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Southern blue whiting (*Micromesistius australis*) stock assessment for the Campbell Island Rise for 2017
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J. Roberts
S.M. Hanchet

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EXECUTIVE SUMMARY

This report documents the stock assessment for the Campbell Island Rise stock of southern blue whiting, using catch history, proportion-at-age, and acoustic survey biomass estimates up to and including the 2016 fishing season. The assessment reported here differs from the previous assessment of this stock, conducted in 2015–16, which modelled the population from 1979 and estimated a non-equilibrium age structure in the first model year. The new assessment models estimated year class strengths from 1958, extended the catch history back to 1971 and assumed an initial equilibrium age structure. The primary data sources were the relative abundance index from the R.V. *Tangaroa* wide-area acoustic biomass surveys carried out from 1993 to 2016, and commercial trawl proportion-at-age data from 1979 to 2016.

The results of the base case assessment model run suggested that the biomass of the Campbell Island Rise stock in 2016 was 239 700 t (95% C.I.s 173 100–328 600 t), and was at 70% B_0 (95% C.I.s 54–86%). The biomass was expected to decline to 36% B_0 by 2021 under an assumption of a constant catch equal to the current TAC in 2016 (40 000 t). The assessment was strongly influenced by the high mature biomass estimates from the last four acoustic surveys in 2009, 2011, 2013 and 2016.

The median estimates of B_0 were similar to those from the 2016 assessment, and higher than earlier estimates of B_0 from this stock. This was primarily due to the strong 2006, 2009, and 2011 year classes which had entered the fishery. There was little doubt that these year classes were strong, although not as strong as the previous large 1991 year class, and that the increase in biomass observed in recent years was due to the recruitment of these strong year classes.

1. INTRODUCTION

In New Zealand waters, southern blue whiting (*Micromesistius australis*) are almost entirely restricted in distribution to sub-Antarctic waters. They are dispersed throughout the Campbell Plateau and Bounty Platform for much of the year, but during August and September they aggregate to spawn near the Campbell Islands, on Pukaki Rise, on Bounty Platform, and near Auckland Islands over depths of 250–600 m, where they are targeted by commercial trawl fisheries (Figure 1). During most years fish in the spawning fishery are 35–50 cm fork length (FL), although occasionally smaller length classes of males (29–32 cm FL) have been observed in the catch (Large & Hanchet 2017).

Commercial fishing has been concentrated on the Campbell Island Rise and, to a lesser extent, the Bounty Platform. The Pukaki Rise and Auckland Islands have generally been smaller fisheries, with much lower annual catches than the Campbell Island Rise and Bounty Platform fisheries (Large & Hanchet 2017).

Figure 1: Relative total density of the commercial catch of southern blue whiting by location, TCEPR data 1990–2016.

Acoustic biomass surveys of the Campbell Island Rise stock using R.V. *Tangaroa* have been carried out approximately every two to three years since 1993 and the most recent survey was carried out in 2016 (O'Driscoll et al. 2018). Stock assessments of the Campbell Island Rise stock have been carried out every one or two years since 1991 with the most recent assessment completed for the 2015–16 fishing year (Dunn & Hanchet 2017).

The results of the 2015–16 assessment were strongly influenced by the value of natural mortality (*M*). Furthermore, when *M* was estimated in the assessment it produced a much more optimistic estimate of stock status, although the *M* estimate was considered implausible by the Deepwater Working Group (Dunn & Hanchet 2017). This led to an investigation of alternative model structures for the estimation of natural mortality in this stock (Roberts & Dunn 2017). This investigation found that the 2015–16

assessment model was unstable at MCMC when the non-equilibrium age structure for the initial state ($C_{initial}$) and M were left unconstrained. The model was stabilised when $C_{initial}$ was discarded in favour of an initial equilibrium age distribution and the model started in 1960 (although this required estimates of catch history in earlier years). However, even with this change in the assumptions over the initial state, the model was still unable to provide a stable unbiased estimate of M. Roberts & Dunn (2017) recommended that the assessment continue to use a value for M of 0.2 with sensitivity analyses at 0.15 and 0.25 until the causes of bias could be identified and corrected.

This report documents the assessment of the Campbell Island Rise stock using data up to and including the 2016–17 fishing year (2016 season). The assessment used the R.V. *Tangaroa* acoustic biomass indices for 1993–2016 and commercial catch proportions-at-age data from the years 1979–2016. A comprehensive summary of available data used for the assessments of southern blue whiting was described by Large & Hanchet (2017).

This report is in fulfilment of the Ministry for Primary Industries Project DEE201611 (To carry out stock assessments of southern blue whiting (*Micromesistius australis*) including estimating biomass and sustainable yields) Objective 2 (To update the stock assessment of the Campbell Island stock, including estimating biomass and sustainable yields).

2. METHODS

2.1 Population dynamics

For the current assessment, a two-sex, single stock and single area Bayesian statistical catch-at-age model for the Campbell Island Rise southern blue whiting stock was implemented in CASAL (Bull et al. 2012) following a similar approach to that used in previous assessments of this stock (e.g., Dunn & Hanchet 2017). The model partitioned the stock into immature and mature fish with two sexes and age groups 2–15, with a plus group at age 15. The model was run for the years 1979 to 2016, with five-year projections run for the years 2017–2021. The annual cycle was partitioned into two time steps (Table 1). In the first time step (nominally the non-spawning season), 90% of natural mortality was assumed to take place. In the second time step (spawning season), fish matured, ages were incremented, and the 2-year-olds were recruited to the population. These were then subjected to fishing mortality and the remaining 10% of natural mortality. A two-sex model was assumed because there are sex-based differences in both the proportions-at-age in the commercial catch for fish aged 2–4 (see Section 0 and their mean length at age (Large & Hanchet 2017).

The stock recruitment relationship was assumed to be Beverton-Holt with a steepness of 0.9, with the proportion of males at recruitment (at age two) assumed to be 0.5 of all recruits. Relative year class strengths were parameterised in the model such that the mean was equal to one.

Southern blue whiting on the Campbell Island Rise are assumed to be mature when on the fishing ground, as they are fished during spawning (Large & Hanchet 2017). Hence, it was assumed that all mature fish were equally selected by fishing. The maximum exploitation rate (U_{max}) was assumed to be 0.8. The proportion of immature fish that mature in each year was estimated for ages 2–5, with fish aged 6 and above assumed to be fully mature (see Dunn & Hanchet 2017 and previous assessments).

Southern blue whiting exhibit large inter-annual differences in growth, presumably caused by local environmental factors, closely correlated with the occurrence of strong and weak year classes (Large & Hanchet 2017). Hence, a standard von Bertalanffy growth curve was not used to determine the mean length at age of fish in the model, but rather an empirical length-at-age matrix. The length-at-age matrix used the empirically estimated mean lengths-at-age from the commercial catch data (Large & Hanchet 2017). Missing estimated mean lengths in the matrix were inferred from the relative size of their cohort and the mean growth of similar ages in other years; and cohorts with unusually small or large increments

were similarly adjusted. For projections, the mean sizes-at-age were assumed to be equal to the estimated sizes-at-age in 2016.

Lengths-at-age were converted to weights-at-age in the model using the length-weight relationship given by Hanchet (1991a), i.e., assuming the relationship weight $= a \times$ length^b for length in centimetres and weight in kilograms. The parameters $a= 0.00515$ and 0.00407, and $b = 3.092$ and 3.152 were assumed for males and females respectively for all model years.

Table 1: Annual cycle of the stock model, showing the processes taking place at each step, and the available observations. Fishing mortality (F) and natural mortality (M) that occur within a time step occur after all other processes. The column headed M is the proportion of M occurring in that time step.

Period	Process	M	Length at age	Observations
1. Nov–Aug	Natural mortality (M)	0.9	–	
2. Sep–Oct	Age, recruitment, fishing mortality (F), and M	0.1	Growth matrix	Proportions at age Acoustic abundance indices

Catches for southern blue whiting have been recorded since 1971 (Large & Hanchet 2017), with an average of 25 000 t annual catch between 1971 and 1977 (Figure 2). However, the locations and hence the stock associated with the catch in this period is not well known. Also age and length sampling of the population from 1979 showed evidence of a very high proportion (greater than 50% by number in the catch proportions-at-age) of old fish (11+) in the Campbell Island Rise population, with the age data suggesting that there was at least one very strong year class spawned in or around 1965 that remained a significant part of the population until the mid-1980s (see also Hanchet et al. 1998).

Without accurate knowledge of the actual catch before 1978, and with strong evidence that the population was not at an equilibrium age structure, recent assessments have assumed a non-equilibrium age structure as the initial state (e.g. Dunn & Hanchet 2017). For these assessments, the population model was initialised with a starting state, labelled $C_{initial}$, in 1979 and the numbers of individuals in the population at the start of the model were estimated for each age group (with both sexes combined and the numbers of males and females within each group assumed to be equal) as independent parameters (see $C_{initial}$ in Bull et al. 2012 for details).

However, an investigation of southern blue whiting models for estimating M found models to be unstable at MCMC when both $C_{initial}$ and M were left unconstrained (Roberts & Dunn 2017). The model was stabilised when $C_{initial}$ was discarded in favour of initial equilibrium age distribution. This required estimation of catch history in years 1971–1978, described in Section 2.2 below.

Differences in model structure between the Dunn & Hanchet (2017) base case model and the current assessment with respect to the initial age distribution and subsequent model parameters are given in Table 2. The DWG recommended that this model structure be adopted for the base case for the current stock assessment.

Table 2: Differences in model structure between the 2016 base case model and the current assessment.

Assessment year	Initial age distribution	Model start year	Initial YCS year	Initial catch year	Reference
2016	Numbers at ages 2–15+ estimated for the year 1979 (C_{intial})	1979	1977	1979	Dunn & Hanchet (2017)
2017	Equilibrium age distribution assumed for year 1960	1960	1958	1971	Roberts & Dunn (2017)

2.2 Observations

Available observations for the Campbell Island Rise stock are described in Large & Hanchet (2017). They include a time series of catches from 1979 to 2016; wide area acoustic biomass estimates, survey age frequency data for immature fish; and proportions-at-age from the commercial catch.

Previous stock assessments have begun from the year 1979 and have used catches for the Campbell Island stock taken from QMRs and MHRs. However, in order to start the model from 1960, catch estimates were required from the start of the fishery. It is known that the Russian fleet fished throughout the New Zealand EEZ from 1971 to 1977, and estimates of the total annual catch are available, but the proportion of the catch taken from the Campbell Island stock could not be determined (Hanchet 1998). For the purposes of the stock assessment it was assumed that the proportion of the catch taken from the Campbell Island stock in the period 1971–1977 equalled the proportion of the catch across the period since 1978 following Hanchet (1998) and Roberts & Dunn (2017). For the period 1978 to 2016–2017 this proportion equalled 0.70 and the resulting estimates for 1971–1977 are given in Table 3.

Table 3: Estimated catches and catch limits (TACCs) (t) of southern blue whiting at the Campbell Islands Rise for 1971 to 2016–17 (source: QMRs, MHRs, Roberts & Dunn 2017; italicised catch figures from 1971 to 1977 were estimated by applying the proportion of the catch taken from the Campbell Island stock since 1978 to the period 1971–1977, following Hanchet (1998) and Roberts & Dunn (2017) ; '−' denotes no catch limit in place).

Fishing year[1]	Estimated catch (t)[2]	Limit (t)	Fishing year[1]	Estimated catch (t)[2]	Limit (t)
1971	*7 260*	–	1995–96	14 959	21 000
1972	*18 010*	–	1996–97	15 685	30 100
1973	*33 856*	–	1997–98	24 273	35 460
1974	*29 458*	–	1998–00	30 386	35 460
1975	*1 660*	–	2000–01	18 049	20 000
1976	*11 929*	–	2001–02	29 999	30 000
1977	*18 453*	–	2002–03	33 445	30 000
1978	6 403	–	2003–04	23 718	25 000
1978–79	25 305	–	2004–05	19 799	25 000
1979–80	12 828	–	2005–06	26 190	25 000
1980–81	5 989	–	2006–07	19 763	20 000
1981–82	7 915	–	2007–08	20 996	20 000
1982–83	12 803	–	2008–09	20 483	20 000
1983–84	10 777	–	2009–10	19 040	20 000
1984–85	7 490	–	2010–11	20 224	23 000
1985–86	15 252	–	2011–12	30 982	29 400
1986–87	12 804	–	2012–13	21 321	29 400
1987–88	17 422	–	2013–14	28 606	29 400
1988–89	26 611	–	2014–15	23 397	39 200
1989–90	16 542	–	2015–16	22 100	39 200
1990–91	21 314	–	2016–17	19 875	39 200
1991–92	14 208	–			
1992–93	9 316	11 000			
1993–94	11 668	11 000			
1994–95	9 492	11 000			

1. Fishing years defined as 1 April to 30 September for 1978; 1 October to 30 September for 1978–79 to 1997–98; 1 October 1998 to 31 March 2000 for 1998–2000; 1 April to 31 March for 2000–01 to current.
2. Estimated catch. Estimates for 1971 to 1977 are taken from Roberts & Dunn (2017).

Previous models have also considered CPUE indices and trawl survey biomass indices. Standardised CPUE indices were last updated by Hanchet et al. (2006) but were not considered to be a useful index of abundance by the Middle Depths Working Group. Hence, we have not used these data in this assessment.

Dunn & Hanchet (2011) modelled observations from the sub-Antarctic trawl survey biomass and age frequencies time series. They found that, although the model fits suggested some consistency with the summer series biomass estimates, in general the trawl survey underestimated biomass at low stock sizes and overestimated biomass at high stock sizes. They concluded that the time series was not particularly useful for monitoring abundance in its present form. Hence, we ignore these data for this assessment.

2.2.1 Wide area acoustic surveys

Acoustic biomass estimates of southern blue whiting available on Campbell Island Rise during the fishing season were available from a wide area survey series conducted by the R.V. *Tangaroa* from 1993 to 2016 (see Table 4 and Large & Hanchet 2017 for details). The primary objective of the acoustic surveys has been to estimate the relative biomass of the adult spawning stock by year. A secondary objective has been to provide biomass estimates of pre-recruit fish and therefore the survey transects extend into 300 m where the younger fish live.

Southern blue whiting acoustic marks were identified as one of three categories of fish; juvenile, immature, or adult. The categories were broadly defined as adult (also known as mature) which consisted mainly of adult fish that were going to spawn that year; immature which consisted mainly of two-year-olds; and juvenile which consisted mainly of one-year-olds. Identification of each mark to a category was typically made at the time based on the results of research tows, the acoustic 'shape' of the mark, and its depth and location.

We used the survey estimates of immature and mature southern blue whiting, and assumed that these were relative estimates of mid-season biomass (i.e., after half the catch for that season has been removed), with a CV equal to the sampling CV estimated from the survey. We ignore the juvenile category in the assessment because biomass estimates of the juvenile category (mainly one-year-olds) were generally low and inconsistent with subsequent estimates of those year classes, and so were unlikely to be a good index of abundance (Dunn & Hanchet 2011).

The acoustic biomass observations were fitted using a lognormal likelihood,

$$-\log(L) = \sum_{i=1}^{n} \left(\log(\sigma_i) + 0.5 \left(\frac{\log(O_i/qE_i)}{\sigma_i} + 0.5\sigma_i \right)^2 \right)$$

$$\text{where } \sigma_i = \sqrt{\log\left(1 + c_i^2\right)}$$

where O_i are the observed biomass estimates at time i with CVs c_i, E_i are the model expected biomasses at time i and q is the catchability constant for the biomass series.

Table 4: R.V. *Tangaroa* juvenile, immature, and mature acoustic biomass estimates (t) and CV for the Campbell Island Rise 1993–2016 using the revised target strength derived by O'Driscoll et al. (2013).

Year	Juvenile		Immature		Mature		Total	Source
	Biomass	CV	Biomass	CV	Biomass	CV	Biomass	
1993	0	0.00	35 208	0.25	16 060	0.24	51 268	(Fu et al. 2013)
1994	0	0.00	5 523	0.38	72 168	0.34	77 691	(Fu et al. 2013)
1995	0	0.00	15 507	0.29	53 608	0.30	69 114	(Fu et al. 2013)
1998	322	0.45	6 759	0.20	91 639	0.14	98 720	(Fu et al. 2013)
2000	423	0.39	1 864	0.24	71 749	0.17	74 035	(Fu et al. 2013)
2002	1 969	0.39	247	0.76	66 034	0.68	68 250	(Fu et al. 2013)
2004	639	0.67	5 617	0.16	42 236	0.35	48 492	(Fu et al. 2013)
2006	504	0.38	3 423	0.24	43 843	0.32	47 770	(Fu et al. 2013)
2009	0	–	24 479	0.26	99 521	0.27	124 000	(Fu et al. 2013)
2011	0	–	14 454	0.17	53 299	0.22	67 753	(Fu et al. 2013)
2013	0	–	8 004	0.55	65 801	0.25	73 805	(O'Driscoll et al. 2014)
2016	775	0.37	4 456	0.19	97 117	0.16	102 348	(O'Driscoll et al. 2018)

2.2.2 Proportions-at-age in the commercial catch

Catch-at-age observations by sex were available from the commercial fishery for 1979 to 2016 from observer data, excluding 1987 (Figure 2). While length data were available for 1987, there were no otoliths aged from 1987, so the age length key was estimated using the length at age keys from 1986 and 1988, and adding or subtracting 1 year's growth respectively, following Hanchet & Ingerson (1995). Commercial catch-at-age data were fitted to the model as proportions-at-age by sex, where associated CVs by age were estimated using the NIWA catch-at-age software by bootstrap (Bull & Dunn 2002). The catch proportions-at-age data were fitted to the modelled proportions-at-age composition using a multinomial likelihood,

$$-\log(L) = -\log(N!) + \sum_i \left[\log\left((NO_i)!\right) - NO_i \log(E_i) \right]$$

where O_i are the observed proportions-at-age i, E_i are the model expected proportions-at-age i, and N is the effective sample size. A robustifying function was used to avoid division by zero errors (see Bull et al. 2012 for more details). Proportions-at-age data were derived from the aged otoliths collected by observers and the length frequency of the catch. Large & Hanchet (2017) described the catch-at-age data available for the assessment models from 1990, and data before 1990 were described by Hanchet et al. (2006). The derivation of the assumed multinomial sample sizes for the proportions-at-age data is described below.

2.3 Model estimation

The model parameters were estimated by minimising an objective function, which is the sum of the negative log-likelihoods from the data, negative-log priors (in a Bayesian analysis), and penalties that constrain the parameterisations, i.e., the objective function in a Bayesian analysis for **p**, the vector of the free parameters, L the likelihood function, and O_i the ith observation was

$$\text{Objective }(\mathbf{p}) = -\sum_i \log\left[L(\mathbf{p} | O_i) \right] - \log\left[\theta(\mathbf{p}) \right],$$

where θ is the joint prior (and penalty) density of the parameters **p**. The observations, likelihoods, penalties, and priors are described later.

Initial model fits were evaluated at the maximum of the posterior density (MPD) by inspecting the model fits and residuals. At the MPD, the approximate covariance matrix of the free parameters was

calculated as the inverse of the approximation to the Hessian and the corresponding correlation matrix also calculated.

To estimate the joint posterior distribution of the parameters in a Bayesian analysis, CASAL uses a straightforward implementation of the Metropolis algorithm (Gelman et al. 1995, Gilks et al. 1998) to execute the Markov Chain Monte Carlo (MCMC). The Metropolis algorithm draws a sample from a Bayesian posterior distribution, and calculates the posterior density π, scaled by an unknown constant. The algorithm generates a 'chain' or sequence of values. Typically the beginning of the chain is discarded and every nth element of the remainder is taken as the posterior sample. The chain is produced by taking an initial point x_0 and repeatedly applying the following rule, where x_i is the current point; (i) draw a candidate step s from a proposed distribution J, which should be symmetric about zero i.e., $J(s)=J(-s)$, (ii) calculate $r=\min(\pi(x_i + s) / \pi(x_i),1)$, and (iii) let $x_i+1=x_i + s$ with probability r, or x_i with probability 1-r.

A point estimate (i.e., the MPD) was produced, along with the approximate covariance matrix of the parameters (as the inverse Hessian) (see Bull et al. 2012 for more detail), and used as the starting point for the chain.

MCMCs were estimated using single chains, with a burn-in length of 1×10^6 iterations, with every 10 000th sample taken from the next 1×10^7 iterations (i.e., a systematic sample of length 1000 was taken from the Bayesian posterior). A prior step was undertaken to improve mixing at MCMC (following the approach of Roberts & Doonan 2016) in which the covariance matrix was recalculated empirically from the 100 samples obtained from a single MCMC chain of length 1×10^6 iterations (no burn in). These initial samples were then discarded and the chain was restarted using the revised covariance matrix.
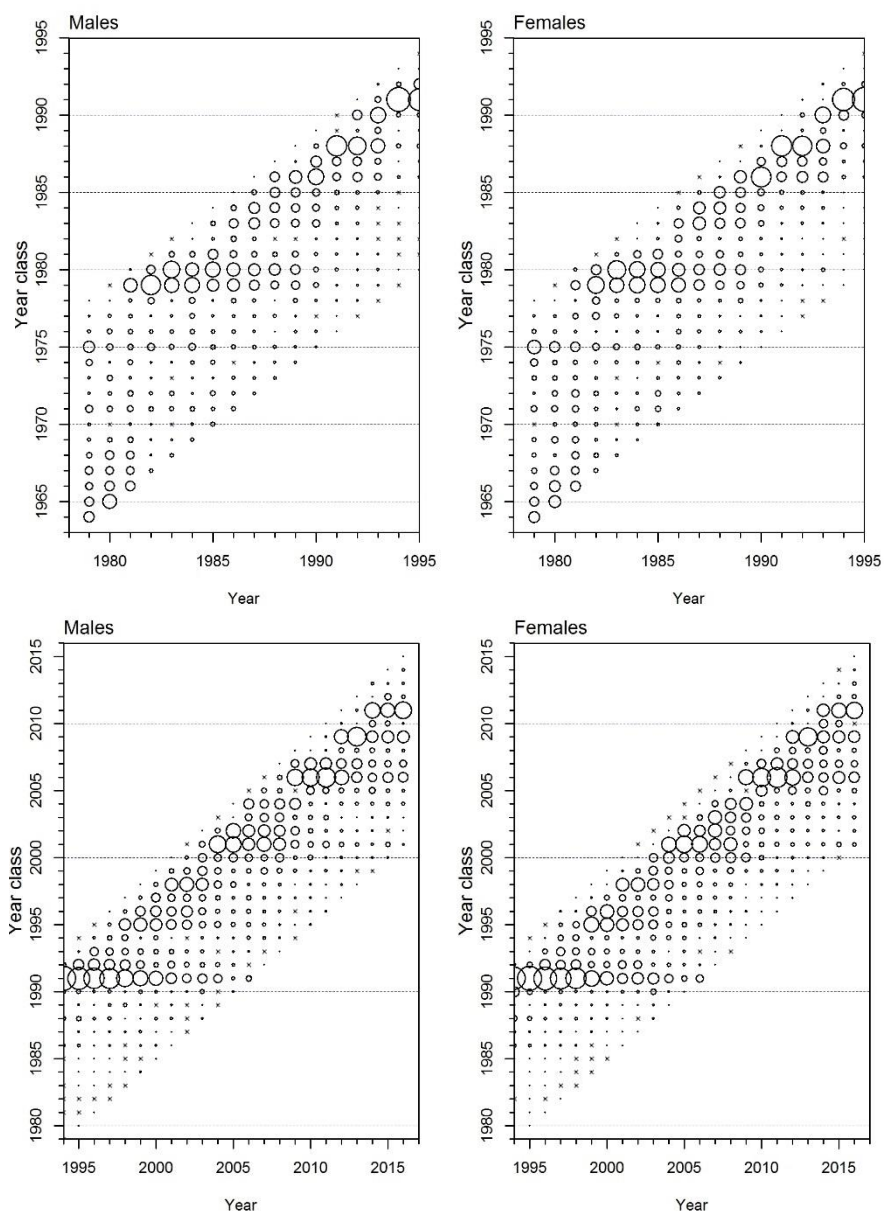
Figure 2: Commercial catch proportions at age for the Campbell Island stock by sex and year class, 1979–2016 for ages 2–15+. Symbol area is proportional to the proportions-at-age within the sampling event.

2.3.1 Prior distributions and penalties

With the exception of natural mortality, which was assumed to be lognormal with mean 0.2 and CV 0.2, the assumed prior distributions used in the assessment were intended to be non-informative with wide bounds (Table 5). The prior assumed for the relative year class strengths was lognormal, with mean 1.0 and CV 1.3, for all year classes.

Table 5: The parameters, number of degrees of freedom (N), priors (including distributions, and means and CVs for the lognormal), and bounds assumed for estimated parameters for the models.

Parameter	N	Distribution	Mean	CV	Lower	Upper
B_0	1	Uniform-log	–	–	30 000	800 000
*Initial population (by sex)	14	Uniform	–	–	2e+2	2e+9
Male maturation ogive	5	Uniform	–	–	0.001	0.999
Female fishing selectivity	5	Uniform	–	–	0.001	0.999
**Year class strength	56	Lognormal	1.0	1.3	0.001	100
Acoustic catchability q						
Mature	1	Lognormal	0.54	0.44	0.1	1.5
Immature	1	Uniform	–	–	0.1	1.5
*Natural mortality (average)	1	lognormal	0.2	0.2	0.075	0.325
*Natural mortality (difference)	1	Normal	0.0	0.05	-0.05	0.05

*Initial population size-at-age ($C_{initial}$) and natural mortality parameters were only estimated on one sensitivity run each.
**37 YCS parameters were estimated for the sensitivity run estimating initial population size-at-age.

A new log-normal prior was developed for the wide area acoustic survey catchability coefficient obtained using the approach of Cordue (1996). The main difference between the revised prior and the original prior used in the 2013 assessment (Dunn & Hanchet 2015) was the inclusion of uncertainty over the tilt angle of southern blue whiting. Individual priors were developed for the key factors, including target strength, acoustic system calibration, target identification, shadow or dead zone correction, and spatial availability and these were then aggregated to develop an overall lognormal prior which had a mean of 0.54 and CV of 0.44 (Table 6).

Table 6: Original and revised 'best' and lower and upper bounds for the factors for the acoustic catchability prior. A lognormal prior with mean 0.54 and CV 0.44 was used for the assessment.

Factor	Original			Revised		
	Lower	Best	Upper	Lower	Best	Upper
Target strength: Uncertainty	0.72	0.90	1.13	0.80	1.00	1.20
Target strength: Tilt angle	-	-	-	0.25	0.70	1.00
Target identification	0.90	1.15	1.45	0.85	1.00	1.15
Vertical availability	0.75	0.85	0.95	0.90	0.95	1.00
Areal availability	0.90	0.95	1.00	0.80	0.90	1.00
System calibration	0.90	1.00	1.10	0.90	1.00	1.10
Combined	0.39	0.84	1.71	0.11	0.60	1.52
Lognormal parameters	mu=0.87, CV=0.30			mu=0.54, CV=0.44		

Natural mortality was estimated to be 0.2 y^{-1} by Hanchet (1991a). When estimated in the current model, natural mortality was parameterised by the average of male and female, with the difference estimated with an associated normal prior with a mean of zero and bounds of 0.05 y^{-1}. The prior on the average natural mortality was assumed to be a normal distribution with mean of 0.2 y^{-1} and CV 0.2 following Dunn & Hanchet (2017).

Penalty functions were used to constrain the model so that any combinations of parameters that did not allow the historical catch to be taken were strongly penalised. A small penalty was applied to encourage the estimates of year class strengths to have mean equal to one.

2.3.2 Process error and data weighting

Lognormal errors, with known CVs, were assumed for the relative biomass and proportions-at-age data. The CVs available for these data allow for sampling error only. However, additional variance, assumed to arise from differences between model simplifications and real-world variation, was added to the sampling variance. The additional variance, termed "process error", was estimated in each of the initial runs (MPDs) using all the available data. Process errors were estimated separately for the proportion-at-age data, and for the acoustic estimates from the wide area surveys.

Estimates of the sample size for the proportions-at-age observations were made via a two-step process. First, the sample sizes were derived by assuming that the relationship between the observed proportions, E_i, and estimated CVs, c_i, followed that for a multinomial distribution with unknown sample size N_j. The estimated sample size was then derived using a robust non-linear least squares fit of $\log(c_i) \sim \log(P_i)$. Second, estimates of the effective sample size, N_j', were obtained by adding additional process error, N_{PE}, to this sample size estimate using Method TA1.8 in Francis (2011) — i.e., from an initial MPD model fit, an estimate of the additional process error was made such that the standardised residuals from the mean observed age and mean expected age in each year had mean equal to one.

Estimates of the process error CV for the biomass observations were made by fitting the process error within each MPD run, where the applied CV c_i' was determined from the process error c_{PE} and the observed CVs c_i by,

$$c_i' = \sqrt{c_i^2 + c_{PE}^2}$$

However, in all models the estimated process error for the acoustic biomass estimates was zero.

2.3.3 Model runs

We considered five model runs: a base case and four sensitivity tests (Table 7). As recommended by the DWG, the base case run had an equilibrium age distribution in the year 1960, YCSs were estimated from 1958 to 2013, the 1971 to 2016 catch history was used, and natural mortality was assumed equal to 0.2. The first three sensitivity tests considered the influence of uncertainty in natural mortality in the model: the first two assumed natural mortality equal to 0.15 or 0.25; and the third allowed for the estimation of the natural mortality rate for males and females. The last sensitivity run was essentially an update of the previous base case assessment from Dunn & Hanchet (2017), which estimated age-specific population size ($C_{initial}$) parameters for the year 1979 and assumed a natural mortality equal to 0.2.

Table 7: Model run labels and descriptions for the model runs.

Model type	Model label	Description
Sensitivity	1.1	Model estimating age-specific population size (C_{inital}) parameters for the year 1979, YCSs estimated for years 1977–2013, catch history for years 1979–2016 and natural mortality equal to 0.20.
Base case	2.1	Model with equilibrium age distribution for the year 1960, YCSs estimated for years 1958–2013, catch history for years 1971–2016, natural mortality equal to 0.20.
Sensitivity	3.1	Model 2.1, but with natural mortality equal to 0.15.
Sensitivity	3.2	Model 2.1, but with natural mortality equal to 0.25.
Sensitivity	3.3	Model 2.1, but with natural mortality estimated.

3. RESULTS

3.1 MPD results

The spawning stock biomass trajectories for the MPD fits are shown for the base case (model 2.1) in Figure 3. The relative year class strengths, and fits to the acoustic indices and mean age are also shown in Figure 3. Fits to the mean age and acoustic indices were generally good, but the model was unable to fit the very high mature and immature biomass seen in the 2009 acoustic survey.

The fits to the mature biomass acoustic indices and the age data looked very similar for the sensitivity runs (not shown). M was estimated to be 0.14 for males and 0.16 for females in model run 3.3.

Figure 3: Results of MPD fits for model run 2.1 showing (a) estimated SSB trajectory (with B_0 shown as a solid line and 20% B_0 and 40% B_0 shown as a dashed lines) 1979–2016; (b) estimated relative year class strength (with the average of one shown as a dashed line); (c) observed (o) and expected (e) mean age fits for the commercial catch-at-age data; and (d) observed (o) and expected (e) mature (black) and immature (red) acoustic biomass indices.

3.2 MCMC results

3.2.1 MCMC diagnostics

MCMC traces were reasonably good for B_0 and for B_{2016} (%B_0) for the base case model (Figure 4) and for the sensitivity model runs (not shown). Plots of relative jump size and autocorrelation were generally good, although the early year class strengths for 1958–1964 parameters performed poorly and may be poorly estimated by these models.
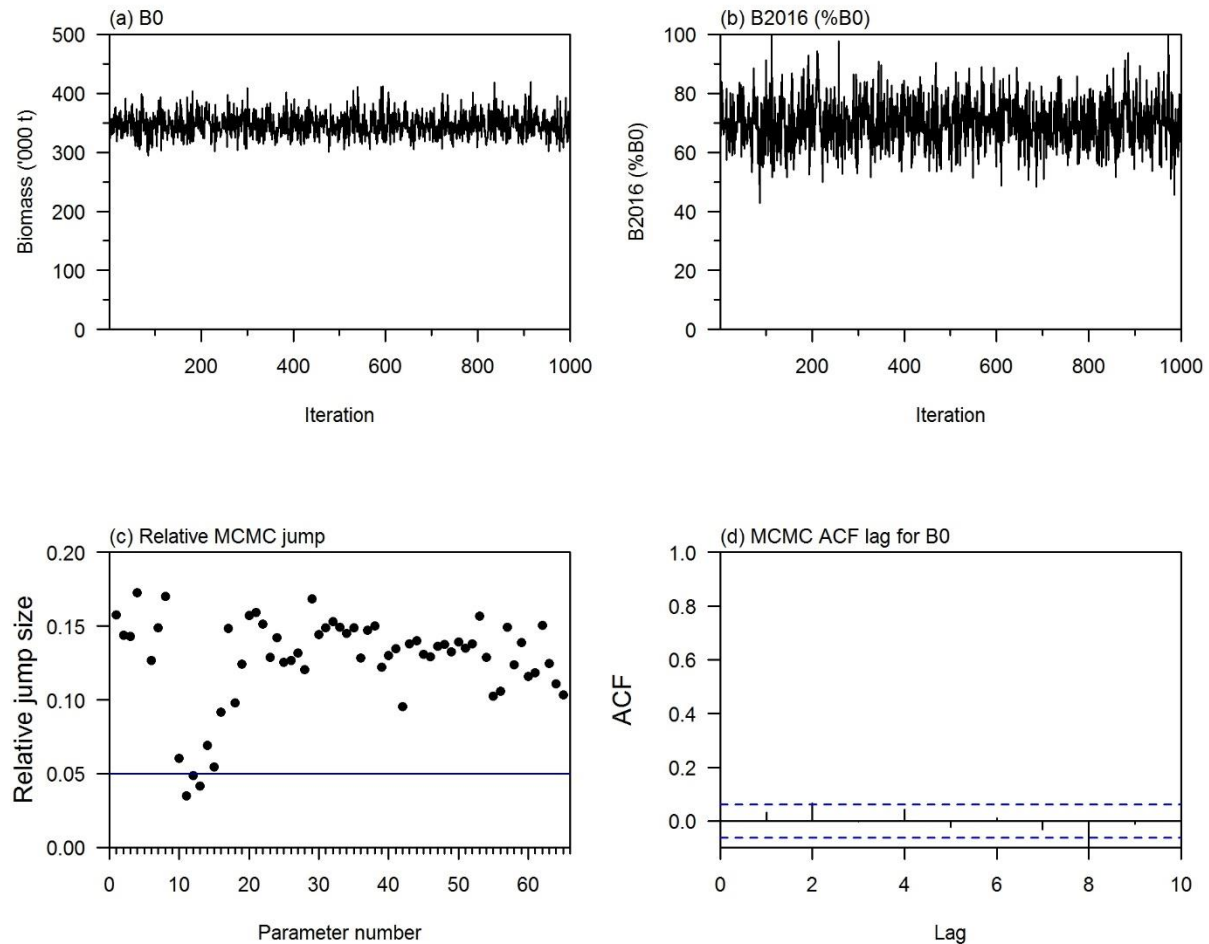

Figure 4: MCMC posterior trace plots for the base case: (a) B_0, (b) current status (%B_{2016}/B_0), (c) median relative jump size for all parameters (x-axis, labelled 1–65 where the line indicates a threshold test for non-convergence – the year class strength estimates for 1959–1961 were the only parameters falling below this threshold), and (d) autocorrelation lag plot for B_0 (where the dashed lines indicate an autocorrelation that is higher than 0.05 or lower than -0.05).

3.2.2 MCMC estimates

Base case model run

The estimated MCMC marginal posterior distributions for parameters of interest are shown for the base case (model run 2.1) in Figure 5 and Figure 6, and the results summarised in Table 8 and Table 9. This run suggested that the spawning stock biomass increased from 1960 to 1970 as a result of a strong year class and no fishery exploitation. It then steadily declined until 1993, followed by a large increase to 1995 resulting from the recruitment of the strong 1991 year class. The spawning population then declined steadily from 1997 until 2008, and then showed a moderate increase to 2015 as the 2006 and 2009 and then 2011 year classes recruited to the fishery. At the start of fishing in 1971, the spawning stock biomass was estimated to be at about 130% B_0. During the late 1980s and early 1990s the biomass was estimated to have dropped below 20% B_0 for several years but then to have increased to about 60% B_0 when the strong 1991 year class entered the fishery. Since then the spawning stock biomass is estimated to have been above 40% B_0.

Year class strength has been highly variable over the course of the fishery. The 1991 year class was about six times stronger than any other year classes until at least 2006, and gave rise to the large increase in biomass seen during the mid-1990s. There were several above average year classes during the mid to late 1990s and early 2000s, but these contributed only a small proportion of the catch and have probably been largely removed from the population. The size of the 2006, 2009, and 2011 year classes was estimated to be at about 3–4 times the average, with large numbers caught in the fishery and large numbers of those year classes observed by recent acoustic surveys.

The estimate of the median mature biomass acoustic q was 0.29, which is less than the prior value of 0.54. This would suggest that the acoustic surveys are underestimating the adult biomass, or that the prior itself was poorly estimated. The estimate of the median immature biomass acoustic q was 0.21, which is less than the estimate for the median mature biomass.

Table 8: Bayesian median and 95% credible intervals of equilibrium (B_0), current spawning stock biomass, and current status (%B_0) for model run 2.1 (base case), and the sensitivity cases.

Model	B_0	B_{2016}	B_{2016} (%B_0)
Base case	345 100 (311 100–389 800)	239 700 (173 100–328 600)	70 (54–86)
1.1	370 300 (327 000–428 500)	246 700 (175 600–340 100)	67 (54–80)
3.1	333 600 (311 400–360 800)	186 400 (138 100–254 500)	56 (44–71)
3.2	424 100 (361 800–521 000)	324 800 (230 300–479 500)	77 (59–100)
3.3	335 200 (307 500–380 500)	209 300 (146 000–313 100)	62 (45–84)

Table 9: Bayesian median and 95% credible intervals of the catchability coefficients (q) for the wide area acoustic biomass indices and estimates of natural mortality for model runs 2.1 (base case) and the sensitivity cases.

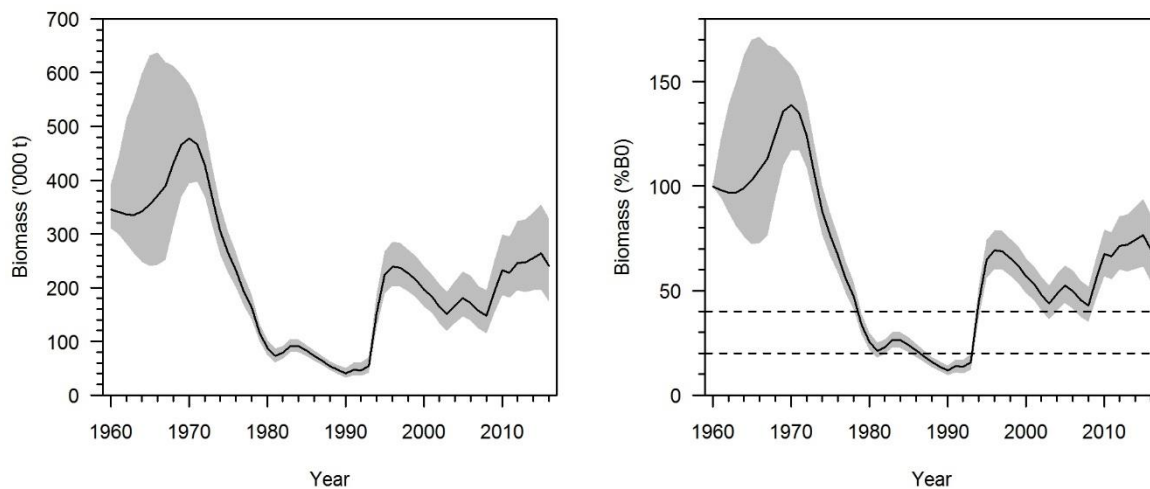| Model	Catchability		Natural mortality	
	Immature	Mature	Male	Female
Base case	0.26 (0.21–0.31)	0.36 (0.29–0.42)	–	–
1.1	0.26 (0.21–0.31)	0.35 (0.28–0.42)	–	–
3.1	0.39 (0.33–0.46)	0.48 (0.41–0.56)	–	–
3.2	0.16 (0.12–0.20)	0.25 (0.18–0.32)	–	–
3.3	0.32 (0.21–0.45)	0.42 (0.30–0.54)	0.17 (0.13–0.21)	0.18 (0.14–0.22)

Figure 5: MCMC median and 95% credible intervals of the trajectory of (left) spawning stock biomass and (right) stock status (%B_0) for the base case.
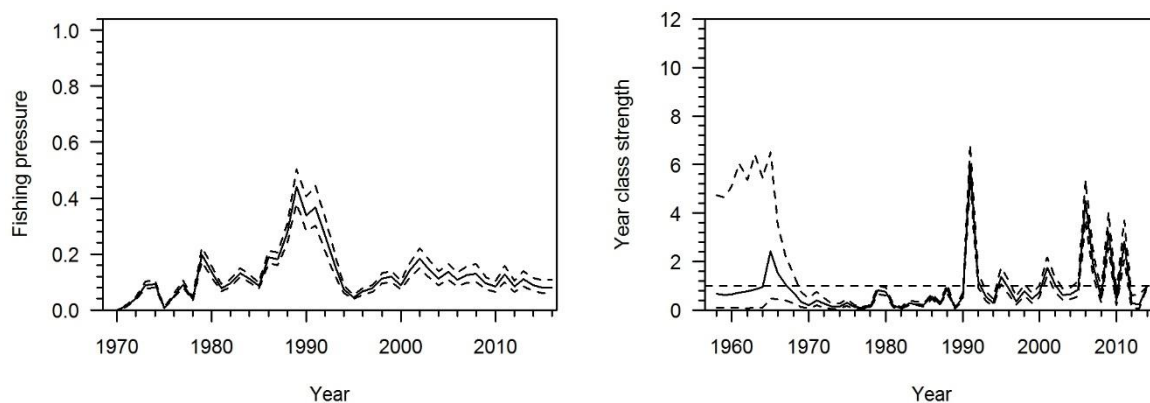

Figure 6: MCMC median and 95% credible intervals for (left) exploitation rates and (right) relative year class strength for the base case.

Model sensitivity runs

The estimated MCMC marginal posterior distributions for parameters of interest are shown for the sensitivity model runs in Figures 7–10, and the results summarised in Tables 8 and 9.

The biomass trajectories for all four sensitivity runs showed very similar patterns to the base case. However, the key difference was in the level of biomass and stock status. As would be expected, the estimate of B_0, current biomass and current status was lower for model run 3.1 but higher for model run 3.2 as a direct result of the assumed value of M. In model 3.3, where M was estimated at 0.17–0.18, the estimates of B_0 and current biomass were lower than for the base case, but the current status was similar to the base case. Biomass estimates and stock status from model run 1.1, which assumed an initial non-equilibrium age structure in 1979, are all slightly higher than the base case.
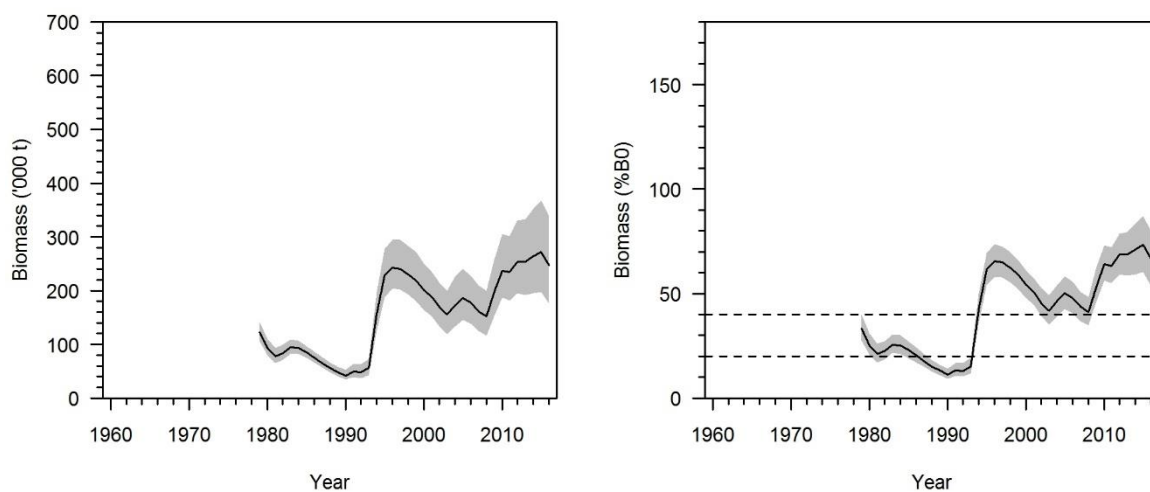
Figure 7: MCMC median and 95% credible intervals of the trajectory of (left) spawning stock biomass and (right) stock status (%B_0) for run 1.1.
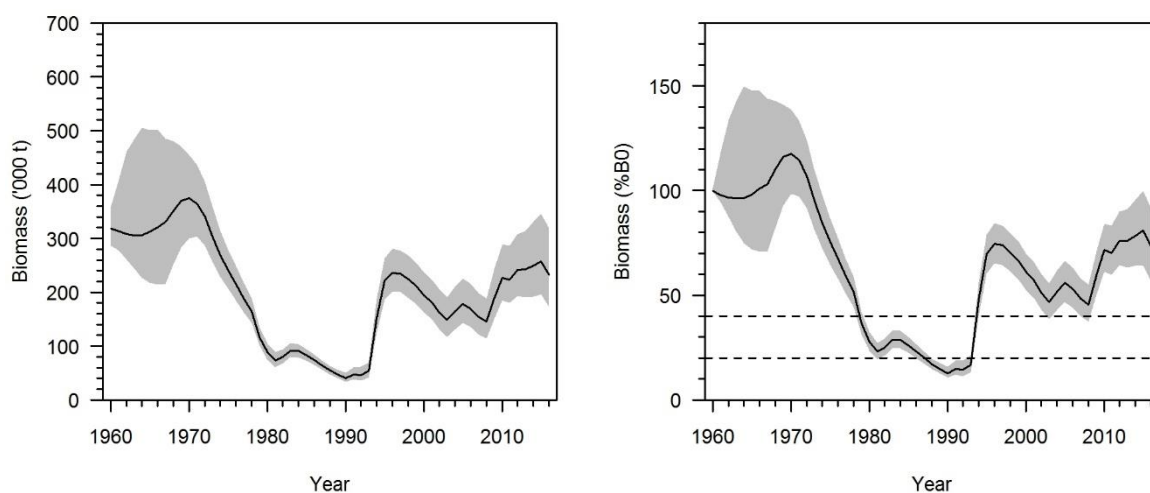

Figure 8: MCMC median and 95% credible intervals of the trajectory of (left) spawning stock biomass and (right) stock status (%B_0) for run 3.1.
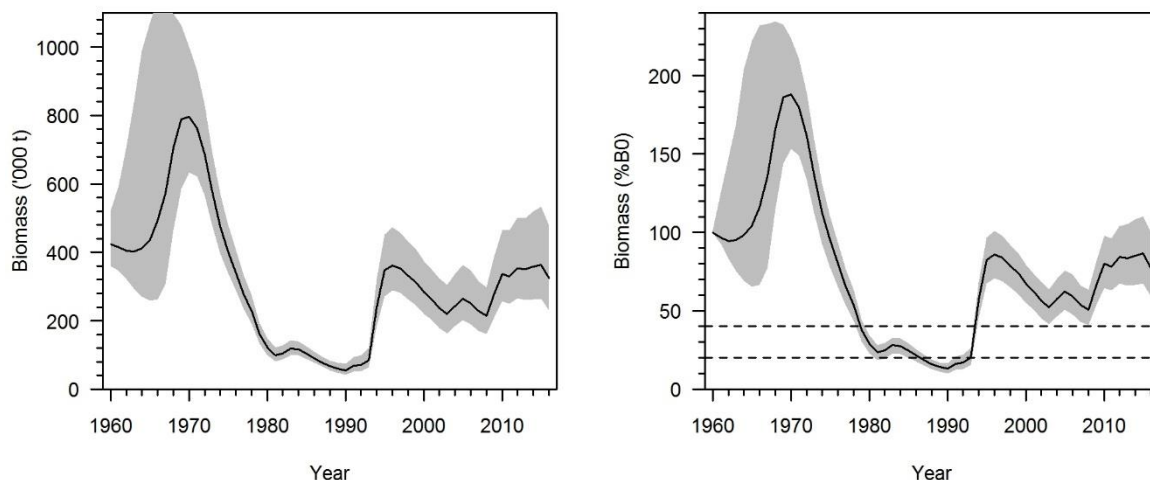
Figure 9: MCMC median and 95% credible intervals of the trajectory of (left) spawning stock biomass and (right) stock status (%B_0) for run 3.2.
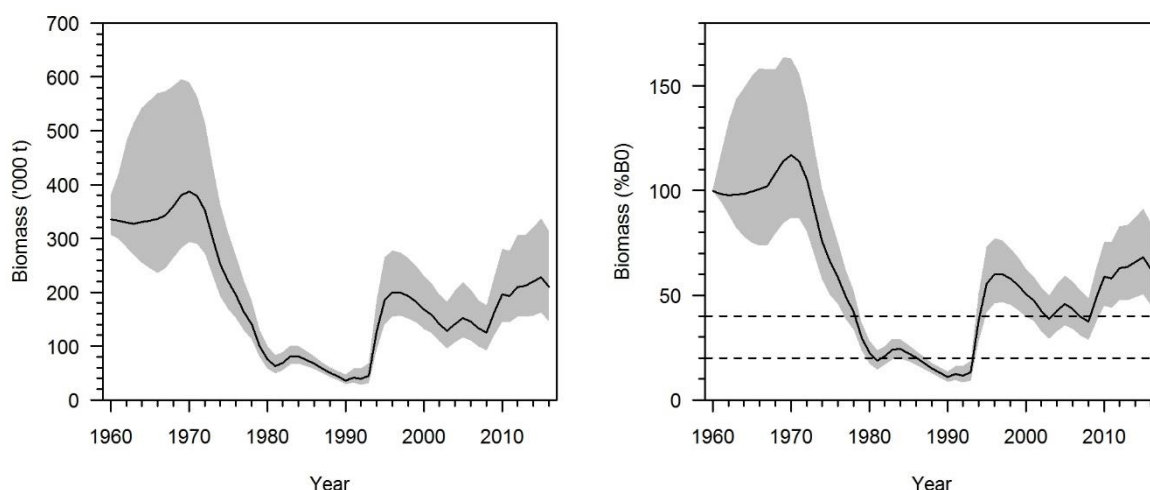

Figure 10: MCMC median and 95% credible intervals of the trajectory of (left) spawning stock biomass and (right) stock status (%B_0) for run 3.3.

3.3 Projections

Projections were made for the base case model and the sensitivity model runs assuming fixed catch levels of 23 000 t and 40 000 t for the years 2018 to 2021. Projections were made using the MCMC samples, with recruitments drawn randomly from the distribution of year class strengths for the period 1958–2013 estimated by the model and applied from year 2014 onwards (Figure 11). Because of the link between mean size at age of fish in the population and the population density, projections assumed that the mean size at age would remain at 2016 estimates, rather than return to the average size at age that might be expected at lower abundances.
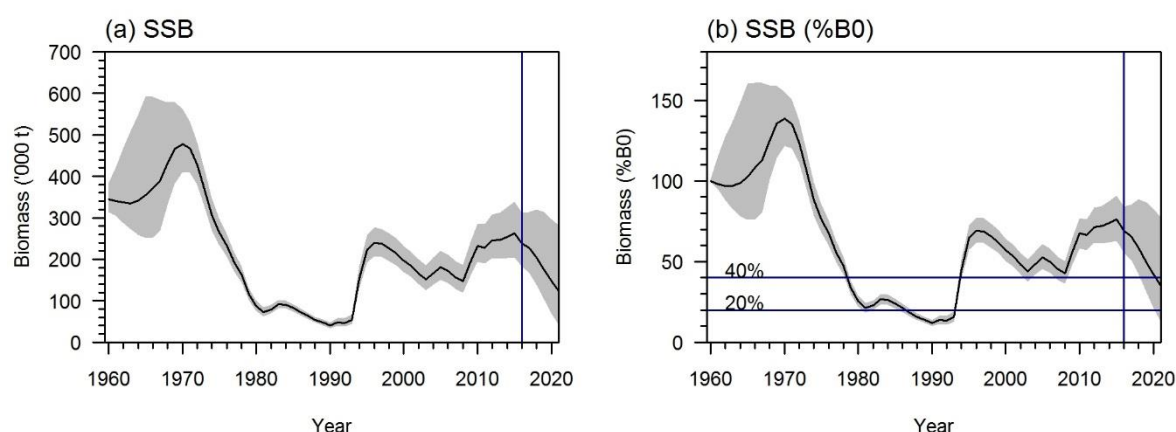
Figure 11: MCMC posterior plots for the median (solid line) and 95% credible intervals for (a) spawning stock biomass and (b) stock status (%B_0) for a catch of 40 000 t for the base case assuming average recruitment. Horizontal lines indicate 40% and 20% B_0, and the vertical line represents the beginning of the projection period (2016–2020).

For each scenario, the probability that the mid-season biomass for the specified year will be less than the threshold level (20% B_0) is given in Table 10. The probability of dropping below the threshold biomass at catch levels of 23 000 t was less than 5% for all models and all years. The probability of dropping below the threshold biomass at catch levels of 40 000 t exceeded 10% by 2020 for runs 3.1 and 3.3 and by 2021 for all models except run 3.2. Under average recruitment conditions the biomass was expected to steadily decline under both catch scenarios in all model runs.

Table 10: Probability that the projected mid-season spawning stock biomass for 2017–2021 will be less than 20% B_0 at a projected catch of 23 000 t and 40 000 t, for model run 1.1 (previous base case), 2.1 (M=0.20 y^{-1}), 3.1 (M=0.15 y^{-1}), and 3.2 (M=0.25 y^{-1}) and 3.3 (M= estimated); run 1.1 assumed average recruitment over the period 1977–2013 for 2014+, whereas runs 2.1, 3.1, 3.2 and 3.3 all assumed average recruitment over the period 1958–2013.

Model	Catch			Pr(B < 0.2B_0)		
	(t)	2017	2018	2019	2020	2021
1.1	23 000	0.000	0.000	0.000	0.001	0.005
	40 000	0.000	0.000	0.001	0.028	0.137
2.1	23 000	0.000	0.000	0.000	0.000	0.004
	40 000	0.000	0.000	0.002	0.038	0.138
3.1	23 000	0.000	0.000	0.002	0.008	0.037
	40 000	0.000	0.000	0.018	0.178	0.418
3.2	23 000	0.000	0.000	0.000	0.000	0.001
	40 000	0.000	0.000	0.000	0.002	0.024
3.3	23 000	0.000	0.000	0.001	0.008	0.023
	40 000	0.000	0.000	0.015	0.111	0.280

4. DISCUSSION

The results of the base case assessment suggested that the spawning stock biomass of the Campbell Island Rise in 2016 was 239 700 t (95% C.I.s 173 100–328 600 t), and was at 70% B_0 (95% C.I.s 54–86%). Projections with an annual catch of 40 000 t suggested that the spawning stock biomass was expected to decline steadily to 36% B_0 by 2021. There was little difference in the biomass trend between the base case and the sensitivity model runs used for the projections, although model run 3.1 (low M) was less optimistic in the current status and projected status and model run 3.2 (high M) was more optimistic.

The initial equilibrium age model structure used here as the base case was considered by the Deepwater and Middle Depths Working Group to be an improvement over the non-equilibrium age structure model used in previous assessments because it increased the number of year classes which could be estimated in the model and had performed well under simulations carried out by Roberts & Dunn (2017). Although the catch history for the early period of the fishery (1970–1977) is not well known, additional sensitivity runs showed that estimates of B_0, $B_{current}$ and stock status were relatively insensitive to higher or lower catches during that period. Furthermore, an extensive trawl survey on the Campbell Plateau carried out by the Japanese research vessel *Kaiyo Maru* in the summer of 1970/1971 caught southern blue whiting with a strongly unimodal size distribution with a peak at 39–40 cm (Hanchet 1991b). Ageing suggested that these fish were mainly age 4+ and 5+ (Anon 1972), which corresponds to the 1965 and 1966 year classes. This is consistent with the above average strength of these year classes estimated in the base case model (Figure 6).

The assessment was strongly influenced by the high biomass estimates from the last four acoustic surveys. These surveys observed some of the highest estimates of adult and immature biomass since the survey series began in 1993. There is no reason to doubt that the 2006, 2009, and 2011 year classes are strong, but the size of the 2011 year class is still not well-estimated. The relative strength of these year classes was virtually identical between the models with the 2006 year class being stronger than the 2009 year class followed by the 2011 year class.

The last assessment of the Campbell Island Rise stock was carried out in 2016 (Dunn & Hanchet 2017). Estimates of B_{1979} and the stock trajectory up to 2016 were similar to the previous assessment, but the current model results had several slight differences: (i) a slightly lower B_0, (ii) a slightly higher current stock status, and (iii) higher estimate of the 2006, 2009 and 2011 year class strengths.

The other important difference between previous assessments and this one was the estimate of B_0, which has been poorly estimated in southern blue whiting models as a consequence of a few very strong year classes strongly influencing estimates of average recruitment. In recent years, the influx of several strong year classes impacted the estimate of average recruitment, and hence resulted in changing estimates of B_0. Although an additional 20 year classes were estimated in the current assessment (from 1958 to 1977), B_0 was estimated to be similar to that in the 2016 assessment.

The estimate of the median mature biomass acoustic q was 0.36 for the base case. This estimate was substantially lower than the mean of the prior of 0.54. This suggests that either the acoustic surveys were underestimating the adult biomass, or that the acoustic adult biomass prior may not have been well-specified (was biased). The main uncertainty in the estimate of the prior is the assumption of the target strength-fish length relationship. We note that further work is underway to determine the average tilt angle of southern blue whiting *in situ*, so that this prior can be more accurately and precisely estimated.

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