

New Zealand Fisheries  
Assessment Report  
2010/1  
January 2010  
ISSN 1175-1584 (print)  
ISSN 1179-5352 (online)

Assessment models for Antarctic toothfish  
(*Dissostichus mawsoni*) in the Ross Sea  
including data from the 2006–07 season

A. Dunn  
S. M. Hanchet

**Assessment models for Antarctic toothfish  
(*Dissostichus mawsoni*) in the Ross Sea  
including data from the 2006–07 season**

A. Dunn<sup>1</sup>  
S. M. Hanchet<sup>2</sup>

<sup>1</sup>NIWA  
Private Bag 14901  
Wellington 6241

<sup>2</sup>NIWA  
P O Box 893  
Nelson 7040

**Published by Ministry of Fisheries  
Wellington  
2010**

**ISSN 1175-1584 (print)  
ISSN 1179-5352 (online)**

©  
**Ministry of Fisheries  
2010**

Dunn, A.; Hanchet, S.M. (2010).  
Assessment models for Antarctic toothfish (*Dissostichus mawsoni*) in the Ross Sea  
including data from the 2006–07 season.  
*New Zealand Fisheries Assessment Report 2010/1*. 28 p.

This series continues the informal  
New Zealand Fisheries Assessment Research Document series  
which ceased at the end of 1999.

## EXECUTIVE SUMMARY

**Dunn, A.; Hanchet, S.M. (2010). Assessment models for Antarctic toothfish (*Dissostichus mawsoni*) in the Ross Sea including data from the 2006–07 season. *New Zealand Fisheries Assessment Report 2010/1*. 28 p.**

We provide an update of the Bayesian sex and age structured population stock assessment model for Antarctic toothfish (*Dissostichus mawsoni*) in the Ross Sea (Subareas 88.1 and SSRUs 88.2A–B), using revised catch, catch-at-age, and tag-recapture data for the 2007 season. The updated reference model resulted in a slightly lower estimate of initial biomass than the 2006 base case. The inclusion of the 2007 recaptures of 2006 tags released had the most substantive impact on the model estimates.

Models using tag-recapture data from all vessels are also presented. Inclusion of all vessels tagging data resulted in a more optimistic assessment. The more optimistic estimate was probably due to the lower recapture rate by non-New Zealand vessels, particularly in the early years. The reason for this is unclear, and may be related to different distributions of fishing effort by different vessels, to poorer survival of tagged fish, or to poorer detection rates. However, if data from all vessels were restricted to 2007 recaptures of 2006 releases, then model estimates were more similar to the 2007 reference model.

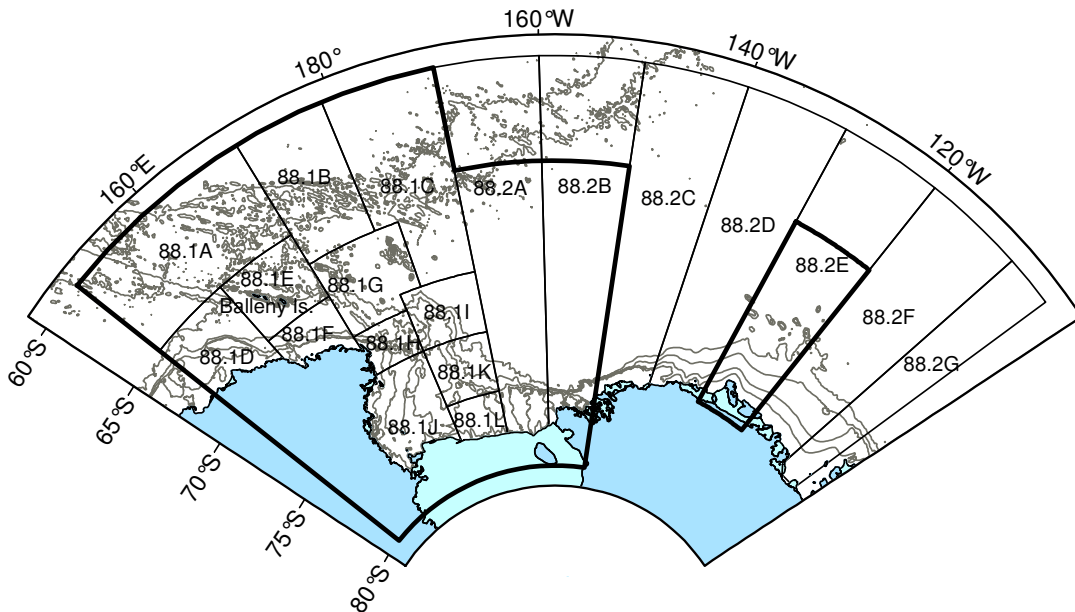
Overall, model fits to the data were adequate, and, as in previous assessments, the tag-release and recapture data provided the most information on stock size. Monte-Carlo Markov Chain (MCMC) diagnostics suggested little evidence of non-convergence in the key biomass parameters. MCMC estimates of initial (equilibrium) spawning stock abundance ( $B_0$ ) for the 2007 reference model were estimated as 71 200 t (95% C.I.s 59 570–87 900 t), and current ( $B_{2007}$ ) biomass was estimated as 81.9% (78.4–85.4%). Estimated yield for the reference case, using the CCAMLR decision rules, was 2700 t.

Similar estimates of initial biomass and yield were obtained for models where the north fishing selectivity was assumed to be logistic (median  $B_0=78\ 480$  t, yield=2988 t), and where the 2007 recaptures from all vessels were added to the New Zealand tag-release and recapture data (median  $B_0=81\ 100$  t, yield=3099 t). If tag-release and recapture data for all vessels in all years were used, then the estimated initial biomass was higher ( $B_0=110\ 130$  t), and corresponding yields were also higher (yield=4200 t).

## 1. INTRODUCTION

The exploratory fishery in the Ross Sea (defined here as Subareas 88.1 and SSRUs 88.2A–B, see Figure 1) was initiated by a New Zealand longline vessel in 1997<sup>1</sup>. Since then, New Zealand vessels, and more recently vessels from other countries, have returned each summer to fish in this area. During that time the fishery for the Ross Sea has increased to about 3000 t per annum, and for 2007, the catch limit for the Ross Sea was set at 3072 t (SC-CAMLR-XXV 2006).

The catch limits adopted by CCAMLR for 2007 were determined from yield estimates from an integrated stock assessment model of Antarctic toothfish in the Ross Sea (Dunn & Hanchet 2006a). That model assumed a single homogeneous area with three geographically defined fisheries (shelf, slope and north, see later). Data included were based on total catch (C2 data); catch-at-age frequencies using the tree regression post-stratification by Phillips et al. (2005) and reported by Hanchet et al. (2006); CPUE indices (Dunn 2006c) including data for the 2006 year; and tag-release and recapture data up to 2006 (Dunn & Hanchet 2006c).



**Figure 1: CCAMLR Subareas 88.1 and 88.2, small scale statistical areas (SSRUs), and the Ross Sea (bounded region). Depth contours plotted at 500, 1000, 2000, and 3000 m.**

Previously, Dunn et al. (2004) introduced CASAL as a method for the assessment of Antarctic toothfish in the Ross Sea. In 2005, Dunn et al. (2005a) extended the model and also investigated an implementation of 2- and 3-area models, following the recommendations of SC-CAMLR-XXIII (2004). While Dunn et al. (2004) found that the single-area model of Antarctic toothfish fishery in the Ross Sea had some deficiencies in representing the observations, Dunn et al. (2005a) found that the data requirements of a multi-area model probably exceeded currently available information on movements and stock structure of Antarctic toothfish. However, they also found that, in a simulation experiment, a single-area model was likely to be conservative (i.e., estimates of current and equilibrium biomass were strongly biased low). Further, preliminary investigations of potential bias of tag based abundance estimators found that, under simplistic assumptions of fish distribution, that the

<sup>1</sup> Note that this report uses the CCAMLR split year that is defined from 1 December to 30 November. Hence, the term “year” refers to the fishing season in which most fishing occurs, e.g., the period 1 December 2004 to 30 November 2005 is labelled the 2005 year.

abundance estimates resulting from the current tagging programme may have under-estimated the true abundance (Dunn 2006a). We note, however, that further investigation of the impact of mixing assumptions for tag data in the Ross Sea is required.

Here, we update the model of Dunn & Hanchet (2006a) by updating the catch, catch-at-age frequencies for 2007, the tag-release and recapture observations for 2007 (Dunn et al. 2007a), and other minor model changes described by Dunn & Hanchet (2007). Although standardised CPUE indices have been calculated annually and included previous models, there has been concern that they are not monitoring vulnerable abundance (Dunn & Hanchet 2006a, Dunn & Hanchet 2007, SC-CAMLR-XXV – appendix F). They are thought instead to be reflecting changes in annual ice conditions, fisher learning and experience, and changes in gear and regulations (Dunn & Hanchet 2007). We therefore did not include CPUE indices in this assessment. In addition, we investigate the sensitivity of the model outputs to an alternative modelling assumption of selectivity, and the inclusion of all and subsets of the non-New Zealand tag-release and recapture data.

## 2. METHODS

### 2.1 Population dynamics

In general, the Ross Sea Antarctic toothfish stock structures used were based on the 2006 base model presented by Dunn & Hanchet (2006a). The models were sex- and age-structured, with ages from 1 to 50, with the last age group a plus group (i.e., an aggregate of all fish aged 50 and older). The annual cycle was broken into three discrete time steps, nominally summer (November–April), winter (May–October), and end-winter (age-incrementation).

The 2007 reference model was based on the model used in the assessment in 2006 (Dunn & Hanchet 2006a, SC-CAMLR-XXV 2006), but updated following the recommendations by WG-SAM (Dunn & Hanchet 2007, WG-SAM 2007). Sensitivities were typically implemented as variations to the reference model. For each model run, the model structure and assumptions are described below.

The models were run from 1995 to 2007, and were initialised assuming an equilibrium age structure at an unfished equilibrium biomass (i.e., a constant recruitment assumption) parameterised by the initial mid-season spawning stock biomass (SSB) and labelled  $B_0$ . Each model was implemented as a single-area, three-fishery model. Here, a single area was defined but the catch was removed using three concurrent fisheries (slope, shelf, and north, see Dunn et al. (2005b)) with each fishery parameterised by a sex-based selectivity ogive, typically using double normal parameterisation (i.e., domed selectivity) or logistic (flat-topped) in the case of one sensitivity.

Recruitment was assumed to occur at the beginning of the first (summer) time step. Recruitment was assumed to be 50:50 male to female, and was parameterised as a year class strength multiplier (assumed to be equal to one for the years 1995–2007), multiplied by an average (unfished) recruitment ( $R_0$ ) and a spawning stock-recruitment relationship (see later for the definition of the spawning stock biomass). The spawning stock-recruitment relationship was assumed to be a Beverton-Holt relationship with steepness of 0.75.

In all cases, selectivity coefficients for males at age were defined to have a range of 0–1, and female selectivity coefficients have range 0– $a_{max}$ . Annual selectivity shifts for the slope and north fisheries were fitted that allowed the selectivity to ‘shift’ to the left or right with changes in the mean depth of the fishery, but were estimated as free parameters for the shelf fishery.

The double normal selectivity used to model the fishing selectivity for the reference model was parameterised using four estimable parameters  $a_1$  (the mode),  $s_L$  (describes the shape of the left hand limb),  $s_R$  (describes the shape of the right hand limb) and  $a_{max}$  (value of the selectivity at the mode). It had value equal to  $a_{max}$  at  $x=a_1$ , and  $0.5a_{max}$  at  $x=a_1-s_L$  or  $x=a_1+s_R$ , i.e., the value of the selectivity at age  $x$  was,

$$\begin{aligned} f(x) &= a_{max} \times 2^{-[(x-a_1)/s_L]^2}, & (x \leq a_1) \\ &= a_{max} \times 2^{-[(x-a_1)/s_R]^2}, & (x > a_1) \end{aligned}$$

To allow for differences in maximum selectivity by sex, the value of  $a_{max}$  was fixed at one for males, but allowed to be estimated for females. The selectivity shift was parameterised by  $a_f(E_f - \bar{E}_f)$ , where  $a_f$  was a shift factor and  $E_f$  was an exogenous variable (and was the catch-weighted mean depth fished of all sets with each year for the slope and north fisheries, expressed in units of years per 1000 m).

In one sensitivity, a logistic selectivity was used for the north fishery (logistic north, see later). Here, the selectivity was parameterised by three estimable parameters  $a_{50}$ ,  $a_{t095}$ , and  $a_{max}$ . It had value equal to  $0.5 \times a_{max}$  at  $x=a_{50}$  and  $0.95 \times a_{max}$  at  $x=a_{50}+a_{t095}$ , i.e., the value of the selectivity at age  $x$  was,

$$f(x) = a_{max} / \left[ 1 + 19^{(a_{50}-x)/a_{t095}} \right]$$

The value of  $a_{max}$  was fixed at one for males, but allowed to be estimated for females. Selectivity shifts were parameterised in the same manner as for the double normal.

Natural mortality was assumed to be constant across age and sex classes, and the value of  $M$  assumed to be  $0.13 \text{ y}^{-1}$  (WG-FSA-SAM 2006).

Fishing mortality was applied only in the first (summer) time step from three concurrent fisheries. The process was to remove half of the natural mortality occurring in that time step, then apply the mortality from the fisheries instantaneously, then to remove the remaining half of the natural mortality. This differs from the more common Baranov catch equation, which implies that natural and fishing mortalities occur continuously and simultaneously. However, the difference in results from using either of these two catch equations was likely to be negligible. Hence, for each fishery  $f$ , an exploitation rate  $U_f$  was applied to the population, i.e.,

$$U_f = \frac{C_f}{\sum_{ij} \bar{w}_{ij} S_{ij} n_{ij} \exp(-0.5tM_{ij})}$$

where  $C_f$  is the catch for fishery  $f$ ,  $S_{ij}$  is the selectivity at age  $i$  and sex  $j$ ,  $\bar{w}_{ij}$  is the mean weight of fish of age  $i$  and sex  $j$ ,  $n_{ij}$  is the number of fish,  $M_{ij}$  is the natural mortality, and  $t$  is the proportion of the year's natural mortality in the time step.

The maximum possible fishing pressure associated with  $f$  was defined as the maximum proportion of fish taken from any age/sex class in the area affected by fishery  $f$  (and constrained to be less than or equal to 0.999), i.e.,

$$\hat{U}_{max}(f) = \max_{i,j} (S_{ij} U_f)$$

The population,  $n_{ij}$ , was then updated using

$$n'_{ij} = n_{ij} \exp(-tM_{ij}) (1 - S_{ij}U_f)$$

The models split the catch into three fisheries, defined as ‘shelf’, ‘slope’, and ‘north’. The definitions of the areas that comprised the fisheries were based on stratifications derived from the tree based regression analysis of the commercial catch-at-length data used to stratify the commercial catch-at-age data by Hanchet et al. (2005) using the tree based regression method of Phillips et al. (2005). Shelf was defined as the SSRUs 88.1E–F, 88.1H–L, & 88.2A–B at a depth of less than 761 m; slope was defined as the SSRUs 88.1E–F, 88.1H–L, & 88.2A–B at a depth of greater than or equal to 761 m; and north was defined as SSRUs 88.1A–88.1C, and 88.1G. The annual reported catch for Antarctic toothfish in the Ross Sea, by area, is given in Table 1.

Estimated illegal, unregulated, and unreported (IUU) catch for the Ross Sea (SC-CAMLR-XXV 2006, Annex 5) is given in Table 2. However, noting that the inclusion of IUU catch had little effect on outcomes in previous model runs (see Dunn & Hanchet 2006a, Dunn & Hanchet 2007), IUU catches were ignored in these model runs.

Length-weight parameters were given by Dunn et al. (2006), and are reproduced below in Table 3. The von Bertalanffy relationship was used to derive the mean length at age relationship (Table 3), with parameters from Dunn et al. (2006). Variability in the length at age relationship was parameterised by a normal distribution, with c.v. = 0.102 (Dunn et al. 2006).

We do not account for maturation in the sex-age structure of the population, but instead specify the time-invariant proportion of male and female fish at age that are mature. Hence, the mid-season spawning stock biomass ( $B$ ) was determined as the biomass of the proportion of fish at age  $i$  and sex  $j$  considered mature, multiplied by the number of fish of age  $i$  and sex  $j$  after half of the natural mortality has been applied, evaluated in the second (winter) time step and summed over  $i$  and  $j$ , within a defined area.

The proportions-at-age and sex mature are not known (Hanchet et al. 2003). Previous work on Antarctic toothfish has assumed 50% maturity at 100 cm (with range 85–115 cm), whilst 50% maturity at 93 cm (range 78–108 cm) is assumed for *D. eleginoides* (SC-CAMLR-XX 2001), but it is plausible that Antarctic toothfish may mature at larger sizes (Livingston & Grimes 2005). We assumed 50% maturity at 100 cm (with 95% range of  $\pm 15$  cm) with a logistic relationship by length, and converted the length-based relationship to an age-based relationship via the von Bertalanffy curve for both sexes combined, i.e., we assumed the proportions mature by age are equal for males and females with the relationship given in Figure 2 and Table 3.

In the age incrementation time step all fish age by 1 year, with the exception of fish in the 50 year plus group — these become the sum of all fish over 50 years and those aged 49.

The population model structure included tag-release and tag-recapture events. Here, the model replicated the basic age-sex structure described above for each tag-release event. The age and sex structure of the tag component was seeded by a tag release event. Tagging was applied to a ‘cohort’ of fish simultaneously (i.e., the ‘cohort’ of fish that were tagged in a given year and time step), and tagging from each year was applied as a single tagging event. The usual population processes (natural mortality, fishing mortality, etc.) were then applied over the tagged and untagged components of the model simultaneously.



**Table 1: Total Ross Sea Antarctic toothfish catch (t) for New Zealand and other vessels by fishery for the years 1997–2007.**

Year	New Zealand			Other			All vessels			Total	Catch Limit <sup>1</sup>
	Shelf	Slope	North	Shelf	Slope	North	Shelf	Slope	North		
1997	0	0	0	0	0	0	0	0	0	0	1 980
1998	8	29	4	0	0	0	8	29	4	41	1 573
1999	14	282	0	0	0	0	14	282	0	296	2 281
2000	64	689	0	0	0	0	64	689	0	752	2 340
2001	112	341	120	1	9	23	113	349	143	604	2 314
2002	10	936	412	0	0	0	10	936	412	1 358	2 758
2003	0	263	691	2	349	469	2	611	1 161	1 774	4 135
2004	46	525	224	97	1 138	147	143	1 663	371	2 177	3 625
2005	8	1179	321	385	1 083	230	393	2 263	551	3 207	3 625
2006	0	1006	333	251	1 367	11	251	2 373	343	2 967	2 964
2007	0	895	265	68	1 549	308	68	2 443	573	3 084	3 072
Total	262	6 145	2 370	804	5 495	1 188	1 066	11 638	3 558	16 260	

1. Catch limit for 88.1 and 88.2 *Dissostichus* spp. combined for the years 1997–2005, and for Subarea 88.1 and SSRUs 88.2A–B for 2006–2007.

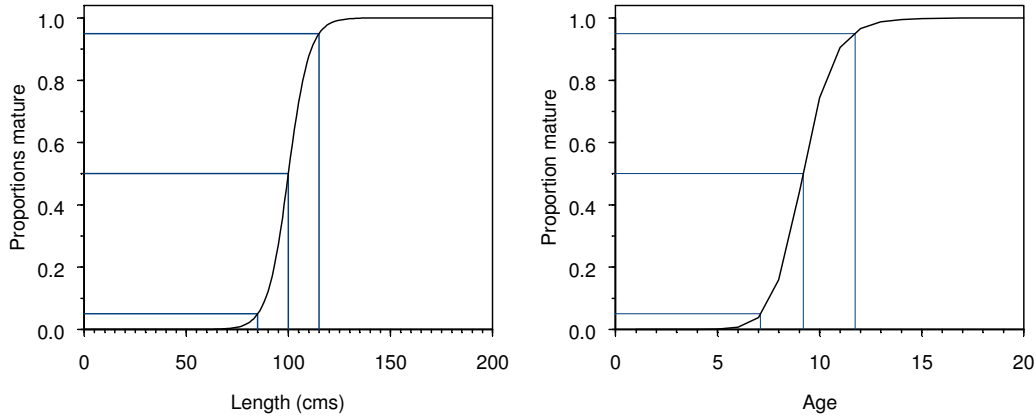
**Table 2: Total IUU catch of *Dissostichus* spp. for Subareas 88.1 & 88.2, for the years 1997–2006 (SC-CAMLR-XXV 2006, Annex 5).**

Year	Subarea 88.1	Subarea 88.2	Total
1997	0	0	0
1998	0	0	0
1999	0	0	0
2000	0	0	0
2001	0	0	0
2002	92	0	92
2003	0	0	0
2004	240	0	240
2005	28	0	28
2006	0	15 <sup>1</sup>	15
Total	360	15	375

1. Associated with SSRU 88.2A, and hence included as catch in the Shelf fishery within the Ross Sea assessment model.

**Table 3: The 2007 reference model biological parameters (natural mortality, growth, length-weight relationship, and length at maturity).**

Relationship	Parameter	Revised parameters	
		Male	Female
Natural mortality	$M$ ( $y^{-1}$ )	0.13	0.13
Von Bertalanffy	$t_0$ (y)	-0.256	0.021
	$k$ ( $y^{-1}$ )	0.093	0.090
	$L_\infty$ (cm)	169.07	180.20
	c.v.	0.102	0.102
Length-weight	$a$ ( $t \cdot \text{cm}^{-1}$ )	1.387e-008	7.153e-009
	$b$	2.965	3.108
Length at maturity (cm)	$L_{50}$ ( $\pm L_{t0.95}$ )	100 ( $\pm 15$ )	100 ( $\pm 15$ )



**Figure 2: Assumed proportions (left figure) mature by length and (right figure) mature by age.**

## 2.2 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  $\mathbf{p}$ , the vector of the free parameters,  $L$  the likelihood function, and  $O_i$  the  $i$ th observation was

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

where  $\theta$  is the joint prior (and penalty) densities of the parameters  $\mathbf{p}$ . The observations, likelihoods, penalties, and priors are described later.

Initial model fits were evaluated at the maximum of the posterior density (MPD), MPD profile distributions (i.e., by evaluating the minimum objective function while fixing one parameter and allowing all other parameters to vary), and 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 Monte Carlo Markov Chain (MCMC). The Metropolis algorithm attempts to draw a sample from a Bayesian posterior distribution, and calculates the posterior density  $\pi$ , scaled by an unknown constant. The algorithm generates a ‘chain’ or sequence of values. Typically the beginning of the chain is discarded and every  $n$ th 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 proposal distribution  $J$ , which should be symmetric 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. 2005 for more detail), and used as the starting point for the chain.

MCMCs were estimated using a burn-in length of  $5 \times 10^5$  iterations, with every 1000<sup>th</sup> sample taken from the next  $1 \times 10^6$  iterations (i.e., a systematic sample of length 1000 was taken from the Bayesian posterior). Chain diagnostic plots, autocorrelation estimates, and single chain convergence tests of Geweke (1992) and Heidelberger & Welch (1983) stationarity and half-

width were used to determine evidence of non-convergence. The tests used a significance level of 0.05 and the diagnostics were calculated using the Bayesian Output Analysis software (Smith 2003).

## 2.3 Observations

### 2.3.1 Proportions-at-age in the catch

The catch proportions-at-age data were fitted to the modelled proportions-at-age composition using a multinomial likelihood, i.e.,

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

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. Proportions-at-age data were derived from the aged otoliths collected by observers and the length frequency of the catch. Hanchet et al. (2007) described the catch-at-age data available for the assessment models, with data available for the years 1998–2007. Effective sample sizes assumed for the proportions-at-age data are described below.

Annual selectivity shifts were estimated in the models. For the slope and north fisheries, the annual selectivity was based on a multiplier of the mean depth fished (weighted by the catch) from all sets within each fishery in each year (Table 4). For the shelf fishery, independent annual selectivity shifts were estimated for the shelf fishery selectivities.

Ageing error was accounted for by modifying the likelihoods for the proportions-at-age data such that  $E_i$  was replaced by  $E'_i$ , where  $E'_i$  were the expected proportions-at-age multiplied by an ageing error misclassification matrix  $A$ . The error misclassification matrix was derived from a normal distribution with constant c.v. = 0.1 (P. Horn, NIWA, pers. comm.).

**Table 4: Weighted mean depth (m), and the average depth fished by fishery (Shelf, Slope, and North), for the years 1998–2007.**

Year	Shelf	Slope	North	All fisheries
1998	697	985	–	908
1999	671	916	–	904
2000	716	974	–	952
2001	654	1 242	1 193	1 121
2002	738	1 118	1 403	1 202
2003	691	1 371	1 477	1 440
2004	662	1 232	1 353	1 215
2005	637	1 200	1 462	1 176
2006	654	1 208	1 379	1 181
2007	658	1 190	1 483	1 233
Average	654	1 187	1 424	1 206

### 2.3.2 Tag-release data

Tag-release data are not technically observations within the models, but rather non-estimable parameters. Numbers of available individual fish tagged and released by New Zealand vessels were initially described by Dunn et al. (2005c), and updated for 2007 by Dunn et al. (2007a). Tag-release events were defined for 2001–2006 and within-season recaptures were ignored

(Table 5). Tag release events were assumed to have occurred at the end of the first (summer) time step, following the (summer) natural and fishing mortality.

Each season's tag-release data were included as a separate member of the model structure, i.e., the model kept account of the numbers of fish tagged in each year separately. Initial tag mortality was assumed to be 10% (Agnew et al. 2005) plus initial tag loss (Dunn & Hanchet 2006b), effectively reducing the number of tagged fish in the population at the time they were tagged.

Instantaneous tag-loss was assumed to occur in equal proportions in the first two time steps, but not in the third. Here, the number of fish in each tagged cohort at time  $j$  (i.e., the number  $n_{ij}$  in any age/sex element  $i$  of the population at time step  $j$ ) was determined by applying the tag loss rate for that cohort,  $l_i$ , by the proportion of tag loss to apply in that time step  $t_j$ , i.e.,

$$n'_{ij} = n_{ij} \exp(-t_j l_i)$$

Tagging appears to result in a check on growth of individual fish (Dunn et al. 2005d). We assume that the effect of tagging on growth is equivalent to a short period of 'no growth' when fish are tagged by adjusting the  $t_0$  von Bertalanffy growth parameter by -0.5 years (Dunn & Hanchet 2007). Here, the effective size at age for a tagged fish is the same as the size at age for an untagged fish at that age less 0.5 years, i.e., the mean size at age for a tagged fish was modelled as,

$$\bar{s}(age) = L_{\text{inf}} \left( 1 - \exp(-k(age - t_0 - g_i)) \right),$$

where  $g_i=0.5$ .

In all cases, the numbers of tagged fish at age were calculated deterministically from the observations of numbers of fish at length. A selectivity was applied to the conversion of fish-at-length to fish-at-age within each fishery, i.e., for each of the shelf, slope, and north fisheries separate tag release events were specified for each year along with the appropriate fishing selectivity. Hence, the numbers of fish tagged at age were determined from the selected length frequency of the untagged fish, the length frequency of tagged fish, and the population state and parameters in the given year and time step of the release event.

### 2.3.3 Tag-recapture observations

Numbers of tagged fish recaptured were given by Dunn et al. (2007a), and are summarised in Table 5. Following Dunn et al. (2005b), we ignored within-season recoveries, and only used data from the between-season recaptures. The estimated number of scanned fish (i.e., those fish that were caught and inspected for a possible tag) was derived from the sum of the scaled length frequencies from the vessel observer records plus the numbers of fish tagged and released. Tag recapture events were assumed to occur at the end of the first (summer) time step, and were assumed to have a detection probability of 100%.

For each year, the recovered tags at length for each release event  $t$  were fitted, in 10 cm length classes (range 40–230 cm), using a binomial likelihood, i.e.,

$$-\log(L) = \sum_i \left[ \log(n_i!) - \log((n_i - m_i)!) - \log(m_i!) + m_i \log\left(\frac{M_i}{N_i}\right) + (n_i - m_i) \log\left(1 - \frac{M_i}{N_i}\right) \right]$$

where  $n_i$  is the number of fish in length class  $i$  that were scanned,  $m_i$  is the number of tagged fish in length class  $i$  that were recaptured from the release event  $t$ ,  $N_i$  is the expected number of fish in length class  $i$  in the population, and  $M_i$  is the expected number of fish in length class  $i$  in the tagged population from  $t$ .

**Table 5: Numbers of Antarctic toothfish with tags released for the years 2001–2006 for (a) New Zealand vessels, (b) all vessels, and the number recaptured in 2002–2007.**

Data source	Tagged fish released		Tagged fish recaptured						
	Year	Number	2002	2003	2004	2005	2006	2007	Total
New Zealand vessels	2001	259	1	1	0	0	0	1	3
	2002	684	–	5	3	5	5	4	22
	2003	858	–	–	7	7	0	5	19
	2004	865	–	–	–	16	11	8	35
	2005	1 518	–	–	–	–	12	9	21
	2006	1 495	–	–	–	–	–	49	49
	Total	5 679	1	6	10	28	28	76	149
All vessels	2001	259	1	1	0	0	0	1	3
	2002	684	–	5	3	5	7	13	33
	2003	858	–	–	13	9	2	9	33
	2004	2 033	–	–	–	23	19	32	74
	2005	3 275	–	–	–	–	26	29	55
	2006	3 040	–	–	–	–	–	89	89
	Total	10 149	1	6	16	37	54	173	287

### 2.3.4 Process error and data weighting

Additional variance, assumed to arise from differences between model simplifications and real world variation, was added to the sampling variance for all observations. Adding such additional errors to each observation type has two main effects, (i) it alters the relative weighting of each of the data sets (observations) used in the model, and (ii) it typically increases the overall uncertainty of the model, leading to wider credible bounds on the estimated and derived parameters.

The additional variance, termed process error, was estimated for each model MPD run, and for each model, the total error assumed for each observation was calculated by adding process error and observation error. A single process error was estimated for each of the observation types (i.e., one for the age data and one for the tag data), with the procedure for calculating the additional process error as described below.

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 the relationship between the observed proportions,  $E_i$ , and estimated c.v.s,  $c_i$ , 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,  $N'_j$ , by adding additional process error,  $N_{PE}$ , to the sample size calculated in (a) above, where,

$$N'_j = 1 / \left( \frac{1}{N_j} + \frac{1}{N_{PE}} \right)$$

i.e., from an initial MPD model fit, an estimate of the additional process error was made by solving the following equation for  $N_{PE}$ ,

$$n = \sum_{ij} \frac{(O_{ij} - E_{ij})^2}{E_{ij}(1 - E_{ij}) \left( \frac{1}{N_j} + \frac{1}{N_{PE}} \right)}$$

where  $n$  was the number of multinomial cells,  $O_{ij}$  was the observed proportions for age class  $i$  in year  $j$ ,  $E_{ij}$  was the expected proportions,  $N_j$  was the effective sample size estimated in (a) above, and  $N_{PE}$  was the associated process error for that class of observations.

Estimates of the over-dispersion for the tag-recapture likelihoods were made using a similar method to that for the proportions-at-age data. First, initial sample sizes for the numbers recaptured and the numbers scanned were assumed from the actual numbers recaptured and numbers scanned. Second, the over-dispersion  $\phi_j$  for each tagging event was calculated from its  $i$  recapture events, from an initial MPD run, where,

$$\phi_j = \text{var} \left( \frac{O_{ij} - E_{ij}}{\sqrt{E_{ij}(1 - p_{ij})}} \right)$$

where  $O_{ij}$  was the observed number of recaptures,  $E_{ij}$  the expected number of recaptures, and  $p_{ij}$  the expected probability of recapture. Over-dispersion terms for each of the recapture events were then combined (i.e., by taking the geometric mean), and the log-likelihood was then modified by multiplying by  $1/\phi$ .

The process error estimates for the 2007 reference model were  $N_{PE} = 719$  and  $\phi = 1.695$ , and had similar values to the process errors estimated in previous years. Process error estimates for the sensitivity models had broadly similar values. Process error estimates for the sensitivity models are given in Section 2.7.

## 2.4 Penalties

Two types of penalties were included within the model. First, the penalty on the catch constrained the model from returning parameter estimates where the population biomass was such that the catch from an individual year would exceed the maximum exploitation rate (see earlier). Second, a tagging penalty discouraged population estimates that were too low to allow the correct number of fish to be tagged. However, in the model runs presented here, these penalties had no significant contribution to the total likelihood.

## 2.5 Priors

Priors were defined for all free parameters in the models. The free parameters, starting values for the minimisation, and bounds are given in Table 6. In models presented here, priors were chosen that were relatively non-informative but also encouraged lower estimates of  $B_0$ . Note that the priors for the parameters were, in general, set to be the same as for the 2006 base model (Table 6).

**Table 6: Number ( $N$ ), start values, priors, and bounds for the free parameters (when estimated) for the model runs.**

Parameter	$N$	Start value	Prior	Bounds		
				Lower	Upper	
$B_0$	1	150 000	Uniform-log	$1 \times 10^4$	$1 \times 10^6$	
Male fishing selectivities	$a_I$	8.0	Uniform	1.0	50.0	
		$s_L$	4.0	Uniform	1.0	50.0
		$s_R$	10.0	Uniform	1.0	500.0
Female fishing selectivities	$a_{max}$	1.0	Uniform	0.01	10.0	
		$a_I$	8.0	Uniform	1.0	50.0
		$s_L$	4.0	Uniform	1.0	50.0
		$s_R$	10.0	Uniform	1.0	500.0
Selectivity shift ( $\text{ykm}^{-1}$ )	$a_f$	3	0.0	Uniform	0.0	50.0
Annual selectivity shift <sup>1</sup>	$E_f$	10	Mean depth	Uniform	-10.0	10.0

1. The bounds for the annual selectivity shifts for the shelf fishery were revised from the 2006 base model [-50,50] to resolve a minimisation problem. Note that these bounds were still outside the range estimated in resulting MCMC calculations, and hence had no impact on the resulting estimates from the models.

## 2.6 Yield calculations

Yields were estimated by projecting the estimated current status for each model under a constant catch assumption using the rules,

- Rule*<sub>1</sub>: Choose a yield  $\gamma_1$  so that the probability of the spawning biomass dropping below 20% of its median pre-exploitation level over a 35-year harvesting period is 10% (depletion probability) .
- Rule*<sub>2</sub>: Choose a yield  $\gamma_2$  so that the median escapement at the end of a 35-year period is 50% of the median pre-exploitation level.
- Rule*<sub>3</sub>: Select the lower of  $\gamma_1$  and  $\gamma_2$  as the yield.

The probability of depletion and the level of escapement were calculated by projecting forward for a period of 35 years (i.e., for the period 2008–2042) under a scenario of a constant annual catch for each sample from the posterior distribution. The depletion probability was calculated as the proportion of samples from the Bayesian posterior where the predicted future SSB was below 20% of  $B_0$  in that respective sample in at least one year for each year over a 35-year projected period. The level of escapement was calculated as the proportion of samples from the Bayesian posterior where the predicted future SSB was below 50% of  $B_0$  in that respective sample at the end of a 35-year projected period. The posterior sample estimates of  $B_0$  were used as a proxy for the pre-exploitation median SSB in each sample. Note that the use of the  $B_0$  proxy will result in a small downward bias of the stock status in each trial and hence a small downward bias in the estimate of yield.

For the yield calculations, recruitment for 2000–2041 was assumed to be lognormally distributed with a standard deviation of 0.6 (Dunn et al. 2006); future catch was assumed to follow the same split between fisheries as that in the four most recent seasons (i.e., based on the distribution of the 2004–2007 catch, 7.4%, 76.5%, and 16.1% of the total future catch was allocated to the shelf, slope, and north fisheries respectively); and that the selectivity shifts were assumed to be the mean of those estimated for the years 1998–2007.

The decision rules were evaluated under two scenarios, (i) assuming that the future constant catch was equal to the 2007 catch limit (CM41-09, CAMLR-XXV 2006), and (ii) the maximum future constant catch that met the decision rule criteria. Note that, in previous years in the Ross Sea fishery, the total catch limit has not always been taken. Ice cover over fishable

depths in the southern SSRUs has meant that fishing vessels were unable to access some of the available catch. We ignore possible ice cover restrictions on future catch, and assume that for each future season, the available catch would be taken, subject to the maximum exploitation rate rule ( $U_{\max}=0.999$ ).

## 2.7 Sensitivity runs

Model runs were conducted for the 2007 reference model and the sensitivity runs described in Table 7. Sensitivity runs were determined as modifications to the 2007 reference model, and were chosen to investigate the effect of alternative assumptions of data weights within the model and the use of either the tag release and recapture data from New Zealand vessels, or the tag-release and recapture data from all vessels.

The purpose of the logistic north model was to investigate the impact on model estimates, if the assumption of a domed selectivity for the north fishery were incorrect. Here, the selectivity for the north fishery was forced to be logistic, i.e., where older fish were assumed to be fully selected.

The all vessels sensitivity used all available tagging data from New Zealand and non-New Zealand vessels combined, for the years since 2004. For the years before 2004, only New Zealand data were used (see Dunn et al. 2007a).

The all vessels 2006 model employed all of the New Zealand tag-release data for 2001–2005 and the tag-release data for all vessels in 2006. Hence, tag recaptures and numbers of scanned fish were for the New Zealand vessels for 2002–2006 and for all vessels in 2007. Effectively, this was the same as the 2007 reference case except that 2006 tag-releases were for all vessels and the recaptured/scanned numbers were the appropriate subset of all vessels in 2007. Here, the sensitivity investigated the impact of adding the 2007 data from the non-New Zealand vessels.

Process error estimates for each of the model runs are given in Table 8.

**Table 7: Labels and description of the sensitivity runs.**

Model run	Description
1 2007 reference	The reference model
2 Logistic north	The reference model, with a logistic selectivity applied to the north fishery
3 All vessels	Same as the reference model, but using tag data from all vessels
4 All vessels 2006	Same as the reference model, but using tag data from all vessels for 2006 only

**Table 8: Process error estimates for catch-at-age ( $N_{PE}$ ) and tag-recapture ( $\phi$ ) observations for the 2007 reference model and the sensitivity models.**

Model	$N_{PE}$	$\phi$
2006 base model <sup>1</sup>	500	2.411
2007 reference	719	1.695
Logistic north	977	1.607
All vessels	772	1.475
All vessel 2006	731	1.667

1. The 2006 base case also included CPUE observations, which had process error  $c_{PE}=0.40$ .



### 3. RESULTS

#### 3.1 MPD results

Objective function values for the MPD estimates for the model runs are given in Table 9 and estimates of initial ( $B_0$ ) and current biomass given in Table 10. Comparison of the objective function values suggested that there were only minor differences in fit between each of the data sets between each of the model runs, except for the likelihoods to the commercial catch proportions-at-age observations.

Updated estimates of the MPD value of the initial biomass gave an estimate of the initial biomass of 70 700 t, similar, albeit lower, than the 2006 base model (c.f., 77 200 t). The negative log-likelihood values for the model did not suggest any evidence of a poor fit to the observations (Table 9).

The logistic north model suggested a slightly higher initial biomass (77 600 t) than for the 2007 reference model, and had very similar fits to the tag data and the catch-at-age observations for the slope and shelf fisheries. However, fits to the north fishery showed some evidence of a slightly less adequate fit when compared with the reference case. Initial biomass estimated for the all vessels case was 109 000 t. This was similar to the values from previous assessments where the estimate of biomass was larger than that obtained from the New Zealand vessels alone. Initial biomass estimates for the all vessels 2006 model were more consistent with the 2007 reference model, with  $B_0 = 81\,200$  t. In general, fits for the observations for the all vessels and all vessels 2006 models were similar to those for the 2007 reference model.

**Table 9: MPD objective function values and number of estimated parameters for all model runs.**

Objective function component	Model run				
	2006 base	2007 reference	Logistic north	All vessels	All vessels 2006
2001 tags recaptured	3.5	6.8	7.0	9.2	7.0
2002 tags recaptured	14.6	27.3	29.2	37.1	31.8
2003 tags recaptured	11.8	28.4	29.4	49.0	32.4
2004 tags recaptured	13.2	28.4	30.6	49.9	33.8
2005 tags recaptured	4.4	15.8	16.0	36.7	24.3
2006 tags recaptured	–	22.8	25.9	33.4	23.7
CPUE (North)	-1.2	–	–	–	–
Catch-at-age (North)	607.9	767.0	896.8	776.0	769.2
CPUE (Shelf)	-1.3	–	–	–	–
Catch-at-age (Shelf)	740.2	915.8	962.6	923.9	917.2
CPUE (Slope)	-4.4	–	–	–	–
Catch-at-age (Slope)	906.9	1 132.3	1 200.0	1 148.9	1136.2
Sub-total (observations)	2 295.7	2 944.5	3 197.4	3 064.0	2 975.5
Penalties	0.0	0.0	0.0	0.0	0.0
$B_0$ prior	11.3	11.2	11.3	11.6	11.3
All other priors	0.0	0.0	0.0	0.0	0.0
Total objective function	2 306.9	2 955.7	3 208.6	3 075.6	2986.8
Number of free parameters	33	34	32	34	34

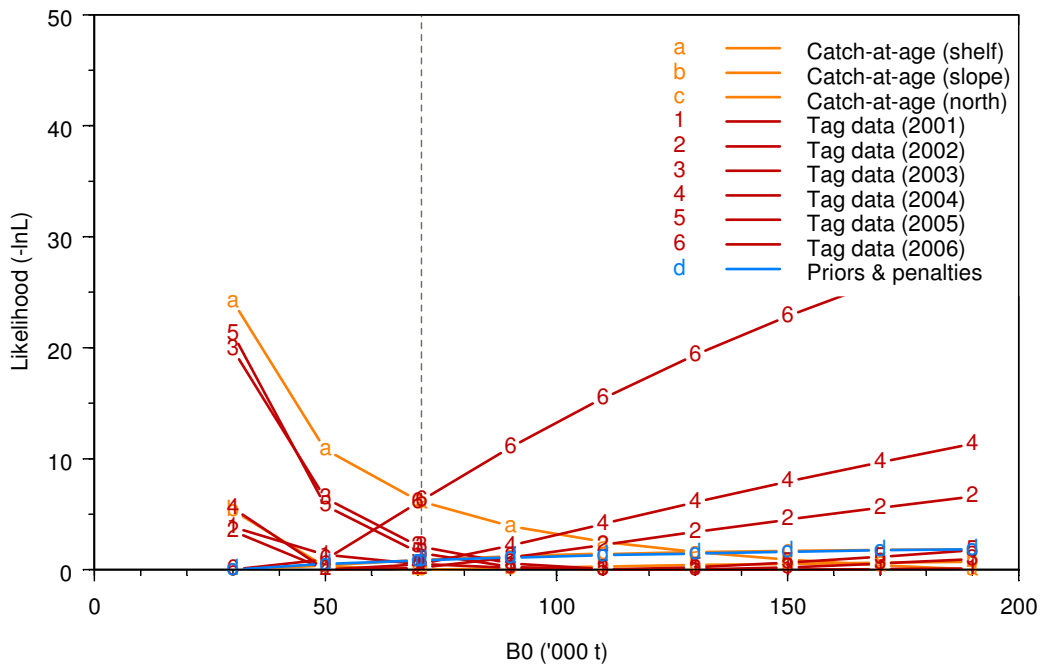
**Table 10: Selected MPD parameter ( $B_0$ ,  $B_{2006}$ , and  $B_{2007}$ ) values for all model runs.**

Model	$B_0$	$B_{2006}$	$B_{2006} (\%B_0)$	$B_{2007}$	$B_{2007} (\%B_0)$
2006 base	77 200	66 500	86.1	–	–
2007 reference	70 700	60 000	84.8	57 900	81.8
Logistic north	77 600	66 900	86.2	64 700	83.4
All vessels	109 400	98 600	90.2	96 500	88.2
All vessels 2006	81 200	70 500	86.8	68 400	84.1

### 3.2 Likelihood profiles

Likelihood profiles for the 2007 reference and the models that included data from all vessels (all vessels and all vessels 2006) are given below (Figures 3–5). For all models, likelihood profiles were carried out by fixing  $B_0$  at values across a range of plausible values (i.e., 30 000–190 000 t), with the remaining parameters (e.g., selectivities) estimated.

The likelihood profiles for the 2007 reference model are shown in Figure 3. Here, the catch-at-age data and tag recaptures from 2003 and 2005 suggested that very low biomass levels were less likely, whilst tag recaptures from 2004 and 2002 suggest very high biomass estimates were less likely. The addition of the 2006 release data were the most dominant of the tag data series. Here, the likelihood profiles suggested that high biomass values were more unlikely than for previous data, and suggested that lower initial biomass values were more likely. As in previous assessments, the profiles were influenced by the shelf catch-at-age proportions, suggesting that low estimates of initial biomass were less likely.



**Figure 3: Likelihood profiles for the 2007 reference model for values of  $B_0$ . Negative log likelihood values rescaled to have minimum 0 for each data set. The dashed vertical line indicates the MPD.**

Similarly, the likelihood profiles for the all vessels model (Figure 4) and the all vessels 2006 (Figure 5) models had similar patterns for the tag data and for the catch-at-age proportions. However, the influence of the tag recapture data was much stronger — a consequence of the increased sample sizes for the tag data. In both cases, the 2006 releases were the dominant data sets, suggesting that very high initial biomass levels were less likely. The all vessel 2006

model also suggested a slightly greater influence of the 2002 and 2004 tag-release data when compared with the 2007 reference case

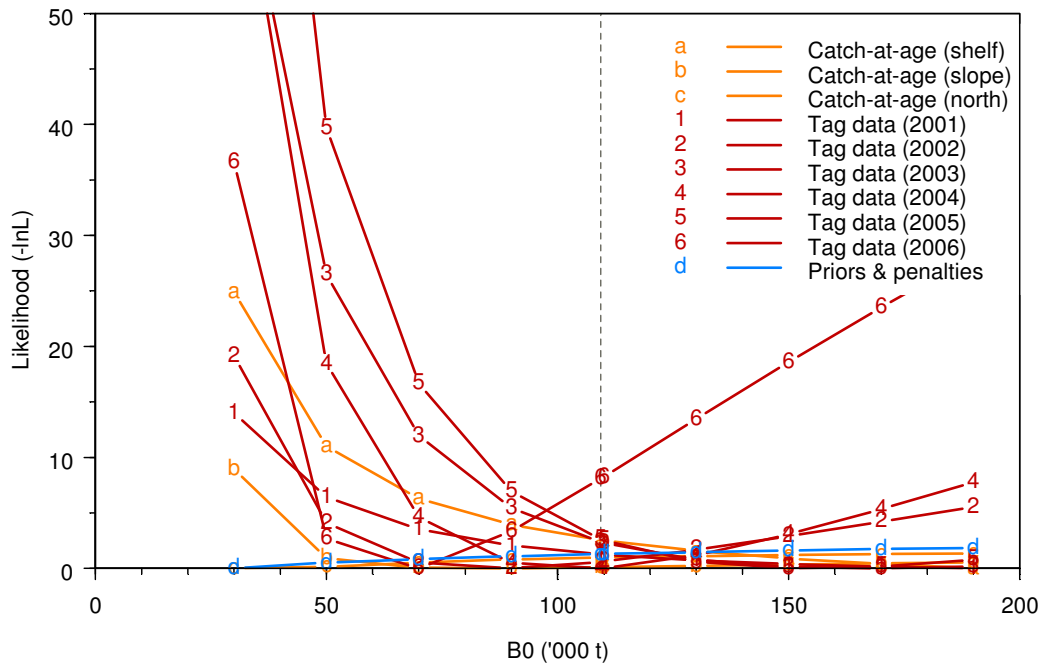


Figure 4: Likelihood profiles for the all vessels model for values of  $B_0$ . Negative log likelihood values rescaled to have minimum 0 for each data set. The dashed vertical line indicates the MPD.

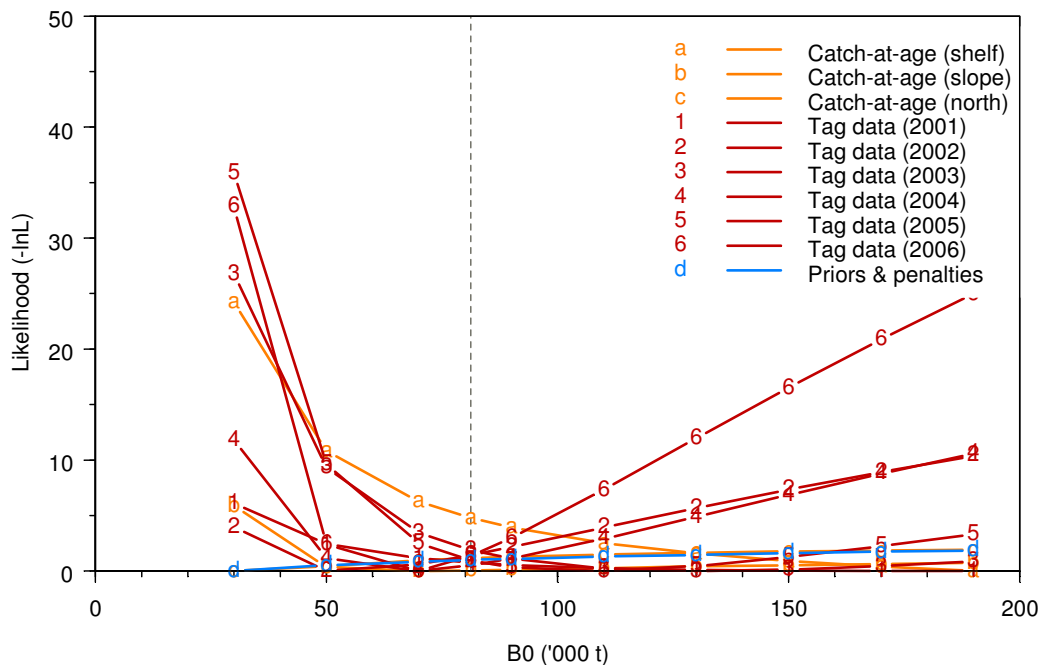
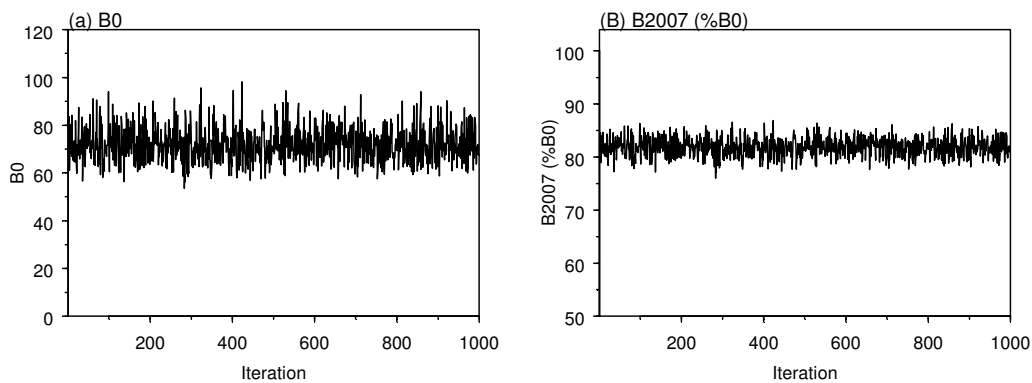


Figure 5: Likelihood profiles for the all vessels 2006 model for values of  $B_0$ . Negative log likelihood values rescaled to have minimum 0 for each data set. The dashed vertical line indicates the MPD.

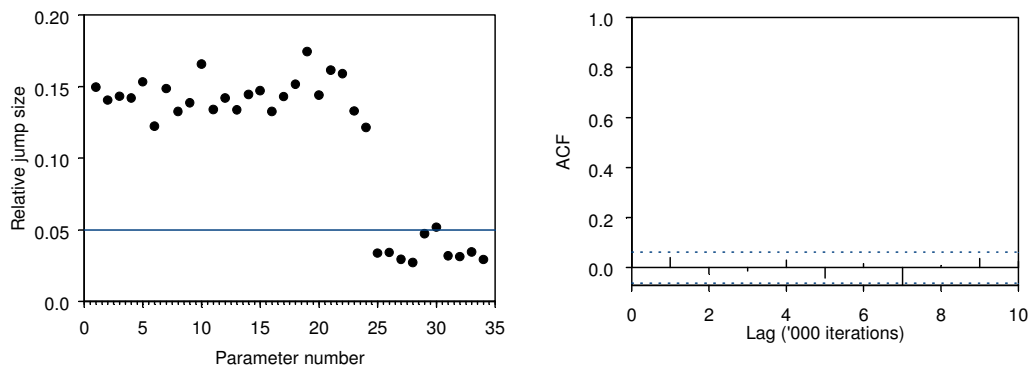
### 3.3 MCMC results

#### 3.3.1 MCMC diagnostics

Trace plot diagnostics of key parameters ( $B_0$  and  $B_{2007}$ ) for the 2007 reference model are given in Figure 6. No evidence of non-convergence from the trace statistics was found in the base or derived biomass parameters. Diagnostic plots suggested no evidence in non-convergence in most parameters ( $B_0$  and the selectivity parameters 1–24, see Figure 7), but there was some evidence in the median MCMC jump statistics for the annual shift parameters for the shelf catch-at-age proportions (parameters 25–34, see Figure 7). Convergence tests of Geweke (1992) and the Heidelberger & Welch (1983) stationarity and half-width tests passed all but some (depending on the test) of the annual shift parameter for the shelf fishery. However, sensitivity analyses of non-convergence in the annual shift parameters suggested that the effect of any non-convergence in these parameters on model conclusions were likely to be minimal (i.e., the choice of the annual selectivity parameters had little effect on resulting estimates of key biomass parameters within individual MCMC samples). Autocorrelation lag plots (Figure 7) suggested reasonable mixing in the MCMC chain with no evidence of autocorrelation. Diagnostics results for the logistic, all vessels, and all vessels 2006 models were very similar.



**Figure 6: MCMC posterior trace plots for (a)  $B_0$  and (b) current biomass ( $\%B_{2007}/B_0$ ) for the 2007 reference model.**



**Figure 7: MCMC posterior diagnostic plots for the 2007 reference model, showing (left) median relative jump size for all parameters (x-axis, labelled 1–35), and (right) ACF lag plot for  $B_0$ .**

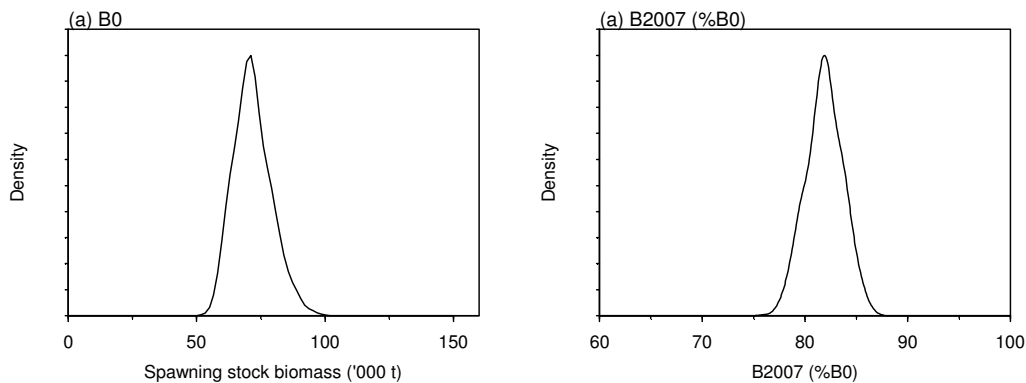
### 3.3.2 MCMC estimates

Key output parameters for all model runs are summarised in Table 11 and posterior estimates of the initial and current biomass for the 2007 reference model are given in Figure 8. MCMC estimates of initial (equilibrium) spawning stock abundance ( $B_0$ ) for the 2007 reference model were 71 200 t (95% credible intervals 59 570–87 900), and current ( $B_{2007}$ ) biomass was estimated as 81.9%  $B_0$  (95% C.I.s 78.4–85.4). While all of the models considered here suggested that the decline in biomass due to fishing has been about 15–20%, there was considerable difference in estimated initial biomass between the 2007 reference and the all vessels models (Table 11). MCMC estimates of initial (equilibrium) spawning stock abundance ( $B_0$ ) for the all vessels and all vessels 2006 models were 110 130 t (96 860–127 540 t) and 81 100 t (70 270–96 150 t) respectively.

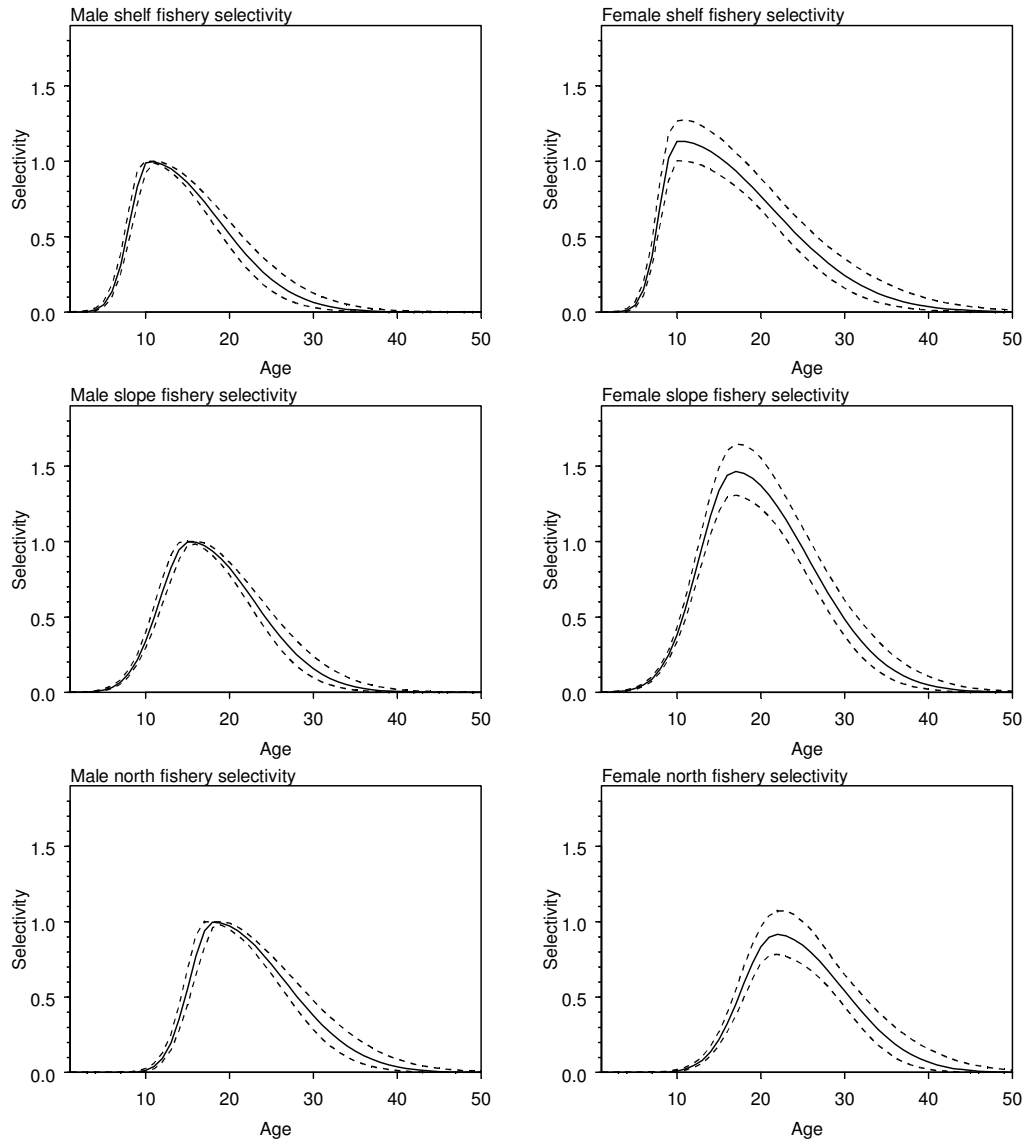
For all models, the diagnostic plots of the observed proportions-at-age of the catch versus expected values did not indicate any strong evidence of inadequate model fit for either the 2007 reference model or the sensitivity models. The 2007 reference model estimated depth shift values of  $11.7 \text{ ykm}^{-1}$  ( $10.3\text{--}13.0 \text{ ykm}^{-1}$ ) for the slope, and  $3.7 \text{ ykm}^{-1}$  ( $2.3\text{--}5.2 \text{ ykm}^{-1}$ ) for the north fishery. Estimated selectivity curves for the 2007 reference model appeared reasonable, with strong evidence of domed shaped selectivity in all of the three fisheries (Figure 9). Estimated posterior densities of the observed and expected number of tags at length, by release event and recapture year, are given in Figure 10. The tag-recapture data were equally as well fitted in the sensitivity models when compared with the 2007 reference model.

**Table 11: Median MCMC estimates (and 95% credible intervals) of  $B_0$ ,  $B_{2007}$ , and  $B_{2007}$  as % $B_0$  for the models.**

Model	$B_0$	$B_{2007}$	$B_{2007}$ (% $B_0$ )
1 2006 base	80 510 (59 920–119 920)	69 790 (49 210–101 190)	86.7 (82.1–90.4)
2 2007 reference	71 200 (59 570–87 900)	58 320 (46 700–75 010)	81.9 (78.4–85.4)
3 Logistic north	78 480 (65 350–96 340)	65 640 (52 510–83 490)	83.6 (80.4–86.7)
4 All vessels	110 130 (96 860–127 540)	97 250 (83 990–114 690)	88.3 (86.7–89.9)
5 All vessels 2006	81 100 (70 270–96 150)	68 920 (57 400–83 250)	84.3 (81.7–86.6)



**Figure 8: Posterior density estimates for (a)  $B_0$  and (b)  $B_{2007}$  as a percent of  $B_0$  for the 2007 reference model.**



**Figure 9: Estimated male and female selectivity ogives for the shelf, slope, and north fisheries for the 2007 reference model (solid lines indicate the median and dashed lines indicate the marginal 95% credible intervals).**

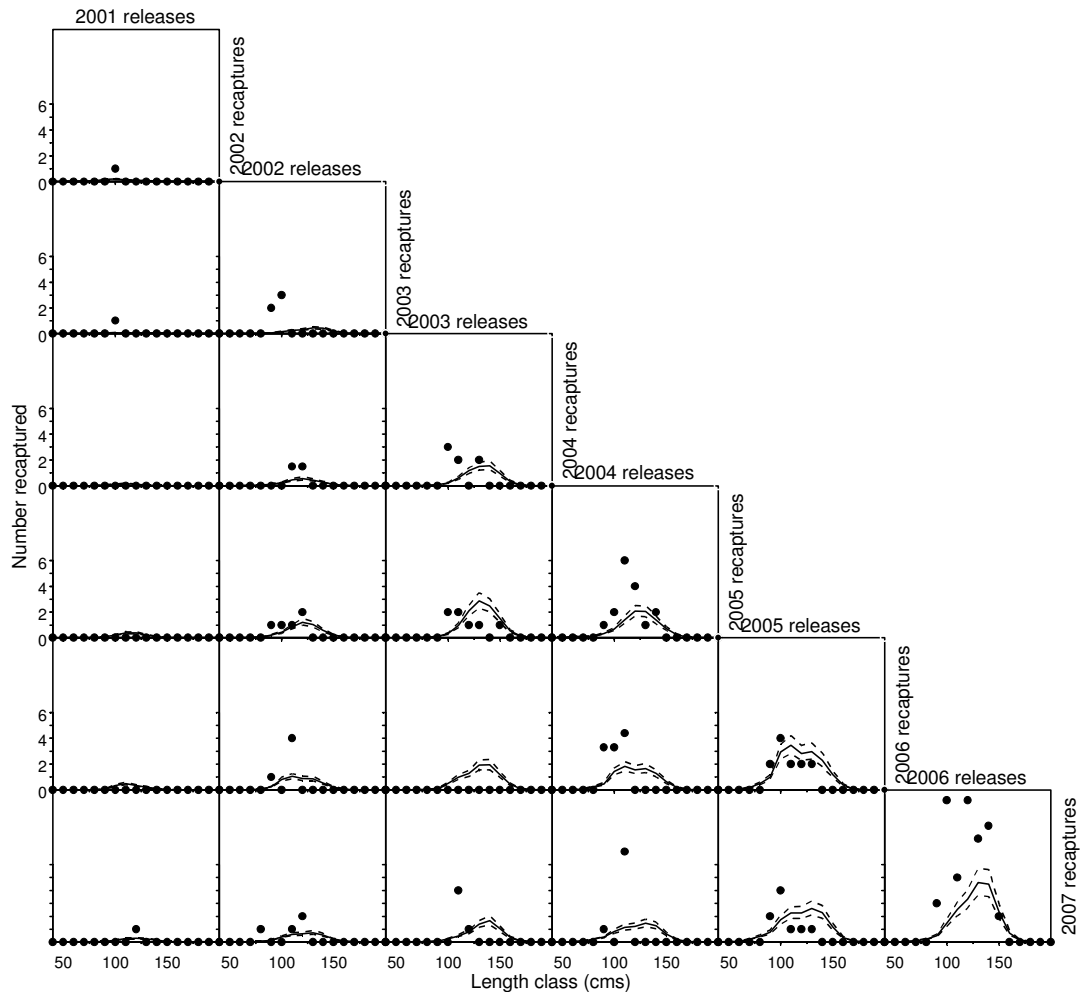


Figure 10: The 2007 reference model observed (points) and posterior estimates (lines, MCMC median and 95% credible intervals) of the number of tags recaptured (y-axis) by length class (x-axis), year of release (columns), and year of recapture (rows).

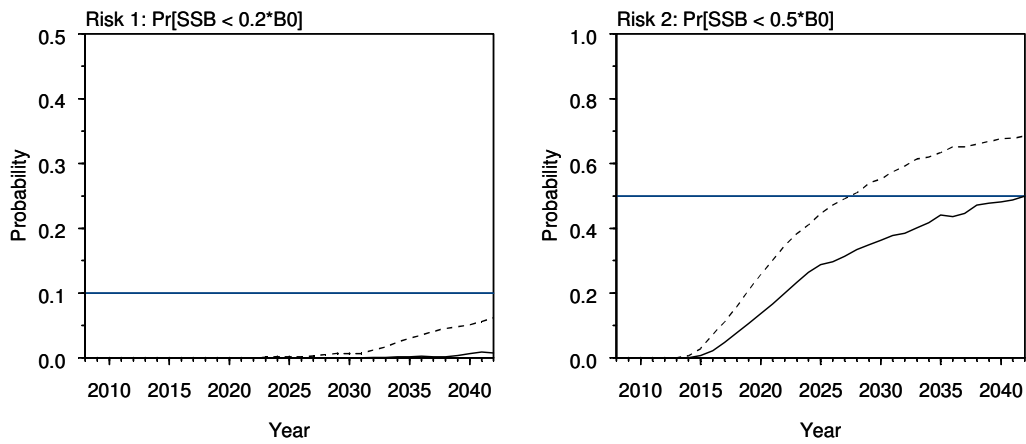
### 3.4 Yield estimates

Yield estimates were based on an assumption of a constant future catches, where the future catch was (i) the current catch limit (3072 t) and (ii) the catch that was maximised subject to the CCAMLR decision rules. The amount of catch from each area (shelf, slope, and north) was assumed to be equal to the mean annual proportion of fish taken from each of these areas for the years 2004–2007, i.e., 7%, 77%, and 16% respectively from the shelf, slope, and north areas.

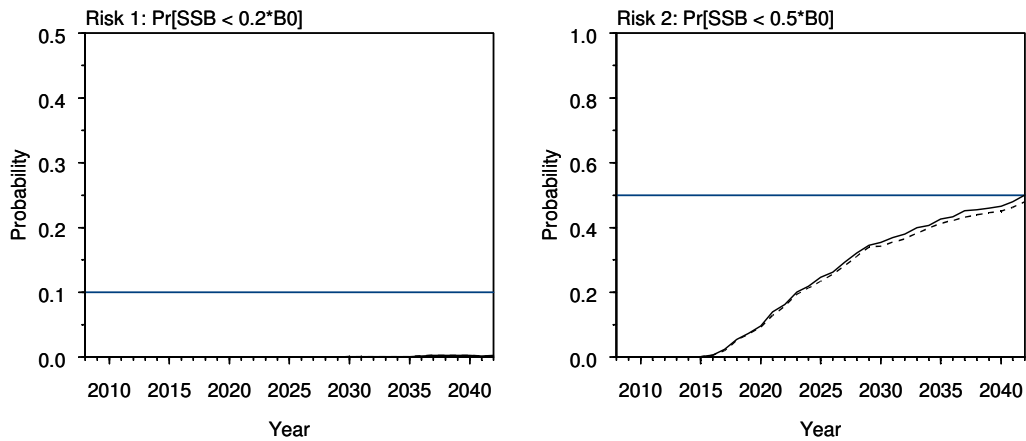
Yield estimates were made for all model runs. Yield estimates for the 2006 base model were derived from MCMC chains reported by Dunn & Hanchet (2006a). For the 2007 reference model, the decision rule risks of a constant catch at the 2007 catch limit (i.e., 3072 t) were calculated as  $\max(Pr[SSB < 0.2 \times B_0]) = 0.06$  and  $Pr[SSB_{+35} < 0.5 \times B_0] = 0.68$  (Table 12). Optimum yields (i.e., the maximum catch satisfying both  $rule_1$  and  $rule_2$ ) for the reference model were a constant future catch of 2700 t (Figure 11). Optimum yields for the logistic and all vessels 2006 models (Figure 12) were similar to the 2006 base model (2988 t and 3099 t respectively). Optimum yields for the all vessels model (Figure 13) were 4200 t, reflecting the higher estimated initial biomass from that model.

**Table 12: CCAMLR decision rule risks ( $rule_1$  and  $rule_2$ ) under the 2007 catch limit and the estimated CCAMLR yield (t).**

Model	Catch limit = 3072 t		Yield (t)	CCAMLR yield	
	$rule_1$	$rule_2$		$rule_1$	$rule_2$
2006 base	0.023	0.500	3 072	0.023	0.500
2007 reference	0.062	0.684	2 700	0.009	0.500
Logistic north	0.010	0.547	2 988	0.004	0.500
All vessels	0.000	0.110	4 200	0.002	0.500
All vessels 2006	0.002	0.479	3 099	0.003	0.499

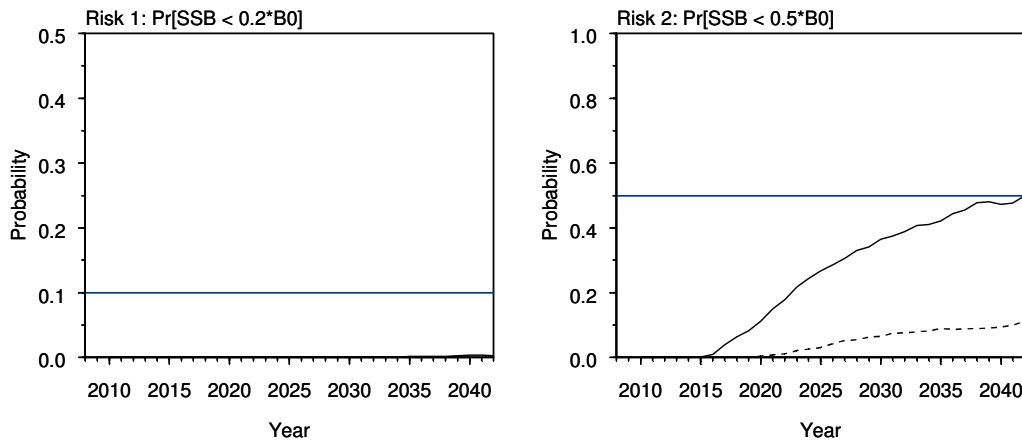


**Figure 11: Estimated risks for the 2007 reference model under the CCAMLR decision rules for (a) probability that  $SSB < 0.2B_0$  and (b) probability that  $SSB < 0.5B_0$  with the (dashed lines) current catch limit (3072 t) and (solid lines) maximum catch that meets the decision rule criteria (2700 t).**



**Figure 12: Estimated risks for the all vessels 2006 model under the CCAMLR decision rules for (a) probability that  $SSB < 0.2B_0$  and (b) probability that  $SSB < 0.5B_0$  with the (dashed lines) current catch limit (3072 t) and (solid lines) maximum catch that meets the decision rule criteria (3099 t).**





**Figure 13:** Estimated risks for the all vessels model under the CCAMLR decision rules for (a) probability that  $SSB < 0.2B_0$  and (b) probability that  $SSB < 0.5B_0$  with the (dashed lines) current catch limit (3072 t) and (solid lines) maximum catch that meets the decision rule criteria (4200 t).

#### 4. DISCUSSION

This report presents preliminary assessment models for Antarctic toothfish (*Dissostichus mawsoni*) in the Ross Sea (Subareas 88.1 and SSRUs 88.2A–B), including data up to the end of the 2006–07 season. The reference model reported here was an update of the 2006 base model assessment (Dunn & Hanchet 2006a) incorporating 2007 catch, tag, and age data together with the revised assumptions agreed at WG-SAM (2007), and ignoring the CPUE indices. This updated assessment resulted in a slightly lower estimate of initial and current biomass than the 2006 base case.

As noted by Dunn & Hanchet (2007), catches reported for 2007 and the inclusion of the IUU catch history had little effect on model estimates of key parameters. The later result was also in line with the results of a simulation experiment by Dunn (2006b), suggesting that the inclusion of IUU catch in such models would result in similar, but less conservative, model outputs.

Inclusion of ‘all vessels’ tagging data for all years resulted in a more optimistic assessment. This was essentially because the number of scanned fish was substantially higher than with the reference model but the number of recaptures increased at a lower rate, particularly for recapture data before 2007. Thus, the overall recapture rate was lower, and biomass was estimated to be higher. The reason for the relatively low number of recaptures in earlier years may be related to the distribution of fishing effort by different vessels (i.e., vessels may be more likely to fish in areas where they had released their own tags), to lower survival of tagged fish, or to lower detection rates on non-New Zealand vessels. If the reference model was modified to have only 2006 releases from all vessels and their recaptures in 2007 (all vessels 2006 model), the resulting estimates were more similar to the reference case .

In general, the fits from all models to the data were adequate, with the tag-release and recapture data providing the most information on stock size, although there was an indication that the 2007 recaptures of the 2006 releases were not as well fitted as the remainder of the tagging data. The most significant impact on the assessment model results was the inclusion of the 2007 tag-recapture data, and, in particular, the recaptures of 2006 releases in 2007. The tags recaptured by New Zealand vessels in 2007 were at a higher rate than in previous years, with most of these being recaptures from 2006 releases (Dunn et al. 2007b). Dunn et al. (2007b) also noted that the locations of the recaptures were highly aggregated and were mostly from four key locations in the Ross Sea, and most fish had moved only short distances. This confirms the concern that the key uncertainty underlying the current model is

the impact of movements and spatial structure in the Antarctic toothfish population, in particular, the level and nature of the bias from non-homogeneous mixing assumptions of tagged fish.

## 5. ACKNOWLEDGMENTS

The authors thank the scientific observers and fishing company staff who collected the data used for this analysis. We also thank the members of the New Zealand Antarctic Fisheries Stock Assessment Working Group for helpful discussions and input into this paper. We thank David Ramm and Eric Appleyard for providing the data extracts from the CCAMLR Secretariat, and assisting in the interpretation of the data. This project was funded by the New Zealand Ministry of Fisheries under project ANT2006/02.

## 6. REFERENCES

Agnew, D.J.; Clark, J.M.; McCarthy, P.A.; Unwin, M.; Ward, M.; Jones, L.; Breedt, G.; Plessis, S.D.; Heerdon, J.V.; Moreno, G. (2005). A study of Patagonian toothfish (*Dissostichus Eleginoides*) post-tagging survivorship in Subarea 48.3 WG-FSA-05/19. 11 p. Marine Resources Assessment Group. Unpublished manuscript presented to the Fish Stock Assessment Working Group of CCAMLR.

Bull, B.; Francis, R.I.C.C.; Dunn, A.; McKenzie, A.; Gilbert, D.J.; Smith, M.H. (2005). CASAL (C++ algorithmic stock assessment laboratory): CASAL user manual v2.07-2005/08/21. NIWA Technical Report 127. 272 p.

CAMLR-XXV (2006). Schedule of Conservation Measures in force 2006/07 season. CCAMLR, Hobart, Australia. 25 October–28 October 2005.

Dunn, A. (2006a). Developments towards an investigation of potential bias from the analysis of tag-release and recapture data in Ross Sea Antarctic toothfish abundance estimates WG-FSA-SAM-06/10. 12 p. National Institute of Water and Atmospheric Research. Unpublished manuscript presented to the Stock Assessment Method Subgroup of the Fish Stock Assessment Working Group of CCAMLR.

Dunn, A. (2006b). Evaluations of the impact of alternative estimates of illegal catch on estimates of CCAMLR yields from a statistical catch-at-age model WG-JAG-06/10. 12 p. National Institute of Water and Atmospheric Research. Unpublished manuscript presented to the Joint Assessment Group of CCAMLR.

Dunn, A. (2006c). Standardised CPUE analysis of the Antarctic toothfish (*Dissostichus mawsoni*) fishery in the Ross sea for the years 1997–98 to 2005–06. WG-FSA-06/47. 13 p. National Institute of Water and Atmospheric Research. Unpublished manuscript presented to the Fish Stock Assessment Working Group of CCAMLR.

Dunn, A.; Ballara, S.L.; Hanchet, S.M. (2007a). An updated descriptive analysis of the toothfish (*Dissostichus spp.*) tagging programme in Subareas 88.1 & 88.2 up to 2006–07 WG-FSA-07/40. 25 p. National Institute of Water and Atmospheric Research. Unpublished manuscript presented to the Fish Stock Assessment Working Group of CCAMLR.

Dunn, A.; Gilbert, D.J.; Hanchet, S.M. (2005a). Further development and progress towards evaluation of an Antarctic toothfish (*Dissostichus mawsoni*) stock model for the Ross Sea. WG-FSA-SAM-05/12. 24 p. National Institute of Water and Atmospheric Research.

Unpublished manuscript presented to the Stock Assessment Methods Subgroup of the Fish Stock Assessment Working Group of CCAMLR.

Dunn, A.; Gilbert, D.J.; Hanchet, S.M. (2005b). A single-area stock assessment model of Antarctic toothfish (*Dissostichus mawsoni*) in the Ross Sea for the 2004–05 season WG-FSA-05/33. 38 p. National Institute of Water and Atmospheric Research. Unpublished manuscript presented to the Fish Stock Assessment Working Group of CCAMLR.

Dunn, A.; Gilbert, D.J.; Hanchet, S.M.; Bull, B. (2004). Development of an Antarctic toothfish (*Dissostichus mawsoni*) stock model for CCAMLR Subarea 88.1 for the years 1997–98 to 2003–04 WG-FSA-04/36. 39 p. National Institute of Water and Atmospheric Research. Unpublished manuscript presented to the Fish Stock Assessment Working Group of CCAMLR.

Dunn, A.; Hanchet, S.M. (2006a). Assessment models for Antarctic toothfish (*Dissostichus mawsoni*) in the Ross Sea including data from the 2005–06 season WG-FSA-06/60. 26 p. National Institute of Water and Atmospheric Research. Unpublished manuscript presented to the Fish Stock Assessment Working Group of CCAMLR.

Dunn, A.; Hanchet, S.M. (2006b). A descriptive analysis of the toothfish (*Dissostichus* spp.) tagging programme in the Ross Sea up to 2005–06 WG-FSA-SAM-06/09. 24 p. National Institute of Water and Atmospheric Research. Unpublished manuscript presented to the Stock Assessment Methods Subgroup of the Fish Stock Assessment Working Group of CCAMLR.

Dunn, A.; Hanchet, S.M. (2006c). An updated descriptive analysis of the toothfish (*Dissostichus* spp.) tagging programme in Subareas 88.1 & 88.2 up to 2005–06 WG-FSA-06/34. 24 p. National Institute of Water and Atmospheric Research. Unpublished manuscript presented to the Fish Stock Assessment Working Group of CCAMLR.

Dunn, A.; Hanchet, S.M. (2007). Revised input parameters and implications for the Antarctic toothfish (*Dissostichus mawsoni*) stock assessment in Subareas 88.1 & 88.2 WG-SAM-07/06. 32 p. National Institute of Water and Atmospheric Research. Unpublished manuscript presented to the Working Group on Statistics, Assessments, and Modelling of CCAMLR.

Dunn, A.; Hanchet, S.M.; Ballara, S.L. (2007b). An updated descriptive analysis of the toothfish (*Dissostichus* spp.) tagging programme in Subareas 88.1 & 88.2 for 2006–07 WG-SAM-07/05. 21 p. National Institute of Water and Atmospheric Research. Unpublished manuscript presented to the Working Group on Statistics, Assessments, and Modelling of CCAMLR.

Dunn, A.; Hanchet, S.M.; Maxwell, K. (2005c). Descriptive analysis of the Antarctic toothfish (*Dissostichus mawsoni*) tagging scheme in the Ross Sea for the years 1997–98 to 2003–04 WG-FSA-SAM-05/10. 18 p. National Institute of Water and Atmospheric Research. Unpublished manuscript presented to the Stock Assessment Methods Subgroup of the Fish Stock Assessment Working Group of CCAMLR.

Dunn, A.; Hanchet, S.M.; Maxwell, K. (2005d). An updated descriptive analysis of the Antarctic toothfish (*Dissostichus mawsoni*) tagging scheme in the Ross Sea for the years 1997–98 to 2004–05 WG-FSA-05/34. 18 p. National Institute of Water and Atmospheric Research. Unpublished manuscript presented to the Fish Stock Assessment Working Group of CCAMLR.

Dunn, A.; Horn, P.L.; Hanchet, S.M. (2006). Revised estimates of the biological parameters for Antarctic toothfish (*Dissostichus mawsoni*) in the Ross Sea WG-FSA-SAM-06/08. 14 p. National Institute of Water and Atmospheric Research. Unpublished manuscript presented to

the Stock Assessment Methods Subgroup of the Fish Stock Assessment Working Group of CCAMLR.

Gelman, A.B.; Carlin, J.S.; Stern, H.S.; Rubin, D.B. (1995). Bayesian data analysis. Chapman and Hall, London. 526 p.

Geweke, J. (1992). Evaluating the accuracy of sampling-based approaches to calculating posterior moments. *In: Bayesian Statistics, 4*. Bernardo, J.M.; Berger, J.O.; Dawid, A.P.; Smith, A.F.M. (eds.). Clarendon Press, Oxford. pp 169–194.

Gilks, W.R.; Richardson, S.; Spiegelhalter, D.J. (eds.) (1998). Markov chain Monte Carlo in practice. *Interdisciplinary statistics*. 399 p. Chapman and Hall/CRC Press, Boca Raton, Florida.

Hanchet, S.M.; Stevenson, M.L.; Dunn, A. (2006). A characterisation of the toothfish fishery in Subareas 88.1 & 88.2 from 1997–98 to 2005–2006 WG-FSA-06/29. 25 p. National Institute of Water and Atmospheric Research. Unpublished manuscript presented to the Fish Stock Assessment Working Group of CCAMLR.

Hanchet, S.M.; Stevenson, M.L.; Dunn, A. (2007). A characterisation of the toothfish fishery in Subareas 88.1 & 88.2 from 1997–98 to 2006–2007 WG-FSA-07/28. 20 p. National Institute of Water and Atmospheric Research. Unpublished manuscript presented to the Fish Stock Assessment Working Group of CCAMLR.

Hanchet, S.M.; Stevenson, M.L.; Horn, P.L.; Blackwell, R.G. (2003). Characterisation of the exploratory fishery for toothfish (*Dissostichus mawsoni* and *D. eleginoides*) in the Ross Sea, and approaches to the assessment of the stocks. *New Zealand Fisheries Assessment Report 2003/43*. 27 p.

Hanchet, S.M.; Stevenson, M.L.; Phillips, N.L.; Dunn, A. (2005). A characterisation of the toothfish fishery in Subareas 88.1 and 88.2 from 1997–98 to 2004–05 WG-FSA-05/29. 27 p. National Institute of Water and Atmospheric Research. Unpublished manuscript presented to the Fish Stock Assessment Working Group of CCAMLR.

Heidelberger, P.; Welch, P. (1983). Simulation run length control in the presence of an initial transient. *Operations Research* 31: 1109–1144.

Livingston, M.E.; Grimes, P.J. (2005). Size at maturity and histological procedures explored to determine spawning activity of female *Dissostichus mawsoni* from samples collected from the Ross Sea in January 2004, December 2004, and January 2005 WG-FSA-05/63. 20 p. National Institute of Water and Atmospheric Research. Unpublished manuscript presented to the Fish Stock Assessment Working Group of CCAMLR.

Phillips, N.L.; Dunn, A.; Hanchet, S.M. (2005). Stratification of catch-at-length data using tree based regression. An example using Antarctic toothfish (*Dissostichus mawsoni*) in the Ross Sea WG-FSA-SAM-05/8. 15 p. National Institute of Water and Atmospheric Research. Unpublished manuscript presented to the Stock Assessment Methods Subgroup of the Fish Stock Assessment Working Group of CCAMLR.

SC-CAMLR-XX (2001). Report of the twentieth meeting of the scientific committee, Hobart, Australia. 22–26 October 2001.

SC-CAMLR-XXIII (2004). Report of the twenty-third meeting of the scientific committee, Hobart, Australia. 25–29 October 2004.

SC-CAMLR-XXV (2006). Report of the twenty-fifth meeting of the scientific committee. CCAMLR, Hobart, Australia. 24–28 October 2005.

Smith, B.J. (2003). Bayesian output analysis program (BOA). Version 1.0 user's manual. Unpublished manuscript. 45 p. University of Iowa College of Public Health. (*see* <http://www.public-health.uiowa.edu/boa>).

WG-FSA-SAM (2006). Report of the WG-FSA Subgroup on Assessment Methods (Walvis Bay, Namibia, 10 to 14 July 2006). WG-FSA-06/6. 66 p.

WG-SAM (2007). Report of the WG-SAM Working Group on Statistics, assessments and Modelling (Christchurch, New Zealand, 9 to 13 July 2007). SC-CAMLR-XXVI/5. 42 p.