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Risk averse policies foster bio-economic sustainability in mixed fisheries

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Abstract

This article examines the role of risk aversion on the sustainable management of mixed fisheries. We consider a bio-economic model of multiple species harvested by a single fleet with uncertain costs of effort. We assume that the regulatory agency aims at reaching MMEY (Multispecies Maximum Economic Yield) by maximizing the expected utility of total profits, where the utility function captures risk aversion. We show analytically that such a risk-averse MMEY mitigates the risk of biological and economic overexploitation of the different species and thus of biodiversity loss. However too high risk aversion also lessens food production. Thus, risk aversion implies a trade-off between different bio-economic goals. These findings are illustrated with the case study of the Australian South East Fishery, where intermediate risk aversion levels allow for balanced bio-economic management objectives, therefore fostering sustainability.

Keywords

Bioeconomics, multispecies fishery, ecosystem-based fisheries management, maximum economic yield, uncertainty, risk aversion.

1 Introduction

Marine ecosystems and fisheries are under pressure worldwide (McWhinnie, 2009). In response, ecosystem-based fishery management (EBFM) has been put forward as an effective and holistic approach for managing world fisheries (Pikitch et al., 2004). This approach generally aims at integrating the ecological and economic complexities of fisheries, and at implementing a multi-criteria framework that allows to sustainably balance ecological and socio-economic objectives (Thébaud et al., 2014; Doyen et al., 2017). The EBFM approach also seeks to embrace the multiple ecological and economic uncertainties that fishermen and managers usually face (Sethi et al., 2005). The general objective of this article is to contribute to EBFM by investigating the bioeconomic performances of multi-species fisheries in a context of economic uncertainty.

Operationalizing the EBFM approach requires new models, to integrate the multiple bio-economic complexities at play (Plagányi, 2007) as well as the uncertainties that are inherent to fisheries (Dowling et al., 2015; Fulton et al., 2016; Sanchirico et al., 2008). These new models should enable to evaluate the capacity for public policies to ensure economic, social and ecological sustainability (Doyen et al., 2012; Péreau et al., 2012; Schuhbauer and Sumaila, 2016).

In that respect, the use of monospecific reference points in multispecies fisheries is increasingly criticized (Legović and Geček, 2010). For instance, monospecific maximum sustainable yield (MSY) targets have been shown to alter the structure of harvested ecosystems (Walters et al., 2005). Moreover, although maximum economic yield (MEY) favor higher biomasses than MSY policies in single-species fisheries (Clark, 2010; Dichmont et al., 2010), it does not account for potential ecological interactions in mixed fisheries (Hoshino et al., 2017). Instead of single-species reference points, there have been attempts at defining multispecies MSY (MMSY) and MEY (MMEY) policies, at which total catches or total profits are maximized (Mueter and Megrey, 2006; Guillen et al., 2013). Such global harvesting policies may however enhance biodiversity losses: while MMSY policies are likely to threaten low-productivity species, MMEY policies induce the overexploitation of stocks with low economic value (Clark, 2006; Tromeur and Doyen, 2018).

Moreover, optimal harvesting policies are generally based on deterministic models that do not account for the multiple bio-economic uncertainties facing fishermen and managers (Grafton et al., 2010). Failing to account for uncertainties can be detrimental to the relevance of fisheries management and public policies. First because fisheries are affected by biological uncertainties. Beddington and May (1977) have shown for instance that the uncertain growth of fish stocks could affect the definition of biological reference points. Likewise, Charles and Munro (1985) suggested that biological uncertainty favors more conservationist policies. Second, fisheries are characterized by uncertainties on economic variables. Uncertainty in fish prices associated with growth stochasticity has for instance been found to induce potential trade-offs between mean profits and their variance (Gourguet et al., 2014). In that perspective, designing portfolios of harvested species has been proposed as a strategy to balance profits with volatility (Edwards et al., 2004). Uncertainty in fishing costs also constitutes a significant source of variability in fisheries revenues. In particular, fuel costs mainly depend on the price of fossil fuels, that are highly volatile assets (Cheilari et al., 2013; Tyedmers, 2004). However, the consequences of cost uncertainty on fisheries management remains far less studied and understood than growth and price uncertainties.

In that context of uncertainty, fishermen as well as managers may be characterized by risk-averse attitudes (Brick et al., 2012). Dealing with bio-economic uncertainty in fisheries management therefore implies to account for attitudes towards risk. In particular, as argued by Sanchirico et al. (2008), managers' objective function must account for risk preferences to develop EBFM tools. Risk-averse behaviors have been shown to affect the definition of optimal sustainable yields (Ewald and Wang, 2010). Baldursson and Magnusson (1997) also showed that consideration of risk implies a reduction in effort form the profit-maximizing level. Accounting for risk aversion in uncertain fisheries may thus help to define management strategies and reference points that allow to balance ecological and economic risks and to promote resilience (Doyen et al., 2017; Grafton et al., 2019).

The aim of this article is to evaluate the bio-economic consequences of risk-averse attitudes under cost uncertainty, as well as to question their relevance in operationalizing ecosystem-based management and multispecies reference points. To do so, we use a bio-economic model with multiple species and a single fleet, and we model preferences by a quadratic utility function. This allows us to derive analytical conditions for a sustainable stochastic MMEY. Thereby, we build a general analytical framework to evaluate how risk-averse attitudes impact sustainability in multispecies fisheries. In particular, we show how risk aversion affects food production, profit variability and biodiversity. These analytical results are illustrated using the case study of the South East Fishery in Australia.

The article is organized as follows. In the following section entitled 'Bio-economic model', we present the bio-economic model describing the management of mixed fisheries relying on risk-averse MMEY policies. In Section entitled 'Results', we examine the impact of risk aversion on ecological and economic performances of these MMEY. Finally, we illustrate in Section 'Case Study' the theoretical results with the example of the South East Fishery in Australia.

2 Bio-economic model

2.1 Multispecies dynamics and equilibrium yield

We consider N species jointly harvested by a single fleet. It is assumed that no ecological interaction occurs between the species. The dynamics of every species $i = 1, \ldots, N$ is described in discrete time as follows:

$$x_i(t+1) = x_i(t)\left(1 + r_i(x_i(t)) - q_i e(t)\right) \tag{1}$$

where $x_i(t)$ denotes the stock of species i at time t, the function $r_i(x)$ its growth rate[1], $q_i \geq 0$ its catchability while $e(t) \geq 0$ stands for the fishing effort at time t. We assume that the growth rate function $r_i(x)$ is continuous, differentiable and strictly decreasing[2] on $\mathbb{R}_+$ namely $r_i^{'}(x) < 0$ for any $x \in \mathbb{R}_+$.

We compute the following equilibrium stocks and efforts for every species i:

$$e(x_i) = \frac{r_i(x_i)}{q_i} \tag{2}$$

or equivalently (as function $r_i(x)$ is a bijection)

$$x_i(e) = r_i^{-1}(q_i e). \tag{3}$$

The harvest at equilibrium for every species i is then defined as follows :

$$h_i(e) = q_i e x_i(e) \tag{4}$$

Such a relation points to the non linear nature of the equilibrium yields for the different species.

2.2 Uncertain profits and MMEY objective

Extending the concept of MEY to the multispecies framework, we consider the multispecies maximum economic yield (MMEY), which aims at maximizing total profits, defined as the difference between total revenues derived from harvesting the different species and the costs (here uncertain) of fishing effort. The total profit $\pi(t)$ of the

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[1] For instance the logistic growth $r_i(x) = r_i\left(1 - \frac{x}{k_i}\right)$ with r_i the intrinsic growth rate and k_i the carrying capacity of species i.

[2] Such an assumtion holds true for the logistic function since $r_i^{'}(x) = -\frac{r_i}{k_i}$.

fishery at time t thus reads as follows :

$$\begin{aligned} \pi(t) &= \sum_{i=1}^{N} p_i h_i(t) - c(t)e(t) \\ &= e(t)\left(\sum_{i=1}^{N} p_i q_i x_i(t) - c(t)\right) \end{aligned} \tag{5}$$

where p_i is the price of species i and $c(t)$ is the uncertain per-unit-effort cost of fishing at time t. Costs of effort $c(t)$ are assumed to vary stochastically through time. The probability distribution of variable cost c is assumed to be independently and identically distributed (i.i.d.) with expectation $\overline{c}$ and standard deviation σ_c, namely:

$$\mathbb{E}[c(t)] = \overline{c} \quad \text{and} \quad \mathbb{V}\mathrm{ar}[c(t)] = \sigma_c^2 \tag{6}$$

Many fisheries are regulated by regional agencies, that apply general directives at regional levels. An example is the Australian Fisheries Management Agency, which objective is to implement an ecosystem-based management of Australia's exclusive economic zone (Scandol et al., 2005). The fisheries management agency then acts as a social planner. We assume that the management objective, inspired by MMEY, is defined so as to maximize the expected utility of aggregated profits (5) at equilibrium:

$$\max_{e \text{ satisfying } (2)} \mathbb{E}(U(\pi(t))) \tag{7}$$

where U is a utility function capturing risk aversion. Risk averse attitudes being common in society (Binswanger, 1980) and in fishermen communities (Brick et al., 2012), we assume the policy maker to be risk-averse (Eeckhoudt et al., 2005). For the sake of simplicity, we rely on a quadratic utility function consistent with portfolio theory or with mean-variance analysis (Edwards et al., 2004; Baldursson and Magnusson, 1997):

$$U(\pi(t)) = U_{\mathrm{a}}(\pi(t)) = \pi(t) - \frac{\mathrm{a}}{2}(\pi(t) - \mathbb{E}[\pi(t)])^2, \tag{8}$$

where $\mathrm{a} \geq 0$ is a coefficient capturing risk aversion of the policy maker. Hereafter, we will simply call it *risk aversion level.* For $\mathrm{a} = 0$, the decision is said to be risk-neutral while for $\mathrm{a} > 0$, it is said to be risk-averse.

Combining (6), (8) and (7), for each value of a, the optimal risk-averse MMEY effort denoted by $e_{\mathrm{a}}^{\mathrm{MMEY}}$ solves the following maximization problem (9) :

$$\max_{e \text{ satisfying } (2)} \sum_i p_i q_i x_i e - \overline{c}e - \frac{\mathrm{a}}{2}\sigma_c^2 e^2. \tag{9}$$

Using equilibrium equation (3), the optimality problem reads as follows

$$\max_{e \geq 0} V_{\mathrm{a}}(e), \tag{10}$$

with the objective function

$$V_{\mathrm{a}}(e) = \sum_i p_i h_i(e) - \overline{c}e - \frac{\mathrm{a}}{2}\sigma_c^2 e^2. \tag{11}$$

In what follows, we study how risk aversion impacts different ecological and economic metrics through the optimal fishing effort. Note that high fishing efforts favor the social objective of maintaining jobs in the fishing industry, while it may induce biological and economic overexploitation of fish stocks, potentially affecting food security and biodiversity.

Hereafter, we assume that the functions $h_i(e) = q_i e x_i(e)$ are concave, namely $h_i''(e) < 0$, and admits a unique optimum[3]. Thus the objective function $V_{\mathrm{a}}(e)$ is also concave.

2.3 Overexploitation

In line with FAO (2016), we consider that a species is biologically overharvested if its biomass is smaller than its MSY biomass where catch at equilibrium is maximal or equivalently when the global harvesting effort in the fishery is larger than the monospe-

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[3] This holds true for the logistic dynamics since, in that case $h_i(e) = q_i k_i \left(1 - \frac{q_i e}{r_i}\right) e$, and thus $h_i''(e) = -2k_i \frac{q_i^2}{r_i} < 0$. Moreover there is a unique e such that $h_i'(e) = q_i k_i - 2\frac{q_i}{r_i} e = 0$.

cific MSY effort of this species. More specifically we define the individual sustainable effort e_i^{MSY} as follows:

$$h_i(e_i^{\text{MSY}}) = \max_e h_i(e) \tag{12}$$

where mono-specific catch at equilibrium[4] $h_i(e)$ is defined in equation (4). A species is thus considered overharvested when

$$e > e_i^{\text{MSY}}, \tag{14}$$

that is when the harvesting effort is larger than its monospecific MSY effort. If it is equal to the MSY effort, the species is said to be fully exploited

Likewise, we consider that a species is economically underexploited if its biomass is smaller than its risk-neutral MEY biomass (where the monospecific risk-neutral utility function at equilibrium is maximal). In a risk-neutral fishery, the optimal MEY strategy is implicitly defined as maximizing the individual expected profit:

$$\overline{\pi_i}(e) = \mathbb{E}(\pi_i(e)) = p_i h_i(e) - \bar{c}e \tag{15}$$

The mono-specific sustainable effort e_i^{MSY} is then defined as follows:

$$\overline{\pi_i}(e_i^{\text{MSY}}) = \max_e \overline{\pi_i}(e) \tag{16}$$

It is well known that the MEY effort is always smaller than the MSY effort[5]. In

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[4] In the case of the logistic dynamics where $h_i(e) = q_i k_i \left(1 - \frac{q_i e}{r_i}\right) e$, using first order conditions of optimality, MSY is explicitly characterized by

$$x_i^{\text{MSY}} = \frac{k_i}{2} \quad \text{and} \quad e_i^{\text{MSY}} = \frac{r_i}{2q_i} \tag{13}$$

[5] For instance for the logistic growth, using first order conditions of optimality, MEY for species i is explicitly characterized by

$$x_i^{\text{MEY}} = \frac{k_i}{2} + \frac{\bar{c}}{2p_i q_i} = x_i^{\text{MSY}} + \frac{\bar{c}}{2p_i q_i} \quad \text{and} \quad e_i^{\text{MEY}} = \frac{r_i}{q_i}\left(\frac{1}{2} - \frac{\bar{c}}{2p_i q_i k_i}\right) = e_i^{\text{MSY}} - \frac{\bar{c} r_i}{2p_i k_i}$$

the monospecific case, MEY is thus always more conservative than MSY.

The next section aims at using this model to assess the bio-economic sustainability of a risk-averse MMEY. We pay a particular attention to the impact of risk aversion on biological and economic overexploitation. We also evaluate aggregated metrics of bio-economic sustainability, such as expected profitability and productivity.

3 Results

In this section, we analyze the impact of risk aversion on various aspects of bio-economic sustainability: employment (measured by the fishing effort), profitability (linked with economic overexploitation), biological sustainability and diversity (linked with biological overexploitation), and food security (linked with catches).

3.1 At MMEY, risk aversion reduces the fishing effort

We first show that the MMEY effort as defined in (7) or (10) is decreasing with respect to the risk aversion level.

Proposition 1. *The optimal effort* $e_{\mathrm{a}}^{\mathrm{MMEY}}$ *is a decreasing function of the risk aversion level* a*. Furthermore, the optimal MMEY effort with risk aversion is always lower than without risk aversion:* $e_{\mathrm{a}}^{\mathrm{MMEY}} \leq e_{0}^{\mathrm{MMEY}}$

Proof. See Appendix A.1.

This result is illustrated in a two-species fishery (Figure 1), where the purpose of risk-neutral MMEY leads to overharvest the species with the lowest growth rate. As expected from Proposition 1, increasing risk aversion reduces the optimal harvesting effort. Reduced efforts imply lower costs and thus lower profit variability. However, a large reduction of fishing activity and effort induced by a high risk aversion may also raise social concerns and question the social acceptability of such a policy, as discussed in Péreau et al. (2012).

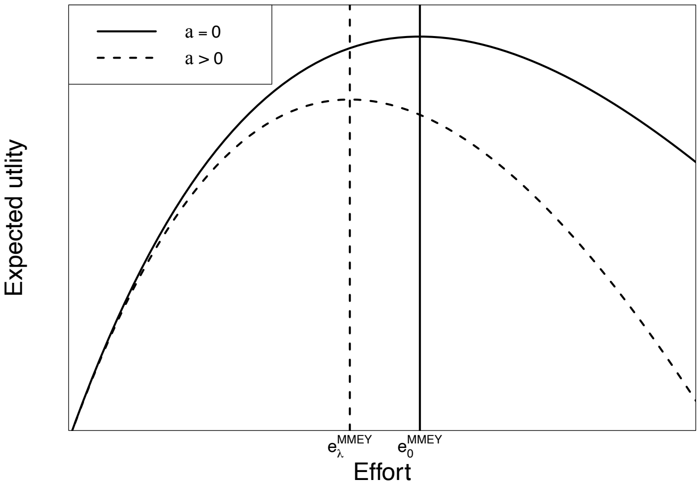


Figure 1: Expected utility versus effort in a risk-neutral (solid line) and in a risk-averse (dashed line) fishery. In the risk-neutral case (a = 0), expected utility is equal to expected profits.

3.2 At MMEY, risk aversion mitigates biological overexploitation

As risk aversion reduces the optimal fishing effort, it is expected to alleviate overexploitation and therefore promote biodiversity conservation. The following proposition claims that for each species there exists a level of risk aversion which avoids the overexploitation of this species at a risk-averse MMEY. A corollary is that there exists a level of risk aversion that precludes biological overexploitation of all species, and therefore guarantees the conservation of the entire ecosystem at MMEY. Let us define the associated risk aversion level

$$\mathrm{a}_{sus}(i) = \max\left(0, \frac{\sum_j p_j h'_j(e_i^{\mathrm{MSY}}) - \bar{c}}{\sigma_c^2 e_i^{\mathrm{MSY}}}\right), \tag{17}$$

with

$$\mathrm{a}_{sus} = \max_{i=1,\dots,N} \mathrm{a}_{sus}(i) \tag{18}$$

We obtain the following proposition:

Proposition 2. *For every species i, the level of risk aversion $\mathrm{a}_{sus}(i)$ defined in (17) is such that for all $\mathrm{a} \geq \mathrm{a}_{sus}(i)$, species i is not overharvested at MMEY:*

$$e_{\mathrm{a}}^{\mathrm{MMEY}} \leq e_i^{\mathrm{MSY}}. \tag{19}$$

Furthermore, the level of risk aversion a_{sus} defined in (18) promotes the biological sustainability of all species.

Proof. It relies on first order optimality conditions and concavity of the objective function. See Appendix A.1 for the details.

Figure 2 exemplifies in a three species fishery the levels of viable risk aversion $\mathrm{a}_{sus}(i)$ with respect to the coefficient of variation $\sigma_c^2/\bar{c}$ capturing cost uncertainty. For every value of $\sigma_c^2/\bar{c}$, we compute a value of $\mathrm{a}_{sus}(1)$ (respectively $\mathrm{a}_{sus}(2)$) that is sufficient to ensure that $e^{\mathrm{MMEY}} \leq e_1^{\mathrm{MSY}}$ (respectively $e^{\mathrm{MMEY}} \leq e_2^{\mathrm{MSY}}$). We do not define a sustainable risk aversion level for species 3, as this species is underharvested even in the risk-neutral situation. According to Proposition 2, the risk aversion level that avoids overexploitation of all species is $\mathrm{a}_{sus} = \mathrm{a}_{sus}(1)$.

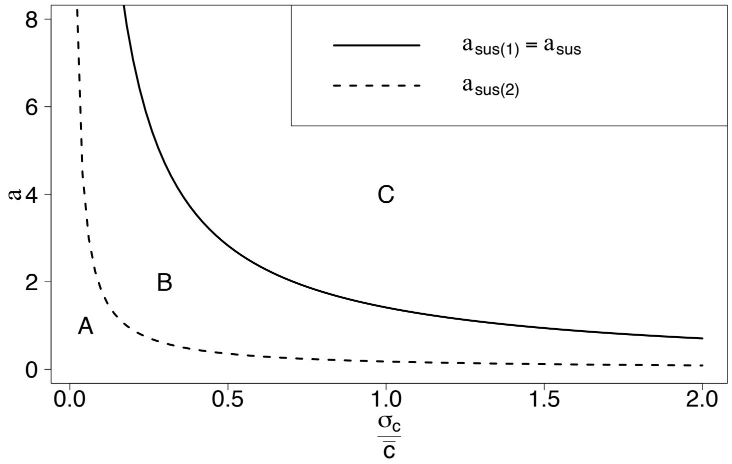


Figure 2: Sustainable risk aversion level a_{sus} for two species in a three species fishery. The dynamics is based on a logistic growth and parameters set to $r_1 = 1$, $r_2 = 2$, $r_3 = 20$, $q_1 = q_2 = q_3 = 1$, $k_1 = k_2 = k_3 = 1$, $p_1 = p_2 = p_3 = 2$, $\bar{c} = 1$. In zone A only species 3 is not overharvested; in zone B species 2 and 3 are not overharvested; in zone C none of the three species is overharvested.

3.3 At MMEY, risk aversion mitigates economic overexploitation

We can apply similar reasonings regarding economic overexploitation using MEY instead of MSY. At MMEY, species are economically overexploited if the MMEY effort is larger than their monospecific MEY effort, as defined in (16). As shown below, risk aversion can also reduce economic overexploitation. Let us define the associated risk aversion level

$$\mathrm{a}_{eff}(i) = \max\left(0, \frac{\sum_j p_j h'_j(e_i^{\mathrm{MEY}}) - \overline{c}}{\sigma_c^2 e_i^{\mathrm{MEY}}}\right),$$

with

$$\mathrm{a}_{eff} = \max_{i=1,\ldots,N} \mathrm{a}_{eff}(i) \tag{20}$$

We obtain the following proposition:

Proposition 3. *For every species i, the risk aversion $\mathrm{a}_{eff}(i)$ is such that for all $\mathrm{a} \geq \mathrm{a}_{eff}(i)$, species i is not economically overharvested:*

$$e_{\mathrm{a}}^{\mathrm{MMEY}} \leq e_i^{\mathrm{MEY}}. \tag{21}$$

Furthermore, at the risk aversion level a_{eff} none of the species is economically overexploited.

Proof. It relies again on first order optimality conditions. See Appendix A.1.

According to Proposition 3, risk aversion may reduce the risk of economic overexploitation in the fishery. Similarly to Proposition 2, this reveals that risk aversion reduces the pressure on stocks, hence improving economic performance.

3.4 At MMEY, too high risk aversion levels restrict food supply

As the MMEY fishing effort is reduced by risk aversion, it gets closer to the individual MSY effort of biologically overharvested species, thus increasing their catches. On the contrary, it moves away from the individual MSY effort of biologically underharvested species, thus reducing their catches. Thus for high risk aversion levels above the global sustainable threshold a_{sus} defined in (18), the catches of every species decline with risk aversion. It results the following proposition.

Proposition 4. *For high risk aversion levels* $\mathrm{a} \geq \mathrm{a}_{sus}$*, at risk-averse MMEY, the optimal catches* $h_i(e_{\mathrm{a}}^{\mathrm{MMEY}})$ *of every species decrease with risk aversion.*

Proof. See Appendix A.1.

This result highlights the potential negative consequences of high risk aversion levels in terms of food production and security. Such an outcome may alter the acceptability of this risk-averse strategy for small scale fisheries where seafood production is critical for local subsistence and food security.

3.5 At MMEY, risk aversion limits expected profit

As fishing efforts are reduced, it also turns out that a risk-averse attitude leads to a decline in expected profit. This negative effect on expected profit increases with risk aversion.

Proposition 5. *The expected profit at the optimal risk-averse effort* $\mathbb{E}[\pi(t, e_{\mathrm{a}}^{\mathrm{MMEY}})]$ *is a decreasing function of the risk aversion level* a*. In particular,* $\mathbb{E}[\pi(t, e_{\mathrm{a}}^{\mathrm{MMEY}})] \leq \mathbb{E}[\pi(t, e_{0}^{\mathrm{MMEY}})]$*.*

Proof. See Appendix A.1.

This proposition is illustrated in Figure 1, where expected profits are represented by the solid line. Shifting from a risk-neutral to a risk-averse MMEY effort leads to a decrease in expected (mean) profits. Risk averse policies thus imply losses in the average possible (random) profits. This illustrates the well-known trade-off between mean-related expectations and risk-related variance or standard deviation (Sanchirico et al., 2008; Gourguet et al., 2014). But note that for risk-averse agents (dashed line in Figure 1), this shift brings higher utility levels, including lower risk and variance.

4 Case study: the South East Fishery in Australia

We illustrate the analytical findings of previous section 'Results' with the Australian South East Fishery (hereafter called SEF). The SEF is a multispecies fishery that plays a major socio-economic role in the coastal communities of south-east Australia. The fishing fleet is mainly composed of trawlers (51 vessels at sea in 2014) that catch about 18000 tons of multiple fish species per year, corresponding to an approximate value of 40 million AUD. In what follows, we first describe the method to calibrate the bio-economic model for the SEF. We then analyse the performance of risk-averse MMEY in terms of profitability, biodiversity conservation, production and diversification. This allows us to exemplify the potential beneficial impacts of risk aversion on bio-economic sustainability.

4.1 Calibration of the bio-economic model

Following existing models for the SEF (Pascoe et al., 2015), the dynamics is based here on a Gompertz growth namely

$$r_i(x) = r_i(\log(k) - \log(x)).$$

To identify the parameters (r_i, k_i, q_i) of dynamics (1) for the different species, we use the method described in Sporcic and Haddon (2016), that defines standardized fishing zones and sums up catch data for each year. We use data on total catch $h(t)$ per year for each species and on the value of catch rates $h(t)/e$. Following Haddon (2010), for each species we choose parameters that minimize the sum of squared residuals between catch data and surplus production models. We select the 8 species that display the best fits. Supplementary details on the calibration method can be found in Appendix A.2.

Price data arise from the *Sydney fish market* for period 1994 to 2008. Daily prices are converted to average monthly prices. Cost data are derived from the *Australian Bureau of Agricultural and Resource Economics and Sciences (ABARES)*. We rely on cost data from 2002 to 2012, and on the total numbers of days at sea for trawlers in year 2009. We deduce an average cost per day and a variance of costs per day for the whole SEF. The variance of costs is about four times higher than the mean value. This could be related to the 30% share of fuel costs in the total costs of the fishery, which also take crew costs, maintenance costs and operational costs into account (Cheilari et al., 2013). The economic environment of the fishery has been coping with important changes due to the substantial oil market fluctuations during 2008 and the economic crisis. Our study is especially relevant in this context of highly varying costs for the fishery. Estimated bio-economic parameters are summarized in Table 1.

Table 1: Calibrated parameters of the selected species for the case study. The estimated average cost of effort is $\overline{c} = 96$ AUD/day while the estimated variance is $\sigma_c^2 = 390$ (AUD/kg)^2. The coefficient of variation in the SEF market is thus $\frac{\sigma_c}{\overline{c}} = 21\%$.

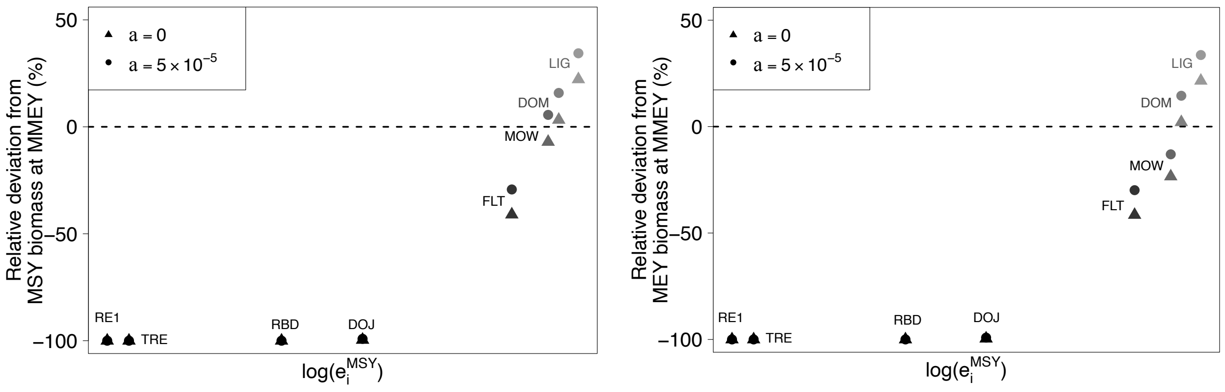
Species i	Abbreviations	growth rate r_i (%/year)	Carrying capacity k_i (t)	Catchability q_i (/year)	Price p_i (AUD/kg)
Ocean Perch	RE1	4.92	12297	1.18×10^{-4}	4.47
Silver Trevally	TRE	0.31	1	6.01×10^{-6}	2.91
Ribaldo	RBD	0.35	1	1.54×10^{-6}	3.79
John Dory	DOJ	2.45	27060	4.90×10^{-6}	8.91
Flathead	FLT	13.34	164315	6.24×10^{-6}	3.86
Morwong	MOW	2.07	66243	6.82×10^{-7}	3.07
Mirror Dory	DOM	14.54	193463	4.31×10^{-6}	3.31
Ling	LIG	14.54	269285	3.56×10^{-6}	6.12

4.2 Biological and economic overexploitation

To illustrate the impact of risk aversion on the SEF, we first compare the effect of risk-neutral and risk-averse situations on biological and economic overexploitation.

The impact of risk aversion on the biological overexploitation of individual species is shown in Figure 3a. We compare the MSY biomass of all harvested species to their biomass at MMEY. To achieve this, we define the deviation from MSY biomass of species i as the difference between MSY biomass of species i and MMEY biomass, normalized by the MSY biomass of species i. Hence, if the deviation is negative, species i is biologically overharvested at MMEY. As expected from Proposition 2, risk aversion reduces the harvesting pressure on all stocks. In particular, it leads to a interesting improvement in the exploitation status of Flathead, Ling, Morwong and Mirror Dory (right hand side of the graph). For instance, Mirror Dory is overharvested in a risk-neutral situation but becomes underharvested with risk aversion. John Dory, Ribaldo, Silver Trevally and Ocean Perch display very low biomass in the risk-neutral situation, so that the relative improvement with risk aversion is not significant.

The impact of risk aversion on economic overexploitation is shown in Figure 3b. To compare the MEY biomass of all species to their biomass at MMEY, we similarly define the deviation of species i from MEY biomass at risk averse MMEY. As expected from Proposition 3, risk aversion reduces the economic overexploitation of all stocks. Again, risk aversion leads to an important improvement in the exploitation status of Flathead, Ling, Morwong and Mirror Dory. The improvement in the exploitation status of John Dory, Ribaldo, Silver Trevally and Ocean Perch is also barely noticeable, due to the low individual optimal efforts of these species.



(a) (b)

Figure 3: Illustration of the impact of risk aversion on biological and economic overexploitation at MMEY. (a) Relative deviations from MSY biomass at MMEY in a risk-neutral and in a risk-averse scenario. (b) Relative deviations from MEY biomass at MMEY in a risk-neutral and in a risk-averse scenario. The darker the symbol of the species, the more endangered it is.

4.3 Policy implications

We then illustrate the trade-offs and synergies between ecological, social and economic objectives with a concluding multi-criteria analysis and graph. As shown in previous Section 4.2, a risk-averse management of the SEF could induce changes in bio-economic performances. This section is dedicated to a better understanding of how these results can help to design sustainable management and policies.

The consequences of risk aversion on multiple management indicators is synthesized in Figure 4, which compares the bio-economic performances of MMEY in a risk-neutral case (a $= 0$), in a low risk aversion case (a $= 5.10^{-5}$), and in a high risk aversion case (a$_{sus}(DOJ) = 4.10^{-3}$). In the latter case, the level of risk aversion was set to avoid the overexploitation of John Dory, according to Proposition 2. This also allows for a sustainable exploitation of Lings, Mirror Dory, Morwong and Flathead, as the monospecific MSY effort of these species is higher than the MSY effort of John Dory. As the sustainable exploitation of Ocean Perch, Ribaldo and Silver Trevally entails even higher levels of risk aversion, we chose to focus our analysis on the sustainable exploitation of John Dory.

Risk aversion reduces the MMEY effort, which implies a reduction in the number of active fishermen. As the effort decreases, expected revenues and total catches are also reduced. This finding is consistent with Propositions 1, 4 and 5. The impact of risk aversion on economic indicators strongly depends on the level of risk aversion: high levels of risk aversion entail a more than 60% reduction in expected profits and catches, while low levels of risk aversion induce moderate decreases.

In Figure 4, economic security is defined here as the inverse of the standard deviation of profits namely $\left(\mathrm{a}\,\sigma_c^2 e^2\right)^{-1}$. Thus, high values capture low profit variability. As expected, this metric increases with risk aversion. In particular, the value of the economic security index in the low risk aversion case is only 20 % that of the high risk aversion case.

To investigate the impact of risk averse MMEY strategy on biodiversity, Figure 4 also plots the values of the Simpson index, which is defined as the inverse of the sum of squared species shares:

$$S(e) = \left(\sum_i \left(\frac{x_i(e)}{\sum_j x_j(e)} \right)^2 \right)^{-1} \tag{22}$$

This index is often used to quantify biodiversity and takes into account the number of species, as well as the share of every species in terms of abundance. High Simpson indices imply large numbers of species or an even distribution among species. It turns out that the Simpson index also benefits from risk aversion. This is consistent with the results regarding the biological overexploitation of species.

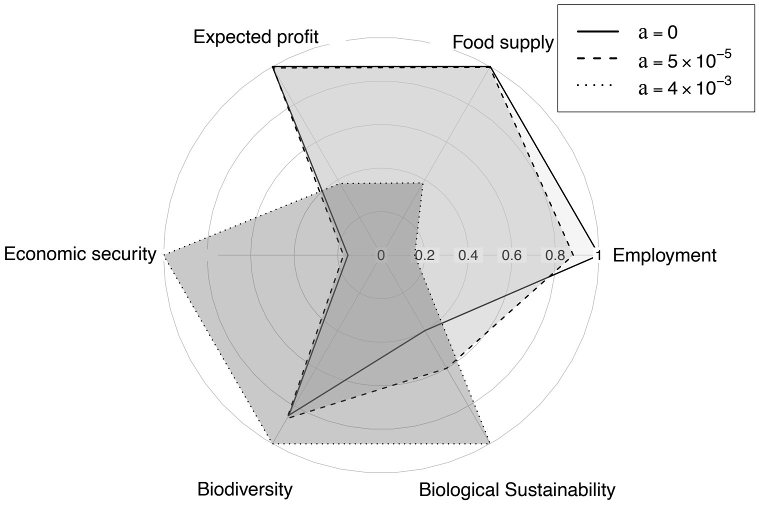


Figure 4: Illustration of the different trade-offs at stake in the South East Fishery for three different levels of risk aversion (a $= 0$, a $= 5 \times 10^{-5}$ and a $= \mathrm{a}_{sus(DOJ)} = 4 \times 10^{-3}$). Each indicator is normalized relatively to its maximum value.

The relationship between the centroid distance from the origin of the radar chart and the level of risk aversion is computed in Figure 5. The centroid is the arithmetic mean position of all indicators, and it thus describes the balance between management objectives. Reduced distance of the centroid from the origin informs on an improved balance between management objectives. In Figure 5, we plot the inverse of the centroid distance from the origin. It turns out that the balance between indicators is maximized at an intermediate level of risk aversion (a $= 7.9 \times 10^{-4}$).

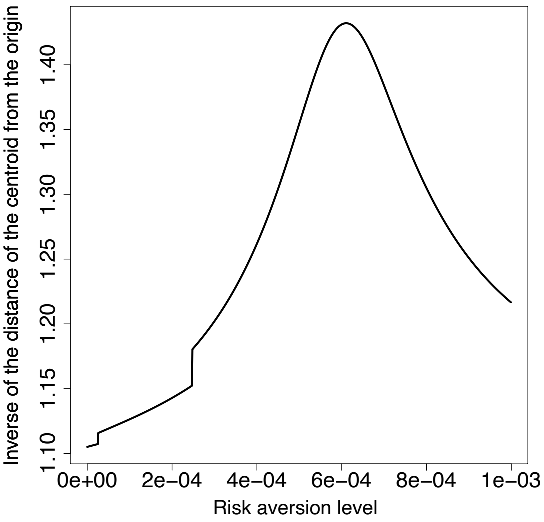


Figure 5: Relationship between the balance between management objectives (measured as the inverse of the centroid distance from the origin) and the risk aversion level a of the policy maker. The maximum of the curve (corresponding to the minimum centroid distance from the origin) is obtained for an intermediate risk aversion level (7.9×10^{-4}).

This case study on the South East Fishery highlights the potential benefits of risk-averse policies when dealing with economic uncertainty. In particular, intermediate levels of risk aversion allow to balance ecological and economic performances, and could thus facilitate the implementation of an ecosystem-based management and sustainability for this fishery.

5 Conclusion

In this article, we examine the consequences of uncertainty and risk aversion on sustainable harvesting strategies in mixed fisheries involving technical interactions. Using a bio-economic model of multiple species harvested by a single fleet and a quadratic utility function, we analyze the impacts of risk-averse attitudes on the outcomes of a multispecies maximum economic yield (MMEY) policy and extend some results from the deterministic case (Clark, 2006; Tromeur and Doyen, 2018). In particular, we show how risk aversion can foster biodiversity by mitigating overharvest and extinction of species while still maximising utility and profitability. These results bring novel insights into the potentially beneficial impacts of precautionary attitudes in mixed fisheries run under MEY policies.

Many fisheries are currently managed by specific regulating agencies, at regional or national levels. An example of such an organization is the Australian Fisheries Management Authority, which is based on the involvement of stakeholders, such as fishery industry members or conservation agencies members (Smith et al., 1999). This co-management may allow to account for fishermen's aversion towards economic risk (Brick et al., 2012). This is especially relevant as fuel costs have experienced large fluctuations in the last decade (Cheilari et al., 2013). Accounting for uncertainty and risk aversion is thus an emerging challenge in fisheries management, and a central objective of the ecosystem-based approach (Doyen et al., 2017).

To adress such issues, we first show that uncertainty in operating costs reduces the optimal MMEY effort as risk aversion increases. Risk aversion in MMEY-driven fisheries thus mitigates the overexploitation of species with low productivity and low value, and helps to maintain biodiversity in fisheries. Thus, accounting for uncertainty and risk aversion in setting effort limits can actually improve the ecological performance of fisheries. This finding is coherent with Andersen (1982), who showed that in a fishery with price uncertainty and risk aversion, the total fishing effort is reduced as the variance of the price increases. Baldursson and Magnusson (1997) also showed that risk aversion could lead to reduced fishing efforts. A similar synergy between reduced variability of profits and ecological performance has been pointed out in agroecosystems by Mouysset et al. (2013). This synergy is due to a diversification of regional land uses, often associated with a focus on less intensive and less profitable (on average) agricultural practices. Thus, as in our study, risk aversion reduces exploitation intensity and profitability, which in turn increases biodiversity. This result also suggests that potential stabilizing subsidies on costs could reduce the effect of risk aversion and thus increase harvesting pressures. In that sense, such stabilizing subsidies could be accounted for as *capacity-enhancing*, following the classification by Sumaila et al. (2010).

Second, we show that risk aversion brings lower levels of mean or expected economic performance. Accounting for uncertainty and risk aversion in fisheries management may therefore hamper the economic efficiency of fisheries. We thus highlight a trade-off between economic performance and economic security, which is associated with a trade-off between economic performance and ecological performance (Cheung and Sumaila, 2008). In addition to lower economic performance, risk aversion induces lower expected (mean) yields. We suggest that risk aversion may partly explain the current problem of undercaught TAC (Total Allowable Catch) the South East Fishery is experiencing. Indeed, according to the Australian Fisheries Management Authority, at the end of 2015, 23 of the 34 species groups under TAC were less than 50% caught. Of the major quota species, only four had catches above 80% of the TACs. This situation is obviously caused by numerous different reasons, but the adoption of risk-averse approaches by local fishery managers may be one of them.

Third, we find that accounting for risk aversion and cost uncertainty can help to manage the trade-offs and synergies between multiple management objectives. In particular, we show that in the Australian South East Fishery intermediate levels of risk aversion improve the balance between management objectives. As risk aversion fosters bio-economic sustainability in such multi-objective contexts, it should be considered a key parameter in ecosystem-based fisheries management approaches (Pikitch et al., 2004).

In this study, we assumed that all species are ecologically independent, while harvesting some species is known to have cascading effects in trophic networks (Finnoff and Tschirhart, 2003). Maximizing total yield in a predator-prey community can for instance induce severe predator depletion and reduce the resilience of the harvested system (Tromeur and Loeuille, 2017). Furthermore, uncertainty in the dynamics of harvested predator-prey communities may lead to unstable dynamics and extinction (Tu and Wilman, 1992). Accounting for other ecological complexities and uncertainties is part of the future extensions of the present work.

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References

Andersen, P., 1982. Commercial fisheries under price uncertainty. Journal of Environmental Economics and Management 9 (1), 11–28.
URL `http://www.sciencedirect.com.ezproxy1.lib.asu.edu/science/article/pii/0095069682900031`

Baldursson, F., Magnusson, G., sep 1997. Portfolio fishing. Scandinavian Journal of Economics 99 (3), 389–403.
URL `http://doi.wiley.com/10.1111/1467-9442.00070`

Beddington, J., May, R., 1977. Harvesting natural populations in a randomly fluctuating environment. Nature 197, 463–465.

Binswanger, H., aug 1980. Attitudes Toward Risk: Experimental Measurement in Rural India. American Journal of Agricultural Economics 62 (3), 395–407.
URL `https://onlinelibrary.wiley.com/doi/abs/10.2307/1240194`

Brick, K., Visser, M., Burns, J., jan 2012. Risk aversion: Experimental evidence from South African fishing communities. American Journal of Agricultural Economics 94 (1), 133–152.
URL `https://academic.oup.com/ajae/article-lookup/doi/10.1093/ajae/aar120`

Charles, A., Munro, G., 1985. Irreversible investment and optimal fisheries management: A stochastic analysis. Marine Resource Economics 1 (3), 247–264.

Cheilari, A., Guillen, J., Damalas, D., Barbas, T., jul 2013. Effects of the fuel price crisis on the energy efficiency and the economic performance of the European Union fishing fleets. Marine Policy 40 (1), 18–24.
URL `http://dx.doi.org/10.1016/j.marpol.2012.12.006http://linkinghub.elsevier.com/retrieve/pii/S0308597X12002473`

Cheung, W., Sumaila, U., may 2008. Trade-offs between conservation and socio-economic objectives in managing a tropical marine ecosystem. Ecological Economics 66 (1), 193–210.
URL `http://linkinghub.elsevier.com/retrieve/pii/S0921800907004764`

Clark, C., 2006. The worldwide crisis in fisheries: Economic models and human behaviour. Cambridge University Press.

Clark, C., 2010. Mathematical bioeconomics: The mathematics of conservation, 3rd Edition. Wiley.

Dichmont, C., Pascoe, S., Kompas, T., Punt, A., Deng, R., 2010. On implementing maximum economic yield in commercial fisheries. Proceedings of the National Academy of Sciences 107 (1), 16–21.

Dowling, N., Dichmont, C., Haddon, M., Smith, D., Smith, A., Sainsbury, K., nov 2015. Guidelines for developing formal harvest strategies for data-poor species and fisheries. Fisheries Research 171 (1), 130–140.

Doyen, L., Béné, C., Bertignac, M., Blanchard, F., Cissé, A., Dichmont, C., Gourguet, S., Guyader, O., Hardy, P.-Y., Jennings, S., Little, L., Macher, C., Mills, D., Noussair, A., Pascoe, S., Pereau, J.-C., Sanz, N., Schwarz, A.-M., Smith, A., Thébaud, O., nov 2017. Ecoviability for ecosystem-based fisheries management. Fish and Fish-

eries 18 (6), 1056–1072.
URL `http://doi.wiley.com/10.1111/faf.12224`

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

Edwards, S., Link, J., Rountree, B., jul 2004. Portfolio management of wild fish stocks. Ecological Economics 49 (3), 317–329.
URL `http://linkinghub.elsevier.com/retrieve/pii/S092180090400148X`

Eeckhoudt, L., Gollier, C., Treich, N., apr 2005. Optimal consumption and the timing of the resolution of uncertainty. European Economic Review 49 (3), 761–773.
URL `https://linkinghub.elsevier.com/retrieve/pii/S0014292103000692`

Ewald, C.-O., Wang, W.-K., may 2010. Sustainable yields in fisheries: uncertainty, risk-aversion, and mean-variance analysis. Natural Resource Modeling 23 (3), 303–323.
URL `http://doi.wiley.com/10.1111/j.1939-7445.2010.00065.x`

FAO, 2016. The state of world fisheries and aquaculture. Tech. rep., Food and Agriculture Organization.

Finnoff, D., Tschirhart, J., 2003. Harvesting in an eight-species ecosystem. Journal of Environmental Economics and Management 45, 589–611.

Fulton, E., Punt, A., Dichmont, C., Gorton, R., Sporcic, M., Dowling, N., Little, L., Haddon, M., Klaer, N., Smith, D., nov 2016. Developing risk equivalent data-rich and data-limited harvest strategies. Fisheries Research 183, 574–587.
URL `https://linkinghub.elsevier.com/retrieve/pii/S0165783616302168`

Gourguet, S., Thébaud, O., Dichmont, C., Jennings, S., Little, L., Pascoe, S., Deng, R., Doyen, L., 2014. Risk versus economic performance in a mixed fishery. Ecological

Economics 99, 110–120.
URL `http://dx.doi.org/10.1016/j.ecolecon.2014.01.013`

Grafton, R., Doyen, L., Béné, C., Borgomeo, E., Brooks, K., Chu, L., Cumming, G., Dixon, J., Dovers, S., Garrick, D., Helfgott, A., Jiang, Q., Katic, P., Kompas, T., Little, L., Matthews, N., Ringler, C., Squires, D., Steinshamn, S., Villasante, S., Wheeler, S., Williams, J., Wyrwoll, P., oct 2019. Realizing resilience for decision-making. Nature Sustainability 2 (10), 907–913.
URL `http://www.nature.com/articles/s41893-019-0376-1`

Grafton, R., Kompas, T., Chu, L., Che, N., jul 2010. Maximum economic yield. Australian Journal of Agricultural and Resource Economics 54 (3), 273–280.
URL `http://doi.wiley.com/10.1111/j.1467-8489.2010.00492.x`

Guillen, J., Macher, C., Merzéréaud, M., Bertignac, M., Fifas, S., Guyader, O., 2013. Estimating MSY and MEY in multi-species and multi-fleet fisheries, consequences and limits: An application to the Bay of Biscay mixed fishery. Marine Policy 40, 64–74.

Haddon, M., 2010. Modelling and quantitative methods in fisheries. CRC press.

Hoshino, E., Pascoe, S., Hutton, T., Kompas, T., Yamazaki, S., nov 2017. Estimating maximum economic yield in multispecies fisheries: a review. Reviews in Fish Biology and Fisheries.
URL `http://link.springer.com/10.1007/s11160-017-9508-8`

Legović, T., Geček, S., 2010. Impact of maximum sustainable yield on independent populations. Ecological Modelling 221, 2108–2111.

McWhinnie, S., 2009. The tragedy of the commons in international fisheries: An empirical examination. Journal of Environmental Economics and Management 57 (3), 321–333.

Mouysset, L., Doyen, L., Jiguet, F., 2013. How does economic risk aversion affect biodiversity? Ecological Applications 23 (1), 96–109.

Mueter, F., Megrey, B., 2006. Using multi-species surplus production models to estimate ecosystem-level maximum sustainable yields. Fisheries Research 81 (2-3), 189–201.

Pascoe, S., Hutton, T., Thébaud, O., Deng, R., Klaer, N., Vieira, S., 2015. Setting economic target reference points for multiple species in mixed fisheries. Tech. rep., Fisheries Research and Development Corporation.

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

Pikitch, E., Santora, C., Babcock, E., Bakun, A., Bonfil, R., Conover, D., Dayton, P., Doukakis, P., Fluharty, D., Heneman, B., Houde, E., Link, J., Livingston, P., Mangel, M., McAllister, M., Pope, J., Sainsbury, K., 2004. Ecosystem-based fishery management. Science 305 (5682), 346–347.

Plagányi, E., 2007. Models for an ecosystem approach to fisheries. Tech. rep., FAO, Rome.

Sanchirico, J., Smith, M., Lipton, D., 2008. An empirical approach to ecosystem-based fishery management. Ecological Economics 64 (3), 586–596.

Scandol, J., Holloway, M., Gibbs, P., Astles, K., jul 2005. Ecosystem-based fisheries management: An Australian perspective. Aquatic Living Resources 18 (3), 261–273. URL `http://www.alr-journal.org/10.1051/alr:2005031`

Schuhbauer, A., Sumaila, U. R., apr 2016. Economic viability and small-scale fisheries âĂŤ A review. Ecological Economics 124, 69–75.
URL `https://linkinghub.elsevier.com/retrieve/pii/S092180091630132X`

Sethi, G., Costello, C., Fisher, A., Hanemann, M., Karp, L., sep 2005. Fishery management under multiple uncertainty. Journal of Environmental Economics and Management 50 (2), 300–318.
URL `http://linkinghub.elsevier.com/retrieve/pii/S0095069605000057`

Smith, A., Sainsbury, K., Stevens, R., dec 1999. Implementing effective fisheries-management systems âĂŞ management strategy evaluation and the Australian partnership approach. ICES Journal of Marine Science 56 (6), 967–979.
URL `http://icesjms.oxfordjournals.org/content/56/6/967.abstracthttps://academic.oup.com/icesjms/article-lookup/doi/10.1006/jmsc.1999.0540`

Sporcic, M., Haddon, M., 2016. Catch rate standardizations for selected SESSF Species (data to 2015). Tech. rep.

Sumaila, U., Khan, A., Dyck, A., Watson, R., Munro, G., Tydemers, P., Pauly, D., 2010. A bottom-up re-estimation of global fisheries subsidies. Journal of Bioeconomics 12 (3), 201–225.

Thébaud, O., Doyen, L., Innes, J., Lample, M., Macher, C., Mahévas, S., Mullon, C., Planque, B., Quaas, M., Smith, T., Vermard, Y., jan 2014. Building ecological-economic models and scenarios of marine resource systems: Workshop report. Marine Policy 43, 382–386.

Tromeur, E., Doyen, L., jul 2018. Optimal harvesting policies threaten biodiversity in mixed fisheries. Environmental Modeling & Assessment.
URL `http://link.springer.com/10.1007/s10666-018-9618-2`

Tromeur, E., Loeuille, N., dec 2017. Balancing yield with resilience and conservation objectives in harvested predator-prey communities. Oikos 126 (12), 1780–1789.
URL `http://doi.wiley.com/10.1111/oik.03985`

Tu, P., Wilman, E., 1992. A generalized predator-prey model : Uncertainty and management. Journal of Environmental Economics and Management 23, 123–138.

Tyedmers, P., 2004. Fisheries and energy use. In: Encyclopedia of energy. Elsevier, pp. 683—-693.

Walters, C. J., Christensen, V., Martell, S. J., Kitchell, J., 2005. Possible ecosystem impacts of applying MSY policies from single-species assessment. ICES Journal of Marine Science 62 (3), 558–568.

A Appendix

A.1 Proofs of propositions

A.1.1 Proof of Proposition 1

Let $e_{\mathrm{a}}^{\mathrm{MMEY}}$ be the effort (unique from assumption) that maximizes the objective function $V_{\mathrm{a}}(e)$ defined in (10), and let $\Delta a > 0$ be a variation of a. We have

$$V_{\mathrm{a}+\Delta a}(e_{\mathrm{a}}^{\mathrm{MMEY}}) = V_{\mathrm{a}}(e_{\mathrm{a}}^{\mathrm{MMEY}}) - \frac{\Delta a}{2}\sigma_c^2(e_{\mathrm{a}}^{\mathrm{MMEY}})^2$$

As $V_{\mathrm{a}+\Delta a}$ reaches its maximum at $e_{\mathrm{a}+\Delta a}^{\mathrm{MMEY}}$, we have

$$V_{\mathrm{a}+\Delta a}(e_{\mathrm{a}+\Delta a}^{\mathrm{MMEY}}) \geq V_{\mathrm{a}+\Delta a}(e_{\mathrm{a}}^{\mathrm{MMEY}})$$

or equivalently

$$V_{\mathrm{a}}(e_{\mathrm{a}+\Delta a}^{\mathrm{MMEY}}) - \frac{\Delta a}{2}\sigma_c^2(e_{\mathrm{a}+\Delta a}^{\mathrm{MMEY}})^2 \geq V_{\mathrm{a}+\Delta a}(e_{\mathrm{a}}^{\mathrm{MMEY}}).$$

Finally, as

$$V_{\mathrm{a}}(e_{\mathrm{a}+\Delta a}^{\mathrm{MMEY}}) - V_{\mathrm{a}+\Delta a}(e_{\mathrm{a}}^{\mathrm{MMEY}}) = \frac{\Delta a}{2}\sigma_c^2(e_{\mathrm{a}}^{\mathrm{MMEY}})^2,$$

we obtain the following relationship:

$$\frac{\Delta a}{2}\sigma_c^2(e_{\mathrm{a}}^{\mathrm{MMEY}})^2 \geq \frac{\Delta a}{2}\sigma_c^2(e_{\mathrm{a}+\Delta a}^{\mathrm{MMEY}})^2$$

As we consider that $\Delta a > 0$ and that fishing efforts are positive, we deduce that

$$e_{\mathrm{a}}^{\mathrm{MMEY}} \geq e_{\mathrm{a}+\Delta a}^{\mathrm{MMEY}},$$

and $\mathrm{a} \to e_{\mathrm{a}}^{\mathrm{MMEY}}$ is thus a decreasing function. As a consequence, the risk-neutral MMEY effort is necessarily larger than a risk-averse MMEY effort : $e_0^{\mathrm{MMEY}} \geq e_{\mathrm{a}}^{\mathrm{MMEY}}, \forall\, \mathrm{a} \geq 0$.

A.1.2 Proof of Proposition 2

We denote by $V'_{\mathrm{a}}(e)$ the first derivative of the objective function $V_{\mathrm{a}}(e)$:

$$V'_{\mathrm{a}}(e) = \sum_i p_i h_i^{'}(e) - \mathrm{a}\,\sigma_c^2 e - \overline{c}. \tag{23}$$

where catch at equilibrium $h_i(e)$ is defined in equation (4). Moreover, as functions $h_i(.)$ are assumed to be concave and thus satisfy $h''_i(e) < 0$, we know that the derivative of $V'_{\mathrm{a}}(e)$, which can be written

$$V''_{\mathrm{a}}(e) = \sum_i p_i h''_i(e) - \mathrm{a}\,\sigma_c^2, \tag{24}$$

is negative.

Let us now identify the expression of the sustainable risk aversion $\mathrm{a}_{sus}(i)$ of the different species i. By definition, species i is not overexploited if $e_\lambda^{\mathrm{MMEY}} \leq e_i^{\mathrm{MSY}}$. As V'_{a} is decreasing with respect to effort e (since V_{a} is concave), we deduce that this is equivalent to $V'_{\mathrm{a}}(e_i^{\mathrm{MSY}}) \leq V'_{\mathrm{a}}(e_\lambda^{\mathrm{MMEY}}) = 0$. It follows :

$$\sum_j p_j h'_j(e_i^{\mathrm{MSY}}) - \overline{c} \leq \mathrm{a}\,\sigma_c^2 e_i^{\mathrm{MSY}}.$$

We deduce that

$$\mathrm{a}_{sus}(i) = \max\left(0, \frac{\sum_j p_j h'_j(e_i^{\mathrm{MSY}}) - \overline{c}}{\sigma_c^2 e_i^{\mathrm{MSY}}}\right).$$

A.1.3 Proof of Proposition 3

Species i is not economically overharvested whenever $e_{\mathrm{a}}^{\mathrm{MMEY}} \leq e_i^{\mathrm{MEY}}$. This condition is equivalent to $\sum_j p_j h'_j(e_i^{\mathrm{MEY}}) - \overline{c} - \mathrm{a}\,\sigma_c^2 e_i^{\mathrm{MEY}} \leq 0$ which holds true for sufficiently high values of risk aversion. As for Proposition 2, we thus have

$$\mathrm{a}_{eff}(i) = \max\left(0, \frac{\sum_j p_j h'_j(e_i^{\mathrm{MEY}}) - \overline{c}}{\sigma_c^2 e_i^{\mathrm{MEY}}}\right).$$

A.1.4 Proof of Proposition 4

As by definition e_i^{MSY} is the optimal effort of catches at equilibrium $h_i(e)$ and $h_i(.)$ is concave, we have

$$\frac{\partial h_i(e^*)}{\partial e} \geq 0 \iff e < e_i^{\text{MSY}}$$

Hence :

$$\frac{\partial h_i(e^*)}{\partial \,\mathrm{a}} = \frac{\partial h_i}{\partial e}(e^*)\frac{\partial e^*}{\partial \,\mathrm{a}} \leq 0 \text{ when } e^* \leq e_i^{\text{MSY}}$$

Moreover, whenever $\mathrm{a} \geq \mathrm{a}_{sus}$, from previous Proposition 2, for every species we have $e^* \leq e_i^{\text{MSY}}$. We conclude that the catches of all species decrease in that case of high risk aversion.

A.1.5 Proof of Proposition 5

The risk neutral problem corresponds to the case where $\mathrm{a} = 0$ and consequently

$$V_0(e) = \mathbb{E}[\pi(t, e)].$$

As e_0^{MMEY} is a maximum for $V_0(e)$, it follows that the expected profit at $e_{\mathrm{a}}^{\text{MMEY}}$ is lower at e_0^{MMEY}, namely

$$\mathbb{E}[\pi(t, e_{\mathrm{a}}^{\text{MMEY}})] \leq \mathbb{E}[\pi(t, e_0^{\text{MMEY}})]$$

More generally, we have

$$\frac{\partial \mathbb{E}[\pi(t, e_{\mathrm{a}}^{\text{MMEY}})]}{\partial \,\mathrm{a}} = \frac{\partial \mathbb{E}[\pi(t, e)]}{\partial e}(e_{\mathrm{a}}^{\text{MMEY}})\frac{\partial e_{\mathrm{a}}^{\text{MMEY}}}{\partial \,\mathrm{a}}$$

From Proposition 1, we know that

$$\frac{\partial e_{\mathrm{a}}^{\text{MMEY}}}{\partial \,\mathrm{a}} < 0. \tag{a}$$

Let us now prove that $\dfrac{\partial \mathbb{E}[\pi(t, e)]}{\partial e}(e_{\mathrm{a}}^{\text{MMEY}}) > 0$. First, since $e_{\mathrm{a}}^{\text{MMEY}}$ is the optimal solution of program (10), we can use the first order condition

$$\frac{\partial V_{\mathrm{a}}(e)}{\partial e}(e_{\mathrm{a}}^{\mathrm{MMEY}}) = 0$$

Using the very definition of the utility function V_{a}, the criteria to optimize can be written

$$V_{\mathrm{a}}(e)) = \mathbb{E}[\pi(t,e)] - \frac{\mathrm{a}}{2}\sigma_c^2 e^2$$

Thus we deduce

$$\begin{aligned}\frac{\partial \mathbb{E}[\pi(t,e))]}{\partial e}(e_{\mathrm{a}}^{\mathrm{MMEY}}) &= \frac{\partial V_{\mathrm{a}}(e)}{\partial e}(e_{\mathrm{a}}^{\mathrm{MMEY}}) + \mathrm{a}\,\sigma_c^2 e_{\mathrm{a}}^{\mathrm{MMEY}} \\ &= \mathrm{a}\,\sigma_c^2 e_{\mathrm{a}}^{\mathrm{MMEY}}\end{aligned}$$

Therefore,

$$\frac{\partial \mathbb{E}[\pi(t,e))]}{\partial e}(e_{\mathrm{a}}^{\mathrm{MMEY}}) \geq 0. \tag{b}$$

By virtue of inequalities (a) and (b), we conclude that

$$\frac{\partial \mathbb{E}[\pi(t, e_{\mathrm{a}}^{\mathrm{MMEY}})]}{\partial\, \mathrm{a}} \leq 0.$$

A.2 Case study

A.2.1 Calibration of models

The theoretical values of catches are compared with the observed values $h^{obs}(t)$ for years 2004 to 2014. We used data on catch per year $h^{obs}(t)$ and catch rates $h^{obs}(t)/e^{obs}(t)$, which gave us yearly values of effort. We were then able to compute our theoretical monospecific harvest from fish for the Gompertz dynamics

$$h_i(e) = q_i e x_i = q_i e k_i \exp(-q_i e / r_i).$$

The parameters (q_i, k_i, r_i) were estimated by minimizing the sum of squared differences between theoretical and observed values :

$$\min_{r_i,q_i,k_i\geq 0} ||h_i - h_i^{obs}||^2 = \min_{r_i,q_i,k_i\geq 0} \sum_{t=2004}^{2014} (h_i(t) - h_i^{obs}(t))^2.$$

We only selected species for which the residual sum of squares was smaller than 10 (see Table 2).

Table 2: Residual sum of squares of the selected species for the Gompertz model. Only species with residuals smaller than 10 were selected.

Species i	Abbreviations	Gompertz
Flathead	FLT	1.05
John Dory	DOJ	0.77
Ling	LIG	1.52
Mirror Dory	DOM	1.91
Morwong	MOW	0.78
Ocean Perch	RE1	0.5
Ribaldo	RBD	1.42
Silver Trevally	TRE	3.18