AN INCIDENTAL CATCH MODEL FOR PORBEAGLE ASSESSMENT AND STATUS EVALUATION

Heather D. Bowlby[1] and Enric Cortés[2]

SUMMARY

*Fisheries landings and associated biological data collection for porbeagle shark (*Lamna nasus*) declined substantially following CITES Appendix II trade restrictions in 2013 due to changes in fleet behavior. This document describes a new stock assessment method that can be used when length-frequency data and CPUE series are not available or reliable for index changes in abundance. The Incidental Catch Model (ICM) is based on the same general premise as data-poor, length-based assessments, in that it uses life history information and equilibrium assumptions to derive a theoretical age-structured population in the absence of fishing. In the ICM, the effect of historical fishing pressure on productivity is taken into account prior to evaluating fishery removals and abundance relative to reference points. The Northwest Atlantic stock was used to demonstrate the method, which can be adapted to assess stocks in the Northeast and South Atlantic by changing life history inputs and the removals series.*

RÉSUMÉ

*Les débarquements des pêcheries et la collecte de données biologiques associées concernant le requin-taupe commun (*Lamna nasus*) ont considérablement diminué suite aux restrictions commerciales découlant de l'inscription de cette espèce à l'annexe II de la CITES en 2013 en raison de changements dans le comportement des flottilles. Ce document décrit une nouvelle méthode d'évaluation des stocks qui peut être utilisée lorsque les données de fréquences de tailles et les séries de CPUE ne sont pas disponibles ou fiables pour les changements d'indice d'abondance. Le modèle de capture accidentelle (« ICM » selon les sigles anglais) est basé sur le même principe général que les évaluations basées sur la taille et pauvres en données, dans la mesure où il utilise des informations sur le cycle vital et des postulats en conditions d'équilibre pour dériver une population théorique structurée par âge en l'absence de pêche. Dans le ICM, l'effet de la pression historique de la pêche sur la productivité est pris en compte avant d'évaluer les prises et l'abondance de la pêche par rapport à des points de référence. Le stock de l'Atlantique Nord-Ouest a été utilisé pour démontrer la méthode, qui peut être adaptée pour évaluer les stocks de l'Atlantique Nord-Est et Sud en modifiant les intrants du cycle vital et les séries de capture.*

RESUMEN

*Los desembarques de las pesquerías y la recopilación de datos biológicos asociados al marrajo sardinero (*Lamna nasus*) disminuyeron sustancialmente tras las restricciones al comercio del Apéndice II de la CITES en 2013 debido a los cambios en el comportamiento de la flota. En el presente documento se describe un nuevo método de evaluación de stock que puede utilizarse cuando no se dispone de datos de frecuencias de talla y series de CPUE o estos no son fiables para los cambios del índice de abundancia. El modelo de captura incidental (ICM) se basa en la misma premisa general que las evaluaciones basadas en la talla y con pocos datos, en el sentido de que utiliza la información del ciclo vital y los supuestos de equilibrio para derivar una población teórica estructurada por edades en ausencia de pesca. En el ICM se tiene en cuenta el efecto de la presión pesquera histórica en la productividad antes de evaluar las extracciones de la pesca y la abundancia con respecto a los puntos de referencia. Se utilizó el stock del Atlántico noroccidental para demostrar el método, que puede adaptarse para evaluar los stocks en el Atlántico nororiental y meridional cambiando las entradas del ciclo vital y las series de extracciones.*

---

[1] Fisheries & Oceans Canada, Bedford Institute of Oceanography, Dartmouth, Nova Scotia, CANADA. E-mail address of lead author: heather.bowlby@dfo-mpo.gc.ca.

[2] National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, Panama City Laboratory, Panama City, Florida 32408, U.S.A. E-mail: Enric.Cortes@noaa.gov

1. Introduction

A fundamental assumption underlying traditional fisheries assessment is that at least one of the data inputs (e.g. catch per unit effort; CPUE) indexes abundance. In situations where there have been profound changes in data indices that are independent of changes to abundance, such models (e.g. Surplus Production, Statistical Catch at Age, Virtual Population Analysis) will not provide information on population size or status relative to current fishery removals (Maunder *et al.* 2006). Similarly, data-poor approaches like length-based assessments rely on having a representative length-frequency distribution from the catches to compare with the equilibrium length distribution of the population (based on life history characteristics) to estimate fishing mortality (Hordyk *et al.* 2015a; Hordyk *et al.* 2015b). For the porbeagle stocks assessed by ICCAT, recent changes to CPUE series and relatively limited data from which to estimate length-frequency means that a new approach to stock assessment is needed.

Here, we propose an Incidental Catch Model (ICM) that can be used to evaluate historical abundance, estimate reference points, determine status (i.e. overfished), and quantify stock response to various levels of future fishing mortality. It incorporates variability in life history inputs (Cortés and Semba 2020) and enables status assessment relative to abundance reference points (i.e. overfished status; Brooks *et al.* 2010). We demonstrate the model by fitting it to data from the Northwest Atlantic porbeagle stock. We chose this example because the predicted abundance trajectory from the ICM could be directly compared to predictions from a fully-integrated Statistical Catch at Age model (Campana *et al.* 2010) to help with validation of the approach.

2. Methods

The ICM is a simulation model with two main parts: (1) a backward-projecting component, used to predict the historical abundance trajectory given the actual time series of removals, and (2) a forward-projecting component that accounts for any reduction in productivity from fishing pressure to assess current status as well as the scope for future removals. The ICM accounts for uncertainty in our understanding of porbeagle life history by simulating over a distribution of values for population productivity. This distribution comes from a Leslie matrix approach to get the theoretical maximum capacity for population growth in the absence of fishing (r_{max}). This type of modeling approach has been previously applied to demographic analyses of cetaceans (Caswell *et al.* 1998; Dans *et al.* 2003), basking shark (Campana *et al.* 2008) and white shark (Bowlby and Gibson 2020). This is the first application of this type of simulation model for stock assessment at ICCAT.

2.1 Simulation Model

It is very likely that porbeagle in the Northwest Atlantic are at low abundance, so we used a simple exponential growth model in the backwards projections to predict changes in population size (N) from the current year (y) to the previous year (y-1), accounting for removals (R):

(1) $N_y = e^r N_{y-1} - R_{y-1}$

which can be rearranged as:

(2) $N_{y-1} = \frac{(N_y + R_{y-1})}{e^r}$

The time series of removals from Task 1 data informs R, spanning the years 1961 to 2018 for the Northwest Atlantic. Thus, the backwards projections start in 2018 and end in 1961. The removals series incorporated an estimation of recent catches from non-reporting fleets based on a ratio estimator, consistent with the previous assessment (Anon 2010). We used the combined size distribution for captures (Santos *et al.* 2020) and the growth function parameters (Cortés and Semba 2020) for the Northwest Atlantic to transform Task 1 biomass into numbers.

The backwards projections occur on an annual time step where population increase happens first and removals take place afterwards. This makes the analysis more precautionary as it slightly exaggerates the effects of removals by modeling them as a discrete rather than a continuous process. Note that we used the population's capacity for growth in the absence of fishing (r_{max}) in these backwards projections. This means that the effect of removals in each year (i.e. the level of exploitation) was calculated relative to the population's theoretical capacity for growth at equilibrium, which is the same premise that underlies length-based assessment approaches (Hordyk et al. 2015a; Hordyk et al. 2015b).

In an age-structured population, fishing mortality (F) and exploitation rates (u) are a function of survival at age (l_x), fisheries selectivity, removals and abundance. Survival at age is conditional on natural mortality (M) and instantaneous fishing mortality (F).

(3) $l_x = \prod_{i=0}^{x-1} e\left(-(M_i + F_i)\right)$

In the backward projections as well as the assessment of status (below), we found the value for F using minimization of the sum of squared residuals between observed removals (R_y) and predicted removals. From the basic relationship between an annual exploitation rate (u) and instantaneous fishing mortality (F):

(4) $u = 1 - e^{-F}$

The number of animals in the population in a given year (N_y) that are vulnerable to the fishery becomes:

(5) $vulnerable = N_y \frac{\sum_{x=sel}^{A} lx}{\sum_{x=0}^{A} lx}$

where *sel* is the age at which animals are fully selected to the fishery and *A* represents maximum age. Here we have approximated a dome-shaped selectivity function by assuming constant fishing mortality rates on juveniles and no fishing mortality on adults. This reflects the length composition data from various fleets, where the vast majority of fisheries captures are immature. We set F to zero above the age at maturity in the calculation of (l_x) and *sel* = 1. Predicted removals are simply *vulnerable*u*.

We used a simple logistic growth model for the forward projections of the simulation, assuming a carrying capacity (K) of 20 million animals (i.e. very weak density dependence) in the Northwest Atlantic.

(6) $N_{t+1} = e^r N_t \left(1 - \frac{N_t}{K}\right)$
(7)

This was done to ensure that the future projections could not grow without bound and thus produce overly optimistic results on the potential for population recovery. We assumed the same selectivity pattern as in the backwards projections and we accounted for the reduction to r_{max} from fisheries removals ($r < r_{max}$) by substituting the estimated value of F into Equation 3. We projected the model forward for 50 years which is roughly 2.5 generation times for porbeagle (generation time ~20 years). Average removals from 2016-2018 were assumed for removals in 2019 and 2020, with the projections starting in 2021. This reflects the earliest year in which management changes could affect the population, given the lag between the 2020 assessment and the implementation of management recommendations.

2.2 Model parameterization

We obtained life history parameters for females from previously published data (refer to **Table 1** in Cortés and Semba 2020). The growth equation came from Natanson et al. (2002) and information on reproductive biology, including the shape of the maturity ogive, litter size, and gestation period came from Jensen et al. (2002). The maturity ogive was updated to include unpublished data from Canada (Cortés and Semba 2020). An estimate of natural mortality (M) is required to determine survival at age. We calculated six different estimators for M (see details in Cortés and Semba 2020) and took the lowest of the suite in a particular iteration to calculate survivorship. Using the life history parameters, we applied the age-structured Leslie matrix approach (equivalent to a life table analysis/Euler-Lotka equation) for females described in Cortés and Semba (2020) to generate a distribution of r_{max} values for input into the ICM. In brief, we used Monte Carlo sampling from predefined statistical distributions for multiple life history parameters to generate the maturity ogive and survivorship at age, as well as values for age at maturity and lifespan. The number of MC samples matched the total number of simulations run for the ICM (n = 5000). Random draws for parameters contributing to the von Bertalanffy Growth Function (VBGF) and maturity ogive came from a multivariate normal distribution to account for covariance. We used a uniform distribution for lifespan with the observed longevity from vertebral aging as the lower bound and seven half-lives (from the K parameter of the growth function) as the upper bound (Cortés and Semba 2020).

2.3 Status evaluation

We assessed whether the stock in the Northwest Atlantic is currently overfished using the SPR_{MER} reference point proposed by Brooks *et al.* (2010): the Spawning Potential Ratio at Maximum Excess Recruitment. This biological reference point is derived entirely from life history data and has been found to accurately predict overfished status relative to Maximum Sustainable Yield (MSY) reference points from traditional stock assessments (Cortés and Brooks 2018). It is calculated as:

(8) $SPR_{MER} = \frac{1}{\sqrt{\hat{\alpha}}}$

Where $\hat{\alpha}$ represents the maximum lifetime reproductive (MLR) rate (Myers *et al.* 1997, 1999). For this model the MLR is defined as the maximum number of female spawners that can be produced by a female spawner throughout her life (as in Bowlby & Gibson 2020). It is calculated from the net reproductive rate or spawners per recruit (SPR) multiplied by maximum age-0 survival (Brooks *et al.* 2010). Overfished status is determined by comparing current abundance with a threshold value. This value typically represents a given proportion (*p*) of the stock size which is expected to produce MSY (e.g. *p* = 1-*M*; Brooks *et al.* 2010). However, ICCAT typically uses MSY directly as the reference point for abundance, which means that the threshold value is *p* = 1.

The threshold value representing the depletion of spawners and recruits at Maximum Excess Recruitment, assuming a Beverton-Holt stock-recruit relationship, is:

(9) $\frac{S_{MER}}{S_0} = \frac{\sqrt{\hat{\alpha}}-1}{\hat{\alpha}-1}$

The population is considered overfished if the level of depletion in an abundance index (*I*) divided by the threshold value in Equation (8) is smaller than the proportion *p*:

(10) $\dfrac{\frac{I_{current}}{I_{unfished}}}{\frac{S_{MER}}{S_0}} < p$

Here, we have assumed that predicted abundance in 1961 represents unfished population size and that abundance in 2018 represents current abundance. Each iteration of the simulation yields a different value for $\hat{\alpha}$ as well as for current (2018) and unfished population size (1961) due to the manner in which variability is incorporated into the model (MC sampling from distributions). Therefore, solving Equation 8 gives a distribution of values that can be compared to *p*.

Similarly, the proportion of simulations that are overfished at a given time step in the forward projections can be found by using predicted future abundance as $I_{current}$. We evaluated the forward predictions at 5-year intervals relative to removals scenarios ranging from 0 to 24,000 animals (0 – 734 mt). In each future year, the proportion of trajectories that are overfished becomes the number of simulations < *p* divided by the total number of simulations.

2.4 Validation

There are no model residuals or deviances output by the ICM that can be used for validation with traditional metrics like AIC or negative loglikelihood (Johnson and Omland 2004). Instead, using different life history parameters or starting the population at a different initial abundance will lead to a different historical abundance trajectory and different estimates of *F*. For this reason, we felt it was necessary to compare model output with a more traditional fishery model to determine if the ICM was a reasonable assessment method. Fortunately, a Statistical Catch at Age (SCA) model that used data up to 2009 was developed for porbeagle in the Northwest Atlantic (Campana *et al.* 2010). The Canadian SCA was a fully integrated assessment, which incorporated the standardization of raw CPUE data within the model. It was sex-based and stage-structured, and partitioned catches according to three different age-length keys, representing catch profiles from three geographic regions. Four model variants were tested, where each incorporated slightly different life history assumptions leading to differences in productivity. Fisheries selectivity for each region was estimated in the model. Although 10 years old, the general structure of the SCA as well as the methods used for fitting (minimization of an objective function via the Laplace approximation in ADMB) are still considered to be among best-practices for data-rich, age-structured assessments (Maunder and Piner 2015).

We qualitatively compared the predicted historical abundance trajectory from the ICM with predictions from the SCA, after standardizing data inputs into the ICM from the SCA. The SCA used a time series of removals derived from NAFO reporting (1961-2009) rather than Task 1 data, which was used as the removals series in the ICM for this comparison. The SCA assumed a Beverton-Holt stock recruitment relationship, and the variant called Model 3 had the most similar reproductive assumptions to the ICM. The VBGF growth parameters were identical.

2.5 Sensitivity Analyses

Recent research has suggested a possible biennial reproductive cycle for porbeagle (Natanson *et al.* 2019). Thus, we present two alternate life history parameterizations: one assuming a reproductive periodicity of one year (High Productivity), and one assuming a 50:50 mix of annually-reproducing and biennially-reproducing females (Medium Productivity). The Medium scenario is expected to better characterize productivity in the North Atlantic stock and was used as the base case in the current assessment. The High scenario is given both for continuity with the previous assessment (Anon 2010) and as a sensitivity analysis.

We also evaluated the sensitivity of the model to the assumed population size in 2018. The Canadian SCA model that was used for validation predicted an abundance of approximately 200,000 individuals in 2009, which was used as median abundance in 2018 in the base case of the ICM. This assumption was made because the ICM produced similar trends in abundance (i.e. similar overall decline rates) as compared to the SCA from this starting size, even though the absolute values predicted for abundance in each year differed. The ICM used a different removals series and had different productivity assumptions, so we would not expect an exact match in predicted abundance between the two models, yet we would expect similar trends. However, to evaluate a higher biomass scenario (representing a stock that was currently at a larger total size), we scaled the historical abundance trajectory so that the ICM predicted a population size of approximately 200,000 animals in 2009 for this sensitivity analysis. This was done by iteratively increasing assumed abundance in 2018 until median abundance in 2009 was 200,000 animals in the backwards projections.

3. Results

3.1 Base Case: mixture of biennial and annual reproduction

The median and 95th percentiles for r_{max} and the SPR$_{MER}$ reference point were 0.059 (0.036, 0.081) and 0.558 (0.415-0.743), respectively (**Table 1**). The backwards projections of the ICM indicated that maximum abundance occurred in the 1960s, with a median predicted population size of ~985,000 animals (top panel; **Figure 1**). Declines continued until 2010, reaching a low of ~170,000, before increasing to the assumed 2018 median population size of 200,000 animals. This means that the stock declined by 79% from maximum historical abundance over the entire time series. The threshold value for biomass at SPR$_{MER}$ was 353,000 animals.

The stock was predicted to be overfished with a very high probability, with 99% of iterations below the critical value of $p = 1$. From the future projections (top panel; **Figure 2**), the stock would remain above the overfished threshold with a > 60% probability until 2070 if removals remained at or below 7,000 animals (**Table 2**). Even if removals (landings plus dead discards) were reduced to zero, it would take until 2035 to rebuild the population to the threshold level for biomass with a > 60% probability.

3.2 Sensitivity Analysis: annual reproduction

If population productivity was higher, the median and 95th percentiles for r_{max} and the SPR$_{MER}$ reference point were 0.079 (0.057, 0.102) and 0.456 (0.339-0.607), respectively (**Table 1).** Predicted population size in 1961 was lower, at 849,000 animals (middle panel; **Figure 1**), leading to a predicted 76% decline over the entire time series. As would be expected from a population with higher productivity, biomass at SPR$_{MER}$ was lower, at 266,000 animals. If reproduction is annual, the stock was still predicted to be overfished with a high probability, with 83% of iterations below the critical value of $p = 1$. From the future projections (middle panel; **Figure 2**), the stock would remain above the overfished threshold with a > 60% probability for 2.5 generations if removals remained at or below 13,000 animals (**Table 3**). Note that it takes less time for the population to recover given that 2018 abundance is much closer to the overfished threshold (middle panel; **Figure 2**).

3.3 Sensitivity Analysis: higher biomass

This sensitivity run used the same life history parameters as in the base case, and had the same values for r_{max} and SPR$_{MER}$ (**Table 1**). Median population size in 2018 needed to be increased to 310,000 animals to give a predicted median abundance in 2009 of 200,000 animals. Population size in 1961 was 994,000 animals (lower panel; **Figure 1**), giving a historical decline rate of 68%. The threshold value for biomass at SPR$_{MER}$ was 349,000 animals. The slight difference in the threshold value from the base case arises from the random MC sampling.

The stock remained overfished if 2018 biomass was 310,000 animals, with 70% of iterations below the critical value of $p = 1$. The future projections (lower panel; **Figure 2**) gave similar results to the high productivity sensitivity analysis; the stock would remain above the overfished threshold with a > 60% probability for 2.5 generations if removals were kept below 14,000 animals (**Table 4**). Similar to the previous scenario, the stock was close to the overfished threshold in 2018 ad could recover more quickly than in the base case scenario (lower panel; **Figure 2**)

3.4 Comparison of Base Case and Sensitivity Analyses

A mix of biennial and annual reproduction reduced the productive capacity of the stock by 25% as compared to annual reproduction. The SPR$_{MER}$ reference point increased from 0.456 (0.339-0.607) to 0.558 (0.415-0.743). Recall that the closer the SPR$_{MER}$ value is to 1, the less exploitation can be permitted (Brooks *et al*. 2010).

The predicted trends in abundance are qualitatively similar among the base case and the two sensitivity analyses (**Figure 1**), but predicted abundance in 1961 was ~136,000 animals less when productivity was higher. This was expected because the population had greater ability to recover from exploitation in the High Productivity scenario so a smaller total number can support a given level of removals.

In all scenarios, the stock reached minimum abundance around 2002 and was predicted to have been increasing since that time. However, all scenarios suggest that the stock is currently overfished and that future removals need to remain low to permit population recovery.

3.5 Model Validation

When productivity inputs and the removals series used in the ICM were matched (as closely as was possible) to the Canadian SCA model, the different approaches produced extremely similar historical abundance trajectories. This could only occur if the ICM was producing similar estimates of the annual exploitation rate as compared to the SCA. Predicted starting population size in the 1960s was just over 800,000 animals from both modeling approaches and declined following a very similar pattern (**Figure 3**). The total predicted decline rate was essentially identical. These results are quite remarkable given how different the data inputs, estimation method and assumptions are in the SCA as compared to the ICM and lends validity to our assertion that the ICM is useful for assessment.

4. Discussion and Conclusions

We have presented a relatively simple simulation model that can be used to assess status relative to biological reference points for porbeagle in the Northwest Atlantic. As in traditional stock assessment, the ICM assumes that the time series of removals is known. However, unlike traditional fisheries assessments, the ICM uses the theoretical productive capacity of the stock at equilibrium to index changes in abundance and to scale historical population size estimates. There was very good correspondence between the abundance predictions of the ICM as compared to a complex, sex-specific and age-structured SCA model. Because of this, we suggest that the ICM can be used to derive advice.

For the stock in the Northwest Atlantic, we used previous assessment results to scale 2018 abundance and to evaluate the sensitivity of the model to that assumption. In the absence of information on current population size, it would still be possible to use the ICM for assessment if a representative CPUE series was available. In this instance, the ICM would be parameterized so that relative changes in abundance matched the trends in the CPUE series (i.e. declined or increased by the same amount over the same number of years), giving a prediction of population size in each year of the removals series. We attempted to fit the ICM to the combined population in the South Atlantic by changing life history parameter input values (e.g. growth coefficients) and the removals series. However, the two CPUE series for the Southern stock showed very different trends over the same years, either

declining strongly in the SW or remaining essentially without trend in the SE. As in other types of fisheries models, the ICM was extremely sensitive to the opposite CPUE trends and the group decided it was not advisable to use the ICM to derive advice until a more representative CPUE series or an estimate of population size was available for the combined Southern stock.

It is very likely that porbeagle in the Northwest Atlantic are currently overfished and remain at low abundance. The future projections demonstrate that the stock in the Northwest Atlantic has very little capacity for removals. Keeping removals below 7,000 animals (base case) would convert to a biomass of 214 mt. For comparison, removals averaged 47 mt from 2014-2018, 143 mt from 2009-2013, 192 mt from 2004-2008 and 611 mt from 1999-2003. The ICM suggests that it is very unlikely that porbeagle in the Northwest Atlantic can sustain the level of removals seen prior to the early 2000s. Keeping current management restrictions in place will likely allow for porbeagle in the Northwest Atlantic to recover, even if current fisheries removals are slightly underestimated.

References

Anon. 2010. Report of the 2009 porbeagle stock assessments meeting (Copenhagen, Denmark, June 22 to 27, 2009). ICCAT Collective Volumes of Scientific Papers. 65(6):1909-2005.

Bowlby, H.D., and A.J.F. Gibson. 2020. Implications of life history uncertainty when evaluating status in the Northwest Atlantic population of white shark (*Carcharodon carcharias*). Ecology and Evolution. 00:1-11. DOI: 10.1002/ece3.6252.

Brooks, E.N., J.E. Powers and E. Cortés. 2010. Analytical reference points for age-structured models: application to data-poor fisheries. ICES Journal of Marine Science. 67:165-175.

Campana, S.E., A.J.F. Gibson, M. Fowler, A. Dorey, and W. Joyce. 2010. Population dynamics of Porbeagle in the northwest Atlantic, with an assessment of status to 2009 and projections for recovery. Collect. Vol. Sci. Pap. ICCAT, 65(6): 2109-2182

Campana, S.E., J. Gibson, J. Brazner, L. Marks, and W. Joyce. 2008. Status of basking sharks in Atlantic Canada. DFO Canadian Science Advisory Secretariat Research Document. 2008/004.

Caswell, H., S. Brault, A.J. Read, and T.D. Smith. 1998. Harbor porpoise and fisheries: uncertainty in analysis of incidental mortality. Ecological Applications. 8:1226-1238.

Cortes, E., and E.N. Brooks. 2018. Stock status and reference points for sharks using data-limited methods and life history. Fish and Fisheries 19:1110-1129.

Cortes, E., and Y. Semba. 2020. Estimates of vital rates and population dynamics parameters of interest for porbeagle shark in the Western North Atlantic and South Atlantic oceans. SCRS/2020/090.

Dans, S.L., M. Koen Alonso, S.N. Pedraza, and E.A. Crespo. 2003. Incidental catch of dolphins in trawling fisheries off Patagonia, Argentina: can populations persist? Ecological Applications. 13:754-762.

Hordyk, A., K. Ono, S. Valencia, N. Loneragan, and J. Prince. 2015b. A novel length-based empirical estimation method of spawning potential ratio (SPR), and tests of its performance, for small-scale, data-poor fisheries. ICES Journal of Marine Science. 72: 217–231.

Hordyk, A.R., K. Ono, K. Sainsbury, N.R. Loneragan, and J.D. Prince. 2015a. Some explorations of the life history ratios to describe length composition, spawning-per-recruit, and the spawning potential ratio. ICES Journal of Marine Science. 72:204–216.

Jensen, C.F., L.J. Natanson, H.L. Pratt Jr., N. E. Kohler, and S.E. Campana. 2002. The reproductive biology of the porbeagle shark (*Lamna nasus*) in the western North Atlantic Ocean. Fisheries Bulletin. 100:727–738.

Johnson, J.B., and K.S. Omland. 2004. Model selection in ecology and evolution. Trends in Ecology and Evolution 19:101-108

Maunder, M.N., and K.R. Piner, 2015. Contemporary fisheries stock assessment: many issues still remain. ICES Journal of Marine Science. 72:7–18.

Maunder, M.N., J.R. Sibert, A. Fonteneau, J. Hampton, P. Kleiber, and S.J. Harley. 2006. Interpreting catch per unit effort data to assess the status of individual stocks and communities. ICES Journal of Marine Science. 63: 1373e1385.

Myers, R.A., G. Mertz, and S. Fowlow. 1997. Maximum population growth rates and recovery times for Atlantic cod, *Gadus morhua*. Fishery Bulletin. 95:762–772.

Myers, R.A., K.G. Bowen, and N.J. Barrowman. 1999. Maximum reproductive rate of fish at low population sizes. Canadian Journal of Fisheries and Aquatic Sciences. 56:2404–2419.

Natanson, L.J., B.M. Deacy, W. Joyce, and J. Sulikowski. 2019. Presence of a resting population of female porbeagles (*Lamna nasus*), indicating a biennial reproductive cycle, in the western North Atlantic Ocean. Fisheries Bulletin. 117:70-77.

Natanson, L.J., J.J. Mello, and S.E. Campana. 2002. Validated age and growth of the porbeagle shark, *Lamna nasus*, in the western North Atlantic Ocean. Fisheries Bulletin. 100:266–278.

Table 1. Demographic rates for the Northwest Atlantic population of porbeagle shark (*Lamna nasus*) used in the ICM model assuming a 50:50 mix of annual and biennial reproduction (Base Case) and the sensitivity run assuming annual reproduction (High Productivity).

Parameter	Base Case			High Productivity		
	Median	0.025	0.975	Median	0.025	0.975
Lambda	1.061	1.037	1.084	1.083	1.059	1.107
rmax	0.059	0.036	0.081	0.079	0.057	0.102
Generation Time	20.116	17.317	21.324	19.317	16.948	20.462
Net Repro Rate	3.520	2.013	6.235	5.281	3.019	9.352
Age-0 Survivorship	0.912	0.894	0.931	0.912	0.894	0.931
Alpha hat	3.208	1.812	5.803	4.811	2.718	8.705
Steepness	0.445	0.312	0.592	0.546	0.405	0.685
SPR_MER	0.558	0.415	0.743	0.456	0.339	0.607
R	0.603	0.543	0.702	0.554	0.508	0.621

Table 2. Percent of simulations that are above the overfished threshold for porbeagle (*Lamna nasus*) in the Northwest Atlantic for the base case of the ICM.

Removals (#)	Removals (mt)	2020	2025	2030	2035	2040	2045	2050	2055	2060	2065	2070
0	0	2	21	47	68	83	92	96	98	99	99	100
1000	31	3	21	44	63	77	87	92	95	97	98	99
2000	61	2	19	40	57	71	81	87	91	94	95	96
3000	92	1	16	35	50	62	72	79	85	88	90	92
4000	122	2	15	32	47	58	66	73	78	82	84	87
5000	153	2	13	27	41	50	58	64	68	72	76	78
6000	183	1	12	25	37	45	52	57	62	65	67	70
7000	214	2	10	22	32	39	46	50	54	57	60	62
8000	245	2	10	19	27	34	39	44	47	50	53	55
9000	275	2	8	17	23	30	34	38	41	43	45	47
10000	306	2	8	14	20	25	29	31	34	36	38	39
11000	336	1	6	13	17	21	25	27	29	31	32	33
12000	367	2	7	11	15	18	21	23	24	26	27	28
13000	398	2	5	9	12	14	16	18	19	20	21	22
14000	428	2	5	7	9	12	13	14	15	16	17	18
15000	459	1	3	5	6	8	9	10	11	11	12	12
16000	489	2	3	4	5	6	7	8	9	9	10	10
17000	520	2	2	3	4	5	5	6	6	6	7	7
18000	550	2	2	2	3	3	4	4	4	5	5	5
19000	581	2	1	2	2	3	3	3	3	3	3	4
20000	612	2	1	1	2	2	2	2	2	2	3	3
21000	642	2	1	1	1	1	1	2	2	2	2	2
22000	673	1	1	1	1	1	1	1	1	1	1	1
23000	703	2	1	0	0	0	0	0	0	0	0	0
24000	734	2	1	0	0	0	0	0	0	0	0	0

Table 3. Percent of simulations that are above the overfished threshold for the sensitivity analysis assuming annual reproduction (High Productivity) in the ICM.

Removals (#)	Removals (mt)	2020	2025	2030	2035	2040	2045	2050	2055	2060	2065	2070
0	0	34	67	91	99	100	100	100	100	100	100	100
1000	31	35	66	88	98	100	100	100	100	100	100	100
2000	61	35	65	85	96	99	100	100	100	100	100	100
3000	92	33	61	80	92	97	99	100	100	100	100	100
4000	122	35	60	78	89	94	97	99	99	100	100	100
5000	153	33	58	74	84	90	94	96	97	98	99	99
6000	183	34	56	70	80	86	90	93	94	95	96	97
7000	214	33	54	67	76	82	86	89	90	92	93	93
8000	245	34	53	64	72	77	81	84	86	87	89	89
9000	275	34	50	61	68	72	76	79	81	82	83	84
10000	306	34	49	58	64	68	71	73	75	76	77	78
11000	336	34	47	55	60	64	66	68	70	71	72	73
12000	367	35	46	53	57	60	62	63	65	66	67	67
13000	398	35	44	50	53	56	58	59	60	61	61	62
14000	428	35	42	47	50	52	53	54	55	56	56	57
15000	459	32	39	43	45	46	48	48	49	50	50	51
16000	489	33	37	40	42	43	44	44	45	45	45	45
17000	520	34	36	38	38	39	39	40	40	40	40	40
18000	550	33	34	34	34	34	35	35	35	35	35	35
19000	581	34	33	32	32	31	31	31	31	30	31	31
20000	612	34	31	30	29	28	28	27	27	27	27	26
21000	642	35	31	27	26	25	24	23	23	22	22	22
22000	673	33	28	24	22	21	20	19	19	18	18	18
23000	703	35	26	22	20	18	17	16	15	15	15	14
24000	734	34	24	20	16	15	14	13	12	12	11	11

Table 4. Proportion of simulations that are above the overfished threshold for the sensitivity analysis assuming a larger population size in 2009 in the ICM.

| Removals (#)	Removals (mt)	2020	2025	2030	2035	2040	2045	2050	2055	2060	2065	2070
0	0	45	73	92	97	99	100	100	100	100	100	100
1000	31	52	77	93	97	99	100	100	100	100	100	100
2000	61	49	75	90	96	97	99	99	99	99	99	100
3000	92	50	72	87	95	96	98	99	99	100	100	100
4000	122	46	68	84	91	95	96	97	97	98	99	99
5000	153	47	68	80	90	94	95	96	97	97	98	98
6000	183	48	64	77	86	91	93	95	96	96	96	96
7000	214	44	61	73	80	85	88	91	92	93	94	94
8000	245	46	63	73	79	84	86	88	88	89	90	91
9000	275	38	51	62	68	74	79	80	82	84	85	85
10000	306	43	55	63	70	74	77	78	80	81	83	83
11000	336	48	59	66	69	72	75	76	77	77	78	79
12000	367	42	50	55	59	61	64	66	67	67	68	69
13000	398	45	51	54	58	60	62	63	65	65	65	66
14000	428	48	52	55	55	57	57	58	59	60	60	61
15000	459	50	52	52	53	53	54	53	52	52	53	54
16000	489	45	47	47	47	47	47	47	46	46	46	47
17000	520	46	44	44	43	43	43	42	42	41	42	41
18000	550	47	43	40	39	38	38	37	36	36	35	35
19000	581	44	39	37	37	35	33	33	32	31	32	32
20000	612	45	39	35	32	30	28	28	28	28	28	27
21000	642	47	42	36	32	30	28	27	25	25	24	24
22000	673	38	28	24	21	19	18	17	17	17	16	16
23000	703	43	30	24	21	18	16	15	15	14	13	13
24000	734	52	36	28	23	18	17	14	13	13	12	12
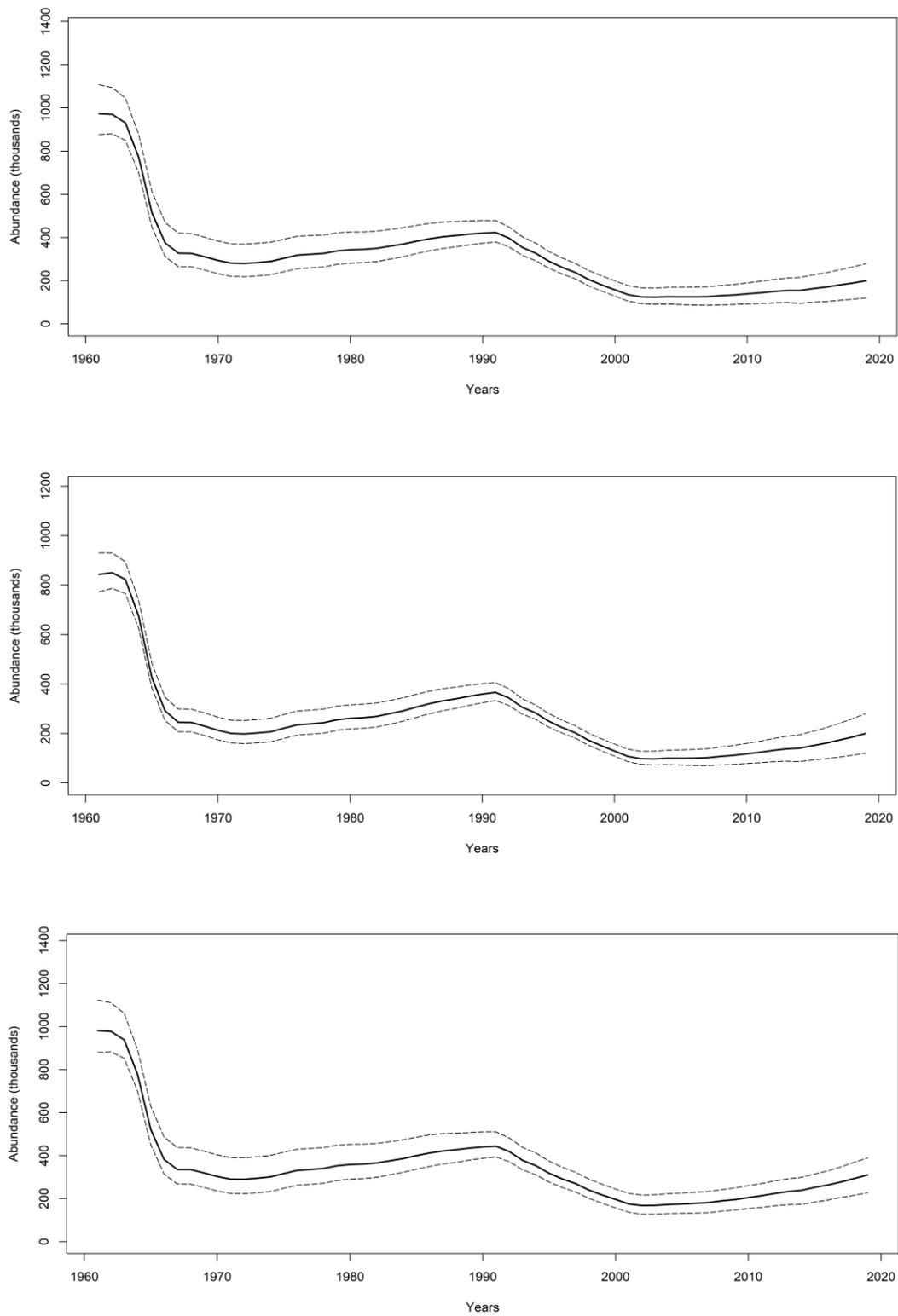
Figure 1. Predicted median abundance trajectory (solid line) and 80th percentiles (dashed lines) for the porbeagle stock in the Northwest Atlantic from 1961 to 2018 for the base case of the ICM (top panel), the sensitivity run assuming higher productivity (middle panel) and the sensitivity run assuming higher abundance in 2009 (lower panel).
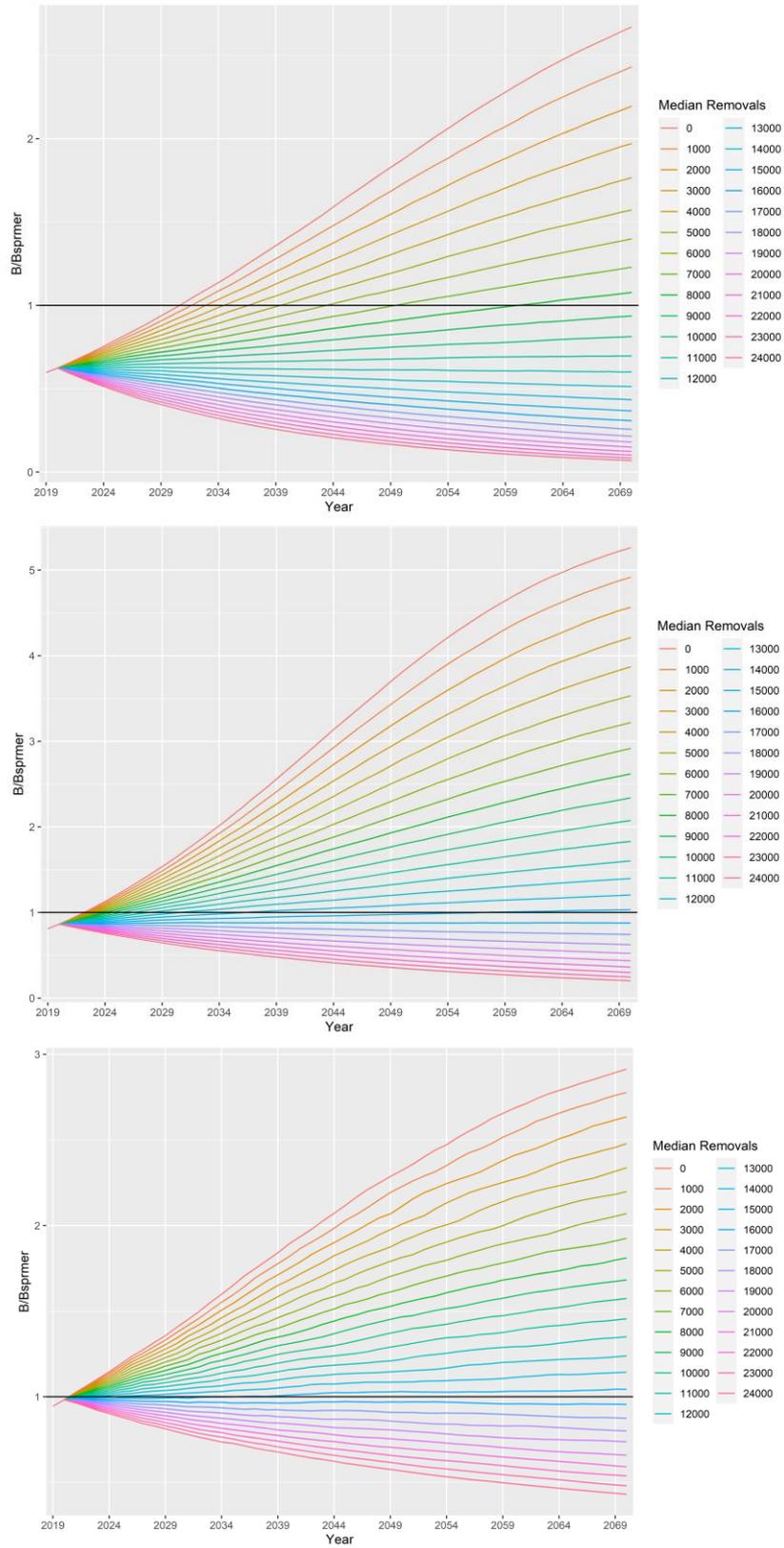
Figure 2. Predicted relative abundance for removals ranging from 0 to 24,000 animals (coloured lines), expressed as the biomass/biomass at SPR$_{MER}$ ratio for the base case of the ICM (top panel), the sensitivity run assuming high productivity (middle panel) and the sensitivity run assuming greater 2009 abundance (lower panel). The horizontal line shows the $B_{SPR_{MER}}$ reference point and the projections extend for 50 years.
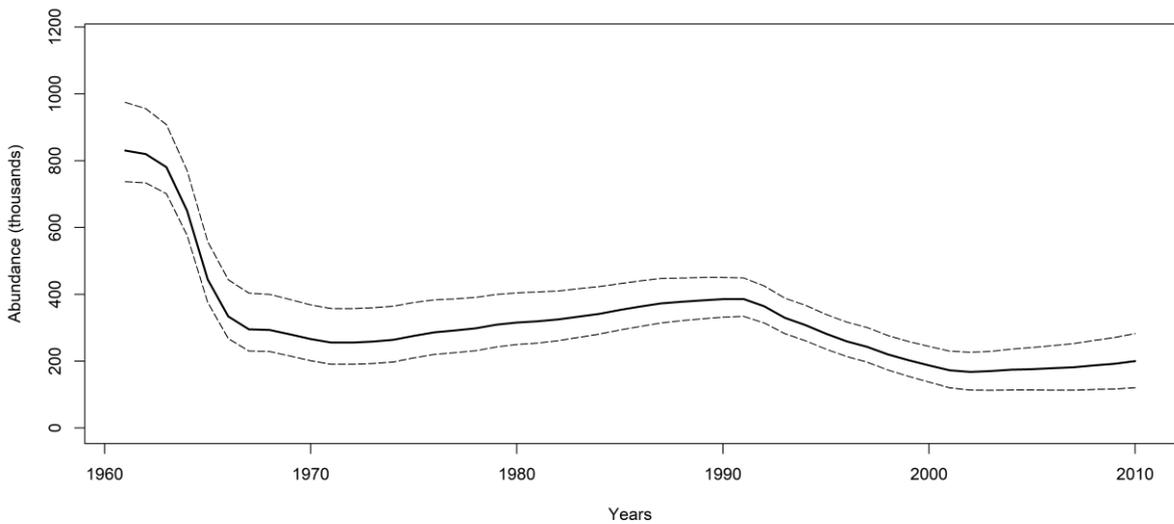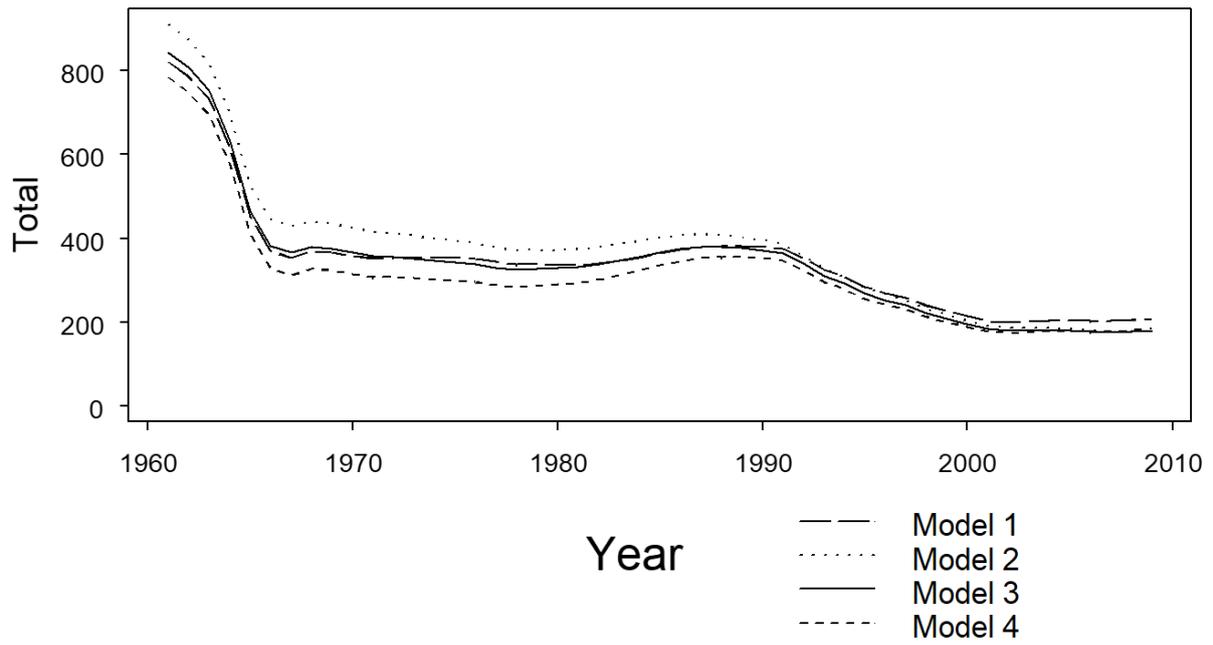
146

Figure 3. Comparison of the predicted historical abundance trajectories from a Statistical Catch at Age Model (Campana *et al.* 2010) for Porbeagle in the Northwest Atlantic (top panel) with output from the ICM over the same number of years, using the NAFO series for removals (lower panel).