Incorporating economics into fishery policies: Developing integrated ecological-economics harvest control rules

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ABSTRACT

Despite the efforts of management authorities, several fish stocks continue to be harvested at unsustainable levels. While integrating economics into catch advice has been identified as key in developing more effective management, few studies investigate how bioeconomic modelling can inform decision-making. At the same time, increasing evidence has been gathered on the occurrence of regime shifts in marine ecosystems, raising greater concerns regarding fish stocks' recovery ability. Policy evaluation methods, however, still resort to compensation population models. As a result, these may overestimate reproductive success, thus jeopardizing the design and evaluation of stock rebuilding policies. This paper addresses those challenges proposing a methodology that approximates a harvest control rule using an age-structured bioeconomic model. As a result, we deliver a policy rule that complies with current advice procedures and endogenously captures economics. Also, we consider concerns on the existence of regime shifts in marine populations and estimate a bioeconomic model with critical depensation. Results from our case study suggest that policies neglecting the existence of critical depensation may compromise stock rebuilding objectives and might even result in fishery collapses. Using bioeconomic modelling to define harvest control rules may enhance policy design aimed at reconciling fisheries' economic returns and stock recovery.

1. Introduction

The existence of regime shifts in the natural environment has been the subject of a long unsettled academic debate. Recently, though, an increasing number of studies has documented their occurrence in marine ecosystems, also suggesting that regime shifts are likely to increase in both frequency and magnitude as a result of climate change (see, e.g., Heinze et al., 2021; Wernberg et al., 2016; Litzow et al., 2014; Möllmann et al., 2015; Conversi et al., 2015). Ecological systems characterized by regime shifts are constituted by multiple equilibria, implying that significant losses in the provision of ecosystem services may accrue if convergence occurs at lower state levels. Adding to that, in some cases recovery may be irreversible or entail extremely long recovery periods (Heinze et al., 2021). Avoiding such low equilibria is thus particularly important when considering commercially relevant fish stocks.

A potential mechanism behind regime shifts in marine populations is the Allee effect, i.e. the reduction in per capita growth as the population abundance decreases (Winter et al., 2020). Alle effects have been historically dismissed as rare or non-existent in marine fish populations. That view, however, was recently challenged by several publications that have identified evidence of its existence across different fish populations (e.g., Perälä and Kuparinen, 2017; Hutchings, 2015; Kuparinen et al., 2014). In addition, and perhaps more important is the finding that while Allee effects may not be revealed by data, they might still be relevant in driving population dynamics (Perälä and Kuparinen, 2017; Hutchings, 2015; Frank and Brickman, 2000). In light of that, several authors have suggested that the design of management plans should account for Allee effects on populations experiencing long recovery periods (Siple et al., 2021; Conversi et al., 2015; King et al., 2015; Gascoigne and Lipcius, 2004). In fact, detecting the presence of Allee effects usually requires obtaining data for low abundance levels, when tipping points may have already been realized, and recovery is either impaired or extremely difficult (Winter et al., 2020). Policy evaluation methods, however, still typically assume compensation population

models. As a result, these may overestimate species' reproductive success, thereby jeopardizing both the design and evaluation of stock rebuilding policies.

Accounting for regime shifts is thus aligned with recent efforts toward implementing an Ecosystem-Based Fisheries Management (EBFM) approach, which has historically focused on ecological and biological concerns. Interestingly, however, a renewed interest has emerged regarding the need for further introducing human dimensions into EBFM. In fact, the discussion on integrating social sciences into fishery management is no longer limited to academic debate but is already taking place at institutional policymaking levels. For instance, the International Council for the Exploration of the Sea (ICES) has taken consistent steps toward integrating socio-economics analysis with policy and management in its recently adopted Strategic Plan (ICES, 2021). In this context, ICES has also started a set of initiatives aiming to facilitate cooperation between marine and social scientists. In particular, ICES launched the Strategic Initiative on Human Dimensions, the Working group on Social Indicators, and the Working Group on Economics (ICES, 2021). The latter with the specific goal of integrating economics into ICES science and policy advice. This is nothing less than a paradigm shift in fishery policies, which will require developing an integrated multidisciplinary approach combining fish ecology and socio-economics knowledge.

This paper addresses the above challenges by proposing a methodology that approximates a Harvest Control Rule (HCR) using an age-structured bioeconomic model. Our contribution is three-fold. First, we develop a method delivering a policy rule that endogenously accounts for economics using a fish population model that is at the basis of modern stock assessment methods and management plans. Second, our policy rule matches a conventional fishery policy tool, facilitating the integration of the proposed method into current legal advice frameworks. Finally, we extend our analysis to consider the existence of Allee effects in the managed population.

Reference-point-based HCRs are a key fishery policy instrument, playing an important role in managing fish stocks in several regions of the globe. Interestingly, HCRs' conceptual foundations are coherent with findings in the economics literature, in which optimal harvest was found to be a function of the biomass stock level (Kvamsdal et al., 2016). This earlier literature, however, was mainly focused on theoretical/ normative analysis resorting to oversimplified growth population models. As a result, in real-world practice economics is typically not embedded in HCRs, but is only an ad-hoc output of management performance evaluations (Benson and Stephenson, 2018). In this context, a new fishery economics literature using more complex population dynamics, namely age-structured bioeconomic models, has contributed to bridging the gap between academic research and policymaking (see Nielsen et al., 2018). Indeed, integrating socio-economics into catch advice using bioeconomic modelling is increasingly advocated as key in developing more effective fishery management (Voss et al., 2017; Dichmont et al., 2010; Beddington et al., 2007). Increasing the complexity of bioeconomic models, though, may come at a cost. One particular difficulty is that when population dynamics are modelled assuming an aged-structured population, optimal harvest is no longer a function of biomass but of the distribution of individuals over age classes (Tahvonen, 2009; Rosa et al., 2018). Accordingly, more complex bioeconomic models cannot be used to derive economically informed HCRs. While that literature has thus contributed to the debate on the introduction of economics into fisheries management by defining annual quotas or studying the impacts of alternative policy scenarios/objectives (see, e.g. Kvamsdal et al., 2016; Emery et al., 2017; Voss et al., 2017, Da Rocha et al., 2010; Da Rocha and Gutiérrez, 2011), it is still open to question how HCRs used in multiannual management plans can be defined through bioeconomic modelling optimization. This paper overcomes that issue by proposing a method that approximates an HCR using an age-structured bioeconomic model. This is done using the Ibero-Atlantic sardine stock as a case study. This stock, supporting the

most important fishery in Portugal, recently faced historically low levels and reduced reproductive capacity. Since 2012 the Ibero-Atlantic sardine fishery has been under strict management and subject to a severe reduction in landings. Population levels, however, have remained surprisingly low, resulting in an unexpectedly long recovery period (ICES, 2018)

Small pelagic fish populations, such as sardines, are paradigmatic examples of species registering stock collapses, and for which several studies identified mechanisms causing a decrease in the reproductive capacity of the species at low abundance levels (Bakun and Cury, 1999; Cury et al., 2000; Liermann and Hilborn, 2001; Zwolinski and Demer, 2012). While no study has explicitly investigated Allee effects in the Ibero-Atlantic sardine stock, several papers have acknowledged the existence of factors that may be negatively affecting the species recruitment potential (Garrido et al., 2017; Garrido et al., 2016; Silva et al., 2013). Considering the potential role of depensation mechanisms in managing overexploited fish populations, we estimate two age-class structured bioeconomic models, assuming both compensatory and depensatory population dynamics. Previous literature has studied optimal resource management in the face of regime shifts but has primarily focused on theoretical analysis making use of simple growth models (see, e.g., Polasky et al., 2011; Ren and Polasky, 2014). We thus also add to the current literature by investigating optimal resource management using applied age-class structured bioeconomic modelling accounting for regime shift mechanisms.

Optimal solutions in bioeconomic models are a function of the underlying population model. Accordingly, optimal harvest and biomass paths obtained using such models explicitly account for the possibility of endogenous collapse if such effects exist in the targeted stock. The methodology proposed in this paper is thus used to obtain two HCRs, one for a population with compensatory dynamics and another for a population with critical depensation on the stock-recruitment relationship, i.e., accounting for Allee effects. We provide a policy evaluation of both HCRs by studying, in a stochastic environment, how they perform in economic terms and in their ability to foster biomass recovery assuming both population models. Results from our case study suggest that policies neglecting the existence of critical depensation may compromise stock rebuilding objectives and might even cause fishery collapses. Using bioeconomic modelling to define harvest control rules may enhance policy design aimed at reconciling fisheries' economic returns and stock recovery.

The rest of the paper is organized as follows. Section 2 briefly describes the bioeconomic models and the methodology used to obtain the HCRs. Section 3 presents results for the HCRs' performance evaluations. Section 4 discusses management implications and provides some concluding remarks on the paper's results. Additional technical information is provided in the supplementary information material.

2. Methods

2.1. Models and data

The Ibero-Atlantic sardine stock population is modelled under an age-class framework. The structure follows usual fisheries stock assessment models, modified to capture specific sardine population dynamics, i.e. the inclusion of intra-seasonal growth, and the specification of a different timing for the harvest season (Rosa et al., 2018).

Considering n age classes, the number of individuals in age class $s = 1, ..., n$ in year $t = 0, 1, ...$ is denoted by $x_{s,t}$. Let γ_s denote the maturity at age in the s-th age class, and ω_s the average weight-in-stock at spawning time. The spawning stock biomass, $x_{0,t}$ is thus given by

$$x_{0,t} = \sum_{s=1}^{n} \gamma_s \omega_s x_{s,t} \tag{1}$$

We assume that spawning occurs in the beginning of each period, and

only a fraction of the eggs survives as recruits. Let R denote the spawner-recruit relationship, so that the number of recruits in the next period is

$$x_{1,t+1} = R(x_{0,t}) \tag{2}$$

Sardines spawn in winter months, which we assume to correspond to the beginning of each period, whereas catches mostly occur in summer months. We thus assume that harvesting occurs after recruitment and at the middle of each period. Let m_s denote the age-specific instantaneous natural mortality rate, so that after half a year, $e^{-m_s/2}$ corresponds to the surviving fraction. Given an effort level, E_t, harvest from age class s is determined by the age-specific catch-effort relationship $h_{s,t} = q_s E_t e^{-m_s/2} x_{s,t}$, where q_s are the age-specific catchabilities. The stock dynamics is therefore given by

$$x_{s+1,t+1} = e^{-m_s/2}\left(e^{-m_s/2}x_{s,t} - h_{s,t}\right), s = 1, \ldots, n-2 \tag{3}$$

$$x_{n,t+1} = e^{-m_{n-1}/2}\left(e^{-m_{n-1}/2}x_{n-1,t} - h_{n-1,t}\right) + e^{-m_n/2}\left(e^{-m_n/2}x_{n,t} - h_{n,t}\right) \tag{4}$$

where Eq. (4) shows that fish accumulate in the older age class according to natural and fishing mortality.

The total annual catch H_t is equal to the sum of harvest over all age classes, evaluated at their average weight-in-catch ω_s^c, so that,

$$H_t = \sum_{s=1}^{n} \omega_s^c h_{s,t} = \sum_{s=1}^{n} \omega_s^c q_s E_t e^{-m_s/2} x_{s,t} \tag{5}$$

To accommodate H_t as the control variable, we rewrite the stock dynamics equations as a function of total catch (Tahvonen et al., 2013). From Eq. (5) we have

$$E_t = \frac{H_t}{\sum_{s=1}^{n} \omega_s^c q_s e^{-m_s/2} x_{s,t}} \tag{6}$$

Combining (6) with the age-specific catch-effort relationship allows us to obtain a function ($G_{s,t}$) transforming total harvest, H_t, into the number of harvested individuals in a given age class s at time t as

$$G_{s,t} = \frac{e^{-m_s} q_s x_{s,t}}{\sum_{s=1}^{n} \omega_s^c q_s e^{-m_s/2} x_{s,t}}, s = 1, \ldots, n \tag{7}$$

Finally, using (7) and $e^{-m_s/2}h_{s,t} = H_t G_{s,t}$, Eqs. (3) and (4) can be written as

$$x_{s+1,t+1} = e^{-m_s} x_{s,t} - H_t G_{s,t} \ \text{s} = 1, \ldots, \text{n} - 2 \tag{8}$$

$$x_{n,t+1} = e^{-m_{n-1}} x_{n-1,t} + e^{-m_n} x_{n,t} - H_t\left(G_{n-1,t} + G_{n,t}\right) \tag{9}$$

Denoting annual net revenues from the fishery as $\pi(H_t)$, letting r denote the annual interest and $b = \frac{1}{1+r}$ the discount factor, the fishery Net Present Value (NPV) can be written as

$$\sum_{t=0}^{\infty} b^t \pi(H_t) \tag{10}$$

The optimization problem thus consists of maximizing (10) subject to the age-structured population dynamics equations (1), (2), (7)–(9) and the usual non-negativity and initial state conditions. The age-specific population parameters are based on the 2014 ICES WGHANSA report (ICES, 2015), economic data on variable fishing cost for the Portuguese purse-seiner fleets were obtained from STECF (2014).[1] Detailed

information on both equations and model parametrization is included in the supplementary information material.

2.1.1. Recruitment functions

We consider two recruitment functions (see Fig. 1). In the first we assume that recruitment follows a standard Ricker formulation (Ricker, 1975), while the second is an extended version that allows for the existence of a spawning stock biomass level associated with zero recruitment, i.e., critical depensation in recruitment.

The first recruitment function is given by:

$$R_t = a x_{0,t} e^{-bx_{0,t}} \tag{11}$$

where R_t denotes the number of recruits (in 10^9 individuals), $x_{0,t}$ is spawning biomass (in 10^6 tons), a is the productivity parameter, and b the (inverse) capacity parameter of the Ricker stock-recruitment relationship. The stock-recruitment function parameters were obtained fitting a log-linearized version of (11) to data collected from ICES (2015) for 1978–2014, namely the number of recruits and biomass of ages 2–7 as an indicator of spawning abundance. We thus estimate the following equation:

$$ln\left(\frac{R_t}{x_{0,t}}\right) = ln(a) - bx_{0,t} + \varepsilon_t \tag{12}$$

where ε_t are independent normally distributed errors with standard deviation σ_ε. The resulting estimates are given by $ln(\widehat{a}) = 3.841 \ (0.276)$, $\widehat{b} = 1.214 \ (0.055)$, with standard error in parentheses.

The second recruitment function follows Chen et al. (2002). This consists of a Ricker type stock-recruitment model that incorporates Allee effects allowing for a non-zero intercept, thereby capturing recruitment failure. The second stock-recruitment model is thus given by:

$$R_t = \left(x_{0,t} - x_C\right)e^{\alpha - \beta\left(x_{0,t} - x_C\right)} \tag{13}$$

where x_c is the Allee effect parameter representing the spawning stock biomass level associated with zero recruitment. Following Chen et al. (2002) we assume that (13) is log-normally distributed and apply maximum likelihood estimation, using a global search algorithm, to the log of expression (13). The estimates obtained are given by $\widehat{x_C} = 78.353 \ (5.56)$, $\widehat{\alpha} = 4.273 \ (0.2296)$, $\widehat{\beta} = 0.001967 \ (0.0006)$, with standard error in parentheses.

2.2. Ecological-economic integrated harvest control rules

Reference-point-based HCRs do not consider economics and as a result do not depend on prices, costs, or other economic factors. The advantage of using bioeconomic models is that optimal harvest and biomass paths explicitly depend on those variables. Also, if the population dynamics used in such models are similar to stock assessment methods, bioeconomic modelling could be directly used to inform
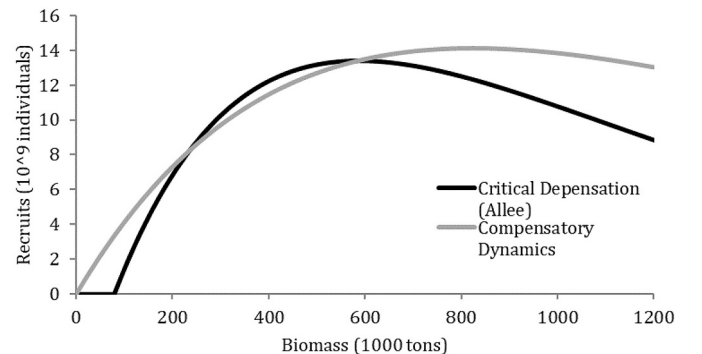
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[1] At the time of development of the model, STECF (2014) and ICES (2015) provided the most recent data for this fishery. While these have been recently updated, using data from older versions of those reports does not affect the analysis offered in this paper. Note that, the applied model serves as a case study illustrating the potential application of the proposed method.

Fig. 1. Recruitment functions.

fishery policy. Until recently, however, fishery economics has been based mainly on the Gordon-Schaefer model, which neglects age/size population structure. Consequently, while the analytical properties of the optimal solution under that framework are well-established, they remain the subject of ongoing research in age-class structured bio-economic modelling literature

Tahvonen, 2009 analyzes that issue using numerical methods. The paper applies a generic age-structured population model and investigates the effects of different properties of the fish population, fishing technology, and other bioeconomic parameters on the model's optimal solution. The author concludes that some general features of the biomass model do not apply when the population is age-structured. Namely, optimal harvest is a function of the distribution of individuals over age classes and is not necessarily an increasing function of biomass. As a result, an optimal saddle path in the Biomass-Harvest state-space does not exist, and the model's optimal solutions cannot be directly translated into an HCR. That being said, though, if the optimal solution of an age class structured bioeconomic model could be reasonably approximated by a policy rule defining annual harvest as a function of biomass, then an HCR could still be defined using that framework. In that case, and in contrast to current policies, the resulting HCR would explicitly account for economics while considering the same population model used in current fishery policy design. In other words, the proposed HCR provides an integrated policy tool combining fish ecology and socio-economic knowledge. Also, note that in that case, applying the resulting HCR generates harvest and biomass paths that approximate the actual bioeconomic optimal solution for any given initial condition of the managed population.

Bearing in mind the above discussion, we solved the problem stated in Section 2.1 for both population models considering several random initial age-class distributions. This resulted in optimal paths rapidly converging to a common age-class structure trajectory, implying that the same convergence occurs in the Harvest-Biomass state-space. Fig. 2 – panels a) and c) – shows the optimal solutions in that state-space. Fig. 2 – panels b) and d) – shows the same information when the first two periods are removed from each optimal path, revealing that converge occurs after approximately two periods only. The resulting common trajectory resembles a saddle path in bioeconomic biomass growth models, defining the optimal harvesting for any given biomass level. This suggests that, while for age-class structured models the optimal harvest depends on age class distribution, for our case study a policy rule may be satisfactorily approximated over a single state variable, i.e., the total stock biomass. Note, however, that for any given initial condition, using a biomass-based policy rule entails initial harvesting levels that differ from the bioeconomic model optimal solution. Appendix 2 in the supporting material discusses this issue in more detail. For the case study considered in this paper we show that solving the model or applying the approximated policy rule results in slight differences in biomass paths and minor NPV losses.

Based on our findings, we thus propose that two ecological-economic integrated HCRs can be defined using the common optimal trajectories in Fig. 2 - panels b) and d). This is done by fitting a polynomial piecewise function to the points in that figure. Given that harvest is approximately zero for very low biomass levels, the first section of that function defines the biomass level for which a moratorium is applied.

Fig. 3 presents the HCRs obtained for the two population models described in Section 2.1. In what follows, *cHCR* denotes the HCR obtained for the bioeconomic model assuming compensation and *cdHCR* for the bioeconomic model assuming critical depensation. Our results highlight that HCRs that explicitly incorporate economics may entail a
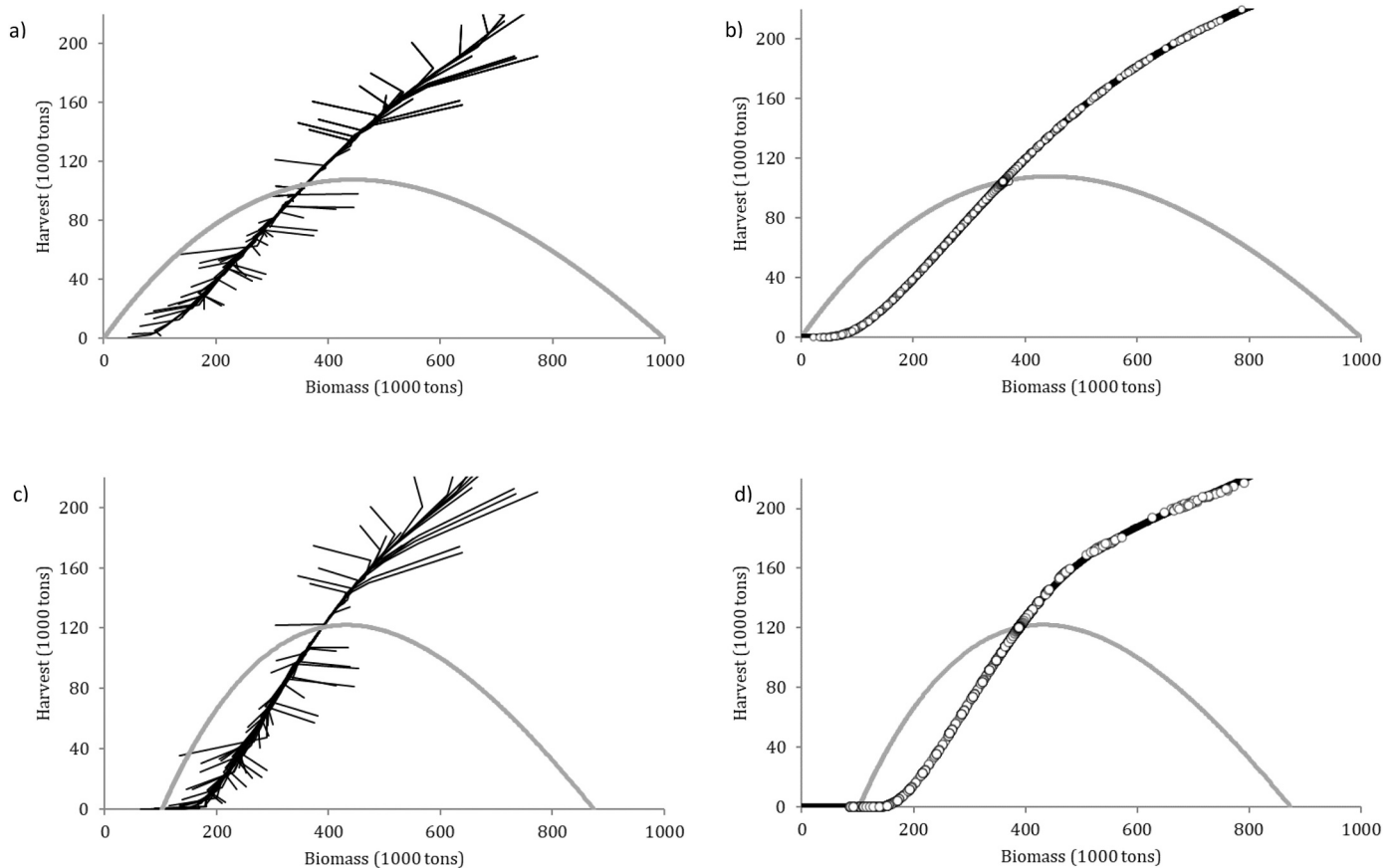

Fig. 2. Panels a) and c): 75 optimal Harvesting-Biomass paths for compensatory and critical depensation population models; Panels b) and d): Optimal Harvesting-Biomass paths after removing the first two periods of each optimal path. Grey lines: Steady-state harvest-biomass curves for both population models.
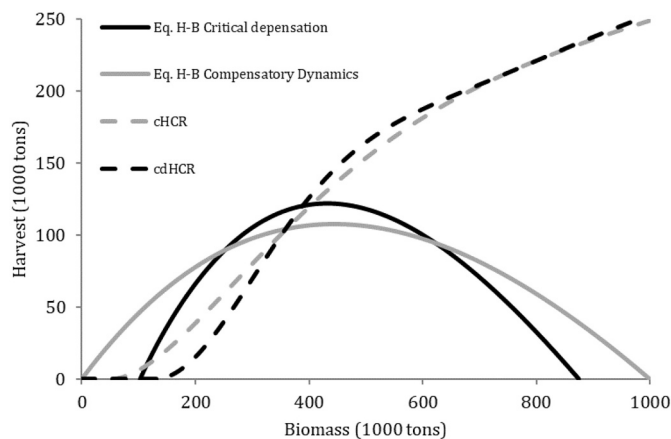
Fig. 3. Harvest control rules with compensation (*cHCR*) and critical depensation (*cdHCR*). Steady-state harvest-biomass curves for both population models.

nonlinear relationship between harvesting and biomass, which contrasts with current biological reference points HCRs (Froese et al., 2011). While fishery closures may still be optimal (the moratoria for *cHCR* and *cdHCR* correspond to 56,000 tons and 153,000 tons respectively), TAC increases at an increasing rate for low biomass levels and increases at a decreasing rate for high enough values of the stock. Also, the HCRs obtained using this method result from a forward-looking optimization problem; optimal paths thus explicitly take into account the reproductive capacity of the fish population and the impact of current harvest in the stream of all future profits. In this case, when the population model exhibits Allee effects in recruitment, optimizing discounted profits implies lower harvesting levels for low stock levels, which allows for a faster recovery of the stock that supports higher harvest levels in the future. As a result, when compared to *cHCR, cdHCR* defines a fishery closure for a higher biomass, and harvesting is lower for low population levels (see Fig. 3).

3. Performance evaluation of harvest control rules

Following usual management evaluation procedures, in this section we provide a simulation performance of the two HCRs obtained in the previous section. Fishing advice processes typically involve estimating a mathematical representation of the managed population (i.e., a simulation population model/operating model) and evaluating the evolution of the managed stock under alternative management strategies, namely candidate HCRs (Punt et al., 2016). While such processes vary in complexity, notably in the treatment of uncertainty of key population model parameters (Butterworth, 2007), these still typically ignore the potential existence of Allee effects in the managed population. Remarkably, though, a large group of experts has recently identified that accounting for density-dependence mechanisms effects is particularly important for precautionary management in the case of small pelagic fishes (Siple et al., 2021).

Adding to current management evaluation processes, our evaluation performance thus applies *cHCR* and *cdHCR* to the two simulation population models described in Section 2.1. The simulation population models are projected considering annual shocks in the recruitment functions. To that end, we use the mean and standard deviation values for the error terms distributions obtained in Section 2.1. Each simulation encompasses 50 years. The evaluation performance of the HCRs is constituted by 2000 simulations. Finally, all simulations assume as initial condition the 2014 age distribution provided in ICES (2015). Biomass after five and ten years were considered indicators of stock recovery while annual harvest and NPV over the simulation horizon (50 years) are used as indicators of economic performance.

Section 3.1 (below) investigates the performance of both HCRs when

the simulation model assumes compensation. In Section 3.2 the same discussion is performed for a simulation model assuming critical depensation in recruitment. In both cases the discussion focuses on the corresponding transition paths. In fact, note that for a given population model both HCRs converge to nearly the same long-run harvest and biomass levels (see Table 1). This is not surprising as the two policy rules cross the equilibrium Biomass-Harvest functions (for both population models) at approximately the same point (see Fig. 3).

3.1. Simulation population model – compensatory dynamics

Small pelagic fishes are characterized by significant recruitment variation, resulting in high fluctuations in population abundance (Siple et al., 2021). In our case, this translates into large shocks in the recruitment functions estimated in Section 2, causing high variability in simulated harvest and biomass paths. Notwithstanding the substantial observed variation, both HCRs result in a relatively continuous and fast recovery of population stock levels (see Fig. 4- upper panel). Under compensatory dynamics, the population carrying capacity is circa 880,000 tons. After five years, the average biomass value is already above 25% of that value. Note, however, that biomass recovery is faster when *cdHCR* is applied. In this case, approximately 80% of all observed biomass simulations at period 5 are already above 25% of the population carrying capacity. After ten years, this is the case for circa 98% of all biomass trajectories.

When compared with *cHCR*, implementing *cdHCR* results in faster recovery due to (i) the greater moratorium and (ii) the lower harvesting determined by this rule for smaller population levels (see Fig. 3). In fact, in year 3, simulated harvests resulting from *cdHCR* are significantly below the levels determined by *cHCR* (see Fig. 4 – middle panel). Note, however, that already in year 5, despite the low catch levels registered for various simulations, the harvest distribution is already relatively the same under both management strategies. In addition, the faster biomass recovery induced by *cdHCR* allows for higher harvest under this rule already in year 10. The counteracting effects of this harvest trajectoies on the fishery profits results in NPV distributions that are approximately equal for both rules (see Fig. 4 – lower panel). Since large variations in recruitment imply large variations in TAC, this translates into large fluctuations in NPV. In this case, both rules generate an average NPV of approximately 289 Million euros and a standard deviation of 33 Million euros.

3.2. Simulation population model – critical depensatory dynamics (Allee effects)

When the stock-recruit relationship in the simulation population model exhibits Allee effects, applying *cdHCR* still guarantees a continuous and fast population recovery (see Fig. 5 – upper panel). In this case, the average biomass after five years equals approximately 245,000 tons (well above 200,000 tons – 25% of the population carrying capacity); after ten years, it amounts to circa 400,000 tons. In contrast, while *cHCR* seems to result in a lower yet continuous biomass recovery immediately after policy implementation, biomass fluctuation thereafter increases significantly (see Fig. 5 – upper panel). In fact, approximately 20% of all biomass trajectories at year 10 are still below the observed 5-year average, indicating the existence of long recovery periods. Also, and

Table 1
Simulation results. Long-run averages for harvest and biomass, values in 1000 tons.

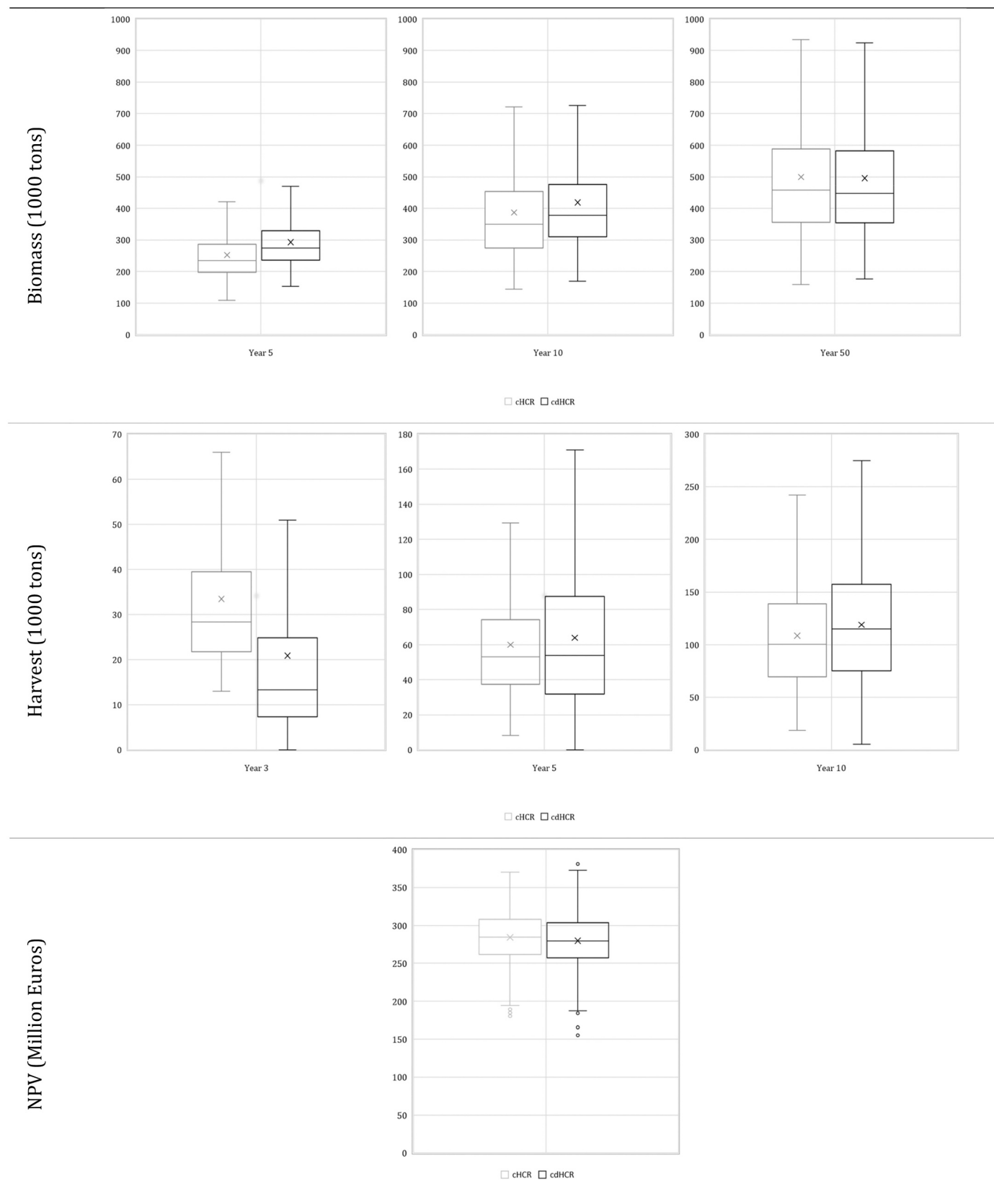|		Compensation	Critical depensation
cHCR	Harvest	143	145
	Biomass	500	500
cdHCR	Harvest	145	150
	Biomass	495	500

Fig. 4. Simulation results – compensatory dynamics. Upper panel: Biomass recovery after five, ten, and 50 years; Middle panel: Harvesting after three, five, and ten years; Lower panel: Net Present Value.
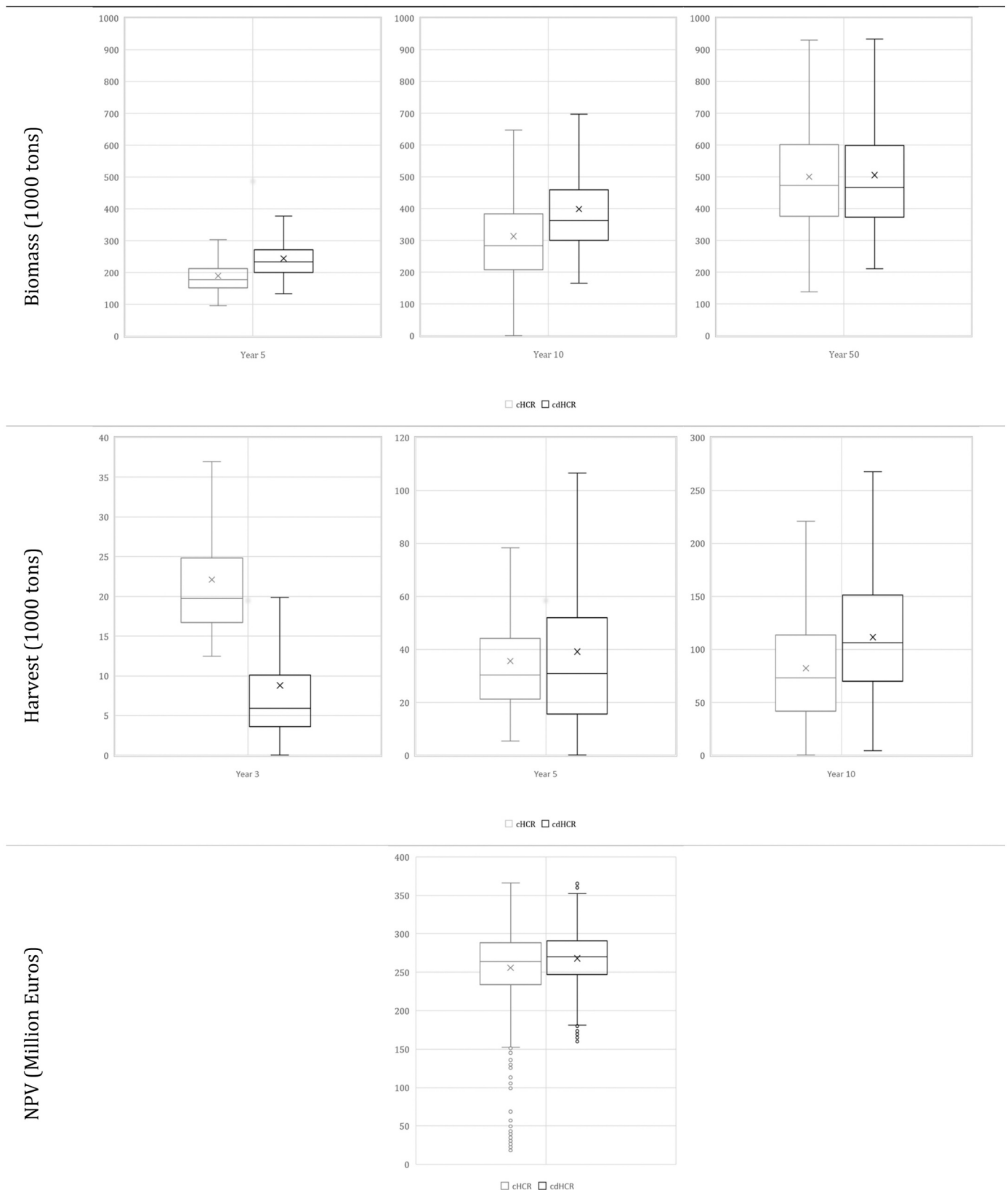
Fig. 5. Simulation results – critical depensatory dynamics. Upper panel: Biomass recovery after five, ten, and 50 years; Middle panel: Harvesting after three, five, and ten years; Lower panel: Net Present Value.

more importantly, some simulations correspond to fishery collapses – the probability of collapse resulting from *cHCR* (measured as the number of simulations in which collapse occurs divided by the total number of simulations) equals 0.02.

The greater variability in biomass resulting from *cHCR* is thus explained by two reasons, (i) the existence of simulations leading to population extinction and (ii) the occurrence of long recovery periods. This is best illustrated by looking at two extreme cases (Fig. 6). In the first, large positive recruitment shocks do not occur in initial periods (Fig. 6 – left-hand side panels); in the second, those are registered at a very early stage (Fig. 6 – right-hand side panels). Note that *cHCR* was obtained assuming a compensation model, but the population simulation model used in the performance evaluation now assumes the existence of Allee effects. For small recruitment shocks, this implied that applying *cHCR* results in initial harvesting levels close to the regenerative capacity of the population (see Fig. 6 – left panels).[2] As a result, stock recovery is hindered, creating a trap in which fishery profits and biomass remain at low levels for extended periods. Fast stock recovery is thus highly dependent on the occurrence of large initial positive recruitment classes.

Extended recovery periods and fishery collapse resulting from *cHCR* severely impact NPV distribution (see Fig. 5 – lower panel): not only is average NPV lower, but variations in economic returns are also greater under this management strategy. Note that the outliers in the first boxplot in Fig. 5 (lower panel) can be roughly divided into two groups. The group located at the bottom of that figure corresponds to the simulations in which collapses occurred. In this case, the lowest registered NPV is approximately 18 Million euros only, while the highest value amounts to circa 69 Million euros. The second group corresponds to simulations in which profits (and harvests) remained low for extremely long periods before recovery took place.

3.3. Discussion and concluding remarks

This paper is the first to propose a method delivering a Harvest Control Rule obtained through age-structured bioeconomic modelling optimization. One of the main advantages of the method is that it integrates economics and ecology into a single tool used in current fishery management policy. In particular, the proposed HCR can be directly included in Management Strategy Evaluation (MSE), acknowledged as the current most sophisticated approach for evaluating the effectiveness of fishery management alternatives (Punt et al., 2016). By developing a simulation-based evaluation focusing on recruitment uncertainty, the previous section illustrates how the method developed in this paper could be potentially used in such a framework.[3]

Our analysis adds to standard MSE in two fundamental aspects. First, MSE management candidate strategies are defined as a set of alternative reference-point based HCRs. Accordingly, while these rules are evaluated for their ability to meet a set of management priorities, identifying an HCR that may best reconcile conflicting objectives resorts essentially

to a trial-and-error simulation process. In contrast, by using a bioeconomic model, the method proposed in this study may deliver HCRs optimizing multiple objectives and/or management concerns. When considering species registering low or very low population levels, a common trade-off involves weighting faster biomass recovery against the negative economic impacts of reducing fishing mortality. The HCRs obtained in this paper endogenously account for such trade-offs: being the result of a forward-looking optimization process, the effects of higher fishing pressure on population recovery and NPV are simultaneously taken into account. Second, our performance evaluation considers two simulation/operating models, explicitly accounting for the existence of depensation mechanisms – which has been recently identified as especially important in improving MSE for small pelagic fishes (Siple et al., 2021).

The analysis offered in Section 3 reveals that when the stock-recruit relationship exhibits Allee effects, applying *cdHCR* results in faster biomass recovery and higher NPV distribution. More significantly, though, is the finding that using a rule derived ignoring the existence of depensation may not only result in unexpectedly long recovery periods but may also have catastrophic consequences by causing fishery collapse. In addition, note that in this paper collapse was defined as a biomass trajectory converging to zero. Collapse, however, is more often defined as biomass being below a given threshold for a considerable period of time (see e.g., Winter et al., 2020). If such a definition was adopted here, the negative impacts resulting from ignoring depensation in the managed population would increase significantly. Our results thus reinforce previous findings on the role of depensation in managing marine fish populations, namely, (i) accounting for Allee effects increased uncertainty regarding marine population recovery time and (ii) faster recovery essentially depended on chance – i.e., the existence of initial large positive shocks in recruitment (Kuparinen et al., 2014). In the presence of depensatory dynamics, *cdHCR* thus outperforms the alternative candidate strategy. On the other hand, however, when the simulation population model is compensatory, a trade-off exists between the two HCRs: applying *cdHCR* results in faster population recovery, but *cHCR* performs better in NPV terms. In addition, note that the simulations' initial biomass is approximately 127,000 tons, and *cdHCR* defines a moratorium for biomass levels below 153,000 tons. As a consequence of that, all simulations for that rule have at least one period for which the fishery is closed. However, that is enough to guarantee that in most cases harvest is not only positive thereafter but is also higher under *cdHCR*. As a result, the fishery closure imposed by *cdHCR* does not have a high impact on NPV. In fact, while *cHCR* performs slightly better when considering fishery NPV, the distribution for this variable is remarkably similar under both rules.

The results from our simulations thus seem to support the choice of *cdHCR* over *cHCR*. If Allee effects exist in the managed population, *cdHCR* performs better in all considered criteria; if depensation is not relevant, *cdHCR* results in faster biomass recovery and entails minor NPV losses compared with *cHCR*. While a final decision regarding HCR choice should consider assessing a broader range of biological and economic uncertainty, this result seems to be aligned with biologists' regular support of precautionary management policies. Note, however, that precautionary management is typically associated with large trade-offs between biomass recovery and economic losses. That, though, is not observed in the analysis offered in this paper. This finding thus highlights one of the merits of the proposed method. Namely, the benefit of using bioeconomic modelling optimization over trial-and-error evaluation of alternative reference point-based HCRs: when the bioeconomic model was modified to explicitly account for depensation, the ecological-economic integrated HCR that was obtained resulted in biomass and harvest paths that endogenously balanced biomass recovery and fishery NPV.

Current fishery advice is the result of a long research effort undertaken primarily by natural scientists. In this context, integrating information from applied bioeconomic modelling into fishery policy is

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[2] Note that growth rate is not a function of biomass alone in age-class structured models. Therefore, in contrast with biomass growth models, it is not possible to plot the relationship between growth and stock level in a Biomass-Harvest/Growth-Rate state space. In this case, biomass growth depends on the distribution of individuals by age-class; population growth in Fig. 6 should thus always be read as an approximation of its actual value.

[3] Designing and implementing an MSE is a complex process that requires developing a series of steps, namely (i) identifying a set of management objectives and priorities, (ii) defining a simulation/operating model of the system to be managed, (iii) identifying candidate management strategies, (iv) using simulation to test the candidate management strategies, and (v) interpreting simulation results (Punt et al., 2016). In this process, a full MSE considers many sources of uncertainty, namely concerning monitoring, estimation and implementation (Siple et al., 2021). In this study we focus on recruitment uncertainty only.
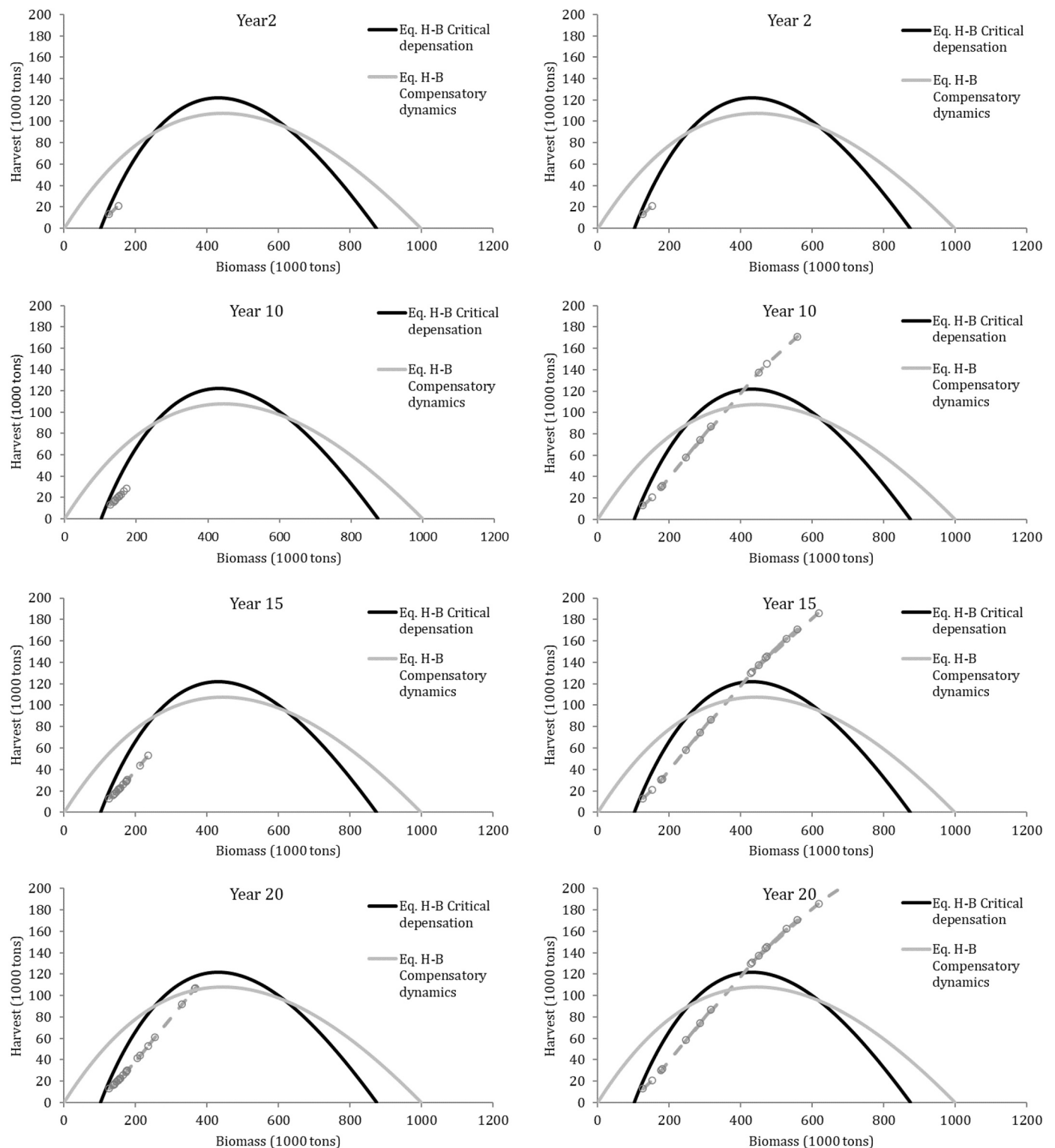
Fig. 6. Left panels: slow recovery simulation, right panels: fast recovery simulation.

relatively new and will require standardizing data collection and evaluation methods, and adapting established practices which are mostly based on biological concepts. (ICES, 2021). This paper contributes to that effort by proposing a method that can be directly integrated into current multiannual fishing advice frameworks. The method requires approximating optimal solutions of a bioeconomic model that depends on age-class distribution, into a function that defines harvest as a function of biomass only. When compared to solving the bioeconomic

model to define annual quotas, applying the obtained HCR will thus necessarily result in suboptimal recovery periods and lower NPVs. Note, however, that this is not a shortcoming of the proposed method, but is due to the characteristics of the original policy instrument itself: reference point-based HCRs are a biomass-based instrument neglecting population age structure. In our case, the optimal harvest and biomass trajectories rapidly converge toward a common path in the Harvest-Biomass state space. Consequently, the obtained HCR reasonably

approximates the properties of the bioeconomic model optimal solution. However, whether that will be the case for any given bioeconomic model ultimately depends on the model parameters and corresponding functional forms. In some cases, guaranteeing that the policy rule reasonably approximates the bioeconomic optimal solution may thus require combining a biomass-based HCR with age-class structure information. In this sense, the findings of this paper also contribute to the discussion regarding the benefits of policy instruments moving beyond a biomass-centered approach (see e.g., Da-Rocha and Mato-Amboage, 2016). Finally, fish populations are usually subject to large fluctuations resulting from uncertainty stemming from environmental shocks. In the case of pelagic species, shocks affecting recruitment largely determine fishing opportunities. This study proposes a method to derive a HCR using a deterministic bioeconomic model. Recently, though, various papers have studied optimal harvesting using age class structured populations under a stochastic setup (see, e.g., Tahvonen et al., 2018; Holden and Conrad, 2015; Kapaun and Quaas, 2013 and Colla-De-Robertis et al., 2019). In this context, investigating the role of uncertainty in the definition of economically informed HRCs also constitutes a promising research avenue.

Appendix A. Supplementary material

Supplementary information to this article can be found online at https://doi.org/10.1016/j.ecolecon.2022.107418.

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