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Is the Northern European hake, *Merluccius merluccius*, management procedure robust to the exclusion of reproductive dynamics?

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ABSTRACT

The purpose of the present study was to investigate the robustness of the management procedure (MP) of Northern European hake to alternative indices of reproductive potential (RP) based on our recently improved understanding of hake reproductive biology using a Management Strategy Evaluation (MSE) framework. MSE simulations allow the testing of the robustness of conventional management systems to uncertainties or different hypothesis about underlying population dynamics. For this purpose, four different reproductive potential (RP) indices of increasing biological complexity (i.e. Spawning Stock Biomass (SSB) using constant maturity; SSB_{MAT} adding variation on maturity, Female Spawning Biomass—FSB, and Total Egg Production—TEP) were estimated and tested. The inclusion of more biological information affected the perception of the population dynamics, the biological reference points (BRPs) as well as the perception of the stock in relation to those biological reference points. In this study, the probability of a wrong perception, i.e. the "perceived" population is above reference limits while the "true" population is overfished, was 13, 5, 3 and 3% for the different RP investigated in the time period between 1978 and 2008. This probability was around 0% in all cases studied for the projected period of 2009–2040. Our results show that (i) when including more information about reproductive biology in the simulation, the number of years below the BRPs are higher in the "perceived" population than in the "true" population, (ii) for the historic period the probability of a wrong perception is diminished when alternative reproductive indices are included when simulating both, 'true' and 'perceived' populations, and (iii) during the initial years of the projected period, although the perception of the population dynamics in relation to BRP is different between SSB_{WG} scenario and alternative RP indices, the outcomes of the management advice of the SSB_{WG} are more restrictive because the wrong perception is conservative, i.e. when "true" population is above B_{pa} the SSB_{WG} is below. Thus, it can be concluded that the MP for European hake is robust to the different hypothesis about alternative RP indices. However, the results of the present study also showed that for the historic period, when the population level was close to BRP level, the perception of the stocks status differs between alternative RP indices. Thus, it would be convenient to include the reproductive biology of the species in the Harvest Control Rules (HCR) and Long Term Management Plan (LTMP) of this stock, especially when the level of the stock is close to BRP level.

1. Introduction

One of the main objectives of any fishery management policy is to maintain adequate rates of stock regeneration to offer resilience to long-term exploitation and associated fishing mortality (Nash et al., 2009; Shelton et al., 2006). This general management objective is achieved by insuring that stock reproductive potential (SRP), being usually considered as the spawning stock biomass (SSB), is kept above a biomass reference limit, below which recruitment may be impaired, and also that fishing mortality is lower than a level (i.e. exploitation reference limit) that would reduce the RP to below this reference limit (Hauge et al., 2007; Kell et al., 2005). Thus, the reproductive biology of any species becomes of crucial importance as it determines the productivity, the resilience of the species to exploitation and its capability of recovery (Morgan et al., 2009). In this context, the stock–recruitment relationship is considered a cornerstone in fisheries management as the biomass reference points are usually dependent on stock–recruitment models (Hilborn and Walters, 1992). Generally, it is assumed that the number of recruits is a function of the RP expressed as adult biomass or SSB (Trippel et al., 1997). This implies a proportionality between SSB and the reproductive potential of the stock (Marshall et al., 2003; Trippel, 1999), which entails, among other implications, that the relative egg production per unit weight of

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mature stock is constant over time (Marshall et al., 1998, 2003) and that production/survival rates of offspring are independent of parental age, body size, and/or body condition (Cardinale and Arrhenius, 2000). Similarly, the estimation of SSB does not commonly incorporate important biological characteristics such as sex ratio, variable maturity ogives, and fecundity. Different studies suggest that the age structure of the population may affect the fecundity and the spawning success of the population (Cardinale and Arrhenius, 2000; Marteinsdòttir and Thorarinsson, 1998) and that the maturity/fecundity of the individuals may exhibit temporal variability (Kraus et al., 2002; Murua et al., 2006). Recently, Morgan et al., 2009, investigated the impact of alternative indices of reproductive potential on the perception of population productivity and dynamics of different commercially harvested North Atlantic fish populations. They concluded that population status in relation to reference points, population productivity and future prospects of stock dynamics and recoveries were dependant on the different reproductive potential index used underlining the importance of including more biological information in the fishery assessment/management process.

The use of a Management Strategy Evaluation (MSE) approach has been widely recognized as valuable tool to test the robustness of Management Procedures (MP) to the uncertainties in the fishery system (Kell et al., 2007). Francis and Shotton (1997) described the sources of uncertainties that arise in the fishery system and categorized them in six groups. The MSE approach considers the interrelation between stock-fleet-management as the dynamic system to be studied (A'mar et al., 2009; Butterworth et al., 1997; Dichmont et al., 2006, 2008; Kell et al., 2004, 2005, 2007; Punt and Smith, 1999) and can be used to evaluate the effect of different type of uncertainties and/or hypothesis on the success of a MP. The MSE simulation approach involves the development of an operating model (OM) to describe the underlying "true" dynamic of the stock and the fishery through time together with a MP model which describes the whole management process from data collection to the management advice (Kell et al., 2007; Kell and Fromentin, 2007). MSE simulations allow testing the robustness of conventional management system to uncertainties or dynamics not considered. For example, the robustness of the management system to more biological realistic dynamics of the real population, which may be the cause of a failure in the MP (Kell et al., 2007).

The population size of the northern stock of European hake, which encompasses ICES Div. IIIa, SubAreas IV, VI and VII and Div. VIIIabd, declined during the late 1990s, and the present level in 2009 is only 40% of the historic level of the 1970s. In 2004 European hake northern stock was considered to be at risk of reduced reproductive capacity and harvested unsustainably (ICES, 2009). Thus, a Recovery Plan was implemented for the northern stock of European hake in 2004 (EC Reg. No 811/2004), with the objective of increasing the SSB to above the precautionary level (140,000 t), establishing a fishing mortality lower than 0.25 constrained by a year-to-year change in Total Allowable Catch (TAC) not exceeding 15% provided that SSB is above 100,000 t (ICES, 2009).

García et al. (2007) showed that the Harvest Control Rules (HCR) implemented for the recovery of the stock yielded the desired results, although those rules should not be maintained after achieving the target spawning biomass. In evaluating the recovery of the European hake northern stock, García et al. (2007) considered process uncertainty in recruitment and process and observation uncertainty in the indices of fish abundance. Operating models using different stock–recruitment models were also tested and, although the results differed, the conclusion about the suitability of the Recovery Plan was the same for different scenarios investigated.

Northern hake stock assessment is presently carried out using eXtended Survivors Analysis (XSA; Shepherd, 1999) where the spawning stock biomass (considered as a proxy of RP) is derived using a constant sex aggregated maturity ogive (Martín, 1991) without accounting for variability in population egg production (i.e. fecundity) (ICES, 2007). However, the dynamics of SSB can also be affected by biological variability in processes such as growth, sex ratio, sexual maturation, and fecundity but still the variability of these processes have so far not been taken into account. Examples of recently changed perception of biological processes in European northern hake that suggest that SSB may be not a good proxy of RP include that of growth due to bias in age estimation (Bertignac and de Pontual, 2007), decreasing trends in reported size at maturity (Domínguez-Petit et al., 2008), and inter-annual variability in fecundity (Murua et al., 2006).

The main objective of this paper is to investigate the effect of biological characteristics that influence the reproductive potential of the northern European hake and, thus, its population dynamics and assessment. Here we study the effect of alternative reproductive potential indices on the dynamics and recovery prospects of the stock, using a more biologically "realistic" data of sex ratio, variable maturity and fecundity. Also, we evaluate the influence of different reproductive potential indices on the stock–recruit relationship and the resulting Biological Reference Points (BRP) as well as on the status of the stock in relation to those BRPs. Thus, the robustness of northern European Hake MP to alternative indices of stock reproductive potential was tested. For this purpose, the MSE simulation algorithm developed in Fisheries Libraries for R (FLR) (Kell et al., 2007) for northern stock of European hake (García et al., 2007, 2008), which includes a biological and fisheries operating model and a MP, has been parameterize in order to account for biological variability in processes determining the magnitude of hake reproductive potential. In the MSE algorithm the "true" population is simulated using alternative indices of reproductive potential while in the MP the perceived population is calculated as in the ICES assessment. The present work develops and extends the preliminary work carried out on hake by Morgan et al. (2009) by evaluating, not only the inclusion of additional biological complexity into RP indices, but also the robustness of actual MP to different RP indices.

2. Material and methods

List of abbreviations and additional formulae are provided in Appendix A.

2.1. The simulation algorithm

The FLR framework (http://www.flr-project.org, Kell et al., 2007) provides a basis to model population dynamics as part of MSE algorithms. The algorithm used in this work is divided into different modules that simulate the various processes in the fishery system. The two main modules of the algorithm are the operating model (OM) which simulates the 'real or true system' (i.e. the biological population and the fishery, their interaction and the implementation of the management advice) and the MP model which simulates the management process (Fig. 1). This simulation algorithm explicitly or implicitly acknowledges different sources of uncertainty in both the "real" system and the management procedures (Francis and Shotton, 1997; Kell et al., 2007; Rosenberg and Restrepo, 1994).

The simulation algorithm takes an initial population, presently from 1978 to 2008, and projects it into the future. The real biological population and fishery are projected, in yearly time steps, using the OM and the MP is applied annually to produce the management advice for the next year. The advice obtained within the MP for a certain year constrains the behavior of the fleets in the next year. The iterations are run in parallel without any interaction.

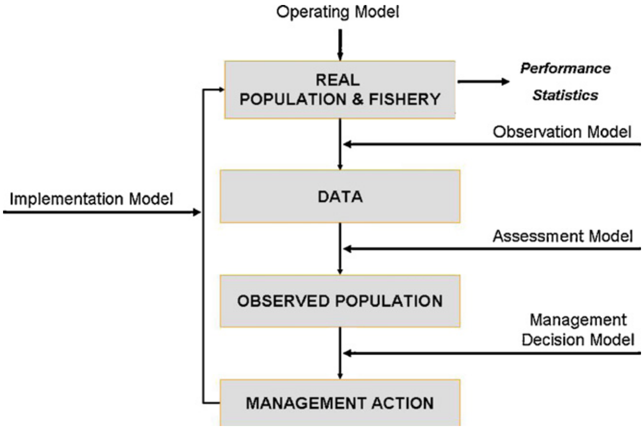

Fig. 1. The simulation is divided into two time periods, (i) the historic state which is conditioned to observed data and (ii) the projection which includes all the models described above.

2.1.1. The initial population

Assuming that the main source of error in Northern Hake assessment is the error in the tuning abundance indices observed; the initial random population is generated adjusting the XSA to the ICES Working Group (WG) datasets (using the same model settings) but replacing the observed abundance indices by a set of boostrapped abundance indices. To generate the random abundance indices, first an XSA is adjusted to the observed landings-at-age matrix and observed abundance indices and the standard error of the log-residuals of the abundance indices are calculated. Then in each of the iterations a random lognormal error, $LN(0, \sigma)$ is multiplied to the original abundance indices, being σ the standard error of the residuals obtained in the initial fit. In this way two random arrays of numbers- and fishing mortalities-at-age are obtained from 1978 to 2008, the years for which landings-at-age data are available to the assessment working group for this stock. The historical effort is assumed to be equal to 1 and, thus, the historical catchability was set equal to the fishing mortality ($F_a = q \times E_a$).

2.1.2. The operating model

The OM consists of an age-structured biological population and a single fishery inducing fishing mortality during the harvesting process. The OM starts in 2009 and for this year the population numbers of age 1 and older are calculated using the numbers and fishing mortalities obtained in the generation of the initial population while the recruits at age 0 are estimated using a Ricker stock–recruitment (SR) relationship. A sensitivity exercise was carried out using different stock recruitment relationships (Beverton and Holt (1957) and segmented regression). AIC model selection indicated that the Ricker model best explained the relationship between RP and recruitment (García et al., AZTI, personal communication). The stock–recruitment relationship was fitted using the numbers at age 0 in the initial population and the different reproductive potential indices investigated. One SR relationship was fit in each of the iterations. In subsequent years, given the matrix of numbers-at-age of the previous years, the population numbers are carried forward using the exponential survival equation and a predefined SR relationship with a lognormal multiplicative random error to generate the recruits (for details see Appendix A). As the real TAC for 2009 is known, it is assumed that in 2009 the fleets will comply exactly with this TAC. The catchability of the fishery is an input parameter and it is assumed constant over time, although it varies through iterations. The effort of the fleet is updated in each year and iteration and is calculated using the Baranov catch equation (Baranov, 1918), constraining it to produce exactly the TAC, given the catchability- and the numbers-, weights- and natural mortality-at-age of the real population.

2.1.3. Reproductive potential indices

Four different reproductive potential index scenarios of increasing biological complexity were used as population RP proxies in the OM to compare the performance of "true" and "perceived" populations in relation to BRPs (see below). The SSB without additional biological information as used by the WG assumes that all reproductive characteristics are constant. The different RP indices were:

$\mathrm{SSB_{WG}}$, estimated as in the ICES WG using a constant sex aggregated maturity ogive

$$\mathrm{SSB_{WG}} = \sum_{a=i}^{j} N_{ay} W_{ay} M_a$$

where N_{ay} is the population number-at-age a in year y, W_{ay} the weight-at-age a in year y, M_a is the proportion mature-at-age a. The age range is the one used in the sequential population analyses of the population.

$\mathrm{SSB_{MAT}}$, estimated incorporating the female variable maturity-at-age for each year.

$$\mathrm{SSB_{MAT}} = \sum_{a=i}^{j} N_{ay} W_{ay} M_{ay}$$

where M_{ay} is the proportion of mature females-at-age a in year y.

Female Spawning Biomass (FSB) incorporating the yearly variable sex ratio-at-age with the yearly variable proportion of female maturity-at-age

$$\mathrm{FSB} = \sum_{a=i}^{j} N_{ay} W_{ay} M_{ay} R_{ay}$$

where R_{ay} is the proportion of females-at-age a in year y.

Total Egg Production (TEP), incorporating a proxy for realised fecundity-at-age:

$$\mathrm{TEP} = \sum_{a=i}^{j} N_{ay} M_{ay} R_{ay} E_{ay}$$

where E_{ay} is the fecundity ogive-at-age a in year y (for details of calculation see Section 2.3.3). In TEP estimation, the weight-at-age is included because it forms part of the estimation of E_{ay} (i.e. fecundity-weight relationship were applied to mean weight-at-age to estimate $E_{ay.}$).

2.1.4. The management procedure

The MP is applied on 2009 and produces the TAC advice for 2010 based on the Harvest Control Rule (HCR) proposed in the Long Term Management Plan (LTMP). The same HCR is applied every year up to 2039. The MP model is divided into three sub-models: (i) the observation model which simulates the data collection and thereby links the MP with the OM, (ii) the assessment process model that generates a 'perceived' population based on the data, including the assessment model and the short term forecast; (iii) the management decision model which uses an HCR based on the perceived population and estimated reference points to derive management advice (Fig. 1).

In the observation model the catch-at-age is considered to be known without error and the abundance indices are generated with a multiplicative random error assuming a linear relationship between catchability and abundance-at-age. The RP index used within the MP is the $\mathrm{SSB_{WG}}$ as used in the ICES WG.

The TAC for year "y" is set based on the assessment carried out in previous year ("$y-1$") with assessment data available up to year "$y-2$". The stock assessment therefore produces fishing mortality estimates for year "$y-2$" and population estimates for the beginning of year "$y-1$" (except for the recruitment). To estimate the fishing mortality in "$y-1$" it is assumed that the fleet catches exactly the TAC set for year "$y-1$" with the selection pattern equal to the average of that observed between years "$y-4$" and "$y-2$". The recruitment in years "$y-1$" and "y" are assumed to be equal to the geometric mean recruitment of the previous 20 years estimated for the "perceived" population, excluding the most recent two years due to the retrospective pattern (as done by the ICES WG).

The following HCR is then applied to calculate the desired fishing mortality for any year y:

$$\tilde{F}_y = \begin{cases} F_{tg} & (1-\alpha)\cdot F_{tg} < \hat{F}_{y-1} < (1+\alpha)\cdot F_{tg}. \\ (1-\alpha) & (1+\alpha)\cdot F_{tg} < \hat{F}_{y-1}\, and\, (1-\alpha)\cdot \hat{F}_{y-1} < F_{PA}. \\ (1+\alpha) & (1-\alpha)\cdot F_{tg} > \hat{F}_{y-1} \\ F_{PA} & (1-\alpha)\cdot F_{y-1} > F_{PA} \end{cases}$$

The target fishing mortality (F_{tg}) in the HCR was set to be equal to $F_{\max} = 0.17$, which was estimated by Scientific, Technical and Economic Committee for Fisheries (STECF) as a proxy for F_{MSY} in the biological evaluation of the LTMP (SEC, 2007a), with an annual maximum variation in fishing mortality equal to 10% (α). The resulting fishing mortality is then converted to catch in order to give TAC advice using the Baranov catch equation and using the perceived population parameters (weigths-, numbers- and selectivity-at-age).

2.2. *The parameterization or conditioning of the operating model*

2.2.1. *The projection*

In each scenario of RP indices, the biological population and fishery were projected until 2040 and for 1000 iterations. Thus, the last management process is run in 2039 and the last 'perceived' population is obtained for 2038.

In the projection of the "true" biological population, maturity, weight, sex-ratio, and fecundity were considered as the mean value of the last three years of the historical values. Regarding natural mortality a constant value of 0.2 was used for all ages and years.

The RP, based on the different reproductive potential indices explained above, determined the number of recruits for the simulated population in the next year class using the S/R model. For each of the RP indices stock–recruitment relationships were fitted using the Ricker model for the 1000 iterations of the initial random population. The parameters and associated variances obtained from fitting the S/R relationships were used to parameterize the S/R model in each of the iterations.

Regarding the parameterization of the fleet, the catchability was set equal to the average catchability between 2007 and 2008 obtained in the initial population. Thus the catchability was constant along years but variable between iterations; as the effort and, hence, fishing mortality was updated in each year and iteration as described in the previous section. In the observation model, the catchability and the standard error of the abundance indices were equal to those obtained in the generation of the initial population and the catch-at-age matrix was taken directly from the fishery without error.

2.2.2. *Biological reference points (BRPs)*

BRPs were estimated in order to compare stock status trajectories and prospects of recovery for the four different scenarios of alternative RP indices, following the ICES WG approach. The WG estimated B_{lim} as the lowest observed biomass in the whole time series (B_{loss}) in 2003 assessment, which corresponds to the

Table 1
Details of the number of hake sampled from 1987 to 2008 to provide biological information (length, sex ratio, maturity and age composition) from the Northern hake fishery used for the operating model.

	Year																	
	1987	1988	1989	1990	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	Total
Min Length (cm)	10	10	9	8	9	12	13	13	13	9	9	25	13	18	26	25	23	
Max Length (cm)	57	65	76	104	104	95	88	79	72	80	85	86	88	84	76	75	88	
No. Sex ratio	491	1448	2802	2219	3924	2476	1909	2083	2336	2073	2125	2257	2048	1534	926	1187	1396	33501
No. Maturity	491	1385	2761	2219	2474	1804	1369	1583	1982	1874	1892	2144	1811	1059	821	952	1081	27945
No. Age	491	1449	3186	2220	3924	2476	1909	2083	2336	2265	2467	2582	2258	1736	926	941	1073	34591

biomass observed in 1994 (ICES, 2009). As such, B_{lim} for alternative RP indices was estimated as B_{loss}. B_{pa} was estimated as being $=B_{\text{lim}} \times e^{1.645 \times 0.2}$ as in the WG (for details of B_{pa} estimation see ICES, 2004; Hauge et al., 2007).

2.3. Data

Data for northern European hake stock were used as reported by the WG (ICES, 2007).

2.3.1. Sex ratio

Sex-ratios for ICES Div. VIIIabd were available for the period 1987–1990 and 1996–2008 and obtained from extensive yearly market sampling carried out in the Basque Country fishing harbours (Table 1). The main fishing fleet targeting European hake in ICES Div VIIIabd was covered using this market sampling, which corresponds to around 15–20% of the total catch of the whole stock (ICES, 2009). In addition to the market sampling, samples obtained from observers onboard commercial fishing vessels within BIOSDEF and DEMASSES EU projects were also used to estimate sex-ratios. Working Group data extended back to 1978, therefore, 3-year averages of the nearest years (1987–1989) were used to extend sex ratios back to 1978 assuming a constant sex ratio from 1978 to 1986. For the period of 1991–1995, where sex-ratio was not available, the average between previous and posterior 3 years (1988–1990 and 1996–1998) was used.

2.3.2. Maturity-at-age

Maturity-at-age data were also available between 1987–1990 and 1996–2008, and were obtained using the same approach as for sex ratio.

2.3.3. Fecundity-at-age

As European hake is considered to show an indeterminate fecundity (Murua and Motos, 2006) no accurate estimate on its total egg production within a given year can be obtained (Murua et al., 2003). Thus, the batch fecundity estimated for the Bay of Biscay hake was used as a proxy for fecundity of this species. This assumes that the number of batches does not vary either with age or year, although hake spawns between 6 and 12 batches per year (Murua et al., 1998, 2006). Therefore, the inclusion of only one batch for the estimation of TEP is considered an underestimation. Batch fecundity–weight relationships were applied to mean weight-at-age to estimate batch fecundity-at-age for all time series. Then, a batch fecundity ogive was estimated, scaling batch fecundity-at-age to the higher batch fecundity-at-age in the whole age/year matrix. In other words, a fecundity ogive was produced, showing a percentage of the maximum batch fecundity production by age, which will allow the estimation of the TEP with the same units as SSB (tonnes).

$$E_{ay} = \frac{BF_{ay}}{Maximum(BF_{ay})}$$

Batch fecundity-weight relationships were available between 1994–1998 and 2001–2006. Batch fecundity–weight relationships outside these periods were derived by pooling the data for all years in which data were available.

3. Results

3.1. Biological variables

Female maturity ogives for European hake showed a trend to earlier maturation during the period examined (Fig. 2a). The inter-annual variation around ages 4 and 5, when most females become mature, was considerable. For example, female age at 50% maturity decreased by almost 1 year over 18 years: occurring around age 5.1 in 1989 to 4.2 in 2007. From these values it is clear that the maturity ogive varied over time whilst the ICES Working group assessment assumed a constant ogive (Fig. 2b). This figure also shows that there are clear differences in the maturity schedule when (i) female only estimation is used and (ii) temporal variation is also considered.

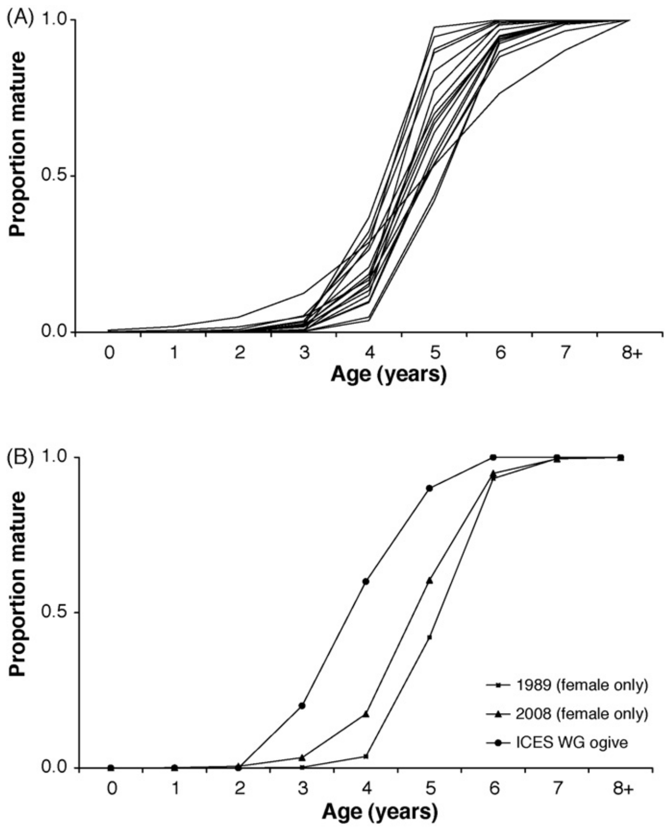

Fig. 2. (a) Female maturity ogives over the historical time series studied and (b) proportion of sex-combined mature at age data from the ICES WG and female mature-at-age estimated from improved biological information in 1989 and 2008.

The proportion of females in each age group, fluctuated around 50% for ages 0 to 4 then increased to around 75% for ages 6 and 7, before approaching 100% in older age classes (Fig. 3). At age 4 the sex-ratio in some years fluctuated between 0.2 and 0.7; whereas in younger ages the fluctuation is less variable between 0.4 and 0.6. The observed values for the proportion of females by age show a clear increasing trend from age 5 onwards, reflecting a differen-

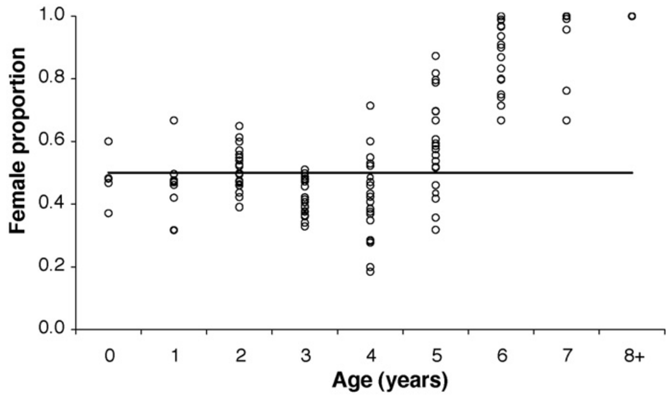

Fig. 3. The proportion of females by age plotted against the midpoint of each age classes.

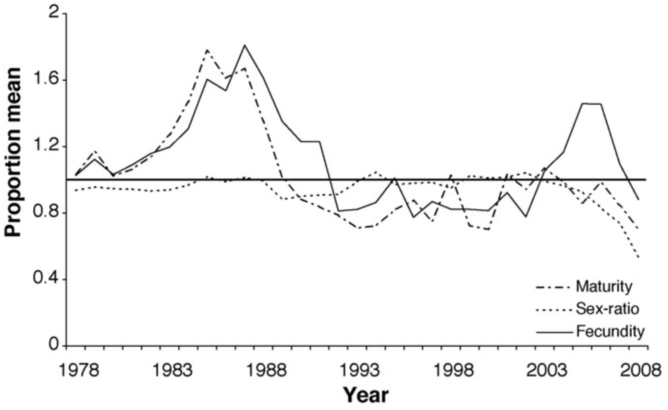

Fig. 4. Time series of standarized weighted mean by population number at age for the proportion of mature, proportion of females and fecundity.

tial longevity and probably growth, maturity and mortality pattern between females and males. Fig. 3 also shows that the sex-ratio varies notably between years.

In order to see the degree of variation of the different reproductive characteristics at the population level, the weighted mean by population number at age of proportion mature, proportion female, and fecundity were calculated for each year. The most variable reproductive characteristic was the fecundity followed by the female maturity ogive, whilst the least variable was sex-ratio showing a maximum change between years of less than 10% for most of the time series with the exception of recent years when the observed changes were between 25 and 40% (Fig. 4).

3.2. The stock–recruitment relationship

Resulting mean parameter values for the stock–recruit relationship are presented in Table 2. Although the Ricker stock–recruitment models were visually very similar (Fig. 5), the variance of the recruitment explained by different reproductive potential indices was different (Table 2). For example, the variance of the recruitment explained by the model was highest for the SSB_{WG}/R relationship which resulted in the lowest corresponding AIC value, followed closely by the TEP/R relationship. For the lowest levels of biomass the estimates of recruitment for SSB_{MAT}, FSB, and TEP are slightly higher than for the SSB_{WG}, while for larger levels of biomass the reverse is true (Fig. 5).

3.3. "True" vs. "perceived" population

Time series of "true" fishing mortality, recruitment and RP were compared to the "perceived" values for each scenario using the alternative indices of reproductive potential, ie. SSB_{WG}, SSB_{MAT}, FSB, and TEP (Figs. 6 and 7). With regard to fishing mortality, in all scenarios, the general trend is comparable and the resulting figures are quite similar between the "true" and "perceived" population for each of the reproductive potential indices. Nevertheless, some bias can be observed in fishing mortality between "true" and "perceived" populations. Fishing mortality, especially in those years when the stock was declining to the lowest values observed (i.e. 1989–1998), was generally underestimated by the "perceived" population in the MP (Fig. 6a). The biomass trends of the different reproductive potential indices were generally similar. However, in all cases, it appears that the precision of the estimated values for each scenario is greater in the "true" population than in the "perceived" population (Fig. 7). All scenarios indicated a large decrease in biomass in the mid-eighties and early nineties, from the peak in 1985 to the lowest observed biomass in 1999 for the "perceived" and "true" population for SSB_{WG}, SSB_{MAT} and FSB scenarios, and in 2000 and 2001 for the "perceived" and "true" TEP populations, respectively. Since then, all time series showed an increasing trend of similar magnitude reaching a level above the B_{pa} by 2008 for all RP scenarios investigated either for "perceived" or "true" populations. For the projected time period from 2009 to 2040, all scenarios investigated showed a similar increasing trend reaching a maximum level of biomass around 2020 after which the population stabilizes around equilibrium showing that the F_{tg} established in the LTMP is a good proxy for F_{MSY}. However, for the TEP scenario, once the maximum level of biomass is reached, the population does not stabilize but shows a fluctuating pattern, which is mainly due to the use of the Ricker stock–recruitment model where recruitment declines at larger population sizes (Fig. 7). As biomass increased to high levels, the recruitment (Fig. 6b) diminished and, consequently, the population numbers/biomass decreased again as the overall fishing mortality is maintained constant in the LTMP; indicating that the F_{tg} established in the LTMP may not be appropriate for the TEP scenario. In fact, the F_{MSY} estimated for TEP is around 0.26 which is significantly higher than the F_{tg}.

There were differences with regard to the first year in which overfishing ($\text{RP} < B_{\text{pa}}$) was observed. For example, 1994 was the first year that the population was estimated to be below B_{pa} for the "true" population of SSB_{WG}, SSB_{MAT} and FSB, whereas it was 1995 for TEP (Fig. 7). The same happened when the population was estimated to have recovered. In the first instance, the "perceived" population for all scenarios and the "true" population for SSB_{WG} were estimated to have recovered by 2004 whereas it was 2002 for "true" SSB_{MAT} and FSB and 2001 for "true" TEP.

The percentage of years, both for historic and projected time series, in which the population status was below B_{pa} differed between the "true" and the "perceived" populations (Table 3). For example, the number of years for which the population was below B_{pa} was 21 (33% of the time series) for the "true" SSB_{WG} and 16 (25%), 12 and 12 (19%) years for the alternative indices of SSB_{MAT}, FSB, and TEP, respectively; whereas it was 23 (37%) for "perceived" SSB_{WG} and 22 for "perceived" SSB_{MAT}, FSB, and TEP. While the number of years for which the population was below B_{pa} was almost the same for SSB_{WG} the difference between the "true" and "perceived" populations was significant for SSB_{MAT}, FSB, and TEP. Accordingly, in all cases it appears that the assessment did not correctly "perceive" that the population was within safe biological limits (Table 3).

For the historic time series (1978–2008), both the "perceived" and "true" populations are above B_{pa} 57, 66, 67, and 68% of the

Table 2
Ricker stock–recruit parameters from different reproductive potential indices used in this study SSB_{WG}, SSB_{MAT}, FSB, and TEP.

Parameter	SSB_{WG}	SSB_{MAT}	FSB	TEP
R^2	0.46	0.32	0.31	0.43
Alpha	2.71	3.31	3.96	3.46
Beta	4.15E−06	5.09E−06	5.96E−06	5.30E−06
AIC	−16.79	−9.97	−9.47	−14.99

Table 3
Summary of the performance of the population relative to BRPs for each reproductive indices (SSB_{WG}, SSB_{MAT}, FSB, and TEP) for the period 1978–2040.

	B_{lim} estimated as the lowest value of the time series or B_{loss} for each alternative index.			
	SSBwg	SSBmat	FSB	TEP
B_{lim}	100000 t.	72990 t.	53515 t.	25077 t.
B_{pa}	140000 t.	101426 t.	74364 t.	34847 t.
No. years "True" $\text{RP} < B_{\text{pa}}$	21 (33%)	16 (25%)	12 (19%)	12 (19%)
No. years "Perceived" $\text{RP} < B_{\text{pa}}$	23 (37%)	22 (35%)	22 (35%)	22 (35%)

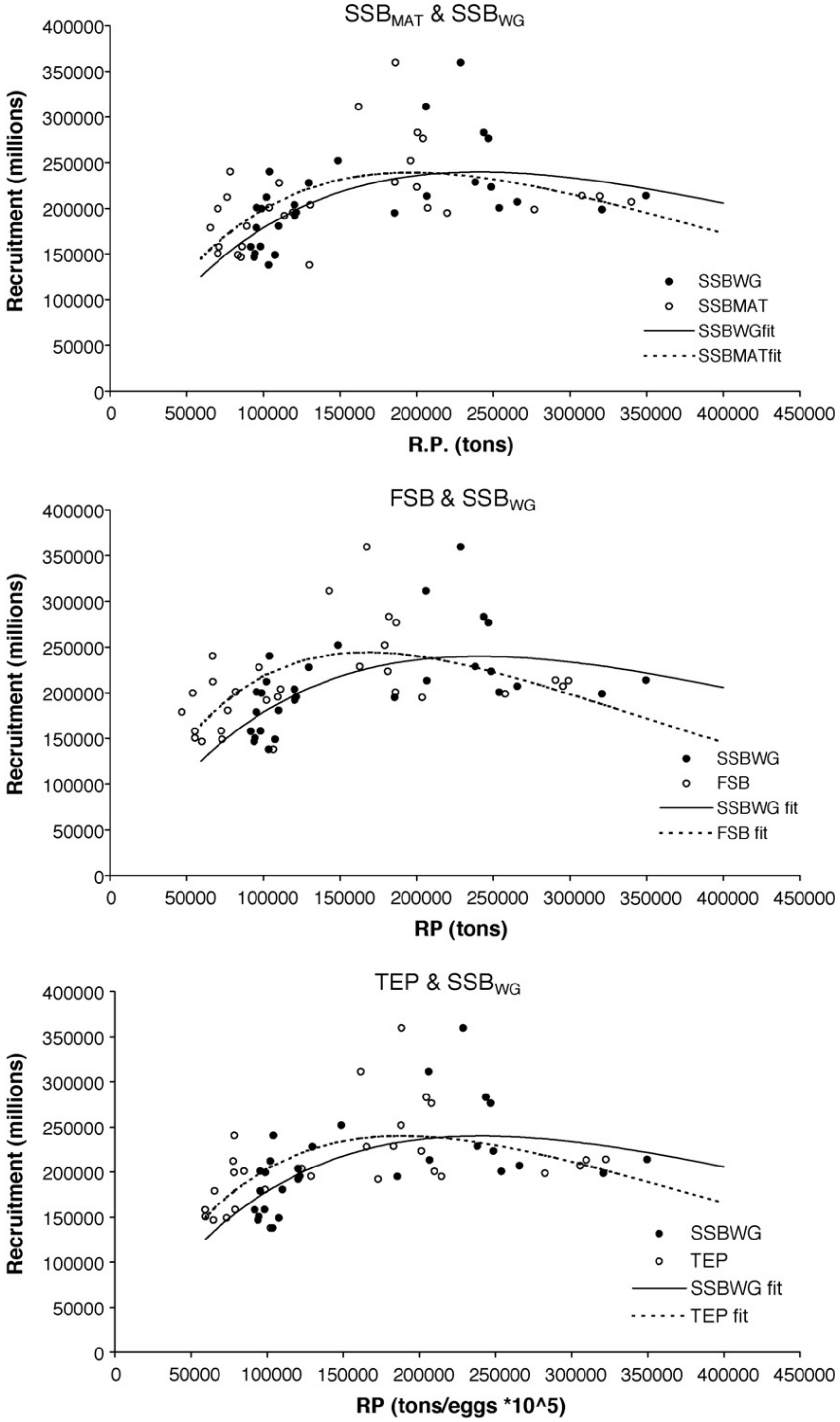

Fig. 5. Working group SSB ($\mathrm{SSB}_{\mathrm{WG}}$) and recruit pairs (filled circles), working group Ricker stock–recruitment curve (solid line), different reproductive potential indices estimates ($\mathrm{SSB}_{\mathrm{MAT}}$, FSB, and TEP) and recruit pairs (open circles), and the corresponding Ricker stock–recruitment relationship (dashed line).

time for each scenario of RP studied ($\mathrm{SSB}_{\mathrm{WG}}$ $\mathrm{SSB}_{\mathrm{MAT}}$, FSB, and TEP) (Table 4). Similarly, the MP or assessment correctly estimates when the "true" population is below B_{pa} 27, 18, 16, and 13% of the time for $\mathrm{SSB}_{\mathrm{WG}}$, $\mathrm{SSB}_{\mathrm{MAT}}$, FSB, and TEP, respectively. Nevertheless, 3, 11, 13, and 16% of the time for $\mathrm{SSB}_{\mathrm{WG}}$, $\mathrm{SSB}_{\mathrm{MAT}}$, FSB, and TEP, respectively, the perception of the working group is pessimistic while the "true" population is above B_{pa}. In this case, however, although the perception is wrong, the outcome of the assessment is more conservative. In 13, 5, 3 and 3% (for $\mathrm{SSB}_{\mathrm{WG}}$, $\mathrm{SSB}_{\mathrm{MAT}}$, FSB, and TEP respectively) of the time the results would not be precautionary because while the "true" population is outside safe biological limits the "perceived" population is within safe biological limits. In these cases, for the historic time period, the largest difference in probability of a wrong and thus not conservative approach is observed in the $\mathrm{SSB}_{\mathrm{WG}}$ scenario which corresponds with the standard RP dynamics assumed by the ICES WG; while the use of alternative RP indices in the simulation of the "true" dynamics decreased the probability of a wrong perception.

For the projected 2009–2040 time period, the probability of the "perceived" population assessing the stock status cor-

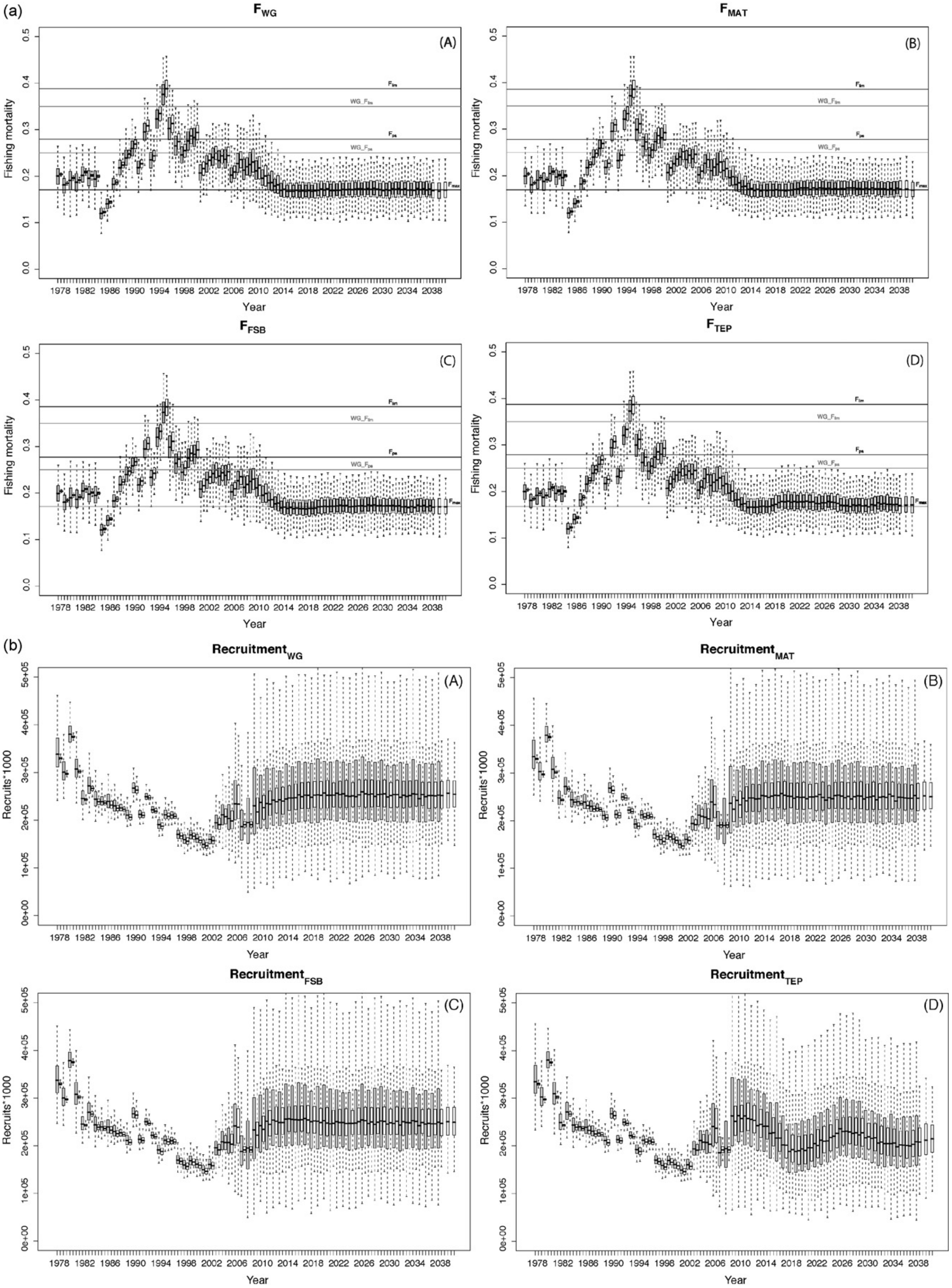

Fig. 6. Time series of (a) fishing mortality (F) and (b) Recruitment from the "perceived" population (filled box-plots) and "true" population (open box-plots); which incorporates different reproductive potential indices. Grey horizontal solid lines represent B_{pa} (140,000 t.) and B_{lim} (100,000 t.) of the "perceived" population. Black horizontal solid lines represent 'true' population BRPs for alternative indices of reproductive potential. The bootstrap 95% confidence intervals are indicated. (A) SSB_{WG}, (B) SSB_{MAT}, (C) FSB, and (C) TEP.

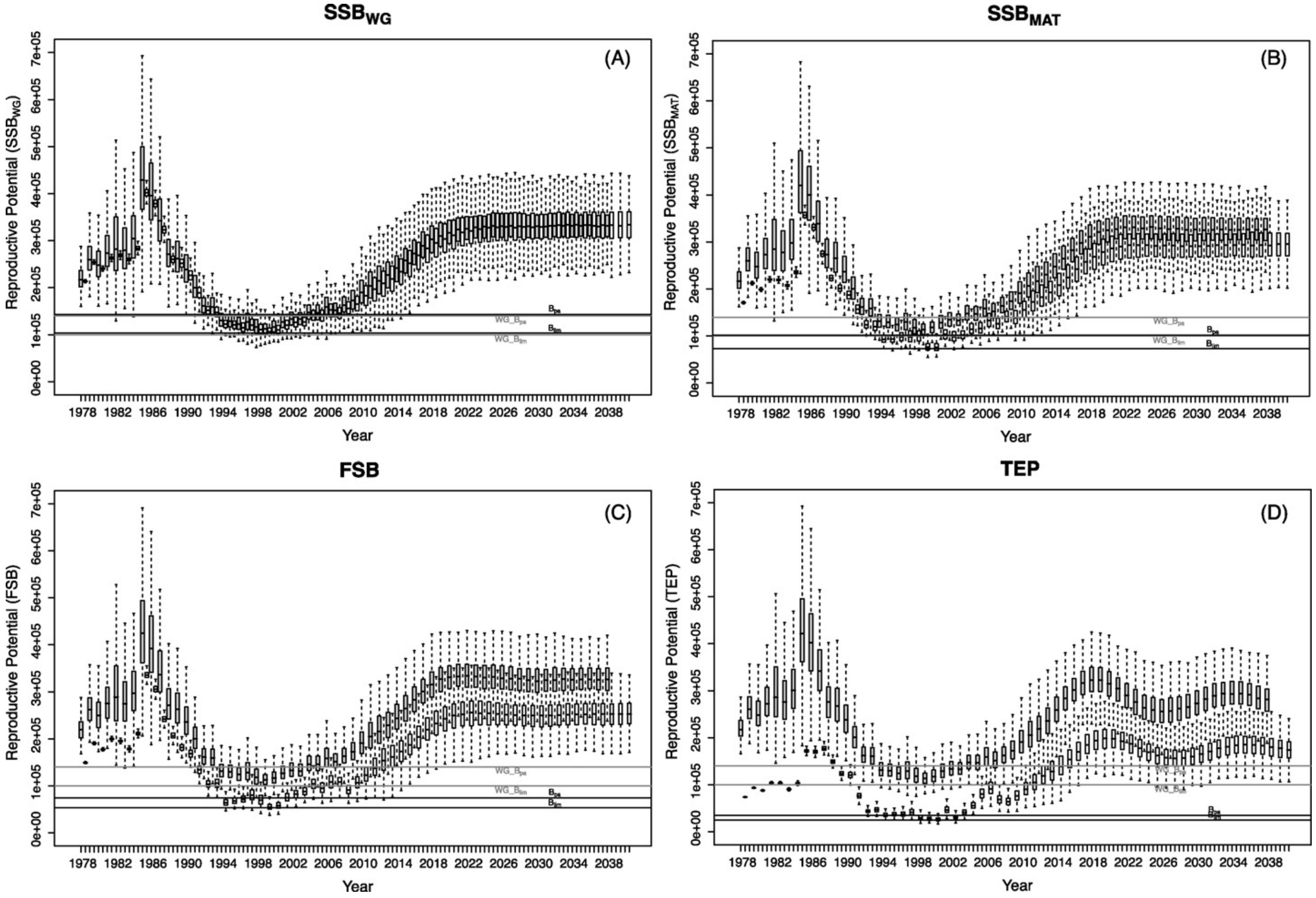

Fig. 7. Time series of SSB from the "perceived" population (filled box-plots) and "true" population (open box-plots); which incorporates different reproductive potential indices [(A) SSB$_{WG}$, (B) SSB$_{MAT}$, (C) FSB, and (C) TEP]. Grey horizontal solid lines represent B_{pa} (140,000 t.) and B_{lim} (100,000 t.) of the "perceived" population. Black horizontal solid lines represent 'real' population BRPs for alternative indices of reproductive potential. The bootstrap 95% confidence intervals are indicated.

rectly in comparison to the "true" population is close to 100% in all scenarios studied (Table 5). However, the values are smoothed by the use of a long term projection, because the main differences are observed in the first years of the projected period when the population is recovering from low values. For example, the probability of RP being below B_{pa} for the "true" scenario of SSB$_{WG}$ and the four scenarios of "perceived" population, was between 7 and 10% in 2009; whereas it was close to 0% in the "true" population of SSB$_{MAT}$, FSB and TEP (Fig. 8).

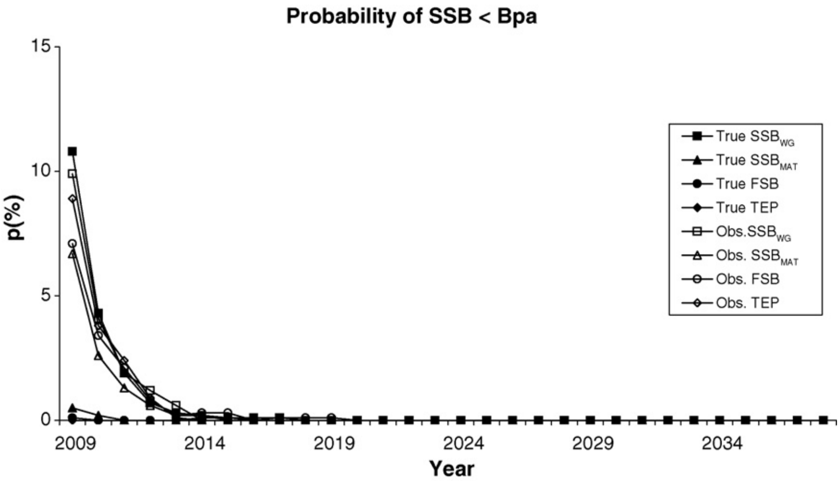

Fig. 8. The probability of being RP < B_{pa} both for "perceived" and "true" population for all scenarios investigated (SSB$_{WG}$, SSB$_{MAT}$, FSB, and TEP) for the projected period (2009-2040).

Table 4

Probabilities of RP being under or over B_{pa} (RP < B_{pa} or RP > B_{pa}) for both the “true” population based on alternative reproductive potential indices used and the “perceived” population in the historic time series from 1978-2008. The bold values represent the probability of a wrong perception and not conservative outcome (i.e. while the “true” population is outside safe biological limits the “perceived;; population is assumed to be within safe biological limits).

		SSB_{WG} “True”		SSB_{MAT} “True”	
		RP > B_{pa}	RP < B_{pa}	RP > B_{pa}	RP < B_{pa}
“Perceived”	RP > B_{pa}	57%	**13**%	66%	**5**%
	RP < B_{pa}	3%	27%	11%	18%
		FSB “True”		TEP “True”	
		RP > B_{pa}	RP < B_{pa}	RP > B_{pa}	RP < B_{pa}
“Perceived”	RP > B_{pa}	67%	**3**%	68%	**3**%
	RP < B_{pa}	13%	16%	16%	13%

Table 5

Probabilities of RP being under or over B_{pa} (RP < B_{pa} or RP > B_{pa}) for both the “true” population based on alternative reproductive potential indices and the “perceived” population in the projected time series from 2009 to 2040.

		SSB_{WG} “True”		SSB_{MAT} “True”	
		RP > B_{pa}	RP < B_{pa}	RP > B_{pa}	RP < B_{pa}
“Perceived”	RP > B_{pa}	99%	0%	100%	0%
	RP < B_{pa}	0%	0%	0%	0%
		FSB “True”		TEP “True”	
		RP > B_{pa}	RP < B_{pa}	RP > B_{pa}	RP < B_{pa}
“Perceived”	RP > B_{pa}	100%	0%	99%	0%
	RP < B_{pa}	0%	0%	1%	0%

4. Discussion

The integration of reproductive biology into fisheries management advice is of major interest to fish biologists, fisheries scientists and managers (Marshall et al., 2006; Morgan, 2008; Morgan et al., 2009). The main outcome of this study is to show that the exclusion of realistic reproductive biology in the assessment process leads to differing perceptions of population dynamics, biological reference points and stock status in relation to BRPs, which, in turn, may affect the advice for fisheries management on the northern stock of European hake. In our investigation the probability of a wrong perception, i.e., the “perceived” population is above reference limits while the “true” population is overfished, was 13, 5, 3 and 3% for SSB_{WG}, SSB_{MAT}, FSB and TEP, respectively, for the time period between 1978 and 2008. This probability was close to 0% in all cases studied for the projected period of 2009-2038. Therefore, our results indicate that (i) when including more information about reproductive biology in the simulation, the number of years below the BRPs are higher in the “perceived” population than in the “true” population (i.e. the probability of a wrong perception increased although the perception of the WG is conservative), (ii) for the historic period, the probability of a wrong perception is diminished when alternative reproductive indices are included when simulating both, ‘true’ and ‘perceived’ populations, and (iii) during the initial years of the projected period, although the perception of the population dynamics in relation to BRP is different between SSB_{WG} scenario and alternative reproductive potential indices, the outcomes of the management advice of the SSB_{WG} are more restrictive because the wrong perception is generally conservative, i.e. when “true” population is above B_{pa} the SSB_{WG} is below. Thus, for the projected period it can be concluded that the Management Procedure of European hake is robust to the different hypothesis about the alternative reproductive potential indices used here. However, for the historic period, when the population level was close to BRP level, the perception of the stocks status differs between alternative RP indices.

In the context of the MSE framework, an evaluation of the Long Term Management Procedure was carried out in 2007 (SEC, 2007b) using the same algorithm used in this study but using the biological parameters applied in the WG. The conclusion of the work presented in SEC (2007b) was that the LTMP plan was robust to the uncertainties analysed (i.e. in the SR model, in the abundance indices, and in the assessment) and the probability of SSB falling below B_{pa} was 0 after 2010; which is in agreement with the results of this study. In that sense, our results showed that the probability of falling below B_{pa} for the “true” and “perceived” population of alternative RP indices is 0 after 2009 (Table 5). However, it is worth noting that the LTMP of Northern hake is a F-based Management Plan and does not take into account the level of SSB in the HCR. In fact, within the MP the SSB does not play any role as recruitment is estimated as a geometric mean of previous year recruitment values. Nevertheless, when the historic time period and first years of the projected period (2009) are considered, i.e., the population had not totally recovered, the perception of population status in relation to BRPs was different. This showed that the probability of a wrong perception was higher in the SSB_{WG} scenario when compared with more complex RP index scenarios. As the LTMP does not take into account the SSB in the MP and the northern hake population has recovered, the effects of reproductive potential indices are diminished when evaluating the LTMP. However, if the population level of SSB should decrease again to levels around the BRPs, it would be necessary to take the reproductive potential into account in the assessment and advice. The results of the present study also show that it would be convenient to include the reproductive biology of the species (SSB or any alternative RP indices used here- SSB_{MAT}, FSB, and TEP) in the HCR of this stock. Therefore, it may be worth integrating reproductive biology in future assessment of the northern stock of European hake as has been recommended for other fish populations (Kell and Bromley, 2004; Marshall et al., 2006; Morgan et al., 2009). As concluded by De Oliveira et al. (2006) a long time series of data is needed to realise an improvement in management advice. Thus, it is important to carry out further sensitivity analysis in order to account for such issues as process and observation errors, other harvest control strategies, cost efficiency analysis, etc.

As with all simulation studies the results are to some extent conditioned on the data used and assumptions made. For example, the sampling area for maturity, sex ratio and fecundity was restricted to only a portion of the total distribution of the stock. However, as this area accounts for around 15–20% of the total catch and provides the majority of biological information used in the ICES WG assessment it was assumed that the biological data (maturity, sex ratio, and fecundity) for Div. VIIIabd are representative for the whole stock until more data becomes available.

Understanding the basis of uncertainty in the stock and recruit relationship is generally the most difficult, outstanding problem in the assessment and management of marine fish stocks (Hilborn and Walters, 1992). As stated by Morgan (2008), the poor fit of traditional stock recruitment relationships, mainly based on SSB, has stimulated the investigation of including variables which are more related to egg and larval production in recruitment models. In essence, to get back to standard spawner-recruit models developed by Beverton and Holt (1957), Ricker (1954) and Shepherd (1982) which originally used the term fecundity (Koslow, 1992; Rothschild and Fogarty, 1989) instead of SSB. However, there is not much literature regarding the improvement of S/R relationships (see Morgan, 2008). The results of this study showed that there were no clear differences in the model fit when fitting the stock recruit relationship to alternative reproductive potential indices (Table 2). In fact, the Akaike information criteria for the fits demonstrated that the best fit was achieved with an SSB_{WG} proxy followed closely by the TEP-recruitment model. Those results are similar to the ones found by Marshall et al. (2006) and Morgan (2008); who showed that alternative indices did not significantly improve the S/R relationships. These are, however, not in agreement with Kraus et al. (2002) and Murawski et al. (2001) who found an improvement in the S/R relationship when more biological information was included. Similarly, Kell and Bromley (2004) demonstrated an improvement in the S/R relationship when corrections for discards and juvenile mortality in directed and non-directed fisheries were considered. An increase in the biological complexity should not be considered only from an S/R improvement point of view, but also whether the integration of alternative indices of reproductive potential into fishery management system result in a better management advice as investigated in this work. In other words, it is necessary to answer the question of how robust the management procedure is to the exclusion of biological complexity into the management process. The MSE simulated "true" versus "perceived" populations for different scenarios described in this paper allowed us to test the management system performance in relation to the inclusion of alternative reproductive indices.

Although the focus of this paper was to investigate the robustness of Northern European hake MP to reproductive dynamics, the present study also reflects the importance of considering biological variability in the assessment. In the case of Northern European hake, the assessment considered a 1:1 sex ratio, both sex-ratio and maturation constant over time, and fecundity is not incorporated. The analysis conducted in this study, however, clearly shows that the sex ratio is skewed towards females in the larger size groups, a phenomenon observed in many other fish species (Jakobsen and Ajiad, 1999; Marshall et al., 2006; Murua, 2003). In this sense, and regardless of new findings in relation to European hake growth (De Pontual et al., 2003, 2006), it is widely recognized that males grow faster than females, up to a specific age, after which the growth rate of males decreases and female growth becomes relatively more rapid. This change in growth pattern has been associated with the onset of maturity (Lucio et al., 2000; Recasens et al., 1998). Thus, females reach a larger size and grow older than males; making the sex ratio skewed to females in the largest length classes (Casey and Pereiro, 1995). This has several implications for fishery assessment and management as pointed out by Marshall et al. (2006). Moreover, it is widely accepted that variables such as sex ratio, maturity and fecundity vary interannually for European hake. For example, Murua et al. (2006) showed that fecundity in this species varies intra- and inter-annually and Domínguez-Petit et al. (2008) reported decreasing trends in size at maturity for northern European hake in the Bay of Biscay. A similar pattern has been observed for many commercially exploited fish stocks worldwide. Morgan et al. (2009) also found, when analysing eight stocks in the North Atlantic, that the underlying reproductive characteristics such as maturity, sex ratio and fecundity vary, resulting in different population productivity. In our study, the proportion of SSB_{MAT}, considering a variable maturity ogive, when compared to the SSB_{WG} was not constant over time (Fig. 9). Instead, SSB_{MAT}/SSB_{WG} was generally lower from 1978 to 1988, increasing slightly to 1990 before decreasing for the remainder of the time period; with exceptions only in 1996 and 2002. The values ranged between a low of 0.68 in 1994 and a maximum of 1.55 in 1988. Similarly, a trend in the FSB/SSB_{WG} and TEP/SSB_{WG} relationships was observed over time. Although the pattern is similar to the one described for SSB_{MAT}/SSB_{WG}, in these cases the ratio is smaller than 1 until 2004 with exceptions between 1987 and 1990. After 2004, while the FSB/SSB_{WG} showed a decreasing trend, the TEP/SSB_{WG} increased to 2006 before decreasing to about 1 in 2008. All these changes in reproductive characteristics illustrate the importance of accounting for basic reproductive biology in the assessment and management processes.

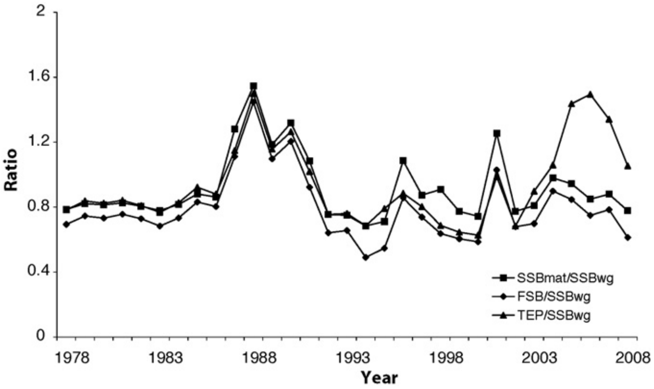

Fig. 9. The ratio of "true" population SSB (SSB_{MAT}, FSB, and TEP), estimated using alternative indices of reproductive potential, to "perceived" SSB_{WG}.

5. Conclusion

In summary, although alternative indices of reproductive potential did not greatly improve the stock recruitment relationship, the inclusion of more biological information to estimate alternative indices of reproductive potential showed that the perception of past and future population dynamics, biological reference points as well as the perception of the stock in relation to those biological reference points (i.e. stock being inside or outside safe biological limits) may be affected. In this context, our results indicated that that the Management Procedure for European hake is robust to different hypothesis about the use of alternative reproductive potential indices. Nevertheless, the results of the present study also showed that for the historic period, when the population level was close to BRP level, the perception of the stocks status differs between alternative RP indices. Thus, it seems sensible to consider the reproductive biology of the species (SSB or any alternative RP indices used here- SSB_{MAT}, FSB, and TEP) in the HCR of this stock, especially when the level of the stock is close to BRP level. The use of the Management Strategy Evaluation simulation framework can be regarded as a valuable tool for testing the suitability of including more biological "realism" into assessment.

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Appendix A.

List of abbreviations

BRP	Biological Reference Points
B_{lim}	Biological Limit Reference Point
B_{loss}	Lowest observed Biomass in the whole time series
B_{pa}	Biological Precautionary Approach Reference Point
F	Fishing Mortality
F_{max}	The value of F that maximises the yield per recruit
F_{MSY}	Fishing mortality that produces the Maximum Sustainable Yield (MSY) in the long term
F_{tg}	Target Fishing Mortality
q	Catchability
FSB	Female Spawning Biomass
HCR	Harvest Control Rule
ICES	International Council for the Exploration of the Sea
LTMP	Long Term Management Plan
MP	Management Procedure
MSE	Management Strategy Evaluation
OM	Operating Model
RP	Reproductive Potential
SPR	Stock Reproductive Potential
SR	Stock Recruitment relationship
SSB	Spawning Stock Biomass
STECF	Scientific, Technical and Economic Committee for Fisheries
TAC	Total Allowable Catch
TEP	Total Egg Production

Historic Initial Population

(a) Datasets and settings:
ICES WGHMM2009 Datasets and XSA settings.

(b) Source of variability:
Parametric bootstrap of indices, using a log-normal distribution, $LN(0, \sigma_{ind,a})$, where $\sigma_{ind,a}$ were equal to the standard error of the log-residuals obtained from an initial XSA fit to the observed indices and catch at age matrix with ICES working group XSA settings.

(c) Population:
Numbers at age and fishing mortality at age arrays from 1000 XSA fits with ICES datasets and settings but replacing tuning indices by the bootstrapped counterparts.

Historic RP indices produced calculated as explained in the text for $\mathrm{SSB_{WG}}$, $\mathrm{SSB_{MAT}}$, FSB, and TEP.

Real population projection

(a) Reproductive Potential indices (RP)

$$\mathrm{RP}_{a,y,i} = \mathrm{RP}_a = \overline{RP}_{(2006-2008)_a}$$

(b) Stock Recruitment relationship (SR)

$$\mathrm{SR}_{y,i} = \mathrm{SR}_i = \mathrm{Ricker}\,(1978-2006)\cdot e^{\varepsilon_i}$$

where $\varepsilon_i \sim \mathrm{N}(0,\sigma_i)$ and σ_i was the standard error estimated in the fit of the i-th SRR.

(c) Catchability (q)

$$q_{a,y,i} = q_{a,i} = \bar{F}_{(2006-2008)XSA_{a,i}}$$

(d) Weight at age (W_a)

$$W_{a,y,i} = W_a = \overline{W}_{(2006-2008)_a}$$

(e) Natural mortality at age (M_a)

$$M_{a,y,i} = M = 0.2$$

(f) Numbers at age (N_a)

$$N_{a,y,i} = \begin{cases} SRR_i\left(RI_{y,i}\right) & a = 0 \\ N_{a,y-1,i}\cdot e^{-(M+F_{a-1,2008,i})} & a > 0 \end{cases}$$

$$RI_{y,i} = \sum_a RI_{a,y,i}$$

(g) Catch at age (C_a)

$$C_{a,y,i} = \frac{F_{a,y,i}}{F_{a,y,i}+M}\cdot\left(1-e^{-(F_{a,y,i}+M)}\right)\cdot N_{a,y,i}$$

Management Procedure

(a) Natural mortality and weight at age same as in the real population.

(b) Tuning indices
For each tuning index (I)

$$I_{ind,a,y,i} = N_{a,y,i}\cdot q_{i_{ind,a}}\cdot\varepsilon_{ind,a,y,i} : \varepsilon\sim LN(0,\sigma_{ind,a})$$

where q_i is the index specific catchability estimated in the XSA initial fit.

In the short term forecast given that management procedure is carried out in year y.

(c) Recruitment (R)

$$\hat{R}_y = \hat{R}_{y+1} = GM(\hat{R}_{xsa,k,i})_{k=y-22}^{k=y-3}$$

where GM stands for the geometric mean.

(d) Selection pattern (S)

$$\hat{S}_{a,y,i} = \hat{S}_{a,y+1,i} = \frac{1}{3}\cdot\sum_{k=y-3}^{y-1}\frac{\hat{F}_{a,k,i}}{\hat{F}_{k,i}} : \hat{F}_{k,i} = \frac{1}{5}\cdot\sum_{a=2}^{6}\hat{F}_{a,k,i}$$

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