Modelling the effects of bycatch on the New Zealand sea lion (*Phocarctos hookeri*) population.

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Prepared for

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Executive Summary

This study was conducted for the New Zealand Department of Conservation (DoC) by the National Institute of Water and Atmospheric Research Ltd. (NIWA) in support of DoC's draft population management plan for New Zealand (Hooker's) sea lions (*Phocarctos hookeri*). It uses an age-structured population simulation model to evaluate the performance of alternative management strategies, or decision rules, for managing the bycatch of sea lions in the squid fishery near the Auckland Islands.

To ensure that the simulation model is realistic, the model is first fitted to sea lion population and squid fishery data, and the key population process parameters are estimated. Bayesian inference is used to estimate the joint posterior distribution of these parameters, and a set of samples of the joint posterior distribution is used as the basis for forward simulations. For model fitting, all fishery and population data sets were updated using the best available information.

Forward simulations are made for 20 years, incorporating additional uncertainty in the form of stochastic process and observation error. Projections are made with no fishing and with a range of management strategies evaluated in previous work. Management strategies are evaluated using two criteria identified in discussion with DoC as relevant to the population management plan, and we present a range of other indicators used in previous evaluations.

1. Introduction

This report describes modelling work undertaken by the National Institute of Water and Atmospheric Research Ltd. (NIWA), on behalf of the New Zealand Department of Conservation (DoC), to address the effect of bycatch of New Zealand or Hooker's sea lions (*Phocarctos hookeri*) in the squid (*Nototodarus sloanii*) trawl fishery, especially near the Auckland Islands in quota management area SQU 6T.

1.1 Background

1.1.1 Sea lions and fishing

The sea lion bycatch problem is well described (e.g. Breen et al. 2003; Wilkinson et al. 2003). *Phocarctos hookeri* is endemic to New Zealand, now occurring mostly in the sub-Antarctic and southern South Island, although the historical range was larger. About 85% of the known sea lion breeding occurs on four Auckland Islands rookeries; breeding also occurs at Campbell Island (Childerhouse et al. 2005) and in Otago (McConkey et al. 2002). Pups remain ashore while their mothers forage at sea, often at considerable distance from the rookery, during the early part of the year.

The New Zealand sea lion is classified as a "threatened species", mainly because of the limited number of breeding sites.

Squid (*Nototodarus* spp.) support an important fishery with annual export values on the order of $100 million, with much fluctuation based on both landings and price. Squid have a one-year lifespan, with rapid growth in spring and summer, and highly variable recruitment appears unrelated to spawning biomass. The fishery begins in January or early February each year and has usually finished by the end of May. Near the Auckland Islands the fishery operates almost entirely by trawling although jiggers operate in calmer waters further north (Sullivan et al. 2005).

Sea lions sometimes enter the trawl nets, probably to catch squid; some are unable to find their way out again and drown. A few come aboard alive and are released, but they are a minority of those caught. To protect sea lions, a 12 nautical mile exclusion zone was imposed around the Auckland Islands in 1986. The Auckland Islands Marine Mammal Sanctuary was established in 1994, prohibiting fishing within this same area, which became part of the Auckland Islands - Motu Maha Marine Reserve in 2004. Most of the observed sea lion bycatch occurs in the SQU 6T squid fishery; some bycatch is known to occur in the squid fishery outside SQU 6T and by other target (e.g. southern blue whiting and scampi) fisheries in the sub-Antarctic.

The squid fishing industry tries to reduce bycatch by using a Code of Practice (Maunder et al. 2000); they have analysed bycatch rates by area, depth, vessel type and date to try to identify patterns that could help them to avoid sea lions (Paul Starr, Starrfish, unpublished data); similar work is ongoing (e.g. Smith & Baird 2005). Sea lion excluder devices (SLEDs) are installed voluntarily in trawl nets in SQU 6T to direct sea lions out of the net through an escape hatch. The devices eject 44-83% (5th to 95th percentiles; median 70%) of sea lions (Breen et al. 2005), but whether survival of ejected sea lions is high is controversial.

1.1.2 Management

The bycatch problem could be managed by DoC under the Marine Mammals Protection Act 1978 ("the Act") if a population management plan (PMP) were in place. There is no PMP at present. In the absence of a PMP the bycatch is managed by the New Zealand Ministry of Fisheries (MFish) under the Fisheries Act, using an annual fisheries-related mortality limit (FRML).

For two years the FRML has been set under an operational management procedure, or decision rule, that was evaluated in 2003 by Breen & Kim (2005). This rule sets the bycatch limit as a function of the pup production estimates from the most recent two years. Before this rule was used, the FRML was set by a different formula based on the work of Wade (1998), which also set the bycatch limit as a function of recent pup production estimates, although less directly. The estimated bycatch is monitored and compared with the FRML during the season; the fishery is stopped, voluntarily or administratively, when the limit is approached. Catch limits, bycatch and season lengths will be shown below.

The work of Breen & Kim (2005) progressed earlier modelling approaches to bycatch management (Maunder et al. 2000; Breen et al. 2003) and comprised several phases:

- defining a simple mathematical population model incorporating key demographic parameters such as survival rates, maturity schedules, pupping rates, etc.,
- estimating population parameters by fitting this model to bycatch data and population data collected by DoC,
- estimating the uncertainty of parameter estimates with Bayesian techniques, especially Markov chain-Monte Carlo simulation (McMC),
- defining a set of alternative "harvest control rules", such as the formula used before 2004 to manage bycatch;
- defining a set of population and fishery indicators to use in measuring rule performance,
- defining likely ranges of environmental and observation uncertainty,
- projecting the population forward for 20 or 100 years, in sets of 5000 stochastic runs, and summarising the indicators for each harvest control rule.

The Minister of Fisheries chose one of the alternative harvest control rules for setting the FRML for 2005[1].

1.1.3 Legal background

To address management of fishing related mortality on NZ sea lions, DoC is preparing a Population Management Plan (PMP). The Act [section 3E(1)] provides for the following to be included in the PMP:

> *(f) The maximum allowable level of fishing-related mortality for the species, in New Zealand fisheries waters, which would allow the criteria specified in section 3F of this Act to be met.*

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[1] The squid fishing year runs from 1 October through 30 September and spans two calendar years, but fishing is limited to the second year and most biological work is conducted in the second year. We refer to a fishing year by its second part, *viz.* the 1992-93 fishing year is called "1993".

The criterion relevant to sea lions specified in section 3F(a) is that the level of fishing related mortality *should allow the species to achieve non-threatened status as soon as reasonably practicable, and in any event within a period not exceeding 20 years*.

DoC requested modelling work from NIWA to assist in establishing a NZ fisheries waters MALFiRM with objectives consistent with the MMPA.

1.2 Terms of reference

The work to be undertaken by NIWA under this contract includes:

1. Updating Breen and Kim model 2004 with the most recent data (up to and including 2005) including pup count data

2. Rerunning the Breen and Kim model on an objective consistent with section 3F (a) Marine Mammals Protection Act 1978.

The work to be undertaken in completing item two includes the following steps:

a) in addition to those 4 approaches already stated in appendix 2, identifying any other approaches that NIWA considers the Breen and Kim model may be used to assist in the establishment of a MALFiRM for NZ fisheries waters;

b) selecting from the range of options identified, a preferred approach and outlining the rationale for selection of that approach and the implications, advantages and limitations of that approach;

c) mathematically representing the interim management objective for New Zealand fisheries waters so that a derivation of the Breen and Kim model may be used to assist in establishing a MALFiRM for NZ fisheries waters;

d) re-running the Breen and Kim model with the mathematically represented interim management objective for New Zealand fisheries waters.

In summary, two key decisions are required: how the Breen and Kim model may be used to assist in informing the NZ fisheries waters MALFiRM (overall approach) and how the current interim management objective may be mathematically represented so that a derivation of the Breen and Kim model may be used to assist in establishing a MALFiRM for NZ fisheries waters.

1.3 Breen-Kim model options

At a meeting held with stakeholders on 18 August, DoC identified and discussed three options to inform establishment of New Zealand fisheries waters MALFiRM described in 2a) of the terms of reference. These were:

1. Using Breen and Kim (2005) to inform the MALFiRM for New Zealand fisheries waters using the current model and objectives (i.e. using the Breen & Kim model as it was used by Breen & Kim (2005) for the Auckland Islands and extrapolating the Auckland Islands results to the whole of New Zealand).

2. Using Breen and Kim (2005) to inform the MALFiRM for New Zealand fisheries waters using the current model with existing or amended objectives and incorporating Campbell Island, Stewart Island and Otago Peninsula populations into the model.

3. Using Breen and Kim (2005) to inform the MALFiRM for New Zealand fisheries waters using the current model and rerunning the model using objectives consistent with the Marine Mammal Protection Act 1978, and providing separately for the Campbell Island and Otago Peninsula populations with calculation of a MALFiRM using the approach of Wade (1998).

At the meeting of the 18th August, 2005, a fourth option was also identified by stakeholders for consideration:

4. Using Breen and Kim (2005) to inform the MALFiRM for New Zealand fisheries waters using the same objectives as in that study, and providing and providing separately for the Campbell Island and Otago Peninsula populations with calculation of a MALFiRM using the approach of Wade (1998).

NIWA considered whether other options should be considered. For the Auckland Islands population, the Breen and Kim (2005) model is considered the best available information, and so is the obvious choice. Data are severely limited outside the Auckland Islands, so the limited-data approach of Wade (1998) is the obvious choice.

These options involve two decisions: first, how to deal with the sea lion populations outside the Auckland Islands rookeries, and second, how to frame the modelling objectives.

Although the Auckland Islands population comprises the majority of the current New Zealand-wide population, there is substantial breeding on Campbell Island, some limited but persistent breeding in Otago, and potential breeding at other sites if the population extends its breeding back into the historical range. Option 1 would ignore breeding outside the Auckland Islands and thus would not address the needs of a PMP.

At the same time, data from breeding outside the Auckland Islands are very sparse. Data from Campbell Island are limited to pup production, with essentially one good recent estimate (Childerhouse et al. 2005). Better data are available from the Otago breeding (McConkey et al. 2002) but numbers involved are very small. The Breen-Kim (2005) model could be fitted to the combined data sets, but the resulting estimate of carrying capacity (K) would be misleading. The model can estimate K for the Auckland Islands, but not for the New-Zealand wide population, because the data are far too limited. Option 2, because it would be likely to give a misleading estimate of K, also would be unlikely to address the needs of the PMP.

Options 3 and 4 require an alternative approach for the sea lion populations outside the Auckland Islands, and the obvious approach to choose is that of Wade (1998), designed for use in data-limited situations. The Wade formula is:

$$C_y^{MALFRM} = 0.5\frac{N_{y-1}^{vuln} + N_{y-2}^{vuln}}{2} R_{\max} F_r$$

where C_y^{MALFRM} is the bycatch limit, N_{y-1}^{vuln} is a conservative estimate of vulnerable numbers in the previous year, $R_{\max}$ is the maximum rate of population increase and F_r is a "recovery factor". N_y^{vuln} was taken as the lower 20th percentile of the total population estimate, obtained from pup production estimates and assumptions about key population parameters (Gales and Fletcher 1996).

Wade (1998) suggested that 0.12 would be a suitable default value for $R_{\max}$ in pinnipeds. The value 0.08 was adopted in New Zealand. This may be too high for *Phocarctos hookeri*: Breen & Kim (2005) estimated a median value of 0.03 for the comparable λ, although it is possible their model structurally limits density-dependence.

The recovery factor F_r allocates the annual population increase between the fishery (in proportion F_r) and the recovering population (in proportion 1-F_r). In the United States, 0.10 is used for "endangered" species, although 0.50 allowed all Wade's (1998) simulated populations to recover as expected. The value used in New Zealand was 0.15. Choice of the value for F_r is not clear-cut. The Wade formula as used in New Zealand was arguably too conservative (Breen & Kim 2005), because it was shown to restrict the fishery severely for only small population consequences, even with $R_{\max}$ = 0.08, so F_r = 0.15 might be too conservative also. A recovery factor of 0.50 should be considered, seeing that it allowed all Wade's (1998) simulated populations to recover as expected.

1.4 Objectives

Options 3 and 4 both would use the Breen & Kim (2005) model to explore MALFiRM rules for the Auckland Islands population, and would use Wade's (1998) approach for other areas. These options differ in how the objectives are formulated. Option 4 would retain the criterion used by Breen & Kim (2005), agreed by the AEWG in 2003. This was: bycatch management should produce a 90% likelihood, for any year, of **either** the population being at or above 90% *K* **or** the population being at or above 90% of where it would have been in the absence of fishing.

At the meeting of 18 August the draft interim management objective for NZ fisheries waters was identified as:

> The New Zealand sea lion population trend increases at a rate that is not reduced by more than 10% compared to the increase that would have been achieved with zero fishing related mortality

Some discussion was held at the meeting of 18 August 2005 at which clarification was sought on the phrase …"***trend increases at a rate…***" as there was some concern that this was an ambiguous objective.

At a meeting with NIWA on 30th August, the Department clarified that the use of the word 'trend' had been inserted to recognise the natural variability that may be occurring in the population that will need to be considered when assessing the population growth in the absence of fishing pressure.

On this basis the objective should be interpreted to read as follows:

> *the New Zealand sea lion population[1] increases at a rate[2] that is not reduced by more than 10% compared to the increase that would have been achieved with zero fishing related mortality*
>
> 1. *Taking into account natural variation*
> 2. *Refers to sea lion population growth over time*

DoC further clarified the objective in discussions with NIWA. They advised that if the New Zealand sea lion population has stabilised independently of the effects of fishing, maintenance of that stable population size will be required. If, however, the population is exhibiting an increasing trend, the rate of increase that would occur in the absence of fishing related mortality must not be reduced by more than 10%. This goal assumes that the sea lion population is either stable or increasing.

However, it is possible that the population could decrease even without fishing-related mortality, for instance through the agency of disease, fluctuations in food or climatic effects.

If a modelled population increased over 20 years in a projection run without fishing-related mortality, it would be simple to find the rule that caused no more than a 10% decrease in the rate of increase. If, however, the population was precisely stable during a run (as might be expected from a deterministic population at K), no fishing-related mortality would be permitted under a strict interpretation of the objective. This would not be a realistic way to manage a population near K, as the Auckland Islands populations may well be.

If the population decreased, as many runs in a set will do in the absence of fishing-related mortality, interpreting the objective as stated above becomes fraught. For instance, it could be interpreted to mandate by catch management that ensured that the rate of decrease did not increase by more than 10%. It is by no means safe to assume that intent for the objective. Further, this interpretation would allow higher bycatch in situations where rates of population decrease were higher, which is counter-intuitive.

In our view, a realistic way to approach the objective is to compare the population behaviour of each run with the behaviour under no fishing-related mortality. If a management strategy allows the population to remain at or above 90% of where it would be in the absence of fishing, this would satisfy the intent of the interim management objective.

In discussion with DoC, it was agreed that Option 3 would be used in this study. Formally stated, the criterion against which bycatch management strategies (or harvest control rules) should be measured is that the probability that the population in any year is at or above 90% of where it would have been in the absence of fishing should be 90% or greater, or

$$P\left(N_t^{Rulen} \geq 0.9 N_t^{Rule0}\right) > 0.90$$

where N_t^{Rulen} is the mature population in year *t* under Rule *n* and N_t^{Rule0} is the mature population in year *t* under no fishing.

Option 4 would use the either/or criterion used by Breen & Kim (2005):

$$P\left[\left(N_t^{Rulen} \geq 0.9N_t^{Rule0}\right)OR\left(N_t^{Rulen} \geq 0.9K\right)\right] > 0.90$$

A proposal to do this was mooted by stakeholders and is a variant of Option 3 above; it has not been discarded as Option 1and 2 were. In response to a request from DoC we examined this criterion also.

2. Data

The terms of reference included "updating the Breen and Kim (2005) model ... with the most recent data (up to and including 2005) including pup count data". The most recent population data were obtained from DoC (Louise Chilvers, pers. comm.). The most recent fishery data - bycatch and effort estimates - were obtained from a recent modelling project presented to the MFish Aquatic Environment Working Group (AEWG) (Breen et al. 2005), here called the Breen-Kim-Starr (BKS) model.

For both the population and fishery data, the complete data sets were reviewed. Numerous minor revisions and updates from new data and were made. As will be shown below, these changes to the data used by the model had small effects on model results. Changes can be identified by comparing data sets of interest in this report with those of Breen & Kim (2005).

2.1 Fishery data

The model uses fishery data in three ways. First, during minimisation and in McMC simulations made to estimate the posterior distributions of parameters, the estimated bycatch is removed from the model population using the model's estimated vulnerability-at-age. Second, in projections the model calculates a catchability coefficient for each year, 1988-present, from the number of tows made, the bycatch estimate and the model's vulnerable population size. This is then used to determine potential bycatch in projections, given effort and vulnerable numbers each year. Third, in projections the model uses the mean and variance of observed "attempted tows" to model unrestricted fishing effort for each projected year. This is the estimated number of tows that would otherwise have been made in a year when the fishery was closed early through the operation of an FRML, obtained by assuming a normal season of 13 weeks and extrapolating the effort made up to the closing time.

There is no single definitive version of the fishery data. Various data have appeared in a variety of publications, mostly ephemera or based on ephemera such as Fisheries Assessment Reports, end-of-season reports from industry, Initial Position Papers (IPPs) from MFish, etc. Fishery data are presented here with sources shown so that values can be compared: differences are usually small.

2.1.1 Fishery data: tows

The unit for fishing effort is trawl tow. Four interpretations of the annual number of tows in SQU 6T are shown in Table 1. For 1988-91 we used the number of tows estimated from bycatch and strike rate estimates presented in the most recent IPP (MFish 2005); we used the estimates from Smith & Baird (2005) for 1992 and data from the Seafood Industry Council (SeaFIC) database (Paul Starr, pers. comm.) for 1993-2005.

Table 1: Total tows in SQU 6T from the various sources shown. The first column uses the IPP estimates of bycatch divided by the IPP estimate of strike rate for each year. The following columns are from Starr (pers. comm.); Smith & Baird (2005) and Baird (2005). Boxes indicate the values uses in modelling.

Year	est total tows	Starr total tows	Smith/ Baird total tows	Baird total tows
1988	1833			
1989	3811			
1990	5318			
1991	3500			
1992	2158		2153	
1993	654	644	656	666
1994	4571	4397	2677	4660
1995	3633	3623	4000	3999
1996	4391	4412	4460	4450
1997	3514	3534	3708	3710
1998	1442	1394	1442	1413
1999	389	392	399	395
[1]2000	1183	1191	1206	1206
2001	568	562	588	580
2002	1647	1651	1635	1645
2003	1393	1383		1365
2004		2555		
2005		2646		

1: Excludes exploratory tows made in SQU 6T before 1 February.

2.1.2 Fishery data: bycatch

The estimated bycatch from four sources is shown in Table 1. The sources are the IPP (MFish 2005), the SeaFIC database (Starr pers. comm.), Baird (2005) and, for 2002-05, results from the BKS model (Breen et al. 2005). Notes from a table of unknown origin suggest that "*for 1988-90 unknown positions of Hooker's sea lion captures were assumed to be in area SQU 6T*" and that for 1993 "*three Hooker's sea lions caught by vessels which targeted scampi have been added to the estimate*". This table shows 17 sea lions caught for 1993.

We used the IPP values for 1988-1992, the SeaFIC estimate for 1993 (because of the problem with scampi bycatch), Baird's (2005) values for 1993-2001 and the Breen et al. (2005) estimates for 2002-05.

Table 2: Estimates of sea lion bycatch from four sources. The BKS estimates are the medians of posterior distributions. Boxes indicate the values uses in modelling.

Year	IPP	Starr	Baird	BKS
1988	33			
1989	141			
1990	117			
1991	21			
1992	82			
1993	17	13	17	
1994	32	31	32	
1995	109	106	109	
1996	101	101	101	
1997	123	123	124	
1998	62	62	63	
1999	14	14	12	
2000	71	71	70	
2001	67	66	64	
2002	84	84	74	61
2003	39	39	39	43
2004	118			117
2005	115			80

2.1.3 Fishery data: attempted effort

Attempted effort calculations are shown in Table 3. The mean of attempted effort, 1988-2005, is 2881 tows, compared with 2871 used by Breen & Kim (2005). The standard deviation around this mean is 1511, compared with 1567 used by Breen et al. (2005).

2.2 Sea lion population data

Population data were supplied by DoC: most recently by Dr. Louise Chilvers, earlier by Dr. Ian Wilkinson and Simon Childerhouse. Some draft autopsy reports were supplied by Dr. Johanna Pierre (pers. comm.). Unless otherwise noted below, data are those provided by Dr. Chilvers. Numerous minor revisions and updates from new data and were made. As will be shown below, these changes to the data used by the model had small effects on model results. Changes can be identified by comparing data sets of interest in this report with those of Breen & Kim (2005).

Table 3: Attempted effort calculations made under the assumption of a 13-week season and constant effort per week within a season.

Year	Tows	Closing date	Weeks	Attempted effort
1988	1833			1833
1989	3811			3811
1990	5318			5318
1991	3500			3500
1992	2153			2153
1993	644			644
1994	4397			4397
1995	3623			3623
1996	4412	4-May	13	4412
1997	3534	28-Mar	8	5743
1998	1394	27-Mar	8	2265
1999	392			392
2000	1191	8-Mar	5	3097
2001	562	7-Mar	5	1461
2002	1651	13-Apr	10	2146
2003	1383			1383
2004	2555			2555
2005	2646	20-Apr	11	3127

2.2.1 Pup production

"Pup production" is the number of births, before any mortality, and is the main source from which the model determines the number of breeding sea lions. Pup production estimates, and their associated reliability codes from 1 (good) to 4 (very poor), are shown in. For consistency with previous work, this study used only those data with reliability codes of 1 or 2.

2.2.2 Pup mortality rates

Data on pup mortality rates through mid-January are shown in. The estimated finite mortality rate is the number of "dead" pups divided by the estimated total births.

Table 4: Pup production estimates for the Auckland Islands rookeries, with reliability codes.

Year	Sandy Bay	code	Dundas	code	Figure of Eight	code	SE Point	code
1943	350	4						
1966	465	2						
1973	525	2	1000	4	29	3		
1975	420	2						
1976	481	2						
1977	428	2						
1978	434	2	2077	2				
1980	193	4						
1981	471	2	2468	3	51	3		
1982	523	2					21	3
1983	142	4						
1984	458	2						
1985	500	2	253	4	47	4		
1986	452	2	1344	2				
1987	473	2	1386	4	105	1		
1990	434	2			120	1		
1991	429	2	1132	4				
1992	489	2	1934	2				
1993	424	1	1870	2	69	1	26	3
1995	467	1	1837	1	143	1	71	1
1996	455	1	2017	1	144	1	69	1
1997	509	1	2260	1	143	1	63	1
1998	477	1	2373	1	120	1	51	1
1999	513	1	2186	1	109	1	59	1
2000	506	1	2163	1	137	1	50	1
2001	562	1	2148	1	94	1	55	1
2002	403	1	1756	1	96	1	27	1
2003	489	1	1891	1	95	1	43	1
2004	507	1	1869	1	87	1	52	1
2005	441	1	1587	1	83	1	37	1

Table 5: From the Auckland Islands rookeries, annual estimates of total pup births, mortality and the mortality rate, 1994-2005 (1994 is not used in the model).

Year	Total pups	Pups alive	Pups dead	Mortality rate
1994	2389	2304	85	3.6%
1995	2518	2206	312	12.4%
1996	2685	2389	296	11.0%
1997	2975	2729	246	8.3%
1998	3021	2350	671	22.2%
1999	2867	2572	295	10.3%
2000	2856	2689	167	5.8%
2001	2859	2468	391	13.7%
2002	2282	1826	456	20.0%
2003	2518	2078	438	17.4%
2004	2515	2347	168	6.7%
2005	2148	2034	114	5.3%

2.2.3 Tagged female pup re-sightings

Data on re-sightings of female pups tagged in 1987 and 1990-93 are shown in Table 6.

Table 6: Re-sightings of tagged female pups in the years shown. For each cohort the number in bold in the year of tagging is the number tagged.

	Tagged in					
Seen in	1987	1990	1991	1992	1993	Total
1987	**101**					
1988	0					0
1989	0					0
1990	0	**156**				0
1991	0	3	**193**			3
1992	2	11	8	**241**		22
1993	0	0	0	1	**214**	1
1994	0	0	0	1	0	1
1995	0	0	0	1	0	1
1996	0	9	11	7	13	40
1997	0	1	0	1	0	2
1998	0	2	5	5	1	13
1999	1	24	37	62	60	184
2000	3	23	47	63	68	204
2001	3	21	38	58	54	174
2002	3	14	25	65	57	164
2003	2	15	30	51	51	149
2004	1	12	27	45	48	133
2005	1	7	20	32	31	91

2.2.4 Branded females and their pups

Data on sightings of breeding females branded in 2000, and their pups, are shown in Table 7. The pup estimates include all the females known definitely to have produced pups plus half the "probable" classification.

Table 7: Numbers of females branded in 2000, subsequent re-sightings and the number of estimated pups produced by this group in the subsequent years shown.

Year	Branded females	Pups
2000	135	
2001	116	99
2002	107	69
2003	94	77
2004	82	66
2005	72	53

2.2.5 Catch-at-age and age frequency of breeding females

The model used the same age distribution of breeding females as that used by Breen & Kim (2005). These data were originally supplied by Simon Childerhouse (pers. comm.).

The age frequency of bycatch comes from autopsy reports. Part of the autopsy procedure includes estimating the age from rings in the tooth (see Gibbs et al. 2003 for details of the procedure). We used the "growth layer groups" estimate where it was available (nearly always) and the "root ridges" estimate where it was not. Some of the aged sea lions had been tagged as pups, hence their true ages were known: for these animals we used the true ages. Where the estimated age was given as a range we used the midpoint; where it was given as a non-integer or as a two-year range we used the nearest integer or lower value respectively.

Ages from 129 animals were used by Breen & Kim (2005); these were obtained from Dickie (1999) and Gibbs et al. (2002; 2003). To these we added ages from twenty-three animals from 1995-96 provided with no reference in a spreadsheet from DoC in 2000 (Ian Wilkinson, pers. comm.), from 13 animals from 2002-03 (Duignan & Jones submitted a) and 24 animals from 2003-04 (Duignan & Jones submitted b). Data are shown in Table 8.

Table 8: Age frequencies of autopsied animals caught by the squid fishery, and of breeding females. Both data sets span several years.

Age	Autopsies	Breeding population
0	3	0
1	3	0
2	10	0
3	22	0
4	18	12
5	30	44
6	27	72
7	17	107
8	12	135
9	11	128
10	15	104
11	8	73
12	6	46
13	2	38
14	3	21
15	1	17
16	0	12
17	0	7
18	0	4
19	0	2
20	0	0
21	1	0

3. Fitting procedures

The 2003 estimation model was used with exactly the same fixed values, assumptions, priors, phases and initial values used by Breen & Kim (2005).

The exception to this statement involves the data set weights. The model uses weights to modify the variance components used in maximum likelihood calculations for each data set, and it estimates a common component of error. The procedure used in stock assessments is to adjust the weights iteratively to obtain standard deviations of normalised residuals (sdnr) that are close to 1 for each data set. Because of the new and revised data used in this study, we adjusted the data set weights to the values shown in, which also shows the resulting sdnrs.

Table 9: Parameter bounds and initial values.

Parameter	Description	Phase	Lower bound	Upper bound	Prior type	Mean	Std. dev.
$\tilde{\sigma}$	common error term	1	0.025	3	uniform	–	–
K	carrying capacity of mature animals	1	1	200000	uniform	–	–
$N1$	number of 1-yr-olds in 1965	1	1	100000	uniform	–	–
$R0$	maximum pups per mature	1	0.1	0.5	uniform	–	–
z	density dependence shape	1	0.25	8	lognormal	2.5	0.3
$S0$	pup survival to mid-January	2	0.5	1	uniform	–	–
$S1$	difference between $S2$ and immature survival rate	2	0.001	0.49	uniform	–	–
$S2$	base survival rate of mature	2	0.5	1	uniform	–	–
$S3$	decline in mature survival rate with age	2	0	0.1	uniform	–	–
m_{50}	age at 50% mature	2	0	15	uniform	–	–
m_{95-50}	difference between ages at 50% and 95% mature	2	0.1	25	uniform	–	–
v_{50}	age at 50% vulnerability	2	0.01	25	uniform	–	–
v_{95-50}	difference between ages at 50% and 95% vulnerability	2	0.01	25	uniform	–	–
σ_k	standard deviation of pup estimate at rookery k	1	5	6000	uniform	–	–
Q_k	proportions in sub-populations	1	0	1	uniform	–	–
λ	the population's maximum annual rate of increase	3	0	1	lognormal	0.08	0.4
$P^{surv,re-sight}$	re-sight probabilities for tagged pups	2	0.001	1	uniform	–	–

Table 10: Data set weights and resulting sdnrs.

Input	Description	Weights	Sdnrs
$w^{pupcounts}$	data weight for pup production estimate estimates	3.00	1.000
w^{Auto}	data weight for autopsy catch-at-age data	17.00	0.948
w^{Popn}	data weight for breeding female proportion-at-age data	19.00	0.953
w^{Tags}	data weight for tagged pup cohort re-sightings data	0.66	1.193
w^{BFpup}	data weight for branded female re-sightings data	0.02	1.066
w^{BF}	data weight for branded female production	8.00	0.446
$w^{pupsurv}$	data weight for pup survival data	0.67	0.972

With the data and fixed values shown, we first obtained point estimates of parameters (these would be the maximum likelihood estimates but for the Bayesian priors). These are the mode of the joint posterior distribution (MPD). To incorporate the uncertainty of parameter estimate we estimated the posterior distributions of parameters by running a single long chain of 20 million Markov chain Monte Carlo (McMC) simulations that began at the MPD, and we saved 5000 regularly spaced samples.

4. Fitting results

4.1 Comparison with 2003

MPD results are shown in and compared with Breen & Kim's (2005) results. Table 11 also compares results from the posterior distributions of estimated and derived parameters. When all the normalised residuals are combined (Figure 1) from the MPD fit, their distribution follows the normal distribution within plus and minus 2 standard deviations.

MPD results showed some substantial changes, but medians of the posterior distributions did not change very much. The objective function (total negative log-likelihood) increased because the model was fitting to more data than in 2003, and changed data weights caused a large increase in the function contribution from branded females. There was little change in parameter estimates or in the model's estimates of λ, pupping rate or the state of the population relative to *K*.

Table 11: PD estimates and summaries of posterior distributions from the 2003 and 2005 fits of the model. NLL: negative log-likelihood. Dataset names are as defined in the text. Parameters are defined in Breen & Kim (2005). The last two indicators, N^{mat}/K and N_0/N^{mat} (mature numbers as a proportion of carrying capacity, and pups per mature female) were based on 2003 numbers in the earlier study and 2005 numbers in this study.

	2003		2005			
	MPD	**Median**	**MPD**	**5%**	**median**	**95%**
NLL						
Total	2797.7	2815.1	5457.6	5471.4	5477.6	5485.3
Pups	310.3	314.7	350.0	353.9	357.7	362.6
Auto	-45.2	-43.1	-52.0	-51.1	-49.7	-47.3
Popn	-51.8	-48.1	-54.2	-54.1	-52.1	-49.1
Tags	2362.7	2369.8	2079.1	2082.3	2086.5	2092.3
BFpups	12.3	11.9	18.9	16.8	17.3	19.9
BF	213.5	213.2	3127.5	3126.5	3127.9	3131.9
Pupmort	-6.8	-8.3	-16.4	-16.3	-15.9	-14.1
Parameters						
$\tilde{\sigma}$	0.103	0.112	0.151	0.136	0.159	0.189
S0	0.866	0.867	0.880	0.855	0.884	0.915
S1	0.084	0.080	0.096	0.095	0.114	0.131
S2	1.000	0.983	1.000	0.996	0.999	1.000
S3	0.018	0.016	0.014	0.013	0.013	0.014
Mat50	6.018	5.645	6.224	5.465	5.921	6.323
Mat95-50	1.821	2.069	1.642	0.917	1.657	2.467
v_{50}	2.86	2.60	2.536	2.031	2.546	3.071
$v_{95\text{-}50}$	0.18	1.21	1.501	0.531	1.568	2.822
σ_1	885	871	663	473	629	858
σ_2	5555	5067	3241	2963	4218	5646
σ_3	622	644	448	347	492	753
σ_4	375	407	234	182	270	436
K	7393	7376	7487	6855	7349	7945
N1	2137	1959	1230	1238	1779	2626
R0	0.500	0.495	0.500	0.489	0.497	0.500
Z	3.085	3.065	5.557	1.930	3.493	6.515
Q_1	0.177	0.178	0.182	0.173	0.182	0.191
Q_2	0.760	0.760	0.755	0.744	0.756	0.767
Q_3	0.042	0.042	0.043	0.037	0.042	0.047
Q_4	0.020	0.020	0.020	0.017	0.020	0.023
$p_{91}^{surv,re-sight}$	0.014	0.017	0.014	0.006	0.020	0.046
$p_{92}^{surv,re-sight}$	0.059	0.062	0.058	0.040	0.063	0.094
$p_{93}^{surv,re-sight}$	0.002	0.003	0.002	0.001	0.004	0.012
$p_{94}^{surv,re-sight}$	0.001	0.003	0.001	0.001	0.004	0.010
$p_{95}^{surv,re-sight}$	0.002	0.003	0.002	0.001	0.004	0.011
$p_{96}^{surv,re-sight}$	0.066	0.071	0.071	0.057	0.079	0.108
$p_{97}^{surv,re-sight}$	0.006	0.008	0.004	0.002	0.007	0.017
$p_{98}^{surv,re-sight}$	0.027	0.031	0.028	0.019	0.034	0.056
$p_{99}^{surv,re-sight}$	0.440	0.485	0.449	0.427	0.510	0.605

	2003		2005			
	MPD	Median	MPD	5%	median	95%
NLL						
$p_{00}^{surv,re-sight}$	0.569	0.630	0.555	0.531	0.635	0.747
$p_{01}^{surv,re-sight}$	0.541	0.601	0.534	0.506	0.611	0.725
$p_{02}^{surv,re-sight}$	0.663	0.734	0.579	0.549	0.663	0.788
$p_{03}^{surv,re-sight}$	0.393	0.434	0.611	0.577	0.701	0.839
$p_{04}^{surv,re-sight}$	–	–	0.645	0.603	0.739	0.899
$p_{05}^{surv,re-sight}$	–	–	0.528	0.478	0.608	0.761
λ	3.3%	3.2%	4.0%	2.5%	3.2%	4.0%
N^{mat}/K	96.2%	95.6%	99.4%	90.3%	94.8%	99.4%
N_0/N^{mat}	37.9%	38.3%	33.3%	33.5%	37.6%	40.4%

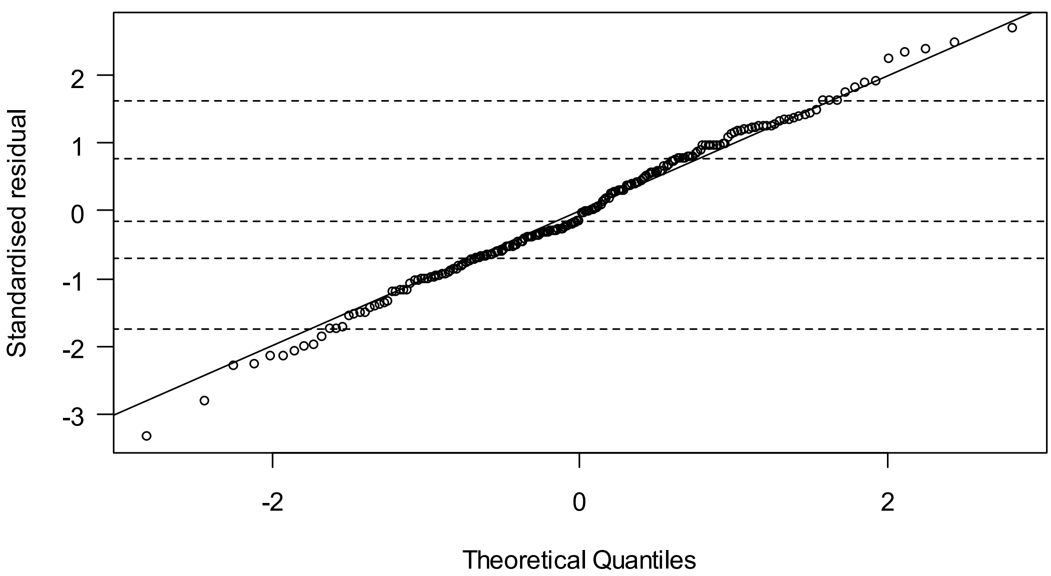

Figure 1: Q-Q plot of all normalised residuals (excluding those from the fits to tag-re-sighting data) from the base case MPD. The dotted lines show the median and 5th, 25th, 75th and 95th percentiles.

4.2 Diagnostics

To determine whether the McMC chains were converged, we first explored the traces, which are plots of the parameter estimate during the run (Figure 2). They are generally well-mixed with no apparent trend except for R0 and S2; these both have a very narrow range and are near the upper bound.

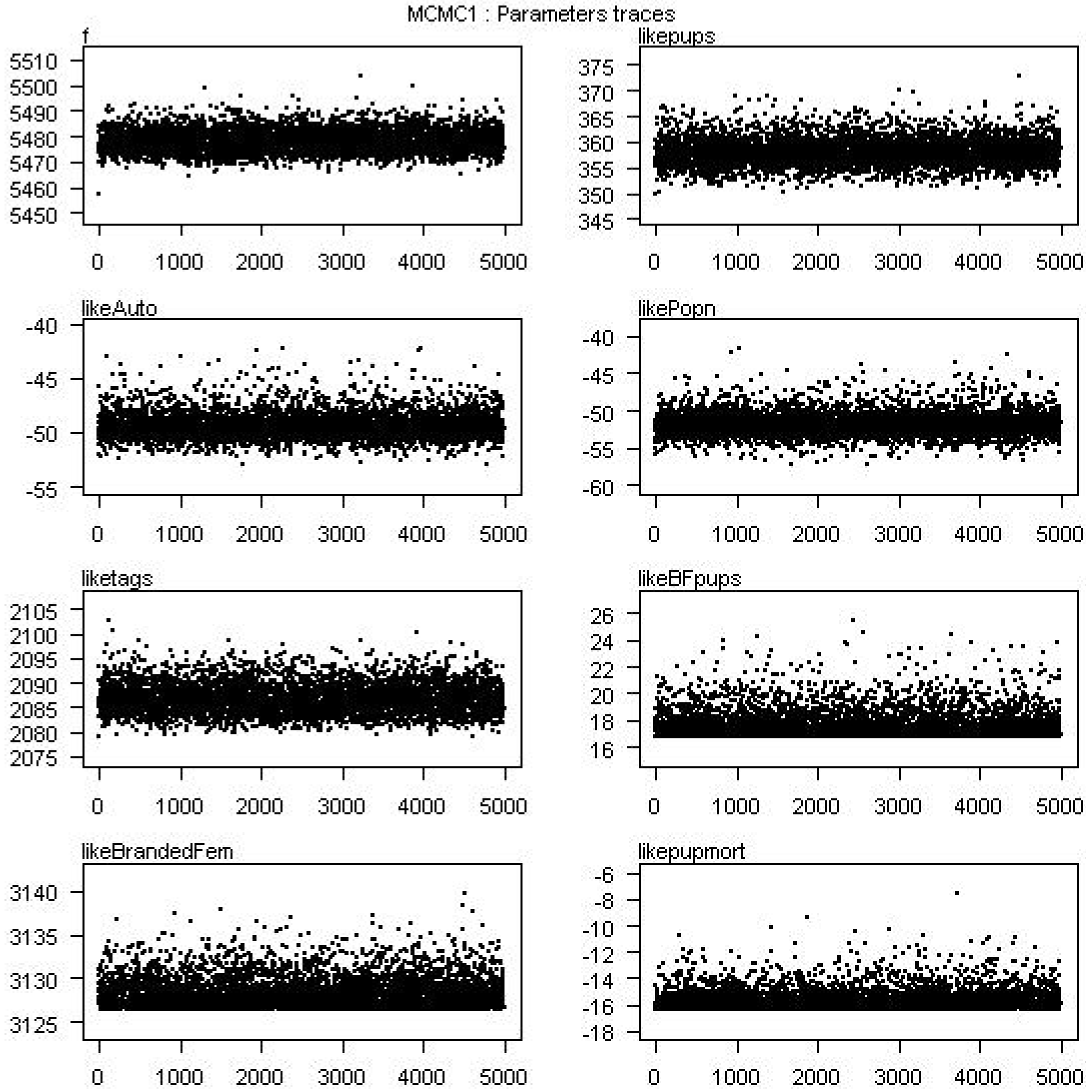

Figure 2: Traces of parameters and selected indicators.

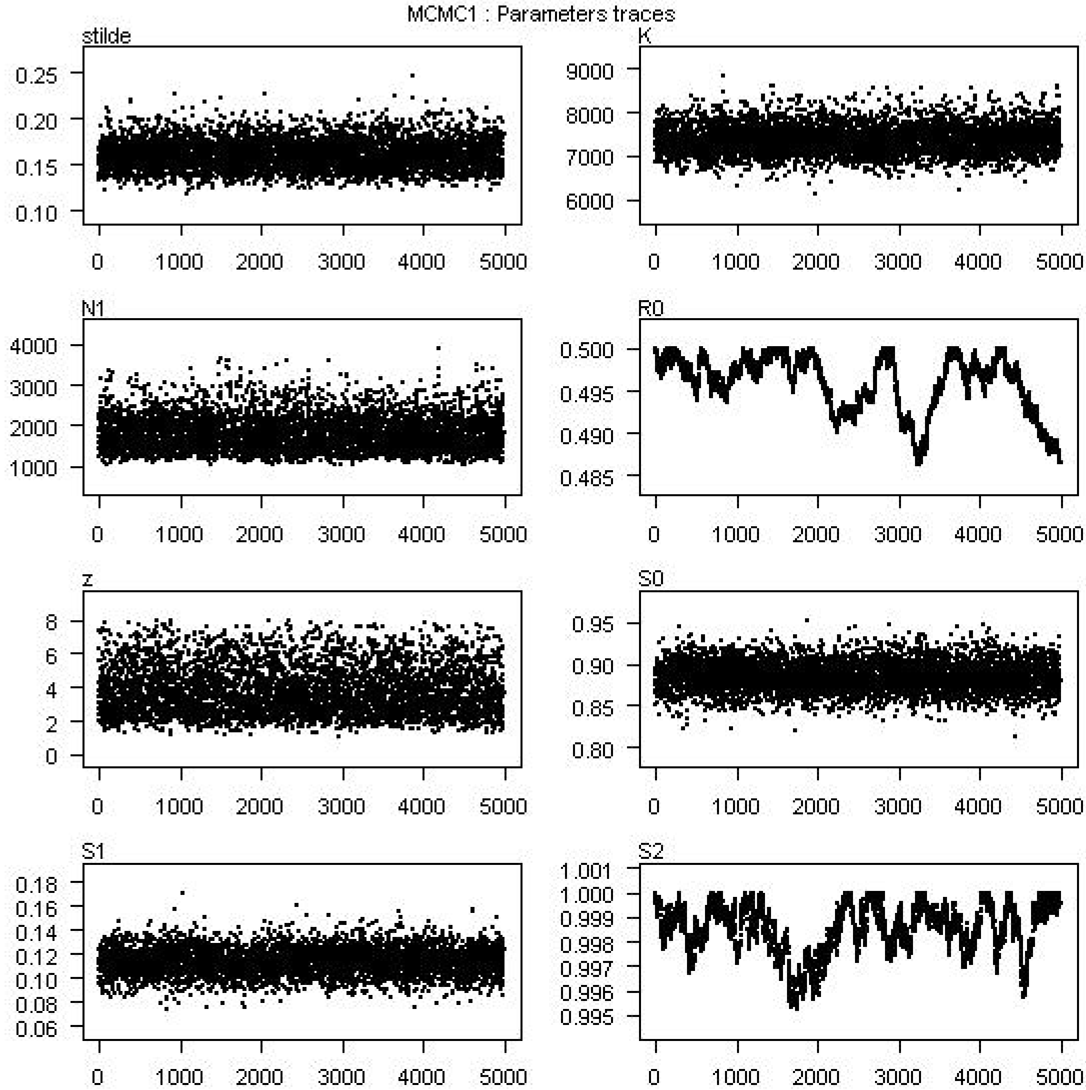

Figure 2: **continued.**

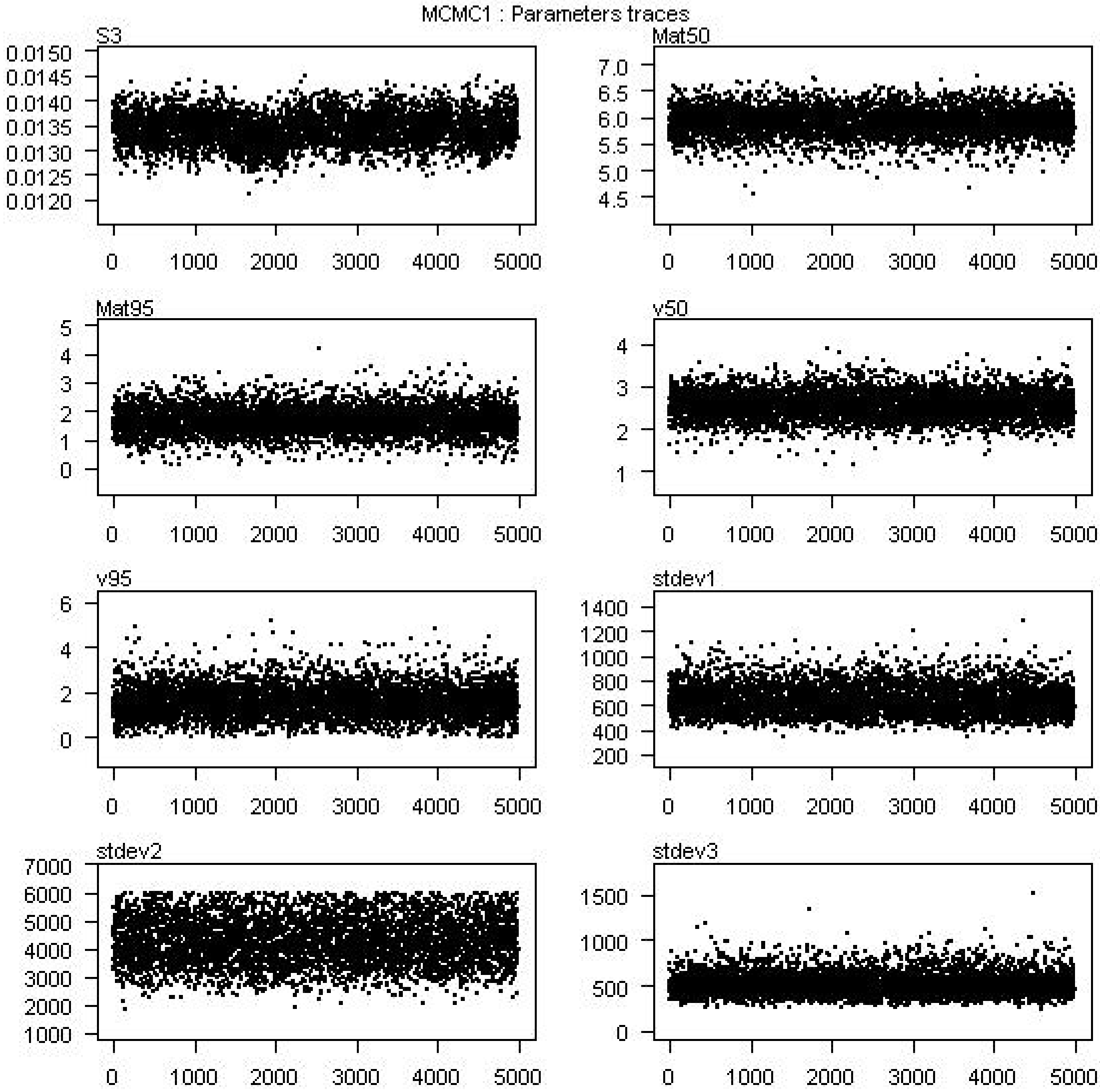

Figure 2: **continued.**

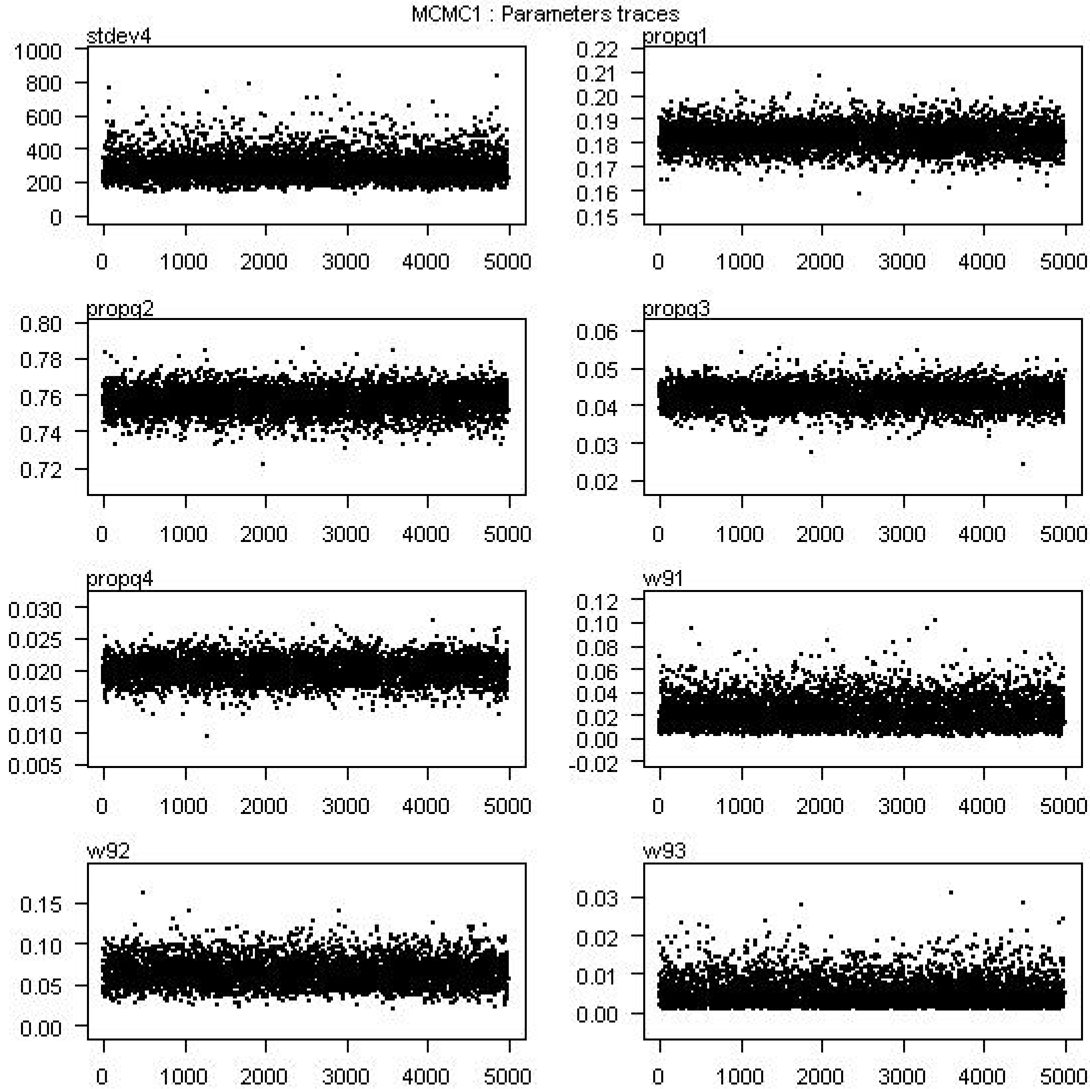

Figure 2: continued.

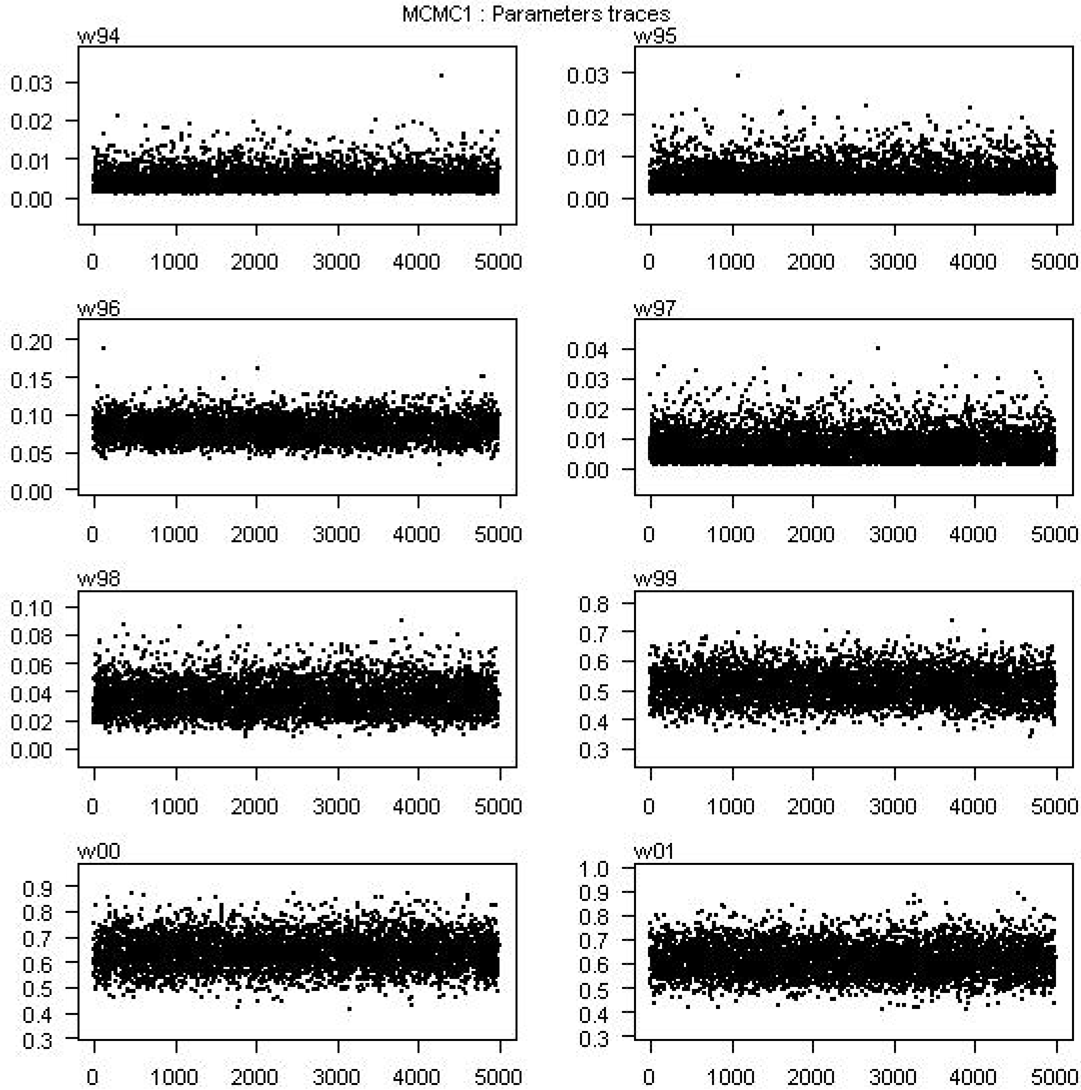

Figure 2: **continued.**

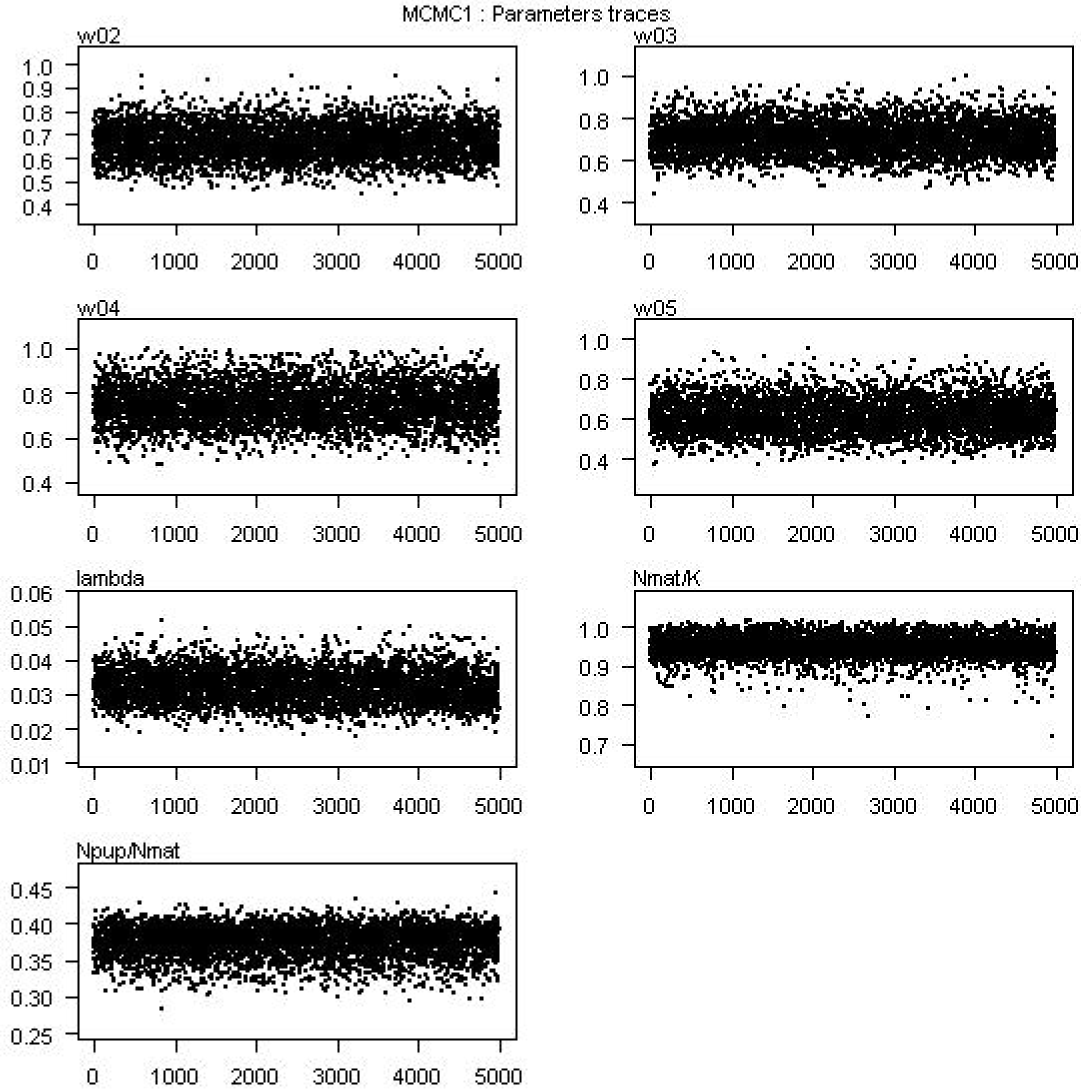

Figure 2: continued.

We also examined the chains using simple diagnostics plots comprising the running median, running 5th and 95th percentiles and moving averages for each parameter and selected indicators (Figure 3). They show no trends for most parameters except for R0 and S2, in which the McMC started from near the upper bound and fluctuates with a trend.

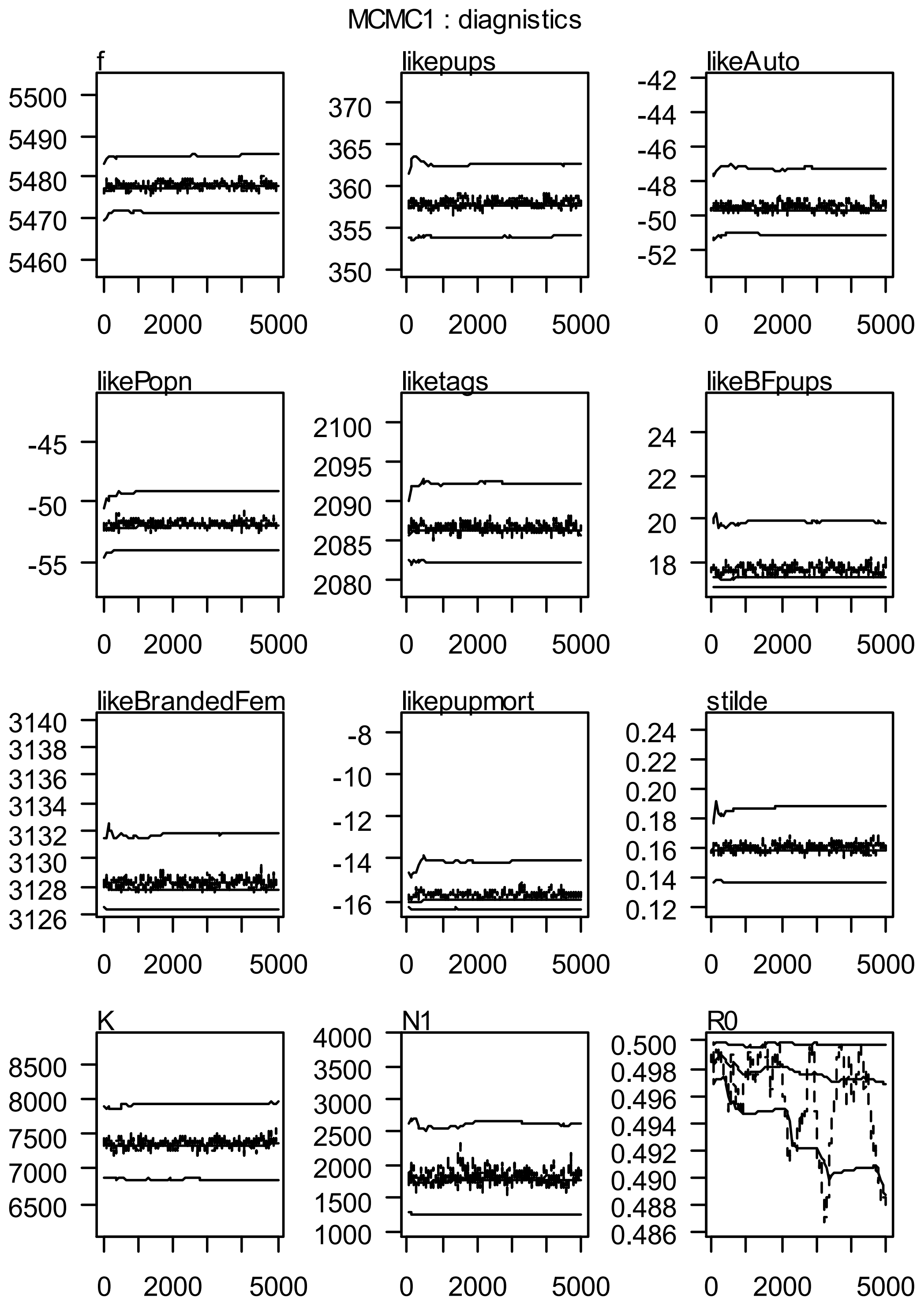

Figure 3: **Diagnostics of parameters. Upper and lower lines show the running 5th and 95th percentiles; the central solid line is the running median; the central dashed line is the moving mean over 20 samples.**

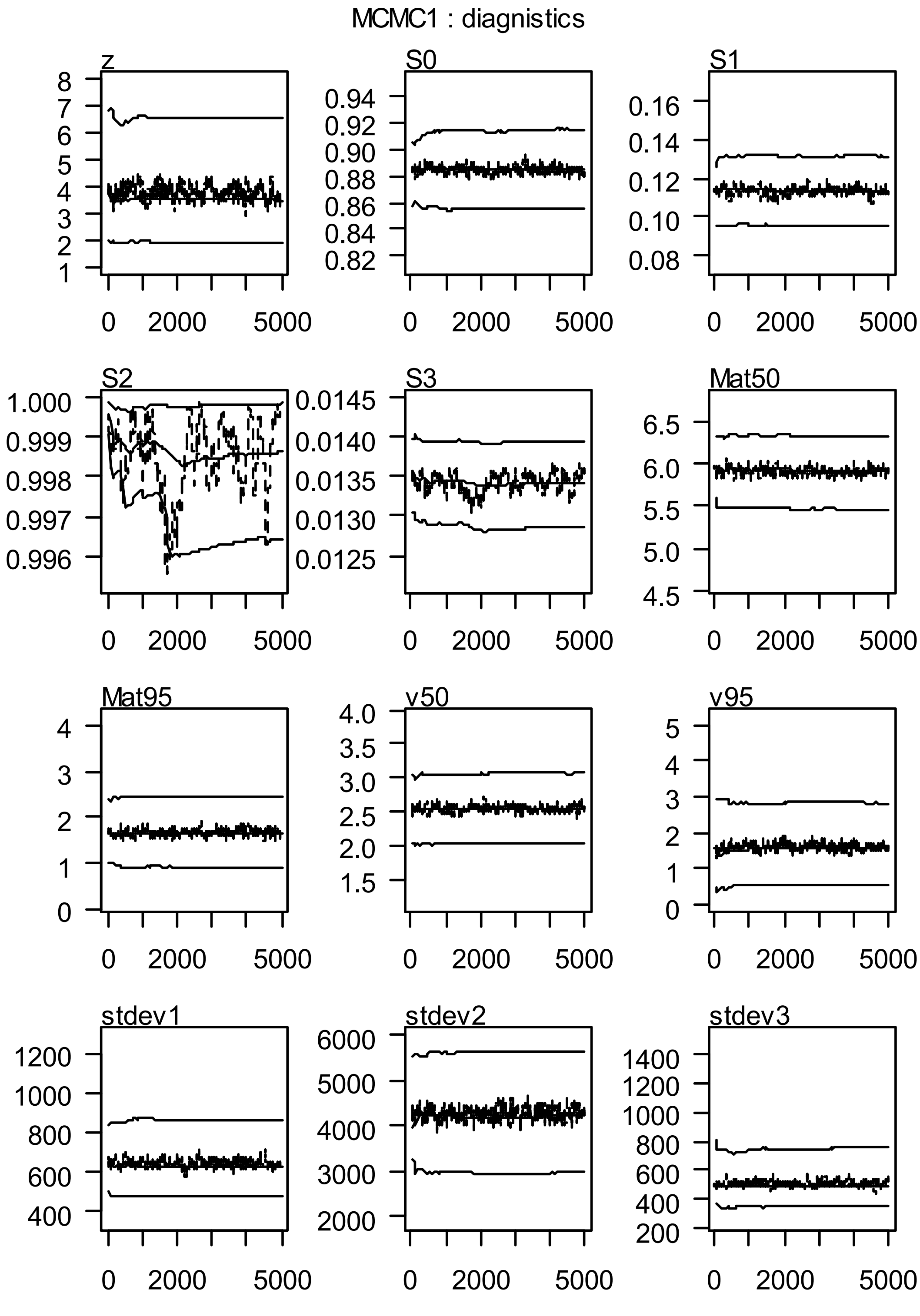

Figure 3: **continued.**

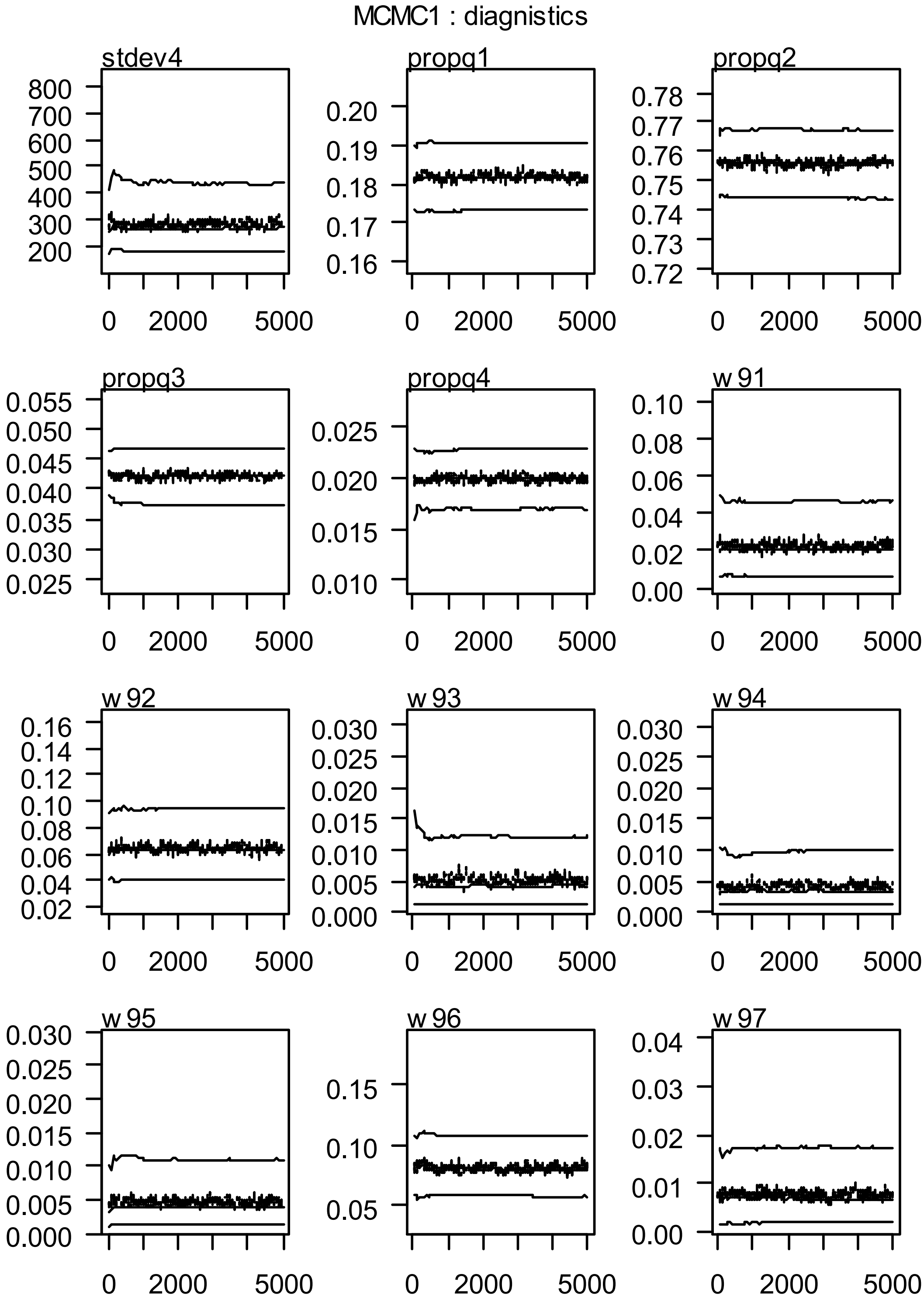

Figure 3: **continued.**

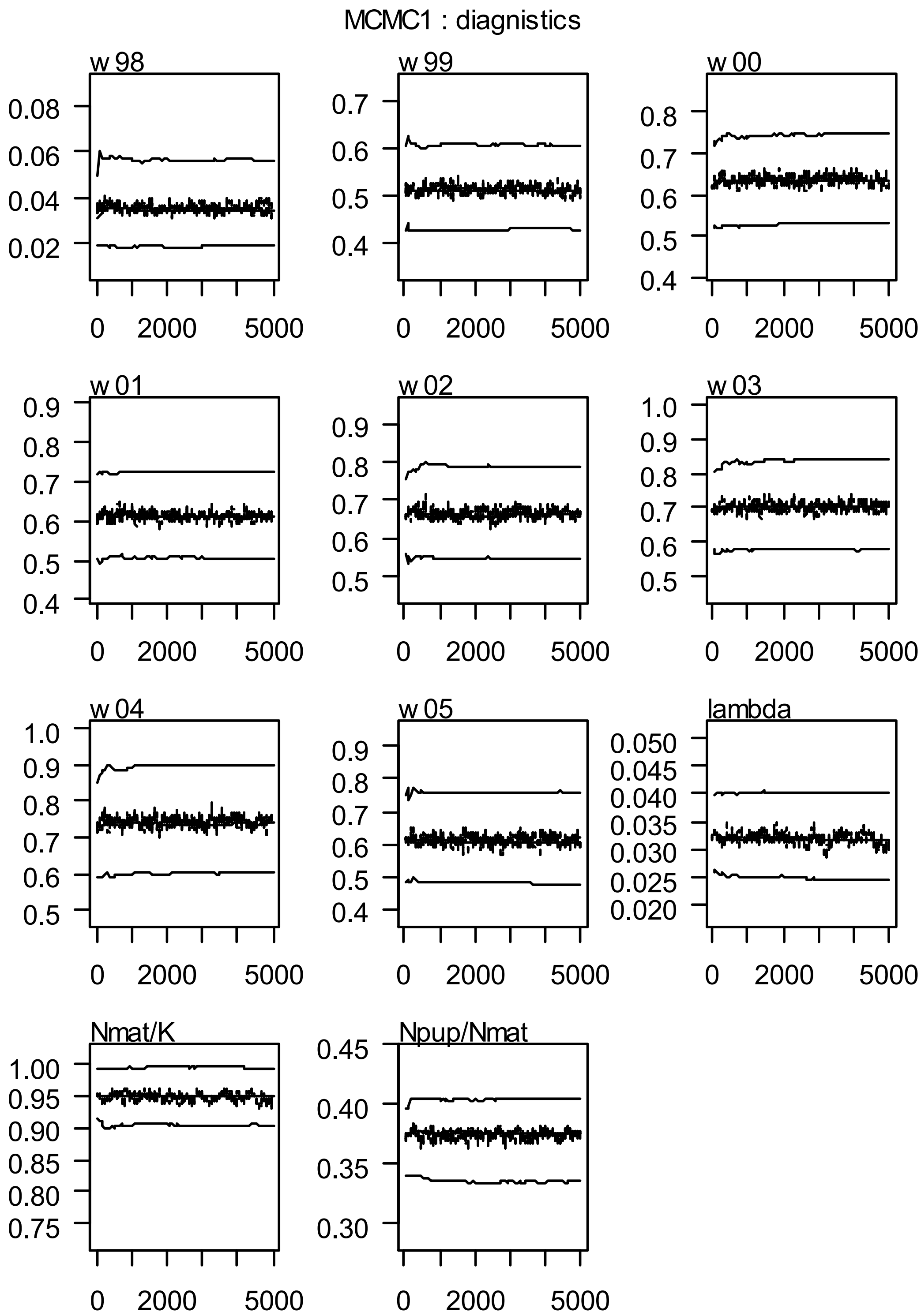

Figure 3: continued.

4.3 Posterior

Posterior distributions (Figure 4) seem well-formed. For most parameters (except the likelihood components, for which the MPD estimate (the round dot in Figure 4)should be less than most of the posterior) MPD estimates are near the centre of the posterior. Exceptions include *N1*, and *R0* and *S2* have MPDs near the upper bound.

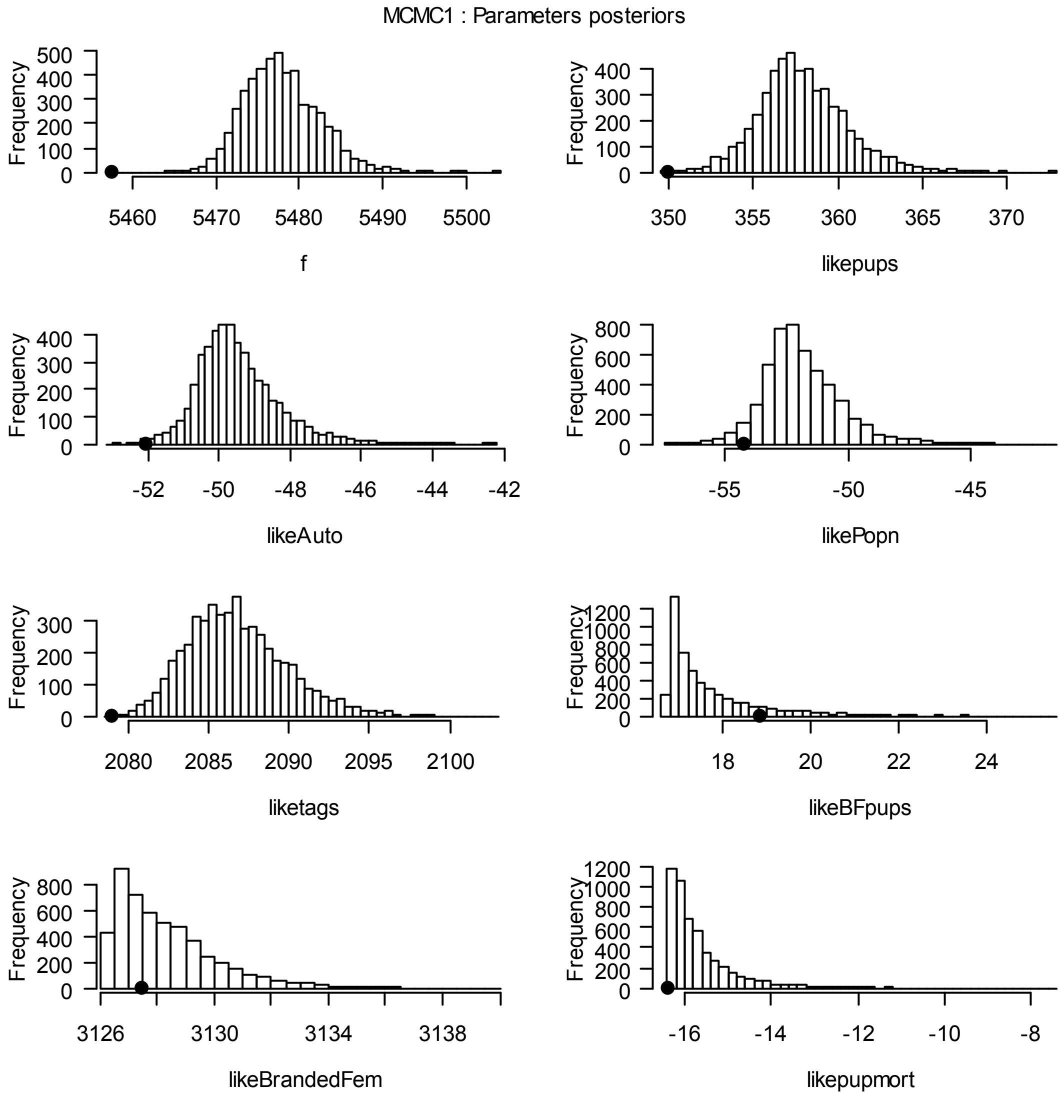

Figure 4: Posterior distributions of parameters and selected indicators. The round dot is the MPD estimate, where available.

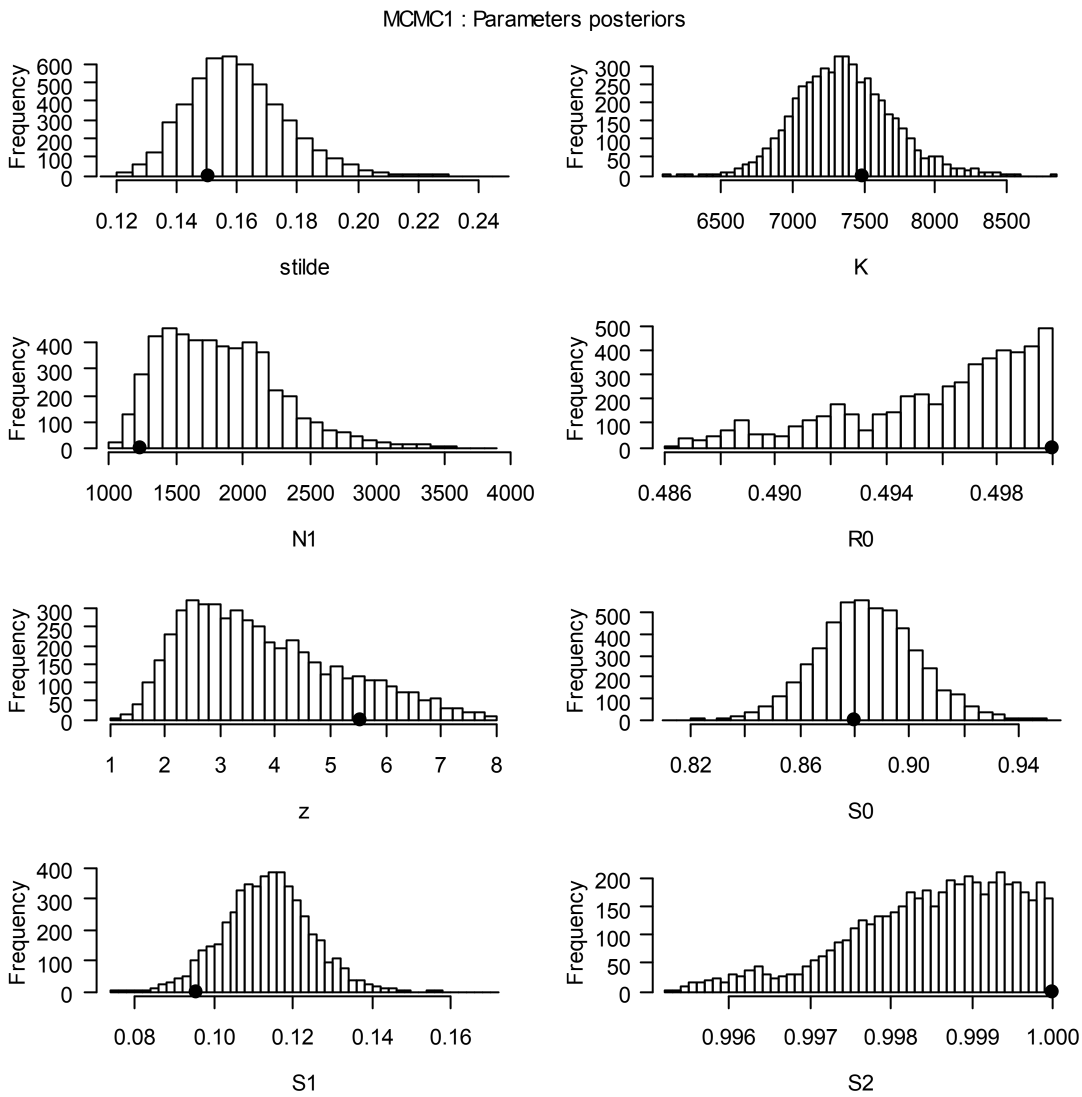

Figure 4: continued.

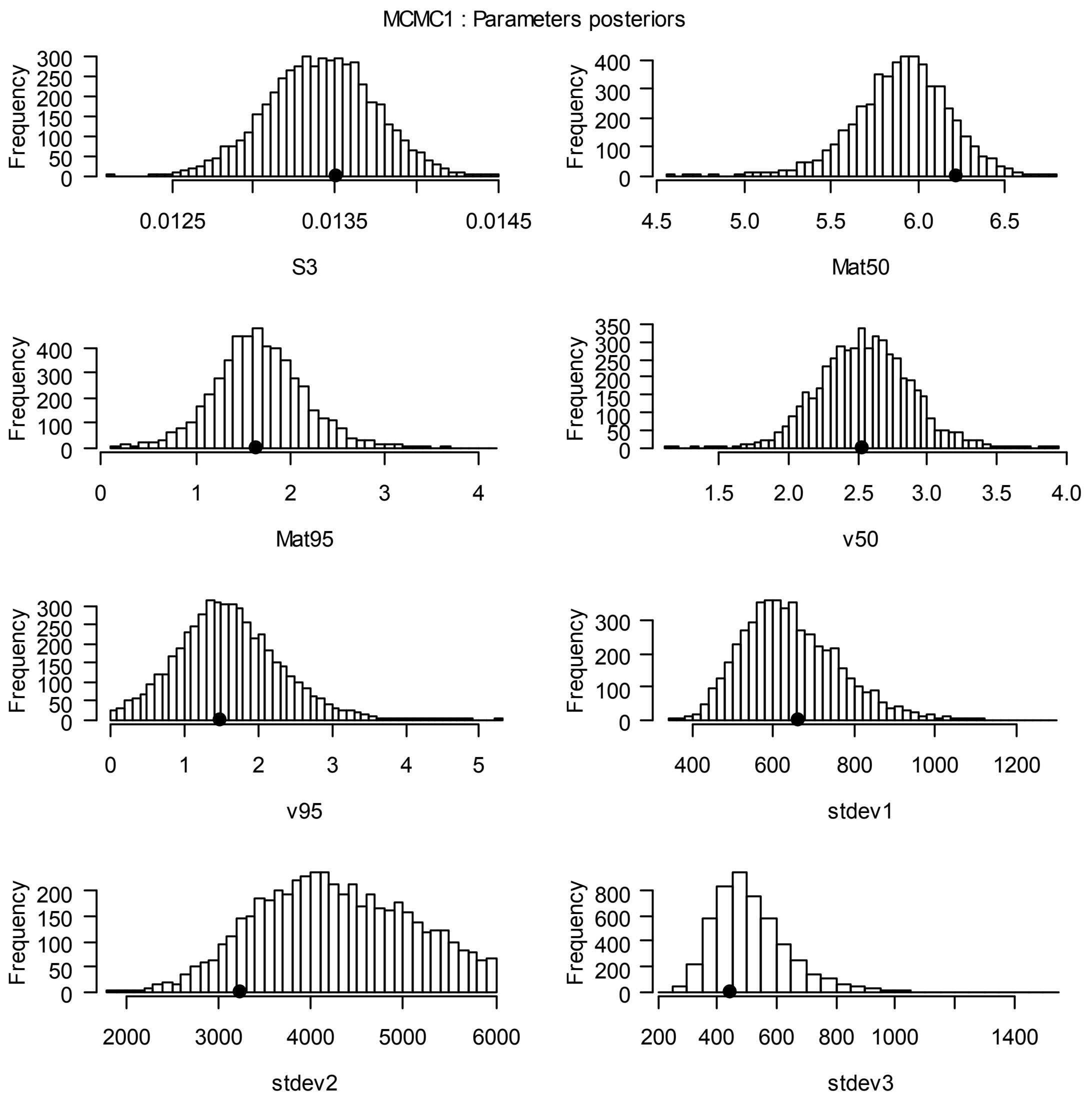

Figure 4: continued.

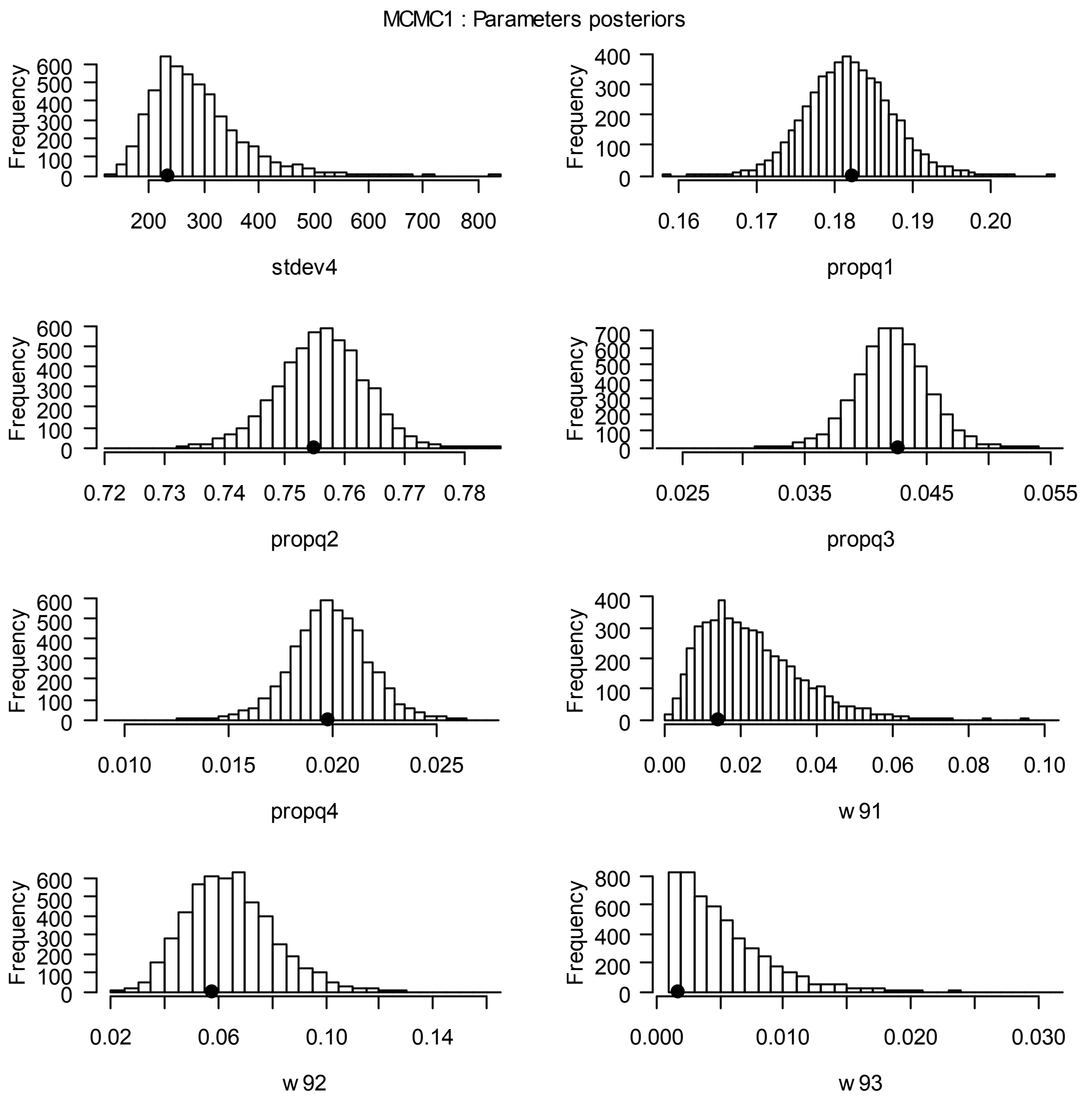

Figure 4: continued.

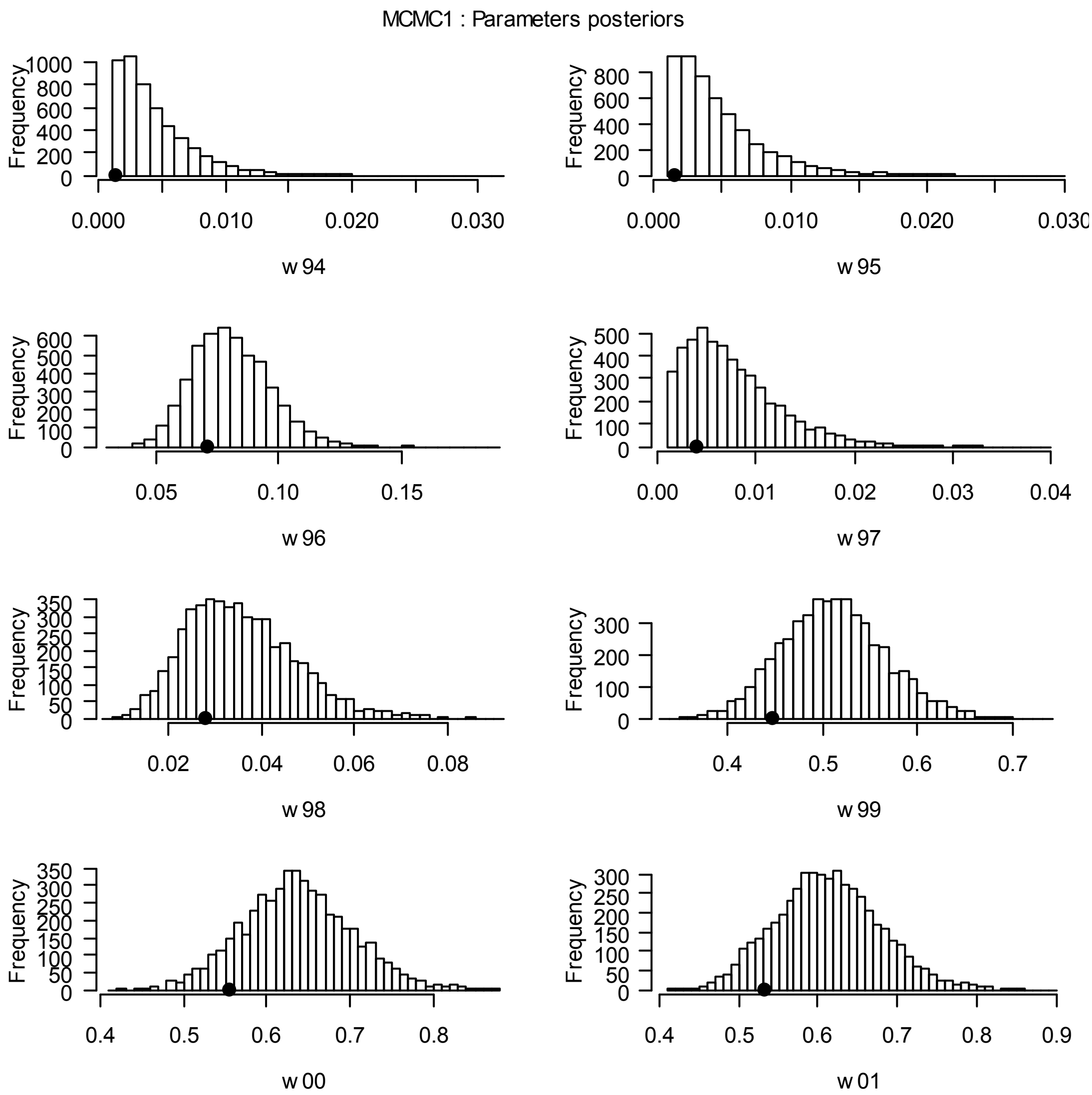

Figure 4: **continued.**

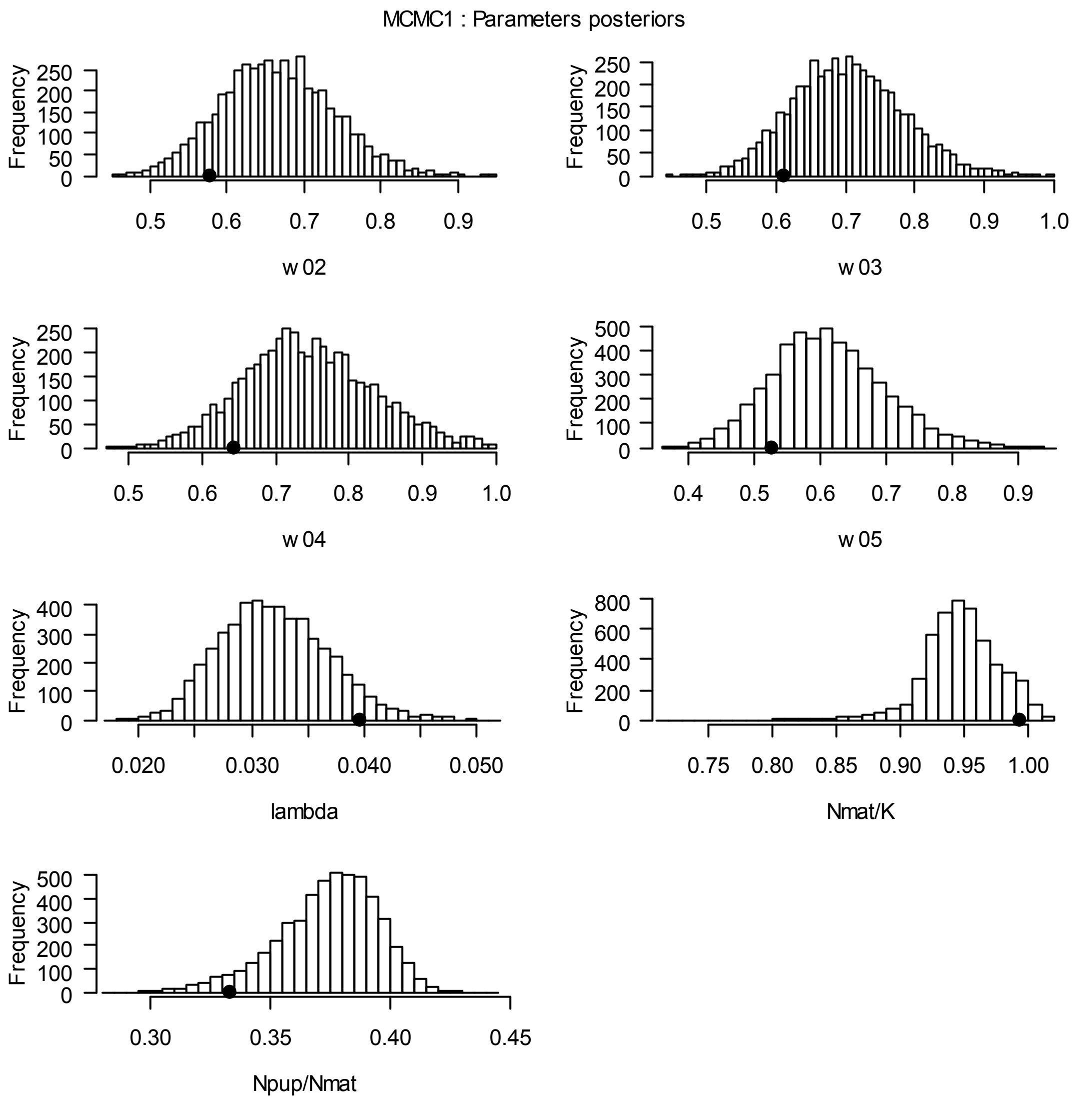

Figure 4: **continued.**

Posteriors of the survival rate-at-age, vulnerability-at-age and maturity-at-age are shown in Figure 5. The variation around the survival rate was largest for pups and older animals; generally this variation was small and was especially tight for ages 6 to 15. Estimated survival rate declined with age for mature animals. Vulnerability-at-age varied considerably between samples, with some overlap between ages, but for animals of 8 years and older vulnerability-at-age was always 1. The variation around maturity-at-age increased with age until 6 years of age then decreased. For animals older than 12, the maturity-at-age was always 1.

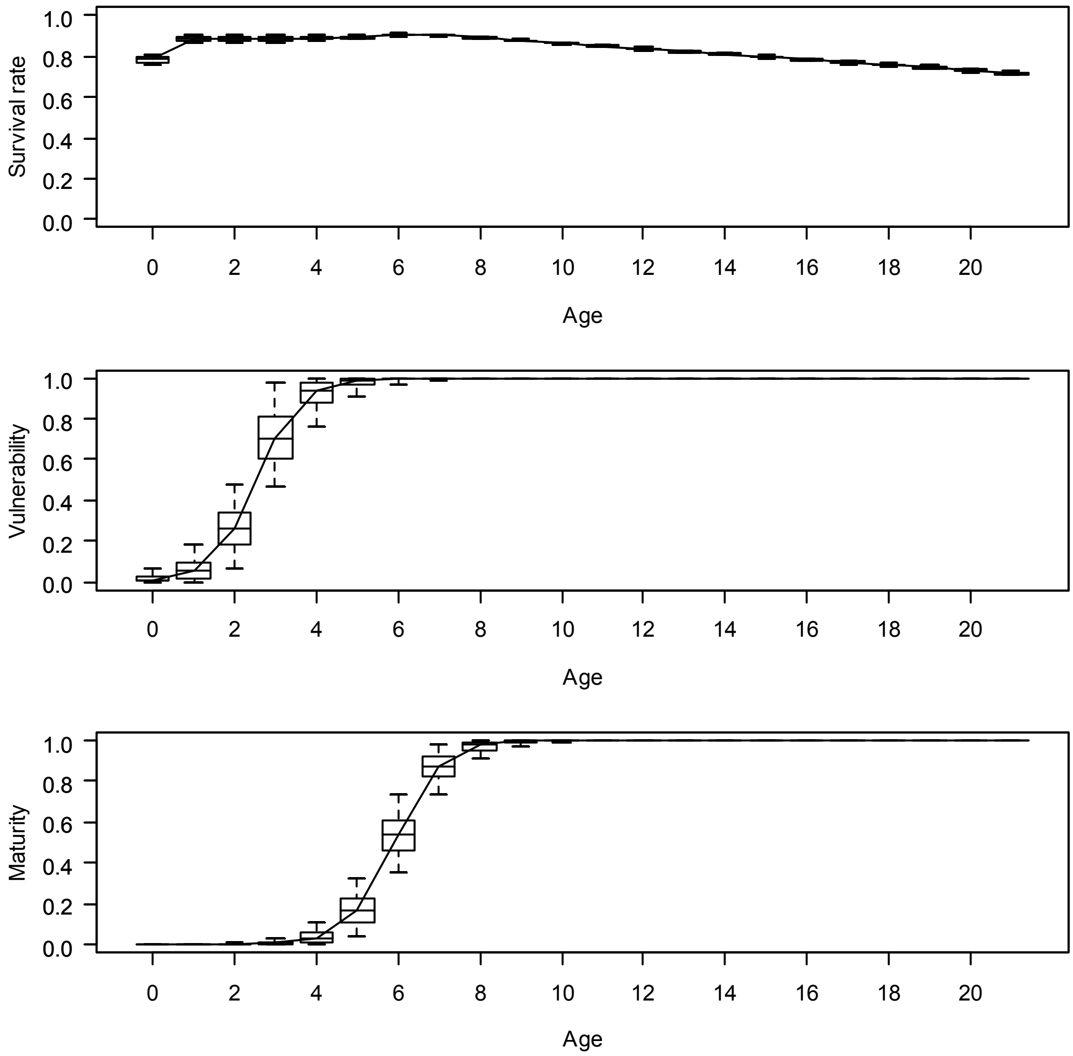

Figure 5: Summaries of the posterior distributions of survival rate-at-age, vulnerability-at-age and maturity-at-age.

The posterior distribution is compared against the assumed prior distribution for the derived parameter λ in Figure 6. The posterior is at the lower end of the prior and is much narrower than the prior, indicating that the model/data combination contain some information about λ.

The prior and posterior for z are compared in Figure 7. The posterior is closely related to the prior, but the model/data combination seems to support higher values than the prior would suggest.

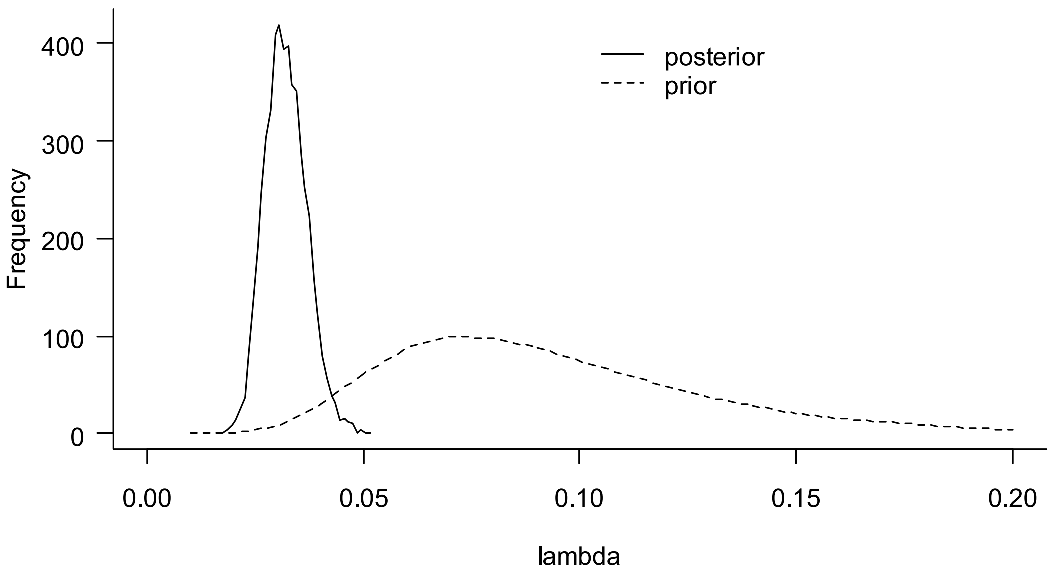

Figure 6: **Posterior (solid line) and the assumed prior distribution of lambda.**

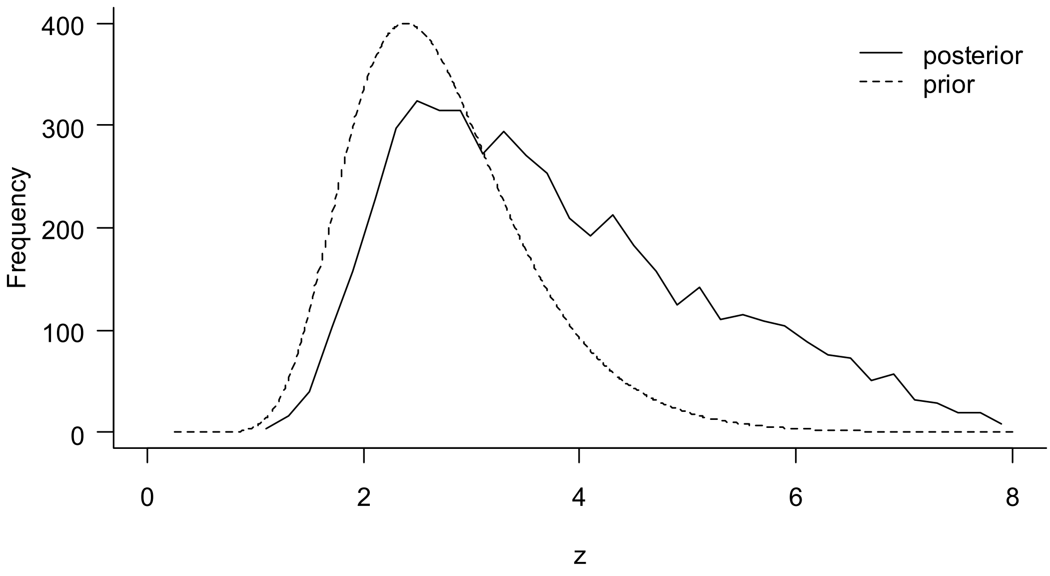

Figure 7: **Posterior (solid line) and the assumed prior distribution of z.**

4.4 Fits and residuals

The model's fits to the data (**Figure 8** to Figure 13) were much the same as in 2003. The fit to pup production estimates is very flat (**Figure 8**), reflecting the model's reconstruction of a stable population. Fits to the tag re-sighting data (Figure 9) were quite good except for the first tagged cohort, from 1987; fits to the branded females were good (Figure 10). Fits to the two -at-age data sets "Auto" and "Popn" (Figure 11 and Figure 12) showed the patterns discussed by Breen & Kim (2005). Fit to the number of pups from branded females ("BFpups") was good (Figure 13).

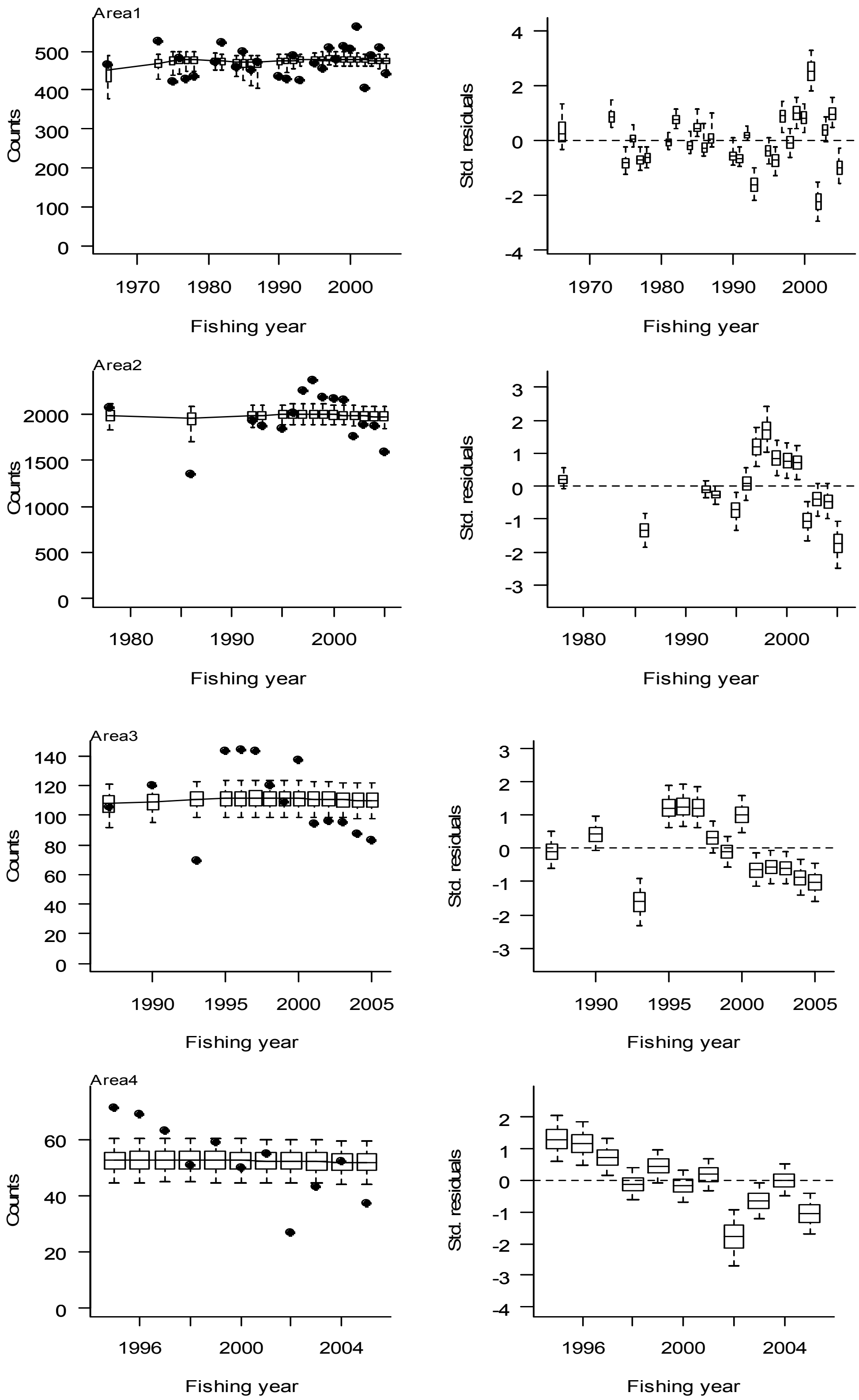

Figure 8: **Fits to pup birth estimates (left) and normalised residuals (right) for (from the top) Sandy Bay, Dundas, Figure of Eight and SE Point. The box plots summarise posterior distributions: median, 25th and 75th percentiles (box) and the 5th and 95th percentiles (whiskers). Dots on the left-hand plots are the observed values.**

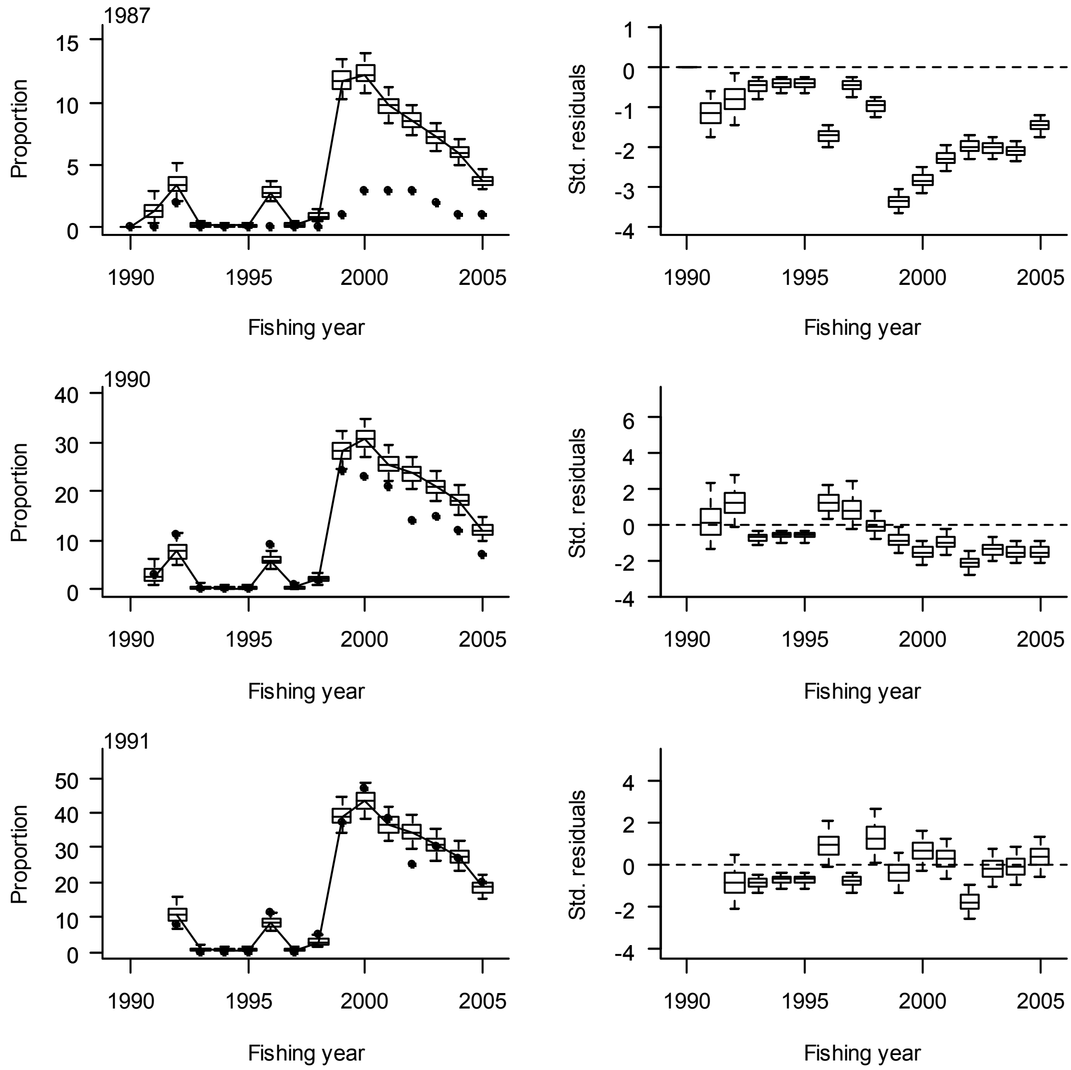

Figure 9: **Fits and residuals to the tagged pup re-sighting data. The year of tagging is given at the top left for each fit plot. The box plots summarise posterior distributions: median, 25th and 75th percentiles (box) and the 5th and 95th percentiles (whiskers). Dots on the left-hand plots are the observed values.**

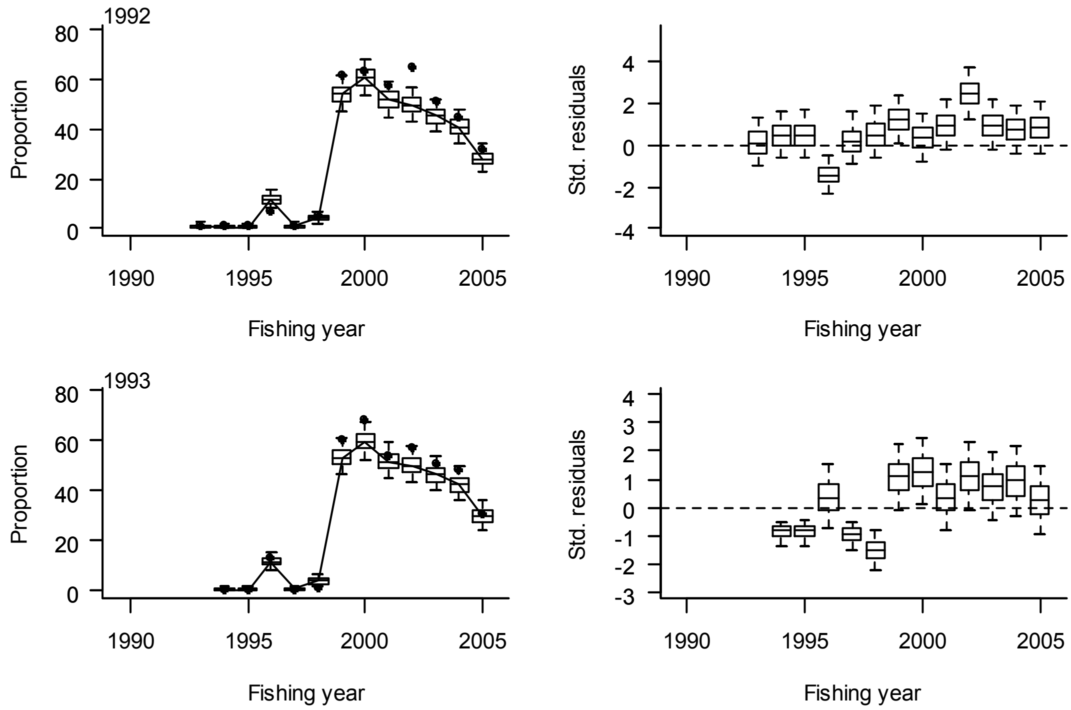

Figure 9: continued.

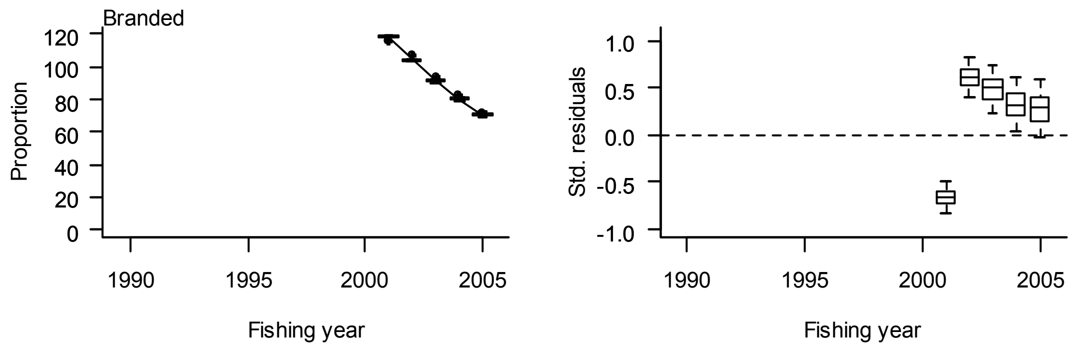

Figure 10: Fits and residuals to the branded female re-sighting data. The box plots summarise posterior distributions: median, 25th and 75th percentiles (box) and the 5th and 95th percentiles (whiskers). Dots on the left-hand plots are the observed values.

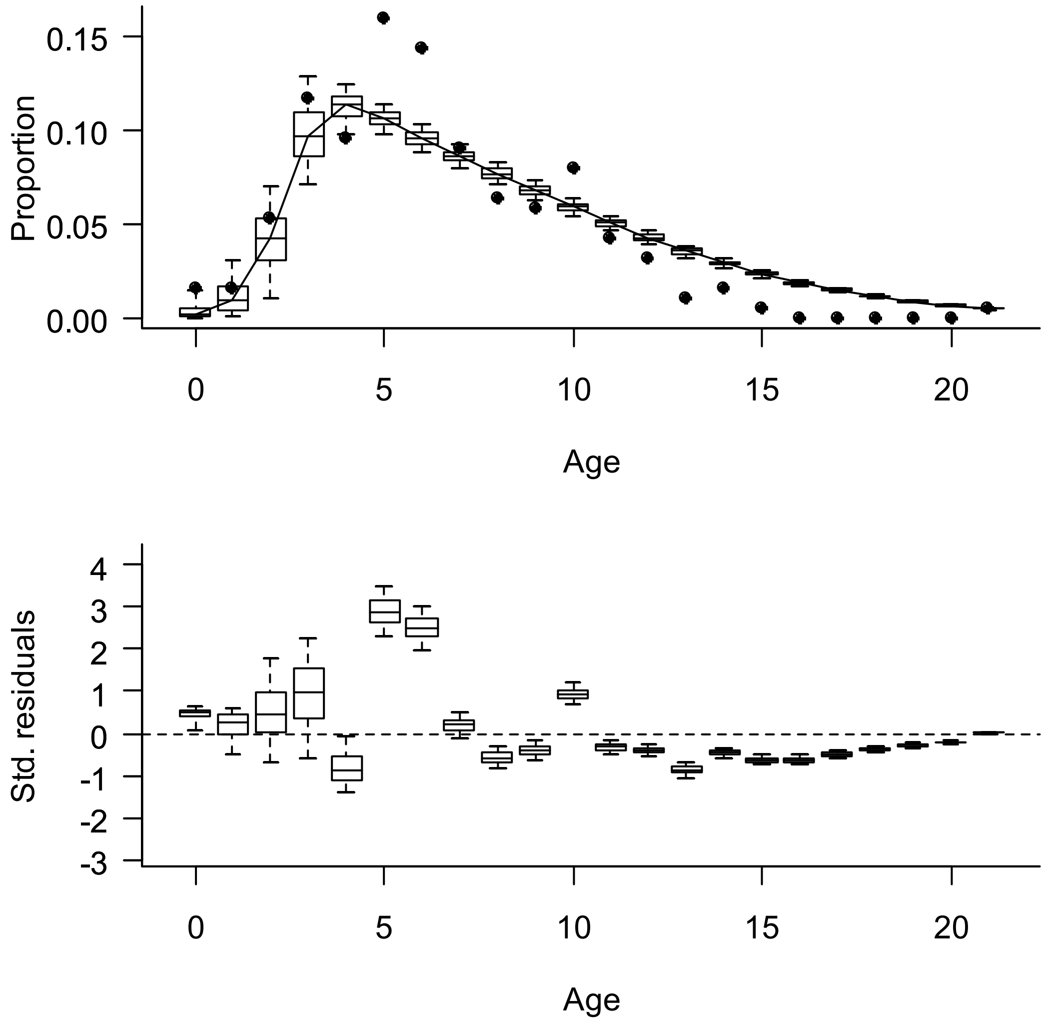

Figure 11: The fit (upper) and residuals for the catch-at-age data. The box plots summarise posterior distributions: median, 25th and 75th percentiles (box) and the 5th and 95th percentiles (whiskers). Dots on the upper plot are the observed values.

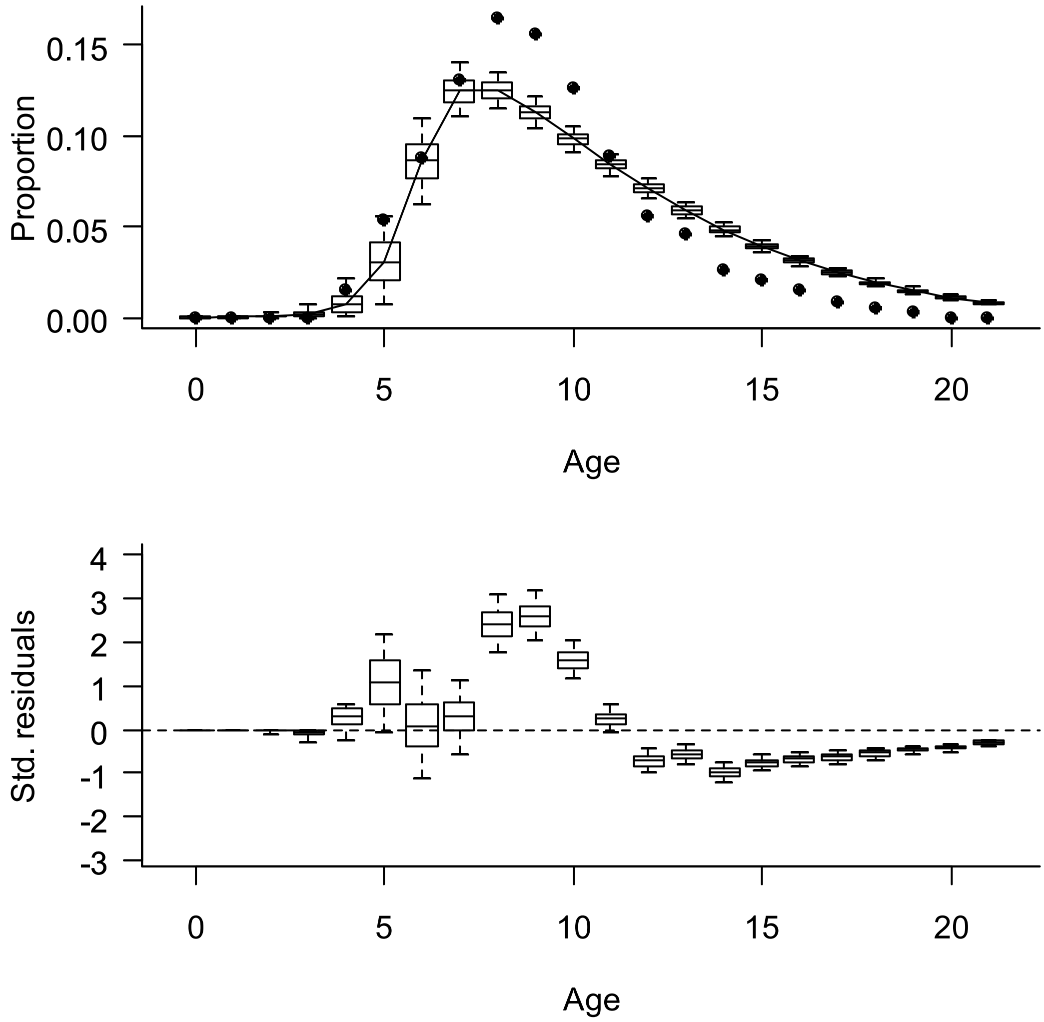

Figure 12: The fits and residuals for the breeding female age data. The box plots summarise posterior distributions: median, 25th and 75th percentiles (box) and the 5th and 95th percentiles (whiskers). Dots on the upper plot are the observed values.

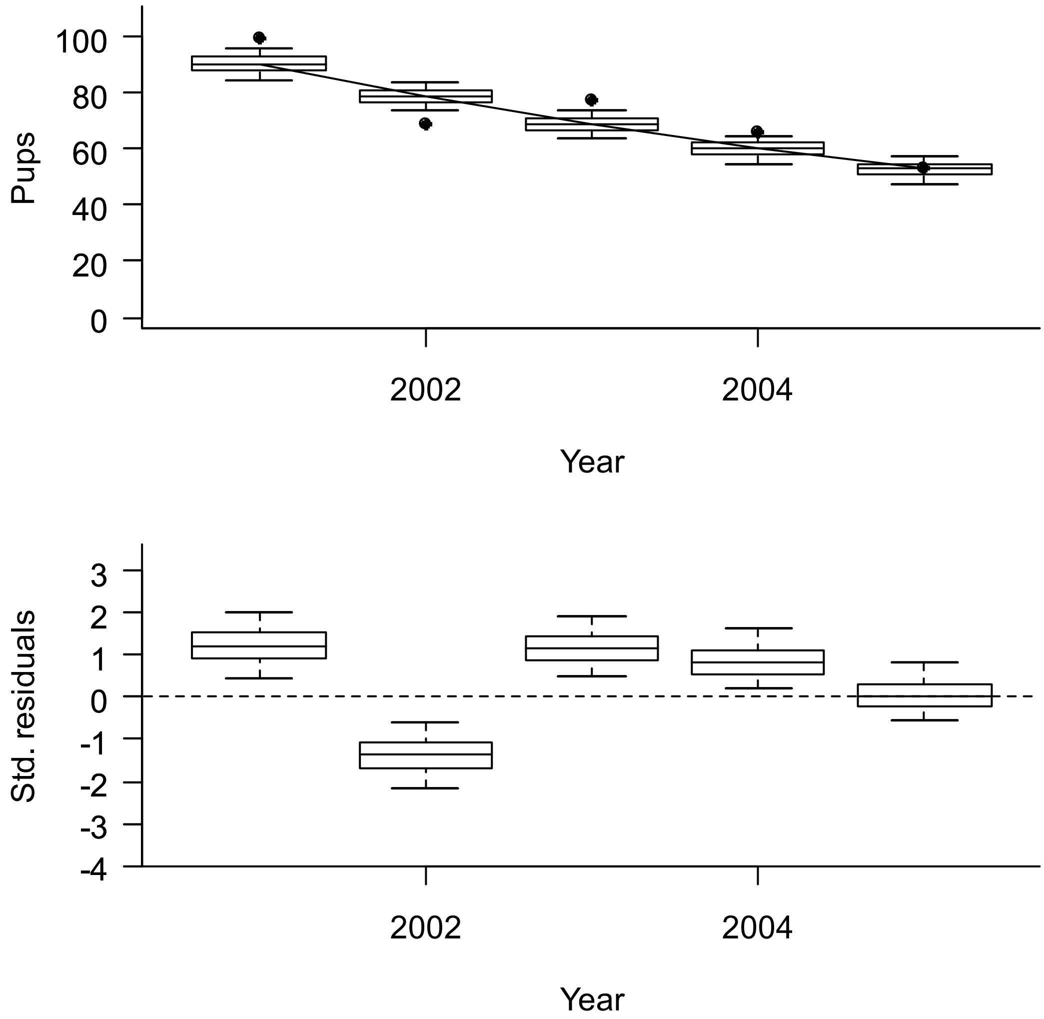

Figure 13: **The fits and residuals for the observed pups from branded females. The box plots summarise posterior distributions: median, 25th and 75th percentiles (box) and the 5th and 95th percentiles (whiskers). Dots on the upper plot are the observed values.**

5. Projection procedures

5.1 Overview

The purpose of fitting the model to data was to produce a simulation model that was highly realistic for use in evaluating different operational management procedures (decision rules) that might be used to manage bycatch.

With the set of 5000 samples of the joint posterior distribution of population parameters, we make sets of forward projections, in which the model is run forward from 2005, in this case for 20 years. The first year affected by a tested rule is 2007, so projections are evaluated for 2007 through 2026. In forward projections, the population parameters used for one run are those in the sample from the joint posterior, except that in every year

- stochastic error is added to survival-at-age;
- stochastic error is added to pup production;
- the catchability coefficient is drawn from a distribution with the same statistical properties as the model estimates for that run;
- fishing effort is drawn from a distribution with the same statistical properties as the observed data and
- observation error is added to the actual pup production to obtain observed pup production.

Specifics of these stochastic processes are described by Breen & Kim (2005). In each set of runs, the bycatch limit is determined by a "harvest control rule", and several of these (described below) are used and compared. Population performance under the different harvest control rules are compared formally with a number of indicators described below.

In the forward projections, the random number sequences are different for each sample of the joint posterior, but they are the same in each set of runs, so that differences between two sets of runs arise only from the harvest control rule and not from the stochastic processes.

5.2 Harvest control rules

In this study three forms of bycatch control rule were used:

- Rule 0 - no fishing
- Rule 3 - a family of rules giving bycatch limits that are linear functions of pup production estimates
- Rule 4 - an alternative adaptive rule discussed by the Breen & Kim (2005), giving bycatch limits that increase as a curved function of pup production estimates.

Rules 0 is used for reference only and generates a set of population trajectories, under no fishing, that are used as the basis for comparing the other rules.

For Rule 3, the annual bycatch limit is calculated as a function of the observed pup production at the Auckland Islands. Log-normally distributed observation error with a c.v. of 0.05 was applied to the model's actual numbers of pups in each year, and pup production estimates were averaged over two years as in recent FRML calculations. The lag described for the real procedure was incorporated: pup production in year y is used, with pup production from year y-1, to develop a bycatch limit for year y+1.

Under Rule 3, each year the model calculates both the no-bycatch limit catch level and the FRML, and sets the actual catch to the lower of these. No further implementation error is simulated.

Before 2004, the FRML was calculated each year from Wade's (1998) formula, using the 20th percentile of estimated population size, which in turn was obtained from pup production estimates using the model of Gales & Fletcher (1996). In this procedure, pup production estimates from both Campbell Island and the Auckland Islands were used.

In the Breen et al. (2003) modelling, a simpler approach was required to evaluate this FRML rule. That modelling addressed only the Auckland Islands population. It would be possible, but was not practical, for the modelling to simulate the operation of the Gales-Fletcher model to obtain the 20th percentile of estimated population size. The approach taken by Breen et al. (2003) was to estimate the relation between Auckland Islands pup production (ignoring Campbell Island estimates) and the resulting FRML from the years in which the procedure had been used. They used that relation and the other constants of the Wade (1998) formula as applied in New Zealand to obtain a simple formula that mimicked the procedure:

$$C_y^{FRML[310]} = 0.02577\left(\frac{N_{0,y-1}^{proj} + N_{0,y-2}^{proj}}{2}\right)$$

where $N_{0,y-1}^{proj}$ is the pup production estimate in year y-1 after observation error has been applied.

This gives approximately the same FRML, for a given pup production at the Auckland Islands only, as would have been obtained from the Wade procedure using both the Campbell Islands and Auckland Islands pup production estimates. The relation is approximate, but the model gives exact performance results for the rule as specified.

Rule 3 as used here is a general family of rules, with the form

$$C_y^{FRML[3n]} = n0.02577\left(\frac{N_{0,y-1}^{proj} + N_{0,y-2}^{proj}}{2}\right)$$

where n is a multiplier. When n = 1, the rule is Rule 310 (the Wade FRML rule); n = 0.5 gives Rule 305; n = 2 gives Rule 320, and so on. Setting n = 0 gives Rule 0. Setting n to some arbitrarily high value, probably near 12, gives Rule 1, no bycatch limit but with fishing effort constrained to recent levels. This general family of rules is useful for exploring the trades-off in fishery and population indicators.

The "Cusp rule" was the product of explorations in early 2004. It is defined as that rule in the Rule 3 family that just meets the population indicators agreed by the AEWG in 2003, based on 100-year runs. The Cusp rule was discovered by varying n

with a simple adaptive algorithm, and turned out to have n = 9.23. That work was not repeated in this study: we use Rule 392 as the "Cusp rule" based on the earlier work.

Rule 4 is an adaptive rule (Breen and Kim 2005) in which the permitted exploitation rate increases as pup production estimates increase. This rule is described by

$$C_y^{FRML[4]} = 102\left(\frac{N_{0,y-1}^{proj} + N_{0,y-2}^{proj}}{2\bar{N}}\right)^2 + 34\left(\frac{N_{0,y-1}^{proj} + N_{0,y-2}^{proj}}{2\bar{N}}\right)^4$$

where $N_{o,y}^{proj}$ is the estimated number of births in year y after observation error is simulated and $\bar{N}$ is the mean pup production estimated from 1999 through 2003. As for Rule 3, this rule was applied with a one-year lag, and each year the fishing sub-model applied the lower of the Rule 4 limit and the no-limit bycatch.

5.3 Indicators

In their raw form, the results of projections were, for each of 20 years in each of 5000 runs (100,000 years), numbers of sea lions-at-age and numbers caught by the fishery. The AEWG in 2003 discussed what criteria should be used to assess the results and developed a list of several population and fishery measurements that were retained for this study.

As discussed in the Introduction, the study evaluated population performance with two main criteria. The first criterion was based on this indicator: for each year of each run in a set (thus 100,000 years in 5000 20-year runs), whether the population at the end of the run was greater than 90% of the population size that would occur in the absence of fishing. The minimum DoC criterion is that this must be true 90% of the time. Formally stated, this is

$$P\left(N_t^{Rulen} \geq 0.9N_t^{Rule0}\right) > 0.90$$

where N_t^{Rulen} is the mature population in year t under Rule n and N_t^{Rule0} is the mature population in year t under no fishing.

The second criterion was based on a similar indicator that included the possibility that the population might be at 90% of K, in which case its position relative to the unfished population could be considered irrelevant. This is the same criterion used by Breen & Kim (2005). Formally stated, this is

$$P\left[\left(N_t^{Rulen} \geq 0.9N_t^{Rule0}\right) OR \left(N_t^{Rulen} \geq 0.9K\right)\right] > 0.90$$

In the 2003 study the AEWG also agreed to examine:

- "*N20/K*": the median of mature numbers at the end of each run as a fraction of K estimated for that run
- *"effortlost"*: the median (of the 5000 runs) of the mean (over the 20 years in each run) of tows lost through the operation of the bycatch control rule during the run, as a measure of cost to the fishing industry,
- "*maxcatch*": the median of maximum annual bycatch in each run,
- "*meancatch*": the median of mean annual bycatch in each run,

- "*Umax*": the median of maximum annual exploitation rate in each run,
- "*nadir*": the median of the population's lowest number of mature sea lions from each run,
- "*nadir*/*K*": the median of population nadirs in each run expressed as a percentage of *K,*
- "*N20*/*K*": the median of the numbers in the final year of each run expressed as a percentage of *K*,
- *"%mat*": the median of the mean percentage of mature animals,
- "*pupmin*": the median of minimum pup production estimates,
- "*pupmax*": the median of maximum pup production estimates, and
- "*puprange*": the median of pup production estimate range.

In addition, to understand the results we chose to examine:

- "*Umean*": the median of mean annual exploitation rate in each run, and
- *"%closed*": the median percentage of seasons closed early through the operation of the bycatch control rule during the run.

6. Projection results

6.1 Two main criteria

Both the main criteria are satisfied under all rules (Table 12): the percentage of years in which the criteria are satisfied is well above 90%, having a minimum of 97% for the Cusp rule. Rules 305 and 310 satisfied both criteria in all years; Rules 320 and 4 came very close.

Table 12: Summary of indicators from each rule. Probabilities (lines 3 and 4) are shown as percentages.

How measured	Indicator	No fishing **Rule 0**	**305**	Wade rule **310**	**320**	**330**	**340**	**350**	**380**	**390**	Rule 392 **Cusp 392**	Adaptive rule **4**
sum over 100,000 years	***(N>90%N0)***		100000	100000	99983	99635	99041	98512	97489	97307	97278	99997
sum over 100,000 years	***[(N>90%N0)OR(N>90%K)]***		100000	100000	99983	99670	99183	98750	97980	97853	97829	99997
percentages	***P(N>90%N0)***	100%	100%	100%	100%	100%	99%	99%	97%	97%	97%	100%
percentages	***P[(N>90%N0)OR(N>90%K)]***	100%	100%	100%	100%	100%	99%	99%	98%	98%	98%	100%
median of posterior	***Nmat0***	6962	6962	6962	6962	6962	6962	6962	6962	6962	6962	6962
median of posterior	***Nmat20***	7174	7068	6991	6894	6850	6830	6819	6806	6803	6803	6901
median of posterior	***N20/K***	97.3%	95.8%	94.7%	93.4%	92.8%	92.5%	92.3%	92.1%	92.1%	92.1%	93.5%
median of posterior	***effortlost***	2906	1655	927	322	123	42	0	0	0	0	368
median of posterior	***maxcatch***	0	36	72	141	205	264	309	322	322	322	143
median of posterior	***meancatch***	0	31	53	77	88	93	96	98	99	99	75
median of posterior	***Umax***	0.00%	0.32%	0.63%	1.24%	1.79%	2.27%	2.62%	2.82%	2.82%	2.82%	1.25%
median of posterior	***meanUproj***	0.00%	0.26%	0.45%	0.67%	0.77%	0.81%	0.84%	0.86%	0.86%	0.86%	0.65%
median of posterior	***nadir***	6696	6633	6579	6503	6465	6446	6434	6416	6414	6414	6508
median of posterior	***nadir/K***	91.6%	90.6%	89.9%	88.8%	88.2%	87.9%	87.7%	87.5%	87.5%	87.5%	88.8%
median of posterior	***%mat***	39.3%	38.9%	38.7%	38.4%	38.3%	38.2%	38.2%	38.1%	38.1%	38.1%	38.4%
% over 100,000 years	***closure***	100.0%	79.7%	54.6%	24.2%	11.4%	5.8%	3.1%	0.7%	0.4%	0.4%	26.9%
median of posterior	***pupmin***	2242	2249	2253	2255	2255	2255	2255	2255	2255	2255	2255
median of posterior	***pupmax***	2956	2958	2960	2962	2962	2963	2963	2963	2963	2963	2961
median of posterior	***puprange***	727	720	716	713	712	711	711	710	710	710	713

6.2 Other indicators

The results for all indicators from all Rules are summarised in Table 12.

Posteriors for *N20/K* are shown in Figure 14. For the Rule 3 family, this has an asymptote at 92% (Table 12). For this and many other indicators, Rule 4 is similar in performance to Rule 320.

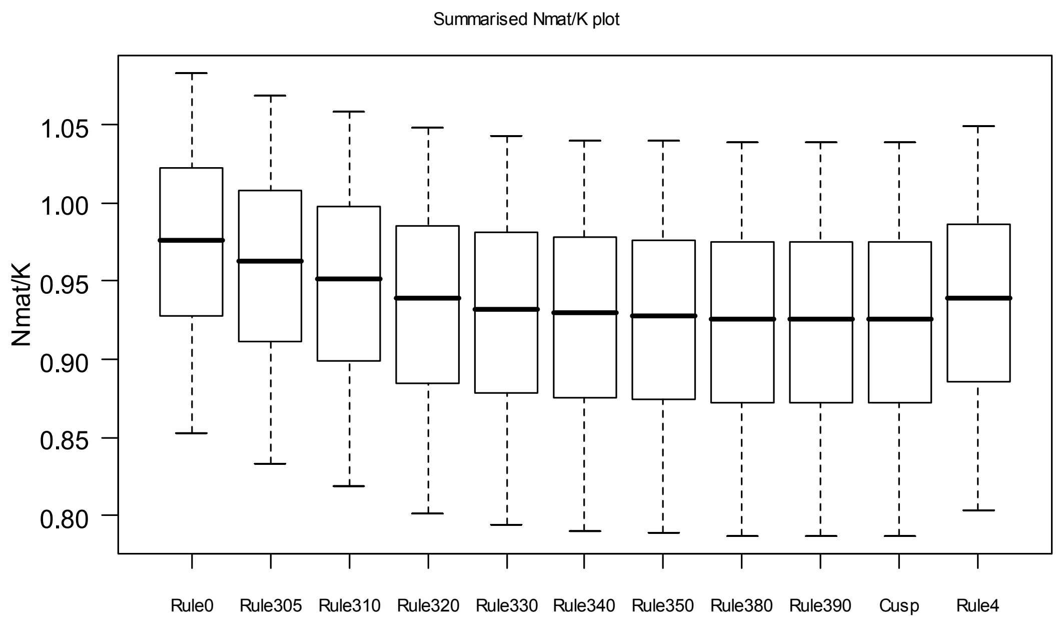

Figure 14: Summary of the posterior distributions of the indicator *N20/K* for different bycatch control rules. For each rule the horizontal line is the median of the posterior distribution, the box encloses the 25th to 75th quantiles and the whiskers show the 5th to 95th quantiles.

Median *effortlost*, the fishing effort lost through operation of the rules, is 927 tows under Rule 310 (Figure 15); this is roughly one-third of the mean attempted effort. This rises to a median of 1655 tows (more than half the effort) under Rule 305, to 322 tows under Rule 320 and to less than 2% of attempted effort after Rule 340. Rule 4 is similar to Rule 320, with a median loss of 368 tows (12%). Under Rule 0 all effort is lost.

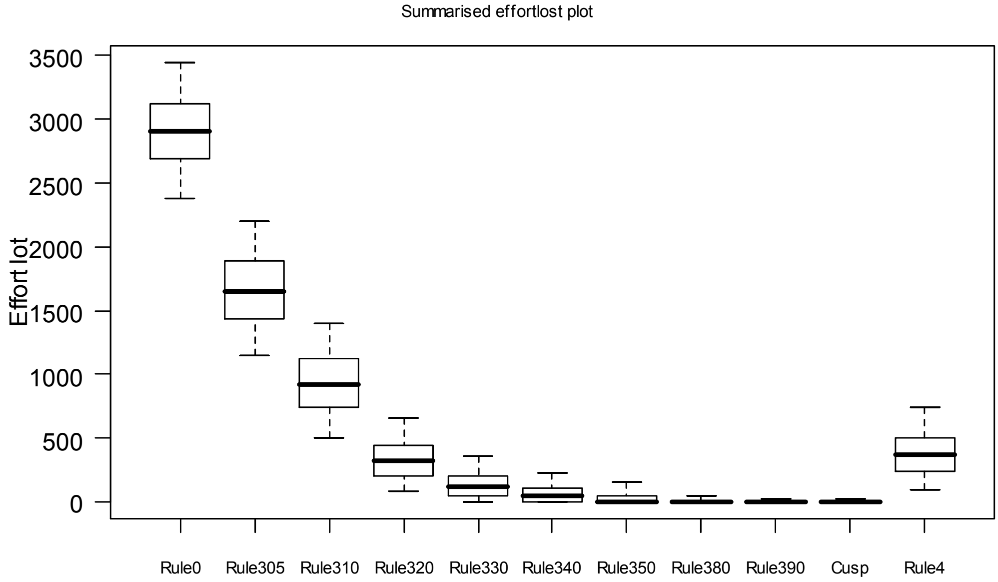

Figure 15: **Summary of the posterior distributions of the indicator *effortlost* for different bycatch control rules.**

The *maxcatch* is roughly linear with the Rule 3 multiplier until Rule 350 (Figure 16), then shows asymptotic behaviour. The median reaches 322 animals at about Rule 380. Rule 4 is roughly similar to Rule 320 at about 143 animals.

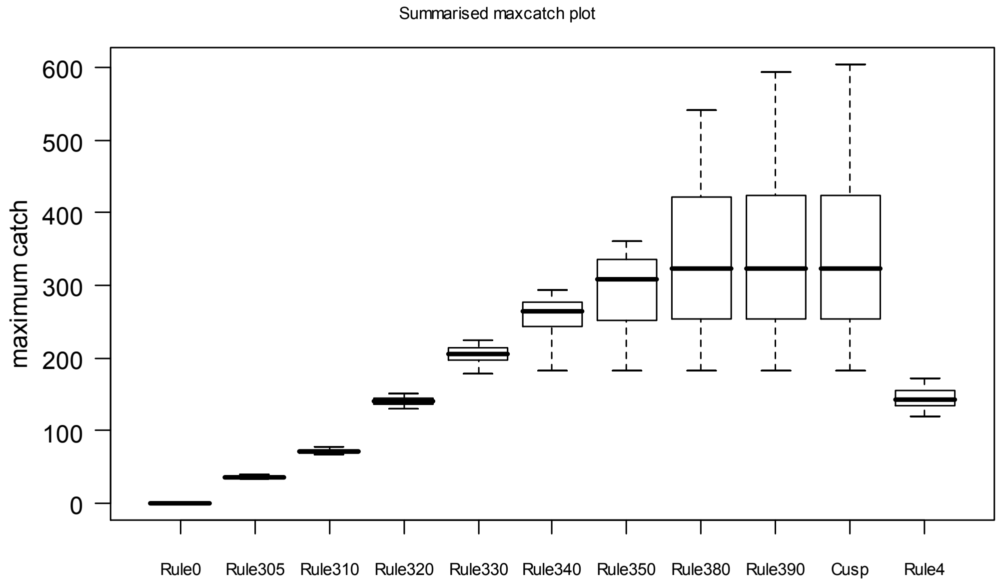

Figure 16: **Summary of the posterior distributions of the indicator *maxcatch* for different bycatch control rules.**

Mean bycatch (Figure 17) is 53 under Rule 310 and 77 under Rule 320. This shows strongly asymptotic behaviour, becoming nearly flat after Rule 340, with an asymptote at 99. Rule 4 is nearly the same as Rule 320.

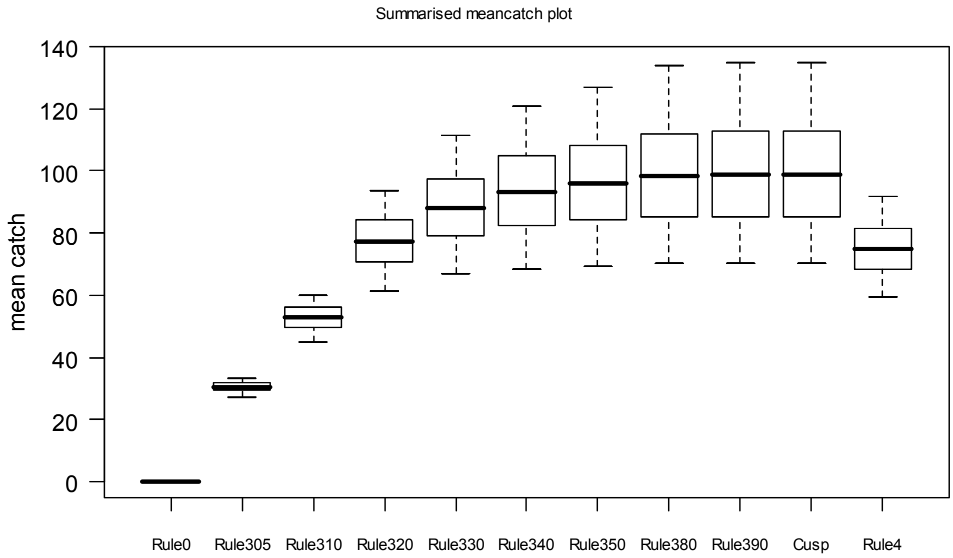

Figure 17: Summary of the posterior distributions of the indicator *meancatch* for different bycatch control rules.

The pattern of maximum exploitation rate, *Umax*, is similar to the maximum bycatch indicator (Figure 18), ranging from 0.32% to 2.82 %, highest for the Rules 380 to 392 (Cusp rule).

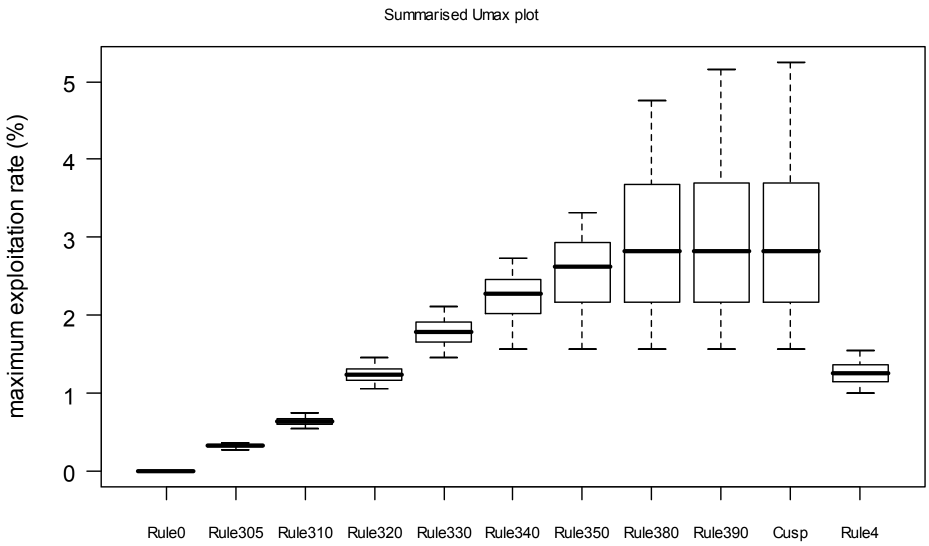

Figure 18: Summary of the posterior distributions of the indicator *Umax* rate for different bycatch control rules.

The median of average exploitation rate (Figure 19) is also highest for Rules 380 to 392 (Cusp rule), reaching 0.86%. This is based on vulnerable animals, which includes many immature sea lions. Rules 320 and 4 are similar in *Umean* with medians near 0.65%.

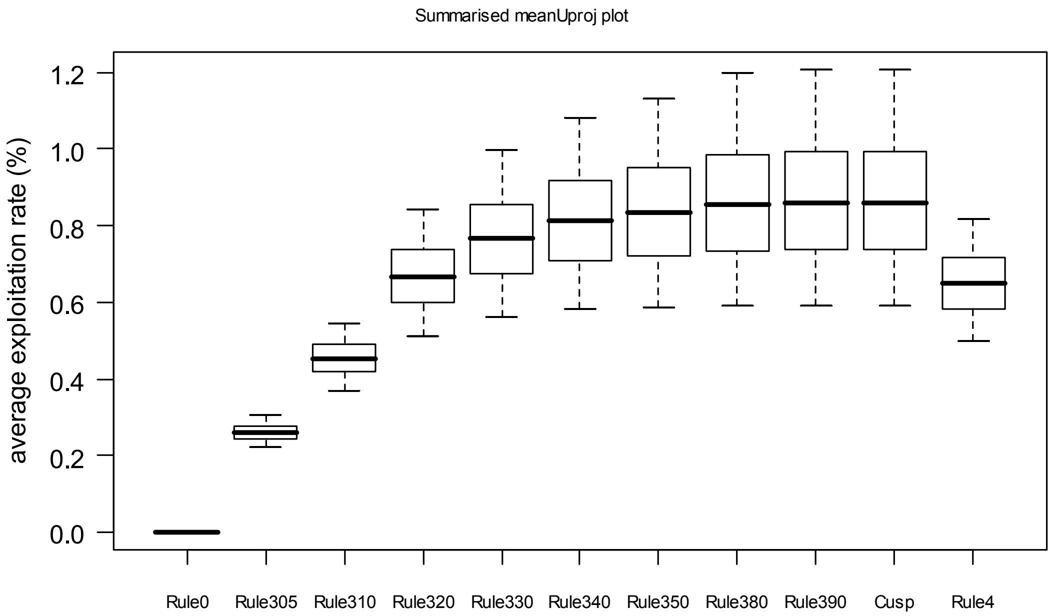

Figure 19: Summary of the posterior distributions of the indicator *Umean* for different bycatch control rules.

The range of *nadir/K* (Figure 20) is narrow: from 87.5% under the Cusp rule to 91.6% under Rule 0. Rules 320 and 4 are the same near 90%. This indicator also shows strongly asymptotic behaviour.

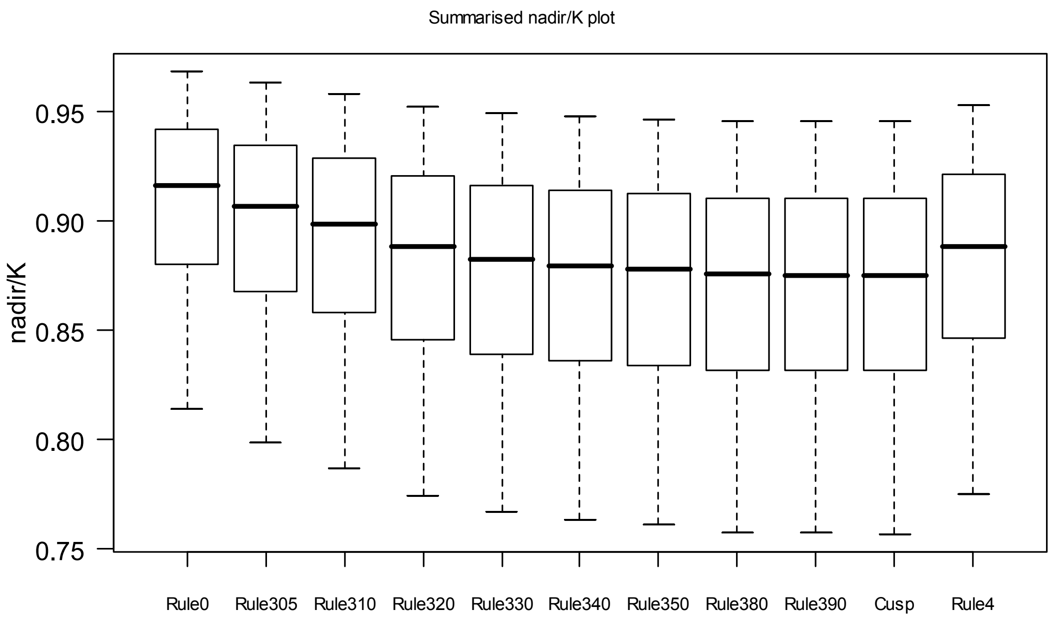

Figure 20: Summary of the posterior distributions of the indicator *nadir/K* for different bycatch control rules.

The mean percentage of mature animals, *%mat* (Figure 21) shows a narrow range between 38% and 39% between Rule 0 and the Cusp rule. There is almost no contrast in this indicator among the other rules.

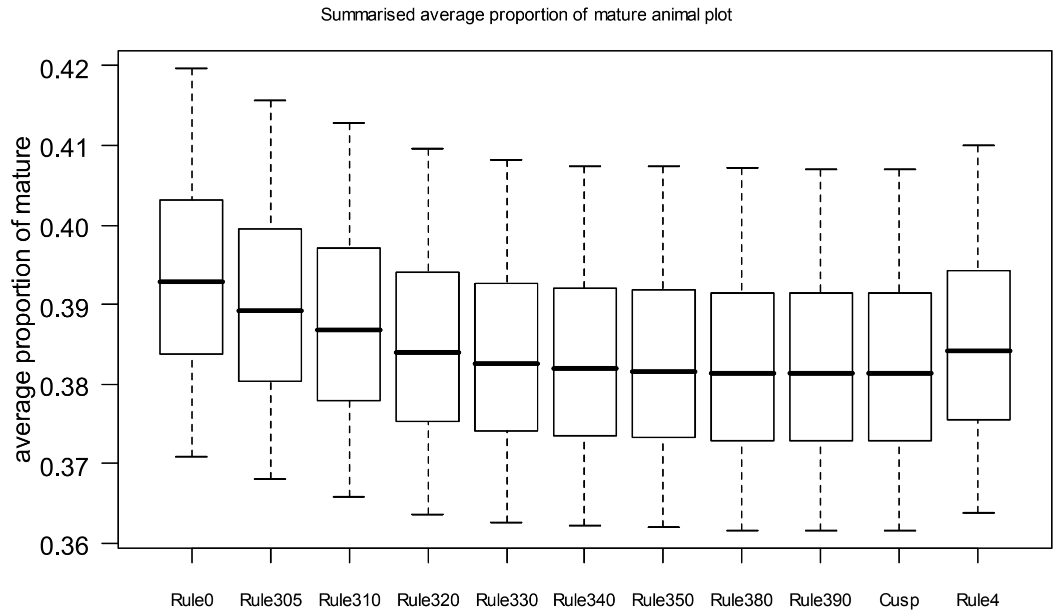

Figure 21: Summary of the posterior distributions of the indicator *%mat* under different bycatch control rules.

The *pupmin*, *pupmax* and *puprange* indicators (Figure 22, Figure 23, and Figure 24) show almost no contrast. The range is about 710 pups under all rules.

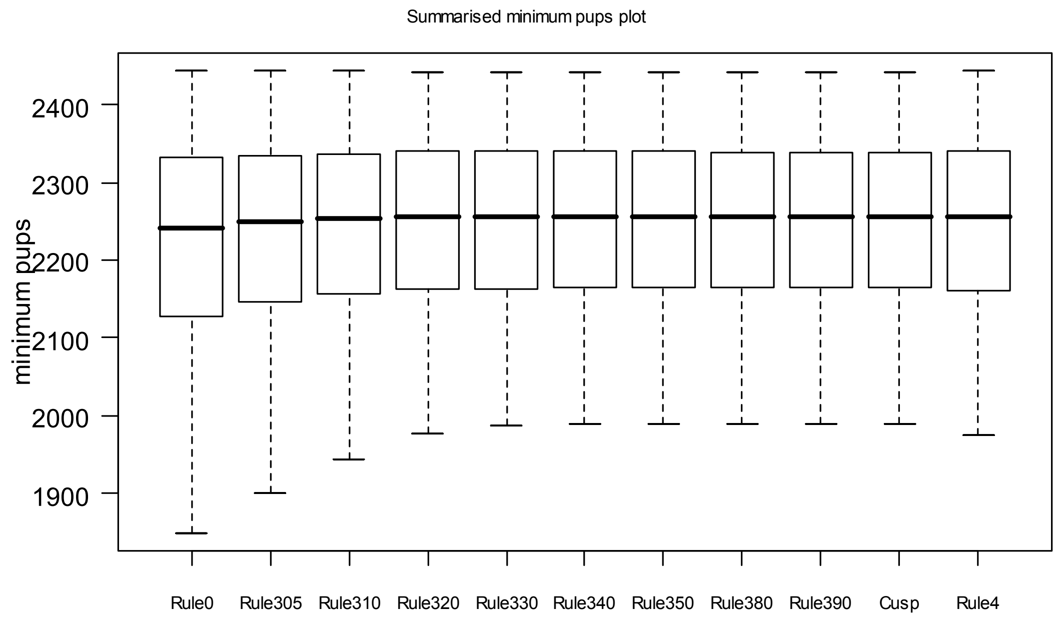

Figure 22: Summary of the posterior distributions of the indicator *pupmin* under different bycatch control rules.

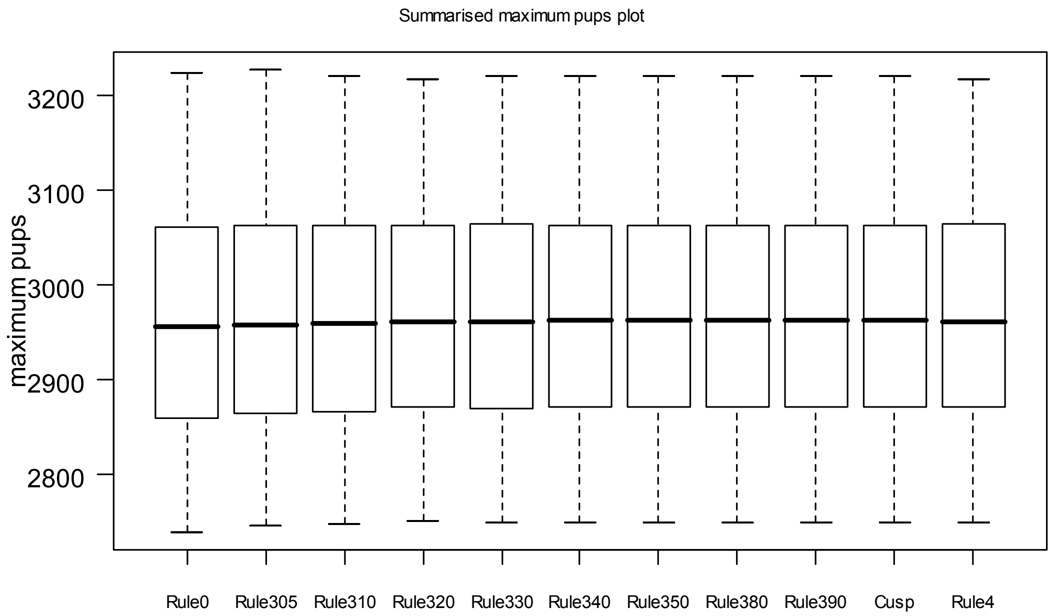

Figure 23: Summary of the posterior distributions of the indicator *pupmax* under different bycatch control rules.

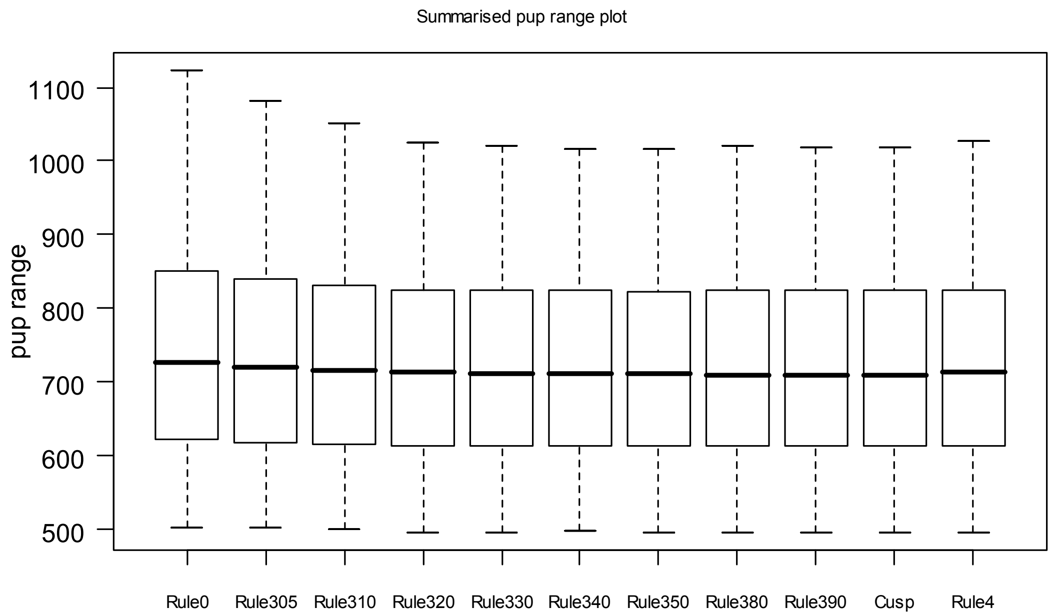

Figure 24: Summary of the posterior distributions of the indicator *puprange* under different bycatch control rules.

The index of seasons closed, *%closed* (Figure 25), ranges from 79.7% under Rule 305 to 24.2% at Rule 320, then decreases very quickly to less than 3.5% at Rule 350. It is near 55% under Rule 310, and Rule 4 is again similar to Rule 320 near 25%.

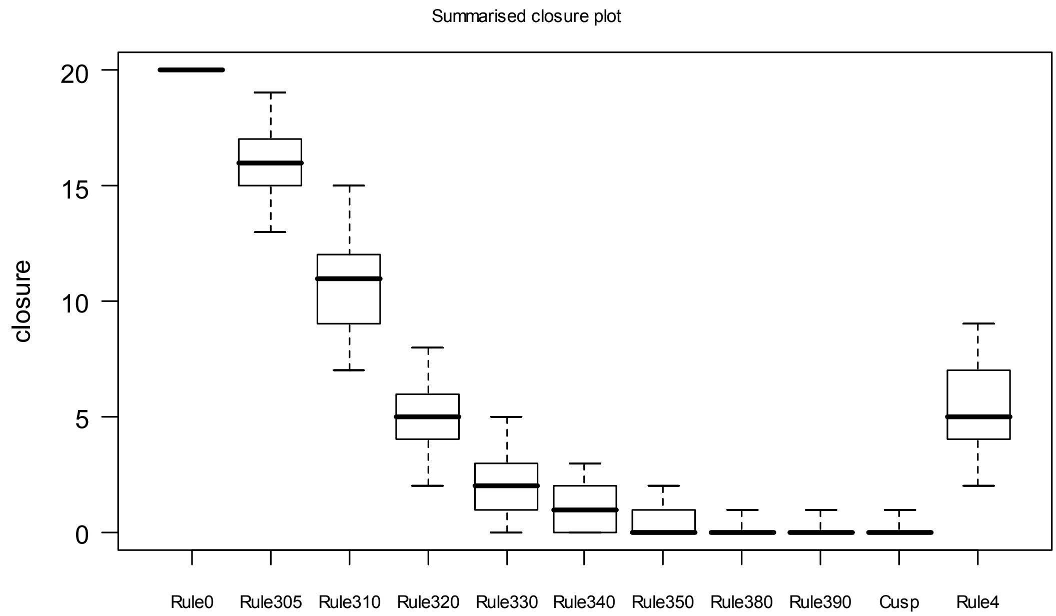

Figure 25: Summary of the posterior distributions of the indicator *%closed* for different bycatch control rules.

6.3 Population consequences

For a randomly chosen run (number 174), the mature population trajectories under Rules 0, 310 and 4 are shown in Figure 26. Differences caused by the different rules are small; the main effect on the population appears to be the stochastic variation in survival and pupping rate.

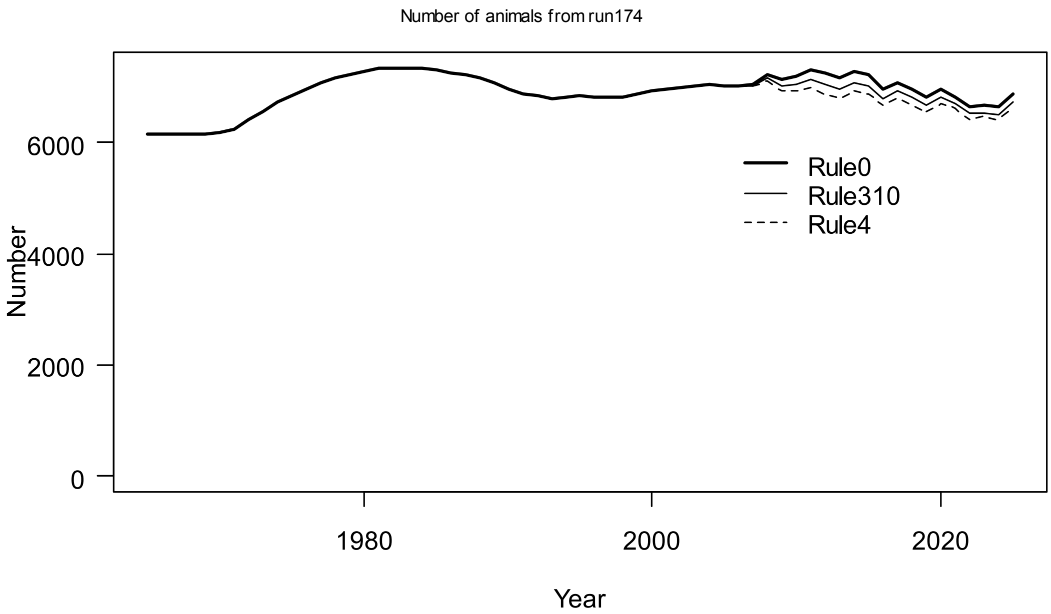

Figure 26: Typical population trajectories (mature numbers) under the four rules indicated from the base case projections.

The distributions of *nadir/K* and *Nmat/K* are compared among these rules in Figure 27 and Figure 28. The distributions shift to the left, towards smaller nadirs or smaller final numbers, as fishing intensity increases. Thus the posteriors for Rule 0 are the right-most and for Rule 4 are the left-most. Distributions from Rule 310 and Rule 4 are similar, with Rule 310 showing higher median values than Rule 4 (Table 11). In both figures, the differences in modes among the four rules are not great.

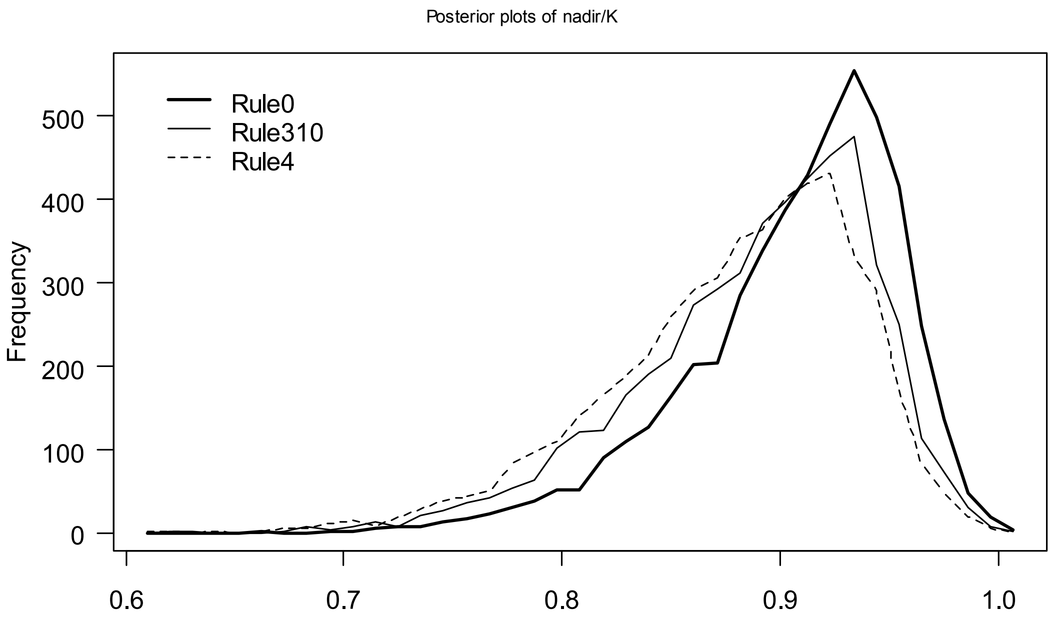

Figure 27: The posterior distributions of the indicators *nadir/K* for each decision rule in the base case projections.

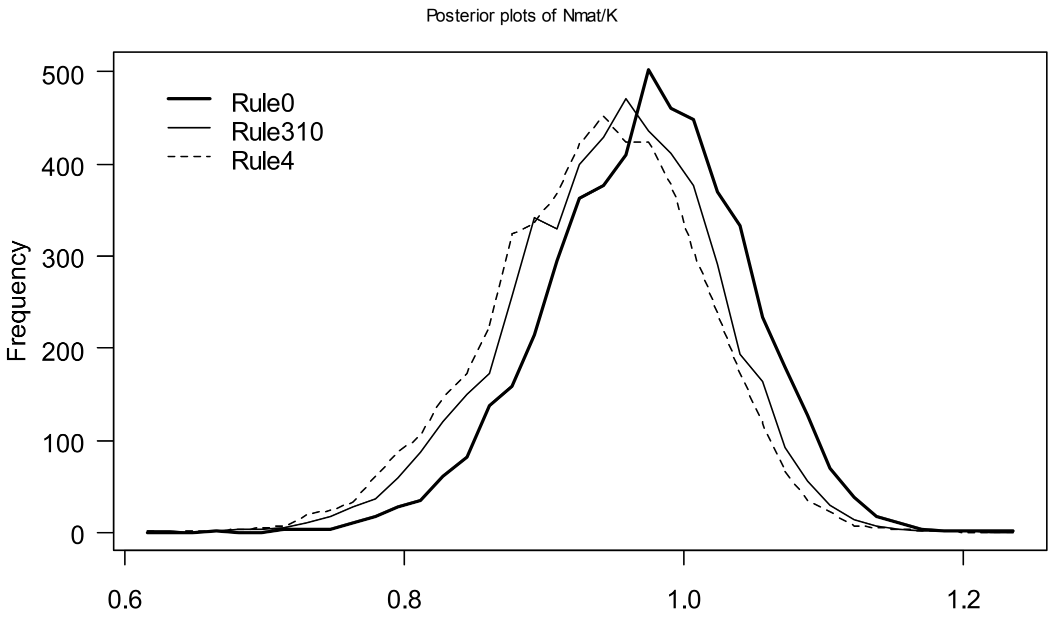

Figure 28: The posterior distributions of the indicators *Nmat/K* for each decision rule in the base case projections.

7. Discussion

7.1 Effect of new data

The difference between this study and that of Breen & Kim (2005) lies in the new and revised data sets and in the slightly modified data weights. The effect of this change on estimated and derived parameters is relatively small (Table 11). Most estimated parameters were near their 2003 estimates. Derived parameters were also similar in their medians. Where parameters such as *z* were poorly estimated in 2003, they were also poorly estimated in this study.

Results based on rule performance are more difficult to compare because the 2003 study used only one 20-year indicator, which was $\left[\left(N_t^{Rulen} \geq 0.9 N_t^{Rule0}\right) OR \left(N_t^{Rulen} \geq 0.9K\right)\right]$. Table 13 shows this indicator compared across the same range of Rule 3 variants from both studies. The 2005 results are slightly different - more pessimistic - than the 2003 results, probably as a result of the recent decrease in pup production. But across the range of the rules examined, the differences are trivial.

Table 13: For each of the rules shown in each of the two studies, the number of years in which the indicator [(N>N0) OR (N>90%*K*)] was true.

Rule	300	305	310	320	330	340	350	380	390	Cusp
2003	100000	100000	100000	99989	99810	99409	99006	98264	98131	98115
2005	100000	100000	100000	99983	99670	99183	98750	97980	97853	97829

7.2 State of the population

Results from this study support the conclusions drawn by Breen & Kim's (2005) study made in 2003. These include:

- The current population is near its carrying capacity, *K*. This conclusion is driven by the relatively stable pup production estimates.
- The model and data sets provide some restriction on the upper limit of λ : even with a prior mean of 0.08, the posterior distribution does not extend past 0.05.
- Productivity estimates are uncertain because there is little contrast in the pup production estimates.
- Bycatch in the squid fishery has only a small effect on the Auckland Islands population over a 20-year time span. Under the Cusp rule, which sets bycatch limits so high they are rarely met, but assuming recent levels of mean effort, the depression of population indicators by fishing is small (Table 11).
- All rules satisfied both the main criteria examined at the 90% threshold, and would have done so with a threshold of 95% certainty. Longer runs would decrease the probabilities of meeting these thresholds.

7.3 Robustness

Model results will be influenced to some extent by sampling or estimation errors in the input data. This uncertainty is reflected to some extent in the Bayesian results for the data to which the model was fit, but is not represented for the bycatch estimates, which the model uses as if they were perfect information. The actual numbers of sea lions caught are imperfectly known except for one year in which the observer coverage was 100%, at least in SQU 6T. In other years the bycatch was estimated from partial observer coverage, and in all years some sea lions may have been caught outside SQU 6T or by fisheries other than the arrow squid fishery.

The effect of systematic errors in bycatch estimates can be predicted. If the bycatch vector is under-estimated, then the model under-estimates the population's density-dependent response to fishing, and the effect of fishing on the population is over-estimated in projections; conversely if bycatch has been over-estimated. Randomly distributed errors with no bias are likely to have little effect on model results.

The conclusions presented here are sensitive to the prior used for λ , which is discussed in detail by Breen & Kim (2005). Without the prior, low values for λ were obtained in that study, and the AEWG thought those values unrealistic. If projections were made from those, the effects of bycatch would be greater and sustainable exploitation rate would be lower.

The performance of Rules 3 and 4 would change to some extent if different values for mean effort were used. Under both rules, the permitted bycatch is often not taken: with higher fishing effort the limit would be reached more often, bycatch would be higher and population indicators would be lower.

The present study modelled the fishery as if it made no effort to release sea lions alive. In practice, the SLEDs probably reduce sea lion mortality, although the success rate is unknown and controversial. However, the conclusions of the study should be robust with respect to limits on the number of dead sea lions, no matter how they died.

8. Acknowledgements

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9. References

Baird, S.J. 2005. Incidental capture of *Phocarctos hookeri* (New Zealand sea lions) in New Zealand commercial fisheries, 2002-03. *New Zealand Fisheries Assessment Report 2005/9*. 13 pp.

Breen, P.A., R. Hilborn, M.N. Maunder & S.W. Kim. 2003. Effects of alternative control rules on the conflict between a fishery and a threatened sea lion (*Phocarctos hookeri*). *Canadian Journal of Fisheries and Aquatic Sciences 60:*527–541.

Breen, P.A. & S.W. Kim. 2005. Exploring alternative management procedures for controlling bycatch of Hooker's sea lions in the SQU 6T squid fishery. Final Research Report For Ministry of Fisheries Project M0F2002/03L, Objective 3. Third revision 19 May 2005. 88 pp.

Breen, P.A., S.W. Kim & P.J. Starr. 2005. Estimating sea lion bycatch in the SQU 6T and SQU 1T fisheries. Report prepared for the Squid Fishery Management Company. *NIWA Client Report: WLG2005-37*. 23 pp.

Childerhouse, S., N. Gibbs, G. McAlister, S. McConkey, H. McConnell, N. McNally & D. Sutherland. 2005. Distribution, abundance and growth of New Zealand sea lion *Phocarctos hookeri* pups on Campbell Island. *New Zealand Journal of Marine and Freshwater Research 39*: 889–898.

Dickie, G. 1999. Population dynamics of New Zealand fur seals (*Arctocephalus forsteri*) and New Zealand sea lions (*Phocarctos hookeri*). MSc Thesis, University of Otago, Dunedin, New Zealand.

Duignan, P.J. & G. W. Jones. submitted a . Autopsy of pinnipeds incidentally caught in commercial fisheries, 2002/2003. Department of Conservation Contract CSP 01/3026, Final Report 28-11-2003

Duignan, P.J. & G. W. Jones. submitted b. Autopsy of pinnipeds incidentally caught in commercial fisheries, 2003/2004. Department of Conservation Contract CSP 01/3026, Final Report 28-11-2004.

Gales, N.J. & D.J. Fletcher. 1996. Abundance, distribution and status of the New Zealand sea lion *Phocarctos hookeri*. New Zealand Department of Conservation, PO Box 10-420, Wellington, New Zealand.

Gibbs, N.J., P.J. Duignan & W.J. Gareth. 2002. Autopsy of pinnipeds incidentally caught in commercial fisheries, 2001/2001. Final Report 11-11-2002 to the Department of Conservation. Institute of Veterinary, Animal and Biomedical Sciences, Massey University, Palmerston North, New Zealand.

Gibbs, N.J., P.J. Duignan & W.J. Gareth. 2003. Autopsy of pinnipeds incidentally caught in fishing operations 1997/98, 1999/2000, and 2000/01. Unpublished report to the Department of Conservation, PO Box 10-420, Wellington, New Zealand.

McConkey, S.D., S. Heinrich, C. Lalas, H. McConnell & N. McNally. 2002. Pattern of immigration of New Zealand sea lions *Phocarctos hookeri* to Otago, New Zealand: Implications for management. *Australian Mammalogy 24*: 107–116.

Maunder, M.N., P.J. Starr & R. Hilborn. 2000. A Bayesian analysis to estimate loss in squid catch due to implementation of a sea lion population management plan. *Marine Mammal Science 16(2)*: 413-426.

MFish. 2005. Proposed operational plan to manage the incidental capture of New Zealand sea lions in the squid (SQU) 6T trawl fishery for the 2005-06 fishing year. Initial Position Paper. Unpublished Report, Wellington. 31 pp.

Smith, M.H. & S.J. Baird. 2005. Factors that may influence the incidental mortality of New Zealand sea lions (*Phocarctos hookeri*) in the squid (*Nototodarus* spp.) trawl fishery in SQU 6T. *New Zealand Fisheries Assessment Report 2005/20*. 35 pp.

Sullivan, K.J., P.M. Mace, N.W.McL. Smith, M.H. Griffiths, P.R. Todd, M.E. Livingston, S.J. Harley, J.M. Key & A.M. Connell. 2005. Report from the Fishery Assessment Plenary, May 2005: stock assessments and yield estimates. Ministry of Fisheries, Wellington.

Wade, P. 1998. Calculating limits to the allowable human caused mortality of cetaceans and pinnipeds. *Marine Mammal Science 14*: 1-37.

Wilkinson, I., J. Burgess & M. Cawthorn. 2003. New Zealand sea lions and squid: Managing fisheries impacts on a threatened marine mammal. pp. 192-207 *In* N. Gales, M. Hindell & R. Kirkwood (Eds.) Marine mammals: Fisheries, tourism and management issues. CSIRO Publishing, Collingwood, Victoria.