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Estimating the economic loss of recent North Atlantic fisheries management

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Abstract:

It is accepted that world's fisheries are not generally exploited at their biological or their economic optimum. Most fisheries assessments focus on the biological capacity of fish stocks to respond to harvesting and few have attempted to estimate the economic efficiency at which ecosystems are exploited. The latter is important as fisheries contribute considerably to the economic development of many coastal communities. Here we estimate the overall potential economic rent for the fishing industry in the North Atlantic to be B€ 12.85, compared to current estimated profits of B€ 0.63. The difference between the potential and the net profits obtained from North Atlantic fisheries is therefore B€ 12.22. In order to increase the profits of North Atlantic fisheries to a maximum, total fish biomass would have to be rebuilt to 108 Mt (2.4 times more than present) by reducing current total fishing effort by 53%. Stochastic simulations were undertaken to estimate the uncertainty associated with the aggregate bioeconomic model that we use and we estimate the economic loss NA fisheries in a range of 2.5 and 32 billion of euro. We provide economic justification for maintaining or restoring fish stocks to above their MSY biomass levels. Our conclusions are consistent with similar global scale studies.

Highlights

► The economic loss of North Atlantic fisheries in 2010 was B€ 12.2. ► Fish stocks would have to recover to levels 2.4 times larger than present to achieve maximum profits. ► Macroscale assessments of governance require simplifications and assumptions. ► Stochastic simulations estimate NA fisheries loss in a range from B€2.5 to B€32. ► Securing that stocks biomass are above MSY is economically justified despite uncertainty on results.

Keywords : Economic assessment ; Fisheries, North Atlantic ; Management efficiency ; Uncertainty

Marine fisheries are an important source of food and livelihood opportunities worldwide (Allison et al., 2009; Garcia & Rosenberg, 2010; Rice & Garcia, 2011). The exploitation state of fish stocks is hotly debated (Branch et al., 2011; Pauly et al., 2002; Worm et al., 2009), but there is a general consensus that marine fisheries food production potential is not achieved (Branch et al., 2011; FAO, 2012). North Atlantic fisheries are nowadays yielding less fish than in recent decades and despite significant improvements (Fernandes and Cook 2013), the state of many of its stocks remains poor. Traditionally, the efficiency of biomass production has been the basis of fisheries management. Therefore, different regulations have aimed at maintaining fish stocks at levels at which they could produce their *Maximum Sustainable Yield* (*MSY*), i.e. the exploitation rate where the response of the stocks to fishing through individual growth and recruitment operates at its maximum capacity. In a deterministic sense, at this level, average fish biomass remains stable over time and the amount of fish that can be sustainably extracted is maximized (Schaefer, 1954). Classic approaches assume that these dynamics operate at a particular stock level, depending on the species' life history and thus, should fisheries management succeed in maintaining each of them at their MSY, the maximum potential of food production from marine ecosystems would be achieved. Using Economic Exclusive Zone (EEZ) and fish species data from the Sea Around Us database, the food production potential wasted due to ineffective management was estimated, i.e., the difference between catch observations and their MSY estimated from historic catch series (Srinivasan et al., 2010). Srinivasan et al. (2010) estimated that catch losses amounted to 7-36% of the reported annual catch, resulting in a landed value loss between $6.4 billion and $36 billion.

In reality, it is ecologically impossible to simultaneously maximize sustainable yield for all species in a multiple species fishery (Link, 2009). Therefore, the productivity of marine ecosystems is expected to be lower than predicted by the sum of single stocks' MSY (Link et al., 2012). The overall productivity and state of exploitation of marine ecosystems have been investigated previously with complex ecosystem models and indicators (Blanchard et al., 2012; Blanchard et al., 2009; Coll et al., 2008; Cury et al., 2008; Merino et al., 2012; Shin et al., 2005), and with single species models applied to entire exploited communities (Guillen et al., 2013; Link et al., 2012; Mueter & Megrey, 2006; Sparholt & Cook, 2009; Worm et al., 2009). For example, 'surplus production models' (SPM), have been used to produce simple representations of the key ecological processes underlying fisheries (Link et al., 2012). SPM can be used to estimate biological reference points (BRP's) such as the biomass level and the rate of exploitation to achieve the MSY of single fish stocks or marine ecosystems.

SPM have allowed the extension of fisheries assessment into other disciplines beyond ecology. For example, the seminal paper by Gordon (1954) introduced the concept of *Maximum Economic Yield* (*MEY*), the bioeconomic reference point at which the economic profits of a fishery are maximized. This concept relies on fish stocks' productivity described by SPM (Schaefer, 1954), the market price of fish and the costs of fishing. A derivation of this model was used to assess the economic efficiency at which the world's fisheries are exploited (Arnason et al., 2009), from which global MEY was estimated based on world's catch, value and costs databases. Arnason et al (2009) highlight the vast economic consequences of inefficient fisheries management and the economic benefit of maintaining fish stocks at healthy levels. Due to the high uncertainty in the data and the simplified model used, the numeric results of Arnason et al (2009) study were presented with caution and with wide confidence intervals. Nonetheless, the global cost of sub-optimal management was estimated to be in a range between $37-67 billion in 2004, with an historic accumulated loss of $2.2 trillion between 1974 and 2004. Arnason et al. (2009) did not explicitly evaluate the cost of rebuilding fish stocks, i.e., the cost of the necessary transition until stocks are recovered and more economic profit is obtained with less fishing effort. More recent research shows that the benefit of rebuilding global fisheries outweighs costs (Sumaila et al., 2012) and that investing in restoring overexploited stocks is economically sound (Crilly & Esteban, 2012). However, it is important to clarify that not all fish stocks are overexploited. For example, forty-three percent of assessed EU stocks were considered overfished in 2012 (Fernandes & Cook, 2013; European Union 2012). In any case, when fishing yields do not correspond to MSY this does not automatically mean a stock is overfished (Hilborn & Stocks, 2010). Hilborn and Stokes (2010) suggest that it would be reasonable to adopt a definition of being overfished as any stock size where the expected yield is 80% or less than MSY, which is the level at which reductions of fishing mortality towards MSY would produce measurable catch increases.

The North Atlantic basin is a dynamic environment for physical and biological processes (Beaugrand et al., 2002; Marshall et al., 2001; Parsons & Lear, 2001) and is home to some of the largest populations of commercially exploited stocks (Trenkel et al., 2013). With this at the background and due to the importance of North Atlantic global climate, BASIN (Wiebe et al., 2009) is a joint EU/North American research initiative with the goal of elucidating the mechanisms uderlying observed changes in the North Atlantic ecosystems and their services, and Euro-BASIN is a programme to implement this vision funded by the European Commission 7th Framework Programme (St. John et al. *introduction article of this issue*). In the context of

Euro-BASIN, this article aims to reflect the economic relevance of fisheries within the North Atlantic basin using some of the methods described above to estimate the economic cost of ineffective fisheries management, defining ‘ineffective’ as a deviation from maximum economic rent (Arnason et al., 2009). To do so, we tested alternative aggregations of fisheries production and economic indicators and parameterized a simple bioeconomic model. The scope and scale of this study is vast and complex and requires simplifications. The ecological complexity, regional differences and dynamics of individual fish stocks in the North Atlantic are simplified in an aggregated single stock of fish, which is exploited by an aggregated single fishery. While this approach has significant ecological difficulties, aggregated fisheries production functions are not new, and have been used to assess the economic efficiency of global fisheries as a single exploited unit (Arnason et al., 2009), at ecosystem level (Crilly & Esteban, 2012; Link et al., 2012; Sparholt & Cook, 2009) and at species-EEZ level (Srinivasan et al., 2010). The implications of this approach and justification for the use of an aggregated model will be discussed in detail throughout the manuscript. Furthermore, we explore the possible impact of parameter uncertainties and the assumptions made to obtain our numeric results. Finally, we discussed the use of multidisciplinary approaches in analyzing marine resources at the basin scale. These results provide background context to the work conducted in Euro-BASIN in the Bio-economic modeling (WP7) and Living resources (WP5) workpackages.

2. Material and Methods

2.1. The data

- *Biological parameters*: Catch data from ICES FishStatPlus database (www.ices.dk), FAO Fishery Statistics (www.fao.org) and Sea Around Us catch database (www.seaaroundus.org) were used to estimate the biological parameters of the surplus production model of the North Atlantic (NA) fisheries from 1950 to 2010. The data used comprise 59 ICES stocks, 18 species and 2 habitats exploited in the North Atlantic for the ICES area (see Table 1). These data were used to explore how alternative levels of stock and taxonomic aggregation could lead to different MSY estimates and indicate the uncertainty that the aggregation process undergone for the NA bioeconomic model. The overall NA basin MSY was estimated using datasets from FAO and was used as input for the bioeconomic model. A series of all the species landed in the NA was used.

- *Economic parameters*: Three main sources of information were used to obtain the economic parameters of the NA fisheries. First, the Sea Around Us database was used to obtain the value of the NA fishery as a whole. Second, a global fishing costs database at fleet segment level (Lam et al., 2011) was used to estimate countries total profits (Table 2). These estimates showed significant differences with the ones reported in the Annual Economic Report of the European Fishing Fleets (JRC, 2012).

2.2 The models

- *Biological parameters estimation*: We used a relatively simple method to obtain plausible MSY estimates and other biological parameters from catch data, based on assumptions on resilience (corresponding to the intrinsic growth rate *r* in the SPM) and the plausible range of relative stock sizes at the beginning of the time series (Martell & Froese, 2010). We used a medium resilience range as defined by Martell & Froese , i.e. $0.2<r<1$, and an initial (in 1950) relative stock size range of 50-90% of carrying capacity *K* or pristine biomass for all stocks (except for 'ghl-arct' ICES stock which was considered of 'low' resilience, $0.05<r>0.5$), and all species, habitat and the total NA. The identification of pairs of *r*-*K* values compatible with the catch time series and the above assumptions was performed using the R-code for batch processing made publicly available in http://www.fishbase.de/rfroese/CatchMSY_2.r for 59 ICES stocks ('ICESct2.csv', catch file processed and also made available by Martell and Froese), for the 18 species targeted in the ICES areas and for the entire NA basin from FAO catch data. The aggregation was a simple summation of catches of all the stocks of each of the 18 species, of all demersal and pelagic species and of all ICES stocks. Similarly, for the NA estimation, all NA species catches were summed to obtain a single catch time series. For each plausible *r*-*K* pair, an estimate is obtained as $MSY=1/4\ r\ K$. This MSY estimation algorithm has been validated against analytical fish stock assessment estimates of MSY (Martell & Froese, 2010). Good agreement was found between stock assessment MSY estimates and the geometric mean of MSY values calculated from the plausible r-K pairs (Martell & Froese, 2010).

- *Aggregated bioeconomic model* (Arnason, 2007; Arnason et al., 2009). This model assumes that the stocks exploited by global fisheries can be modeled as a single fish stock with an aggregate biomass growth function and a fishing industry operating exclusively in the area. The economic performance from fisheries is estimated with the value of the global landings calculated with an aggregated harvest function (SPM by Schaefer 1954) and an aggregated fishing cost function relating current fishing effort to fisheries costs. Incorporating NA fisheries into a single fishery allows for a model with a manageable number of parameters. This model requires 4 biological parameters: (i)

Global MSY, (ii) total ‘carrying capacity’ or ‘unexploited biomass level’, (iii) fish biomass growth in the last year and, (iv) a ‘schooling’ parameter; and 5 economic parameters from the fishing industry: (i) Landings, (ii) value of landings and, (iii) total profits from fishing in the last year, (iv) ‘fixed costs ratio’ and, (v) ‘elasticity of demand’ with respect to total biomass. The MSY has already been explained and ‘unexploited biomass’, fish biomass growth, landings, value of landings and profits are self-explanatory. The ‘schooling’ parameter describes the spatial distribution behavior of fish and ranges between 0 and 1. The lower the parameter the more aggregated the fish, e.g., small pelagic stocks like anchovies, sardines, mackerel etc. When this parameter is close to 1, fish are homogeneously distributed in space, e.g., demersal species such as hake or plaice. For our analysis we fixed this parameter as 1, to assume that all fish are homogeneously distributed throughout the NA. However, the impact of this parameter on the final calculations is explored in the Appendix. The ‘fixed costs ratio’ describes the fraction of the total costs incurred by the fishing industry that are not originated by labor, fuel, capital and other factors of production such as maintenance, repair, supplies and gear costs. We considered this ratio to be 0 as in the global study (Arnason et al., 2009). Assuming a zero value means that fishing effort is measured as the size of fishing industry and not by its activity (if inactive, fleets would still generate fixed fishing costs). The elasticity of demand to biomass expresses the price of fish as dependent on the global marine commercial fish biomass. This elasticity is positive: when there is overexploitation, biomass is at a low level, the proportion of low value fish is higher and mean price is smaller; in the other direction, if fish stocks recover from overexploitation, the average size and trophic level of caught fish increased and price does so likewise. This is a manifestation of the “fishing down the food web” effect (Pauly et al 1997).

The bioeconomic model also required additional input that was obtained as follows: from the FAO catch data series we obtained catches in 2010 (Y_{2010}) and biomass growth in 2010 ($G(x)_{2010}$). We used the difference between Y_{2010} and Y_{2009} as $G(x)_{2010}$ which assumes that catch changes were only caused by abundance changes rather than management or other factors. For the value of catches, the Sea Around Us database was used to complement FAO data ($Value_{2010}$). The profits of NA fisheries in 2010 was obtained summing the national profits (Table 2) obtained from Lam et al (2011) applied to NA value of catch.

(# Insert Table 2 #)

We plot the classical equilibrium catch-biomass curve (Schaefer, 1954) and different potential profit curves (*iso_Ψ*) defined by the profits in 2010 (Ψ_{2010}), price at equilibrium (p_{eq}), costs of fishing per unit of effort (c) and biomass at equilibrium (equation 1).

$$iso\_\Psi = \frac{\Psi_{2010}}{p_{eq} - c \cdot B_{eq}} \quad \text{(equation 1)}$$

The points where *iso_Ψ* trajectories meet the catch and biomass equilibrium curve are a feasible sustainable profit, catch and biomass equilibrium points. The maximum feasible *iso_Ψ* is searched to identify the *MEY* of North Atlantic fisheries. Further transformations of the basic equations by Gordon-Schaefer required to plot the curves are explained in the Appendix. Economic loss is then calculated as the difference between this MEY value and realized profits in 2010.

Stochastic simulations: The economic loss estimated with the deterministic model for 2010 was re-estimated allowing for uncertainty on the input parameters (Table 3): (i) random values in a range of ±30% of the initial parameters ('sim 1'), (ii) lognormal distributions for MSY and K as provided by the Martell and Froese (2010) estimation model and random for the others ('sim 2'), (iii) lognormal distributions for MSY and K and random for Catch 2010 ('sim 3') and, (iv) random within ±30% for all parameters but MSY, *K* and Catch 2010 which were kept constant ('sim 4'). For all stochastic simulations the model was run for 10^5 iterations to equilibrium.

3. Results

The total MSY for all the ICES stocks combined was estimated to be between 6.68 and 9.75 million tonnes, depending on the level of catch aggregation from which the estimates were calculated (Table 1 and Figure 1).

Biological parameters were estimated for each of the ICES stocks and were then aggregated into species, habitat and total ICES areas. The total MSY estimate for the ICES fisheries decreases exponentially with the level of aggregation, with MSY estimates 30% lower when using ICES area aggregation (largest aggregation) compared to estimates from stock level aggregation (lowest aggregation). Although we don't use the estimated MSY by ICES area for the basin-scale analysis, the differences between estimates arising from different levels of aggregation were used to calculate the confidence limits of MSY in the North Atlantic, which were used as inputs for the bioeconomic model.

(# Insert Figure 1 #)

(# Insert table 1 #)

We used the time-series of total aggregated NA landings applied to the algorithm by Martell and Froese (2010) to estimate an MSY of 13.7 Mt (s.d.=0.04) (Figure 2). Historically North Atlantic fisheries were considered to be under development up to 1970s, when total landings started to exceed MSY considerations. From 1980 until the early 2000s, the total catch has fluctuated near this estimated global MSY. Since then landings have decreased to levels approximately 80% of basin MSY. This model also estimated the carrying capacity parameter (K) or unfished biomass for North Atlantic fish resources to be 170 Mt (Table 3).

(# Insert Figure 2 #)

The bioeconomic model estimated that NA fisheries could generate B€ 12.85 of profits compared to the current B€ 0.63 (Figure 3). In addition, this equilibrium model shows the biomass level (45Mt, 26% of K) if current profits were to be maintained. In summary, this figure indicates that allowing stocks to rebuild to the biomass consistent with MEY (108 Mt, 63.5% of the unexploited biomass) would allow multiplying profits 20 fold. In other words, NA fisheries are only generating 5% of their economic potential. It must be noted that the catch at MEY is estimated to be 12.66 Mt, only 25% larger than the catch level in 2010.

(# Insert Table 3 #)

(# Insert Figures 3 and 4 #)

Using the classic revenue-cost against fishing effort curve by Gordon (1954) (Figure 4), the effort level that would lead to the economic maximization of North Atlantic fisheries is estimated to be 47% of current effort. Thus, should fishing effort increase 10% above current levels, the NA fisheries would incur economic losses. Figure 4 also shows that assuming equilibrium conditions NA fisheries in 2010 were near the “Bioeconomic Equilibrium” (BE), the point at which the fishery rents are dissipated as fishing costs are equal to the revenues from fishing.

(# Insert Figure 5 #)

The computations above are subject to uncertainties, and thus we added a level of stochasticity to our model’s input parameters, which indicated that the economic loss of North Atlantic fisheries in 2010 ranged between B€2.5 and B€32 when all parameters were randomly fluctuating with a 30% coefficient of variation (Figure 5). MSY and Y_{2010} are the most important sources of variation when estimating the economic losses of fisheries (Arnason, 2007; Arnason et al., 2009) and Figure A.1 (Appendix). Besides, significant uncertainty was propagated into estimates of stocks' carrying capacity (sdLog=0.24). Therefore, specific simulations investigating the impact of those three parameters were performed. Uncertainty in estimation was moderately reduced by varying the three parameters through lognormal distributions and generating random values with a uniform distribution with bounds ±30% for the others ('sim 2') or assuming them constant ('sim 3'). The simulations 'sim 3' and 'sim 4' confirmed that these parameters generated the largest uncertainty on the final estimates of economic loss. For 'sim 4', fixing MSY, K and Y_{2010} the variability of the loss estimate was reduced significantly, ranging between B€ 6 and 19 with 95% confidence. The most important result from these simulations is that the model is more sensitive to biological parameters and therefore, biological parameterization is more important than economic parameterization.

4. Discussion

We have provided an assessment of the economic losses due to the choices taken in the management of North Atlantic fisheries. We have used methods previously implemented in the assessment of the economic losses of global fisheries (Arnason et al., 2009). Such a focus on the North Atlantic, in the context of the Euro_BASIN project, is motivated by the fact that its fisheries have a long history and economic importance, with significant catch-independent and dependent data sets, which are managed at different scales and with different degrees of success and failure.

The catch and value of North Atlantic fisheries have declined significantly in the last decade, partially due to management restricting catches (see below). The economic opportunity lost through the inefficient management of North Atlantic fisheries in 2010 was estimated to be B€ 12.2. This echoes the results of a bioeconomic model built imposing strong assumptions on North Atlantic basin biological productivity and economic data of the fishing fleets operating in the area. Arnason et al 2009 estimated the global economic loss of marine fisheries due to overexploitation to be in a range of $B 37-67. North Atlantic landings correspond to approximately 12% of global catches and the economic loss of NA fisheries represents ~33% of global losses. This may be caused by the relative larger price of NA fisheries in comparison to other areas (Sumaila et al., 2007) and by the historical overfishing history of North Atlantic fisheries (FAO, 2012). For the North East Atlantic, Crilly and Esteban (2012) estimate that restoring fish stocks could deliver up to £4.43 billion per year in profits, approximately 41% of our estimate for the entire NA.

Fisheries assessment provides information on the state of exploitation of marine resources and is generally performed at stock level, a harvested unit which dynamics are driven by recruitment, growth, natural mortality and fishing. Because of the limited number of fish stocks with stock assessment data at the basin-scale, the catch based approached employed in this study allow us to include a wider range of fish stocks. However, the catch based approach is based on the assumption that catch reflects fish abundance and productivity. This principle is controversial, especially when management interventions change through the history of catch time-series (Pauly et al., 2013). However, catch-based methods are widely used to assess data-poor fisheries and to produce large scale overviews of the state of fisheries (Fernandes et al., 2013; FAO, 2012; Lleonart & Maynou, 2003; Pauly et al., 2003, Vasconcellos and Cochran, 2005). Data on North Atlantic fisheries' are abundant, especially for ICES-assessed stocks. A specific problem arises because data are not available at the basin scale, one of the challenges that the Euro-BASIN project tries to address. Also, the proportion of assessed stocks in relation to total catch differs across regions of the NA. For example, more than 90% of the North Sea catch (areas IVa-c) corresponds to assessed stocks but in the Celtic Sea (VIIe-k) this number is less than 40% (Gascuel et al., 2012). Using data that are only available from ICES statistical area may thus provide a biased view of the status of fisheries in the NA basin. Besides, multi-species MSY is less than the sum of single stocks', as demonstrated in this and other studies (Link et al., 2012; Sparholt & Cook, 2009). Multi-species MSY could have been estimated with ecosystem models as well. Fish species dynamics are regulated through trophodynamic interactions and energetic fluxes across trophic levels (Pauly et al., 2000; Shin & Cury, 2004), which are reflected in the ecosystem's size spectra (Blanchard et al., 2009). For example, capelin, cod and herring interact in the Barents Sea food web (Lindstrøm et al., 2009). However, these models are relatively complex in relation to SPM (Coll et al., 2008). We favor the use of a simplified aggregated surplus production model because these models can produce robust estimates of multispecies environments (Sparholt & Cook, 2009) allowing for comparison across areas towards the practical implementation of the ecosystem-based fisheries management (Link et al., 2012). Also, this model provides a consistent platform to produce a macro scale assessment of North Atlantic fisheries in combination with economic information.

We acknowledge that the use of an aggregated economic model requires significant simplifications of complex ecological processes, and masks geographical differences in ecosystems productivity and management efficiency. For example, let us look at two cod stocks in the Irish and Icelandic Seas. Recent annual landings of Irish Sea cod have been lower than 5 k t with prospects for zero catch in 2013 and a stock which is currently outside biological limits (ICES 2012a). In contrast, the Icelandic cod's TAC for 2012 was 177 k t and the stock is considered inside safe biological limits (ICES 2012b). The overall fishing effort recommended to achieve MEY for NA fisheries would not be expected to be applied homogeneously to all stocks. The aggregative approach by-passes stock-specific responses and assumes that fishing effort reductions would have to focus those stocks catalogued as “overexploited” or “under overexploitation”, and that benefits from adequate fishing management will be especially notable for the most productive areas of the North Atlantic.

Based on our analysis, North Atlantic fisheries remained within the estimated MSY range from 1964 to 2005. Then, total landings declined significantly and the estimation model associates this to overall overexploitation of NA resources. However, marine ecosystems are driven by multiple drivers that change over time; therefore, a constant historical MSY may not be realistic. The constant MSY estimated in the first part of this work is used for the subsequent economic assessment and should be considered with caution. Catch reductions can be caused by multiple factors, including overexploitation, environmental variability or implementation of catch restrictions. Overexploitation is defined by Hilborn and Stokes (2010) when catches are below 80% MSY, which equates to when declining yields are obtained with increased fishing effort (Schaefer 1954). The same is concluded from the biomass-catch diagram shown in Figure 3. When MSY is exceeded for extended periods and if fishing effort is maintained beyond the level corresponding to MSY, yield will decrease as the available biomass has fallen below the point at which MSY is achieved (50% of its unexploited level, in this case). That is, biomass decreases with increasing catch until the point when biomass reductions will result in lower catches if fishing effort is not increased. In multispecies fisheries apparent MSY levels can be maintained by targeting previously undeveloped fisheries simultaneously with declining stocks. Using theoretical models, it has been shown that this feature can precede a sequential collapse of geographically distant fisheries (Merino et al., 2010; Merino et al., 2011).

This model does not consider environmental effects on the productivity of the NA basin. In reality, fish stocks, especially small pelagic fish (70-80% of total NA catch), are highly vulnerable to environmental variability (Barange et al., 2009; Chavez et al., 2003; Fernandes et al., 2010; Hsieh et al., 2009). However, it is also evident that the impacts of particular environmental conditions differ between species. For example, Icelandic capelin catch averaged 1 Mt from 1979 to 2002 (13% the yields from ICES assessed stocks) when it started declining to 15 kt in 2008. This decline is reflected in the overall NA trend and it could be caused by temperature changes (Carscadden et al., 2013 (In press)). However, other stocks such as herring (yielding ~2Mt in the last decade) seem to be favored by current conditions and have recovered from overexploitation faster than expected (Nash et al., 2009), which could counterbalance the negative environmental impact on capelin on the basin scale trend. Another example is blue whiting whose catches have displayed a dramatic “boom and bust” dynamic over the past two decades (ICES, 2011). Landings during the 1980s and early 1990s were typically between 500 and 1000 kt, but increased to 2400 kt in 2004 as a result of a suite of good year classes. At this point, blue whiting was the largest fishery in the North Atlantic, ahead of herring, and the third largest marine capture fishery in the world (FAO, 2010). The subsequent decline of the fishery has, however, proved to be equally dramatic (ICES, 2011). The alternation between warm and cold regimes is associated to alternative species proliferation (Chavez et al., 2003), including multidecadal regime shifts (Alheit et al., 2009). However, investigating each of the environmental drivers affecting fish stocks in the North Atlantic in order to better estimate individual MSYs would mean losing focus on the principal objective of this study and its scale.

A third factor resulting in catch reduction is management restriction. Generally, closures and drastic catch limitations are the consequence of overexploiting resources and subsequent fishery crises (Finlayson, 1994; Lazkano et al., 2012; Nøstbakken & Bjørndal, 2003; Worm et al., 2009). Historically, fish stocks have collapsed due to a myriad of unfavorable environmental conditions and excessive fishing pressure (Alheit et al., 2009; Chavez et al., 2003; Merino et al., 2013; Watson et al., 2006) and which triggered consequent catch restrictions (Worm et al., 2009). However, we would like to stress that, particularly the catch reduction in the last ten years of the data series, should be attributed not only to historical overfishing but also to management driven catch limitations. For example, under the EU framework, the Common Fishery Policy and the Financial Instrument for Fisheries Guidance (FIFG) a remarkable reduction of fishing boats has been accomplished (Fernandes and Cook 2013). In addition, since 2005 emergency and recovery plans have applied under the EU adopted MSY framework aiming to reduce fishing mortality towards achieving MSY for different stocks which is already improving fisheries economic indicators (Cardinale et al., 2013). Furthermore, this approach is followed by the International Council of the Exploration of the Sea (ICES) and other international agreements (FAO, 2012). To sum up, some of the catch reductions reflected in the basin scale trend (Figure 2) are aligned to the implementation of international efforts to restore fish stocks and this can potentially bias the parameter estimation procedure used in this study by estimating as economic loss what in reality may be a short term economic loss “invested” in stocks recovery towards more profitable fisheries.

The parameters used in the bioeconomic model can be controversial too: For example, classically, the supply-demand relation is considered as inverse: the lesser the catch, the higher the price. However, the positive elasticity parameter used here was taken from Arnason et al (2009) which aligns with a global perspective of the state of marine fisheries, as the “Fishing down the food web” concept (Pauly et al 1997) does. However, this is expected to have low impact in our numeric results: The estimated catch increase when moving towards MEY would be small, so the expected price changes would be small too. A different matter is the potential impact of exogenous variables on North Atlantic fish demand and therefore, in the price equation used in this document. We do not consider the impact of aquaculture expansion on the price of wild fish nor the impact of imports that might act as less priced substitutes to North Atlantic fish. Both factors could presumably reduce the price of North Atlantic fish and therefore, the potential economic profit of North Atlantic fisheries would be reduced. Finally, our model is based on estimates of current profits of NA fisheries, estimated with value and fishing costs databases, and without considering the effects of subsidies. According to Sumaila et al (2012), 31% of landed value in world fisheries is subsidized and therefore, the current profits for the fishing companies are presumably larger than the B€0.63 used to parameterize our bioeconomic model.

The implementation of ecosystem-based fisheries management (EBFM) requires the development of models to assess the economic performance of the fishing industry in combination with their impact upon marine ecosystems (Gascuel et al., 2012). The bioeconomic model used here was parameterized with a global estimate of NA ecosystems productivity and the sum of the economic performance indicators of the countries operating in its waters. In contrast to the biological part, the aggregation of the economic parameters was additive, we estimated the NA value of catch and net economic profits as the sum of the national estimates. The values shown in table 2 were obtained collating catch and value data from the Sea Around Us database and estimating the fleet specific costs of fishing using costs per tonne estimates from Lam et al (2011). Fishing costs and net profit values were also available from alternative reports. For example, the Annual Economic Report (AER) on the EU fishing fleet (JRC, 2012) provides estimates of many fishing indicators of EU countries. However, this report aggregates all EU countries fishing operations in waters beyond the NA. Using costs of fishing per tonne of catch in the NA allows for assigning the fishing costs only to the operations targeting North Atlantic fish. However, the cost structure provided in the AER is more detailed than in our approach. The net profit of EU fleets operating in the NA estimated in the AER is B€-0.236. Had this value been used as input to our bioeconomic model, our estimated loss would have been even larger. Additional sources of information on the economic performance of Russian, Norwegian, US and Canadian fleets (FAO, 2007; Kitts et al., 2010; NOAA, 2011) could improve the economic parameter estimation process. However, as seen in Figure 5, the most determinant set of parameters are those related to ecosystems productivity.

Our approach is based on deviations from biological and economic reference points. The economic loss pivots around the concept of *Maximum Economic Yield*, an equilibrium point where the net economic return from a fishery can be maximized sustainably, as assumed in previous studies (Arnason et al., 2009; Crilly & Esteban, 2012; Sumaila et al., 2012). This reference point is estimated with a graphical procedure (Figure 3). Large benefits will be considered as unsustainable as they do not meet the parabola and; lower than the optimal will cross it twice, one for high levels of biomass and the other at biomass levels below that corresponding to MEY. It is important to note that the recovery of the stocks towards MEY biomass would not produce major changes in the overall catch from the NA. In 2010 10.8Mt of fish was landed whereas for the MEY total catch would be 12.66 Mt. Therefore, a catch increase of 26% would produce a net economic gain of 2000%, but would require a 53% reduction in fishing effort. According to this, the economic benefit of restoring stocks would outweigh its potential food security implications (Garcia & Rosenberg, 2010; Rice & Garcia, 2011; Srinivasan et al., 2010). By reducing the fishing effort, costs would reduce linearly as revenues would increase potentially until the MSY peak. Then, further effort reductions would make revenues reduce too until its gradient equals fishing costs lines slope. As a result, a fishing effort reduction would produce a logarithmic increase in profits. Therefore, the profit increase would be more substantial at the initial stages of reduction. For example, if total fishing effort was reduced to 70% of current levels, total fish biomass (not each and every stock) would recover to MSY and profits would increase up to B€10.8 (1725% more than in 2010). Therefore, accepting the hard transition of reducing the size of the industry to 47% of current level, it is important to note that moderate reductions would also produce large economic benefits as well as improving resource conservation significantly.

The reduction of fishing effort will have negative short term costs in the form of reduction of catch towards stocks recovery, loss of a notable number of current jobs provided by fisheries and costs to dismantle a number of the fishing boats currently operating in the North Atlantic. Therefore, it will require investments to reallocate fishermen in alternative activities, scrap fishing vessels and other compensations to the fishing industry. Crilly and Esteban (2012) and the work by Sumaila et al. (2012) demonstrate that after a short transition the benefits of restoring fish stocks outweighs the costs incurred and investments required to reduce fishing mortality. This conclusion holds notwithstanding the high uncertainty in estimates and the assumptions made to enable large scale assessments of governance (Cash & Moser, 2000; Christensen & Walters, 2004; Jennings et al., 2008; Wilbanks & Kates, 1999). In addition, restoring fish stocks would avoid reducing the risk of fisheries collapses and its dramatic economic consequences. For example, the collapse of cod produced an increase of 30% of unemployment in some areas of Newfoundland and more than $3 billion were spent to restructuring adjustments for workers in the fishing sector, among other social implications (Hamilton and Butler, 2001). However, it is also true that fishing mortality reductions haven't always produced the stocks' recovery predicted by fisheries assessment models. For example, a combination of environmental changes and fishing pressure are responsible of Atlantic cod populations failure to recover (Hilborn and Litzinger, 2009).

A single estimate of economic loss is intuitive but can be simplistic given the number of parameters involved in the computation. In order to add consistency to our results and to offset the uncertainty associated with our methods, four stochastic experiments were conducted with the bioeconomic model. The results of these experiments provide two conclusions: First, allowing as much as a 30% random variation in the input parameters, the estimated economic loss of North Atlantic fisheries is measured in billions of euro. Second, the model is particularly sensitive to three biological parameters: MSY, K and catch in the last year. Reducing the uncertainty on these parameters reduces the standard deviation of the estimates significantly. In contrast, fixing the other five parameters produces only moderate reductions of variability on the economic loss of fisheries. Therefore, we emphasize the relevance of adequate commercial and fishery independent data collection programs in order to improve the stock assessment process. Despite uncertainties on the current scale of North Atlantic basin productivity, we conclude that an overall fishing effort reduction is recommended, with not only ecological benefits but significant and demonstrable economic consequences.

To conclude, our analysis supports the work conducted under the Euro-BASIN project by providing a basin-scale framework for the economic analysis of the efficiency of North Atlnatic fisheries management. In the future this analysis needs to take into consideration the way European fisheries management, in particular, is evolving. The reform of the European Common Fisheries Policy identifies MSY as a management target, consistent with our analysis. It also highlights the need to implement a discard ban, which should come hand in hand with the needed improvement in the monitoring and reporting of fishing activities. While our analysis is conducted at the basin scale, regionalization of management is a process that would need to be considered in future monitoring programs and modeling approaches. Significantly, the CFP reform also identifies the need to collect environmental, social and economic data and use these as criteria to allocate fishing rights. Future Euro-BASIN initiatives would have to consider the above in developing their workprogramme, as well as approaches to better understand market price formation (exports and competition with products from other areas) and how to influence consumer demand for species that traditionally have been less preferred.

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Appendix

Table A1. Necessary transformations to run the bioeconomic model using the parameters shown in Table 3 (Arnason, 2007; Arnason et al., 2009; Gordon, 1954; Schaefer, 1954).

{Insert Table A1}

Figure A1. Sensitivity analysis of the economic loss in 2010 for different parameters.

{Insert Figure A1}

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Figure captions

Figure 1. Maximum Sustainable Yield estimates for the stocks assessed by ICES at different aggregation levels. The aggregation level is the number of units considered: 59 stocks, 18 species, 2 habitats and 1 for all the ICES stocks as a single fishery. Boxes show the geometric mean of the estimate, 0.25 and 0.75 quantiles. The intervals limit the estimate to a 99.5% confidence (see text).

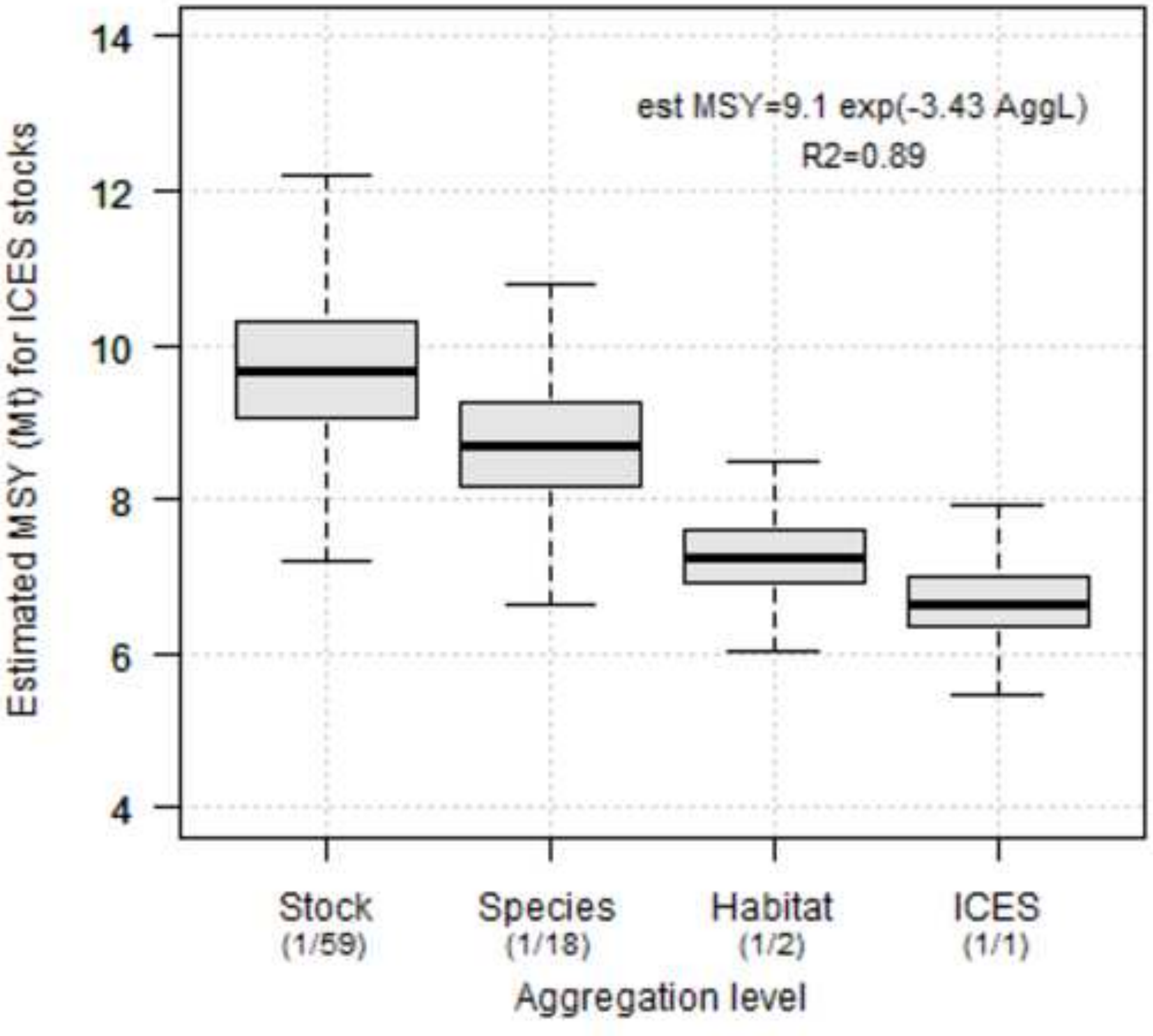
Figure 2. Top: Historical landings of North Atlantic fisheries according to FAO and estimated corresponding MSY. Below: Density distribution of annual total landings and distribution of plausible total MSY values using the approach by Martell and Froese (2013).

Figure 3. Graphical estimation of North Atlantic fisheries Maximum Economic Yield. The crossing point between different potential profit trajectories (*iso*_Ψ) and the biomass-catch equilibrium curve determines the catch and biomass level that will lead to MEY. The MEY for North Atlantic fisheries is B€ 12.85. The *iso*_Ψ =0.63B€ corresponds to profits in 2010.

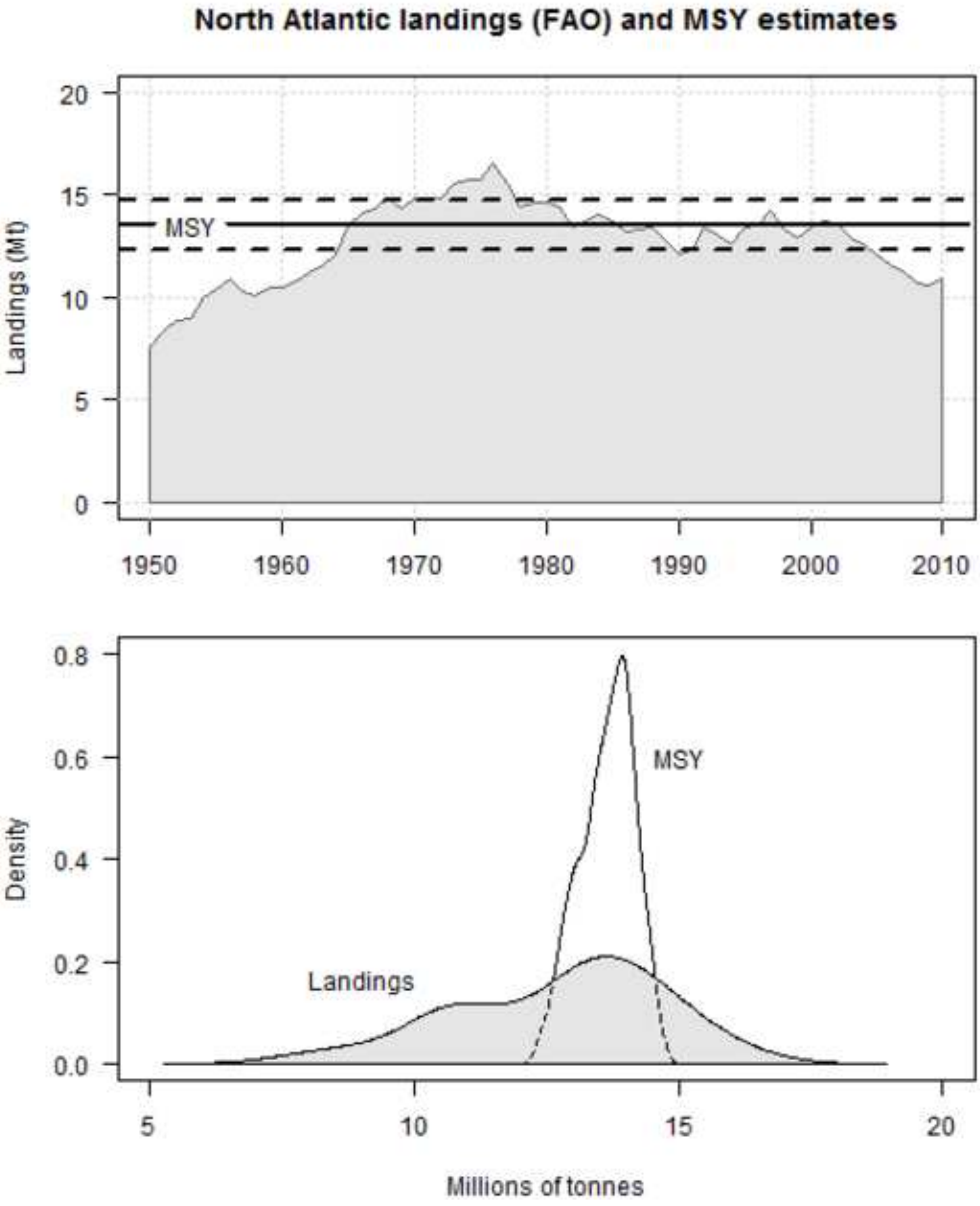
Figure 4. Gordon-Schaefer's model equilibrium for North Atlantic fisheries. Current (2010) and economically optimum fishing efforts indicated with dotted lines. Net profits are calculated as the difference between value of catch and costs of fishing. MEY is the *Maximum Economic Yield,* i.e. the maximum difference between value of catch and costs of fishing.

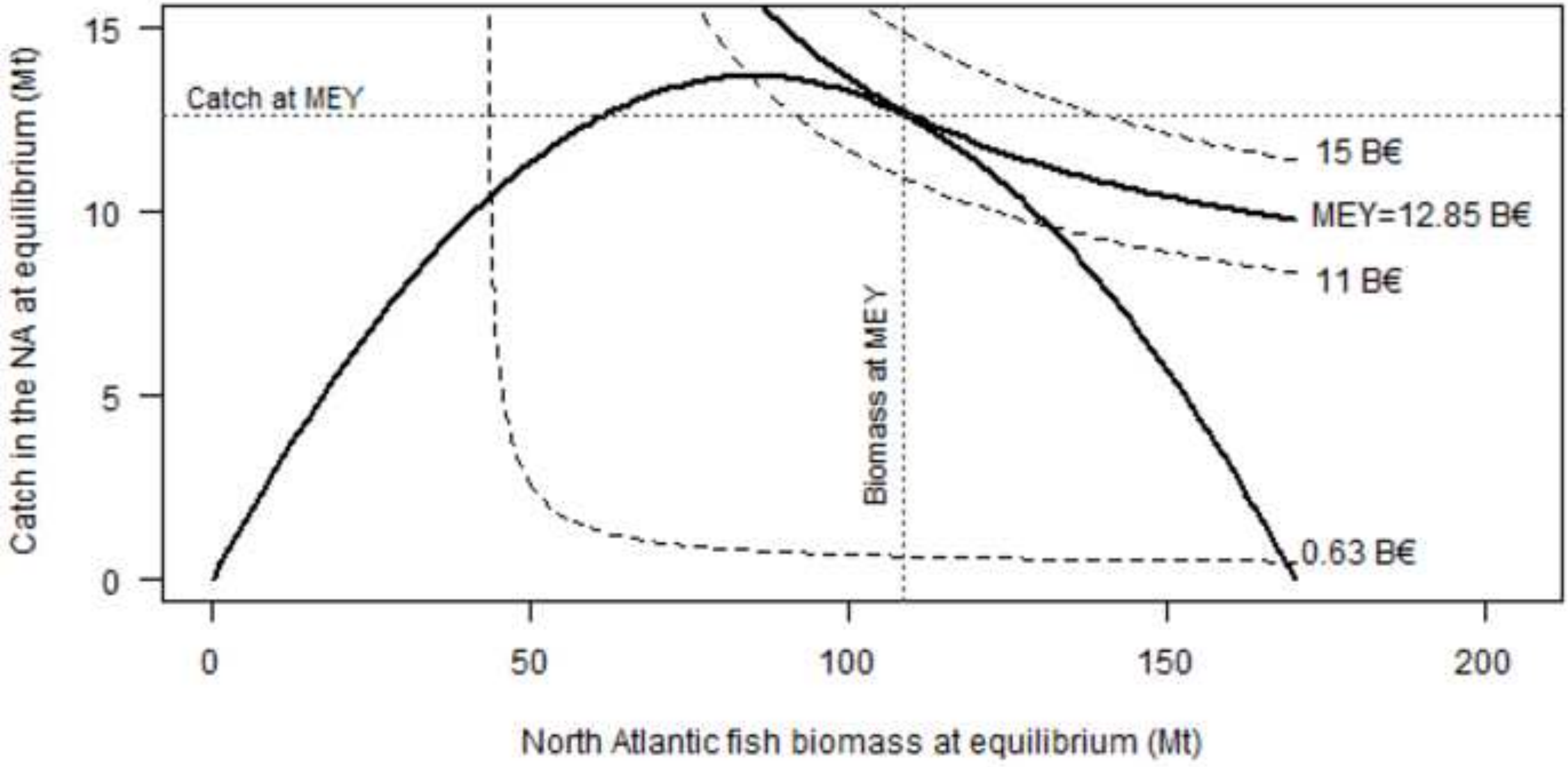
Figure 5. Results of stochastic estimates of the economic loss of North Atlantic fisheries in 2010: 'sim 1' random fluctuation (±30%) of the parameters of the bioeconomic model (see table 3); 'sim 2' log-normally distributed MSY, K and Y_{2010} and random fluctuation for the others (±30%); 'sim 3' same as previous but with other parameters kept constant at values shown in table 3; 'sim 4' MSY, K and Y_{2010} constant and random fluctuation for the others (±30%).

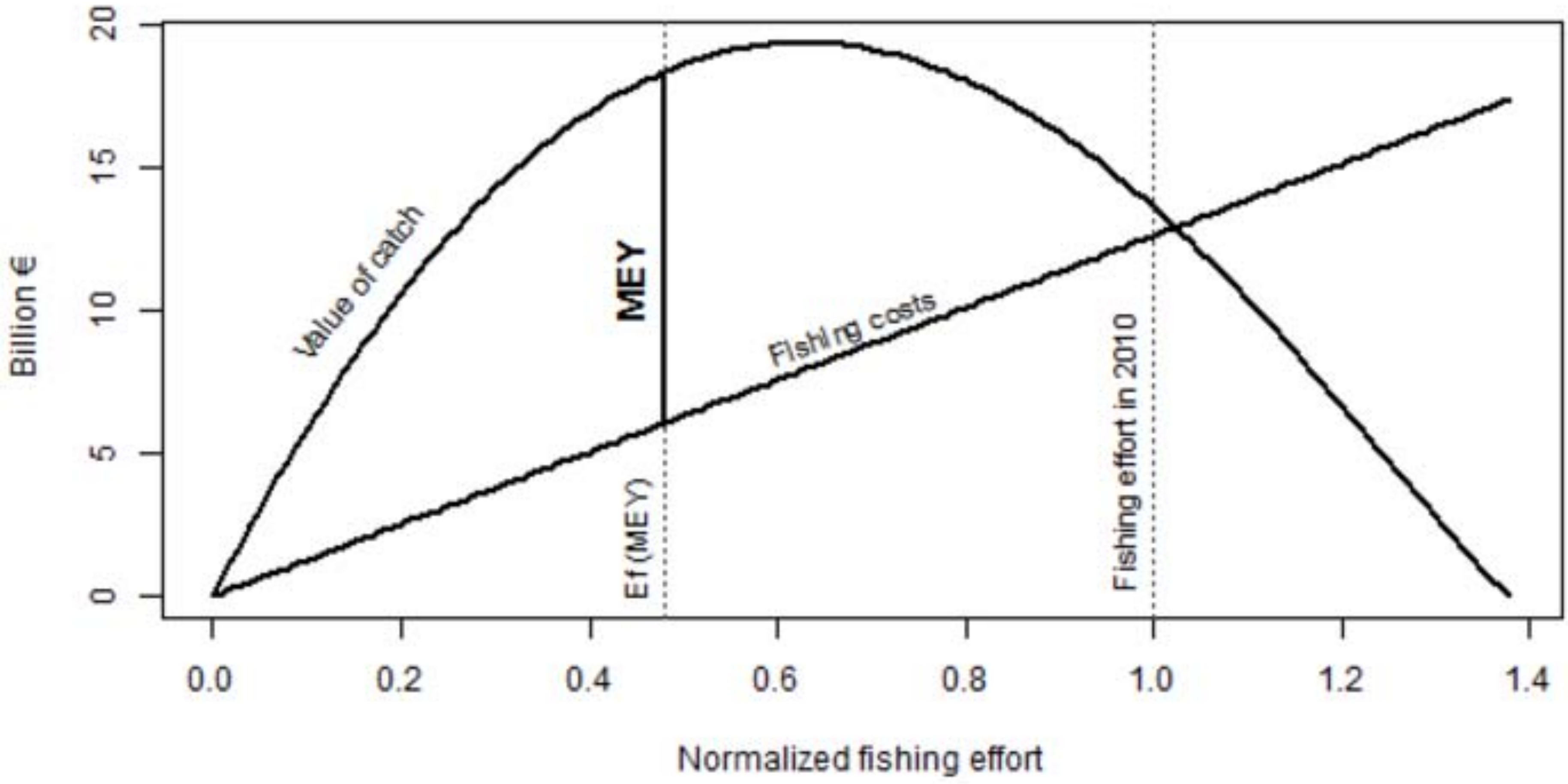
Figure A1. Sensitivity analysis of the economic loss in 2010 for different parameters.

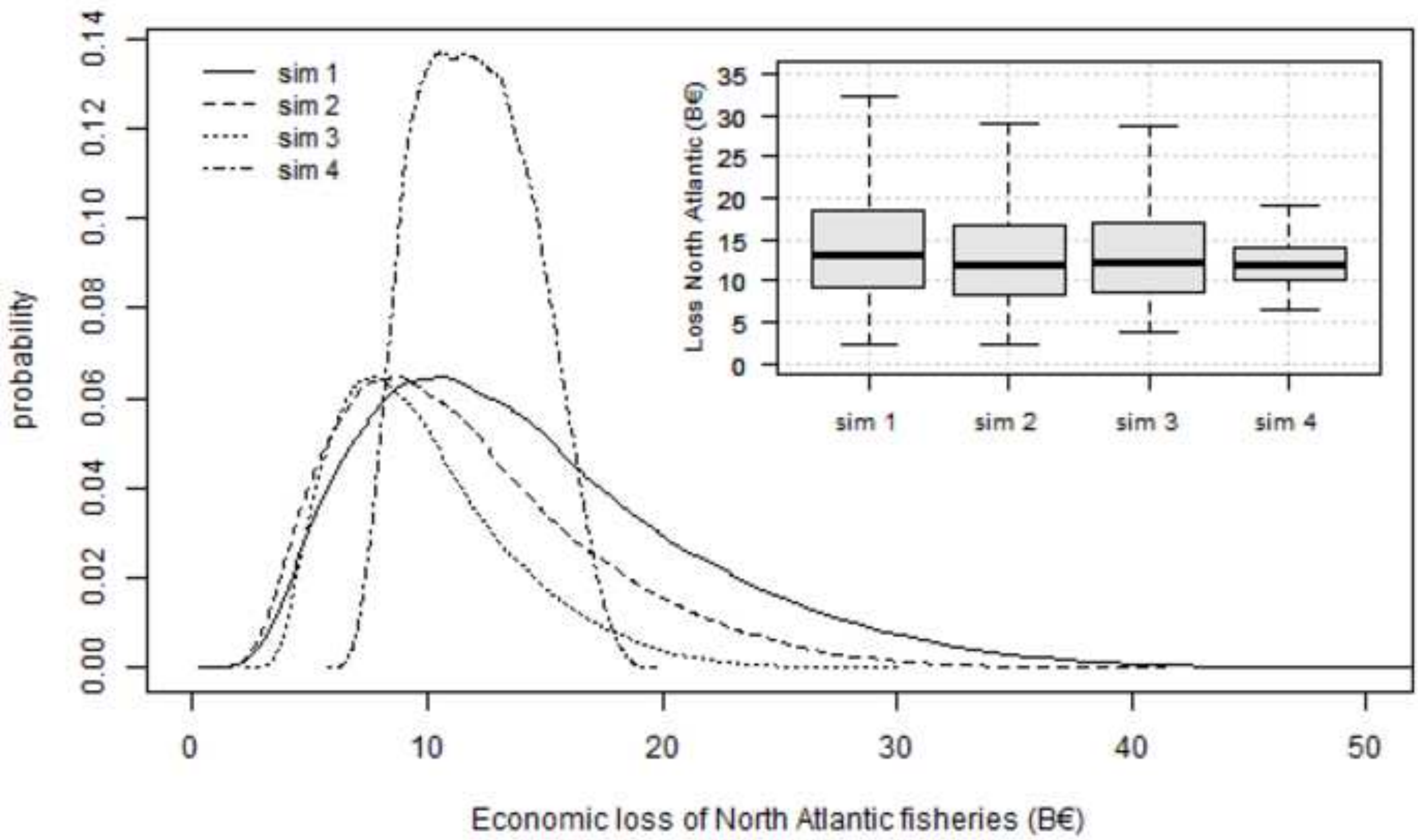


est MSY=9.1 exp(-3.43 AggL)
R2=0.89
Estimated MSY (Mt) for ICES stocks
14
12
10
8
6
4
Stock
(1/59)
Species
(1/18)
Habitat
(1/2)
ICES
(1/1)
Aggregation level

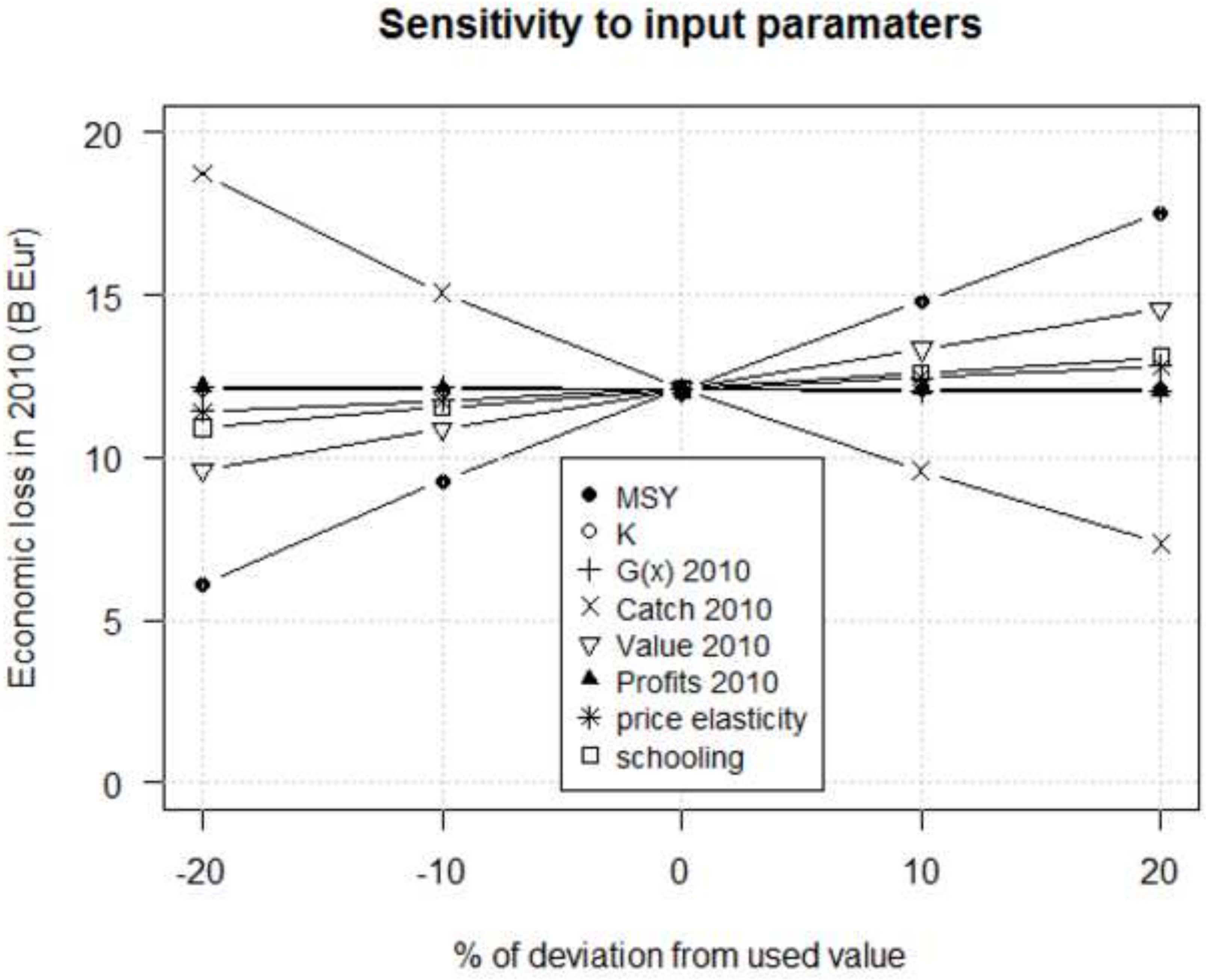

North Atlantic landings (FAO) and MSY estimates
Landings (Mt)
20
15
10
5
0
MSY
1950
1960
1970
1980
1990
2000
2010
Density
0.8
0.6
0.4
0.2
0.0
MSY
Landings
5
10
15
20
Millions of tonnes


Catch in the NA at equilibrium (Mt)
North Atlantic fish biomass at equilibrium (Mt)
Catch at MEY
Biomass at MEY
15 B€
MEY=12.85 B€
11 B€
0.63 B€
0
5
10
15
0
50
100
150
200


Billion €
0
5
10
15
20
0.0
0.2
0.4
0.6
0.8
1.0
1.2
1.4
Normalized fishing effort
Value of catch
Fishing costs
MEY
Ef (MEY)
Fishing effort in 2010



sim 1
sim 2
sim 3
sim 4
probability
0.00
0.02
0.04
0.06
0.08
0.10
0.12
0.14
0
10
20
30
40
50
Economic loss of North Atlantic fisheries (B€)
Loss North Atlantic (B€)
0
5
10
15
20
25
30
35
sim 1
sim 2
sim 3
sim 4


Sensitivity to input paramaters
Economic loss in 2010 (B Eur)
20
15
10
5
0
-20
-10
0
10
20
% of deviation from used value
● MSY
○ K
+ G(x) 2010
× Catch 2010
▽ Value 2010
▲ Profits 2010
✳ price elasticity
□ schooling

Tables

Table 1. Estimated Maximum Sustainable Yield for different levels of aggregation of ICES-stocks.

Total *MSY (Mean Mt, sdLog)*	*Habitat* *MSY (Mean Mt, sdLog)*	*Species* *MSY (Mean kt, sdLog)*	*ICES-Stock* *MSY (Mean kt, sdLog)*
ICES (6.68, 0.07)	Demersal2.26, 0.03)	Cod (1364.61, 0.03)	cod-2224 (38.91, 0.09)
			cod-2532 (181.77, 0.06)
			cod-347d (256.87, 0.05)
			cod-arct (737.22, 0.05)
			cod-coas (59.71, 0.1)
			cod-farp (24.48, 0.05)
			cod-iceg (374.87, 0.06)
			cod-iris (8.72, 0.07)
			cod-scow (16.41, 0.12)
		Greenland halibut (20.38, 0.20)	ghl-arct (20.16, 0.2)
		Haddock (419.40, 0.04)	had-34 (281.57, 0.07)
			had-7b-k (12.61, 0.18)
			had-arct (164.18, 0.13)
			had-faro (18.11, 0.06)
			had-iceg (81.20, 0.18)
			had-rock (4.67, 0.12)
			had-scow (23.36, 0.07)
		Megrim (1.23, 0.11)	mgb-8c9a (1.23, 0.11)
		Anglerfish (0.46, 0.14)	mgw-8c9a (0.46, 0.14)
		Nephrops (5.15, 0.09)	nep-8ab (5.68, 0.11)
			nop-34 (0.18, 0.13)
		Plaice (122.48, 0.04)	ple-celt (1.18, 0.07)
			ple-eche (4.32, 0.07)
			ple-echw (1.68, 0.08)
			ple-iris (3.38, 0.05)
			ple-nsea (114.97, 0.04)
		Sole (31.93, 0.06)	sol-bisc (5.27, 0.08)
			sol-celt (1.12, 0.06)
			sol-eche (4.73, 0.11)
			sol-echw (0.97, 0.05)
			sol-iris (1.40, 0.08)
			sol-kask (0.77, 0.09)
			sol-nsea (21.4, 0.04)
	Pelagic (4.99 Mt, 0.08)	Capelin (867.50, 0.07)	cap-icel (866.05, 0.07)
		Herring (2028.94, 0.13)	her-2532-gor (245.28, 0.09)
			her-30 (61.63, 0.15)
			her-3a22 (121.54, 0.15)

			her-47d3 (538.76, 0.07)
			her-irls (20.81, 0.05)
			her-noss (1202.19, 0.2)
			her-riga (33.45, 0.12)
			her-vasu (92.75, 0.09)
			her-vian (79.26, 0.06)
		Horse Mackerel (260.41, 0.09)	hom-west (259.44, 0.09)
		Mackerel (697.43, 0.07)	mac-nea (696.15, 0.07)
		Saithe (366.03, 0.04)	sai-3a46 (160.77, 0.06)
			sai-arct (193.75, 0.08)
			sai-faro (53.57, 0.15)
			sai-icel (68.40, 0.09)
		Sandeel (646.50, 0.08)	san-ns1 (357.36, 0.09)
			san-ns2 (72.04, 0.11)
			san-ns3(259.59, 0.09)
		Sardine (149.70, 0.10)	sar-soth (150.34, 0.1)
		Sprat (378.03)	spr-2232 (375.41, 0.13)
		Blue whiting (1254.76, 0.16)	whb-comb (1241.12, 0.16)
			whg-47d (37.18, 0.14)
		Whiting (39.74, 0.09)	whg-7e-k (13.09, 0.08)

Table 2. Data used to parameterize the bioeconomic model. Catch and value were built combining data from international databases (FAO and Sea Around Us). Cost parameters are from Lam et al (2011). Additionally, the net profits of EU member states as reported in the Annual Economic Report on the European Fishing Fleet (JRC, 2012).

Country	Catch (t) (2010)	Value of catch (MEur) (2010)	Operational costs (Eur/t)	Total Costs (Eur/t)	Estimated Op. profits (MEur)	Estimated Net Profits (MEur)	*Net Profits (AER, 2010)*
Belgium	21,907	78.40	1,007.9	2,139.1	56.32	31.54	*-8.2*
Denmark	827,936	309.88	1,001.4	1,147.4	-519.23	-640.12	*-34.1*
Estonia	89,752	32.61	765.2	936.1	-600.90	-742.46	*8.1*
Finland	121,169	24.54	980.1	1,176.7	-63.42	-81.07	*1.2*
France	312,162	731.22	1,053.5	1,259.9	603.58	578.57	*-5.5*
Germany	193,536	211.31	1,108.4	1,324.9	-134.68	-202.26	*2.0*
Iceland	1,057,988	697.00	1,119.6	1,352.6	480.32	435.23	-
Ireland	285,527	63.65	981.6	1,169.0	-974.82	-1,173.11	*-33.4*
Latvia	77,085	14.72	644.0	803.1	-169.16	-214.59	*4.5*
Lithuania	21,371	19.92	673.0	828.5	-31.96	-43.95	*8.2*
Netherlands	283,377	49.83	1,004.8	1,206.4	28.36	24.05	*4.6*
Norway	2,555,186	5,027.08	610.7	733.8	4,854.01	4,819.15	-
Poland	113,579	223.46	874.1	1,067.1	-2,009.95	-2,503.27	*30.9*
Portugal	201,730	102.33	801.4	949.8	11.31	-5.55	*-38*
Russian Fed	997,827	1,305.52	1,276.0	1,533.7	1,048.10	996.12	-
Spain	399,448	245.59	1,211.3	1,427.4	-963.10	-1,178.74	*-250.2*
Sweden	210,552	137.83	567.3	757.6	-88.77	-164.80	*-1.5*
UK	578,677	293.55	973.5	1,167.5	88.57	47.72	*74.6*
Canada	1,046,985	1,416.26	1,652.3	1,825.5	460.11	359.91	-
US	743,143	1,444.67	1,435.4	1,604.5	377.93	252.28	-

Table 3. Parameters for the bioeconomic model (Arnason, 2007; Arnason et al., 2009), primarily derived from data for 2010 (see text).

Parameter	*Explanation*		*Value*	*Units*
MSY			13.7	Mt
K	Carrying capacity		170	Mt
$G(x)_{2010}$	Biomass growth		0.283	t/yr
Y_{2010}	Fisheries catches		10.14	Mt
Value_{2010}	Value of catches		12.43	M€
Price_{2010}	Base price of catch		1.22	€/t
Profits_{2010}	Fishery profits		0.63	B€
C_{2010}	Cost of effort		11.8	M€
Price elasticity	Price elasticity		0.2	€/Mt
Schooling (b)	Schooling parameter		1	-
Ef_{2010}	Fishing effort		1	Normalized

Table A1. Necessary transformations to run the bioeconomic model using the parameters shown in Table 3 (Arnason, 2007; Arnason et al., 2009; Gordon, 1954; Schaefer, 1954).

Biological parameters	$\alpha = 4 \cdot \frac{MSY}{K};\ \beta = \frac{\alpha}{K}$
Biomass in 2010	$B_{2010} = (\alpha/2\beta) \cdot (1 - \left(1 - \frac{4\beta(Y_{2010} + G(x)_{2010})}{\alpha^2}\right)^{0.5}$
Price in 2010	$p_{2010} = Value_{2010}/Y_{2010}$
Cost of unit of effort	$c = \frac{Value_{2010} - \Psi_{2010}}{Ef_{2010}}$
Catchability in 2010	$q_{2010} = Y_{2010}/(B_{2010} \cdot Ef_{2010})$
Biomass at equilibrium	$B_{eq} = (\alpha - q_{2010}Ef_{eq}) \cdot K/\alpha$
Catch at equilibrium	$Y_{eq} = q_{eq}Ef_{eq}B_{eq}^{b}$
Price at equilibrium	$p_{eq} = p_{2010}(\frac{B_{eq}}{B_{2010}})^{b}$
Profits at equilibrium	$\Psi_{eq} = Y_{eq}p_{eq} - cEf_{eq}$