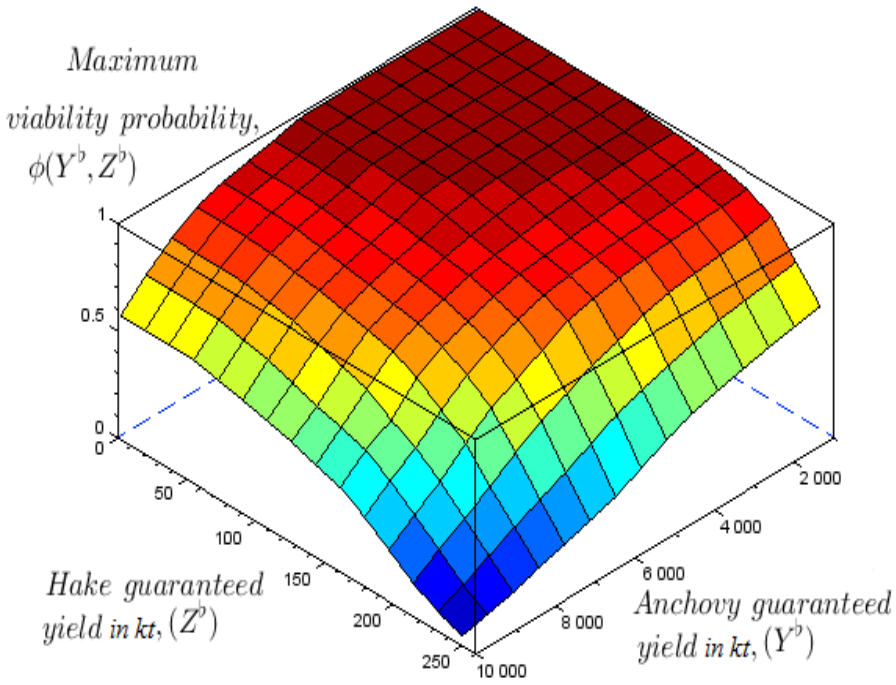


Figure 3.7: Maximal viability probability w.r.t. various minimal catch levels of anchovy and hake, the uncertainty set $\mathbb{S}_E$ and probability distributions $\mathbb{P}_{dd}$

To resume, this second assumption on the probability distribution of uncertainties substantially changes the room for risk-taking in our case study. Content of the information available, we favor the uniform distribution because it puts all empirical uncertainties on the same footing. Assigning less weight to relatively more outlying empirical uncertainties over 10 points does not fall within a precautionary approach.

Trade-off analysis between production and preservation requirements

Figure 3.8 displays the maximum viability probability, $\phi_E(y^\flat, Z^\flat)$ as a function of anchovy minimal biomasses and hake minimal yields, obtained for the uncertainty set $\mathbb{S}_E$, equipped with the uniform probability distribution, $\mathbb{P}_u$. We test for anchovy minimal biomasses within the range $y^\flat \in$[3,000 kt; 10,000 kt], and hake minimal yields within the range $Z^\flat \in$[0 kt; 250 kt]. On the other hand, the hake minimal biomass, $z^\flat$, and anchovy minimal yield, $Y^\flat$, thresholds are kept equal to IMARPE's recommendations.

The outcome of this trade-off analysis is quite surprising. It seems that whatever the

conservation objective set on the anchovy stock, $y^\flat$, (beyond 3000 kt), the maximum viability probability only varies w.r.t. $Z^\flat$. As can be expected, $\phi_E(y^\flat, Z^\flat)$ is decreasing in $Z^\flat$. The explanation to this absence of trade-off is that the anchovy minimal yield $Y^\flat$ =2,000 kt must be satisfied at all times, while, given the set of uncertainty scenarios retained, there apparently always exists an effort strategy that satisfies, at all times, the range of anchovy minimal biomass threshold we consider, if this catch constraint is too. If it was not the case, the associated maximum viability probability would be zero.

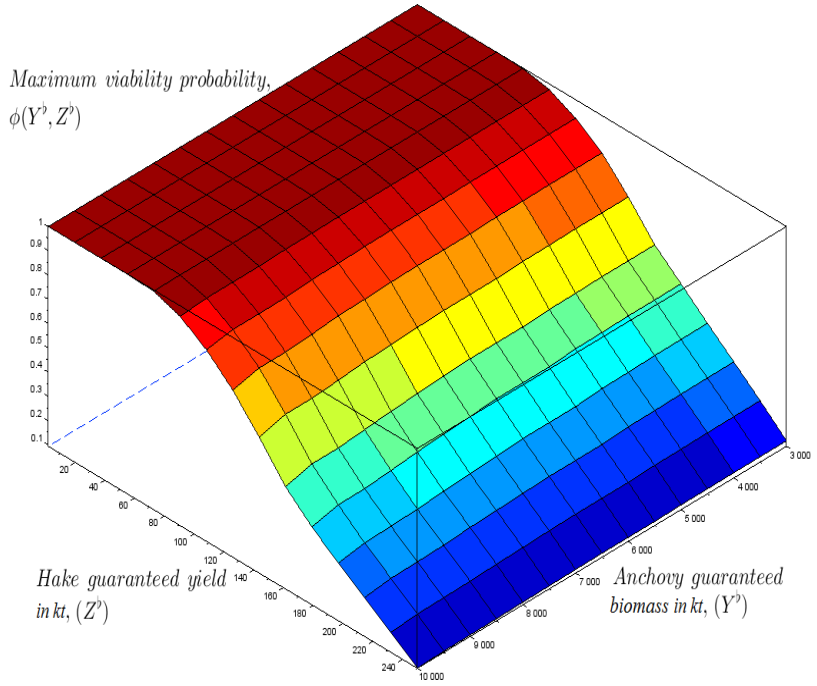


Figure 3.8: Maximal viability probability w.r.t. minimal biomass levels of anchovy and minimal catch levels of hake, the uncertainty set $\mathbb{S}_E$ and the uniform probability distribution

3.3 The robust viability approach

In this section we present the robust viability theory, focusing on the concept of robust viability kernel. We then derive the robust viability kernel numerically, in the case of the anchovy–hake Peruvian fisheries under uncertainty, and compare it to that obtained in the deterministic case.

Drawing on the concept of kernel, in contrast to the analysis carried in section 3.2, initial

states now vary, whereas the set constraints and scenarios are fixed. Wherever a fishery stands, the robust viability kernel enables to predict whether economic and conservation objectives can be satisfied, at all times, over a time span. By delineating the set of *admissible* biomasses, the robust viability kernel also shades light on the set of states to be avoided, which information may be of relevance for the regulator.

The deterministic and robust viability approach meet in that both do not allow for trade-offs between objectives or time periods. The definition of the deterministic and robust viability kernel given below builds upon the same generic ecosystem model under uncertainty in (3.1), and the same approach to sustainability constraints, introduced in §3.2.1.

As exposed earlier, we do not advocate the robust viability approach as a fully suitable decision tool for fishery management, since resulting harvesting strategies may impose economic losses to society, which are not necessarily justified when the ecological risk is not devastating or irreversible. Indeed, recall from Figure 3.6, that the minimal yields guaranteed with a probability one are relatively much lower than when admitting little risk. Our aim is to display the management possibilities that the viability theory allows for in two extreme cases:

- no uncertainty,
- full uncertainty.

It is a manner of emphasizing to what extent does accounting for risk modifies management recommendations.

3.3.1 The robust viability kernel

The *robust viability kernel* $\mathbb{V}\text{iab}^R(t_0)$ (De Lara and Doyen [21]) is the set of initial states $(y(t_0), z(t_0))$ for which there exists a control strategy γ as in (3.5), such that, for any uncertainty scenario $(\varepsilon_y(\cdot), \varepsilon_z(\cdot)) \in \prod_{t=t_0}^{T-1} \mathbb{S}(t)$ in (3.4), the state path $\{(y(t), z(t))\}_{t=t_0,\dots,T}$ as in (3.6), and control path $\{(v_y(t), v_z(t))\}_{t=t_0,\dots,T-1}$ as in (3.7), satisfy:

- preservation (minimal biomass levels), $\forall t = t_0, \dots, T$,

$$y(t) \geq y^\flat \ , \quad z(t) \geq z^\flat \ ,$$

- production requirements (minimal catch levels), $\forall t = t_0, \ldots, T-1$,

$$v_y(t)y(t)\mathcal{R}_y\big(y(t), z(t), \varepsilon_y(t)\big) \geq Y^\flat \ , \quad v_z(t)z(t)\mathcal{R}_z\big(y(t), z(t), \varepsilon_z(t)\big) \geq Z^\flat .$$

States belonging to the robust viability kernel are also named *robust viable states*. Characterizing robust viable states makes it possible to test whether or not minimal biomass and catch levels can be guaranteed for all times of a time horizon, despite of uncertainty. Here, *guaranteed* means that biomasses and catches never fall below the minimal thresholds as in the inequalities (3.8) and (3.9).

The robust viability kernel can be computed numerically by means of a dynamic programming equation associated with dynamics (3.1), state constraints (3.8) and control constraints (3.9) (see §C.3 in Appendix and De Lara and Doyen [21]).

3.3.2 The deterministic viability kernel

The deterministic version of the framework exposed in §3.3.1 corresponds to the case where the uncertainties $(\varepsilon_y(t), \varepsilon_z(t)) = (0, 0)$ for all $t = t_0, \ldots, T-1$, that is, the uncertainty sets in (3.2) are reduced to the singleton $\mathbb{S}(t) = \{(0, 0)\}$. In that case, the robust viability kernel coincides with the so-called *viability kernel* $\mathbb{V}\mathrm{iab}(t_0)$ (Aubin [4]), defined in §C.2 in Appendix.

The following Proposition 5 gives an analytical expression of the deterministic viability kernel under conditions on the guaranteed levels in (3.8) and (3.9). The proof, adapted from De Lara et al. [24], is given in §C.2 in Appendix.

Proposition 5 *If the minimal biomass thresholds $y^\flat$, $z^\flat$ and catch thresholds $Y^\flat$, $Z^\flat$ are such that*

$$y^\flat\mathcal{R}_y\big(y^\flat, z^\flat, 0\big) - y^\flat \geq Y^\flat \ \text{ and } \ z^\flat\mathcal{R}_z\big(y^\flat, z^\flat, 0\big) - z^\flat \geq Z^\flat \ , \tag{3.13}$$

for $T \geq t_0 + 2$, the deterministic viability kernel is given by

$$\mathbb{V}\mathrm{iab}(t_0) = \Big\{ \ (y, z) \in \mathbb{R}^2_+ \mid y \geq y^\flat, z \geq z^\flat, y\mathcal{R}_y\big(y, z, 0\big) - y^\flat \geq Y^\flat, z\mathcal{R}_z\big(y, z, 0\big) - z^\flat \geq Z^\flat \ \Big\} \ . \tag{3.14}$$

The interpretation of conditions (3.13) is as follows. A the point $(y^\flat, z^\flat)$ of minimum biomass thresholds, the surplus $y^\flat \mathcal{R}_y\big(y^\flat, z^\flat, 0\big) - y^\flat \geq Y^\flat$ and $z^\flat \mathcal{R}_z\big(y^\flat, z^\flat, 0\big) - z^\flat \geq Z^\flat$ are at least equal to the minimum catch thresholds $Y^\flat$ and $Z^\flat$, respectively. Notice that the expression (3.14) does not depend on the horizon T (where $T \geq t_0 + 2$): for any initial state in the deterministic viability kernel $\mathbb{V}\mathrm{iab}(t_0)$, there exists a strategy such that the constraints (3.8) and (3.9) are satisfied for all times from t_0 to infinity.

3.3.3 Application to the anchovy-hake Couple in the Peruvian Upwelling Ecosystem (1971–1981)

Here again, we apply the robust viability analysis to the Peruvian anchovy–hake fisheries. We consider the same two-species prey-predator dynamical model, and preservation and production requirements supported by IMPARE, as in §3.2.5.

We compute the robust viability kernel numerically, testing different assumptions on the uncertainty sets $\mathbb{S}(t)$ in (3.2), to appraise the sensitivity of the size and content of the robust viability kernel with respect to the set of uncertainty scenarios.

3.3.4 Choice of uncertainty sets

We now specify the uncertainty sets $\mathbb{S}(t)$ in (3.2), in which the uncertainties $\varepsilon_y(t)$ and $\varepsilon_z(t)$ in (3.10) take their values. Here again, we choose to focus on stationary uncertainty sets $\mathbb{S} = \mathbb{S}(t)$.

First, we consider the same uncertainty sets, $\mathbb{S}_E$ and $\mathbb{S}_{ER}$ in §3.2.6, derived from the empirical uncertainties' value. Then, we identify and only consider extreme uncertainties in §3.3.4, producing worst-case scenarios. In §3.3.5, we will explain these choices in light of the corresponding robust viability kernels.

Uncertainty sets reduced to extreme values

Through numerical simulations, we found that the set of robust viable states is sensitive to few extreme points of the uncertainty set $\mathbb{S}_{ER}$. This is why we consider the following two uncertainty sets, $\mathbb{S}_M$ and $\mathbb{S}_H$.

- The uncertainty set $\mathbb{S}_M$ is composed of two extreme uncertainty couples taken from the set $\mathbb{S}_{ER}$:

$$\mathbb{S}_M = \{(\bar{\varepsilon}_y^{min}, \bar{\varepsilon}_z^{min}), (\bar{\varepsilon}_y^{min}, \bar{\varepsilon}_z^{max})\} \subset \mathbb{S}_{ER} \, . \tag{3.15}$$

- The uncertainty set $\mathbb{S}_H$ is obtained by increasing the values in $\mathbb{S}_M$ increased by 20%:

$$\mathbb{S}_H = 1.2 * \mathbb{S}_M \, . \tag{3.16}$$

The uncertainty couple $(\bar{\varepsilon}_y^{min}, \bar{\varepsilon}_z^{min})$ corresponds to low growth factor for both species, whereas $(\bar{\varepsilon}_y^{min}, \bar{\varepsilon}_z^{max})$ affects negatively the prey growth and positively the predator growth.

3.3.5 Discussion on the viability kernels

Now, we present the robust viability kernels computed on the basis of the following ingredients: the dynamical model of harvested ecosystem in the Peruvian upwelling and sustainability constraints in §3.2.5, the various uncertainty sets (including the deterministic case) laid out in §3.3.4, a time horizon of uncertainty scenarios $T = 10$. In §3.3.5, we compare the viability kernels: the deterministic, the robust resulting from the uncertainty set $\mathbb{S}_E$ and that obtained from the uncertainty set $\mathbb{S}_{ER}$. In §3.3.5, we turn to the uncertainty sets built upon "extreme uncertainties" and we scrutinize how these sets impact the robust viability kernels.

Deterministic viability kernel, robust viability kernel and empirical uncertainties

Figure 3.9 displays the deterministic viability kernel and the robust viability kernels associated with dynamics (3.10), constraints (3.8) and (3.9), and with the uncertainty sets $\mathbb{S}_E$ and $\mathbb{S}_{ER}$, respectively. The horizontal and vertical lines represent the minimal biomass safety levels $y^\flat$ and $z^\flat$. In §C.3 in Appendix, we detail how the robust viability kernels are computed numerically, with the scientific software Scicoslab.

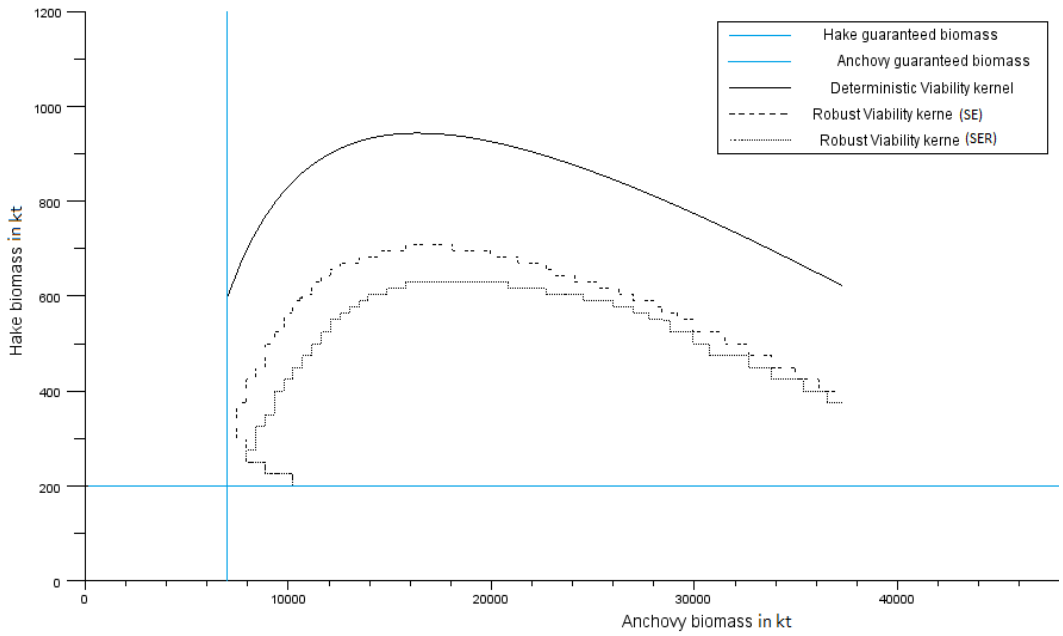


Figure 3.9: Deterministic and robust viability kernels for the uncertainty sets $\mathbb{S}_E$ and $\mathbb{S}_{ER}$

The deterministic viability kernel

Replacing the growth rates $\mathcal{R}_y$ and $\mathcal{R}_z$ in (3.14) by their expressions (3.10) yields the expression of the deterministic viability kernel:

$$\begin{aligned}\mathbb{V}\text{iab}(t_0) &= \Big\{ \ (y,z) \mid y \geq y^\flat, z \geq z^\flat, y\big(R - \tfrac{R}{\kappa}y - \alpha z\big) - y^\flat \geq Y^\flat, z\big(L + \beta y\big) - z^\flat \geq Z^\flat \ \Big\} \\ &= \Big\{ \ (y,z) \mid y \geq y^\flat, \tfrac{1}{\alpha}[R - \tfrac{R}{\kappa}y - \tfrac{y^\flat + Y^\flat}{y}] \geq z \geq \max\{\tfrac{z^\flat + Z^\flat}{L+\beta y}, z^\flat\} \ \Big\}. \end{aligned} \tag{3.17}$$

The humped shape of the upper frontier of the deterministic viability kernel in Figure 3.9 stems from the logistic dynamics of the anchovy stock. Indeed, from the expression of $\mathbb{V}\text{iab}(t_0)$ in (3.17), we deduce that the upper frontier is characterized by

$$\frac{1}{\alpha}[R - \frac{R}{\kappa}y - \frac{y^\flat + Y^\flat}{y}] = z \Leftrightarrow y\big(R - \frac{R}{\kappa}y - \alpha z\big) = Y^\flat + y^\flat \Leftrightarrow y\mathcal{R}_y(y,z,0) = Y^\flat + y^\flat \ .$$

Hence, before a tipping anchovy biomass level $\overline{y}(z) = \frac{\kappa(R-\alpha z)}{2R}$, the species' growth rate $\mathcal{R}_y(y,z,0)$ increases with y, whereas it decreases for y beyond $\overline{y}(z)$. In other words, for the hake population — the predator (z) — to reach high biomass levels the growth rate of the anchovy population — the prey (y) — must be sufficiently high. Thus the increasing slope

of the frontier of the deterministic viability kernel, until $\overline{y}(z) = \frac{\kappa(R-\alpha z)}{2R}$. Conversely, for the anchovy biomass to reach high levels, the hake biomass must be relatively low. Thus, the declining slope of the frontier of the deterministic viability kernel as the biomass y approaches K.

We conducted a sensitivity analysis of the deterministic viability kernel to the value of the Lotka-Volterra model parameters in §3.2.5. We find that a $\pm 10\%$ variation in any parameter slightly changes the size of the deterministic viability kernel, but not its overall shape, as long as the condition on minimal thresholds set in Proposition 5 is satisfied. Even in absence of harvesting, $(Y^\flat, Z^\flat) = (0, 0)$, the shape remains the same.

When the condition in Proposition 5 is violated, the deterministic viability kernel appears to be empty. This actually makes sense: in absence of harvesting, equation (3.13) in Proposition 5 gives:

$$\mathcal{R}_y\big(y^\flat, z^\flat, 0\big) - 1 \geq 0 \quad \text{and} \quad \mathcal{R}_z\big(y^\flat, z^\flat, 0\big) - 1 \geq 0. \tag{3.18}$$

On the other hand, setting harvesting efforts equal to zero in the dynamics in (3.10), writing (3.10) in the form of two differential equations, and setting them greater or equal to zero gives the same expressions in (3.18). Rewritten them yields:

$$y \geq \frac{(1-L)}{\beta} \quad \text{and} \quad z \leq \frac{1}{\alpha}[R - \frac{R}{\kappa}y - 1]. \tag{3.19}$$

Thus, (3.13) imposes no more than that at the point $(y^\flat, z^\flat)$ the growth rates of each species, respectively, is positive. As long as minimal biomass thresholds, $(y^\flat, z^\flat)$, satisfy inequalities in (3.19), the anchovy-hake ecosystem in the Peruvian Upwelling, as it is modeled, admits a non-empty viability kernel. The condition (3.19) corresponds to the condition of co-existence of both species in absence of harvesting. In presence of harvesting it becomes (3.17). Indeed, the viability kernel is delineates by the range of biomass couples for which the growth rate of each species is positive or null given minimal catch requirements.

Gap between the deterministic kernel and the robust ones

In Figure 3.9, we observe an important gap between the deterministic kernel and the robust ones. A share of the states identified as viable by the deterministic approach is in fact

excluded when uncertainty is taken into account. Indeed, there exists no effort strategy capable of guaranteeing preservation and production minima for biomass couples standing outside the robust kernels, given the chosen scenarios sets and time horizon. Furthermore, we cannot tell whether the effort strategies advocated by the deterministic approach for an initial biomass couple belonging to the robust kernels guarantee sustainability objectives over time in presence of uncertainty.

Figure 3.10 displays the deterministic viability kernel $\mathbb{V}\text{iab}(t_0)$, the robust viability kernel $\mathbb{V}\text{iab}_E^R(t_0)$ and stochastic viability kernels of confidence level 30%, 50%, 70% and 90% obtained for the uncertainty set $\mathbb{S}_E$ equipped with the uniform probability distribution $\mathbb{P}_u$. These stochastic viable states are the set of initial states $(y(t_0), z(t_0))$ for which there exists a control strategy γ as in (3.5), such that, for any uncertainty scenario $(\varepsilon_y(\cdot), \varepsilon_z(\cdot)) \in \prod_{t=t_0}^{T-1} \mathbb{S}(t)$ in (3.4), the state path $\{(y(t), z(t))\}_{t=t_0,\ldots,T}$ as in (3.6), and control path $\{(v_y(t), v_z(t))\}_{t=t_0,\ldots,T-1}$ as in (3.7), satisfy constraints (3.8) and (3.9) with a probability at least equal to β and $T = 10$.

As one can observe, the lower is the confidence level the wider is the stochastic viability kernel. Indeed, as a higher level of constraint violations is accepted, more biomass endowments are likely to comply with the lower level of expectation regarding sustainability constraints. The interest for plotting stochastic viable sets is it that it tells us that the determinist viability kernel comes close to the stochastic viability kernel of confidence level 70%. However, it is not possible to state whether, starting from a point in this set, the harvesting path advocated by the deterministic viability approach would guarantee targeted sustainability constraints with a minimum probability of 70%, since controls are computed regardless of disturbances.

Sensitivity of the robust viability kernel to uncertainty sets and to the time horizon

Since $\{(0, 0)\} \subset \mathbb{S}_E \subset \mathbb{S}_{ER}$, where the uncertainty sets $\mathbb{S}_E$ and $\mathbb{S}_{ER}$ are given in §3.2.6, we expect the corresponding robust and deterministic viability kernels to satisfy

$$\mathbb{V}\text{iab}_{ER}^R(t_0) \subset \mathbb{V}\text{iab}_E^R(t_0) \subset \mathbb{V}\text{iab}(t_0) \; . \tag{3.20}$$

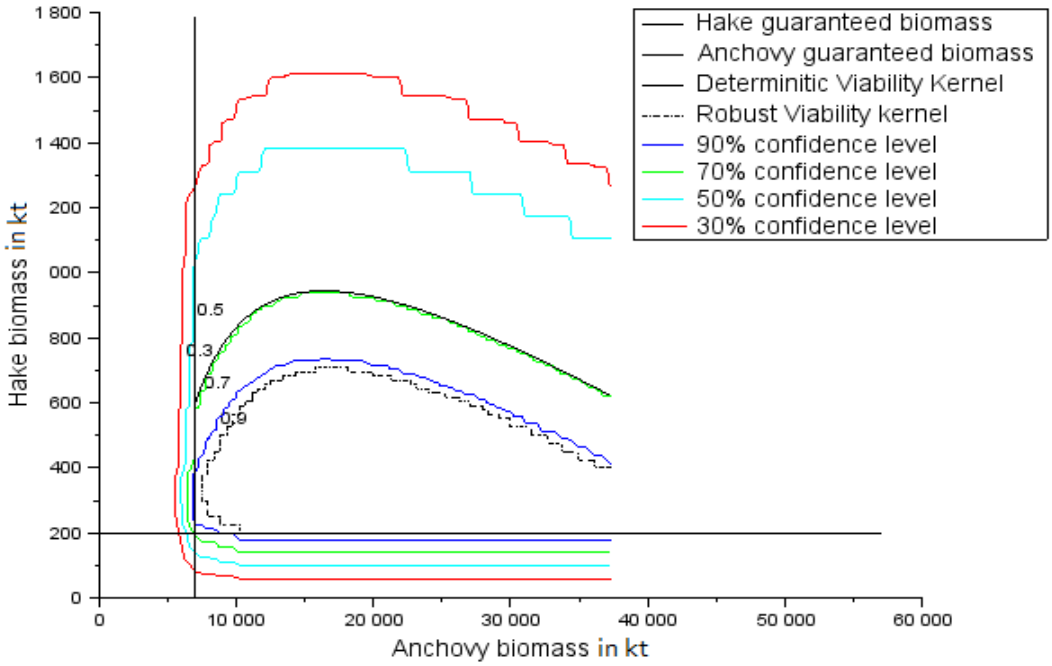


Figure 3.10: Deterministic, stochastic and robust viability kernels for the uncertainty set $\mathbb{S}_E$

We indeed observe strict inclusions in Figure 3.9. This confirms our initial guess that, by exposing the ecosystem dynamics to a denser set of scenarios $\mathbb{S}_{ER}$ instead of $\mathbb{S}_E$, fewer initial states should be likely to allow for an effort strategy guaranteeing all sustainability constraints at all times.

Besides, we look at the influence of uncertainties ε_y and ε_z in $\mathbb{S}_{ER}$, separately, on the set of robust viable states. When drawing only ε_y in $\mathbb{S}_{ER}$, (i.e. $\{\varepsilon_z(t) = 0 | t = t_0, \ldots, T-1\}$), the robust kernel is smaller then the deterministic one but greater than $\mathbb{V}\text{iab}^R_{ER}(t_0)$. When reversing, $\{\varepsilon_y(t) = 0 | t = t_0, \ldots, T-1\}$, we obtain the deterministic kernel. The disturbances affecting the hake population alone, ε_z, have no influence on the set of viable states. This last outcome is surprising, especially as the dynamics resulting from the joint action of both disturbances narrows the set of robust viable states beyond that obtained when only accounting for ε_y. In fact, this empirical observation meets the findings drawn in §3.3.5 and reveals that the disturbance directly affecting the anchovy growth factor has critical implications on the ecosystem dynamics.

Lastly, we examine the sensitivity of the robust viability kernel $\mathbb{V}\text{iab}^R_{ER}(t_0)$, to the length

of the time horizon. It appears that as soon as $T \geq 7$, the set of robust viable states is stable and homogenous (time independent).

Robust viability kernel and extreme uncertainties

Figure 3.11 displays the deterministic viability kernel (3.17) once again, and the two robust viability kernels associated with dynamics (3.10), constraints (3.8) and (3.9), and with the uncertainty sets $\mathbb{S}_M$ and $\mathbb{S}_H$, respectively, as defined in §3.3.4.

Extreme uncertainties

Since $\mathbb{S}_M \subset \mathbb{S}_{ER}$, we know that:

$$\mathbb{V}\text{iab}^R_{ER}(t_0) \subset \mathbb{V}\text{iab}^R_M(t_0). \tag{3.21}$$

However, in practice the inclusion is not strict: our numerical results show that the robust viability kernels $\mathbb{V}\text{iab}^R_M(t_0)$ and $\mathbb{V}\text{iab}^R_{ER}(t_0)$ are equal. More precisely, whatever the set of uncertainty couples we add to $\mathbb{S}_M$, with values in the rectangle $[\bar{\varepsilon}_y^{min}, \bar{\varepsilon}_y^{max}] \times [\bar{\varepsilon}_z^{min}, \bar{\varepsilon}_z^{max}]$, the resulting robust viability kernel is the same. On the other hand, when we eliminate one of the two uncertainty couples in $\mathbb{S}_M$ in (3.15), we observe that the robust viability kernel increases.

The fact that the couple $(\bar{\varepsilon}_y^{min}, \bar{\varepsilon}_z^{max})$ produces worse adverse ecological and economic consequences is quite intuitive, whereas it is less obvious for the couple $(\bar{\varepsilon}_y^{min}, \bar{\varepsilon}_z^{min})$, given the nonlinear relationships linking both species.

Expended extreme uncertainties

Now, we consider the uncertainty set $\mathbb{S}_M$ in (3.15) and the corresponding viability kernel $\mathbb{V}\text{iab}^R_M(t_0)$. By numerical simulations, we explore the sensitivity of $\mathbb{V}\text{iab}^R_M(t_0)$ to changes in extreme uncertainties values.

- When, we increase $\bar{\varepsilon}_z^{max}$, all other things kept equal in $\mathbb{S}_M$, we observe that the viability kernel is enlarged.

- When, we increase (in absolute value) $\bar{\varepsilon}_y^{min}$ and $\bar{\varepsilon}_z^{min}$ simultaneously, all other things kept equal in $\mathbb{S}_M$, the viability kernel is empty beyond a 25% increase of these two extreme uncertainties.

- When we increase all uncertainties in $\mathbb{S}_M$ by more than 20% (a 20% increase corresponds to $\mathbb{S}_H$), the robust viability kernel is empty.

Thus, the viability kernel displays contrasted patterns when submitted to different increases in extreme uncertainty values. A possible explanation comes from (3.4), which reflects an "independence" assumption of uncertainties w.r.t time. Due to this assumption, scenarios can display arbitrary evolutions, switching from one extreme to another between time periods. Such scenarios deserve the label of worst-case scenarios as they narrow the possibility of guaranteeing ecological and economic objectives at all times. Hence, amplifying the distance between our extreme uncertainties shrinks the robust viability kernel.

The questions that rises is to what extent should the distance between decisive uncertainties be increased? As just mentioned, beyond a 20% increase of the uncertainties in $\mathbb{S}_M$, the robust viability kernel is empty. Thereby, the worst case scenario clearly belongs to the set of scenarios derived from $\mathbb{S}_M$. Nonetheless, it is not of interest as such disastrous forecast does not permit any management options, while we have no idea of its likeliness. In this context, the notion of worst case is contingent on the level of caution that one chooses to adopt, and applies to uncertainty scenarios rather than to the value of uncertainties, taken individually.

Retrospective analysis of the Peruvian anchovy-hake fisheries trajectories between 1971 and 1981

In Figure 3.11, the circles indicate the biomass observations of the anchovy-hake couple over 1971–1981. Only one circle, marked by a cross, stands within the robust viability kernel $\mathbb{V}\text{iab}_M^R(t_0)$, corresponding to the initial biomass couple estimated in 1971 [9]. Starting from that date, there theoretically existed a harvest strategy providing, for the next 10 years, at least the sustainable yields $Y^\flat$ =2,000 ktons and $Z^\flat$ =5 ktons, and guaranteeing biomasses over the preservation thresholds $y^\flat$ =7,000 ktons, $z^\flat$ =2,000 ktons, whatever the uncertainties

[9] Notice that it is the biomass couple taken as our initial value in the stochastic viability analysis in §3.2.7.

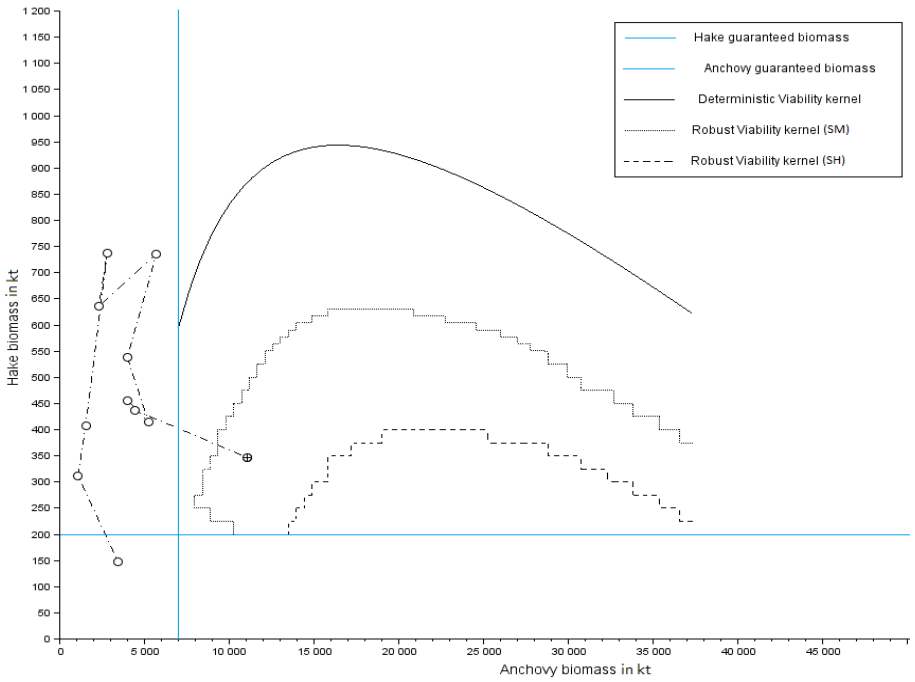


Figure 3.11: Comparing the deterministic and robust viable kernels for uncertainty sets $\mathbb{S}_M$ and $\mathbb{S}_H$

stemming from $\mathbb{S}_H$, or more exactly from the rectangle $[\bar{\varepsilon}_y^{min}, \bar{\varepsilon}_y^{max}] \times [\bar{\varepsilon}_z^{min}, \bar{\varepsilon}_z^{max}]$. In reality, the catches of year 1971 were very high, and the biomass trajectories were well below the biological minimal levels for 14 years.

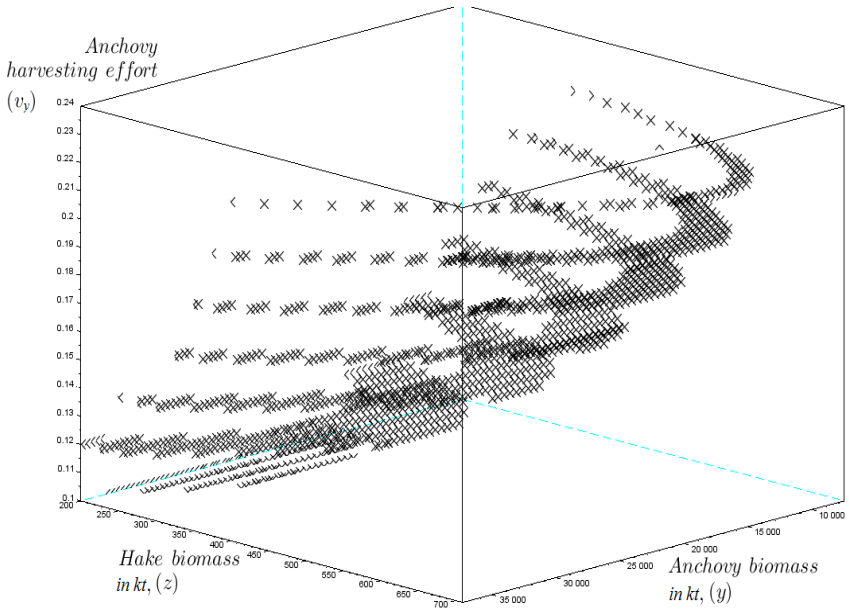
3.3.6 Robust harvesting strategies

In this last section, we examine the effort strategies produced by the robust approach. As exposed in §3.2.2, an effort strategy γ is a sequence of mappings from biomasses towards efforts, producing state paths $\{(y(t), z(t))\}_{t=t_0,\dots,T}$ as in (3.6), and control path $\{(v_y(t), v_z(t))\}_{t=t_0,\dots,T-1}$ as in (3.7), given an initial state $\big(y(t_0), z(t_0)\big) = \big(y_0, z_0\big)$.

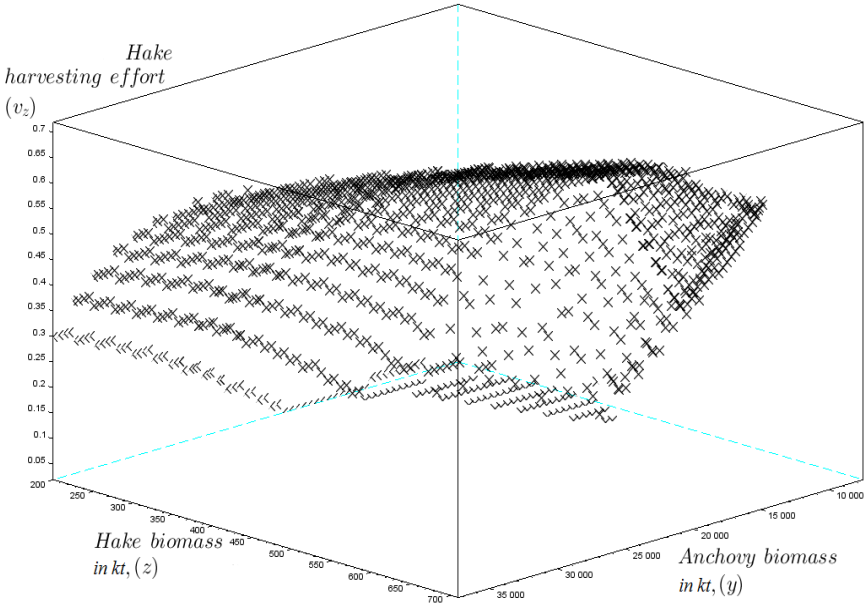
In our robust numerical application, successful effort strategies satisfy dynamics (3.10), preservation and production objectives (3.8) and (3.9). Figure 3.12 displays the robust effort strategy apparent in period $t = 0$, with $\mathbb{S} = \mathbb{S}_E$: harvesting effort of each species (v_y, v_z) are plotted as a function of species biomass levels (y, z). In Figure 3.12(a), one sees emerging the humped shape of the anchovy logistic growth function. Harvesting effort adapts to the

surplus growth of the species. What is more, fishing effort rises with the hake biomass, z, and is decreasing in the level of the anchovy biomass, y. Indeed, the lower is the anchovy stock, the higher must be fishing effort to meet the minimal yield requirement (3.9). Also, the higher is the hake stock, the stronger is the rivalry for the anchovy resource, and so the higher is anchovy harvesting effort. Nonetheless, as the anchovy stock is important, the deviation in harvesting effort required to adjust to stock variations remains low: $v_y \in [0.1, 0.24]$.

Figure 3.12(a) displays a harvesting pattern which is less obvious to interpret. In a symmetrical manner, the hake harvesting effort is decreasing in the species stock level. Logically, hake harvesting effort also varies inversely to the size of the anchovy biomass: when the anchovy species is abundant, hake harvesting effort decreases and vice versa. Thus anchovy and hake harvesting effort tend to vary in the same direction. Yet, the range in which the hake effort varies is much wider: $v_y \in [0.02, 0.72]$. At some point, it is remains sustainable to harvest more than half of the species biomass. This stems from the fact that minimal production and conservation requirements are low.



(a) Anchovy harvesting effort:v_y



(b) Hake harvesting effort:v_z

Figure 3.12: A robust effort strategy $(v_y, v_z) = \gamma_t(y, z)$ for $t = 0$, ecosystem dynamics in (3.10) and uncertainty set $\mathbb{S}_E$

As an indiction, Figure 3.13 delineates the set of biomass couples $\big(y_0, z_0\big)$ within the robust viability kernel $\mathbb{V}\text{iab}_E^R(t_0)$, for which there exists a constant robust effort strategy (in yellow). The robust effort couple is the same — $(v_y, v_z) = \gamma_t(y(t), z(t)$ — for all times

$t = t_0, \ldots, T-1$, with $T = 10$. The orange area corresponds to the set of initial biomass couples $\big(y_0, z_0\big)$, for which there exists a robust effort strategy where the robust effort couple takes only two values over $T-1$ (the first one holds until $t = 7$, the second until $t = 0$). For all the remain biomass couples in $\mathbb{V}\text{iab}_E^R(t_0)$, the robust effort strategy shifts of effort couple twice, no more (the first value holds until $t = 7$, the second until $t = 5$ and the third one until $t = 0$).

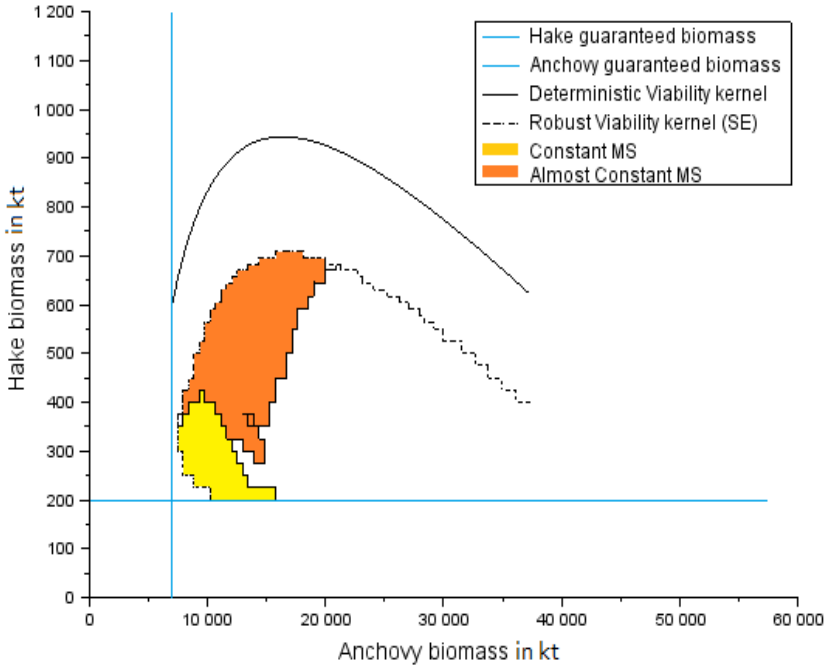


Figure 3.13: Possible robust effort strategies w.r.t. biomass couples in $\mathbb{V}\text{iab}_E^R(t_0)$

3.4 Conclusion

This work is a theoretical and practical contribution to ecosystem sustainable management under uncertainty. We proceed to a stochastic and robust viability analysis of the anchovy-hake couple in the Peruvian upwelling ecosystem.

The stochastic approach permits us to test the sensitivity of the trade-offs between policy objectives to assumptions on the set of uncertainties affecting the ecosystem dynamics, and their probability distribution. Uniformly distributed, the composition of the uncertainty

set, within a finite space, is not of great influence. We observe an important gap between objectives that can be envisaged when admitting a low level of risk and when strictly avoiding risk taking. Low risk taking reveals a wide production potential of this fishery, that can achieved in accordance with minimum stock requirement levels.

On the other hand, we show that, for a same uncertainty set, the weight affected to uncertainties considerably changes the set of policy objectives that can be envisaged with a high level of guarantee. This outcome puts forward flimsiness of management recommendations under uncertainty, when few biological data is available.

Nonetheless, in the event where available stock estimates allow to identify the stochastic process underlying an ecosystem, the stochastic viability framework appears as an operational framework to support the identification quota levels and harvesting strategies consistent with preservation and production objectives, with a high confidence level.

The robust analysis focuses on the concept of kernel and aims at illustrating the distance between the management possibilities advocated by the viability theory in two extreme cases: no uncertainty versus full hedge against risk. The robust viable kernel is an insightful mean to display the impact of uncertainty on the possibility of a sustainable management. Wherever a fishery stands, the set of robust states enables to predict whether economic and ecological objectives can be guaranteed over a time span, despite of uncertainty.

For the anchovy-hake couple in the Peruvian upwelling ecosystem, we have shown to what extent taking into account uncertainty affects the conclusions drawn from the deterministic case. By making allowance for uncertainties in the ecosystem dynamics, effort strategies guaranteeing all sustainability constraints at all times exist for fewer initial states than in the deterministic case.

In addition, we have been able to shed light on the uncertainties that really matter for a precautionary approach. Indeed, by computing several robust viable kernels, we have realized that only few important uncertainties matter, and that they correspond to extreme cases. Assessing which uncertainties truly impact the robust viability kernel can help the decision-maker to focus on those uncertainties that are relevant for sustainable management.

What is more, we have shown that not only the absolute value of extreme uncertainties matters, but also the possible arbitrary evolutions of scenarios, switching from one extreme

to another between time periods. We la We label such scenarios worst-case scenarios, as they delimit the set of robust viable states. However, because the distance between extreme uncertainties can always be amplified, the notion of worst-case is contingent on the level of caution that one chooses to adopt.

In rather common situations where very little is known about uncertainties, the robust framework contents itself of poor assumptions on sets rather than possibly unjustified probabilistic ones. However, we have seen that the robust viability kernel can be empty and that the robust approach imposes unjustified economic losses to society when risk is not catastrophic or irreversible.

Bibliography

[1] F. Alvarez-Cuadrado and N. V. Long. A mixed Bentham-Rawls criterion for intergenerational equity: Theory and implications. *Journal of Environmental Economics and Management*, 58(2):154–168, 2009.

[2] E. Anderson. Taxes vs. quotas for regulating fisheries under uncertainty: A hybrid discrete-time continuous-time model. *Marine Resource Economics*, 3(3), 1986.

[3] J. E. Anderson and L. Young. The optimality of tariff quotas under uncertainty. *Journal of International Economics*, 13(3):337–351, 1982.

[4] J-P. Aubin. *Viability Theory*. Birkhäuser, Boston, 1991. 542 pp.

[5] J. R. Beddington, D. J. Agnew, and C. W. Clark. Current problems in the management of marine fisheries. *Science*, 316(5832):1713–1716, 2007.

[6] J. R. Benddington and R. M. May. Harvesting natural populations in a randomly fluctuating environment. *Science*, 197(4302):463–465, 1977.

[7] C. Béné and L. Doyen. Sustainability of fisheries through marine reserves: a robust modeling analysis. *Journal of Environmental Management*, 69(1):1–13, 2003.

[8] C. Béné, L. Doyen, and D. Gabay. A viability analysis for a bio-economic model. *Ecological Economics*, 36:385–396, 2001.

[9] J. Bentham. *Bentham: A Fragment on Government*. Oxford at the calrendon press, 1891.

[10] D. S. Butterworth and A. E. Punt. Experiences in the evaluation and implementation of management procedures. *ICES J. Mar. Sci.*, 56(6):985–998, 1999.

[11] L. E. Blume C. P. Simon. *Mathematics for Economists*. W.W. Norton and Company, Incorporated, 1994.

[12] Laetitia Chapel, Guillaume Deffuant, Sophie Martin, and Christian Mullon. Defining yield policies in a viability approach. *Ecological Modelling*, 212(1-2):10 – 15, 2008.

[13] G. Chichilnisky. An axiomatic approach to sustainable development. *Social Choice and Welfare*, 13(2):219–248, 1996.

[14] G. Chichilnisky. An axiomatic approach to choice under uncertainty with catastrophic risks. *Resource and Energy Economics*, 22(3):221–231, 2000.

[15] G. Chichilnisky. Avoiding extinction: Equal treatment of the present and the future. Economics discussion paper, Kiel Institute for the World Economy, 2009.

[16] C. W. Clark. *Mathematical Bioeconomics*. Wiley, New York, second edition, 1990.

[17] C. W. Clark and G. P. Kirkwood. On uncertain renewable resource stocks: Optimal harvest policies and the value of stock surveys. *Journal of Environmental Economics and Management*, 13(3):235 – 244, 1986.

[18] C. W. Clark, G. R. Munro, and U. R. Sumaila. Limits to the privatization of fishery resources. *Land Economics*, 86(2):209–218, 2010.

[19] P. Cury, C. Mullon, S. Garcia, and L. J. Shannon. Viability theory for an ecosystem approach to fisheries. *ICES J. Mar. Sci.*, 62(3):577–584, 2005.

[20] P. Dasgupta and J. Stiglitz. Tariffs vs. quotas as revenue raising devices under uncertainty. *The American Economic Review*, 67(5):pp. 975–981, 1977.

[21] M. De Lara and L. Doyen. *Sustainable Management of Natural Resources. Mathematical Models and Methods*. Springer-Verlag, Berlin, 2008.

[22] M. De Lara and V. Martinet. Multi-criteria dynamic decision under uncertainty: A stochastic viability analysis and an application to sustainable fishery management. *Mathematical Biosciences*, 217(2):118–124, 2009.

[23] M. De Lara, L. Doyen, T. Guilbaud, and M-J. Rochet. Is a management framework based on spawning-stock biomass indicators sustainable? a viability approach. *ICES Journal of Marine Science*, 64(4):761–767, 2007.

[24] Michel De Lara, Eladio Ocaña Anaya, and Jorge Tam Ricardo Oliveros-Ramos. Ecosystem viable yields. *Environmental Modeling & Assessment*, 17(6):565–575, 2012.

[25] W. G. Doubleday. Environmental fluctuations and fisheries management. *International Commission for the Northwest Atlantic Fisheries. Serial paper*, 1:141–150, 1976.

[26] L. Doyen and M. De Lara. Stochastic viability and dynamic programming. *Systems and Control Letters*, 59(10):629–634, 2010.

[27] L. Doyen, M. De Lara, J. Ferraris, and D. Pelletier. Sustainability of exploited marine ecosystems through protected areas: a viability model and a coral reef case study. *Ecological Modelling*, 208(2-4):353–366, 2007.

[28] L. Doyen, O. Thébaud, C. Béné, V. Martinet, S. Gourguet, M. Bertignac, S. Fifas, and F. Blanchard. A stochastic viability approach to ecosystem-based fisheries management. *Ecological Economics*, 75(0):32–42, 2012.

[29] K. Eisenack, J. Sheffran, and J. Kropp. The viability analysis of management frameworks for fisheries. *Environmental Modeling and Assessment*, 11(1):69–79, 2006.

[30] G. Fishelson and F. Flatters. The (non)equivalence of optimal tariffs and quotas under uncertainty. *Journal of International Economics*, 5(4):385 – 393, 1975.

[31] S. Garcia, A. Zerbi, C. Aliaume, T. Do Chi, and G. Lasserre. The ecosystem approach to fisheries. Issues, terminology, principles, institutional foundations, implementation and outlook. *FAO Fisheries Technical Paper*, 443(71), 2003.

[32] H. S. Gordon. The economic theory of a common property resource: the fishery. *Journal of Political Economy*, 62:124–142, 1954.

[33] R. Hilborn and C. F. Walters. *Quantitative fisheries stock assessment. Choice, dynamics and uncertainty*. Chapman and Hall, New York, London, 1992. 570 pp.

[34] R. Hilborn, E. K. Pikitch, and R. C. Francis. Current trends in including risk and uncertainty in stock assessment and harvest decisions. *Canadian Journal of Fisheries and Aquatic Sciences*, 50(4):874–880, 1993.

[35] IMARPE. Trabajos expuestos en el taller internacional sobre la anchoveta peruana (TIAP), 9-12 Mayo 2000. *Bol. Inst. Mar Peru*, 19:1–2, 2000.

[36] IMARPE. Report of the first session of the international panel of experts for assessment of Peruvian hake population. March 2003. *Bol. Inst. Mar Peru*, 21:33–78, 2004.

[37] P. A. Larkin. An Epitaph for the Concept of Maximum Sustained Yield. *Transactions of the American Fisheries Society*, 106(1):1–11, January 1977.

[38] T. Lauck, C. W. Clark, M. Mangel, and G. R. Munro. Implementing the precautionary principle in fisheries management through marine reserves. *Ecological Applications*, 81, Special Issue:72–78, 1998.

[39] N. V. Long and V. Martinet. Combining rights and welfarism: A new approach to intertemporal evaluation of social alternatives. CESifo Working Paper Series 3746, CESifo Group Munich, 2012.

[40] D. Ludwig and C. J. Walters. Measurement errors and uncertainty in parameter estimates for stock and recruitment. *Canadian Journal of Fisheries and Aquatic Sciences*, 38(6):711–720, 1981.

[41] D. Ludwig and C. J. Walters. Optimal harvesting with imprecise parameter estimates. *Ecological Modelling*, 14(3):273 – 292, 1982.

[42] V. Martinet. A characterization of sustainability with indicators. *Journal of Environmental Economics and Management*, 61(2):183 – 197, 2011.

[43] V. Martinet and L. Doyen. Sustainable management of an exhaustible resource: a viable control approach. *Resource and Energy Economics*, 29(1):19–37, 2007.

[44] V. Martinet, L. Doyen, and O. Thébaud. Defining viable recovery paths toward sustainable fisheries. *Ecological Economics*, 64(2):411–422, 2007.

[45] V. Martinet, O. Thébaud, and A. Rapaport. Hare or Tortoise? Trade-offs in recovering sustainable bioeconomic systems. *Environmental Modeling and Assessment*, in press, 2010.

[46] R. M. May, J. R. Beddington, J. W. Horwood, and J. G. Shepherd. Exploiting natural populations in an uncertain world. *Mathematical Biosciences*, 42(3):219–252, 1978.

[47] R. Mendelssohn. Determining the best trade-off between expected economic return and the risk of undesirable events when managing a randomly varying population. *Journal of the Fisheries Research Board of Canada*, 36(8):939–947, 1979.

[48] M. J. Miranda and P. L. Fackler. *Applied Computational Economics and Finance*. The Ohio State University & North Carolina State University MIT Press, 2002.

[49] L. J. Mirman and D. F. Spulber. Uncertainty and markets for renewable resources. *Journal of Economic Dynamics and Control*, 8(3):239–264, 1984.

[50] W. Nordhaus. *A question of balance: Weighing the Options on Global Warming policies*. Yale University Press, 2008.

[51] J. A. A. De Oliveira and D. S. Butterworth. Developing and refining a joint management proceduree for the multispecies South African pelagic fisheries. *ICES J. Mar. Sci.*, 61: 1432–1442, 2004.

[52] A. Punt and R. Hilborn. Fisheries stock assessment and decision analysis: the bayesian approach. *Reviews in Fish Biology and Fisheries*, 7(1):35–63, 1997. ISSN 0960-3166.

[53] A. Punt and R. Hilborn. Bayesian stock assessment methods in fisheries. Technical report, FAO, Rome, 2001.

[54] J. Rawls. *A theory of Justice*. Clarendon, Oxford, 1971.

[55] W. J. Reed. A stochastic model for the economic management of a renewable animal resource. *Mathematical Biosciences*, 22(0):313–337, 1974.

[56] K. J. Sainsbury, A. E. Punt, and A. D. M. Smith. Design of operational management strategies for achieving fishery ecosystem objectives. *ICES Jounal of Marine Science*, 57:731–741, 2000.

[57] M. B. Schaefer. Some aspects of the dynamics of populations important to the management of commercial marine fisheries. *Bulletin of the Inter-American tropical tuna commission*, 1:25–56, 1954.

[58] M. P. Sissenwine. The effect of random fluctuations on a hypothetical fishery. *International Commission for the Northwest Atlantic Fisheries. Serial paper*, 2:137–144, 1977.

[59] S. J. Smith, J. J. Hunt, and D. Rivard. *Risk Evaluation and Biological Reference Points for Fisheries Management Canadian Special Publication of Fisheries and Aquatic Sciences No. 120*. NRC Research Press, 1993. 442 pp.

[60] R. M. Solow. Intergenerational equity and exhaustible resources. *Review of Economic Studies*, 41:29–45, 1974. Symposium on the Economics of Exhaustible Resources.

[61] M. Spence. Nonlinear prices and welfare. *Journal of Public Economics*, 8(1):1–18, 1977.

[62] N. Stern. *The Economics of Climate Change: the Stern Review*. Cambridge University Press, 2008.

[63] M. L. Weitzman. Prices vs. quantities. *Review of Economic Studies*, 41(4):477–491, October 1974.

[64] M. L. Weitzman. Landing fees vs harvest quotas with uncertain fish stocks. *Journal of Environmental Economics and Management*, 43(2):325 – 338, 2002.

Appendix C

C.1 Numerical computation of maximal viability probabilities

The dynamic programming equation adapted from De Lara and Doyen [21] and Doyen and De Lara [26] associated to dynamics (3.1) preservation (3.8) and production (3.9) minimal thresholds is given by

$$\left\{ \begin{array}{rcl} V_T(y,z) = & \mathbf{1}_{\mathbb{A}}(y,z), & \\ V_t(y,z) = & \mathbf{1}_{\mathbb{A}}(y,z) \max\limits_{(v_y,v_z)\in[0,1]^2} \mathbb{E}_{(\varepsilon_y,\varepsilon_z)}\big[\mathbf{1}_{\mathbb{B}(y,z,\varepsilon_y,\varepsilon_z)}(v_y,v_z) V_{t+1}(G(y,z,v_y,v_z,\varepsilon_y,\varepsilon_z))\big], & \end{array} \right. \tag{C.1}$$

where $(\varepsilon_y(t),\varepsilon_z(t))_{t=t_0,\dots,T-1}$ are *i.i.d.* with a uniform marginal distribution on $\mathbb{S}$ in (3.2). The function G denotes the dynamics (3.1)

$$G(y,z,v_y,v_z,\varepsilon_y,\varepsilon_z) = \left\{ \begin{array}{l} y\mathcal{R}_y\big(y,z,\varepsilon_y\big)\big(1-v_y\big)\ , \\ z\mathcal{R}_z\big(y,z,\varepsilon_z\big)\big(1-v_z\big)\ , \end{array} \right.$$

the set $\mathbb{A}$ stands for the subset of biomasses satisfying conservation objectives

$$\mathbb{A} = \{(y,z) \mid y \geq y^\flat\ , z \geq z^\flat\} = [y^\flat, +\infty[\times[z^\flat, +\infty[\ ,$$

and the set $\mathbb{B}$ stands for the subset of efforts satisfying minimal production requirements

$$\mathbb{B}(y,z,\varepsilon_y,\varepsilon_z) = \{(v_y,v_z) \mid v_y y\mathcal{R}_y(y,z,\varepsilon_y) \geq Y^\flat,\ v_z z\mathcal{R}_z(y,z,\varepsilon_z) \geq Z^\flat\}\ .$$

The notation $\mathbf{1}_{\mathbb{A}}(y,z)$ is the indicator function of the set $\mathbb{A}$: it takes the value 1 when $(y,z) \in \mathbb{A}$ and 0 else. The same holds for $\mathbf{1}_{\mathbb{B}(y,z,\varepsilon_y,\varepsilon_z)}(v_y,v_z)$.

Because we investigate several levels of catch thresholds, $Y^\flat$ and $Z^\flat$, the resolution requires a powerful computational tool since we have to solve the dynamic programming equation (C.1) for each couple $\big(Y^\flat, Z^\flat\big)$. We proceed by discretizing over state and control variables and catch thresholds. The numerical application in §3.2.4 was performed with the scientific software Scicoslab.

C.2 The deterministic viability kernel

The *deterministic viability kernel*, $\mathbb{V}\text{iab}(t_0)$, associated with the following dynamics (C.2), and constraints (C.3) and (C.4), for $t = t_0, \ldots, T$, is the set of viable states defined as follows. A couple (y_0, z_0) of initial biomasses is said to be a *viable state* if there exist a trajectory of harvesting efforts (controls) $\big(v_y(t), v_z(t)\big) \in [0,1],\ t = t_0, \ldots, T-1$, such that the state path $\{\big(y(t), z(t)\big)\}_{t=t_0,\ldots,T}$, and control path $\{\big(v_y(t), v_z(t)\big)\}_{t=t_0,\ldots,T-1}$, solution of[1]

$$\left\{ \begin{array}{rcl} y(t+1) & = & y(t)\mathcal{R}_y\big(y(t), z(t)\big)\big(1 - v_y(t)\big)\ , \\ z(t+1) & = & z(t)\mathcal{R}_z\big(y(t), z(t)\big)\big(1 - v_z(t)\big)\ , \end{array} \right. \tag{C.2}$$

starting from $\big(y(t_0), z(t_0)\big) = (y_0, z_0)$ satisfy the following goals:

- preservation (minimal biomass levels): for all $t = t_0, \ldots, T$

$$y(t) \geq y^\flat\ , \quad z(t) \geq z^\flat\ , \tag{C.3}$$

- and production requirements (minimal catch levels): for all $t = t_0, \ldots, T-1$

$$v_y(t)y(t)\mathcal{R}_y\big(y(t), z(t)\big) \geq Y^\flat\ , \quad v_z(t)z(t)\mathcal{R}_z\big(y(t), z(t)\big) \geq Z^\flat\ , \tag{C.4}$$

We now turn to the proof of Proposition 5 in §3.3.2.

Proof. Consider $y^\flat \geq 0$, $z^\flat \geq 0$, $Y^\flat \geq 0$, $Z^\flat \geq 0$. We set

$$\mathbb{V}_0 = \Big\{(y,z) \in \mathbb{R}^2_+ \,\Big|\, y \geq y^\flat, z \geq z^\flat\Big\}$$

[1]Equation (C.2) is (3.1) with the uncertainty couple $(\varepsilon_y, \varepsilon_z) = (0,0)$ (corresponding to the deterministic case). Notice that the growth rates $\mathcal{R}_y$ and $\mathcal{R}_z$ do not include uncertainty variables, as was the case in §3.2.1.

and we define a sequence $(\mathbb{V}_k)_{k\in\mathbb{N}}$ inductively by

$$\begin{aligned}\mathbb{V}_{k+1} = \{\ (y,z)\in\mathbb{V}_k \mid \exists (v_y,v_z)\in[0,1] \text{ such that } & yv_y\mathcal{R}_y(y,z)\geq Y^\flat, zv_z\mathcal{R}_z(y,z)\geq Z^\flat,\\ \text{and } & y' = y\mathcal{R}_y(y,z)(1-v_y),\ z' = z\mathcal{R}_z(y,z)(1-v_z),\\ & \text{are such that } (y',z')\in\mathbb{V}_k\big\}\ .\end{aligned}$$

For $k=0$, we obtain

$$\begin{aligned}\mathbb{V}_1 &= \left\{(y,z)\left|\begin{array}{l} y\geq y^\flat, z\geq z^\flat \text{ and, for some } (v_y,v_z)\in[0,1],\\ v_y y\mathcal{R}_y(y,z)\geq Y^\flat, v_z z\mathcal{R}_z(y,z)\geq Z^\flat,\\ y\mathcal{R}_y(y,z)(1-v_y)\geq y^\flat, z\mathcal{R}_z(y,z)(1-v_z)\geq z^\flat\end{array}\right.\right\}\\
&= \left\{(y,z)\left|\begin{array}{l} y\geq y^\flat, z\geq z^\flat \text{ for which there exist } (v_y,v_z) \text{ such that}\\ \frac{Y^\flat}{y\mathcal{R}_y(y,z)}\leq v_y\leq \frac{y\mathcal{R}_y(y,z)-y^\flat}{y\mathcal{R}_y(y,z)} \quad\text{and}\quad 0\leq v_y\leq 1,\\ \frac{Z^\flat}{z\mathcal{R}_z(y,z)}\leq v_z\leq \frac{z\mathcal{R}_z(y,z)-z^\flat}{z\mathcal{R}_z(y,z)} \quad\text{and}\quad 0\leq v_z\leq 1\end{array}\right.\right\}\\
&= \left\{(y,z)\left|\begin{array}{l} y\geq y^\flat, z\geq z^\flat,\\ \sup\{0,\frac{Y^\flat}{y\mathcal{R}_y(y,z)}\}\leq \inf\{1,1-\frac{y^\flat}{y\mathcal{R}_y(y,z)}\}\\ \sup\{0,\frac{Z^\flat}{z\mathcal{R}_z(y,z)}\}\leq \inf\{1,1-\frac{z^\flat}{z\mathcal{R}_z(y,z)}\}\end{array}\right.\right\}\\
&= \left\{(y,z)\left| y\geq y^\flat, z\geq z^\flat, \frac{Y^\flat}{y\mathcal{R}_y(y,z)}\leq \frac{y\mathcal{R}_y(y,z)-y^\flat}{y\mathcal{R}_y(y,z)}, \frac{Z^\flat}{z\mathcal{R}_z(y,z)}\leq \frac{z\mathcal{R}_z(y,z)-z^\flat}{z\mathcal{R}_z(y,z)}\right.\right\}\\
&= \left\{(y,z)\left| y\geq y^\flat, z\geq z^\flat, Y^\flat\leq y\mathcal{R}_y(y,z)-y^\flat, Z^\flat\leq z\mathcal{R}_z(y,z)-z^\flat\right.\right\}\ .\end{aligned}$$

Then, for $k=1$, we obtain

$$\begin{aligned}\mathbb{V}_2 &= \left\{(y,z)\left|\begin{array}{l} y\geq y^\flat, z\geq z^\flat \text{ and, for some } (v_y,v_z)\in[0,1],\\ v_y y\mathcal{R}_y(y,z)\geq Y^\flat,\ v_z z\mathcal{R}_z(y,z)\geq Z^\flat\\ \text{and such that } (y',z')\in\mathbb{V}_1\\ \text{where } y' = y\mathcal{R}_y(y,z)(1-v_y),\ z' = z\mathcal{R}_z(y,z)(1-v_z)\end{array}\right.\right\}\\
&= \left\{(y,z)\left|\begin{array}{l} y\geq y^\flat, z\geq z^\flat \text{ and, for some } (v_y,v_z)\in[0,1],\\ v_y y\mathcal{R}_y(y,z)\geq Y^\flat,\ v_z z\mathcal{R}_z(y,z)\geq Z^\flat,\ y'\geq y^\flat,\ z'\geq z^\flat,\\ Y^\flat\leq y'\mathcal{R}_y(y',z')-y^\flat,\ Z^\flat\leq z'\mathcal{R}_z(y',z')-z^\flat\\ \text{where } y' = y\mathcal{R}_y(y,z)(1-v_y),\ z' = z\mathcal{R}_z(y,z)(1-v_z)\end{array}\right.\right\}.\end{aligned}$$

We now make use of the property (see De Lara et al. [24]) that, when the decreasing sequence $(\mathbb{V}_k)_{k\in\mathbb{N}}$ is stationary, its limit is the viability kernel $\mathbb{V}\text{iab}(t_0)$. Hence, it suffices to show that $\mathbb{V}_1\subset\mathbb{V}_2$ to obtain that $\mathbb{V}\text{iab}(t_0)=\mathbb{V}_1$.

Let $(y,z) \in \mathbb{V}_1$. We have that

$$y \geq y^\flat, \quad z \geq z^\flat \text{ and } \quad y\mathcal{R}_y(y,z) - y^\flat \geq Y^\flat, \quad z\mathcal{R}_z(y,z) - z^\flat \geq Z^\flat \ .$$

Let us set $\hat{v}_y = \frac{y\mathcal{R}_y(y,z) - y^\flat}{y\mathcal{R}_y(y,z)}$, which has the property that $y' = y\mathcal{R}_y(y,z)(1-\hat{v}_y) = y^\flat$. We prove that $\hat{v}_y \in [0,1]$. Indeed, on the one hand, we have that $\hat{v}_y \leq 1$ since $1 - \hat{v}_y = y^\flat / y\mathcal{R}_y(y,z)$, where $y^\flat \geq 0$. On the other hand, since by assumption $y\mathcal{R}_y(y,z) - y^\flat \geq Y^\flat \geq 0$, we deduce that $\hat{v}_y \geq 0$. The same holds true for $\hat{v}_z$ and $z' = z\mathcal{R}_z(y,z)(1-\hat{v}_z) = z^\flat$. By (3.13), we deduce that

$$y'\mathcal{R}_y(y',z') - y^\flat = y^\flat \mathcal{R}_y(y^\flat, z^\flat) - y^\flat \geq Y^\flat \text{ and } z'\mathcal{R}_z(y',z') - z^\flat = z^\flat \mathcal{R}_z(y^\flat, z^\flat) - z^\flat \geq Z^\flat \ .$$

The inclusion $\mathbb{V}_1 \subset \mathbb{V}_2$ follows, hence $\mathbb{V}\text{iab}(t_0) = \mathbb{V}_1$, and (3.14) holds true. □

The *viable controls* attached to a given viable state $(y,z) \in \mathbb{V}\text{iab}(t_0)$ are the admissible controls (v_y, v_z) such that the image by the dynamics (C.2) is in $\mathbb{V}\text{iab}(t_0)$.

Corollary 6 *Suppose that the assumptions of Proposition 5 are satisfied. The set of viable controls associated with the state (y,z) is*

$$\left\{ (v_y, v_z) \in [0,1]^2 \;\middle|\; \begin{array}{l} \frac{y\mathcal{R}_y(y,z) - y^\flat}{y\mathcal{R}_y(y,z)} \geq v_y \geq \frac{Y^\flat}{y\mathcal{R}_y(y,z)}, \quad \frac{z\mathcal{R}_z(y,z) - z^\flat}{z\mathcal{R}_z(y,z)} \geq v_z \geq \frac{Z^\flat}{z\mathcal{R}_z(y,z)}, \\ y'\mathcal{R}_y(y',z') - y^\flat \geq Y^\flat, \ z'\mathcal{R}_z(y',z') - z^\flat \geq Z^\flat \end{array} \right\} ,$$

where $y' = y\mathcal{R}_y(y,z)(1-v_y)$, $z' = z\mathcal{R}_z(y,z)(1-v_z)$.

C.3 Numerical computation of robust viability kernels

We first sketch how to establish a dynamic programming equation associated with dynamics (3.1), and preservation (3.8) and production (3.9) minimal thresholds. Then, we depict a numerical discretization scheme to solve this equation numerically.

C.3.1 Dynamic programming equation

The dynamic programming equation associated with dynamics (3.1), and preservation (3.8) and production (3.9) minimal thresholds is given by[2]

$$\begin{aligned} V_T(y,z) = &\ \mathbf{1}_{\mathbb{A}}(y,z), \\ V_t(y,z) = &\ \mathbf{1}_{\mathbb{A}}(y,z) \max_{(v_y,v_z)\in[0,1]^2} \min_{(\varepsilon_y,\varepsilon_z)\in\mathbb{S}(t)} \left[\mathbf{1}_{\mathbb{B}(y,z,\varepsilon_y,\varepsilon_z)}(v_y,v_z) V_{t+1}\big(G(y,z,v_y,v_z,\varepsilon_y,\varepsilon_z)\big)\right], \end{aligned} \tag{C.5}$$

for all $t = t_0, \ldots, T-1$, where the continuous function G denotes the dynamics (3.1)

$$G(y,z,v_y,v_z,\varepsilon_y,\varepsilon_z) = \begin{cases} y\mathcal{R}_y\big(y,z,\varepsilon_y\big)\big(1-v_y\big)\ , \\ z\mathcal{R}_z\big(y,z,\varepsilon_z\big)\big(1-v_z\big)\ , \end{cases} \tag{C.6}$$

where $\mathbb{A}$ stands for the subset of biomass satisfying conservation objectives (3.8)

$$\mathbb{A} = \{(y,z) \mid y \geq y^\flat\ , z \geq z^\flat\} = [y^\flat, +\infty[\times[z^\flat, +\infty[\ , \tag{C.7}$$

and where $\mathbb{B}(y,z,\varepsilon_y,\varepsilon_z)$ stands for the subset of catches satisfying minimal production requirements (3.9)

$$\mathbb{B}(y,z,\varepsilon_y,\varepsilon_z) = \{(v_y,v_z) \in [0,1]^2 \mid v_y y\mathcal{R}_y(y,z,\varepsilon_y) \geq Y^\flat,\ v_z z\mathcal{R}_z(y,z,\varepsilon_z) \geq Z^\flat\}\ . \tag{C.8}$$

The notation $\mathbf{1}_{\mathbb{A}}(y,z)$ is the indicator function of the set $\mathbb{A}$: it takes the value 1 when $(y,z) \in \mathbb{A}$ and 0 else. The same holds for $\mathbf{1}_{\mathbb{B}(y,z,\varepsilon_y,\varepsilon_z)}(v_y,v_z)$.

It turns out that, for all $t = t_0, \ldots, T$, the robust viability value function V_t is the indicator function $\mathbf{1}_{\mathbb{V}\mathrm{iab}^R(t)}$ of the robust viability kernel $\mathbb{V}\mathrm{iab}^R(t)$ (see De Lara and Doyen [21]). The sketch of the proof is as follows, by backward induction.

By (C.5), we have that $V_T = \mathbf{1}_{\mathbb{A}} = \mathbf{1}_{\mathbb{V}\mathrm{iab}^R(T)}$. Now, assume that $V_{t+1} = \mathbf{1}_{\mathbb{V}\mathrm{iab}^R(t+1)}$. When the operation $\min_{(\varepsilon_y,\varepsilon_z)\in\mathbb{S}(t)}$ is performed in (C.5), the result is 1 if, and only if, for all uncertainties $(\varepsilon_y,\varepsilon_z) \in \mathbb{S}(t)$, we have both $\mathbf{1}_{\mathbb{B}(y,z,\varepsilon_y,\varepsilon_z)}(v_y,v_z) = 1$ and $\mathbf{1}_{\mathbb{V}\mathrm{iab}^R(t)}\big(G(y,z,v_y,v_z,\varepsilon_y,\varepsilon_z)\big) = 1$, that is, both efforts (v_y,v_z) satisfy minimal production requirements (3.9) and the images $G(y,z,v_y,v_z,\varepsilon_y,\varepsilon_z)$ by the dynamics G belong to the viability kernel $\mathbb{V}\mathrm{iab}^R(t)$. Then, the operation $\max_{(v_y,v_z)\in[0,1]^2}$ yields 1 if, and only if, there is at least one control (v_y,v_z) — indeed achieved by continuity of the dynamics G in (C.6) — such that (3.9) is satisfied and

[2] What follows is a simple extension of the results in De Lara and Doyen [21] and Doyen and De Lara [26].

$G(y, z, v_y, v_z, \varepsilon_y, \varepsilon_z) \in \mathbb{V}\text{iab}^R(t)$. The term $\mathbf{1}_{\mathbb{A}}(y, z) = 1$ if, and only if, the conservation objectives (3.8) are satisfied. To end, we obtain that $V_t(y, z) = 1$ if, and only if, there exists at least one control (v_y, v_z) such that the conservation objectives (3.8) and the production requirements (3.9) are satisfied, and that the images $G(y, z, v_y, v_z, \varepsilon_y, \varepsilon_z)$ by the dynamics G belong to the viability kernel $\mathbb{V}\text{iab}^R(t)$ for all uncertainties $(\varepsilon_y, \varepsilon_z) \in \mathbb{S}(t)$. By a simple extension of the results in De Lara and Doyen [21] and Doyen and De Lara [26], we have just characterized $\mathbb{V}\text{iab}^R(t)$.

C.3.2 Numerical resolution of the dynamic programming equation

Now, we expose how we proceed to find the robust viability kernel numerically thanks to the dynamic programming equation (C.5).

We discretize biomass, harvesting effort and uncertainty values. A top loop for time steps embraces two nested loops for state variables y and z, respectively. Next, loops over uncertainties nested in loops over harvesting efforts allow us to obtain the set of images associated with a biomass couple (some of these steps are actually done through matrix computing). Images for target constraints that are not satisfied are set equal to zero. We then project these images on the value function grid of the previous period, through linear interpolation. At given efforts, we retain the minimum value obtained over all uncertainty couples. Then, we retain the highest value produced by an effort couple among all tested. It is this value that is multiplied with the value function of the current time period, at the location of the biomass couple at stake. The robust viability kernel is defined by the set of grid points where the value function is equal to 1. This implies that biomass couples for which, at a date t, all images do not fall between four 1 in the interpolation are excluded from the robust viability kernel (in the sense that we provide robustness with respect to grid approximation).

Chapter 4

Conclusion

This dissertation addresses the consequences of the biological and market interactions between two production processes of food fish: the aquaculture and capture fishery sectors. It focuses on the consequences of these interactions from an economic, ecologic and food security point of view.

Biological interactions between both sectors stem from the fact that aquaculture relies on natural populations for the feeding of 46.1% of its total global production (including aquatic plants). The fish stocks targeted to feed farmed fish constitute preys for species that are part of the same food web, which in turn, may be targeted for direct human consumption. The multi-sector dynamic model developed in Chapter 1 emphasizes how critical these interactions may be to the sustainability of each sector. It is shown that when these biological interactions are high, aquaculture leads to a decline in the feed fish stock and the wild edible fish stock, a decrease in wild edible fish supply and an increase in its price. What is more, for high levels of income, aquaculture provokes the collapse of the wild edible fishery though it would have remained alone.

These results are derived assuming fisheries operate in open access, non-cooperatively, although an efficient exploitation of resources would call for a global optimization framework. Indeed, the aquaculture and capture fishery sectors produce negative externalities and call for public intervention to regulate their activity in order to remain viable economically and ecologically. In this respect, the following investigation that should be undertaken on the basis of the theoretical framework of Chapter 1, is to seek the condition on regulatory instruments

(tax, quota) to control harvesting effort, that achieve the joint optimization of both sectors' surplus, in a sustainable way.

While this research prospect should provide qualitative insights on how to set management instruments, Chapter 3 applies an operational methodology — the viability approach — to define harvesting effort strategies in consistency with economic and conservation objectives, and ecosystem dynamics under uncertainty. In particular, a numerical application of the viability analysis to the anchovy-hake couple in the peruvian upwelling system is carried. Both species are related by a prey-predator relationship.

This application is a concert example of biological interaction between the aquaculture and capture fishery industries since the peruvian anchovy stock generates an important share of world supply of farmed fish feed, while hake is targeted for direct human consumption. The study enables to quantify the trade-offs between minimal production and preservation thresholds entailed by the prey-predator relationship linking both species. In particular, this chapter investigates how to control harvesting effort according to a precautionary approach with respect to uncertainties affecting the ecosystem dynamics, without inflicting too high economic costs to society.

In terms of further research perspective, it would be interesting to observe how our results are affected by constraining variations in the level harvesting effort from one period to another. Indeed, in facts, a fleet might not be able to considerably modify its harvesting capacity in a short time span. Thus, imposing smooth variation in controls appears as a more realistic assumption.

Another interesting extension to this work would be to replace minimal catch requirements on both species by a modelling of the aquaculture and capture fishery production structure, so as to define catch requirements endogenously. This formalization would permit to explore the influence of the efficiency of the aquaculture technology, the price of feed fish or fuel on catch requirements, and the likeliness of satisfying them content of the prey-predator dynamics relating both targeted species.

Besides biological interactions, the aquaculture and capture fishery production come to interact at the market level. Indeed, the introduction of farmed fish on stalls can affect food fish price dynamics, and in turn, the supply of both sectors so as the pressure that applies

to wild stocks. The extent to which farmed fish affects fish food price dynamics depends on the degree of substituability between wild and farmed fish.

The theoretical model established in Chapter 1 accounts for the demand side and assumes farmed and wild product are substitutes. An analytical examination of the influence of the degree of substituability between both fish products on steady state outcomes shows that consumer preferences may reverse them. In fact, a high degree of substitutions always seems to be beneficial from an ecological and welfare point of view. Indeed, if there where no competition between wild and farmed fish products, this would allow fishermen to earn short-term rents. However, these rents may not be sustained if investment in overcapacity and the associated fishing pressure are not limited. Indeed, if a fishery is in open access, the absence of consumption adjustment to a rising price notifying a scarce resource may threaten its sustainability, thus weakening the ecosystem to which it belongs. Quaas and Requate [1] demonstrate that the inelastic demand behaviors resulting from consumers preference for food fish diversity is likely to cause sequential collapse of fish stocks under open-access fishery.

However, even if aquaculture production may alleviate pressure on a given wild species, it is not possible to qualify and quantify the net effect of farming on ecosystems' sustainability owing to the numerous linkages between inherent species. Indeed, to ensure economic viability, fishermen rarely target a single species. Thus, a lower profitability of a species due to harsh price competition from the aquaculture sector may bring fishermen to report fishing pressure on other fish species within their bundle of catches, threatening the health of the ecosystem.

Chapter 2 addresses market competition between farmed and wild fish empirically, within the French sea bass and sea bream markets. It is found that market interactions between wild and farmed fish differ from one species to another. In fact, econometric studies investigating consumer behaviour towards farmed and wild fish products are not unanimous on this question. This outcome highlights the need to carry additional studies on different species and within other market places to better understand the factors critical to the integration of aquaculture production.

The behavior of consumers towards farmed products is conditioned by their perception of the aquaculture industry but also by local demand factors such as consumption habits and

income. According to our study it also appears that consumers may be more sensitive to the seafood production process when it comes to higher-value species. On the other hand, consumers willing to spend a limited budget on food fish seems to pass over the production process attribute.

Consumers awareness that intensive aquaculture techniques causes environmental pollution or distrust regarding the health implications of change in species natural diet calls for an industry applying environmentally friendly technics in one case, and relying on forage fish to feed dependent farmed species in the other. Reliable and transparent certifications for these production features, conferring supply a price premium, should provide incentives to develop quality aquaculture subsectors.

Nonetheless, an ecologically sustainable aquaculture requires more costly production technics, while the exhaustibility of marine resource is a limit to fed aquaculture production. Food security and economically constrained populations require a large scale food fish production which cannot be based on natural populations to fill the supply gap of the fishery industry. Thus, it is likely that the fish food market becomes increasingly segmented, with respect to wild versus farmed fish, but also that the aquaculture supply splits into a high value and low value segment.

The aquaculture industry requires intensive research to develop cost-effective substitutes to forage fish, so as to maintain its growth rate and the nutritional intakes of farmed fish. In addition, there is a need for stronger supervision of its environmental impacts to prove sustainable and compatible with other resource users. As exposed, fed aquaculture implies biological interactions with capture fisheries, but the waste loads it releases into the wild in certain areas also has consequences on the activity of various economic sectors relying on costal and fresh water resources such as small scale artisanal fishermen or anglers. It seems obvious that the sustainability and efficiency of the aquaculture and capture fishery sectors involves an integrated approach to fish market and to fisheries resources management. A global analysis of fisheries dependent sectors will enhance the socioeconomic benefit withdrawn by society from this natural resource.

Bibliography

[1] M. F. Quaas and T. Requate. Sushi or fish fingers? Seafood diversity, collapsing fish stocks, and multispecies fishery management. *Scandinavian Journal of Economics*, 115: 381–422, 2013.