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Ecolabel certification in multi-zone marine protected areas can incentivize sustainable fishing practices and offset the costs of fishing effort displacement

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ABSTRACT

As area-based marine conservation coverage expands to meet global targets, tension with fishing activities increases. While fully protected areas (FPAs) provide the largest range of long-term social-ecological benefits, their establishment has been constrained by difficulties arising from the short-term costs of protection, and associated limitations in economic incentives and in the resources required for effective implementation. Building on an existing bio-economic model for self-financed FPAs, we examine the economic and operational feasibility of establishing an ecolabel approach to balance the costs endured by fishers when implementing an FPA. Optimal increased profits can be achieved by designating the ecolabelled self-funded managed-fishing area for 20–25% of FPA. Multi-zone MPAs with a price premium derived from catch ecolabel certification inside partially protected areas surrounding FPAs provide incentives to help fishers engage into adopting sustainable fishing practices. Here we pave the way for more innovative approaches towards transformative changes for fisheries sustainability.

1. Introduction

In face of increasing climate change impacts and biodiversity loss, governments, as well as private companies and local communities need to re-think their relationships with nature (IPBES, 2022). In the ocean, overexploitation is one of the main threats on marine ecosystems (IPBES, 2019). From about 20 million tons of seafood in 1950 worldwide, about 90 million tons were extracted each year from the world ocean in 2018, with one third of stocks considered overexploited (The State of World Fisheries and Aquaculture, 2020). Despite the establishment of international principles to guarantee responsible fishing practices and ensure the conservation and sustainable management of marine resources on a global scale (FAO. Food and Agriculture Organisation, 1995), and progress in applying these principles in certain regions, unsustainable fishing practices as well as overexploitation persist in many parts of the world (Hiddink et al., 2011; Cashion et al., 2018; Pitcher et al., 2022; Reis-Filho and Loiola, 2022). As fishing makes a fundamental contribution to food, employment, recreation, trade, and economic and cultural well-being for people around the world, there is an urgent need for better fisheries management, for both nature and people.

Fisheries management has often been focused on ecological sustainability by setting and enforcing conservation measures such as catch limits (Hornborg et al., 2020), leaving the social and economic dimensions of fisheries management largely implicit. However, environmental outcomes are dependent on how social and economic dimensions are addressed (Fulton et al., 2011; Garlock et al., 2022; Thébaud et al., 2023). In many regions, limited capacity or political will to establish strong enforcement policies result in poor regulation of fishing activities, ultimately leading to illegal practices (Reis-Filho and Loiola, 2022). Income losses may also encourage illegal behavior, leading to unsustainable catch levels, habitat destruction and biodiversity loss (Desai and Shambaugh, 2021). There is a need to implement effective conservation and access regulation measures. This would enable both the protection of the production and reproduction potential of fisheries resources, and provide self-sustaining access regulations to enable

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transformative changes towards profitable and sustainable fisheries.

Marine Protected Areas (MPAs) represent an opportunity to improve biodiversity protection and the effectiveness of fisheries management, with solutions building on multidisciplinary knowledge of the ecological, economic and institutional dimensions of marine resources management (Whitehorn et al., 2019). However, to date, their design and the processes for their establishment have often met with difficulties, despite their expected long-term ecological and social benefits. MPAs are recognized as an efficient area-based management tool for marine conservation (Edgar et al., 2014; Chaigneau and Brown, 2016; Giakoumi et al., 2018; Rasheed, 2020; Grorud-Colvert et al., 2021). Fully and highly protected MPAs, the most effective levels of protection (Zupan et al., 2018; Grorud-Colvert et al., 2021), can deliver ecological outcomes within their boundaries (e.g., increases in fish biomass), and fisheries outcomes outside their boundaries, through spillover and larval replenishment (Halpern et al., 2009; Di Lorenzo et al., 2016, 2020). Countries parties to the Convention on Biological Diversity (CBD) have recently agreed within the Kunming-Montreal Global Biodiversity Framework to effectively conserve 30% of land and ocean by 2030. Over the past 20 years, the global coverage of MPA has shifted from less than 5 million km^2 to about 28 million km^2 (UNEP-WCMC and International Union for Conservation of, 2021). However, in many places, meeting these areal targets is being done at the expense of the quality and effectiveness of MPAs (De Santo, 2013; Claudet, 2018; Claudet et al., 2020, 2021; Roessger et al., 2022). Among these 7.9% of the world's coastal and marine waters designated as MPAs, only 3.6% are effectively implemented (with management tools in place), while 1,6% are not effectively managed and 2.1% are only proposed (Sala et al., 2018). As a result, more than half of the existing MPAs can be qualified as "paper parks". Also, among the implemented MPAs, most are under-protected (Roessger et al., 2022), despite the fact that lower levels of protection lead to limited ecological or socioeconomic benefits (Costello et al., 2016; Zupan et al., 2018; Smallhorn-West et al., 2022; Turnbull et al., 2021), except if coupled with a fully protected area (Zupan et al., 2018).

As the areal coverage of well protected MPAs expands, the extent of potential constraints imposed on fisheries is increasing, and is at the root of difficulties in implementing effective MPAs around the world (Notohamijoyo et al., 2020; Ponte, 2008). Lack of inclusion of stakeholders in decision-making processes, as well as consideration of the short-term impacts and how these could be compensated, act as barriers internationally to the implementation of fully and highly protected MPAs (Schultz et al., 2022). The ecological benefits of MPAs need a few years to materialize (Navarro et al., 2018; Kayal et al., 2020), with costs endured by fishers in the short-term (Stevenson et al., 2013; Rojas-Nazar et al., 2015). This entails resistance upstream of MPA implementation decisions. In addition, MPA effectiveness is also limited by resource constraints to support governance of established MPAs (Schultz et al., 2022). Key factors required to increase support and effectiveness of MPAs include enhancing stakeholder participation, providing financial support for management and increasing management accountability under improved governance arrangements (Schultz et al., 2022).

Only limited transformative approaches have been proposed to counteract these barriers to the establishment of MPAs with full and high levels of protection. Millage et al. (2021) proposed an interesting approach to address poaching in MPAs. Their work focused on the feasibility of funding surveillance in an MPA through the establishment of a "conservation financed area" within a fully protected area. Here, we propose a novel and complementary approach to the work of Millage et al. (2021), by adapting their bio-economic model to consider the barriers relating to the short-term economic costs experienced by fishers. We consider a multi-zone MPA scenario where fishers can engage into adopting sustainable fishing practices inside partially protected areas surrounding fully protected areas, with price premiums for their catch associated with an ecolabel certification. We aim to assess the extent to which such a certification system could (1) offset the short-term conservation costs to the fisheries sector, (2) ensure the long-term sustainability of stocks, and (3) pave the way towards transformative changes in fishing practices. We discuss operationalization pathways for such an ecolabel certification system in multi-zone MPAs, considering the importance of co-management arrangements in multiple social-cultural contexts.

2. Method

2.1. A marine protected areas with self-financed enforcement and ecolabelled fishing practices

We modeled a system spatially divided into two zones: an MPA, and an area open to fishing (F). The MPA could be further divided into two zones: a Fully Protected Area (FPA) and a Partially Protected Area (PPA) acting as a buffer zone between the FPA and the open area. Our approach builds on Millage et al.'s model (2021) that considered entry fees for fishers to fish in the PPA. The funds thus recovered add to a pre-existing exogenous enforcement budget that can help finance the enforcement of the MPA, to curb illegal fishing. Millage et al. (2021) showed that by designating 25–50% of the MPA as restricted access, the cost of fishing illegally became greater than the cost of legally accessing a more productive, regulated fishing area.

Here, we complemented this model by adding the possibility for fishers operating in the PPA to benefit from the sale of ecolabelled fish catch (Fig. 1). We assumed that with such a label, fish caught in this area could be sold with a price premium, or that the quality of the fish landed and the perception that they are sustainably caught could ensure a better market share for fishers. Such a price premium would at least cover the costs associated with obtaining and using an ecolabel (Asche and Bronnmann, 2017). This PPA, which we called "Ecolabelled and Financed Area for Conservation" (EFAC), would effectively enable the development of sustainable fishing practices aimed at maximizing the value of catch. We assumed that this would materialize by fishing effort levels set so as to achieve Maximum Economic Yield (MEY). The benefits to fishers derived from implementing such a strategy and the associated ecolabel could be considered akin to a "Payment for ecosystem services", which has already been adopted in the fisheries sector (Begossi et al., 2011; Failler et al., 2019). Indeed, under such schemes, fishers are financially rewarded for adopting fishing techniques and intensity designed to ensure sustainable harvests.

With this adapted model of a multi-zone MPA, we considered the conditions under which the loss of fishing grounds as a result of implementing an FPA could be offset by the benefits derived from fishing in the EFAC, while still effectively deterring illegal fishing under the MPA's self-financing framework. The effectiveness of this multi-zone MPA design has never been tested and its originality lies in the simultaneous consideration of conservation performance and economic viability of the fishery.

2.2. Bio-economic model

The evaluation of the ecological and economic relevance of an eco-certification system in a marine protected area was carried out by adapting a bio-economic model initially developed by Millage et al. (2021) to focus on enforcement costs. Following Millage et al. (2021), our model is established in discrete time and considers a monospecific exploited metapopulation divided into two patches i: an open access fishing area (patch F), and a multi-zone MPA (patch MPA). The two populations grow independently according to a logistic growth. The biomass of the population of patch i at time t can be written as:

$$g\left(X_{i,t}\right) = X_{i,t} + rX_{i,t}\left(1 - \frac{X_{i,t}}{K}\right) - H_{j,t} \tag{1}$$

with r, the intrinsic growth rate, K, the carrying capacity of the environment and $H_{j,t}$ the catch of individuals by the fleet j in the

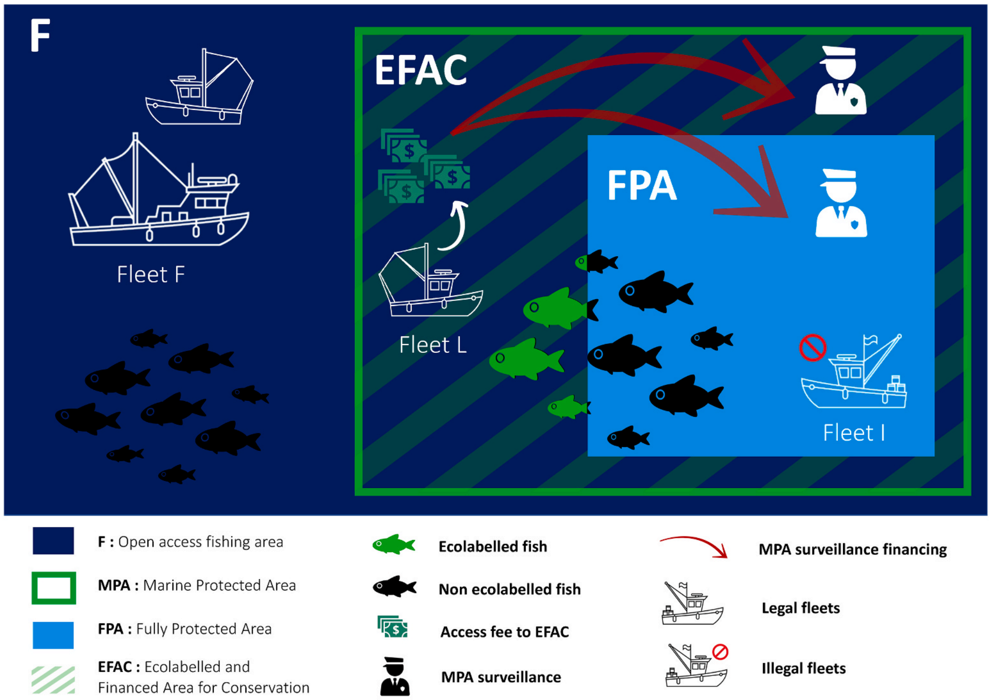

Fig. 1. Schematic organization of a multi-zone marine protected area (MPA) system with an ecolabelled and financed area for conservation (EFAC) surrounding a fully protected area (FPA) and adjacent to an open access area (F). Fishers wishing to benefit from the EFAC pay an access fee used for MPA surveillance and obtain higher revenues from ecolabelled catch.

corresponding patch i, in time t. Movements of the species between the two patches were modelled through a dispersion matrix (D). At each time step, a fraction of each patch's biomass is redistributed in the other patch while the remaining fraction remains in the original patch. The distribution is uniform in each patch and age and population structure were not considered. The biomass in each patch at time $t+1$ can be represented as follows:

$$D = \begin{bmatrix} d_{F,F} & d_{MPA,F} \\ d_{F,MPA} & d_{MPA,MPA} \end{bmatrix} \tag{2}$$

$$\boldsymbol{X_{F,t+1}} = g(X_{F,t})d_{F,F} + g(X_{MPA,t})d_{MPA,F} \tag{3}$$

$$\boldsymbol{X_{MPA,t+1}} = g(X_{MPA,t})d_{MPA,MPA} + g(X_{F,t})d_{F,MPA} \tag{4}$$

with $d_{F,F}$, the fraction of the stock in F that remains in F and $d_{F,MPA}$, the fraction of the F stock that moves into the MPA, and vice versa. We considered three fishing fleets j: a fleet 'F' that operates in the open access patch (F), a fleet 'L' that operates in the EFAC area (L for Lease area), and an illegal fleet 'I' that operates in the entire MPA. Total harvest of each fleet j is dependent on their fishing effort $E_{j,t}$ (expressed in fishing days), the catchability of the prey, q, and the fraction of biomass available in the targeted patch, $f(X_t, L, D)$:

$$\boldsymbol{H_{j,t}} = qE_{j,t}[f(X_t, L, D)] \tag{5}$$

Fishers derive a revenue from their catches, $pH_{j,t}$. They also pay the costs of their fishing activity $cE_{j,t}$, and depending on the fleet j, an EFAC access fee or a fine may be added, such that the net profit of each fleet may be written as:

$$\boldsymbol{\pi_{F,t}} = pqE_{F,t}X_{F,t} - cE_{F,t}^{\beta} \tag{6}$$

$$\boldsymbol{\pi_{L,t}} = pqE_{L,t}X_{MPA,t}L - cE_{L,t}^{\beta} - \chi E_{L,t} \tag{7}$$

$$\boldsymbol{\pi_{I,t}} = pqE_{I,t}(X_{MPA,t} - H_{L,t}) - cE_{I,t}^{\beta} - \theta\varphi E_{I,t} \tag{8}$$

with p the price of fish, c a fishing cost parameter, β a coefficient that determines the shape of the cost curve ($\beta > 1$ implies a more than proportional increase in fishing costs with effort), χ the EFAC access fee, θ the probability of detection of an illegal fleet, and φ the fine paid after an illegal fleet has been apprehended. In Millage et al. (2021) the initial model (scenario 0) has the fishing effort of each fleet gravitates at each time period t towards the patches with the highest marginal profit. They are calculated so as to maximize the short-term profit of fishers (Costello et al., 2015).

$$\boldsymbol{E_{F,t}} = \left[\frac{pqX_{F,t}}{\beta c}\right]^{\left(\frac{1}{(\beta-1)}\right)} \tag{9}$$

$$\boldsymbol{E_{L,t}} = \left[\frac{pqX_{MPA,t}L - \chi}{\beta c}\right]^{\left(\frac{1}{\beta-1}\right)} \tag{10}$$

$$\boldsymbol{E_{I,t}} = \left[\frac{pq(X_{MPA,t} - H_{L,t}) - \theta\varphi}{\beta c}\right]^{\left(\frac{1}{\beta-1}\right)} \tag{11}$$

A monitoring budget for the MPA (B) was also considered and corresponds to the sum of the rents obtained by the EFAC ($\chi\ E_{L,t}$) and an exogenous monitoring budget (b, potentially $= 0$). This is associated with monitoring costs (α) per unit of monitoring effort (δ) so that:

$$\boldsymbol{B} = b + \chi E_L \tag{12}$$

$$\boldsymbol{\delta} = \frac{b + \chi E_L}{\alpha} \tag{13}$$

$$\boldsymbol{\theta} = 1 - e^{-\mu\delta} \tag{14}$$

The enforcement effort δ is directly related to a probability of detecting illegal fishing effort (θ), which also depends on a parameter μ: the speed of detection of illegal fishing effort as a function of the applied enforcement effort. The smaller the area under enforcement, the faster illegal fishing will be detected.

2.3. Modeling the ecolabeling scheme

We first considered the implementation of sustainable fishing practices in the EFAC for the fleet L by computing fishing effort that maximizes the short-term economic profit of the fleet fishing in the EFAC (Maximum Economic Yield, MEY). While this is different to the level of effort that would maximize the net present value of the fishery in the EFAC, this seemed closer to the reality of how levels of fishing effort might be established in this area, while also ensuring that the levels of harvest remain lower, and biomass levels higher, than those associated with Maximum Sustainable Yield. MEY effort was therefore obtained by calculating the first derivative of the profit and setting this to 0 (i.e., $\left.\frac{\partial \pi_L}{\partial E_L} = 0\right)$ (see appendix A). The two other fleets F and I were assumed to be operating under an Open Access strategy (OA), subject to enforcement capacity in the FPA, leading towards zero profit levels. Indeed, in such a strategy, i.e., at bionomic equilibrium (see appendix B), incomes equal costs (i.e., $\pi_F = \pi_I = 0$). Corresponding fishing efforts for each fleet are defined as follows:

$$\boldsymbol{E_{F,t}} = \left[\frac{pqX_{F,t}}{c}\right]^{\left(\frac{1}{(\beta-1)}\right)} \tag{15}$$

$$\boldsymbol{E_{L,t}} = \left[\frac{pqX_{MPA,t}L - \chi}{\beta c}\right]^{\left(\frac{1}{\beta-1}\right)} \tag{16}$$

$$\boldsymbol{E_{I,t}} = \left[\frac{pq\left(X_{MPA,t} - H_{L,t}\right) - \theta\varphi}{c}\right]^{\left(\frac{1}{\beta-1}\right)} \tag{17}$$

These management strategies and corresponding fishing effort levels (short-term MEY for fleet L and OA for fleets F and I) constitutes our control scenario (scenario 1).

The ecolabelled fish price in the EFAC, p_L, is expected to be higher than the fish price p in the open access area, under the assumption that the fish caught in the lease area are larger and/or of better quality benefitting from the full protection of the FPA (Lester et al., 2009; Zupan et al., 2018; Jaco and Steele, 2020) (scenario 2). The ecolabelled price can also secure better market shares due to consumer preference for ecolabelled seafood. This results in the possibility of securing higher prices for the ecolabelled products compared to those obtained in the open access area or in the illegally fished FPA. p is therefore multiplied by a parameter ε (found to range between 1.1 and 1.9 after calibration, to achieve population equilibrium) to obtain p_L. The range of the values that can be taken by p_L is discussed further. The fish caught by the illegal fleet is at identical price p than the one from the F patch.

$$\boldsymbol{p_L} = p\varepsilon \tag{18}$$

We considered the impact of lower detectability of poaching μ in the EFAC, as compared to the FPA, where the sole presence of a fishing boat, easier to observe, is an infringement (scenario 3). Conversely, we assumed that fleet monitoring requires more effort in the EFAC because of the need to monitor the fishing activity of multiple fishing vessels.

$$\theta_L = 1 - e^{-\mu_L \delta} \tag{19}$$

$$\theta_{FPA} = 1 - e^{-\mu_{FPA} \delta} \tag{20}$$

with μ_L the speed of detection of an illegal fleet in the EFAC and μ_{FPA} the speed of detection of an illegal fleet in the FPA ($\mu_L < \mu_{FPA}$) and the corresponding probability of detecting an illegal fleet (θ_L and θ_{FPA}).

Finally, entry fees to the EFAC were set to be dependent on the profit π_L that fishers make from fishing in this area (scenario 4):

$$\chi = \pi_L^o \tag{21}$$

with o an exponent that describes the nature of the relationship between the fisher's profit in the EFAC and the access fee that must be payed to access this area (increasing non-linearly with $o = 0.5$).

Six scenarios (Table 1) were thus considered and compared: the control scenario was compared with the results of Millage et al. (2021) while scenarios 2, 3, 4, and 5 were compared with the control scenario.

All simulations were performed in R (R Core Team, 2021). The model was calibrated with biological and economic parameter values from a purse seine fishery for a tuna species, *Katsuwonus pelamis* (Skipjack), managed on the basis of a vessel days scheme in the Pacific Ocean (Villaseñor-Derbez et al., 2020) (see Appendix C for reference values). The simulations were run over 50 year time horizons. For each simulation, the EFAC size that maximizes the equilibrium biomass of the system and the profit of the fishers in EFAC was recorded. Sensitivity analyses of the model were conducted by varying the value of the following parameters: EFAC access fees, the price of ecolabelled fish, enforcement costs and dispersion patterns (results presented in Appendix D). The ranges retained for these parameters were selected such that an equilibrium fish population can be achieved over the simulation horizon.

3. Results

Under Millage et al. (2021)'s self-financed surveillance regime of an MPA, when the entire MPA is fully protected as an FPA, i.e., when the proportion of the Conservation Financed Area (CFA) is equal to 0, the optimal biomass of the system is at its lowest level (74% of K, Fig. 2A, black curve). There is indeed an economic advantage to fishing illegally in the FPA given the lack of funding for MPA monitoring and resulting low probability of detection and sanction. Similarly, when the entire MPA is subject to regulated fishing access, i.e., the proportion of the CFA is equal to 100%, the biomass of the system is also at low levels. Even in the absence of poaching, fishing effort is applied throughout the system and there is no refuge area for the exploited population. Conversely, an intermediate FPA size, between 25 and 50% of the CFA, allows the biomass of the system to reach 84% of its carrying capacity K, with a

Table 1
Scenarios to assess the proposed ecolabelling scheme.

Scenarios	Equations and/or parameters	Source
Scenario 0 – Fishery short-term profit maximization	$E_{F,t} = \left[\frac{pqX_{F,t}}{\beta c}\right]^{\left(\frac{1}{(\beta-1)}\right)}$ (9) $E_{L,t} = \left[\frac{pqX_{MPA,t}L - \chi}{\beta c}\right]^{\left(\frac{1}{(\beta-1)}\right)}$ (10) $E_{I,t} = \left[\frac{pq\left(X_{MPA,t} - H_{L,t}\right) - \theta\varphi}{\beta c}\right]^{\left(\frac{1}{\beta-1}\right)}$ (11)	Millage et al. (2021)
Scenario 1 – Control (OA + MEY)	$E_{F,t} = \left[\frac{pqX_{F,t}}{c}\right]^{\left(\frac{1}{(\beta-1)}\right)}$ (15) $E_{L,t} = \left[\frac{pqX_{MPA,t}L - \chi}{\beta c}\right]^{\left(\frac{1}{\beta-1}\right)}$ (16) $E_{I,t} = \left[\frac{pq\left(X_{MPA,t} - H_{L,t}\right) - \theta\varphi}{c}\right]^{\left(\frac{1}{\beta-1}\right)}$ (17)	(Eide, 2021, p. 89)
Scenario 2 - Fish price	$\boldsymbol{p_L = p\ \varepsilon\ (\varepsilon = 1.9)}$ (18)	Teisl et al. (2002)
Scenario 3 – Detection speed	μ_L (5e −6) and $\boldsymbol{\mu_{FPA}}$ (5e − 4)	–
Scenario 4 – Access fee	$\chi = \boldsymbol{\pi_L^o}$ (21)	–
Scenario 5 – Combined scenarios	$\boldsymbol{p_L, \mu_L, \mu_{FPA}}$, **o** *(Similar values than previously)*	–

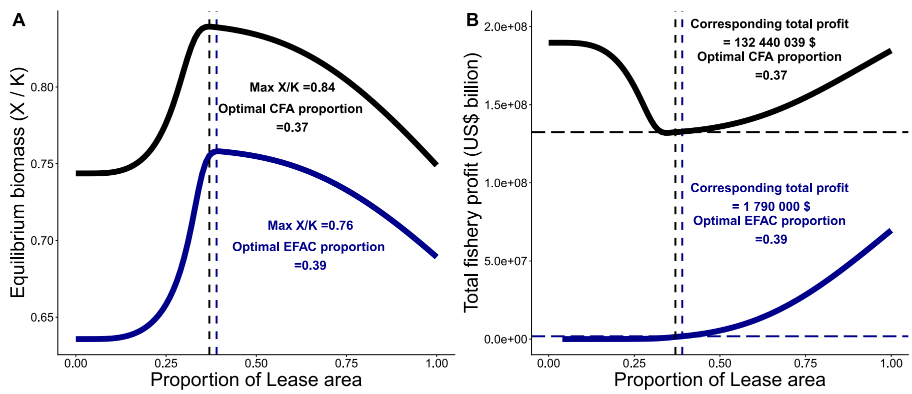

Fig. 2. Total population equilibrium relative to carrying capacity (X/K) (A) and total fishery profit (B) for different proportion of the Ecolabelled and Financed Area for Conservation (EFAC) in the marine proetcted area compared between scenario 0 (Millage et al. (2021)'s model) represented with black curves, and the control scenario (scenario 1, open access strategy for fleets F and I and Maximum Economic Yield (MEY) for fleet L) represented with blue curves. Vertical dashed lines indicate the lease area size corresponding to the highest possible equilibrium biomass in each scenario, whereas horizontal dashed lines indicate the corresponding profit. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

higher density of individuals in the MPA compared to the open access area.

In the reference scenario, when fishing fleets F and I operate under open access and fleet L operates under MEY (scenario 1), the optimal proportions of CFA and EFAC are 37% and 39% of the MPA, respectively (Fig. 2A, comparing blue and black curves). While the size of the EFAC appears similar to that of the CFA of Millage et al. (2021), clear differences in biomass and total profit are observed. The equilibrium biomass is about 8% lower than the one calculated by Millage et al. (2021) (76%). Similarly, the total profit of the fishery is 74 times lower compared to the profit obtained under the conditions established by Millage et al. (2021) (Fig. 2B).

Scenarios 2, 3, 4 and 5 (Fig. 3) allow us to identify the influence of each parameter of the bio-economic model on the equilibrium biomass and the overall profit of the fishery relative to the reference situation (control scenario 1). In all scenarios, the biomass of the system reaches a maximum of 76% of the carrying capacity. However, depending on the model input parameters' modification, total biomass is maximized for different EFAC sizes and profit values. When we apply a higher ecolabelled price to the catch from the EFAC area (scenario 2), we observe a decrease by 18% in the proportion of the EFAC area required to achieve maximum biomass compared to the control scenario (Fig. 3A). Corresponding profit for the fishery reaches $2,000,000/year compared to the initial $1,790,000/year (Fig. 3B). The decrease in the speed of detection of an illegal fleet in the EFAC (scenario 3) results in an increase in the proportion of the EFAC in the MPA of 7%, as compared to the control scenario (Fig. 3C), to achieve maximum biomass. However, the profit increases significantly, with $3,760,000/year, i.e., more than a two-fold increase compared to the reference scenario (Fig. 3D). In scenario 4, the establishment of access fees dependent on the profit of the fleet L barely affects the optimal proportion of EFAC. The two areas represent just over 35% of the MPA, while profit is divided by about 1.4, decreasing to $1,210,000/year (Fig. 3E and F). When the three policy changes are combined (scenario 5), a 17% decrease in the EFAC is observed compared to the control scenario. We can observe an offsetting effect on the total profit of the fishery, which is not as high as in scenario 2, but it is higher than for the control scenario and scenario 4 ($2,450,000/year, i.e., a 1.4 fold increase compared to the control scenario).

In summary, our results show that the ecolabelling of fish associated with sustainable fishing practices always results in a significant decrease in the EFAC size required to achieve maximum biomass, in favor of a larger FPA. On the other hand, the adjustment of monitoring costs (and associated detection speed) and entry fees have less influence on the optimal EFAC size (i.e., between 2 and 7% difference). Each scenario has a significant influence on the total fishery profit, which is strongly dependent on the price of the ecolabelled fish and on the detection speed of an illegal fleet in the EFAC. It is interesting to note that each of the simulated policy changes implies an increase in the total fishery profit, compared to the control scenario, except for scenario 4. In addition, for an almost identical equilibrium biomass, the total profit of the fishery always increases exponentially when there is a price premium for the fish caught in the EFAC. In fact, the presence of an ecolabel on fishery products from the EFAC generates maximum profits when the EFAC area equals 100% of the MPA (scenarios 2 and 5). However, under such a scenario, i.e., a well-managed fully accessible MPA without an FPA, the equilibrium biomasses are close to 0. The absence of a price premium for fish caught in the EFAC does not counterbalance this tendency in scenarios 3 and 4, in which biomasses are also reduced, although not down to zero.

4. Discussion

We proposed a new approach to the design and management of Marine Protected Areas (MPAs), with the ambition to reconcile conservation objectives with the economic and social well-being of local stakeholders. We introduced the concept of "Ecolabelled and Financed Area for Conservation" (EFAC), associated with a price premium for fish caught in this zone, to foster sustainable fishing practices in the vicinity of a Fully Protected Area (FPA). We modelled the ecological and socioeconomic outcomes of the proposed multi-zone MPA and assessed whether it could offset the short-term conservation costs to the fisheries sector, paving the way towards transformative changes in fishing practices. Under the assumptions made to represent our modelled MPA, the implementation of an ecolabel on fishery products from an EFAC can lead to an increase in the total fishery profit with an equilibrium total biomass at 76% of its carrying capacity. The optimal proportion of the EFAC ranges from 20 to 25% of the total size of the MPA.

The implementation of an EFAC supposes a modification of fleets' fishing effort regimes, resulting in biomass and profit differences between the contrasted scenarios. While fleet L is assumed to operate to maximize short-term profit (MEY), fleets F and I are assumed to operate under open access equilibrium, leading towards zero profit levels. This contributes to explain the lower total fishery profits we observe, compared to the scenario considered by Millage et al. (2021), as well as the higher level of fishing effort, leading to lower biomass. While the static nature of MEY and open access equilibrium limits their reliability as practical fishery management tools (Seijo et al., 1998), they can provide useful indications as to the direction towards which alternative management situations may lead a fishery.

Bio-economic equilibriums are expressed for the case of perfect markets, for which the price of a product is determined when demand meets the good supply. However, it is necessary to ensure that the fishing effort applied in the EFAC is sustainable and lower than in the open access area. Even if the situation of bio-economic equilibrium is idealized and rare, price formation often leads to prices close to those that would be found in perfect markets (Eide, 2021). Although lower than the results of Millage et al. (2021), the equilibrium biomass values and economic benefits obtained under the ecolabelling scheme make

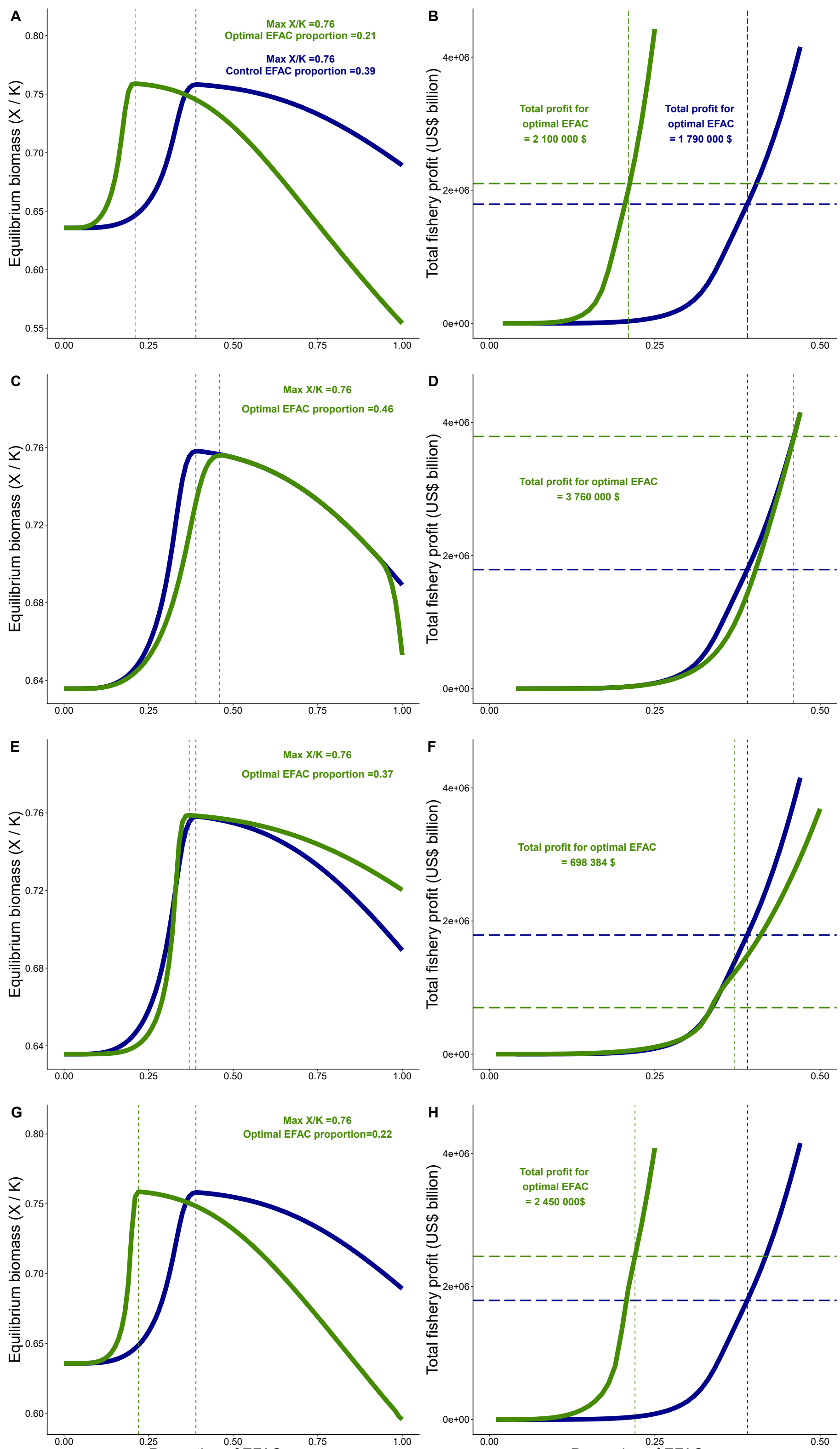

Fig. 3. Total population equilibrium relative to carrying capacity (X/K) (A, C, E, G) and total fishery profit (B, D, F, H) for different proportion of the Ecolabelled and Financed Area for Conservation (EFAC) in the marine protected area and compared across the 4 scenarios considered (scenario 2 “fish price” in panels A and B, scenario 3 “detection speed” in C and D, scenario 4 “access fee” in E and F, and scenario 5 “all combined” in G and H). Results assuming an open access strategy for fleets F and I (scenario 1, the control scenario) are represented in blue whereas the modified scenarios are in green. Vertical dashed lines indicate the lease area size corresponding to the highest possible equilibrium biomass in each scenario (note the different scale for total fishery profit panels), whereas horizontal dashed lines indicate the corresponding profit. The equilibrium biomass in the control scenario is equal to 0.76 of the carrying capacity for an EFAC proportion of 0.39 generating a gain of $ 1,790,000 (blue curves). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

sense for markets that initially tend toward bio-economic equilibrium fisheries management.

In multi-zone MPAs, increasing the size of FPAs at the expense of buffer zones allows a larger fraction of mobile commercial fish to remain within the FPA (Villegas-Ríos et al., 2021). This may also increase recruitment of individuals (Almany et al., 2007; Berumen et al., 2012). Here, reducing the size of the EFAC area to about 20-25% of the MPA could therefore contribute to improving overall conservation objectives, with the increase in fish prices due to the ecolabel price premium also allowing fishery profits to increase. This system can provide strong socio-economic and ecological benefits. The implementation of an MPA with a well-managed EFAC and an ecolabel generates synergies between conservation and socio-economic objectives, conciliating conservation targets and local stakeholders' interest.

The use of ecolabels has increased in fisheries to improve or maintain market share (Teisl et al., 2002; Anderson et al., 2021). In our model, the price premium of the ecolabel was limited to twice the initial price. This is of the same order of magnitude as what has been observed in Pacific tuna fisheries where the "Dolphin Safe label" has been adopted (Teisl et al., 2002). Both in the case our model is based on, and with the Dolphine Safe label, most of the tuna are sold on a similar north American market (Villaseñor-Derbez et al., 2020). Besides, since the cost of accessing the EFAC area is dependent on the profit of the fishers (following a non-linear increasing function, see equation (21)), increases in the price premium might lead to access fees being perceived as too high. In turn, this could become an incentive to increased poaching, reducing the sustainable model of the proposed MPA zoning scheme.

We considered in our model the potential impacts of ecolabelled fish price's variation via sensitivity analysis (appendix D). However, we did not account for the potential changes of fish price (with or without premium) resulting from modifications in the fish supply from the MPA, and according to potential supply and demand changes in the market (Auger et al., 2010; Ramachandran and Parappurathu, 2020). Indeed, we considered that the fishery is a price-taker, so that production of fish from the MPA, both legally and non-legally caught, does not affect the price of the fish on the market. In addition, the ecolabelled and non-ecolabelled products should belong to different markets, shouldn't be considered as substitutes and should be processed via different marketing channels. This implies that the supply of ecolabelled fish would not impact the price of non-ecolabelled fish, or vice versa.

The success of an ecolabel is largely dependent on its perception by consumers. Over the past two decades, ecolabels have become strong instruments to empower consumers towards environmental awareness and sustainable consumption patterns (Giacomarra et al., 2021), in particular in the fisheries sector (Vitale et al., 2017). However, consumer's willingness to pay price premiums for sustainably certified seafood products varies among species, countries, brand, consumer's awareness and perception of local and global causes, demographic parameters, cultural heritage, economic conditions, market, and supply chains (Asche and Bronnmann, 2017; Vitale et al., 2017; Menozzi et al., 2020; Kitano and Yamamoto, 2021). There is inevitably a need to adapt ecolabelling strategies according to the context (Menozzi et al., 2020), especially where demand for eco-labeled fish is moderate (Pérez-Ramírez et al., 2015; Blomquist et al., 2015). Limited understanding of what sustainability refers to is one of the main reasons why consumers might not be receptive to ecolabels (Winson et al., 2022). Consumers show better willingness-to-pay when they have been adequately informed on the standard and underlying requirements, such as the conservation status of the commercial species, and when they trust the certification processes (Uchida et al., 2014; Vitale et al., 2017). Of particular relevance in the context of MPAs is the fact that consumers often attach importance to the location of origin of the products they purchase. This could provide a strong basis for ecolabelling strategies such as the one we describe if the certification processes in place are perceived as being strictly enforced. Beyond ensuring that fishing activities allowed in the EFAC are well managed, the operationalization of our ecolabel scenario should also guarantee that the fishing activities do not endanger ecosystem components other than the target stock and that the MPA meets its biodiversity conservation objectives (Booth et al., 2021).

When proposing the implementation of an ecolabel on fish products from MPAs, it is important to assess its operational and economic feasibility, in particular the additional costs involved. Millage et al. (2021) assume that costs of administering the lease area program are implicitly factored into the access fee to the managed fishing area. Similarly, part of the costs of implementing an MPA ecolabel could also be included in the EFAC access fees. However, costs of setting up such a program are likely to be higher than those of creating the Conservation Financed Area. The benefits of optimizing the price of fish in EFAC (scenario 2 or 5), could then be used to compensate these costs. Of course, the EFAC access fee should not exceed the additional benefit fishers receive from fishing in that area (Mainardi, 2019), in order to preserve the incentives to fish sustainably.

The MPA ecolabel could benefit from a reference framework and a control plan defined by law and set by decree, in line with the FAO guidelines on sustainable fishing (1995). The reference framework should be divided into a production and a marketing standard. For the production standard, it would be appropriate to adopt a certification unit of the type: 1 species x 1 gear or fishing method x 1 fishing area (EFAC). In this way, the EFAC area could allow an eco-certification for different exploited species and fishing methods. For the marketing standard, the origin of the ecolabelled fish could be ensured by establishing separate fishing access regulations for the open access area and for the EFAC. The landings should be clearly associated with a defined area, enabling the identification of sustainably caught fish. At further stages in the supply chain, the ecolabel tag should be included at all levels of processing, packaging and sale. This will ensure clear separation of certified and non-certified products.

Finally, the eco-certification of fishery products from an MPA could benefit from being co-developed with local stakeholders. Co-management approaches involving close collaboration between fishers, scientists, and managers, including bottom-up management regimes and the allocation of exclusive access rights to fishers, appear promising for achieving conservation, fisheries management, and fishing community welfare goals (Guidetti and Claudet, 2010; Bennett et al., 2019; Shah et al., 2019; Di Franco et al., 2020; Smallhorn-West et al., 2020).

5. Conclusion

Lost fishing grounds and the associated short-term increase in fishing costs are often presented as barriers to the implementation of fully protected areas. Here, we showed that setting up an ecolabel approach could balance the short-term costs endured by fishers after a fully protected area implementation, and promote transformations towards more sustainable fishing practices. This new marine protected area design could be a solution to reconcile conservation objectives with the economic and social well-being of stakeholders. This novel approach could leverage multiple challenges to the establishment of fully protected areas and pave the way to reach the 2030 protected areas targets of the global policy agenda.

Declaration of competing interest

The Authors declare no conflicts of interest.

Data availability

No data was used for the research described in the article.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.esg.2023.100184.

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