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Contents

1 Introduction

Bioeconomic modeling has become one of the key tools for fisheries management. Any bioeconomic model integrates biological and economic aspects with the purpose of characterizing the status of natural resources. If this integration is appropriate the model can be successfully applied to assess the bioeconomic impact of different management strategies.

In developing a bioeconomic model it is crucial to include the goal of managers since this is the measure used to rank the different management strategies. As pointed out by Lleonart et al. (2003) the objectives of fishery managers are diverse and often contradictory. They may seek be to maximize fishing profits, revenues or yield, to minimize catch fluctuations, to avoid the risk of collapse of the resource, to maintain employment, etc. Nevertheless, bioeconomic fishery models typically consider that the goal is to find exploitation paths that optimize profits or other economic variables. This has been the typical approach from the pioneering surplus production models (Gordon, 1954; Scott, 1955) up to more sophisticated models that include the age-structure of the population (Clark, 1990; Lleonart et al., 2003; Tahvonen, 2009; Da Rocha et al., 2010; Skonhoft et al., 2012).

Basic bioeconomic models have also been developed to incorporate some new concerns in fishery management. Issues such as marine biodiversity (Alexander, 1954; Bertram and Quaas, 2017), harvest control rules (Da Rocha and Mato-Amboage, 2016), quota management (Costello et al., 2008; Little et al., 2009; Marchal et al., 2011), and strategic behavior (Jensen and Vestergaard, 2002; Merino et al., 2007; Aanesen and Armstrong, 2016) have enriched the understanding of bioeconomic models as a tool for management purposes.

Unwanted catches have also been regarded as a key issue in commercial fishing worldwide (Kelleher, 2005). Most unwanted catches are discarded and returned to the sea with a low survival rate, especially for fish (Revill, 2012; Guillen et al., 2014). In the case of the EU, discard levels vary considerably from one location and gear to another. For instance for the

Galician fleet discard rates range from insignificant (coastal trolling fleet) to 43.5% (offshore trawling fleet) of total catches (Vázquez-Rowe et al., 2011). In the Mediterranean Sea, Tsagarakis et al. (2014) find that discards account, on average, for 18.6% of total catches, and are concentrated mainly in the trawls despite their relatively low contribution to catches. However, other biological factors such as the depth of waters (Sánchez et al., 2004) and the fishing intensity (Sánchez et al., 2007) also foster discards in the Mediterranean Sea. On the other hand, for most species there is a greater variability in discard rates across regions than across fisheries, suggesting that a region-by-region approach to discard reduction would be more meaningful (Uhlmann et al., 2014). All these concerns about discards have been taken into account in the latest reform of the EU Common Fisheries Policy (Article 15, EU (2013)), which includes a land obligation (LO) that requires all unwanted catches to be kept on board, landed and counted against quotas. This landing obligation will enter fully into force in all EU waters by 2019.

Although the LO will not be fully in force until 2019, its effects have already been studied in light of a bioeconomic modeling framework. For instance Batsleer et al. (2016) apply the Dynamic State Variable Model proposed in Clark and Mangel (2000) and state that restrictive quotas do not necessarily lead to a reduction in discards when a discard ban is not properly enforced. Prellezo et al. (2016) use the FLBEIA simulation bioeconomic model (Garcia et al., 2013; García et al., 2016) to estimate the effects of the LO on the Basque trawling fleet operating in the Bay of Biscay. Their results are mixed. The LO is likely to have a negative short term effect on the economic performance of the fleet, but at the same time there are likeky to be private incentives to improve selectivity to reduce discards.

Following this research line, our stated purpose in the MINOUW project was to update the software behind the bioeconomic framework that, which we named SASOM (Stochastic Age-Structured Optimization Model) with the goal of including discard analysis in determining reference points. This main objective has now been achieved by extending the multi-species model presented in Da Rocha et al. (2012b). These new developments have been applied to estimate the economic impact of the LO on the small-scale fisheries in Galicia. The main finding of this study is that future catches under the LO are likely to be only 50% of those expected in the absence of the LO. These results are published in Villasante et al. (2016b). We describe the main characteristics and the results of this study in Section 4.

Moreover, new bioeconomic developments also enables policies that seek to mitigate unwanted catches to be assessed. Several policies come to mind that seel to bring about reductions in unwanted catches. Some focus on promoting the adoption of fishing technologies that improve selectivity and effectiveness in harvesting target species (Catchpole et al., 2006; Kraak et al., 2013; Condie et al., 2013). Others focus more on market control by banning the sale of unwanted catches (Tolotti et al., 2015) though this may stimulate the black market, especially for juvenile fish (Catchpole et al., 2017). In fact the LO policy promoted by the EU is a combination of both these types of measure. We use the new model developments to analyze the bioeconomic effects of these two types of policy in the Southern Iberian hake stock. In particular reference points for this mixed fishery are computed under some scenarios representing the two policies as the steady state solutions of a dynamic optimal management problem. The optimization problem takes into account that species are caught simultaneously in unselective fishing.

The rest of the report is structured as follows. Section 2 details the characteristics of the SASOM framework and the developments made by the MINOUW partner UPV-EHU (jontly with the Universidade de Vigo) to provide advice on different aspects of fishery management. Section 3 sets out the multi-species model used to assess some policies regarding the LO regulation which is a modification of the model presented in Da Rocha et al. (2012b). Section 4 describes how these new developments are applied to estimate the economic impact of the LO on small-scale fisheries in Galicia. In Section 5 the new model developments are applied to the Southern Iberian hake stock in order to analyze the bioeconomic effects of two types

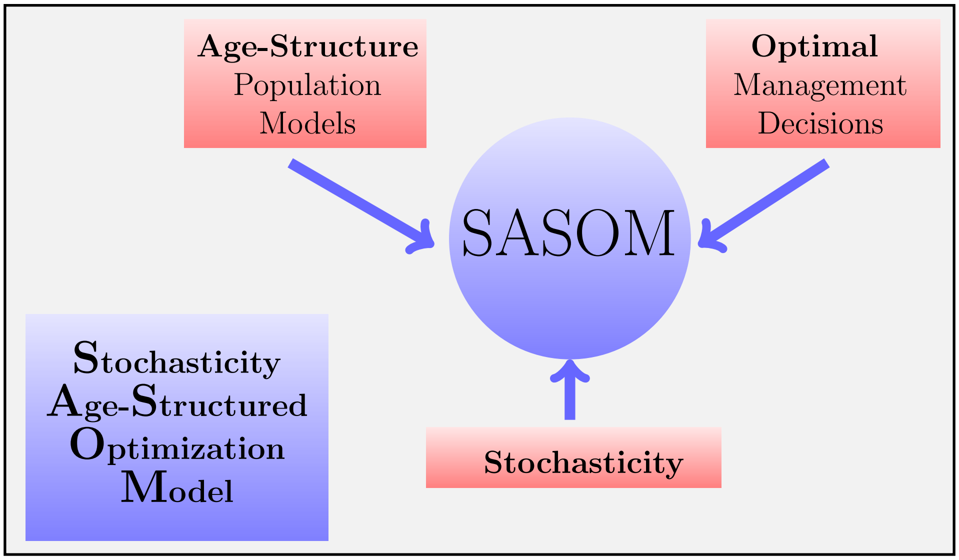

Figure 1: SASOM framework. Main characteristics

of policy intended to reduce unwanted catches. Section 6 concludes.

2 The SASOM framework

The key aspects of the SASOM framework are the following: *i)* Optimality. Solutions derived from SASOM always represent the optimal responses of agents given the economic and biological settings of the fishery; *ii)* Age-Structure Populations. The resource is structured in cohorts, i.e. in groups of fish that have the same age and probably the same size, weight and maturity time; *iii)* Stochasticity. There are shocks affecting economic and/or biological aspects of the fishery. Figure 1 summarizes the main characteristics of the SASOM framework.

The SASOM framework has been already used by the MINOUW partner UPV-EHU (jontly with the Universidade de Vigo) to provide advice on fisheries management in three dimensions[1]: reference points, biomass risk and floating targets.

[1]The research team has also used a methodological approach close to the SASOM framework to advise

2.1 SASOM and reference points

SASOM has been used to determine optimally the reference points of fisheries in which managers are considered as agents with particular economic objectives. This framework enables us not only to find steady state reference points but also the transitional trajectory from the current position of the fishery to the long-run values.

The SASOM framework has been developed to assess recovery and management plans based on the control of reference points. For instance, Da Rocha et al. (2010) uses it to include endogenous disinvestment decisions. This extension enables the Southern Hake recovery plan to be assessed considering other social constraints such as maintaining the fleet size. The long-term management plan for Northern Hake is also analyzed under this framework in Da Rocha and Gutiérrez (2011).

Since SASOM is based on age-structured populations, pulse fishing may arise as the optimal strategy in fisheries (Hannesson, 1975). However Da Rocha et al. (2012a) shows that second-best solutions based on stationary policies would minimally reduce the discounted profit of the fishery with respect to the global optimal solution, which would imply the cyclical closure of fisheries for some periods. Moreover, the optimality of pulse solutions is affected by the selectivity of the fishing technology. In particular, imperfect selectivity reduces the optimal pulse length (Da Rocha et al., 2013). Nevertheless, changes in the metrics used to define the reference point (e.g using the log(MSY)) and/or optimal harvesting control rules that stabilize employment and biomass around the optimal stationary values are two possible

on equity concerns. For instance Da Rocha and Sempere (2016) incorporate heterogeneous exploitation firms into a dynamic stochastic framework. Heterogeneity (caused by firm-specific shocks to production opportunities) leads firms to make entry/exit decisions endogenously. Therefore, in this setup the distribution of the firms in the fishery is endogenously determined. This study shows that a reform consisting of allowing ITQs to be traded as a separate asset has two effects. First, it leads to a greater concentration of production in the industry, as the most efficient firms will produce more. Second, it directly converts a non-tradable asset into a tradable one. This is equivalent to giving a lump sum transfer to all firms. The first effect implies a concentration in revenues, while the second implies a redistribution of wealth. We have not considered this research line as part of the SASOM framework because there is not need to consider an age-structure population for the resource.

ways of eliminating optimal pulse fishing solutions (Da Rocha et al., 2012c). Appropriate non constant discount rates that take into account the intertemporal scarcity of the resource may also be useful in smoothing optimal captures over time (Da Rocha et al., 2016).

The management of fisheries with reference points is quite challenging for the case of mixed fisheries in which fleets target different species together. Da Rocha et al. (2012b) provide an algorithm for characterizing reference points for this type of fishery as the steady-state solution of a dynamic optimal management problem under the SASOM framework. This study also shows that in the case of the European Northern Hake Stock this mixed management is superior to single-species management because it leads the fishery to higher discounted profits, with higher long-term spawning-stock biomass for all species involved.

The algorithm developed in Da Rocha et al. (2012b) has been used to study other mixed fisheries. For instance, Sampedro et al. (2016) uses it to simulate the management scenarios that enable the effectiveness of the collaboration between stakeholders and scientists to be assessed in the management of the fisheries in Iberian Atlantic waters. García et al. (2016) also uses the algorithm to calculate the multistock fishing mortality reference points for assessing the economic impact of the LO policy in the Spanish demersal fleet operating in the Iberian Sea region. They find that at fishery level, multistock reference points mitigate the decrease in net present value that results from the implementation of the LO.

The SASOM approach has several advantages for the calculation of optimal reference points. First, it is applicable to any fishery (single or mixed) whenever the biological parameters of the population are known or can be estimated. Second, the steady-sate values of the reference points are relatively easy to calculate in computational terms. The transition paths toward those long-run values are harder, but not impossible, to obtain. This means that SASOM enables the present value of the economic indicators for the fishery to be assessed, not only the value of the fishery in the long run. Thirdly, the solutions can be restricted to smooth solutions, thus avoiding those that imply the closure of the fishery. The main drawback of the approach is that there is no friendly software that allows its immediate calculation. In al cases it is necessary to adjust the algorithm to the main characteristics of the fishery.

The main task of our work into the MINOUW project has been to modify the software code underlying the algorithm proposed by Da Rocha et al. (2012b) in such a way as to analyze the effects of the LO requirement on the reference points and on the rest of the bioeconomic variables of fisheries.

2.2 SASOM and biomass risk

The SASOM has been also used to analyze the status of the biomass in fisheries where the biological models are not perfectly known. It is commonly assumed that in these uncertain contexts a robust approach implies the use of simple rules synthesized in harvest control rules (HCR).

Da Rocha and Mato-Amboage (2016) develop a theoretical model based on the SASOM methodology where the goal of the fishery manager is to establish an HCR that stabilizes the resource close to a target point. In this context, that reference point is exogenously given and the manager must avoid the risk of the stock dropping below a limit point. The article shows that the gains from adding age structure to the biological population depend on the accuracy with which they are estimated. As precision increases, the advantages of introducing age structure rather than using simple biomass based rules increase. Moreover, unlike simpler rules, including age structure in HCRs reduces stock volatility and generates a positive correlation between variances in biomass and yield.

This line of research is currently under development with a view to including robustness methods in the design of HCRs using the control theory proposed by (Hansen and Sargent, 2001, 2007). In particular, we design HCRs that explicitly include scientific uncertainty by assuming that managers understand that the perceived dynamics -a version of Hannesson

(1975) with stochastic recruitment- represent an approximation of the real (operating) model. Following the approach in Hansen and Sargent (2001, 2007), we characterize robust HCR by distorting the perceived dynamics with the worst case estimate of the operating model. Our preliminary findings are: i) Managers infer that the operating model (which is generating the perceived recruitment shock) is more persistent than the perceived model. As a result robust HCRs must be designed by assuming a more persistent process for recruitment than the one implied by the perceived model; and ii) Constant effort HCRs (which provide precautionary advice when recruitments are uncorrelated) are not robust under scientific uncertainty. A robust HCR has a steeper ratio between fishing mortality and biomass than a non-robust one. Rather than decreasings fishing mortality reference points for exploitation, the optimal robust response to scientific uncertainty is to increase biomass limits when knowledge of the stock status decreases.

This line of research opens up many policy issues related to the management of fisheries with HCRs. Although they originally lay outside the remit of the MINOUW project, discards are one of the issues that could be addressed when these results have been consolidated.

2.3 SASOM and floating targets

The new European Union multi-annual plans set floating band target regimes for controlling reference points for fishing mortalities, borrowing the ideas set out in Hilborn (2010) and recently expanded to multispecies analysis by Rindorf et al. (2016). This regime consists of setting lower and upper bounds on fishing mortalities levels around the maximum sustainable yield (MSY) target points; fishing mortality floats freely within the band and managers intervene occasionally when it crosses the limits of the band.

To assess this type of floating target, Da Rocha et al. (2017) extend the SASOM setting to build a multi-species multi-fleet state variable model where the individual behavior of each fleet endogenously determines the aggregate performance of the fishery. Fleets optimally select what level of fishing effort to apply by considering aggregate mortality as given. The introduction of a balance constraint enables total fishing mortality to be determined consistently with individual decisions. The optimal decisions taken by individual fleets generate endogenous catchability functions which enable total fishing mortalities be calculated as a combination of the contemporaneous abundances and stochastic processes affecting the fishery. These catchability functions are used to project future abundances as a dynamic system that depends on current abundance. Since total fishing mortality levels are not constant over time but are endogenously determined by the behavior of fleets, the abundance projections represent the *endogenous status quo*, which is the counter-factual scenario with which any policy scenario should be compared.

Da Rocha et al. (2017) use this methodology for endogenizing fishing mortality to assess the robustness of the floating band for fishing mortality levels in the Mediterranean demersal fishery in northern Spain. They find that the projections for the endogenous mortality float around the target F in quiet a narrow band, and state that this type of management regime is supported by the analysis.

This methodology for endogenizing fishing mortality is generic and can be applied to evaluate any fishing policy. In particular, the methodology seems very well suited to analyzing the effectiveness of the LO requirement since many fisheries are exploited in multi-species and multi-fleet scenarios.

3 Methods

The bioeconomic model that we use is based on the multi-species setting developed in Da Rocha et al. (2012b). Figure 2 represents the logic of the model. It comprises two main model-boxes. One of them represents the biological aspects of the fishery. This model-box has inputs such as recruitments and parameters representing selectivity, weights, maturity levels, and natural mortality levels. The other box represents the economic model, which usually means the managers decision problem based on economic elements such as the cost and demand functions and the discount rate when the decision is a long-term one. Other social restrictions such as the preservation of jobs in fleets can also be considered by the economic model. The managers decision problem is solved taking into account the biological model and as result optimal reference points for reference points (fishing mortality levels) are obtained. This optimal decision enables the stock to be evaluated in terms of economic indicators such as the net present value of the yield.

The biological part of the model is a standard multi-species age-structured model used for stock assessment. Assume that there are n species in the fishery. The stock of species $s = 1, ...S$ is broken down into $A(s)$ cohorts. i.e. in each period t there are $A(s) - 1$ initial old cohorts for species s and a new cohort is born.

Let $Z_{s,a,t}$ be the mortality rate that affects the population of species s at age a^{th} in period t^{th}. This mortality rate can be decomposed into fishing mortality, $F_{s,a,t}$, and natural mortality, $M^{s,a}$, so $Z_{s,a,t} = F_{s,a,t} + m_{s,a}$. While the fishing mortality rate may vary from one period and one age to another, natural mortality is constant over periods.

We assume that species s are fished simultaneously in relatively unselective fishing operations q_s and that the landing and discard selection patterns, $\bar{p}_{s,a}$ and $\bar{d}_{s,a}$, of each species are constant. Therefore, for each unit of effort, E_t , the fishing mortality for each age and species is given as $F_{s,a,t} = (\bar{p}_{s,a} + \bar{d}_{s,a})q_j E_t$. If a multiproduct technology is considered, such that for a given level of effort each species is captured in fixed proportions (Leontief, 1944) at each age, then the fishing mortality multiplier can be defined with no loss of generality as effort $F_t = E_t$, and the original landing and discard selection pattern, $p_{s,a} = \bar{p}_{s,a}q_j$ and $d_{s,a} = \bar{d}_{s,a}q_j$ can be rescaled to express the fishing mortality for each age and species as $F_{s,a,t} = (\bar{p}_{s,a} + \bar{d}_{s,a})q_j E_t = (p_{s,a} + d_{s,a})F_t$.

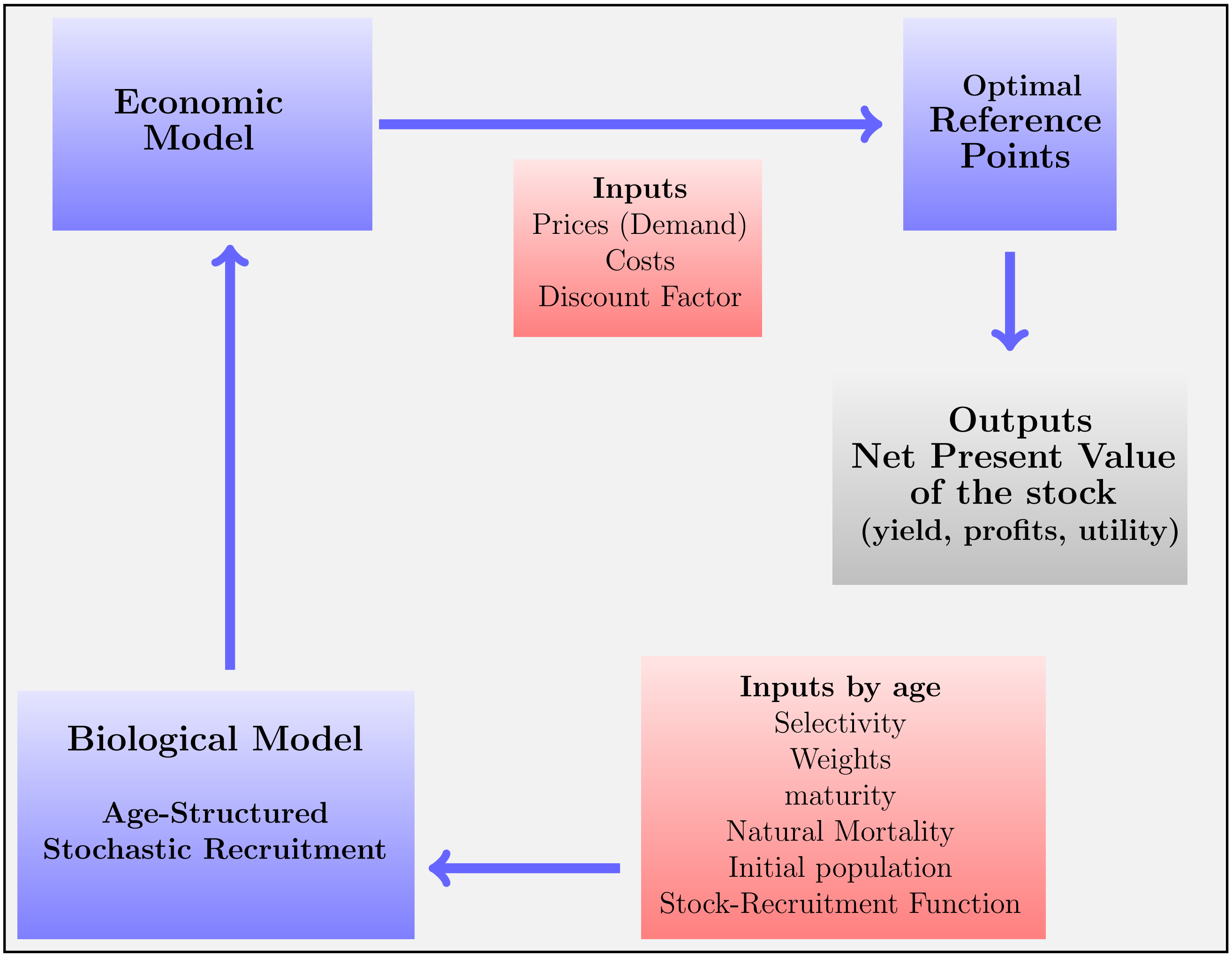

Figure 2: Optimal Reference Points with SASOM framework

The population of species s decreases at an exponential rate in accordance with the mortality rate $Z_{s,a,t}$. Formally, $N_{s,a+1,t+1} = e^{-Z_{s,a,t}} N_{s,a,t}$, where $N_{s,a,t}$ represents the abundance of species s for age a at period t. Notice that by backward substitution $N_{s,a,t}$ can be expressed as a function of recruitment (see Da Rocha et al. (2012b) for more details)

The size of a new cohort (recruitment) of species s is given by the the Shepherd (1982) stock–recruitment (S–R) relationship,

$$N_{s,1,t+1} = \frac{\alpha_s SSB_{s,t}}{1 + (SSB_{s,t}/K_s)^{b_s}},$$

where $SSB_{s,t} = \sum_{a=1}^{A(s)} \omega_{s,a}\mu_{s,a}N_{s,a,t}$ is the spawning-stock biomass, which is a function of the spawning-stock weight at age $\omega_{s,a}$ and the maturity fraction $\mu_{s,a}$. α_s, b_s and K_s are parameters with specific values for species s.

Finally, the fishing yield for sale for each species and age is given by Baranov's equation (Baranov, 1918). Formally, the yield of species s is calculated as

$$Y_{s,t} = \sum_{a=1}^{A(s)} \frac{p_{s,a}F_t}{Z_{s,a,t}} \left(1 - e^{-Z_{s,a,t}}\right) N_{s,a,t}.$$

Notice that discard patterns appear as an element of the mortality rate per age and species, $Z_{s,a,t}$, but this does not influence the fraction of the mortality that determines the yield for sale $p_{s,a}F_t/Z_{s,a,t}$.

The economic model is represented as a dynamic management problem in which the net present value of an economic indicator such as yield, value of yield or profits is maximized. For instance, when the economic indicator is the fishery yield the function to be maximized can be expressed as

$$Y_t = \sum_{t=0}^{\infty} \beta^t \sum_{s=1}^{S} \sum_{a=1}^{A(s)} \frac{p_{s,a}F_t}{Z_{s,a,t}} \left(1 - e^{-Z_{s,a,t}}\right) N_{s,a,t},$$

where $0 \leq \beta \leq 1$ is the discount factor which represents the willingness of the manager (or society) to trade-off the value of fishing today against the benefits of increased yields in the future, measured by higher biomass and recruitment. The optimal reference point emerges as the steady-state solution of this dynamic problem. In Da Rocha et al. (2012b) it is proved that this steady-state solution is just a generalization of F_{msy}.

4 The effects of discards on small-scale fisheries in Galicia (North West Spain)

The new developments of the SASOM framework presented in Section 3 have been applied to investigate the economic and biological impact of the EU LO in small-scale fisheries (SSF) in Galicia. The main results of this section are published in Villasante et al. (2016b)

4.1 Small-scale Galician fisheries

Galicia, in north-wester of Spain, is the main fishing region of the country. Fishing contributes strongly to the regional gross domestic product, and the region accounts for over than 40% of the country's fleet, 50% of catches reported by Spanish vessels fishing in European Union waters and more than 60% of total employment in fishery-related sectors (Villasante, 2016).

The Galician SSF is comprised mainly of small vessels, averaging 6 m in length and mostly under 12 m, operating daily from Monday to Friday, with an average tonnage of 2.2 GT. It uses a wide variety of passive gears, known as "artes menores", such as traps (*nasas*) for octopus or crabs, hooks and lines (*palangrillos*), and nets such as gill and trammel (*beta, trasmallo, miño*) and small seines (*xeito*), exploiting a diverse range of species, most of which are not subject to TAC regimes. 10 main commercial species are harvested in the Galician SSF: common octopus, velvet crab and common prawn, mostly captured in traps (*"nasas"*); European sole, European seabass and spider crab, all of which are mostly harvested by trammel nets (*"miños"*); and European hake, horse mackerel, pouting and red mullet mainly captured with gillnets.

The discard rate for SSF ranges between 5% and 18% depending on the type of commercial species harvested though it can be significantly higher for some sedentary resources, e.g. 74%

for goose barnacle and 49% for razor clam, mainly due to minimum landing size restrictions (Villasante et al., 2016a).

4.2 Impact of LO on the Galician small-scale gillnet fishery ("Betas")

The economic impact of the LO on the Galician multispecies SSF is felt mainly by the gillnet fishing fleet (called "Betas"), which harvests European hake as its main species (*Merluccius merluccius*) mixed with Atlantic horse mackerel (*Trachurus trachurus*), mackerel (*Scomber scombrus*), pouting (*Lophius piscatorius*) and red mullet (*Mullus surmuletus*) as secondary species. Figure 3 shows the spatial distribution of the Betas fishing fleet and its main characteristics along the Galician cost.

The bioeconomic model outlined in Section 3 is used to estimate the maximum economic losses arising from the LO in two phases. In the first phase the model is used to compute the maximum discounted net present value of the sum of economic benefits arising from the multispecies small-scale gillnet fishery. The algorithm used is described in (Da Rocha et al., 2012a,b, 2013) while a detailed description of the model used appears in the supplementary materials in (Da Rocha et al., 2016).

In the second phase the number of fishing days and the expected catches of the fishery are estimated for 2016-2026 under two different scenarios: one without the LO and the other with it. The economic impact of the LO in both scenarios is then measured as follows.

In both scenarios the sequences of hake TACs are assumed to remain constant, i.e. we assume that the percentage of quotas allocated (not in absolute terms) is the same due to the application of the stability principle, which means that quotas will respond to changes in the stock size over time. These sequences of TAC are then used to estimate the expected fishing days and catches by the fleet without the LO. Both scenarios assume full compliance

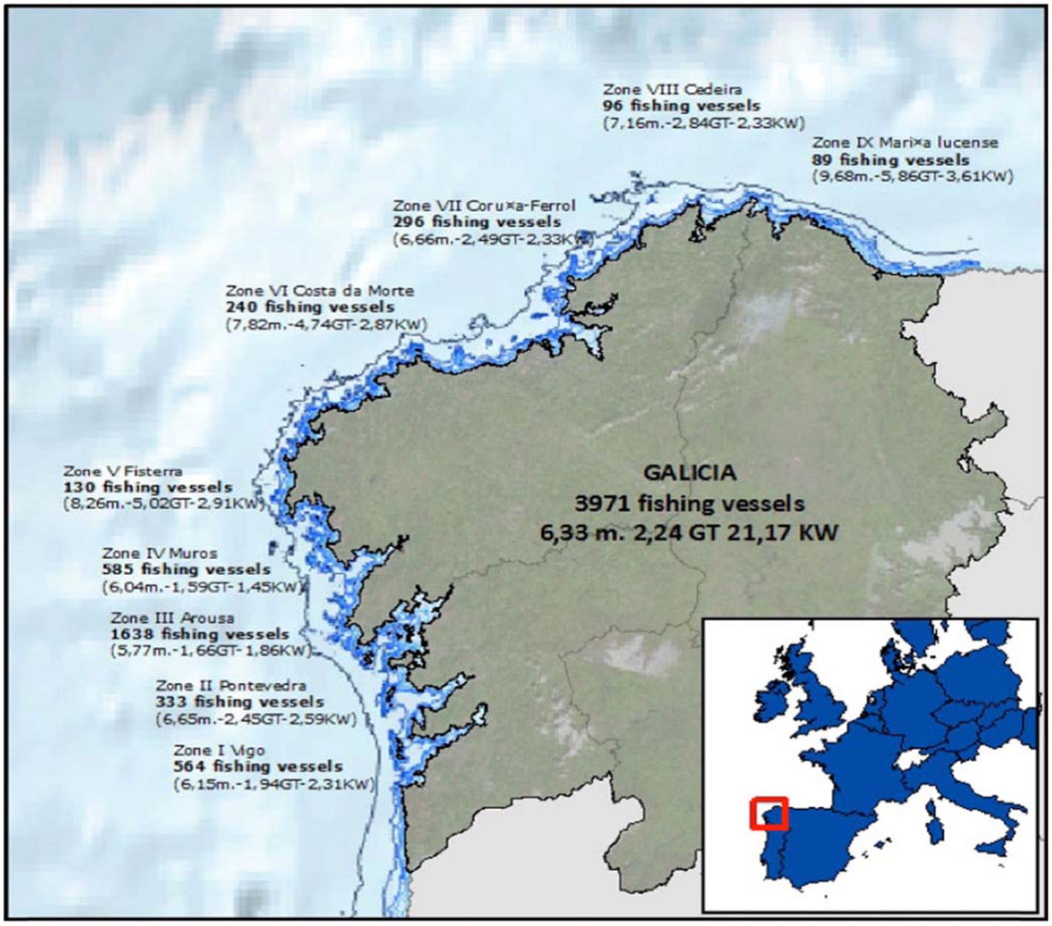

Figure 3: Spatial distribution of the Galician small-scale ("Betas") fishing fleet (2015). Source: Villasante et al. (2016b)

with TAC which is in accordance with the historical use of quotas by this fleet (ICES, 2007, 2011).

Introducing of the LO modifies fishing days and catches for the species harvested in the fishery. Therefore, although both scenarios are computed by assuming the same TACs, the introduction of LOs affects the trend in the SSB of the species and the profits of the fishery. The trend in the SSB is calculated by the ICES Working Group for the Bay of Biscay and Iberian Waters Ecoregion (WGBIE) by using a length-based model with GADGET (ICES, 2014a,b).

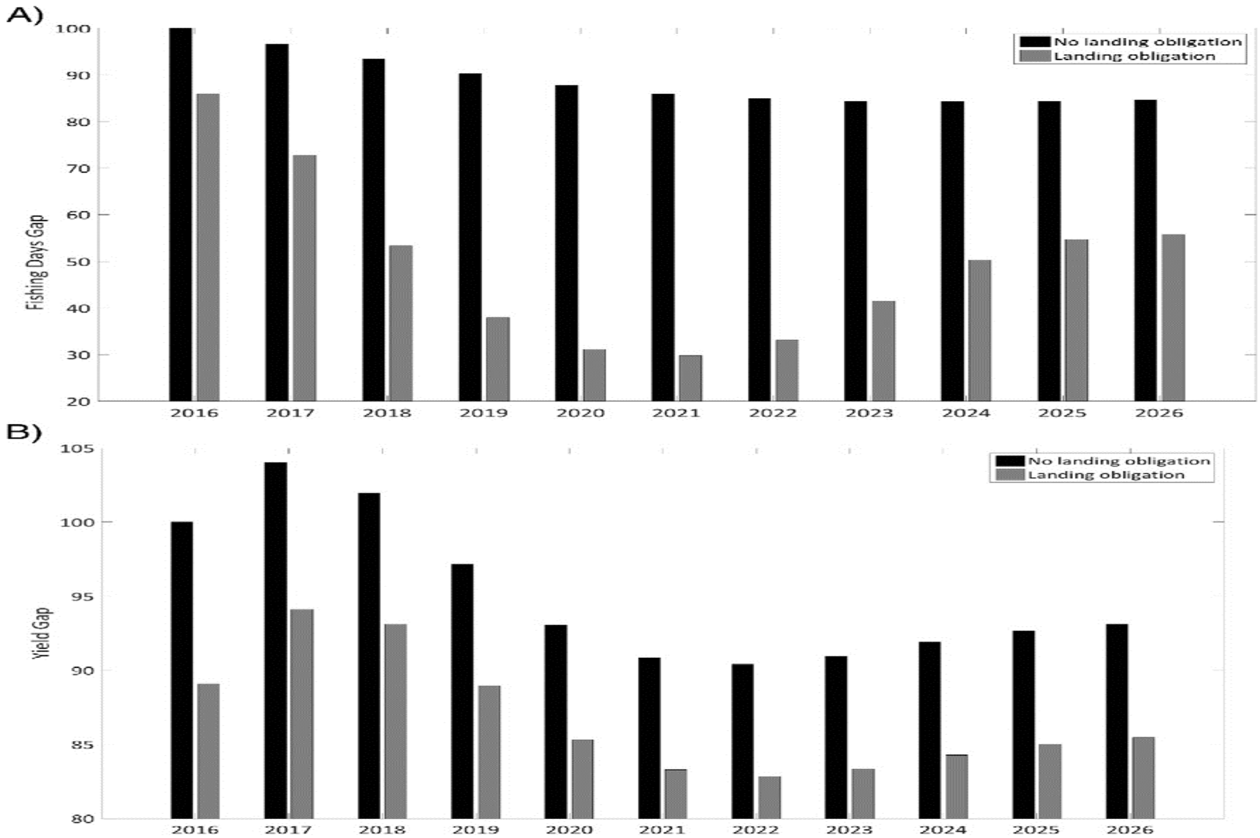

Figure 4: Fishing days A) and yields (catches) B) estimated for the Galician gillnet fleet "with" and "without" LO scenarios. Source: Villasante et al. (2016b)

Analytical results are outlined in Figure 4. It can be seen that the introduction of the LO policy in 2016 would lead to short and long-term losses in fishing days and yields for the small-scale hake fishery. The expected fishing days for the hake fishery under the LO would be reduced most (50%) in the first five years following the implementation of the policy.

The economic impacts of the LO would be observed in the first year of adoption of the policy, with a loss of fishing time equivalent to 30% of fishing days per annum compared to the status quo. The reduction in fishing days would rise to 70% in the fifth year of implementation of the LO, and a difference of more than 50% of the 2016 value would be seen until the year 2026.

The LO would also affect the capacity of the fishing fleet to harvest the allocated TACs, due to the loss of fishing time at sea. The future yield (catches) under the LO would be only 50% of the catches expected without it, regardless of the total volume of quotas allocated to the fleet.

5 Policies for Mitigation Unwanted Catches in the Southern Iberian Hake Stock

The model presented in Section 3 has also been applied to the Southern Iberian Hake Stock (SIHS) to analyze the bioeconomic effects of two types of policy intended to reduce unwanted catches. On the one hand we analyze how the bioeconomic variables of the fishery may vary when selectivity is improved. On the other hand, we study the effects of imposing a ban on the sale of juveniles. Both policies are analyzed first under the *ceteris paribus* criteria. The results of a combination of both policies are then studied.

5.1 The Southern Iberian Hake Stock

The Northern limit of the SIHS is the Spanish–French frontier and the Southern limit is the Straits of Gibraltar (see Figure 5). The SIHS is a fishery managed with the advice of the International Council for the Exploitation ofthe Sea (ICES), and it includes all fisheries in subareas VIIIc and IXa. Hake (*Merluccius merluccius*) is caught in a mixed fishery with other demersal (e.g. megrim, monkfish and nephrops) and pelagic (e.g. blue whiting, sardine and horse mackerel) species by the Spanish and Portuguese fleets (trawls, gillnetters, longliners and artisanalfleets). Spain accounts for most of the landings. Total landings and discards were 11786t and 2292t, respectively, in 2015. Total catches in 2015 were up by 4% on 2014. The fishery is managed by TAC, effort control and technical measures. On the basis of the

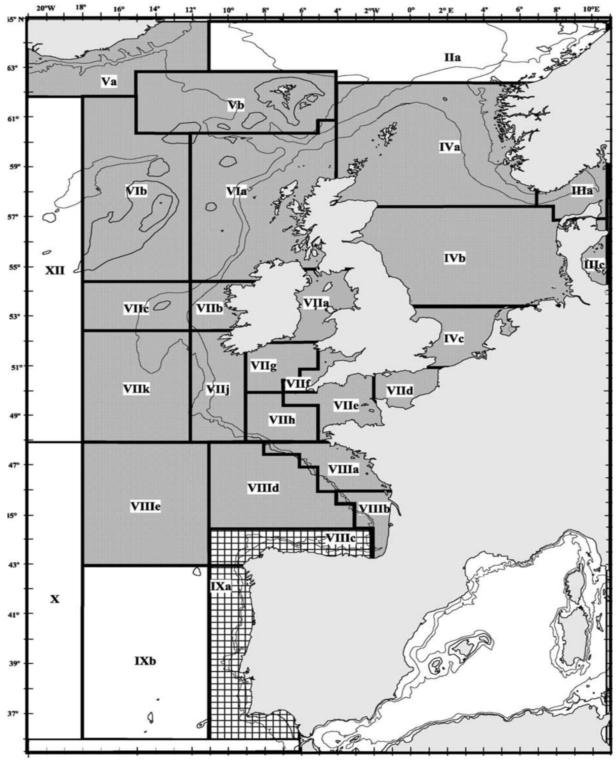

Figure 5: The Southern Iberian Hake Stock includes ICES subareas VIIIc and IXa (in small square design).

transition to the MSY approach, the ICES advised that landings for SIHS should be no more than 6078 t in 2016 and 8049 t in 2017. Under the EU LO in 2016, this means that landings should be the same as catches. Nevertheless, the agreed TAC for SIHS was 13826t in 2015 and 10674t in 2016.

A recovery plan for the SIHS was implemented in 2006 (EC, 2005). This plan aims to rebuild the stock to within safe biological limits by decreasing fishing mortality by a maximum of 10% a year with a TAC constrain of 15%. An SSB target (35000 t) is not considered suitable under the new assessment model. This regulation includes effort management by limiting days at sea, which are updated every year. The effort from fishing trips which catch less than 8% of hake is excluded from the regulation (ICES, 2016).

The parameterization used for the age-structured population is the same as that used in the ICES Working Group on the Assessment of Southern Shelf Stocks of Hake, Monk, and Megrim (ICES, 2015), as presented in Tables A.1 and A.2 in the Appendix. The assessment of the SIHS depends on whether or not discards are included in the analysis (Fernández et al., 2010), so discard patterns are taken into account for hake. Three secondary species are considered for analyzing the mixed fisheries: megrim *lepidorhombus whiffiagonis* (MGW), megrim *lepidorhombus boscii* (MGB) and monkfish *lophius piscatorius* (MON).

Hake recruitments were obtained using the Shepherd (1982) stock–recruitment relationship which was estimated from recruitment and SSB data for 1978–2006. This fit gives $\alpha = 14.774$, $K = 12134.22$ and $b = 1.604$. For the secondary species, the expected recruitment is considered as constant over time. In particular recruitment (in thousands) is 2504 for MGW, 24016 for MGB, and 855 for MON.

Reference points for single and mixed fisheries are computed as the steady-state solution of a dynamic optimal management problem in which the yield is maximized. The optimization problem takes into account that: (i) species are caught simultaneously in unselective fishing operations; and (ii) there is intertemporal discounting equal of 0.95 (i.e. an interest rate close to 5 per cent). See Da Rocha et al. (2012b) for a similar approach with the Northern Iberian Stock of Hake.

5.2 Improving fishing selectivity

The LO requirement imposed by the EU Common Fisheries Policy seeks to promote changes in fishing technologies so as to improve selectivity and effectiveness in harvesting target species (Catchpole et al., 2006; Kraak et al., 2013; Condie et al., 2013).

In this section we show how the bioeconomic variables of the SIHS would change if the selectivity were improved, *ceteris paribus*. To that end an age-species specific parameter, $0 < \gamma_{s,a} < 1$, is introduced into the mortality rate and into the Baranov yield equation.

Formally, the yield of species s is calculated as

$$Y_{s,t} = \sum_{a=1}^{A(s)} \frac{(1-\gamma_{s,a})p_{s,a}F_t}{Z_{s,a,t}} \left(1 - e^{-Z_{s,a,t}}\right) N_{s,a,t},$$

where

$$Z_{s,a,t} = m_{s,a} + (1-\gamma_{s,a})(d_{s,a} + p_{s,a})F_t,$$

and $\gamma_{s,a}$ represents the reduction, in percentage terms, in the selectivity parameter. If $\gamma_{s,a} = 0$ represents the benchmark situation, then applying $\gamma_{s,a} = 0.30$ means that with the same fishing effort the fishing mortality for species s and age a will be 70% of the benchmark fishing mortality, i.e. a 30% lower. In this sense, higher $\gamma' s$ in young ages can be understood as indicative of more selective technologies.

Table 1 illustrates the results of improving selectivity by using $\gamma_{s,a} = 0.90$. The mixed nature of the fishery means that a single reference point needs to be calculated for the management of the resource. The first row in Table 1 shows the optimal fishing mortality under various scenarios. In the status quo the optimal reference point is $F = 0.70$. When the selectivity parameter is reduced 90% for age 0, the optimal reference point increases to $F = 0.73$; when this reduction is extended to ages 0 and 1 the optimal reference points increases to $F = 1.11$, and so on.

The results show that the highest yield for the main species (hake) occurs when selectivity is improved for ages 0, 1 and 2 when the hake yield increases to 5.77 times more than in the status quo scenario. However the highest SSB is found when the improvement in selectivity also includes also age 3. In this case the yield of hake is 5.42 times greater than in the status quo.

Overall, reducing the selectivity parameters by 90% for the two lower ages leads to the greatest improvements in terms of hake yield and total yield compared with the status quo scenario. Moreover hake discards are down by a significant figure of more than 20 percentage

Table 1: Improving selectivity by age ranges in the SIHS

Reduction in Selectivity Parameters: $\gamma_{s,a} = 0.90$							
	Status quo	Age Range Policy Application					
		0	0-1	0-2	0-3	0-4	0-5
F	0.70	0.73	1.11	1.66	1.88	2.12	2.61
	Yield						
HKE	2910	4309	13707	**16783**	15760	13824	9779
MGW	147	147	150	151	151	150	150
MGB	1353	1354	1359	1349	1345	1341	1333
MON	1066	1055	888	624	532	446	302
	SSB						
HKE	2686	3880	13339	23629	**26928**	24087	12332
MGW	553	546	471	380	348	317	261
MGB	4867	4814	4202	3498	3263	3039	2634
MON	2673	2551	1363	529	364	246	106
	Change wrt status quo						
Yield HKE	1.00	1.48	4.71	**5.77**	5.42	4.75	3.36
Total Yield	1.00	1.20	2.50	**2.88**	2.71	2.41	1.80
Hake discards / Hake yield	0.31	0.31	0.14	**0.10**	0.10	0.12	0.21

points on the status quo.

This analysis has been repeated for other reductions of the selectivity parameters. Table 2 shows some of the results for the sensitivity analysis. The main conclusion is that only when selectivity changes significantly are the results positive in terms of increasing yield and reducing discards. These results are summarized in Figure 6, where the changes in hake yield with respect to the status quo are plotted for the reductions in selectivity parameters and the age ranges over which those reductions are applied.

To finish this section, it is worth mentioning that the results presented show the effects of the improving selectivity only in the yield but not in the profits of the fishery. To calculate the profits we would need to know the cost of modifying and adapting the fishing technology. Most experts consider that in many cases, including small-scale fisheries, selectivity is difficult

Table 2: Improving selectivity. Sensitivity analysis.

Change wrt status quo. Hake yield							
Reduction Parameter	Status quo	Age Range Policy Application					
		0	0-1	0-2	0-3	0-4	0-5
0.90	1.00	1.48	4.71	**5.77**	5.42	4.75	3.36
0.70	1.00	0.99	0.94	0.89	0.88	0.91	0.95
0.50	1.00	0.99	0.91	0.85	0.86	0.89	0.91
0.10	1.00	0.98	0.80	0.69	0.71	0.75	0.79
Change wrt status quo. Total hake							
Reduction Parameter	Status quo	Age Range Policy Application					
		0	0-1	0-2	0-3	0-4	0-5
0.90	1.00	1.20	2.50	**2.88**	2.71	2.41	1.80
0.70	1.00	0.99	0.96	0.93	0.93	0.95	0.98
0.50	1.00	0.99	0.94	0.90	0.91	0.93	0.95
0.10	1.00	0.99	0.89	0.83	0.84	0.86	0.89
Hake discards / Hake yield							
Reduction Parameter	Status quo	Age Range Application					
		0	0-1	0-2	0-3	0-4	0-5
0.90	0.31	0.31	0.14	**0.10**	0.10	0.12	0.21
0.70	0.20	0.22	0.23	0.23	0.23	0.22	0.21
0.50	0.23	0.23	0.25	0.25	0.25	0.24	0.24
0.10	0.28	0.28	0.30	0.30	0.30	0.30	0.30

to improve without high costs (Villasante et al., 2016c). In any case, the increase in yield achieved by improving selectivity can be seen as an upper bound for the cost of incorporating the new technology into standard harvesting operations.

5.3 Policy for a ban on the sale of juveniles

Unwanted catches can be reduced through market control policies that ban their sale (Tolotti et al., 2015). However this may foster the black market for juvenile fish (Catchpole et al., 2017).

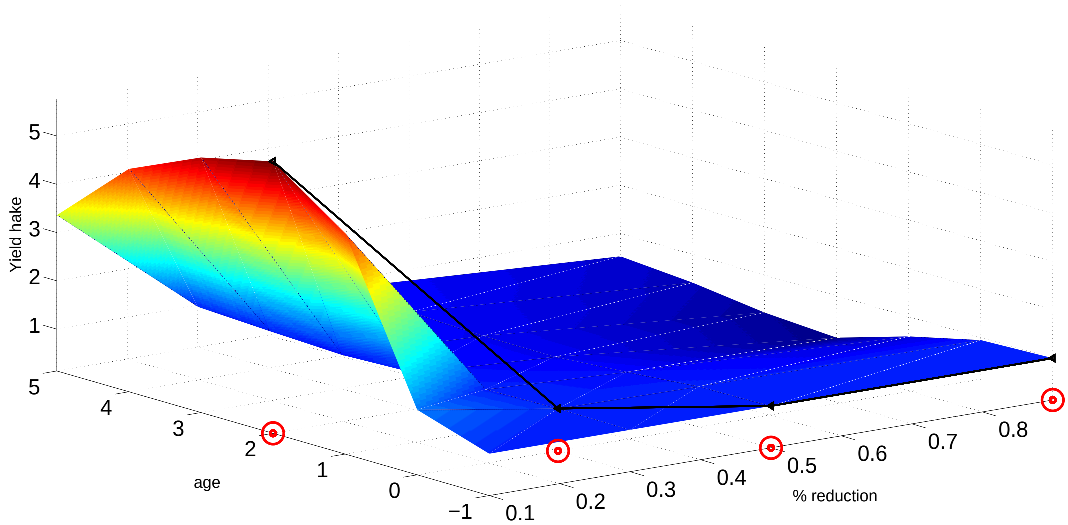

Figure 6: Impact on yield of decreasing discards. Labels show the optimal policy. Label -1 is the status quo (zero reduction).

In this section we show how the bioeconomic variables of the SIHS would change if the sale of some age ranges were banned, *ceteris paribus*. To that end we take the ban into account in the Baranow yield equation. For instance, if the ban on sales is imposed for ages $a = 1, 2, ..j$, the yield of species a is calculated as

$$Y_{s,t} = \sum_{a=j+1}^{A(s)} \frac{(1-\gamma_{s,a})p_{s,a}F_t}{Z_{s,a,t}} \left(1 - e^{-Z_{s,a,t}}\right) N_{s,a,t}.$$

Table 3 illustrates the results of banning the sale of juveniles in the market under various scenarios. The first row shows the optimal reference point applied in this mixed fishery under the policy scenarios. In the status quo, the optimal reference point is $F = 0.70$. When the sale of juveniles of age 0 is banned, the optimal reference point stays the same because in the status quo those fishes (eggs) are not sold; when this ban is extended to ages 1 and 2 the optimal reference point is $F = 0.64$ and so on. The more ages that are covered by the ban in the market the lower the fishing mortality applied to the fishery is.

Table 3: Effects on the SIHS of a ban on sales of juveniles

	Status quo	Age Range Policy Application					
		0	0-1	0-2	0-3	0-4	0-5
F	0.70	0.70	0.64	0.48	0.38	0.32	0.28
	Yield						
HKE	2910	2912	**3513**	2137	840	294	97
MGW	147	146	129	102	71	30	14
MGB	1353	1353	1348	1256	987	632	300
MON	1066	1066	1072	1026	865	652	457
	SSB						
HKE	2686	2689	4437	8411	10655	12112	**13202**
MGW	553	553	565	600	621	635	645
MGB	4867	4867	4971	5268	5450	5570	5659
MON	2673	2674	2920	3674	4174	4515	4775
	Change wrt status quo						
Yield HKE	1.00	1.00	**1.21**	0.73	0.29	0.10	0.03
Total Yield	1.00	1.00	**1.08**	0.85	0.56	0.34	0.19
Hake discards / Hake yield	0.31	0.31	**0.38**	0.88	2.39	6.93	21.04

The results show that the highest yield for the main species (hake) was 21% higher than in the status quo scenario, when the ban on sales affects ages 0 and 1. As expected, the highest SSB is found when the ban affects all ages because this is equivalent to the recovery of the fishery after a closure. Discards also increase when the ban affects older fishes. Notice that with this policy discards do not entail any cost for fishermen and the fishing technology does not change.

Overall, banning the sale of juveniles of ages 0 and 1 leads to the greatest improvements in hake yield (21% up) and total yield (8% up) compared with the status quo scenario. However, discards of hake increase 7 percentage points with respect the status quo.

This analysis is supplemented by the possibility of improving selectivity simultaneously. Table 4 illustrates the results of both scenarios for the hake yield, the total yield and the discards/yield ratio for hake with respect to the status quo. The figures shown in the first row of the three boxes correspond to the results of applying a pure sales ban policy, so these are the same as the results that appear in the lower box of Table 3. The figures in the second row of the three boxes correspond to the results of applying a mixed policy in which the ban on sales is applied simultaneously with an improvement in selectivity that represents a reduction of the 90% in fishing mortality. Comparing the two policies, we following can be observed:

i) The mixed policy leads to significantly higher increases than the pure sales ban policy in the hake yield and total yield than in the status quo for all age range scenarios.

ii) The mixed policy leads to a smaller discard/yield ratio for hake than in the status quo scenario, and the extent of the reduction increases as the age range is extended. However, with the pure ban this result is reversed; the discard/yield ratio increases on the status quo figure by an amount that increases as the age range is extended.

iii) Applying the pure sales ban policy for ages 0 and 1 leads to the highest hake yield and total yield and to the smalest increase in the hake discards/yield ratio. However, when this policy is applied jointly with an improvement in selectivity, the best results appear when it is applied over the broadest age range.

6 Summary and Conclusions

Basic bioeconomic models have been developed to incorporate new concerns in fisheries management. Unwanted catches are one of the issues have recently come to be regarded as key for commercial fisheries. The LO included in the latest reform of the EU Common Fisheries Policy emphasizes the significance of this matter.

The main aim of the UPV-EHU (jointly with the Universidade de Vigo) as a partner in the MINOUW project has been to update the software underlying the bioeconomic framework

Table 4: Pure vs mixed sales ban policies

Reduction Selectivity Parameter γ	Status quo	Age Range Policy Application					
		0	0-1	0-2	0-3	0-4	0-5
	Change wrt status quo yield						
0.00 (Pure policy)	1.00	1.00	**1.21**	0.73	0.29	0.10	0.03
0.90 (Mixed policy)	1.00	1.00	2.47	7.40	9.74	10.40	**10.55**
	Change wrt status quo SSB						
0.00 (Pure policy)	1.00	1.00	**1.08**	0.85	0.56	0.34	0.19
0.90 (Mixed policy)	1.00	1.00	1.11	1.48	1.69	1.81	**1.88**
	Hake discards / Hake yield						
0.00 (Pure policy)	0.31	0.31	**0.38**	0.88	2.39	6.93	21.04
0.90 (Mixed policy)	21.04	21.01	8.53	2.84	2.16	2.02	**1.99**

named SASOM (Stochastic Age-Structured Optimization Model) with a view to including discard analysis in the determination of reference points. This main goal has been achieved by extending the multi-species model presented in Da Rocha et al. (2012b).

These new developments enable policies that seek to mitigate unwanted catches to be assessed. In particular, they are applied here to two case studies: Small-scale fisheries in Galician and the Southern Iberian Hake Stock.

On the one hand, the new bioeconomic developments have been used to estimate the economic impact of the LO on small-scale fisheries in Galicia. The main finding of this study is that future catches under the LO may be only 50% of the amount expected in the absence of the LO. These results are published in Villasante et al. (2016b).

On the other hand, advances in the SASOM framework have been applied to the Southern Iberian Hake Stock to study the bioeconomic effects of two types of policy intended to reduce unwanted catches. On the one hand we study how the bioeconomic variables of the fishery may vary if selectivity is improved. On the other hand we study the effects of imposing a ban on the sale of juveniles. Both policies are analyzed first under *ceteris paribus* criteria, then the results of a combination of the two policies are studied.

With respect to the first policy, our results show that reducing the selectivity parameters a 90% for the two early ages leads to the highest improvements in terms of yield of hake and total yield comparing with the status quo scenario. Moreover discards of hake reduces significantly with more than 20 percentage points with respect the status quo.

With respect to the second policy, our findings indicate that banning the sale of juveniles of ages 0 and 1 leads to the greatest improvements in terms of hake yield (21% up) and total yield (8% up) compared with the status quo scenario. Moreover, discards of hake are up by 7 percentage points on the status quo.

The results for the sales ban policy are reversed when a mixed policy is considered in which the ban is applied simultaneously with an improvement in selectivity. This mixed policy leads to reductions in the discard/yield ratio for hake with respect to the status quo which increases as the age range is extended. However, with the pure ban this result is reversed; the discard/yield ratio increases on the status quo figure by an amount that increases are the age range is extended.

All the research described in this report have been is presented in the following **publications** produced with the support of MINOUW:

[**1**] Da Rocha, J.M., García-Cutrin, J., Gutiérrez, M.J., Jardim, E., (2017, forthcoming). "Endogenous Fishing Mortalities: a State-Space Bioeconomic Model". **ICES Journal of Marine Science.** DOI 10.1093/icesjms/fsx067.

[**2**] Da Rocha, J.M., García-Cutrin, J., Gutiérrez, M.J., Touza, J., (2016). "Reconciling yield stability with international fisheries agencies precautionary preferences: the role of non constant discount factors in age structured models". **Fisheries Research** 173, 282-293. DOI http://dx.doi.org/10.1016/j.fishres.2015.08.024.

[**3**] Da Rocha, J.M., García-Cutrin, J., Prellezo, R., Sempere, J., (2017). "The social cost of fishery subsidy reforms". **Marine Policy** 83, 236-242.

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[**4**] Da Rocha, J.M., Prellezo, R., Sempere, J., Taboada Antelo, L., (2017). "A dynamic economic equilibrium model for the economic assessment of the fishery stock-rebuilding policies". **Marine Policy** 81, 185-195. http://dx.doi.org/10.1016/j.marpol.2017.03.029

[**5**] Da Rocha, J.M., Sempere, J., (2016, forthcoming). "ITQs, Firm Dynamics andWealth Distribution: Does Full Tradability Increase Inequality?". **Environmental Resource Economics.** http://dx.doi.org/10.1007/s10640-016-0017-3

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[**7**] Villasante, S., Pita, C., Pierce, G.J:, Pazos Guimeráns, C., Garcia Rodrigues, J., Antelo, M., Da Rocha, J.M., García Cutrín, J., Hastie, L.C., Sumaila, U.R., Coll, M., (2017). "To land or not to land: How do stakeholders perceive the zero discard policy in European small-scale fisheries?". **Marine Policy** 71, 166-174.

This research has also been presented, at various stages, at the following **congress and conferences:**

[**1**] 23rd Annual Conference of the European Association of Environmental and Resource Economists (EAERE), Athens (Greece), 28th June – 1st July, 2017.

[**2**] 5th Workshop on Age-structured models in Natural Resource Economics, Lütjenburg, Germany, 17th - 19th May, 2017.

[**3**] 41 Simposio de la Asociación Española de Economía (SAEe), Bilbao (Spain), 15th - 17th December 2016.

[**4**] VI Congress of the Spanish-Portuguese Association of Environmental and Natural Resource Economics (AERNA), Aveiro (Portugal), 5th - 7th September 2016.

[**5**] Joint Research Center (JRC). Seminar. Ispra (Italy), 17th-18th February 2016.

[**6**] 21st Annual Conference of the European Association of Environmental and Resource Economists (EAERE), Helsinki (Finland), 24th - 27th June 2015.

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A Appendix

Table A.1: SIHS. Parameters of the age-structure model for hake

Ages	Abundances	Mortalities			Maturity	Weight (Kg)
	(M)	Natural	Landings	Discards		
0	92516	0.4	0.00	0.06	0.00	0.01
1	61742	0.4	0.26	0.23	0.08	0.12
2	37602	0.4	0.83	0.10	0.70	0.46
3	10508	0.4	0.92	0.04	0.98	1.10
4	3150	0.4	0.92	0.01	0.99	2.01
5	486	0.4	0.92	0.01	1.00	3.09
6	137	0.4	0.92	0.00	1.00	4.26
7	28	0.4	0.92	0.00	1.00	5.46
8	6	0.4	0.92	0.00	1.00	6.62
9	2	0.4	0.92	0.00	1.00	7.71
10	1	0.4	0.92	0.00	1.00	10.00

Source: ICES Working Group on the Assessment of Southern Shelf Stocks of Hake, Monk, and Megrim (ICES, 2015).

Table A.2: SIHS. Parameters of the age-structure model for secondary species

Mergrim *lepidorhombus whiffiagonis* **(MGW)**					
Ages	Abundances (2011, M)	Mortalities		Maturity	Weight (Kg)
		Natural	Fishing		
0	2504	0.2	0.012	0.34	0.062
1	4357	0.2	0.096	0.90	0.092
2	780	0.2	0.134	1.00	0.123
3	704	0.2	0.177	1.00	0.158
4	751	0.2	0.283	1.00	0.193
5	362	0.2	0.294	1.00	0.241
6	275	0.2	0.294	1.00	0.396
Mergrim *lepidorhombus boscii* **(MGB)**					
Ages	Abundances (2011, M)	Mortalities		Maturity	Weight (Kg)
		Natural	Fishing		
0	24016	0.2	0.000	0.00	0.003
1	17855	0.2	0.001	0.55	0.037
2	23296	0.2	0.079	0.86	0.070
3	10209	0.2	0.275	0.97	0.088
4	6879	0.2	0.513	0.99	0.110
5	3734	0.2	0.535	1.00	0.145
6	2766	0.2	0.401	1.00	0.186
7	1417	0.2	0.401	1.00	0.265
Monkfisk *lophius piscatorius* **(MON)**					
Ages	Abundances (2011, M)	Mortalities		Maturity	Weight (Kg)
		Natural	Fishing		
0	855	0.2	0.013	0.01	0.001
1	920	0.2	0.120	0.09	0.681
2	366	0.2	0.202	0.28	1.922
3	162	0.2	0.288	0.53	3.843
4	161	0.2	0.334	0.73	6.398
5	212	0.2	0.319	0.85	9.483
6	105	0.2	0.272	0.91	12.956
7	76	0.2	0.219	0.94	16.748
8	12	0.2	0.173	0.96	20.834
9	29	0.2	0.136	0.98	25.132
10	37	0.2	0.108	0.98	29.497
11	5	0.2	0.088	0.99	33.788
12	4	0.2	0.072	0.99	37.914
13	15	0.2	0.030	1.00	52.467

Source: ICES Working Group on the Assessment of Southern Shelf Stocks of Hake, Monk, and Megrim (ICES, 2015).

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