Ministry for Primary Industries
Manatū Ahu Matua

The 2014 demographic assessment of the Snares Islands population of Southern Buller's albatross (*Diomedea bulleri bulleri*)

New Zealand Government

Growing and Protecting New Zealand

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EXECUTIVE SUMMARY

Southern Buller's Albatross is estimated in the level-2 risk assessment, L2RA, to be at very high risk of direct fishing-related mortality within the EEZ. Fully-quantitative modelling of Southern Buller's Albatross population on the Snares Island using data up to 2007 suggested that the population had increased substantially from the 1950s, but that population growth may have slowed and potentially reversed. This report describes an updated demographic assessment of the Snares Islands population of Southern Buller's albatross (SBA), including new mark-recapture data from 2008–2014 and a new census estimate from 2014.

A number of models with varying levels of complexity were considered, which differed in terms of the observational datasets being used. Model 3, which incorporated mark-recapture and census data, was considered to have provided the best assessment of the SBA population. This model estimated annual varying adult survival rates and constant breeding probabilities. This model fitted well to both the census and mark-recapture observations, and estimated no increase in the size of the breeding population between 2002 and 2014. Adult survival has declined since 1990 and the 2008–2014 estimate (0.924) was marginally lower than the 2004–2007 period (0.932).

The estimates of breeding rate (0.83 and 0.57 for breeders and non-breeders, respectively) and probability of breeding success (0.73) are similar to those of a previous assessment (0.84, 0.58 and 0.92, respectively; Francis et. al. (2008)). There was considerable variability in some of the key demographic rates (e.g. breeding rate), and it is not known if this was due to noise in the data, model structure or natural variation in demographic processes. The ability to quantify these variabilities was hindered by a large reduction in resighting effort since 2008. Also since 2008, there have been changes in the monitoring of breeding status and the cessation of monitoring of breeding success, precluding an assessment of temporal variation in reproductive rates. Also, the numbers of non-breeders may have been overstated from the mark-recapture field study during this period. These are likely to have biased the estimates of some of the demographic rates.

A 10-year population projection (2015–2024) was carried out using model 3 with demographic rates (e.g. survival, breeding probabilities, etc.) for future years assumed to be the same as the most recent estimates. The projection suggested that at the current demographic rates, the population is likely to increase by 5.7% in the next 10 years.

Model 4 incorporated fishery bycatch estimates of SBA from the trawl and longline fisheries between 1999 and 2012 and assumed that this bycatch was from the Snares population. A sensitivity model (4.1) also considered the effects of cryptic mortalities from these fisheries (estimates of bycatch were multiplied by the cryptic mortalities multiplier). These models resulted in the same estimates of population size up to the current year. Some major limitation of these models include that the uncertainties of the bycatch estimates are not accommodated; these captures apply to the whole EEZ and cannot be separated between Snares and Solander; and that there are no estimates prior to 1998–99.

1. INTRODUCTION

This work addresses Ministry for Primary Industries (MPI) project PRO2013-17: "*Repeat quantitative modelling of Southern Buller's Albatross*" and it fulfils objective (1) "*To update the fully quantitative population model of Southern Buller's Albatross to assess population trend and key demographic rates for this population*" and (2) "*To use the model to predict future trends assuming recent average demographic rates in this species*".

Southern Buller's albatross (*Thalassarche bulleri bulleri*) is endemic to New Zealand and breeds only on The Snares (48°02′S, 166°36′E) and Solander Islands (46°35′S, 166°54′E) with populations of 8713 and 4912 breeding pairs respectively, as estimated in 2002 (Sagar & Stahl 2005). This species was listed as "Near Threatened" by the IUCN on the basis of the number of breeding sites and population size, although this assessment did not distinguish it from Northern Buller's Albatross *T. bulleri plate*i (IUCN 2012).

Most adult southern Buller's albatross (hereafter SBA) breed annually, with each pair producing a single egg in January-February and the surviving young fledging in August-September (Sagar & Warham 1998). After breeding, adults migrate to waters off Chile and Peru before returning to the breeding islands for the next breeding season (Stahl & Sagar 2000a). SBA forage in Australasian waters during breeding (Stahl & Sagar 2000a; Stahl & Sagar 2000b), where they have been recorded in the bycatch of trawl and longline fisheries since the late 1980s (Murray et al. 1993; Robertson et al. 2003). Recent data from Chile show that Buller's albatross is one of several bycatch species in the pelagic longline swordfish fishery (Birdlife Global Seabird Programme 2008).

SBA was estimated in the level-2 risk assessment, L2RA, to be at very high risk of direct fishing-related mortality within the EEZ, based on capture data from 2007 to 2011 (Richard & Abraham 2013b). Fully-quantitative modelling of the SBA population on Snares Islands using data up to 2007 suggested that the population had increased substantially from the 1950s, but that population growth may have slowed and potentially reversed (Francis & Sagar 2012). A reduction in adult survival rate was also estimated. Substantial new information is now available, including a population survey in early 2014, and updating the existing model to include the new demographic data will provide better information on the current trend and assist in the interpretation of the L2RA results.

The SBA population model was initially developed as part of a five-year Ministry of Fisheries Project PRO2006-02. One of the main objectives of that project was to assess the population status of selected seabird populations in relation to fisheries risks to population viability (Francis et al. 2007). A generic software package SeaBird was developed which allows flexibility in specifying the population dynamics and parameter estimation. SeaBird was subsequently used in project PRO2006-02 to assess SBA (Francis et al. 2008; Francis & Sagar 2012), black petrel (Francis & Bell 2009), white-capped albatross (Francis 2010; Francis & Bell 2012), and Gibson's albatross (Francis et al. 2011). This report updates quantitative models of SBA reported by Francis et al. (2008) to include new data from the 2014 whole-island census count and the mark-recapture observations from the three study areas from 2008 to 2014.

2. METHODS

2.1 Study population

The model described in this report only concerns the SBA population breeding on Snares Islands/Tini Heke (because it is more intensively studied than the population on the Solander Islands).

2.2 Data

The analysis involved three types of data, (1) mark-recapture data from the study areas, (2) population census data (these are the abundance data for the whole SBA population on Snares Islands), and (3) estimated fishing-related mortality.

2.2.1   Mark recapture data

The geographical coverage of banding and resighting effort for SBA within Snares Islands has varied substantially over the period 1948–2014. To ensure reliable estimates of population parameters, we restricted attention to the three areas in which effort has been most consistent – Mollymawk Bay (MB), Lower Punui Bay (LPB), and Upper Punui Bay (UPB) – hereafter referred to as the study areas. Before 1992, researchers did not distinguish between these last two areas, simply recording Punui Bay (PB).

Banding began in 1948, when 159 breeding adults were banded (Richdale & Warham 1973), mostly in PB. The last recorded sighting of one of these birds was in 1993, when the bird must have been over 50 year old (because the youngest known-age breeder recorded was 6 year old – see below). More birds were banded sporadically during the 1960s and 1970s, including the first banding of chicks in 1972 (although only 130 of the 859 banded chicks were from one of our study areas). Before 1992, breeding status was not usually recorded, so the observations simply recorded the presence or absence of birds (but chicks were distinguished from adults). Data from individuals banded in 1948 were highly variable (Sagar et al. 2000), which may have resulted from the lack of information between banding and first recapture (a 13-year period), heterogeneity in the life of the bands (they were handmade), and/or temporally variable recapture probabilities, with noticeably lower recaptures in 1971 and 1973 (Figure 1).

The first multi-year study started in 1992, and the three study areas have been revisited every year since. Up until 2004, two visits were made each year and the number of total field days ranged from 27 to 44 (Table 1). During the first visit in February/March (at the end of the egg laying period) all breeders within the study areas were recorded, and those not yet banded were banded. In the second visit, in July/August (towards the end of the breeding period), the success of each breeding pair was noted (where 'success' meant that a chick had hatched and was still alive within a few weeks of fledging) and chicks were banded. In most trips, areas immediately adjacent to the study areas (referred to as "non-study areas", up to 300 m from PB and 100 m from MB, depending on time available) were also examined Since 2005, only a 4–9 day visit during early April (the hatching period) was made (see Table 1). During all of these visits as many banded non-breeding birds as possible were also recorded. Because of the reduced effort 2006–2014 compared with 1992–2005 the breeding information of the resighting records is not consistent across these two periods. The one short visit per year 2006–2014 means that: (1) birds that attempted to breed, but failed before the visit would be coded as a non-breeder if recaptured, and so this state is likely to be over-emphasised; (2) states indicating breeding success (i.e. raising a chick that flies from the island) cannot be included.

Overall, more than 2000 chicks and 1500 adults have been banded in the three areas since 1948 (Table 2). In what follows we will distinguish between two types of mark-recapture

data: known-age (for birds banded as chicks) and adult of unknown-age (for those banded as adults).

Since 1992, about 40 adults have been banded each year, although much higher numbers of adults were banded in the first two years as the study areas were established (Figure 2). Over 400 breeders (about 220 in MB, and 100 in each of LPB and UPB) were observed each year before 2005 (Figure 3). Since then, the total number of breeders observed each year has generally been less than 350, an apparent result of the much shortened visit to the area. Adults show very strong fidelity to an area once they start breeding. None of the visits between 1992 and 2015 has found that a breeding bird from the study areas has moved to another area to breed.

A total of 2772 birds were banded as chicks between 1992 and 2004, some chicks were banded in 2011–2014, but these will not be resighted for a number of years, therefore they were not included in this analysis. Between 1992 and 2004, 1997 birds were banded in the study areas (about two thirds in MB), and the number banded each year have shown an increasing trend in all three areas (Figure 4). Because all chicks were banded each year (except for in 1995 when the field team had to abandon the survey work as a poor weather event was approaching the island, which resulted in fewer chicks being banded that year), the increase in the number of banded birds represents a genuine trend in the number of chicks recruited to the study areas. About 24% of the chicks banded in the study areas have been resighted in subsequent years, and most of them were resighted inside the study areas (Table 3). Birds were usually resighted after 4 or 5 years after tagging, but usually as pre-breeders, and it wasn't until 9 or 10 years later, that they were resighted as breeders (Figure 5). The oldest known-age birds recaptured in the three study areas for the first time in 2014 were from the 1998 cohort, and so were 16 years old. This indicates that many more years of recapture effort are required to obtain reliable estimates of the survival of these known-age birds. However, given that the mean age of first breeding of SBA at The Snares is 10–12 years (Francis & Sagar 2012) more birds from the later cohorts are likely to be recorded breeding in future.

Of the 2772 birds banded as chicks since 1992, 300 have since been observed breeding, at ages between 6 y and 14 y (
Table 4). Amongst the 300 birds, 213 bred in their natal area. It appeared that these birds strongly tended to breed close to where they were hatched. However, the evidence of natal fidelity is inconclusive given the large proportion of chicks never resighted. Because chicks do not always breed in the area in which they were hatched, if a bird has not been resighted since it was banded as a chick, and it is much more than 14 y since banding, then we can be confident that one of two things has happened: *mortality* (the bird has died) or *relocation* (it has chosen to breed in an area that is rarely or never observed). However, it is not possible to distinguish between these explanations.

All observations of banded birds that were in the pre-breeding state were dropped from the models. For birds banded as chicks, this meant ignoring resighting before the first year they were seen breeding; for other birds, it meant treating the year the bird was first seen breeding as the year of banding (before 1992, when breeding status was not recorded, we assumed no birds were banded as pre-breeders). The reason is that these pre-breeding observations can be misleading. Once birds start breeding they will always breed in the same area, but pre-breeders may be sighted in areas other than where they will eventually breed (Sagar & Warham 1998). Therefore we cannot tell whether a bird sighted (in the study areas) as a pre-breeder, but never as a breeder, has died or simply chosen to breed elsewhere. Another complicating factor is that the percentage of birds (other than chicks) that were not breeding in the year of banding has varied widely from year to year from 0% to 79%, averaging 18% and much of this variation derived from year-to-year changes in resighting effort, rather than changes in the population.

Table 1: Number of field days of the band-recapture study between 1994 and 2014

Year	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004
Days	27	38	26	30	27	32	35	36	44	30	30

Year	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014
Days	22	4	5	6	7	7	9	7	10	7

Table 2: History of banding of Southern Buller's albatross (SBA) in selected areas (MB = Mollymawk Bay; PB = Punui Bay; LPB = Lower Punui Bay; UPB = Upper Punui Bay) on Snares Islands.

						Number banded			
		Chicks				Adults			
Year	Researcher	MB	PB	LPB	UPB	MB	PB	LPB	UPB
1948	Richdale						127		
1961	Stonehouse					111	53		
1969	Bennington					176			
1970	Hay					4			
1972	Horning	64	66						
1975	Horning					38			
1977	Sagar					40			
1992–2004	Sagar	1059		503	435	425		206	209
2005–2014	Sagar					127		45	103

Table 3: Number of birds resighted, for birds banded as chicks since 1992. The rows separate the birds depending on their natal areas, which are characterised as 'study' (i.e., Mollymawk Bay, Upper Punui Bay, or Lower Punui Bay) and 'non-study' (i.e., areas immediately adjacent to the study area). The columns separate the birds depending on where they were resighted and whether they have bred.

	Resighted in study area		Resighted in non-study area		Total banded
	Breeding site	Undetermined	Breeding site	Undetermined	
Natal area					
study	172	210	45	34	1997
non-study	15	17	57	60	775

Table 4: Age at first (observed) breeding, by natal and breeding area, for birds that were banded as chicks since 1992 and have since been observed breeding. The rows separate the birds depending on their natal and breeding areas, which are characterised as 'Study' (i.e., Mollymawk Bay, Upper Punui Bay, or Lower Punui Bay) and 'Other'. Table entries are numbers of birds.

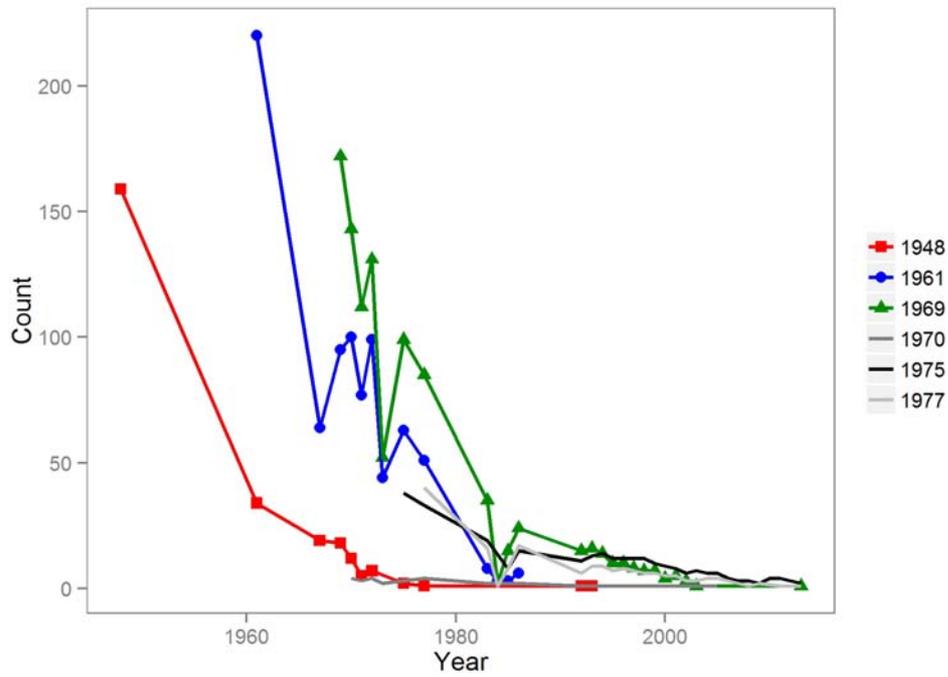| Natal area	Breeding area					Age at first breeding					
		6	7	8	9	10	11	12	13	14+	Total
Other	Other	0	0	1	1	9	13	6	14	15	59
Other	Study	0	0	1	1	1	3	1	1	7	15
Study	Other	0	0	1	8	7	7	9	17	22	54
Study	Study	2	1	3	7	25	23	34	17	60	172
											300

Figure 1: Numbers of birds banded (the first point in each line) and recaptured (from second point onwards in each line) for adults banded in the study area in 1948, 1961, 1969, 1979, 1975, and 1977.
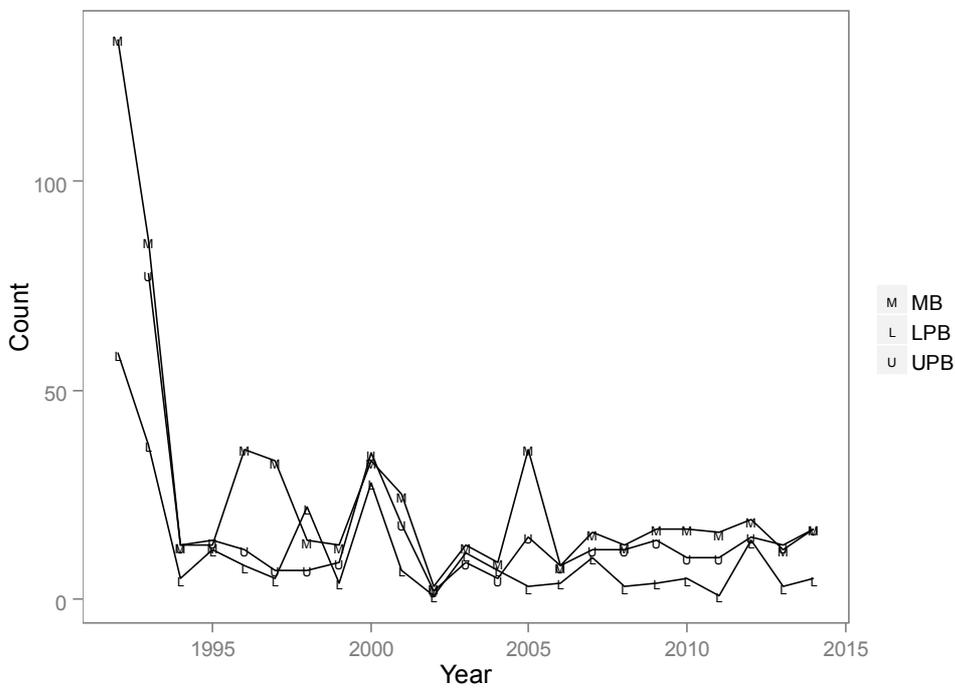

Figure 2: Annual numbers of adults banded in the study areas between 1992 and 2014.
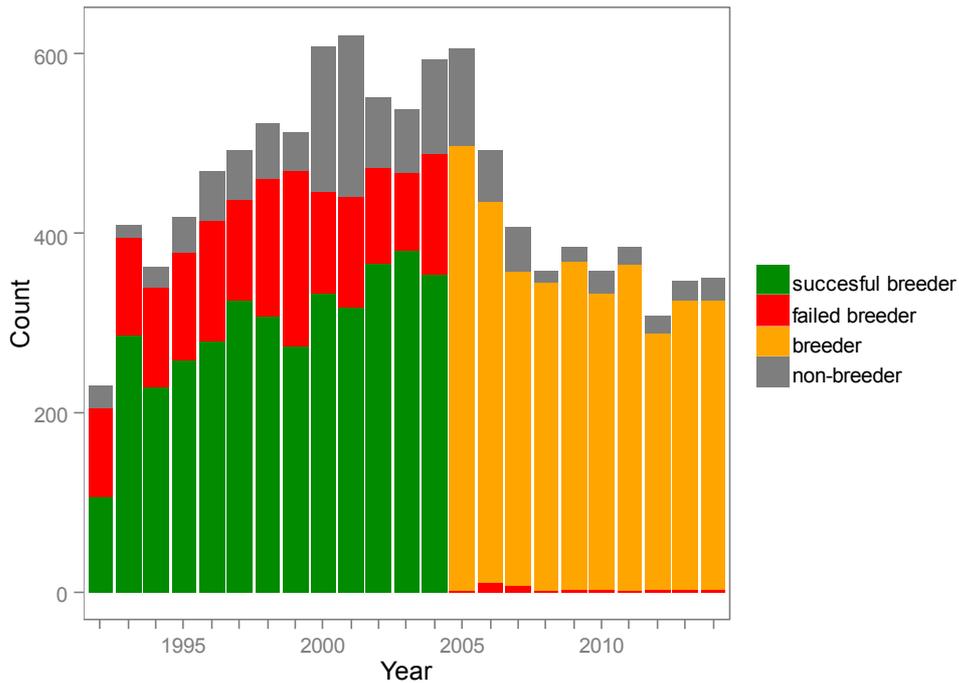
Figure 3: Annual numbers of birds observed in the study area (either newly, or previously, banded as adults in the study area) by breeding status between 1992. Note that since 2005 whether the breeding has been successful or not has generally been unknown.
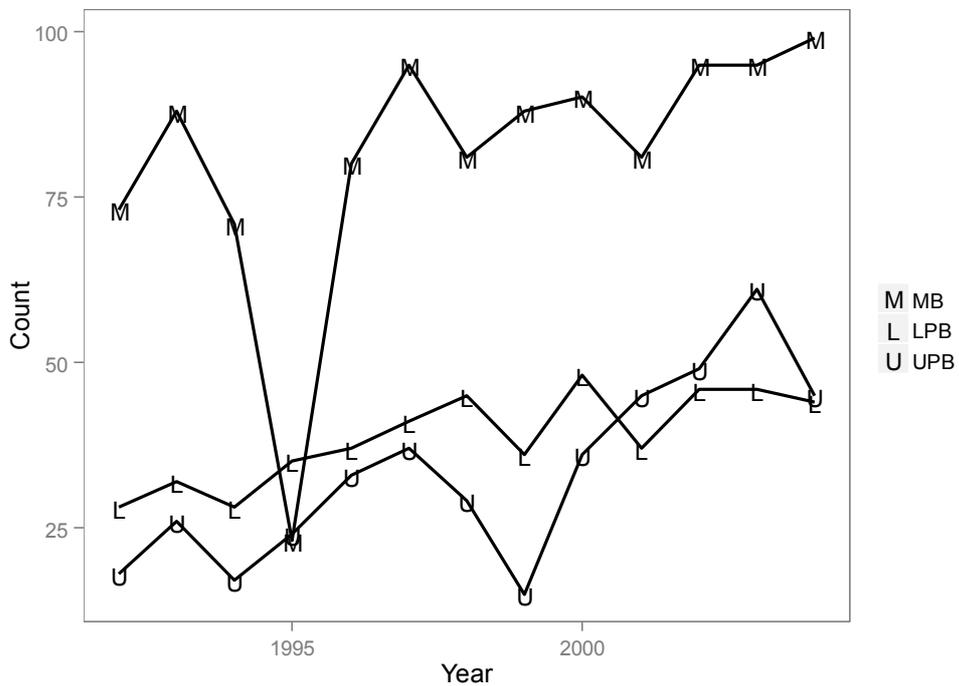

Figure 4: Annual numbers of chicks banded in the study areas between 1992 and 2005.
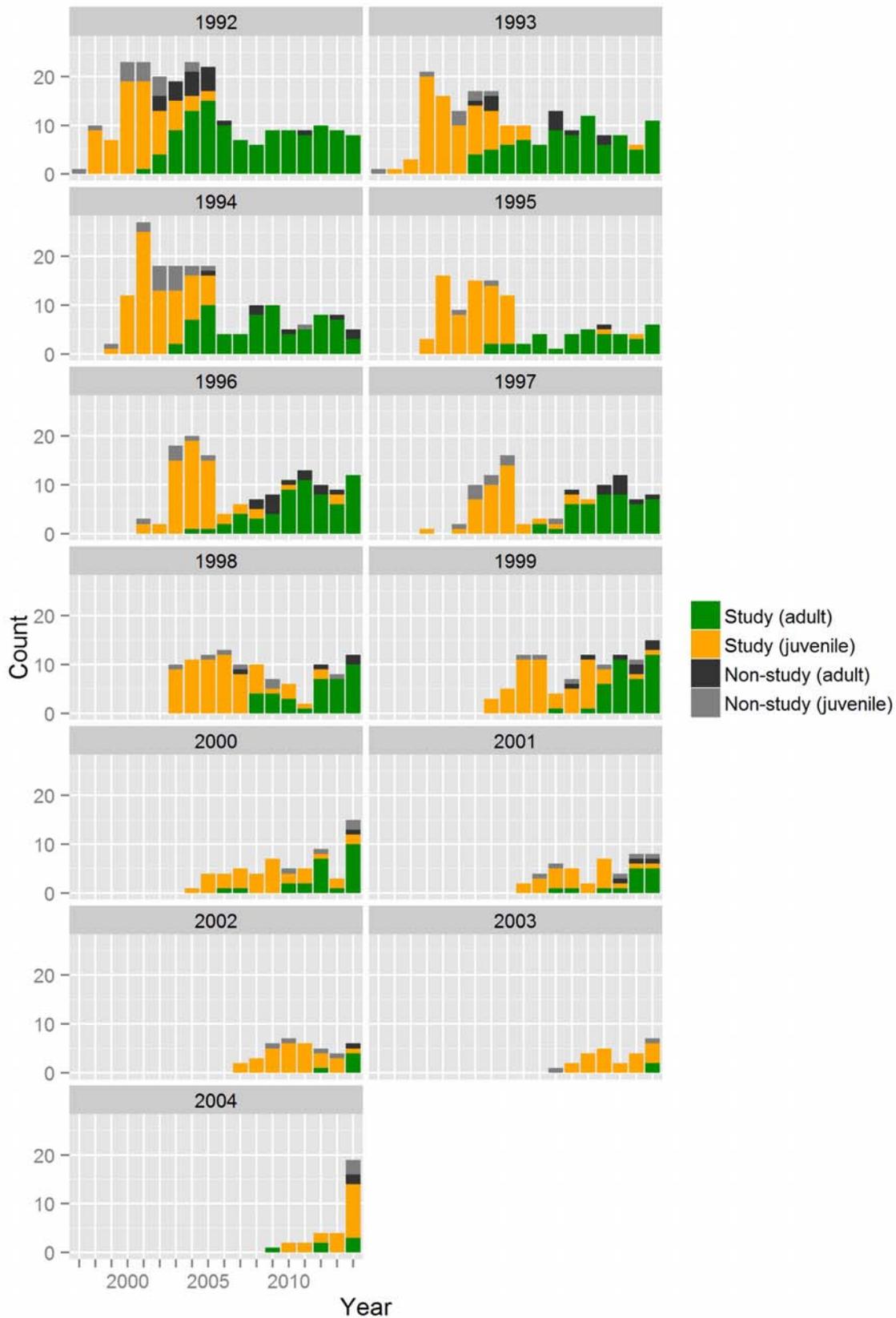
Figure 5: Annual numbers of birds observed by breeding status between 1997 and 2014 for each cohort of birds banded as chicks in the study area between 1992 and 2004. The breeding status indicates whether the birds have been resighted as a breeder or non-breeder, and whether the breeding area has been within with the study area.

2.2.2   Census data

There have now been five whole-island counts of the numbers of breeding pairs of SBA on North East Island and Alert Stack, The Snares, during the period 1969–2014 (Sagar 2014). The results of these counts show that the population increased markedly, more than doubling between 1969 and 2002 (Table 5). The survey of North East Island and adjacent Alert Stack in 2014 resulted in an estimate of 8047 breeding pairs, which was very similar to the 8165 breeding pairs estimated in 2002, indicating that the increase in size of the breeding population over the period 1969–2002 had not continued.

The whole-island surveys during 1969 and 2014 used similar methods and was completed at a similar time of year (Sagar 2014). The timing of survey was scheduled to occur close to the end of laying, when most birds sitting on a nest were presumed to be incubating (The laying period of SBA at The Snares extends from late December to the end of February, with most of the eggs laid by late January). SBA are monogamous, usually nest annually, and do not re-lay within a season if the single egg laid is broken (Sagar & Warham 1998). Therefore, counts of incubating birds plus broken and abandoned eggs are presumed to represent the number of pairs of birds breeding each year.

On North East Island and Alert Stack, ground counts were completed wherever access to nests was possible. Included in ground counts were birds incubating an intact egg, and nests with an abandoned or broken egg or an egg that had rolled out of the nest. Abandoned, broken and eggs rolled out of the nest were assumed to represent pairs of albatrosses that had attempted to breed that season. Birds breeding in inaccessible areas were counted from vantage points, at distances of up to 500 m, using binoculars. In the majority of such situations counts were made by all three observers independently and averaged. Where vantage point counts with an average of more than 100 occupied nests occurred, the total counted by each observer were required to be within 10% of the average. Where the average was less than 100 the totals were required to be within 5%. The counts were repeated until they met the above criteria. For both ground and vantage-point counts birds standing on nest mounds were not included in the totals. Abandoned and broken eggs could not be counted from vantage-point counts, and so totals from these counts are considered to represent the minimum number of breeding pairs.

The ground counts were both accurate and precise (Sagar & Stahl 2005). Francis et al. (2008) estimated coefficients of variations (CV) for the vantage counts by making distributional assumptions on these margin of errors. The method is described in Appendix A. We applied this method to the 1992 to 2014 censuses and the estimated CVs ranged between 0.6% and 1.3% (Table 6), suggesting that the vantage-point counts were generally very precise. For the 2014 survey, counts from individual observers were recorded, so the CV can be directly estimated from these counts. This estimate (0.7%) was smaller than that obtained from the fore-mentioned method. If we further assume, that the CV for the total ground counts in these censuses lay between these estimates and zero, then the overall CV for each census must lie between the values given in the second and third rows of Table 6, that is, less than 1%, and possibly as low as 0.2%.

Francis et al. (2008) argued that that 1969 census data was reliable and the observed increase in the size of the breeding population over the period 1969–2002 is likely to be real. The timing of the 1969 census was about 2–3 weeks earlier than the later census and was completed by a single observer. However these differences are not thought to have affected the results significantly. Despite the greater number of observers used, the later censuses located no concentrations of SBA nests not marked on the 1969 sheets, showing that all of them had been located in the earlier census. A direct comparison of the field sheets between the 1969 counts and later census suggested that the distribution of the concentration of nests in the 1969 were almost identical to later census, but the difference was mainly in the number of nests counted in each area, with fewer SBA counted in 1969. The five census counts imply an average population growth rate of 2.1% per year. Annual population growth rates calculated separately for the census subareas range from 1.4% to 3.6% (left panel, Figure 6); the corresponding rates for the five study areas range from 1.2% to 3.7% (right panel, Figure 6).

Table 5: Census counts of the number of breeding southern Buller's albatross (SBA) on Snares Islands (tabulated values are double the original counts, which were of breeding pairs).

Year	Count
1969	8 896
1992	15 366
1997	16 484
2002	17 426
2014	17 408

Table 6: Estimated CV (%) of the total vantage-point counts, and the total counts, from the censuses of The Snares in 1992, 1997, 2002, and 2014.

	1992	1997	2002	2014
CV (vantage count)	1.3	0.9	0.6	1.1
CV (total count)[1]	0.9	0.7	0.4	0.8
CV (total count)[2]	0.7	0.4	0.2	0.4

[1] Assuming total ground counts have the same CV as the total vantage-point counts
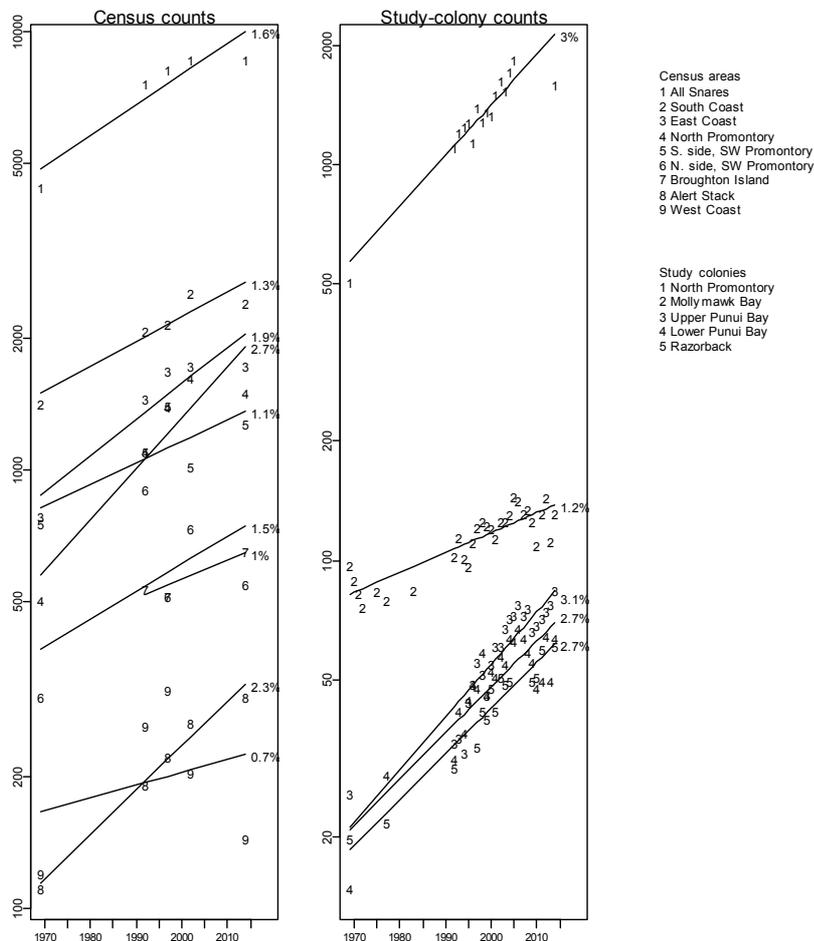
[2] Assuming the ground counts are without error

Figure 6: Plots of census and study-area counts of pairs of breeders with fitted trend lines and estimated annual percentage growth rates (calculated from the slopes of the trend lines). Data are from (Sagar 2014).

2.2.3 Fishing-related mortality data

Francis et al. (2008) collated estimates of fishing-related mortality of SBA in New Zealand fisheries for fishing years 2000/01–2003/04, primarily based on analyses carried out as part of Ministry of Fisheries project ENV2005-01. The estimates were derived from the trawl (TR), surface longline (SLL), and bottom longline (BLL) fisheries by "scaling up" observed bycatch by the ratio of total effort to observed effort for each combination of fishing method, FMA, and quarter. Total bycatch of SBA over 2000/01–2003/04 were obtained for 11 groups of trawl fisheries (as defined by combinations of FMA and quarter), and 6 groups of SLL fisheries (as defined by combinations of fleet, FMA, and quarter). The bycatch estimates for each fishery subgroups were included in the model (SBA 24) as observations to allow the model to determine the annual bycatch of SBA since 1990 (assuming that in each of the fishery subgroup, the bycatch is proportional to the product of fishing effort and the abundance of adults).

Richard&Abraham (2013a) estimated the capture of a number of species groupings of seabirds in New Zealand trawl and longline fisheries from observer records using Bayesian statistical models. They used Generalised Linear Models that predicted the logarithm of the expected captures during a fishing event as a linear function of a number of covariates. The coefficients of the covariates were determined by fitting the model to observed capture data, and the coefficients were then used to estimate the expected number of captures from unobserved fishing events. The estimated number therefore represents captures that would have been observed had observers been present on all fishing vessels ("observable captures"). Models were built for fishery subgroups as defined by fishing method, target species, and size of the vessel. Captures of SBA were estimated from 2002–03 to 2011–12 for trawl fisheries, and from 1998–99 to 2011–12 for longline fisheries (Table 7). The model-based estimates have the advantage of accounting for any biases in observer coverage that make the observations non-representative of the fishing effort. Furthermore, the use of negative-binomial model are appropriate for data that contained many events without any captures, and a few events with multiple captures (over-dispersion); the Bayesian model incorporated random effects, which compensate for the fact that, observers generally record data from all fishing on entire trips, therefore observations are not a random sample of all fishing effort. In addition, the estimation is driven directly by the observed captures, there are fewer assumptions in carrying out the estimates than are required for the risk assessment approach described below.

Richard&Abraham (2013b) carried out an extensive assessment of the potential impact of fishing-related mortalities and estimated the risk to a wide range of seabird species from New Zealand fisheries. They suggested that estimates based on observer bycatch records will under-represent the total number of fishing-related seabird mortalities as seabirds involved in fatal interactions with commercial fisheries may not be captured by fishers or recorded by observers. They calculated potential seabird fatalities by incorporating such cryptic mortalities which are not included in the number of observable captures. They estimated cryptic mortality multipliers which give the number of potential fatalities as a multiple of the number of observable captures. Separate multipliers were estimated depending on the fishing method and species type. For surface-longline fisheries, the estimates were based on values from a 15-year study which counted the number of seabirds that were caught when lines were set (Brothers et al. 2010). For trawl fisheries, the estimates were based on 190 hours of observations on the number of warp strikes and subsequent fatalities in the South African deepwater hake fishery in 2004 and 2005 (Watkins et al. 2008), and observations of warp strikes recorded in New Zealand trawl fisheries between the 2004–2005 and 2008–2009 fishing years (Abraham 2010). The estimated cryptic mortality multipliers are 8.66 for trawl and 2.08 for bottom and surface longline fisheries (Table 8). There are no data available regarding cryptic mortality in bottom-longline fisheries so the same multiplier as for surface-longline fisheries was used (Richard & Abraham 2013b). The estimation of cryptic mortality multipliers included uncertainty. However the uncertainty is limited to statistical uncertainty from the underlying data sources and does not include uncertainty associated with extrapolating from results obtained in different fisheries, and in different jurisdictions (Richard & Abraham 2013b).

Table 7: Estimated captures of SBA (mean 95% confidence interval) and by year and fishery from Richard&Abraham (2013a).

Year	Trawl		BLL		SLL	
	Captures	95% c.i	Captures	95% c.i	Captures	95% c.i
1999			61	(18–130)	218	(164–285)
2000			58	(17–124)	217	(155–289)
2001			59	(17–124)	228	(163–292)
2002			40	(11–87)	291	(219–380)
2003	80	(31–172)	51	(17–104)	277	(208–361)
2004	95	(39–211)	39	(12–80)	194	(148–246)
2005	209	(106–422)	81	(26–165)	99	(73–129)
2006	93	(44–176)	70	(23–142)	100	(72–132)
2007	59	(23–119)	118	(39–238)	158	(125–197)
2008	110	(59–197)	109	(39–217)	99	(75–133)
2009	83	(47–144)	82	(26–167)	107	(83–137)
2010	71	(34–144)	85	(27–171)	158	(129–191)
2011	105	(56–194)	76	(24–153)	106	(80–136)
2012	162	(92–309)	58	(20–115)	109	(83–139)

Table 8: Mean and 95% confidence interval of the number of potential seabird fatalities per observed capture in the longline and trawl fisheries for SBA (from table 8 of (Richard & Abraham 2013b).

Fishery	Mean	95% CI
Bottom and surface longline	2.08	1.79–2.44
Trawl	8.66	5.67–12.78

2.3 Model

The Snares Islands population of SBA was modelled using the SeaBird software package (Francis et al. 2008). SeaBird uses a multi-state, integrated modelling approach which simulates population processes and estimates parameters using Bayesian or likelihood-based methods. SeaBird allows analyses of multiple types of observational data including abundance (absolute or relative), fishing-related mortality, and mark-recapture observations, thereby allowing population growth rates and demographic parameters to be estimated within the same modelling framework. The approach used to analyse these observations in SeaBird is based on a generalisation of the widely-used Cormack-Jolly-Seber method, which is used to estimate survival rates and resighting probabilities (Cormack 1964; Jolly 1965; Seber 1965). The generalisation and extension of this method in SeaBird allows multi-state information (e.g. frequency and success of breeding) and incomplete observations in the mark-recapture data to be analysed.

2.3.1 Previous models of SBA

The SBA models developed by Francis et al. (2008) incorporated census counts 1969–2002, mark-recapture observations 1948–2007, and bycatch estimates 2001–2004. Models of increasing complexity were developed to explore various aspect of the data and alternative model structures. The aim was to investigate the information content of the various data sets, and to optimise model configurations. Below is a summary of the models developed by Francis et al. (2008) and some of the decisions those authors made.

SBA11 used the presence/absence observations of birds banded as adults and suggested that the mismatch between observed and predicted number of recaptures from about 15–20 years after banding could be attributed to potential band loss (it was suggested that some types of bands were more prone to loss than others). The bias of predicted recaptures was much alleviated if the resighting observations of each individual bird was restricted to a maximum of 20 years (SBA 12) or 15 years (SBA 13, and subsequent models). This restriction was applied to the birds banded prior to 1992, because the bands were made of aluminium or monel metal, both of which wore quickly, leading to band loss. All birds banded since 1992 have been banded with stainless steel bands, which have shown no sign of wear so the full set of resighting observations was used.

SBA17 investigated various model configurations to estimate adult survival using birds banded as adults. The optimal model was selected based on the Akaike Information Criterion, or AIC (Akaike 1974). The mark-recapture observations were split into seven sets, each named according to the year and location of banding. Resighting probabilities were allowed to vary between study areas and between breeders and non-breeders. The year blocks for the time-varying parameter of adult survival were investigated.

SBA18 used just the known-age mark-recapture data in order to estimate the 'juvenile' parameters: age at first breeding; and juvenile mortality (i.e., that which occurs in the years prior to first breeding). It was found that the model is not able to distinguish between juvenile mortality and relocation. A parametric form was also determined to define age at first breeding (see Section 3.1 below).

SBA19 used all mark-recapture data including birds tagged as adults and known-age birds. This model showed that the information in the known-age and adult mark-recapture data sets are almost independent. The model was used to estimate the probability that, when a bird first breeds, this occurs in one of the study areas, and it was found that this parameter is confounded with juvenile survival.

SBA21 used mark-recapture and census data and estimated the initial population size. SBA21 generated almost identical estimates of demographic parameters as SBA19 but suggested a greatly different population trend (this is shown by running SBA20 assuming parameters estimated from SBA19). This implied that the mark-recapture data contain no information about the absolute size of the population, and only indirect information about trends in population size. SBA21 was considered as the base model that provided the best assessment of the SBA population (Francis & Sagar 2012).

SBA24 incorporated fishing-related mortality information. This model assumed that fishing-related mortality affected only adult SBA and was proportional to the product of fishing effort and the abundance of adults. Model SBA24 generated virtually the same population trajectory as the base model but the estimates of post-1990 adult survival increased by about 0.016 on average.

2.3.2 Model structure and parameters

SBA19, SBA21, and SBA24 of Francis et al. (2008) were updated in this analysis and the updated models are named models 2, 3, 4, respectively, in this report. These models incorporate new data available since 2007: Model 2 included only the mark-recapture observations; Model 3 included both mark-recapture and census data; Model 4 was the same as Model 3, except that it incorporated fishing-related mortality. Model 3 is considered to be the base model by the AEWG. The structure and parameters of the base model are described below.

The model partition divided the SBA population into 20 classes, corresponding to chicks (class 0), pre-breeders of ages 1 to 13 (classes 1–13), and adults (classes 14–19). The adults were divided into six classes by breeding status and location (Table 9). The movement of seabirds between partition classes is determined by a transition matrix, which was defined as follows:

$$
T_{ij} = \begin{cases}
1 & \text{if } j = i+1 \text{ and } i < 5 \\
\text{Pbrstd} \times \text{P1stbr}(i+1) \times \text{Psuccess} & \text{if } 5 \leq i \leq 12 \text{ and } j = 14 \\
\text{Pbrstd} \times \text{P1stbr}(i+1) \times (1 - \text{Psuccess}) & \text{if } 5 \leq i \leq 12 \text{ and } j = 15 \\
(1 - \text{Pbrstd}) \times \text{P1stbr}(i+1) \times \text{Psuccess} & \text{if } 5 \leq i \leq 12 \text{ and } j = 17 \\
(1 - \text{Pbrstd}) \times \text{P1stbr}(i+1) \times (1 - \text{Psuccess}) & \text{if } 5 \leq i \leq 12 \text{ and } j = 18 \\
1 - \text{P1stbr}(i+1) & \text{if } 5 \leq i \leq 12 \text{ and } j = i+1 \\
\text{Pbrstd} \times \text{Psuccess} & \text{if } i = 13 \text{ and } j = 14 \\
\text{Pbrstd} \times (1 - \text{Psuccess}) & \text{if } i = 13 \text{ and } j = 15 \\
(1 - \text{Pbrstd}) \times \text{Psuccess} & \text{if } i = 13 \text{ and } j = 17 \\
(1 - \text{Pbrstd}) \times (1 - \text{Psuccess}) & \text{if } i = 13 \text{ and } j = 18 \\
\text{Tbrbr} \times \text{Psuccess} & \text{if } i = 14 \text{ or } 17 \text{ and } j = i, \text{ or} \\
& i = 15 \text{ or } 18 \text{ and } j = i-1 \\
\text{Tbrbr} \times (1 - \text{Psuccess}) & \text{if } i = 14 \text{ or } 17 \text{ and } j = i+1, \text{ or} \\
& i = 15 \text{ or } 18 \text{ and } j = i \\
1 - \text{Tbrbr} & \text{if } i = 14 \text{ or } 15 \text{ and } j = 16, \text{ or} \\
& i = 17 \text{ or } 18 \text{ and } j = 19 \\
\text{Tnbrbr} \times \text{Psuccess} & \text{if } i = 16 \text{ or } 19 \text{ and } j = i-2 \\
\text{Tnbrbr} \times (1 - \text{Psuccess}) & \text{if } i = 16 \text{ or } 19 \text{ and } j = i-1 \\
1 - \text{Tnbrbr} & \text{if } i = 16 \text{ or } 19 \text{ and } j = i \\
0 & \text{otherwise}
\end{cases}
$$

where T_{ij} is the probability that a bird in class i will move to class j in the next year (assuming it survives). Chicks transition into pre-breeders at age 1. Pre-breeders transition into breeders with probability P1stbr(i), which is the probability that a bird will breed at age i, given that it survives to that age and has not previously bred. When a bird first breeds, this occurs in one of the study areas with probability Pbrstd. Within each area, adults transition between breeders and non-breeders following probability Tbrbr (probability of being a breeder this year, given the bird being a breeder last year), or Tnbrbr (probability of being a breeder this year, given the bird being a non-breeder last year). Transition into breeders are further split into a successful breeder class (Nsbrstd, Nsbroth), and a failed breeder class (Nfbrstd, Nfbroth) according to Psuccess, which is the probability that a bird that bred would successfully raise a chick. Note that adult birds never move between the study areas and the other areas.

Table 9: Description of partition classes for the base model. N1-N13 refer to pre-breeders of ages 1 to 13; classes 14–19 all contain adults which are classified according to whether the bird successfully bred (sbr) or bred but failed to rear a chick (fbr) or did not breed (nbr), and whether it is in the study (std) or other (oth) areas.

Classes	Name	Description
N0	Nchick	Chicks
N1–13	N1–N13	Juveniles from ages 1 to 13
N14–16	Nsbrstd, Nfbrstd, Nnbrstd	Adults in the study area (std), classified by breeding status (successful, failed, or nonbreeder)
N17–19	Nsbroth, Nfbroth, Nnbroth	Adults in areas other than the study area (oth)

It was assumed that P1stbr(i) was a linear function of age in logit space, i.e.,

$$\text{logit}(\text{P1stbr}(i)) = \text{logit}(\text{P1stbr}(9)) + (i-9) \times \text{oddsmult}$$

where the logit function is defined as logit(p) = log[p/(1–p)] P1stbr(9) and oddsmult are parameters estimated in the model. It is assumed here that P1stbr(i) = 0 for ages less than 6, and that P1stbr(14) = 1.

Estimated parameters are summarised in Table 10, and they include those that define transition probabilities (see above), the number of adult birds in 1948 (N0), annual survival probabilities for pre-breeders ($P_{surv,juv}$, assumed to be the same for all years and for ages) and for adults ($P_{surv,adu}$, assumed to be time-varying), and resighting probabilities (assumed to be time-varying and estimated annually). Altogether, 132, parameters were estimated, including 6 for transition, 14 for survival, and 111 for resighting probabilities (in 6 vectors).

Table 10: Description of estimated parameters for the base model. Parameters of length greater than 1 were time-varying. MB, LPB and UPB relate to the three study areas

Name	Length	Description
N0	1	Number of adults in population in 1948
$P_{surv, adu}$	13	Annual probability of survival for adults
$P_{surv, juv}$	1	Annual probability of survival for juveniles
Pbrstd	1	Probability that first breeding occurs in the study area
P1stbr(9)	1	Probability a bird will first breed at age 9, given that it survives to that age
oddmult	1	Slope of linear relation between logit(P1stbra) and age a
Tbrbr	1	Probability of being a breeder this year, given the bird being a breeder last year
Tnbrbr	1	Probability of being a breeder this year, given the bird being a nonbreeder last year
Psuccess	1	Probability that breeding is successful
PrPB, PrMB	10, 13	Early (pre-1992) resighting probabilities in PB, or MB
PrMBbr, PrLPBbr	22, 22	Recent (post-1992) resighting probabilities for breeders in MB or LPB
PrMBnbr, PrLPBnbr	22, 22	Recent (post-1992) resighting probabilities for non-breeders in MB or LPB

Other population processes included in the model were recruitment and survival. Each year, the number of chicks recruited into the population was equal to half the number of successful breeders (assuming a male to female ratio of 1). The survival was parameterised as juvenile (i.e., that which occurs in the years prior to first breeding) and adult survival. The base model doesn't explicitly include fishing-related mortality so the estimated survival (both $P_{surv,juv}$ and $P_{surv,adu}$) should have included both natural mortality and fishery induced mortality.

The mark-recapture observations fell into ten sets (seven for birds tagged as adults and three for birds tagged as chicks), each named according to the year and location of banding (Table 11). The pre-1992 mark-recapture data contained no observations of breeding status and a single vector of resighting probabilities was estimated for each area. The post-1992 observations included breeding status and resighting-probabilities were estimated as separate vectors for breeders (e.g., PrLPBbr) and non-breeders (e.g., PrLPBbr). Resighting probabilities were assumed to be 0 for all juvenile classes (Nchick and N1–N13) and for adults outside the study area (classes Nsbroth, Nfbroth, Nnbroth). Note that analogous post-1992 resighting parameters (PrMBbr & PrMBnbr) were estimated for areas MB and UPB (PrUPBbr & PrUPBnbr) (Francis et al. 2008).

The (known-age) data associated with each study area were of two types: birds observed breeding in that area; and birds that were born in that area but which have not been observed breeding in any study area (i.e., they've either not been observed breeding or have been observed breeding outside the study areas). Birds banded (as chicks) outside the study areas and resighted in non-study areas were not included (because resighting effort in non-study areas was highly variable). These data were mainly used to estimate 'juvenile' parameters: juvenile survival, age at first breeding, and the probability of first breeding occurring at one of the study areas. Resightings of pre-breeders were ignored because these birds sometimes visit areas other than the ones at which they eventually choose to breed.

Birds tagged as adults before 1992 contained no observations of breeding status. They were assumed to be tagged as breeders, but resighted with breeding status unknown (modelled as a 'composite' class in SeaBird; it means that the bird was observed to have been in one of a user-defined group of partition classes, but exactly which class it was in was unknown). The observations of these data were restricted to a maximum of 15 years to mitigate the potential issue of tag loss (see Section 2.3.1). Some of birds banded in the early years were rebanded and the vectors of observations for these birds were split at the year of rebanding. Thus, for example, a bird banded in 1948 and rebanded in 1969 was entered as two vectors: one for 1948–1969 with the original band number, and another for 1969–2012 with the new band number (note that the way the likelihoods are calculated ensures that the 1969 observation is not counted twice). The restriction to 15 years of observations does not apply to birds banded after 1992 because the band loss was unlikely to be an issue (see Section 2.3.1).

For birds tagged as adults after 1992, about 18% of the adult birds that were banded were not breeding in the year of banding (but this percentage varied widely from year to year – from 0% to 79% – primarily because of changes in research priorities). Data from these birds can be misleading because it is not possible to distinguish between pre-breeders, which may not yet have decided in which area they will eventually breed, and birds that have previously bred in the area where they were banded, but were not breeding that year. With the former type of bird, we cannot distinguish between a bird that subsequently dies and one that decides to breed elsewhere (and is thus not observed). To avoid these ambiguities, all observations before the first observed breeding were ignored in all models (so the year in which a bird was first observed breeding was treated as the year of banding).

Model 2 included only the mark-recapture observations, and used a simplified partition where the breeders were not differentiated between successful and failed breeders (class 14 and 15 were combined into one class, and so are classes 17 and 18). The model did not have the parameter Psuccess and the transition matrix is reduced accordingly (e.g. the probability of transitioning into a breeder class is the combination of the transition probability into a successful and a failed breeder class). This particular model concerns only demographic rates and is not used to estimate the population size. Therefore the breeding success rate was not considered and annual recruitment was fixed at an arbitrary constant.

Model 3 incorporated the census data. The census observations were assumed to be normally distributed with a CV of 0.06 because a simple simulation experiment showed that year-to-year variation in the transition probabilities, Tbrbr and Tnbrbr, (which is not allowed for in model 3) can cause this amount of variation in the number of breeders (Francis et al. 2008). A variation of model 3 (model 3.1) was also considered which allowed both Tbrbr and Tnbrbr to be time-varying.

Model 4 incorporated fishing-related mortality into the model but dealt with this information in a different way than model SBA24. SBA24 estimated fishing-related mortality from 1990 onwards by fitting to the estimated observable captures between 2001 and 2004, with predicted total captures assumed to be proportional to the product of fishing effort and the abundance of adults in a number of fishery sectors. SBA24 did not estimate fishing-related mortality before 1990 and therefore the survival estimates before 1990 included both natural and fishing-related mortality (note that this did not mean that the model assumed no fishing-related mortality before 1990). This approach was adopted in SBA24 because reliable bycatch estimates of SBA were available for only a very short

period of time (2001–2004). However this approach effectively made strong assumptions that for each fishery, a unit of effort would catch the same proportion of the SBA population each year. This assumption was not supported by bycatch estimates from other sources (see figure 30B in Francis et al. (2008)). Model 4 included the estimates of observable SBA captures from TR, SLL, and BLL (see Table 7) as fishery induced mortality which were directly applied to the population in the corresponding period. This assumed that only adults were vulnerable to fishing, that breeders and non-breeders were equally vulnerable, and that there was no cryptic mortality. Model 4 did not estimate fishing-related mortality before the start of the estimated captures.

An additional model, model 4.1 was also considered. This model included the estimates of total fishing-related mortality (see Section 2.2.3). The total fishing-related mortality was calculated by applying the corresponding cryptic mortality multiplier (Table 8) to the estimates of observable bycatch for each of the TR, SLL, and BLL fisheries (see Table 7). The total fishing-related mortalities concern the bycatch for the entire SBA population within the New Zealand. We further assumed that 63% of the mortality is related to the population of SBA on Snares Islands. This percentage is based on the estimates of population size for SBA on the Snares and Solander Island in 2002 (8713 and 4912 breeding pairs, respectively, see Sagar&Stahl (2005)). Model 4.1 did not estimate fishing-related mortality before the start of the estimated captures.

All the models considered are summarised in Table 12.

Table 11: Characteristics of the seven sets of observations used in the base model: number of vectors of banded birds (total number and those being included in the model); number of resighting observations (total number and number being included); whether breeding status was observed; and which vector(s) of resighting probabilities was associated with it. MB, LPB and UPB relate to the three study areas.

Dataset	Number banded		Number observed		Breeding status observed	Resighting probability vector
	Total	Included	Total	Included		
1948 PB	141	141	83	43	No	PrPB
1961 PB	53	53	136	122	No	PrPB
1961 MB	132	131	350	320	No	PrMB
1969–1977 MB	265	265	1082	804	No	PrMB
1992–2013 MB	600	600	4361	4216	Yes	PrMBbr, PrMBnbr
1992–2013 LPB	253	253	2021	1957	Yes	PrLPBbr, PrLPBnbr
1992–2013 UPB	318	318	2223	2161	Yes	PrMBbr, PrMBnbr
1992–2004 MB (known-age)	1056	1056	449	274	Yes	PrMBbr, PrMBnbr
1992–2004 LPB (known-age)	495	495	199	124	Yes	PrLPBbr, PrLPBnbr
1992–2004 UPB (known-age)	447	447	199	129	Yes	PrMBbr, PrMBnbr

Table 12: A description of model runs conducted.

Model	Description
2	Used the full mark recapture data (updates model 19 of Francis et al. (2008))
3	Base model using the full mark recapture data and census data
3.1	Model 3, but allowing Tbrbr and Tnbrbr to be time-varying
4	Model 3 with the addition of observable fishing-related mortality
4.1	Model 4, but fishing-related mortality including cryptic mortality

3. RESULTS

3.1 Model 2

Model 2 included only the mark recapture data and it updated model SBA19 of Francis et al. (2008). Estimated adult survival and resighting probabilities were very similar to SBA19 (Figure 7), although the adult survival for 2004–2007 was estimated to be higher than from SBA19 (0.933 compared to 0.920). This small difference is not surprising as estimates of survival for more recent years are usually associated with higher uncertainty due to the well-known confounding of resighting and survival probabilities. The survival estimate for 2008–2012 was 0.928, marginally lower than the previous period.

Estimates of resighting probabilities (Figure 7) are almost the same as for SBA19 (for comparable periods). There was a noticeable drop in resighting probability estimates from 2005 onwards for both breeders and non-breeders and for all study areas. This results from the much reduced resighting effort during this period (Table 2).

The estimated breeding probability was 0.833 for breeders in the previous year, and 0.564 for non-breeders in the previous year. These estimates are similar to those from SBA19 (Table 13).

The estimate of juvenile survival was 1 (as in SBA19). This parameter has been shown to be highly correlated with Pbrstd (the probability that first breeding occurs in the study area, estimated to be 0.07). Juvenile survival was difficult to estimate because the study site does not cover the entire SBA population, therefore it is not possible to distinguish between mortality and relocation, if a bird has not been resighted since it was banded as a chick.

The probability, $P_{br,a}$ that a bird will first breed at age a can be calculated using the following recursive equations:

$$P_{br,6} = c\text{P1stbr}(6)$$
$$P_{br,a} = P_{br,a-1}\text{P}_{\text{surv,juv}}\text{P1stbr(a)}\big(1 - \text{P1stbr}(a-1)\big)/\text{P1stbr}(a-1)$$

where the constant c is adjusted so that $\sum_{a=6}^{13} P_{br,a} = 1$). Estimated $P_{br,a}$ is shown in Figure 8. Maximum $P_{br,a}$ occurred at about age 12 –as in SBA19 – but $P_{br,a}$ was skewed to the right in Model 2 relative to SBA19. The estimates from Model 2 were informed by more observations (299 birds resighted as first breeders (
Table 4), as opposed to 99 observations in SBA19. In addition, the SBA19 estimates were based on a much shorter resighting history for the known-age birds, and since 2007 many more birds have come back to the study area to breed for the first time.

The fits to mark-recapture observations were examined by comparing observed and expected numbers of birds resighted each year (Figure 9). The observed number in each year was the total number of birds banded and resighted (breeders, non-breeders and chicks) for all study areas combined (accounting for area-specific resighting probability). The expected number was the predicted number of birds in the model partition assuming that recruitment into the population each year is the number of birds tagged. Overall there was a reasonably good match between expected and observed numbers. The expected number exceeded observed since 1976, due to the maximum year of 15 applied to resighting observations of birds banded prior to 1992 (i.e. observations older than 15 years were discarded, see Section 2.3.1).

Table 13: Comparison of breeding probability estimates from Model 2 of this assessment and SBA19 (Francis et al. 2008). Tbrbr and Tnbrbr are the probability of breeders and non-breeders in yr-1 breeding in the current year

	Tbrbr	Tnbrbr
SBA19	0.838	0.581
Model 2	0.833	0.566
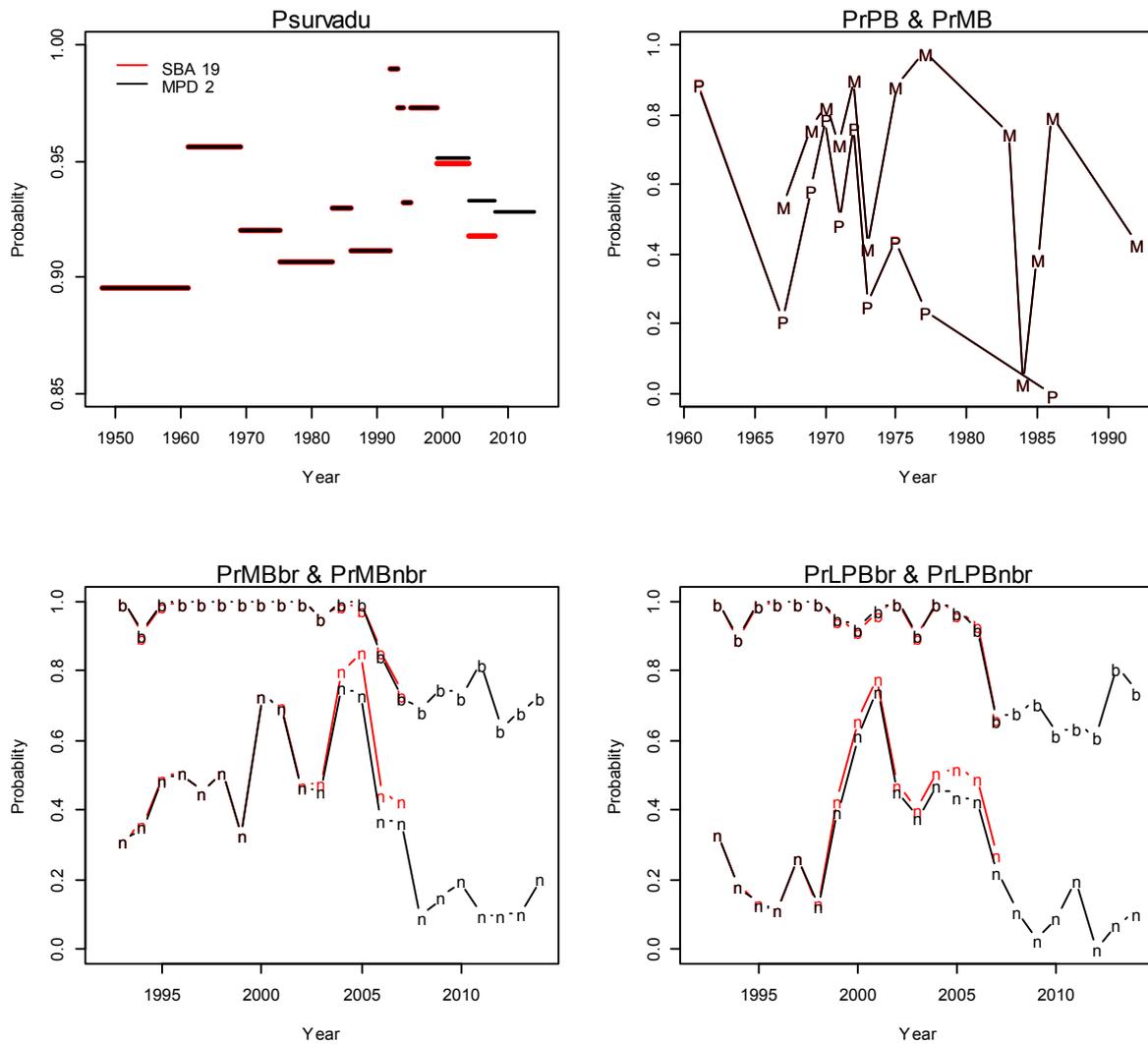
Figure 7: Comparison of estimates of the survival and resighting parameter estimates that are common to SBA19 from Francis et al. (2008) and Model 2. "M" is the resighting probability for MB and "P" is for PB. "b" is the resighting probability for breeders and "n" is for non-breeders.
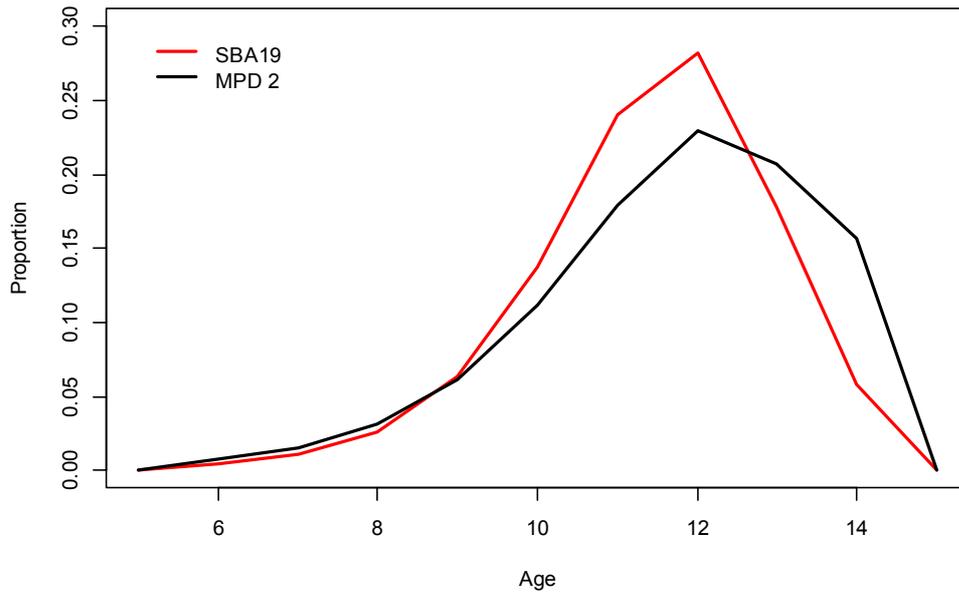
Figure 8: Comparison of estimates of age at first breeding from SBA19 and Model 2.
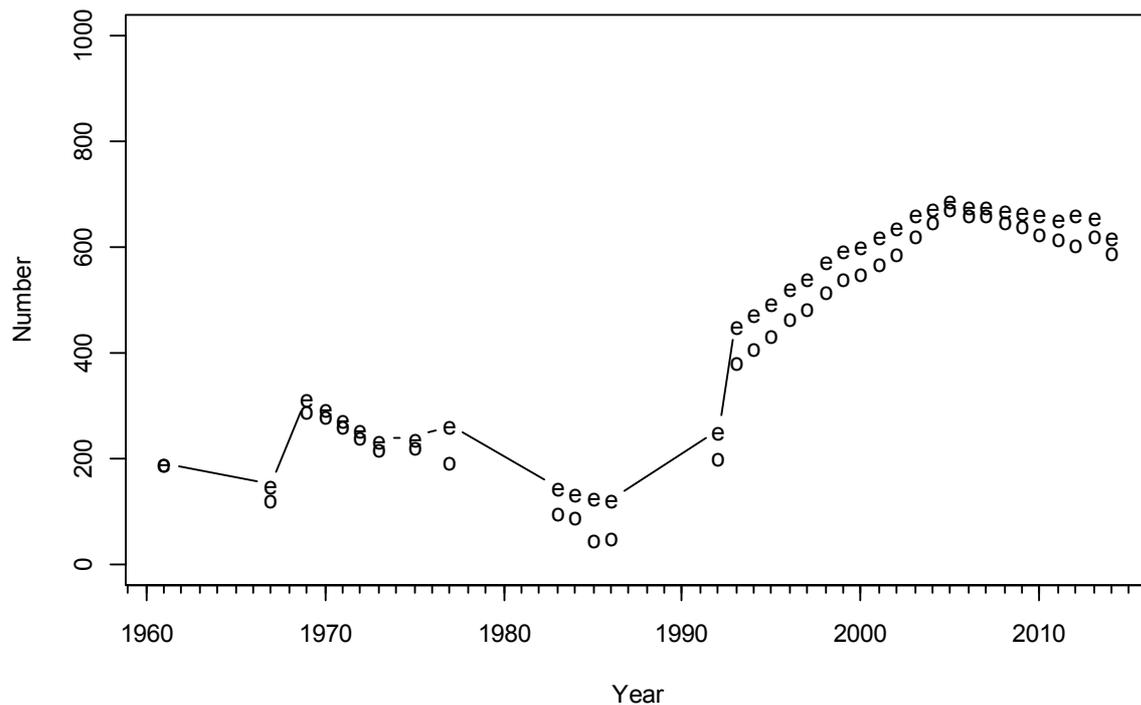
Figure 9: Observed ('o') and expected (e) numbers of birds seen for Model 2. Observed number in a year included birds resighted and tagged for all study areas combined (accounting for resighting probabilities).

3.2 Model 3 and 3.1

Mark-recapture observations provide no information about the absolute size of a seabird population (except that the minimum population cannot be less than the tagged population), although it does indirectly inform trends in population size. Model 3 is the same as model 2, except that it also includes census observations (Table 5). There are two additional parameters – N0 and Psuccess – and

the model partition was altered to allow the estimation of Psuccess. The partition and estimated parameters for model 3 are given in Table 9 and Table 10, respectively.

This model estimated that the population has been growing over the past 60 years at an average rate of 2.8% per year, with slightly higher growth rates between the mid-1980s and early 2000s, although growth was estimated to have slowed or stopped since then (Figure 10). Estimated breeding population size was within the confidence intervals for census observations, with about 80% of the total adult population breeding each year.

The sensitivity of estimates to fitting to census data as well as mark-recapture data is shown by comparison of models 2 and 3. $P_{surv,juv}$ was lower (and, to compensate, Pbrstd was higher) in model 3 (fitting to census) (Table 14). The other difference occurred in compensatory parameters - $P_{surv,adu}$ was much higher for the periods 1948–60 and 1983–91, and to compensate PrPB was much lower in 1961 and the PrMB estimate was lower in 1992 (Figure 11).

For those years in which the study areas were visited twice we can directly calculate breeding success (outside of the model). The direct estimates showed substantial year to year variation and the mean value (0.69) was close to the estimate from Model 3 (0.72) (Figure 12). The direct estimates appear to have an upward trend (suggesting an increasing trend in breeding success) though the trend was not significant if the initial very low estimate was removed.

The trend in model estimates of annual chick production agreed with the numbers banded each year in the study areas (Figure 13).

An examination of the mark-recapture data suggested that some birds are 'good' breeders (because they are observed breeding year after year) and others are not. In other words there appears to be between-bird heterogeneity in breeding rate. This heterogeneity is not allowed for in the model, where rates of breeding are controlled by the transition parameters Tbrbr and Tnbrbr, which are assumed to be the same for all birds. As a simple test of this assumption we restricted attention to birds that were banded as breeders since 1992 and counted how many consecutive years after banding each bird was observed breeding (see Appendix 3 for this method). For model 3, there was reasonable agreement between expected and observed breeding frequency, suggesting that although the breeding rate might differ among individual birds, the average pattern over the population can be predicted reasonably well (Figure 14).

Model 3.1 was exactly the same as model 3 except that the transition parameters, Tbrbr and Tnbrbr, were allowed to vary from year to year. This model was no better in terms of AIC (an additional 44 parameters reduced the objective function by 22.5). The estimated numbers of breeders from model 3.1 showed more year-to-year variation (as was expected), but in terms of trend was similar to that of model 3, except in the last 10 years, where the number of breeders dropped sharply in 2006 and 2007, and then increased sharply in 2008 (Figure 15). This drop derives from the corresponding fall in all three study areas of the observed number of breeders in 2006 and 2007 (see Figure 3), although resighting effort was reduced at this time and the true number of breeders in the population may not have dropped (Figure 16). The model explained the fall in observed numbers as a reduction in breeding probability in these years (Figure 17). The increase in the number of breeders since 2008 was due to the higher than average estimate of breeding probability since 2008. The breeding probability for non-breeders was estimated to be close to 1 from 2008 (Figure 17) and this is not credible. The annual count of birds observed as nonbreeders for two consecutive years varied between 1 and 25 from 1995–2006 (mean = 9), but was zero in all years from 2008 due to the low resighting probability of non-breeders in this period – and this is likely to have biased the breeding probability estimate for this later period.
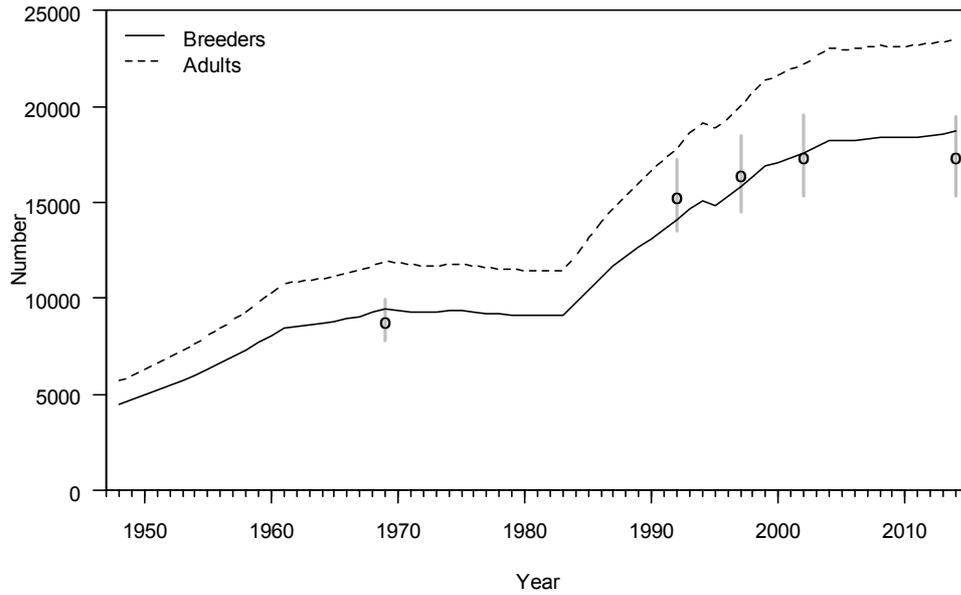
Figure 10: Model 3 estimates of number of breeders (solid line) and adults (broken line) in each year. Also shown are the census observations of numbers of breeders, with assumed 95% confidence intervals (vertical lines).
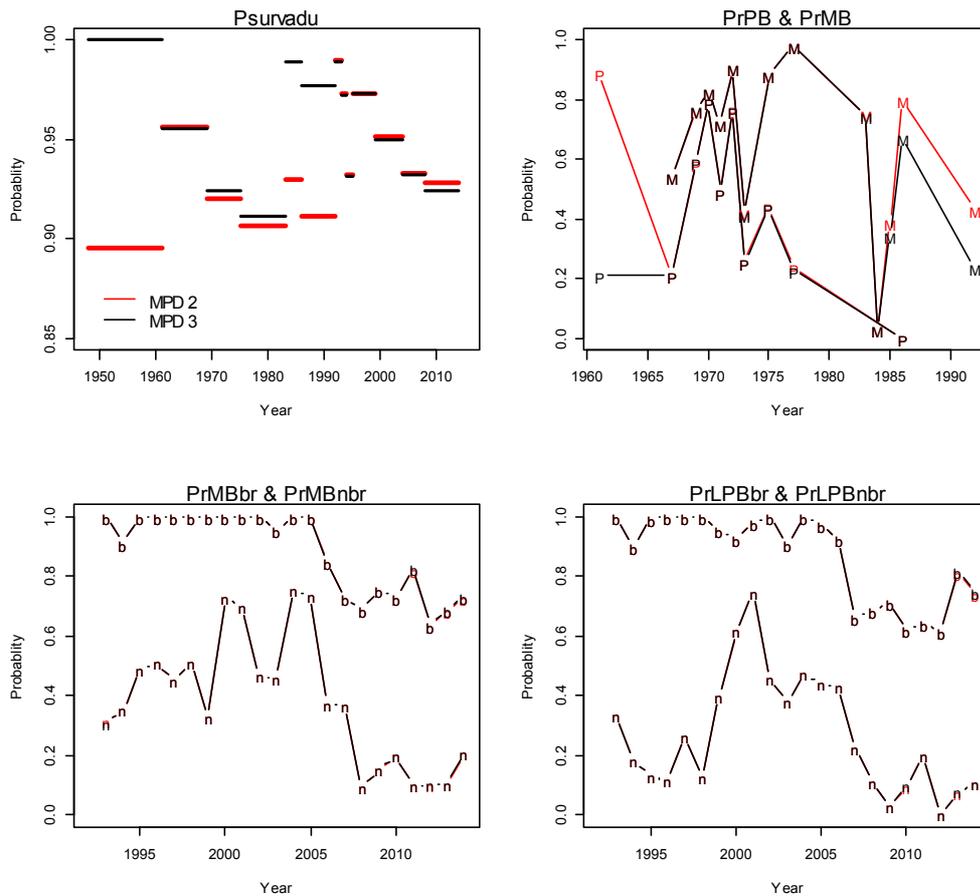

Figure 11: Comparison of estimates of the survival and resighting parameter estimates that are common to Models 2 (red) and 3 (black).
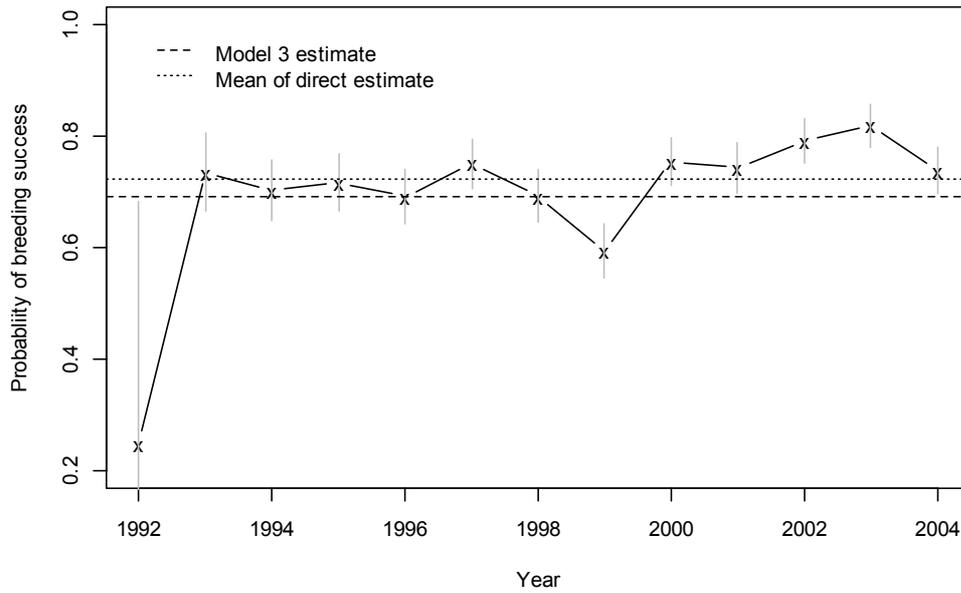
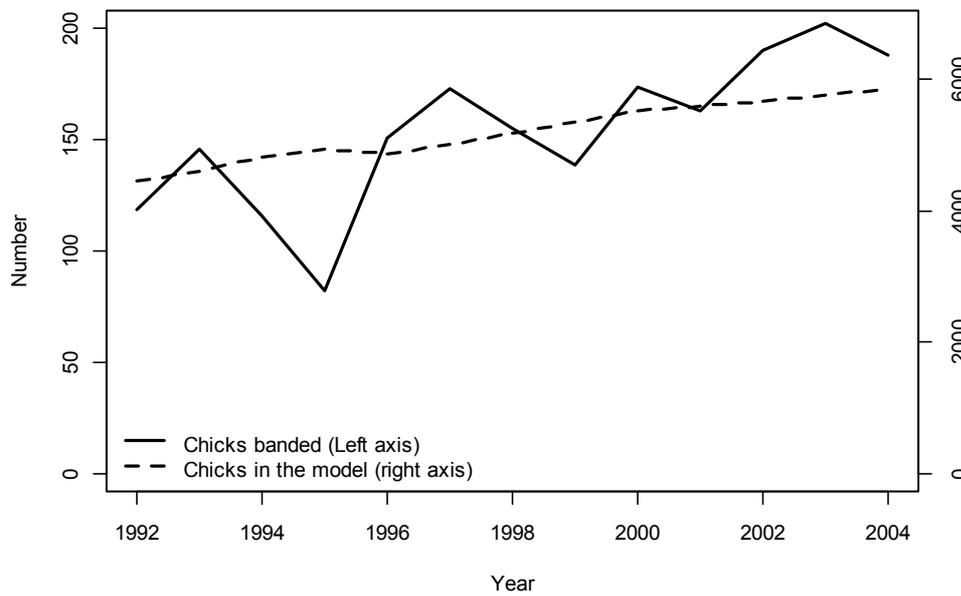**Figure 12: Comparison of direct estimates of breeding success ('x', with 95% confidence intervals as vertical bars) with that from Model 3.**

Figure 13: Comparison between the numbers of chicks banded in the study areas and the estimated total number of chicks from Model 3.
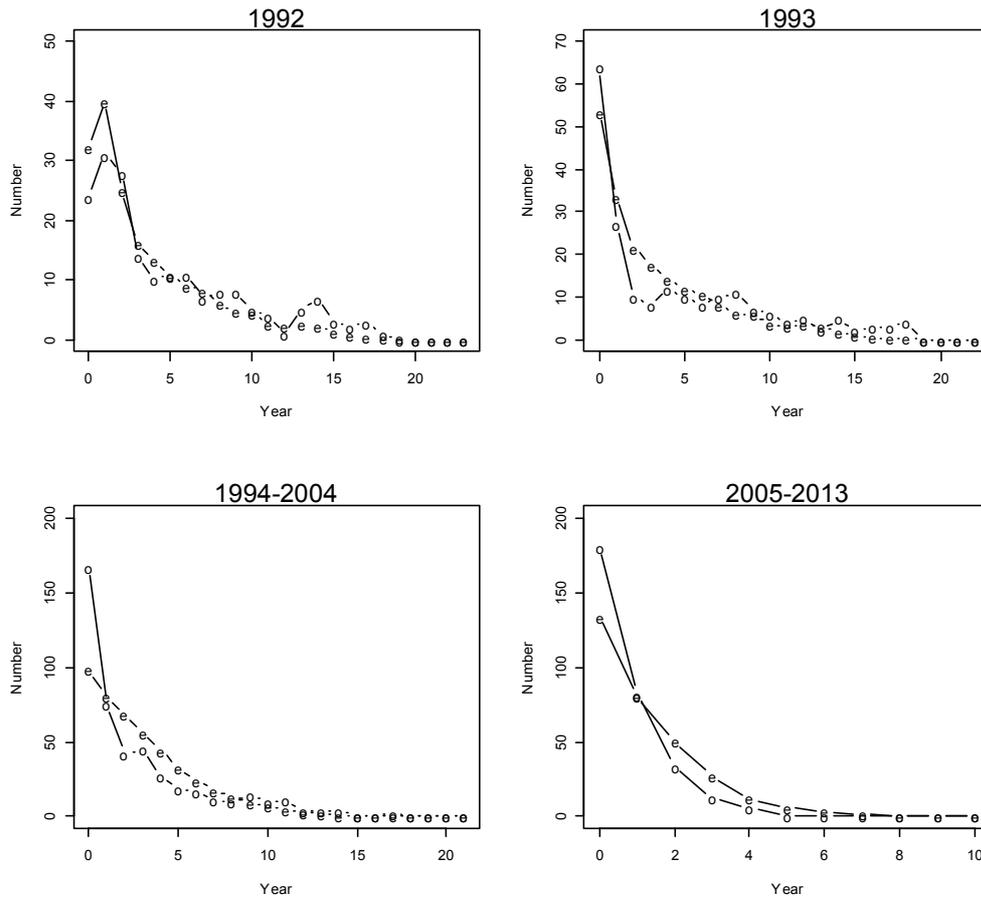
Figure 14: Expected and observed frequency distributions for the number of consecutive years of observed breeding following banding for birds banded as breeders in 1992, 1993, 1994–2004, and 2005–2013.
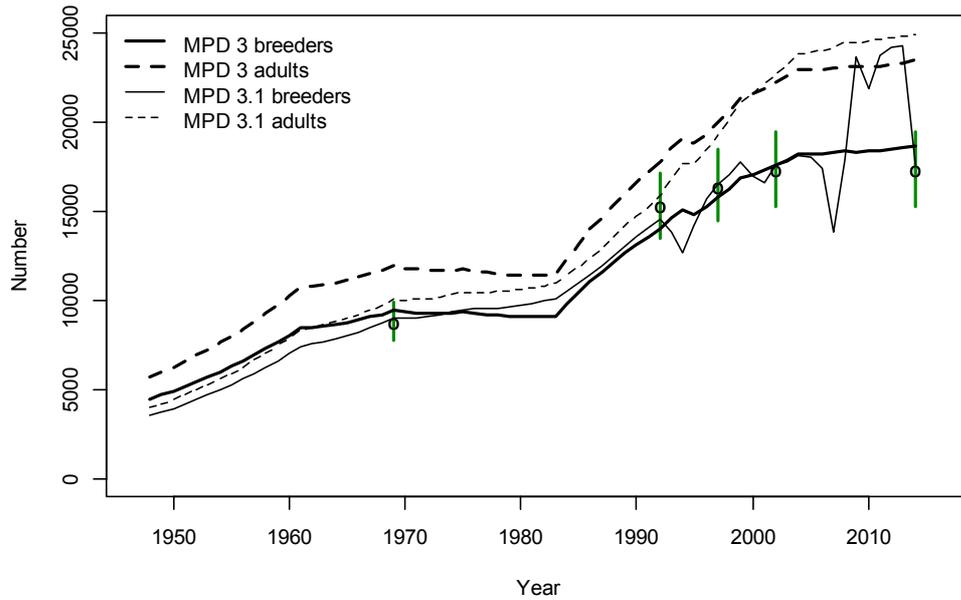
Figure 15: Comparison of estimates from Model 3 (heavy lines) and 3.1 (light lines) of numbers of breeders (solid lines) and adults (broken lines) in each year. Also shown are the census observations of numbers of breeders, with assumed 95% confidence intervals (vertical lines).
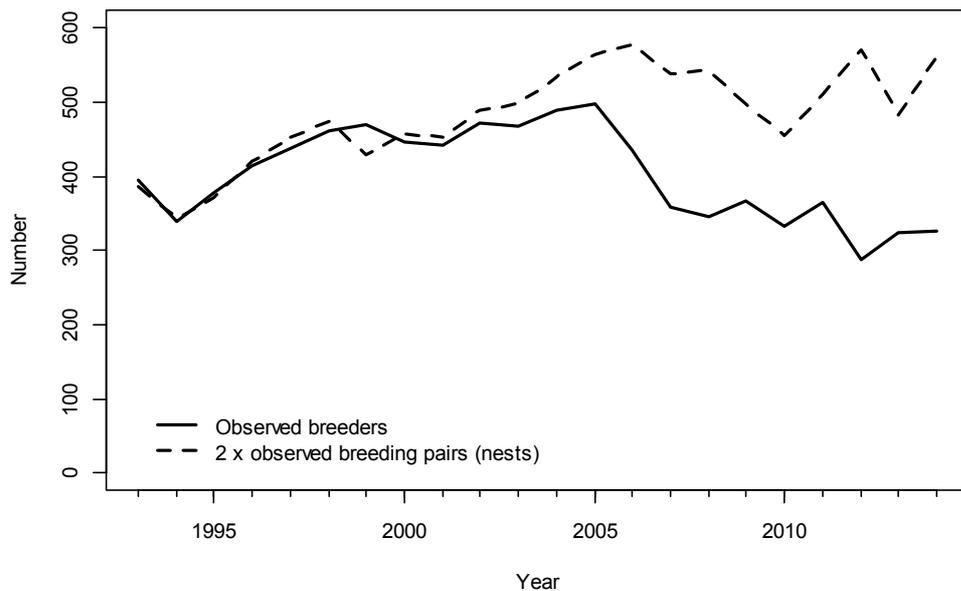

Figure 16: Total numbers of breeders (solid line) observed each year in the three study areas. Also shown (as a dotted line) is another estimate of the numbers of breeders obtained by doubling the inferred number of breeding pairs from model 3.
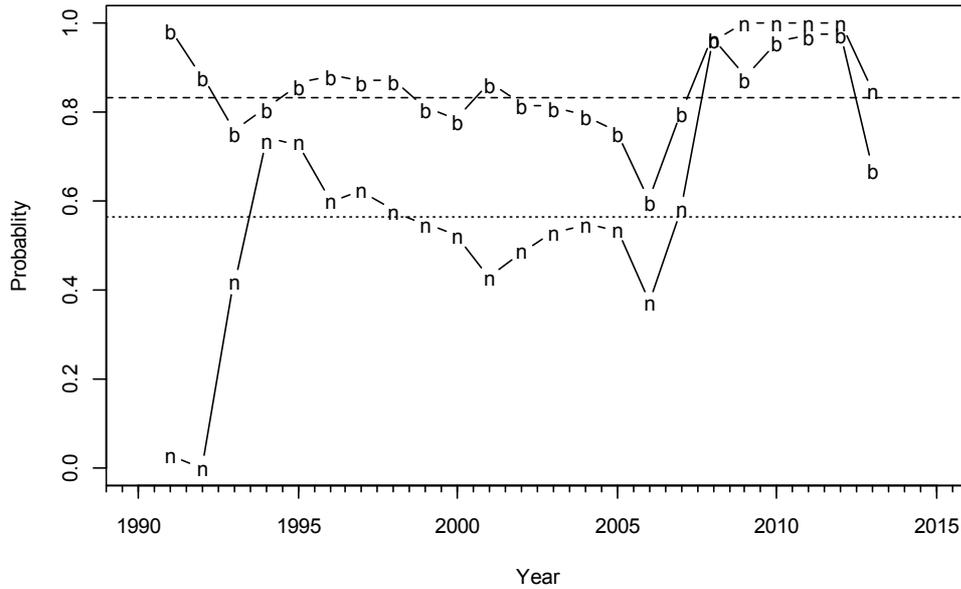
Figure 17: Estimated transition probabilities (annual probability of breeding given breeding status the previous year) from Model 3.1. Also shown, as horizontal dotted lines, are the estimates from Model 3, in which these parameters do not vary from year to year.

Table 14: Comparison of selected parameter estimates from Model 2 and Model 3.

	Model 2	Model 3
N0	–	5713
Pbrstd	0.069	0.235
$P_{surv, juv}$	1.000	0.902
$P_{surv, adu}$		
(1948–60)	0.896	1.000
(1961–68)	0.957	0.955
(1969–74)	0.921	0.924
(1975–82)	0.907	0.912
(1983–85)	0.930	0.989
(1986–91)	0.912	0.977
(1992)	0.990	0.989
(1993)	0.973	0.973
(1994)	0.932	0.931
(1995–98)	0.973	0.973
(1999–03)	0.951	0.950
(2004–07)	0.933	0.932
(2008–14)	0.928	0.924
Tbrbr	0.832	0.833
Tnbrbr	0.565	0.566
P1stbr(9)	0.064	0.046
oddmult	2.094	2.230
Psuccess	–	0.725
Mean age (1stbr)	11.7	11.7

3.3 Model 4 and 4.1

Models 4 and 4.1 used the same data as model 3 but incorporated different estimates of fishing-related mortality. Model 4 used estimated observable bycatch as in Table 7 and model 4.1 used potential fatality estimates by applying the cryptic mortality factors in Table 8. The effect of including bycatch data is to partition the survival estimates into fishery induced mortality and remaining "natural" mortality (assuming no other anthropogenic mortality).

The capture data were organised into three groups of fisheries –trawl, surface longline, and bottom longline. The estimated captures are higher for the SLL than for the BLL. For SLL, captures were relatively high between 1999 and 2004, then almost halved since. Captures in trawl fisheries are variable and the 2005 estimate is more than double that of any other year. All three fisheries were assumed to catch only adults. Breeders and non-breeders (from both study and non-study areas) were assumed to be equally vulnerable, and model 4 assumed there to be no cryptic mortality.

The two models showed broadly similar trajectories, which was essentially the same population trajectory as model 3 (Figure 18). Model 4.1 shows a slight declining trend since the early 2000s. The only estimated parameters that changed appreciably were the post-2000 estimates of adult survival (Figure 19), because for these years this parameter now concerns the non-bycatch proportion of total mortality. The adult survival estimate for 2008–2014 increased by 0.014 when observable bycatch was included and by 0.036 if the potential fatality estimates were used.
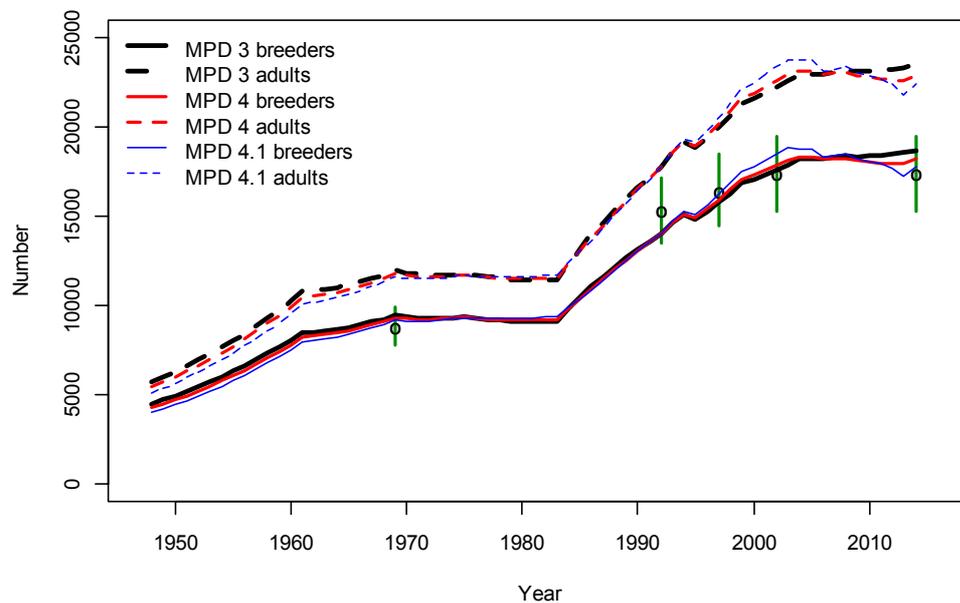

Figure 18: Comparison of estimates from Model 3 (heavy black lines), 4, (heavy red lines), and 4.1 (light blue lines) of numbers of breeders (solid lines) and adults (broken lines) in each year. Also shown are the census observations of numbers of breeders, with assumed 95% confidence intervals (vertical lines).
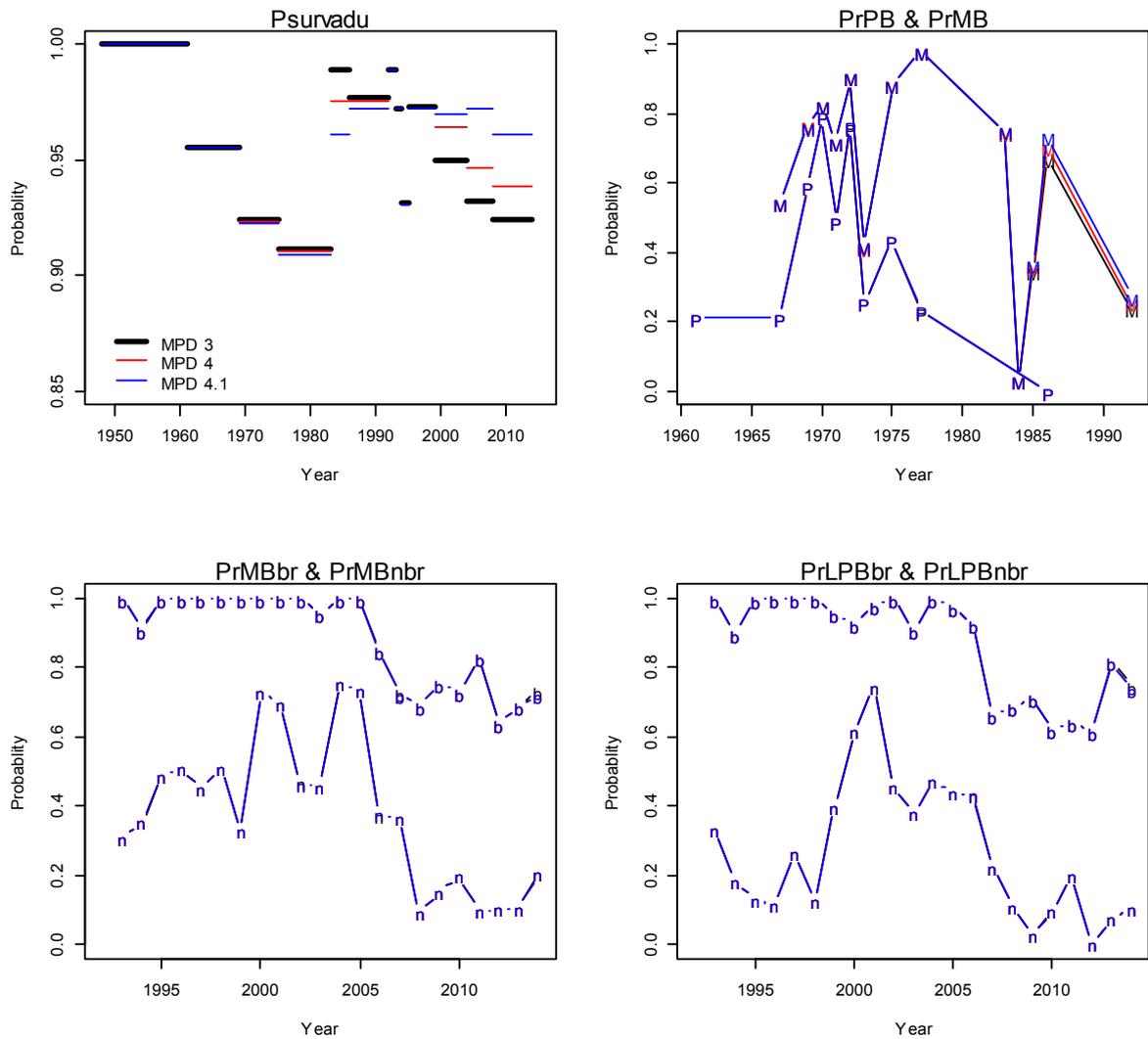
Figure 19: Comparison of survival and resighting probability estimates from models 3 (black), 4 (red), and 4.1 (blue).

3.4 Projection

A 10-year projection (2015–2024) of the population trajectory was carried out using model 3 (Figure 20). Demographic rates for future years were assumed to be the same as the most recent model estimates (e.g. survival of the 2008–2014 year block). To incorporate uncertainty in the parameter estimates in the projection, the parameters were drawn from a multinomial distribution with the mean assumed to be the MPD estimates, and the covariance assumed to be the inverse of the Hessian matrix at the MPD estimates. Three hundred samples were drawn and 300 projected population trajectories were obtained. The projections suggested that at the current demographic rates, the population is likely to increase by 5.7% in the next 10 years. Among the 300 projections, 285 predicted that the population size in 2024 will be greater than that in 2014.
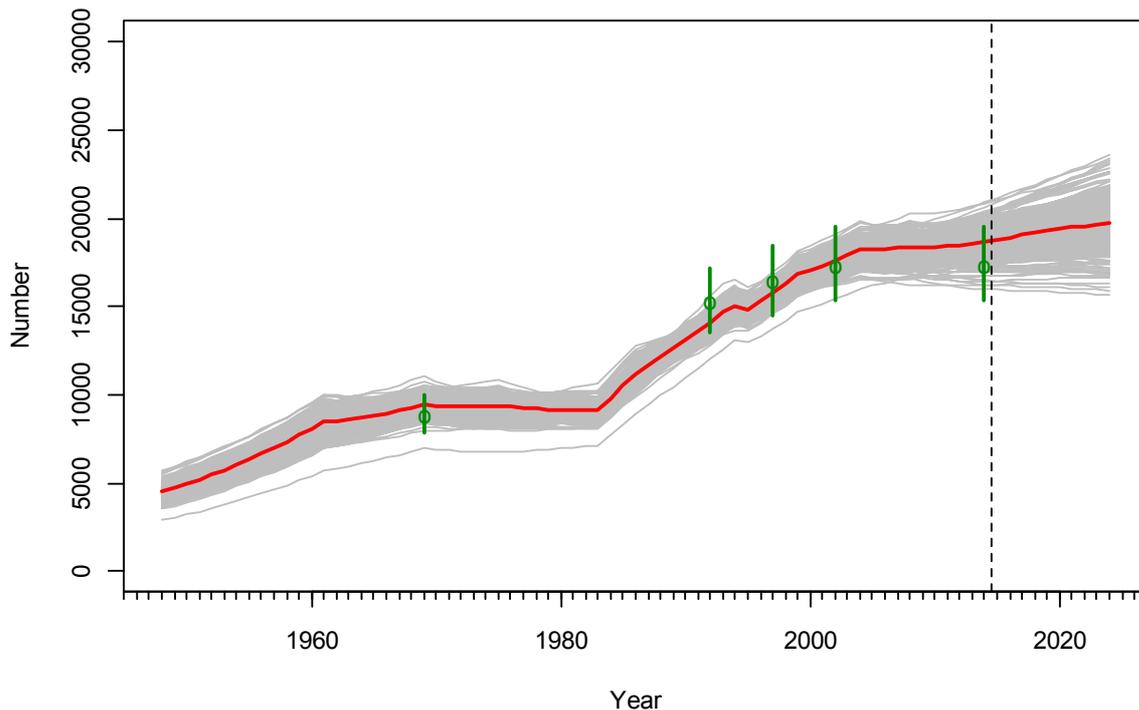
Figure 20: Predicted number of breeders from a 10-year projection (2015–2024) carried out for model 3. The red line is the projection using MPD estimates of parameters. Grey lines are based on parameters sampled from a multinomial distribution (see Section 3.4).

3.5 MCMC

Standard errors estimated via the inverse Hessian matrix using the maximum-likelihood method were very small (typically less than 0.01) and they appeared substantially to have underestimated parameter uncertainty. For some key parameters, 95% confidence intervals derived from likelihood profiles were 4–11 times wider than those inferred from the Hessian-based standard errors (Francis & Sagar 2012). Further investigation in the projection using arbitrarily inflated standard errors on estimated parameters resulted in implausible population trajectories (contradictory to census estimates), suggesting that true uncertainty of these parameters are likely to be small given the assumptions on the population structure and dynamics of SBA. Some preliminary investigations in estimating full Bayesian posterior distributions of parameters via MCMC simulations were conducted, and the results are given in Appendix C.

4. DISCUSSION

This work updates the quantitative modelling of the Snares Islands population of southern Buller's albatross, using field observations collected since 2008. We updated models developed by Francis et al. (2008) that had varying levels of complexity. The updated models gave very similar estimates to the previous assessment and there were no apparent contradictions between the outputs of model runs undertaken in this assessment. Good fits to mark-resighting and census observations suggested that the model structures used were appropriate and the demographic rate estimates obtained were a good representation of information coming from the data. However, since 2006: resighting effort was reduced, information on breeding success was unavailable for most identified breeders and the number of non-breeders was likely to have been overstated. This may have caused the estimated breeding probability to be lower than the true value.

The adult population was estimated to have doubled over the last 60 years, although this growth now appears to have stopped. The adult survival rate is likely to have declined since 1990, but for the most recent years (2008 onwards) has increased slightly. We note that, since adult survival was the only year-varying demographic parameter (in all model runs except 3.1), a decline in adult survival was the only possible demographic explanation for the changing population trajectory inferred by the latest census. Changes in resighting methods after 2006 precluded meaningful year-varying estimates of breeding parameters in recent years and so their potential effect on changing population was not properly investigated. However, demographic rate estimates were relatively insensitive to fitting to census data (compared with just mark-recapture data) and a small decline in adult survival was well supported by the integrated model.

There seemed to be a possibility that this falling survival rate may be an artefact caused by a combination of the dramatic drop in resighting effort in recent years and the well-known confounding between resighting probabilities and survival rates. A simulation experiment was devised to investigate these possibilities, which simulated resighting observations assuming reduced resighting effort in later periods (Francis et al. 2008). The results from this experiment showed that the drop in resighting effort did not appear to induce a strong trend in survival estimates. Further, any trends arising from random patterns in the data were typically small. We conclude that our estimated fall in survival rates since 1992 is likely to be real.

The early mark-recapture data were restricted to a maximum of 15 years of resighting as potential band loss in later years might have biased survival estimates. The evidence of potential band loss was implied by the gap between observed and predicted number of resightings for the early data, particularly for some banding sequences (Francis et al. 2008). However, it could also be that survival declines with increasing age. Unfortunately, it is very difficult to distinguish between the two hypotheses (band loss and age-dependent survival) with the data currently available for SBA. In the 1960s, 70s and 80s most of the considerable effort was put into checking the MB study area, with occasional checks for banded birds in PB. Usually there was a different observer each season, but the time interval between visits to the island were variable (part of the Buller's breeding season only). However, there was no specific study of Buller's in 1983–1987 and resighting effort was far less than in previous years.

Juvenile survival is notoriously difficult to estimate in albatross species, because it is strongly confounded with relocation parameters. Although adults show very strong fidelity to an area once they start breeding, chicks do not always breed in the area in which they were hatched (Sagar & Warham 1998) If a bird has not been resighted since it was banded as a chick, and it is much more than 14 years since banding, then we can be confident that one of two things has happened: mortality (the bird has died) or relocation (it has chosen to breed in an area that is rarely or never observed). However, it was not possible to distinguish between these explanations.

The base model did not incorporate estimates of fishing-related mortality, although the survival probabilities estimated by the base case model implicitly accounted for deaths from all causes including fisheries. Capture data appeared to be of little or no use in estimating trends in abundance, which were driven by estimates of total (natural plus fishing) mortality from the mark-recapture data. The capture estimates do allow us to estimate how much of that mortality is related to fishing, and therefore estimate "natural" mortality. The model estimates by Richard&Abraham (2013a) have associated uncertainties (see Table 7), but these uncertainties are not currently accommodated by SeaBird, which treat the bycatch observation as accurate estimates, which is unlikely to be true. Therefore SeaBird will underestimate the uncertainty of the natural survival accordingly. The observable capture estimates used in model 4 included birds from the Solander Island population which is approximately half the size of The Snares population. However, there are some limitations associated with these estimates: (1) captures apply to the whole EEZ and individuals breeding on Snares and Solander cannot be differentiated; (2) estimates of SBA captures are likely to have

included some Northern Buller's Albatross due to difficulties with on-board differentiation of these from SBAs; (3) there are no estimates prior to 1998–99 because earlier observer data are not considered to be reliable, and no estimates were made for other methods (such as potting, set netting, trolling, or purse seining) as observer coverage in these fisheries has generally been low; (4) the on-board capture records (on which the model estimates are based) may under-represent the total number of fishing-related fatalities, as entangled or hooked seabirds may not be retained when gear is hauled, but may be lost before being brought on deck (Richard & Abraham 2013a); (5) these estimates do not consider the demographic consequences of those captures for the seabird populations and they are assumed to represent the fishing induced mortality in this report.

Estimates of total fishing-related mortality (incorporating cryptic mortality) were also considered (model 4.1), but these estimates are highly uncertain (Richard & Abraham 2013b). Estimation of cryptic mortality was primarily based on two studies (Watkins et al. 2008; Brothers et al. 2010) that were conducted in fisheries in South Africa and Australia, respectively. The study of cryptic mortality in trawl fisheries was based on results from a single trip (Watkins et al. 2008), and so is very limited. Moreover, estimation of cryptic mortality only considered birds that were killed but not brought on-board the vessel. It did not include possible problems such as seabird bycatch not being reported when the observer is off duty, or seabird carcasses not being seen by the observer because of the volume of catch (Richard & Abraham 2013b). The Aquatic Environment Working Group (AEWG) suggested using the annual potential fatality estimate (scaled by fishing effort and population size) as an alternative estimate of cryptic mortality (rather than applying the cryptic mortality multiplier to the observable estimate as in model 4.1) in future studies. This would allow the model to incorporate fishing induced mortality prior to 1998–99 when there is no reliable estimate. This approach can be further extended to construct "scenario analyses" to investigate the effect on population from various levels of fisheries mortality. In addition, the potential effect on the change of catchability from various sorts of bycatch mitigation measures (Richard & Abraham 2013a) could be investigated in projection analysis.

Fishery discards are an important component of the diet of chicks (Francis & Sagar 2012), but it is not known whether the associated positive effect on population growth (e.g., from increased breeding success) is greater or less than the negative effect of bycatch (incidental mortality of seabirds caused by fishing) (Francis & Sagar 2012). This aspect of the effect of fishing was not considered.

The models examined in this report showed broadly similar population trajectories. The base model showed a reasonably flat trend since 2000, and model 4.1 estimated a slight declining trend over the same period (which appears to indicate that fishing-related mortality was possibly high). Because the estimates of fishery captures (including cryptic mortality) are unlikely to be accurate, and model 4.1 did not separate the fishing-related mortality from natural mortality before 1990, projections using this model will be highly uncertain. However, based on the overall trend in the estimated population trajectory and key demographic rates (e.g. adult survival, breeding probability) from the base model, we believe that the fisheries risk to the viability of this population over the last 60 years appears to have been small.

5. ACKNOWLEDGMENTS

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APPENDIX A: CV ESTIMATION FOR CENSUS OBSERVATIONS

To use the census counts in SeaBird we need to have some idea of how precise they are. What follows is an attempt to estimate approximate coefficients of variations (CV) for the census counts.

In constructing a census count, C, of breeding pairs in a specified area the area was divided into n subareas in which ground counts were done, and a further m subareas in which counts were from vantage points, using binoculars. Thus, we may write $C = \sum_{i=1}^{n} G_i + \sum_{j=1}^{m} V_m$, where the G_i are ground counts and the V_j are vantage-point counts.

Each V_j was the mean of two independent counts by different people. When the two counts differed by more than a specified amount (10% for counts greater than 100; 5% for smaller counts) they were repeated until the difference was less than this amount. Typical differences between pairs of vantage-point counts were 5–6% for counts greater than 100, and 2–3% for smaller counts. These typical percentage differences were used to calculate approximate standard errors (s.e.s) for the total vantage-point counts as follows.

Denote the individual counts in sub-area j as V_{j1} and V_{j2} [so $V_j = 0.5(V_{j1} + V_{j2})$] and assume that the V_{jk} were approximately normally distributed with mean T_j (the true count in sub-area j) and CV, c_j, where $c_j = \gamma_1$ when $T_j < 100$ and $c_j = \gamma_2$. Therefore, the s.e. of the total vantage-point count is $S = \left[\sum_j 0.5 c_j^2 T_j^2 \right]^{0.5} \approx \left[\sum_j 0.5 c_j^2 V_j^2 \right]^{0.5}$. Now, $V_{j1} - V_{j2}$ is normally distributed with mean 0 and s.d. $2^{0.5} c_j T_j$, which is approximately equal to $2^{0.5} c_j V_j$. Thus the median absolute difference, $|V_{j1} - V_{j2}|$, is approximately equal to $0.6745 \times 2^{0.5} c_j V_j$ (0.6745 is the upper quartile of the standard normal distribution). If we assume that the 'typical differences' quoted in the previous paragraph are medians, and, to be conservative, use the upper values of the quoted ranges (i.e., 3% and 6%), we can estimate $\gamma_1 = 0.03 / (0.6745 \times 2^{0.5})$ and $\gamma_2 = 0.06 / (0.6745 \times 2^{0.5})$. Substituting these estimates, together with the recorded values of the V_j, into the above equation for S, we obtained the s.e.s in Table A1.

Table A1: Total counts and total vantage-point counts, with estimated s.e.s for the latter, from 2014 census of breeding pairs on the Snares. The total counts were taken from table 3-1 of Sagar (2014).

	Total count	Vantage count	s.e.
N.Prom	1 599	327	1.8
W.Coast	146	138	1.1
NofSW.Prom	551	590	3.1
SofSW.Prom	1 288	484	3.8
S.Coast	2 425	918	9.7
E.Coast	1 733	314	4.9
Alert.St	305	305	8.4
BroughtonI	657	79	NA
AllSnares	8 704	3 047	32.7

APPENDIX B: CALCULATING BREEDING PROBABLITY FOR CONSECUTIVE YEARS

We described how to calculate the expected number of birds that are observed breeding for consecutive years after banding. We count how many consecutive years after banding each bird was observed breeding. For example, for birds tagged as breeders in 1992, the count was 0 for those birds not observed breeding in 1993, 1 for those birds observed breeding in 1993 but not in 1994, and so on up to a maximum count of 22 (for birds observed breeding every year from 1993 to 2014). The expected number of birds with a count of n was calculated as $E_n = m_{\mathrm{MB}} P_n^{\mathrm{MB}} + m_{\mathrm{UPB}} P_n^{\mathrm{UPB}} + m_{\mathrm{LPB}} P_n^{\mathrm{LPB}}$ where m_{COL} is the number of birds banded as breeders in year y in area COL P_n^{COL} is the probability that such a bird will have a count of n, given by

$$P_n^{COL} = \begin{cases} 1 - X_y^{COL} & n = 0 \\ X_y^{COL} ... X_{y+n}^{COL} \left(1 - X_{y+n}^{COL}\right) & 1 \le n < 22 \\ X_y^{COL} ... X_{y+n}^{COL} & n = 22 \end{cases}$$

$X_y^{\mathrm{COL}} = \mathrm{Psurv\_adu}_y \, \mathrm{T}_{\mathrm{brbr}} \left(1 - \mathrm{Pr}_{y+1}^{\mathrm{COL}}\right)$, and $\mathrm{Pr}_y^{\mathrm{COL}}$ is

$X_y^{COL} = P_{surv,adu_y} \mathrm{T}_{\mathrm{brbr}} \left(1 - \mathrm{Pr}_{y+1}^{\mathrm{COL}}\right)$, and $\mathrm{Pr}_{y+1}^{\mathrm{COL}}$ is the appropriate resighting probability (either PrMBbr of PrLPBbr).

APPENDIX C: PRELIMINARY INVESTIGATIONS ON CONDUCTING MCMC

Models runs 2, 3, and 4 were conducted at the MPD phase. The initial attempt to estimate Bayesian posterior distributions of parameters via MCMC simulations for the base model (Model 3) was not successful as the MCMC samples of estimated parameters stuck at their MPD estimates. The causes of this are not exactly known, though initial investigations suggest that it relates to the way that SeaBird calculates the initial proposal distribution used for MCMC sampling, which can become problematic when there is a large number of estimated parameters (e.g. when all the resighting probabilities were allowed to vary annually).

The number of estimated parameters was reduced by reducing the length of the resighting probability vectors (model 5). For the pre-1992 mark-recapture data, the resighting probability PrMB (for Mollymawk Bay) and PrPB (for Punui Bay) were assumed to be the same for all years. For the post-1992 data, the resighting probability was estimated only for two distinctive periods: 1992–2005 and 2006–2014 for Mollymawk Bay, and 1992–2006 and 2007–2014 for Lower Punui Bay. This is based on the trend in the estimated annual resighting probabilities from Model 3 (see figure 11 of Fu & Sagar 2015). Estimates of adult survival probabilities were marginally different comparing model runs 3 and 5 (Figure C1).

A preliminary MCMC run was conducted using the revised model (model 5). The MCMC simulations were successful with 5500 samples taken from 550 000 iterations. A number of parameters had poor posterior traces (Figure C2), including: oddsmult (this may be due to the small number of birds resighted first breeding at the study sites, or confounding estimation of maturation parameters); and juvenile survival (this parameter was confounded with Pbrstd). However, the key demographic parameters such as adult survival and breeding probabilities appeared to be well estimated (Figure C3), and the 95% credible interval of their posterior distribution was narrow (Figures C4 and C5).
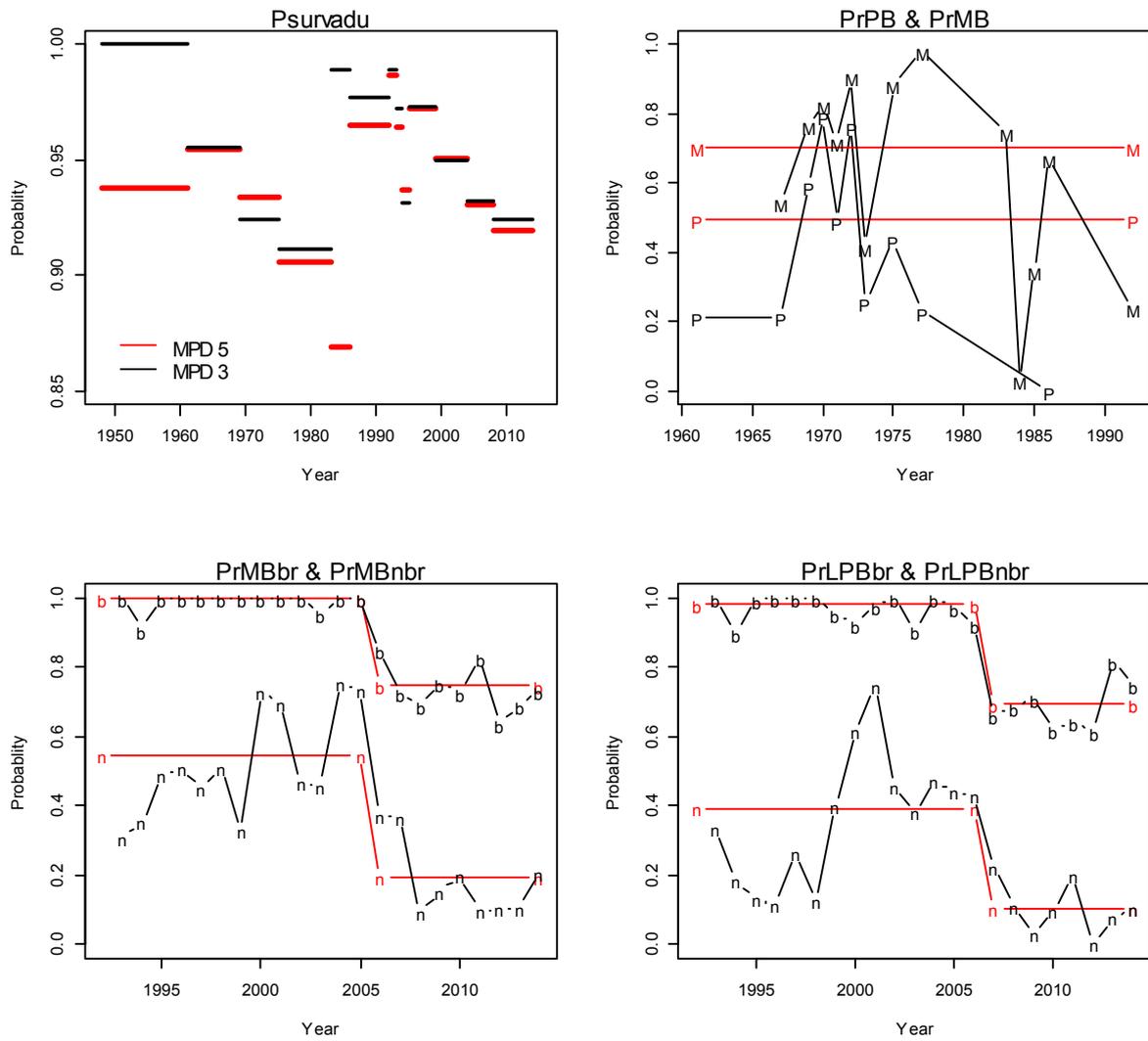
Figure C1: Comparison of estimates of the survival and resighting parameter estimates that are common to models Model 5 (red) and Model 3 (black).
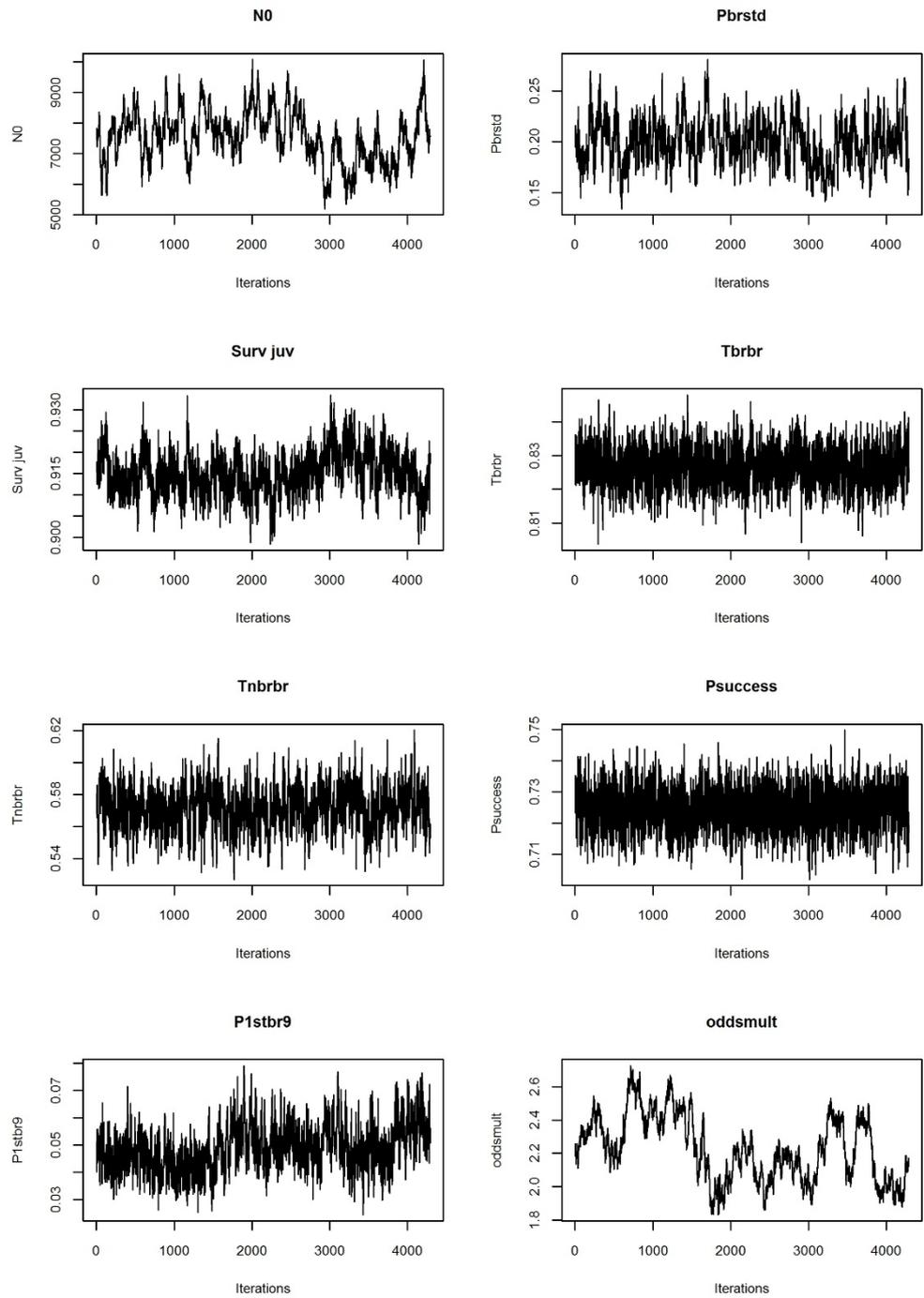
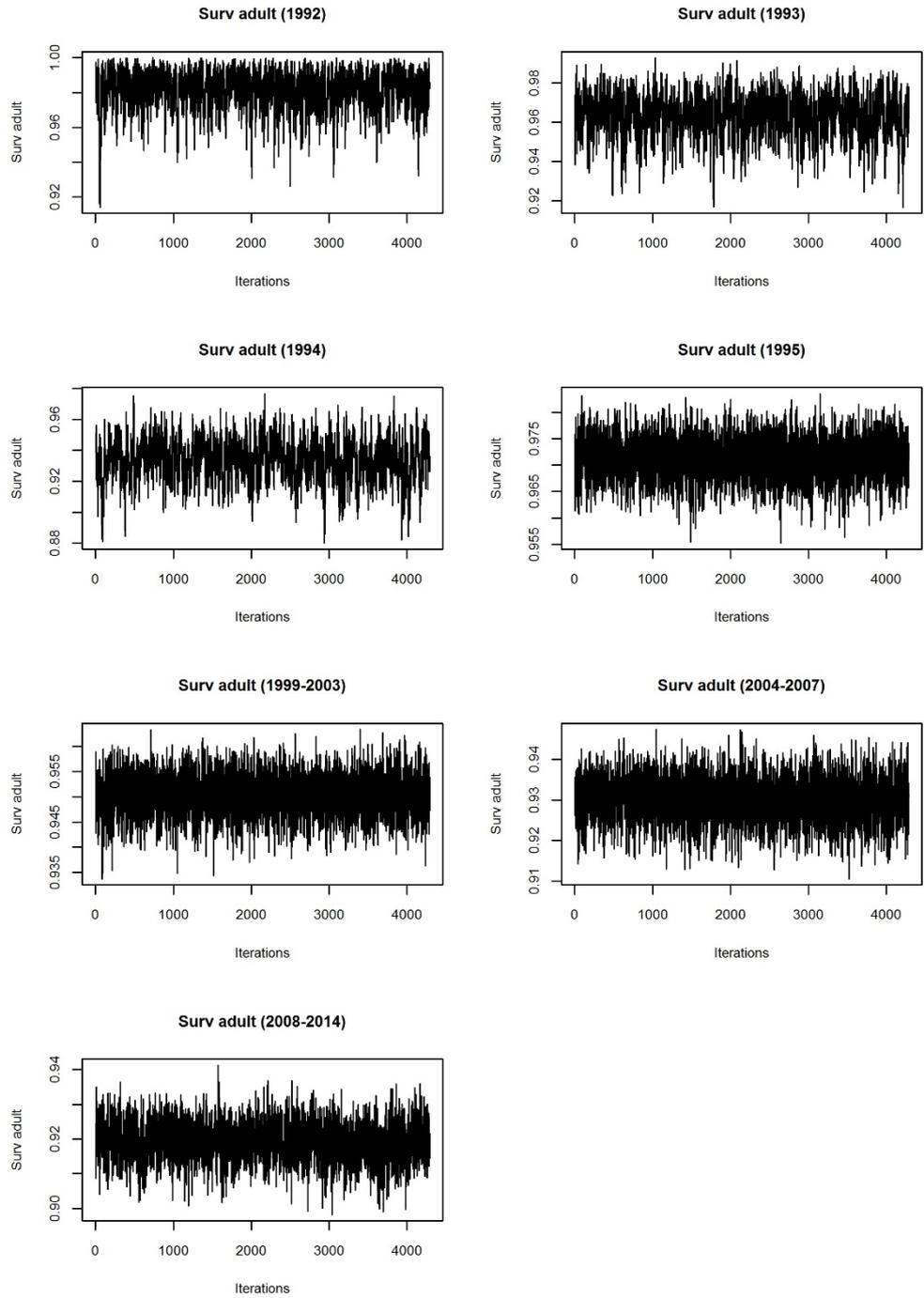**Figure 2: Traces of estimated parameters from MCMC for model 5.**

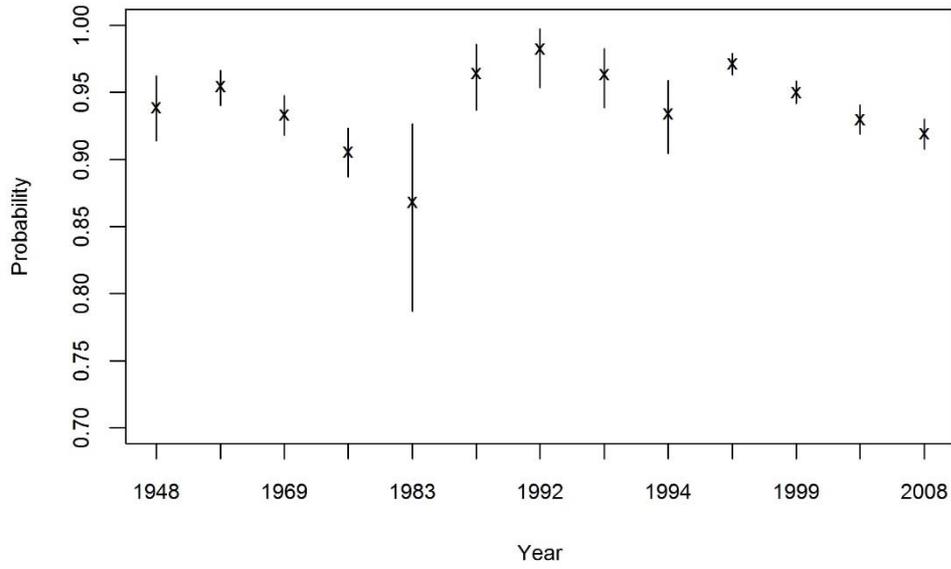**Figure C3: Traces of estimated adult survival parameters from MCMC for model 5.**

Figure C4: Median and 95% CI of the posterior distribution of estimated adult survival from MCMC for model C5.
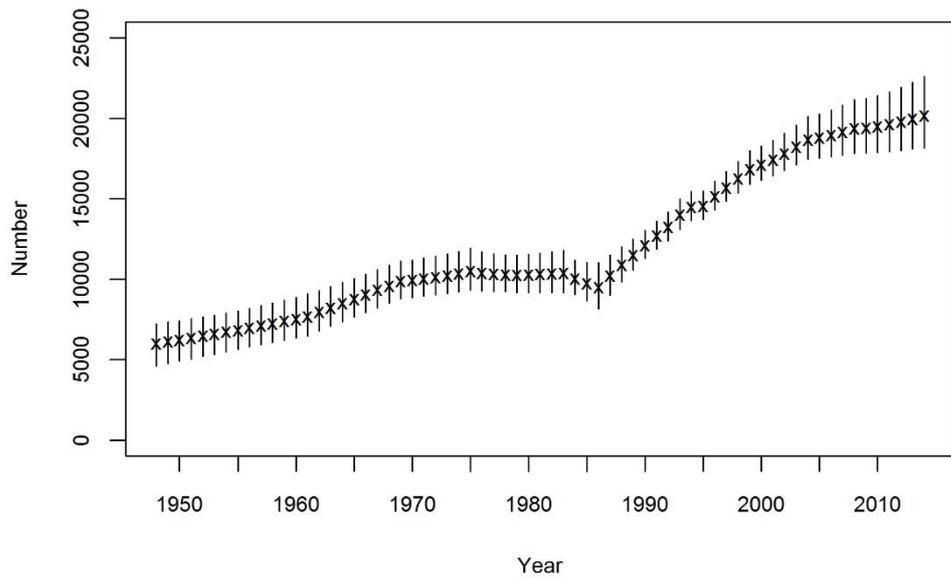

Figure C5: Median and 95% CI of estimated number of annul breeders from MCMC for model 5.