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A stochastic viability approach to ecosystem-based fisheries management

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Abstract:

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

Highlights

► Stochastic viability is a useful approach for ecosystem based fisheries management. ► It is applied to the hake and nephrops fisheries of the Bay of Biscay. ► The status quo strategy (2006) is not viable. ► Co-viability strategy can guarantee both ecological and economic sustainability.

Keywords: Ecosystem based fisheries management ; Viability ; Stochastic ; Nephrops ; Hake

1 Introduction

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

Indicators and their associated reference points are key to the implementation of such sustainability strategies. For example, the objective of the precautionary approach promoted by the International Council for the Exploitation of the Sea (ICES) is to maintain spawning stock biomass above a limit reference point Blim, while keeping fishing mortality below a limit Flim. To achieve this in a context of high uncertainty on the current level of both indicator and reference point, operational precautionary reference points Bpa and Fpa are used. This approach proposes to preserve a minimum quantity of reproducers to avoid recruitment accidents that would endanger the sustainability of the stock and consequently the fishery. Although such a precautionary approach has had positive effects in Europe on some severely depleted stocks, the overall state of European fish stocks remains grim. A first criticism of this approach is that it adopts a viewpoint which is too 'ichthyocentric', as it focuses on the conservation of fish populations and stocks only. Social and economic considerations are not included and left to the fishery managers' calls. Excluding these considerations from the assessment of trade-offs associated with alternative management strategies leaves managers with limited scientific ground on which to base decisions, a lack of clarity in the objectives pursued, and greater potential for stakeholder conflict and resistance to the implementation of management options, particularly in the face of uncertainty. Formal approaches to including multiple objectives in evaluation of management options, such as Management Strategy Evaluations (Sainsbury *et al.*, 2007), has been shown to positively contribute to agreement being reached on fisheries regulation options. Increasingly, it is also stressed that single stock assessments should be replaced by more complex multi-species and/or multi-fisheries analyses that account for interactions between species and/or fisheries. For example, in the United States, the National Marine Fisheries Service (NMFS) began to include ecosystem effects on stocks and fishery effects on ecosystems in its individual groundfish stock assessment reports to the North Pacific Fisheries Management Council in 2002. As part of its strategic goals, the NMFS has now replaced single-species management with ecosystem-based management, balancing ecological and social objectives. There is nowadays widespread acceptance that more integrated perspectives are needed to manage marine fisheries sustainably.

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

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

Northern hake is conjointly caught by several mixed demersal fisheries, and is an economically important species for several major European fishing fleets [2]. Spain accounts for the main part of the landings with nearly two thirds of landings in recent years, while French fleets contribute to a quarter of the total landings, and other countries (UK, Denmark, Ireland, Norway, Belgium, Netherlands, Germany, and Sweden) contributing small amounts (STECF, 2008). Total landings decreased steadily from 66,500t in 1989 to a low 35,000 tons in 1998, and fluctuated at around 40,000 tons since then. During the same period, the estimated biomass of reproducers (spawning stock biomass SSB) decreased to levels close to the minimum biomass level recommended by the ICES, $B_{\text{lim}} = 100,000$ tons. In 2003, the ICES considered that this stock presented a risk of collapse. Various management regulations were then introduced to attempt to restore the sustainability of the hake fishery. In particular a series of technical measures was proposed (EC Council Regu-

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

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

lations N 1162/2001, 2602/2001 and 494/2002) to improve the selectivity of the fishing gear and protect juveniles. Subsequently a hake recovery plan was introduced (Council regulation EC Reg. No 811/2004), recommending a reduction in fishing mortality to the precautionary level $F_{\text{pa}} = 0,25$ to allow for a recovery of the SBB above $B_{\text{pa}} = 140,000$ tons.

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

The viability (or viable control) approach aims at identifying desirable combinations of states and associated controls that ensure the 'good health', safety or effectiveness of the system (Béné *et al.*, 2001). By identifying the conditions that allow desirable objectives to be fulfilled over time, considering both present and future states of a renewable resources system, the viability approach conveys information relevant for policy and decision makers. Viability does not aim at identifying optimal or steady state paths for the co-dynamics of resources and exploitation, but instead, is closely related to the maximin, or Rawlsian, approach (Heal, 1998) with respect to intergenerational equity (Martinet & Doyen, 2007). Viability may in particular allow for the satisfaction of both economic and environmental constraints and is, in this respect, a multi-criteria approach (Baumgartner & Quaas, 2009). Tichit *et al.* (2007) shows how the so-called Population Viability Analysis (PVA) developed in conservation biology (e.g. Morris & Doak (2003)) addresses issues comparable to those of the viability approach. Viability analysis has been applied to renewable resources management and especially to fisheries (see, e.g., Béné & Doyen (2000); Béné *et al.* (2001); Doyen & Béné (2003); Eisenack *et al.* (2006); Martinet *et al.* (2007)), but also to broader (eco)-system dynamics (Mullon *et al.* (2004); Chapel *et al.* (2008); Doyen *et al.* (2007)). Cury *et al.* (2005) illustrate how the viability approach can potentially be useful to integrate ecosystem considerations into fisheries management. Relationships between viability, sustainable management objectives and reference points as adopted in the ICES precautionary approach are discussed in DeLara *et al.* (2007).

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

The paper is organized as follows. Section 2 describes the bio-economic model of the system, together with the ecological and economic constraints. Section 3 presents the results related to the PVA, EVA and CVA approaches. The following section discusses those results and explores the potential usefulness of co-viability in relation to ecosystem-based fisheries management. A series of conclusion follows.

2 The bio-economic model

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

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

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

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

where

- $M_{s,a}$ is the natural mortality rate of individuals of species s at age a;
- $F_{s,a,f}$ is the current (here 2008) fishing mortality rate of species s at age a due to fleet f, and
- the controls $u_f(t)$ are multipliers of the current fishing mortality $F_{s,a,f}$ for fleets f. As we assume that there is no control on the third fleet "others", we fix its fishing mortality by writing

  $$u_3(t) = 1.$$

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

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

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

$$N_{s,1}(t+1) = \varphi_s\Big(SSB_s(N_s(t)), \omega(t)\Big),$$

where

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

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

[5] `http://wwz.ifremer.fr/peche/Le-role-de-l-Ifremer/Observation/`

- $SSB_s(N_s)$ is the spawning stock biomass of species s

$$SSB_s(N_s) = \sum_{a=1}^{A} \gamma_{s,a} \upsilon_{s,a} N_{s,a} \ ,$$

  with $(\gamma_{s,a})_{a=1,\dots,A}$ being the proportions of mature individuals at age a and $(\upsilon_{s,a})_{a=1,\dots,A}$ being the weights of individuals at age a,
- the function φ_s represents the specific stock-recruitment relationship of each species s,
- $\omega(t)$ stands for the uncertainties (environmental or demographic) affecting the stock recruitment relationships through different possible scenarios Ω.

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

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

where $B_1^{\mathsf{lim}} = 54,521$ tons, $\mathcal{N}(\overline{B}_1, \sigma_1)$ stands for a Gaussian distribution[6] and $\overline{B}_1 = 241,776$ for the estimated mean of the hake recruitment (in tons) over the period 1992-2006 while the standard deviation $\sigma_1 = 58,459$ measures the dispersion of the recruitment. Note that the risk of collapse of the stock is captured by the linear declining value for SSB_1 below the critical level $B_1^{\mathsf{lim}} = 54,521$ tons.

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

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

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

[6] We have also tested a uniform distribution but it does not significantly modify the whole results.

2.2 Catches and gross incomes

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

$$C_{s,a,f}(t) = N_{s,a}(t)u_f(t)F_{s,a,f}\frac{1-\exp\left(-M_{s,a}-\sum_{f=1}^{3}u_f(t)F_{s,a,f}\right)}{M_{s,a}+\sum_{f=1}^{3}u_f(t)F_{s,a,f}} \tag{2}$$

The gross income of each fleet's catch is then estimated by incorporating the market prices of the species, recorded for different commercial categories (corresponding to different age groups), along with the estimates of the discard rates (see tables 1 and 2 for details), so that:

$$\mathrm{Inc}_f(t) = \sum_s p_{s,a}\sum_{a=1}^{A+1}\upsilon_{s,a}\,C_{s,a,f}(t)(1-d_{s,a,f}) \tag{3}$$

where

- $p_{s,a}$ is the market price of individuals of species s at age a,
- $\upsilon_{s,a}$ is the mean weight of individuals of species s at age a, and
- $d_{s,a,f}$ is the discard rate of individuals of age a by the fleet f.

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

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

$$\pi_f(t) = \alpha_f \mathrm{Inc}_f(t) - (c_f^{var}.e_f(t_0) + c_f^{fix})k_f(t_0).u_f(t). \tag{4}$$

where

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

The whole set of these economic parameters is displayed in the table 3 in appendix.

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

$$F_{s,a,f} = q_{s,a,f}.e_f(t_0).k_f(t_0), \tag{5}$$

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

3 A viability diagnosis

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

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

$$SSB_s(N_s(t)) \geq B_s^{\mathsf{pa}}, \qquad s = 1, 2 \qquad t = t_0, \ldots, T, \tag{6}$$

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

$$SSB_1(N_1(t)) \geq B_1^{\mathsf{pa}} = 75,784, \qquad t = t_0, \ldots, T.$$

where $75,784$ t corresponds to the (new) precautionary level introduced by the EU in 2006. As there is no ICES precautionary or limit spawning biomass level for nephrops, we set its viability threshold to zero with $B_2^{\mathsf{pa}} = 0$.

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

$$\pi_f(t) > 0, \qquad f = 1, 2, \qquad t = t_0, \ldots, T. \tag{7}$$

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

CVA: The co-viability approach requests that both stock conservation and economic viability of the fleets are guaranteed conjointly. This requires complying both with the SSB constraints (6) and the economic constraints (7).

For all three approaches, we choose to deal with uncertainty in a probabilistic sense. We therefore perform a stochastic viability analysis. For this, we consider a probability $\mathbb{P}$ on scenarios $\omega(.) \in \Omega$, a confidence rate $\beta \in]0, 1]$

together with a time horizon $T > 0$, and we aim at identifying the controls (fishing multipliers u_1 and u_2) that satisfy the following condition [7]:

$$\mathbb{P}_{\omega(.)}\Big(N(t) \text{ satisfies the constraints, } t = t_0, \ldots, T \Big) \geq \beta \tag{8}$$

In the case of the PVA, using the stochastic viability approach means that we consider the constraint (6) and compute the viability probability $\textsc{pva}(u_1, u_2)$ associated to the SSB constraint (6):

$$\textsc{pva}(u_1, u_2) = \mathbb{P}_{\omega(.)}\Big(N(t) \text{ satisfies (6), } t = t_0, \ldots, T \Big)$$

Similarly, in the EVA context, $\textsc{eva}(u_1, u_2)$ denotes the economic viability probability associated to the economic constraints (7), namely:

$$\textsc{eva}(u_1, u_2) = \mathbb{P}_{\omega(.)}\Big(N(t) \text{ satisfies (7), } t = t_0, \ldots, T \Big)$$

Finally for the CVA perspective, $\textsc{cva}(u_1, u_2)$ denotes the co-viability probability associated to both ecological requirements (6) and economic constraints (7):

$$\textsc{cva}(u_1, u_2) = \mathbb{P}_{\omega(.)}\Big(N(t) \text{ satisfies (6) and (7), } t = t_0, \ldots, T \Big)$$

In terms of decision, given a level of risk $1 - \beta$, we aim at identifying viable fishing intensity vectors, namely $u(t) = (u_1, u_2)$ expressed as multipliers of the baseline $u = (1, 1)$, that satisfy viability condition (8). In this context, of particular interest are the controls that maximize the viability probabilities, that is, $\max_u \textsc{pva}(u)$, $\max_u \textsc{eva}(u)$, and $\max_u \textsc{cva}(u)$.

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

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

4 Results

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

4.1 Population Viability Analysis

Figure 1(a) shows the numerical approximation of the viability probabilities for the PVA case. A viability 'frontier' appears in dark blue, delimiting a viability control space within which combinations of fishing intensity (u_1, u_2) are such that the viability probability to satisfy (6) is close to one, namely $\mathrm{PVA}(u_1, u_2) \approx 1$. Outside this viability space, i.e. above the frontier, the viability probability declines progressively toward zero in the red zone where fishing intensity is too high and condition (6) is not satisfied any longer namely $\mathrm{PVA}(u_1, u_2) \approx 0$. This corresponds to combinations of high catch mortality that drive the stock of hake below the precautionary level $B_1^{\mathsf{pa}} = 75,784$ tons. The 'edge' of the viability frontier indicates intermediate situations where $\mathrm{PVA}(u_1, u_2)$ lies between 1 and 0. This corresponds to 'risky' fishing strate-

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

[9] By 'viable trajectories', we mean trajectories satisfying the constraints (6) and/or (7), depending on PVA, EVA or CVA contexts.

gies for which the sustainability of the hake stock is at stake. In particular, the status quo strategy $u_{\text{BLA}} = (1, 1)$ which is slightly above the frontier has not a satisfying PVA (in fact $\text{PVA}(u_{\text{BLA}}) \approx 75\%$) informing on its underlying ecological risk.

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

4.2 Economic Viability Analysis

The results of the economic viability probability EVA are shown in figure 1(b). A viability domain exists (in blue), which contains control states (u_1, u_2) for which the viability probability to guarantee profit condition (7) is close to 1, i.e. $\text{EVA}(u_1, u_2) \approx 1$. Outside the boundary of this viability space, the probability to maintain the economic viability of the system decreases. Intermediate situations appear around of the viability space, where the probability declines rapidly from 1 to 0. Beyond this, in color red, the probability to maintain the two fisheries' profitability is nil. The right part of the red area corresponds to unviable economic situations due to *long-term* (indirect) effects of fishing on the stocks: excessive fishing mortality drives the stocks down to levels which are too low to ensure profitabilities. This especially occurs for large fishing mortality of hake fleet. The dynamics at work on the two other sides of the viability space are of a different nature. The two sharp-edged boundaries running parallel to the axes reflect the *short-term* (direct) effects of the economic constraints (7). They illustrate in particular the fact that minimum fishing mortality are necessary to ensure that constraint (7) is satisfied. Note that the status quo strategy $u_{\text{BLA}} = (1, 1)$ has an EVA close to 0%.

4.3 Co-Viability Analysis

Finally, combining the ecological and economic constraints, figure 1(c) illustrates the results of the computation of the co-viability analysis CVA. In that case, the viability space is reduced to an area close (but not exactly) to the intersection of the PVA and EVA viability spaces. The short-run and long-run effects of the constraints described for the PVA and EVA cases still hold. In particular we recognize the direct short-term economic effect of constraint (7) on the lower and left side of the viability space and the effect of the ecological constraint (6) on the upper side. The long-term effect of (7) is not visible as the constraint (6) affects the system before it can come into play. Note however that situations may occur where the long-term effect of (7) may appear before constraint (6). The fact that the co-viability control space does not exactly coincide with the intersection of the PVA and EVA viability spaces proves that complex and non linear mechanisms occurs through the dynamics, interactions and uncertainties at play.

4.4 Status quo strategy: neither viable nor sustainable

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

$$u_{\mathrm{BLA}} = (1, 1).$$

Figure 2 displays ten trajectories randomly generated under this scenario reflecting recruitment uncertainties and stochasticity. The figure first shows that, this scenario is not ecologically viable since the hake biomass trajectories frequently passes under the ICES precautionary threshold $B_1^{\mathsf{pa}} = 75,784$ tons. Conjointly, the figure also reveals that this scenario is not viable from the economic viewpoint especially for the nephrops fleet: over the 20-year simulations, numerous profit trajectories become negative. The profitability of hake fleet is also at stake although the risk is very low compared to the nephrops trawlers. Overall, this indicates that 2008 had been a favorable year for catches and profits, but that this was not sustainable. Catch intensity could not be maintained at this level if ecological and economic viability of the system are to be ensured.

4.5 Optimizing scenario: a high bio-economic risk

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

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

where the discount rate is set to $r = 10\%$. The associated optimal effort multipliers take the values $u_{\mathrm{PV}} \approx (2.5, 0.25)$. Figure 3 displays ten trajectories randomly generated for this scenario. The figure shows that the paths are not very safe from the ecological viewpoint, as the spawning biomass trajectories fall below the hake limit reference point at several occasions over the 20 year horizon. This is due do the significant increase in the hake fishing effort induced by the adoption of the maximization strategy. This large increase in catch effort does not, however, lead to a sustainable profitability of the hake fleet. In other words, the strategy which consists in maximizing the present value of the total income is not simply ecologically risky. It also implies that the economic viability of the hake feet is not warranted. Note also the severe reduction in the nephrops fleet's effort imposed by the strategy with $u_2 \approx 25\%$.

4.6 Ecological Conservation scenario: viable but not sustainable

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

$$\mathrm{PVA}(u_{\mathrm{PVA}}) = \max_u \mathrm{PVA}(u)$$

This strategy is equivalent to a no-take strategy $u_{\mathrm{PVA}} = (0, 0)$, where no catch is extracted for both fisheries. This strategy obviously satisfies the ecological conditions as illustrated by the very high SSB trajectories in figure 4 for both hake and nephrops stocks. This, however, is clearly not a satisfying solution in economic terms as both fleets incomes are nil.

4.7 Economic scenario: sustainable and almost viable

Symmetrically to the previous ecological scenario, we now consider an 'extreme' viable strategy u_{EVA} characterized by minimal economic risk as follows:

$$\mathrm{EVA}(u_{\mathrm{EVA}}) = \max_u \mathrm{EVA}(u)$$

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

$$|u_{\text{EVA}} - u_{\text{BLA}}| = \min\Big(|u^* - u_{\text{BLA}}|,\ \text{EVA}(u^*) = \max_u \text{EVA}(u)\Big)$$

This EVA scenario corresponds to a strategy where we aim at identifying the controls (fishing multiplyers) u that maximize the probability of economic viability $\text{EVA}(u)$, and then, amongst those solutions, to retain these with the values closest to current level namely $u_{\text{BLA}} = (1, 1)$. The rationale underlying such an option is to minimize the "costs" of changes and thus account for inertia or rigidity in behaviours. Computation indicates that one solution to achieve this maximum EVA strategy is $u_{\text{EVA}} \approx (0.92, 0.54)$. Figure 5 shows one series of trajectories obtained under this strategy. As expected, the levels of profit generated for both fleets remain positive throughout time. An interesting result is that, despite the potentially relatively detrimental nature of this strategy from an ecological conservation point of view, both hake and nephrops average SSB levels appear to be higher under this particular EVA strategy than they were under the initial baseline scenario BLA. This outcomes emerge because maximizing $\text{EVA}(u)$ necessitates fishing mortality levels which do not impact too severely the stocks, so as to maintain long-term catch rates. The fact that economic viability constraints may imply biological viability has also been stressed by Béné *et al.* (2001) and Martinet *et al.* (2007).

4.8 Co-viability scenario: A win-win situation

Finally, we consider the co-viability strategy u_{CVA}, that is, a strategy that maximizes CVA probability mixing ecological and economic constraints:

$$\text{CVA}(u_{\text{CVA}}) = \max_u \text{CVA}(u)$$

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

$$|u_{\text{CVA}} - u_{\text{BLA}}| = \min\Big(|u^* - u_{\text{BLA}}|,\ \text{CVA}(u^*) = \max_u \text{CVA}(u)\Big).$$

Figure 6 displays a series of trajectories obtained under this u_{CVA} strategy. In the particular case studied here, the combination of fishing intensity used for the simulations is $u_{\text{CVA}} \approx (0.9; 0.2)$. Under this strategy, viability probabilities are maximum, that is, $\text{CVA}(u_{\text{CVA}}) = \text{EVA}(u_{\text{CVA}}) = \text{PVA}(u_{\text{CVA}}) = 1$. When compared to the u_{EVA} scenario above, it is worth emphasizing the very important reduction of fishing intensity requested under u_{CVA} for nephrops fleet.

The intuition here is that the CVA strategy accounts for the fact that catches made by the nephrops fleet weaken the viability probability of the hake stock through the impact of the by-catch. In this sense, CVA permits to better balance the stock and fleet requirements, and more generally ecological and economic goals, than any other strategy. However the severe reduction in fishing for the nephrops fleet imposed by this co-viability strategy may question its acceptability.

Finally note that although the probabilities $\textsc{eva}(u_{\mathrm{CVA}}) = \textsc{pva}(u_{\mathrm{CVA}}) = 1$ resulting from the co-viability strategy are unique, co-viability itself can be obtained through different combinations of fishing controls, as illustrated in figure 1(c). This characteristic constitutes a notable difference with optimal control theory where single control solutions are usually identified. In the case of co-viability, a space of solution is identified.

5 Discussion and perspectives

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

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

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

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

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6 Appendix

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

Table 1

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

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

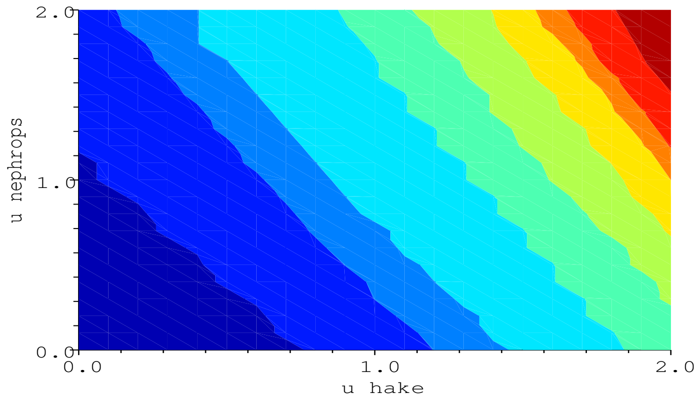
Table 2

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

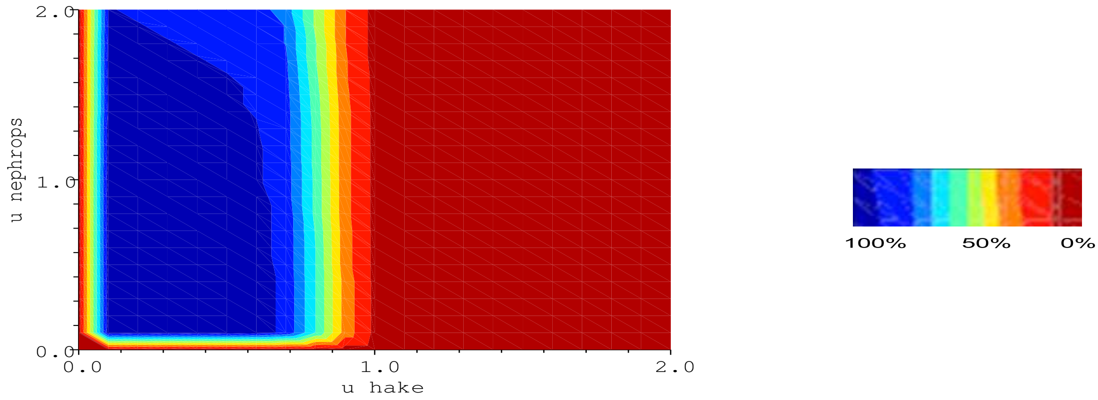
Fleets	$k_f(t_0)$	$e_f(t_0)$	c_f^{var}	c_f^{fix}	α_f
Nephrops trawlers $f = 1$	87	180	379	257604	1.48
Gill netters $f = 2$	116	180	481	211432	1.52

Table 3

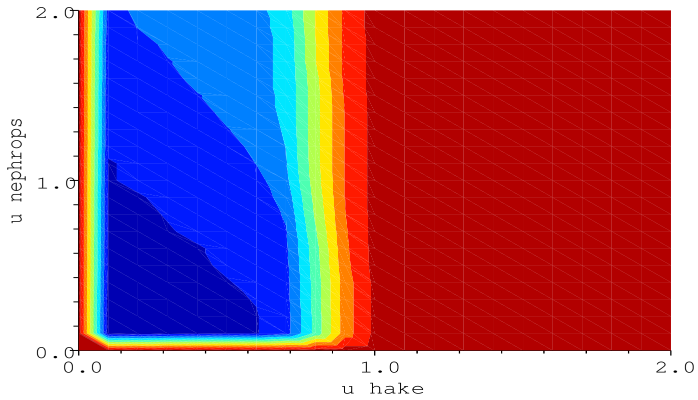
Economic parameters for fleets $f = 1, 2$: Initial number of vessels $k_f(t_0)$, effort by vessel $e_f(t_0)$ (day at sea), variable c_f^{var} (€ by vessel by day), fixed costs c_f^{fix} (€ by vessel) and multiplyer of extra fishing income α_f . *source : Ifremer, SIH, DPMA, 2008*



(a) Population viability probability $\textsc{pva}(u_1, u_2)$



(b) Economic viability probability $\textsc{eva}(u_1, u_2)$



(c) Co-viability probability $\textsc{cva}(u_1, u_2)$

Fig. 1. Viability probabilities $\textsc{pva}(u_1, u_2)$, $\textsc{eva}(u_1, u_2)$ and $\textsc{cva}(u_1, u_2)$ as a function of effort multipliers u_1 and u_2. The viability control space (proba$\approx$ 100%) is in blue. The non viability space (proba$\approx$ 0%) is in red.

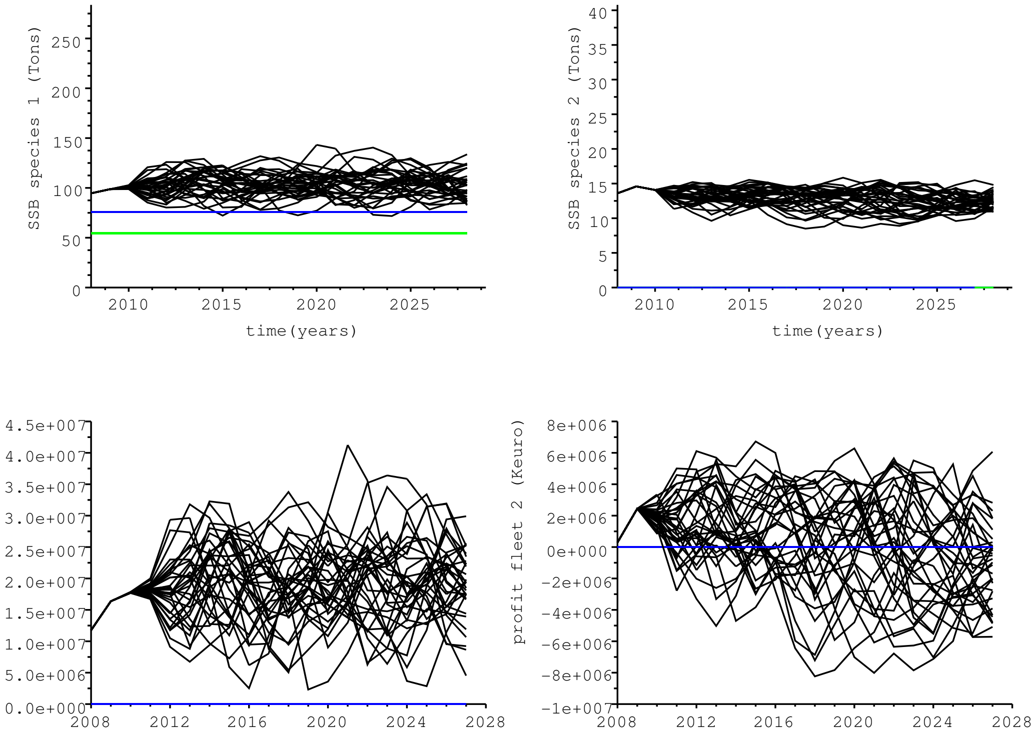


Fig. 2. Trajectories (in black) under the 2008 baseline scenario $u_{\text{BLA}} = (1, 1)$ and viability thresholds (in blue). Top diagrams: spawning biomass $SSB(t)$ for hake (left) with its precautionary biomass level B^{pa} (in blue) and its limit level B^{lim} (in green), and nephrops (right). Bottom: profits $\pi(t)$ for the two fleets with zero viability threshold in blue. See text for comments.

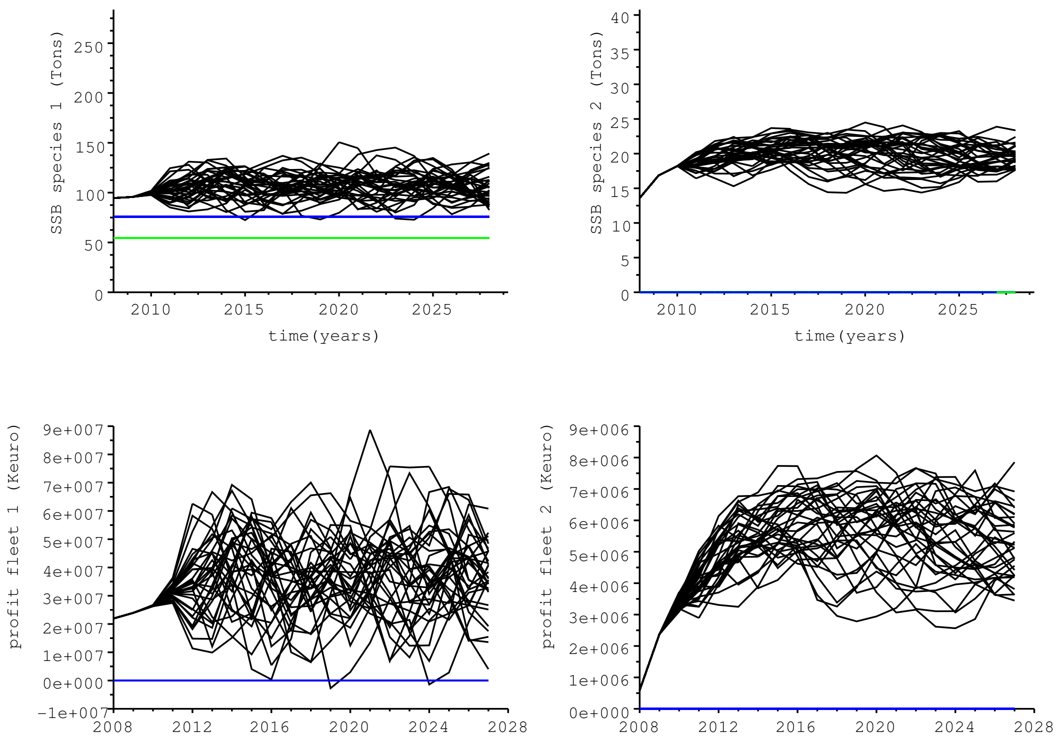


Fig. 3. Trajectories under the optimizing present value scenario PV with effort multiplier $u_{\mathrm{PV}} = (2.5, 0.25)$. Top diagrams: spawning biomass $SSB(t)$ for hake (left) and nephrops (right). Bottom: profits $\pi(t)$ for the two fleets. See text for details.

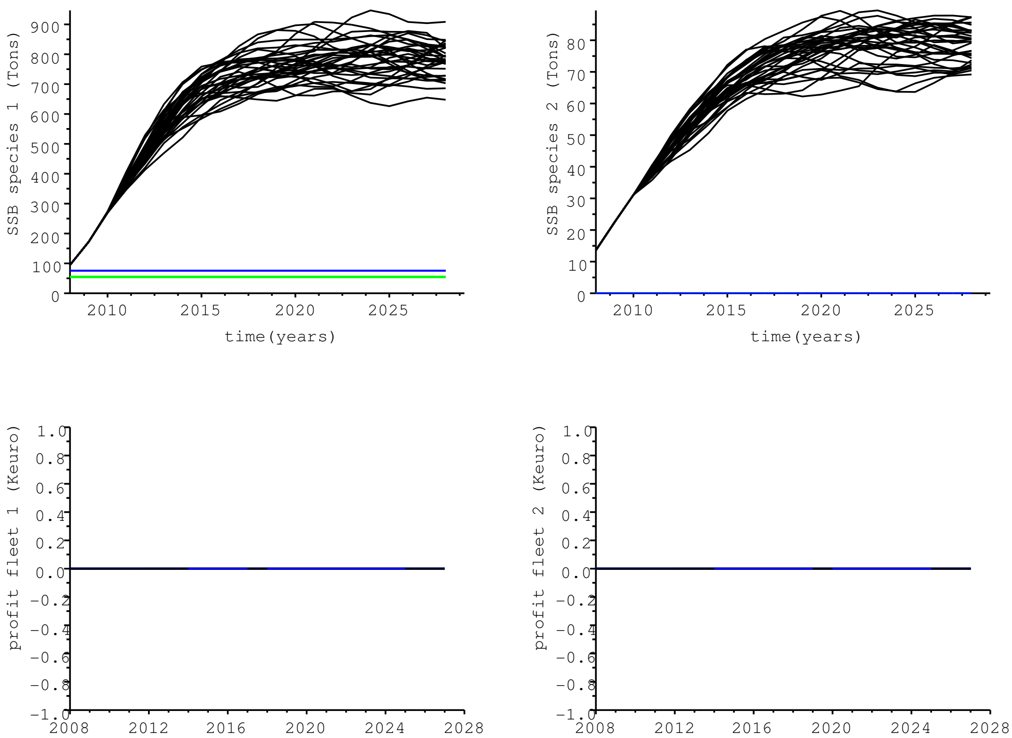


Fig. 4. Trajectories under the ecological (no-take) strategy, i.e. $u_{\mathrm{PVA}} = (0, 0)$. Top diagrams: spawning biomass $SSB(t)$ for hake (left) and nephrops (right). Bottom: no profit is generated.

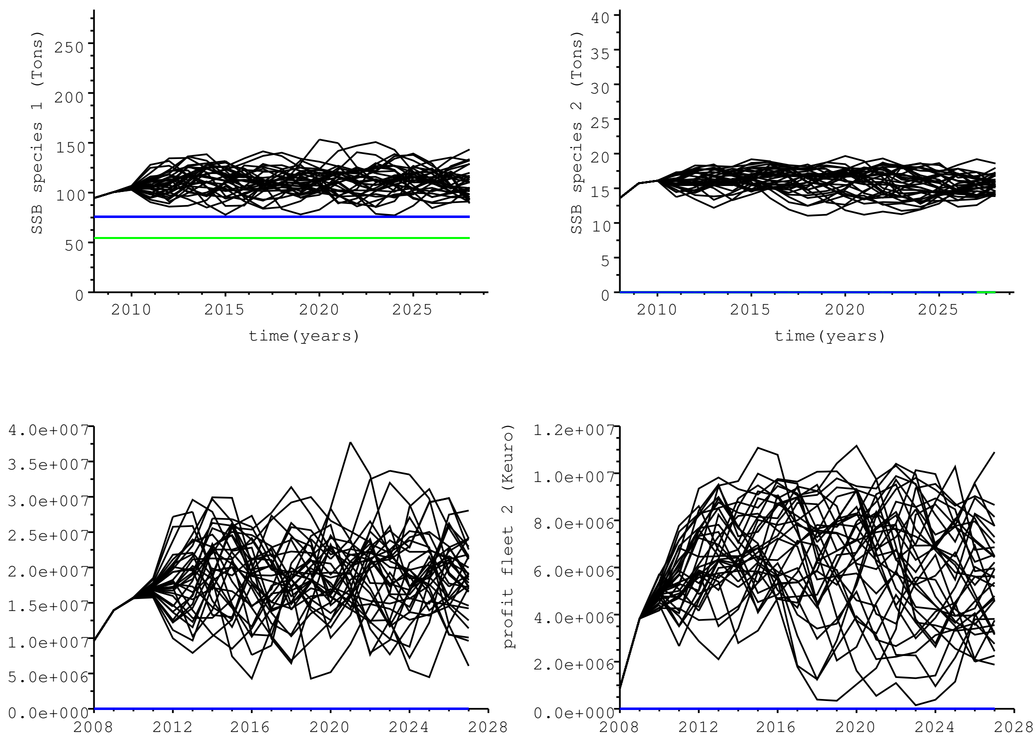


Fig. 5. Trajectories under the economic scenario EVA with effort multiplier $u_{\text{EVA}} = (0.92, 0.54)$. Top diagrams: spawning biomass $SSB(t)$ for hake (left) and nephrops (right). Bottom: profits $\pi(t)$ for the two fleets. See text for details.

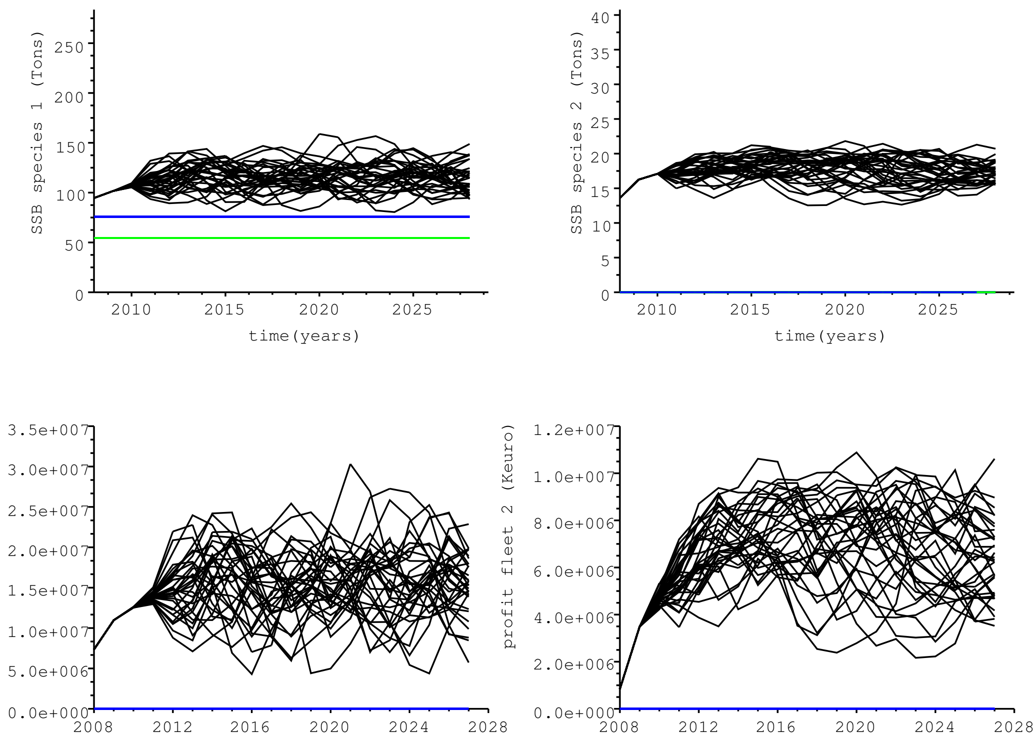


Fig. 6. Trajectories under the co-viability scenario CVA with effort multiplier $u_{\mathrm{CVA}} = (0.9; 0.2)$. Top diagrams: spawning biomass $SSB(t)$ for hake (left) and nephrops (right). Bottom: profits $\pi(t)$ for the two fleets. See text for details.