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A Statistical Assessment of the Status and Trends of Antarctic and Sub-Antarctic Seabirds

A STATISTICAL ASSESSMENT OF THE STATUS AND TRENDS OF ANTARCTIC AND SUBANTARCTIC SEABIRDS



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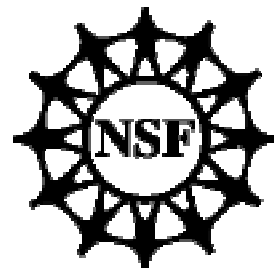
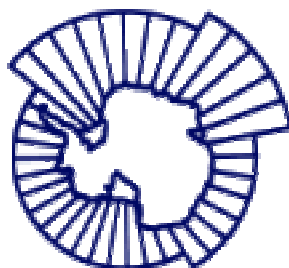
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ABSTRACT

A workshop held under the auspices of SCAR, CCAMLR & NSF in Montana in May 1999 examined statistically all available long-term population data for Southern Ocean seabirds. For the workshop, long-term was defined as either: i) continuous for more than 10 years, or ii) discontinuous for more than 10 years, but with more than 50% coverage. A total of 61 long-term data sets for 25 taxa was examined. Species and regional syntheses were undertaken where data permitted.

Significant decreases in populations are evident for those species known to be caught on longline fisheries (albatrosses, Southern Giant Petrels and *Procellaria* spp.). Substantial changes were noted for many of the penguin populations examined, but these varied in terms of degree and direction among species and geographical areas, so that no completely consistent overall pattern emerged. However, decreases in all penguins other than King Penguins in the Subantarctic was a feature of the data.

For some species of unfavourable conservation status (BirdLife International 2000) very limited, if any, data are available for assessment of population trends (eg Tristan Albatross *Diomedea dabbenena*, White-chinned Petrel, Spectacled Petrel *P. conspicillata* and Grey Petrel *P. cinerea*). Other species with notably inadequate data include Light-mantled and Atlantic Yellow-nosed Albatrosses.

No time-series population data appear to exist for Antarctic Petrels, a species that is endemic to the Antarctic continent and adjacent waters. Other characteristic Antarctic species for which there is little current population monitoring include Cape and Snow Petrels and Southern Fulmar.

Priorities for collateral data to assist future assessments, for other analytical techniques and for additional data are indicated.

INTRODUCTION

Since the 1980s, the Scientific Committee on Antarctic Research (SCAR) through the Bird Biology Subcommittee (BBS) of the Working Group on Biology (WGB), has provided advice and information on the status and trends of Southern Ocean seabird populations to the Scientific Committee of the Commission for the Conservation of Antarctic Marine Living Resources (SC-CAMLR) (SCAR Bird Biology Subcommittee 1988, SCAR 1992, Woehler & Croxall 1997). These three reviews were largely qualitative but provided initial broad-scale summaries of known population trends. This is the fourth review of the status and trends of Antarctic and Subantarctic seabird populations produced by the SCAR BBS for SC-CAMLR and the first to provide comprehensive statistical assessments of available population data.

In response to a request in 1998 from SC-CAMLR for a statistical assessment of available population data, the SCAR BBS organised a Southern Ocean Seabird Population Workshop to examine statistically long-term population data for Southern Ocean seabirds. The Workshop was convened by EJ Woehler (Australian Antarctic Division), hosted by WR Fraser and DL Patterson (Montana State University), and was held 17-21 May 1999 at Bozeman, Montana, USA. Researchers holding long-term seabird population data sets (see Methods, below) were invited to contribute their data to the Workshop and participate in the analyses and interpretation. A list of participants is given in Appendix 1.

The present review was prepared at the May 1999 Workshop, tabled for review at the SCAR BBS meeting in Tokyo, Japan, in July 2000 and submitted to CCAMLR in 2001.

METHODS

i. Data Sets

A planning meeting was held at the 7th SCAR Symposium on Antarctic Biology in Christchurch, New Zealand in August 1998, where many holders of long-term data sets developed a suite of criteria to be used in selecting those data to be analysed at the Workshop. These criteria were:

- a. Continuous annual data extending for more than 10 years
- b. Discontinuous annual data series extending for more than 10 years, but with more than 50% of years covered
- c. Data of sufficient quality to be used as indicative of trends
- d. Other data available

The Australian Antarctic Data Centre (AADC) developed and hosted a password-protected web-based data base for the Workshop, similar to the web-based data base developed for the CCAMLR Area 48 Workshop in 1998 (Anonymous 1998). At the time of the Montana Workshop opening, more than 1300 records had been submitted for 23 species at 65 localities. Not all submitted data met the pre-determined criteria, and statistical analyses were confined to only those data sets that met criteria a and b.

ii. Data Analyses

The analyses sought to identify whether there was a statistically significant trend in the data. A trend was defined to be an overall linear increase or decrease in the data set. Linear regression was used to evaluate the trend. Because the data were counts, the specific statistical model used was a generalized linear model with a Poisson error structure (ie Poisson regression) (McCullagh & Nelder 1989). We also checked to see if the specific error structure affected the results by using models with Normal and Poisson with over-dispersion error structures for selected data sets. The conclusions did not change and therefore only the results based on models with a Poisson error structure are presented. We used residual analysis to evaluate how well the linear model fit the data and to identify potential outliers and influential points. Potential outliers and influential points were removed and analyses rerun to see if conclusions substantially changed. Significance of the model was calculated and we used percentage deviance explained to summarize the overall fit of the model to the data. Significance was assessed at an alpha of 0.05.

There were obvious non-linear trends in the data sets. To understand these patterns, a variety of methods were suggested, especially polynomial regression, piece-wise linear regression with a single breakpoint (PLR), and generalized additive models (GAM) (Hastie & Tibshirani 1990), all of which were used in an exploratory fashion. Polynomial regression is a straight-forward extension of the linear model used for trend analysis. In all but a few cases, a polynomial of order 2 was fitted. Significance of the model and proportion deviance explained was calculated. Piece-wise linear regression is a technique to find a point where the slope changes in the trend line. Only a single change point can be found and the algorithm used (see below) is done in the context of a Gaussian error structure. Therefore all counts were log-transformed for analysis. Significance of the model and proportion variance explained was calculated. Generalized additive models are an extension of generalized linear models where a smoother is used to characterize non-linear patterns. This technique can show patterns that are not easily modelled with polynomials. This technique has a tendency to overfit the data, particularly where sample sizes are low (ie it follows the data points too closely). In all cases, we used a Poisson error

structure. We felt this modeling technique was most reliable if there were at least 10 counts in a data set. For these data sets, we used the default smoothing span used in the software package. If the default span did not produce a model that converged, then the sample size was deemed inadequate. For species with a single sparse data set (ie less than 10 data points), we used the technique in an exploratory manner and changed the default span to produce a model that converged; we noted this in Table 1 when this occurred. We used a loess smoother for the models. We tested whether the generalized additive model was an improvement over the linear model and calculated a "proportion deviance explained". In a sense, this is actually a measure of how closely the model follows the data (with a change in span, you can get the deviance explained to be 1 and the data points connected by lines or you can change the span so the model mimics the behaviour of the linear model). However, by using the default span rather than changing the span manually for the larger data sets, we avoided "forcing" a non-linear model onto the data. We then calculated the proportion deviance explained by the smoother (ie $1 - (\text{residual deviance} / \text{null deviance})$) and used it to help judge whether or not the generalized additive model was an improvement over the linear model. All analyses except PLR were performed in S+2000 (MathSoft 1999); the PLR was performed in Statistica 4.0 (StatSoft 1995).

In the text, %VE is used to represent the proportion of deviance explained (linear and polynomial generalized linear models), the proportion variance explained (PLR), and the proportion of deviance explained with a smoother (generalized additive model). We wanted to use the simplest model possible. Therefore, we judged the nonlinear models to be an improvement over the linear model if the %VE increased by at least 15% (and was significant at an alpha of 0.05 or less for the polynomial and additive models). Otherwise, the linear model was judged to be adequate for describing the trend seen in the data. For some species where sample size was low but were the only data available for a species, the nonlinear models were used in an exploratory fashion.

Time-series analyses of data sets were not undertaken, even though one or two data sets submitted to the Workshop may have been usefully analysed by such techniques. Workshop attendees decided to apply a common analytical approach to all data sets, and leave more detailed or specialised analyses such as time-series analysis to data holders. However, it was hoped that such analyses might be attempted for a subsequent assessment of status and trends.

iii. Reporting of results

The CCAMLR system for reporting years has been adopted in this report, ie 1998/99 is reported as 1999. Annual rates of change in populations reported here are compound rates derived from population sizes predicted from the linear regression models from all the data or from the separate linear regression models from PLR.

RESULTS

A total of 61 long-term data sets for 25 taxa was examined in the Workshop (Table 1). Accounts of population trends at each site where long-term data were examined are presented below, as are regional overviews where appropriate. A summary of all statistical results is given in Table 1. Although we did not specifically investigate the internal consistency of the data submitted, most, if not all, are derived from standardised long-term studies, within which achieving internal consistency and avoiding systematic bias has been a primary requirement. In the text we indicate those data wherein bias due to inconsistency in data collection might exist. We provide references to papers which include further details of the methodology relevant to the conduct of the study but note that this remains an issue that may need to be addressed in greater detail.

Species accounts

Emperor Penguin *Aptenodytes forsteri*

One data set from Pointe Géologie is available for a period of 45 years with yearly counts since 1963 (Fig. 1). The counts were obtained from counting eggs and chicks lost throughout the breeding season and adding the number of chicks fledged. The estimate of the breeding population was compared with oblique photographs taken at 300m above the huddles of incubating males. There was a decrease in the population with a significant break in the series in 1979 (Table 1). The population at Pointe Géologie decreased (0.9% per annum: PA) until 1978, then halved within five years and decreased since 1984 (1.2% annually). A prolonged decrease in adult survival related to a period of warming and of reduced sea extent was the major cause of the decrease (Barbraud & Weimerskirch 2001). Additional information on other populations is necessary to assess whether the trend of the Pointe Géologie population is representative. Future censuses should focus on counts of incubating males just after laying to allow comparisons.

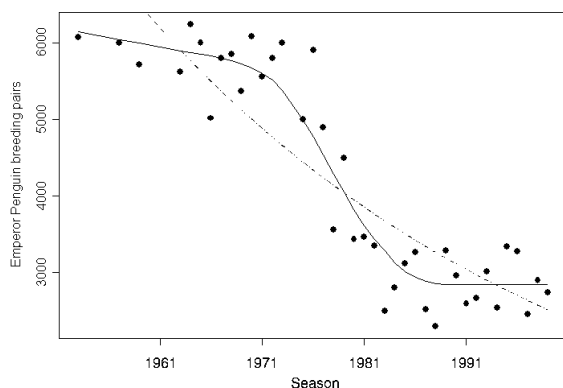


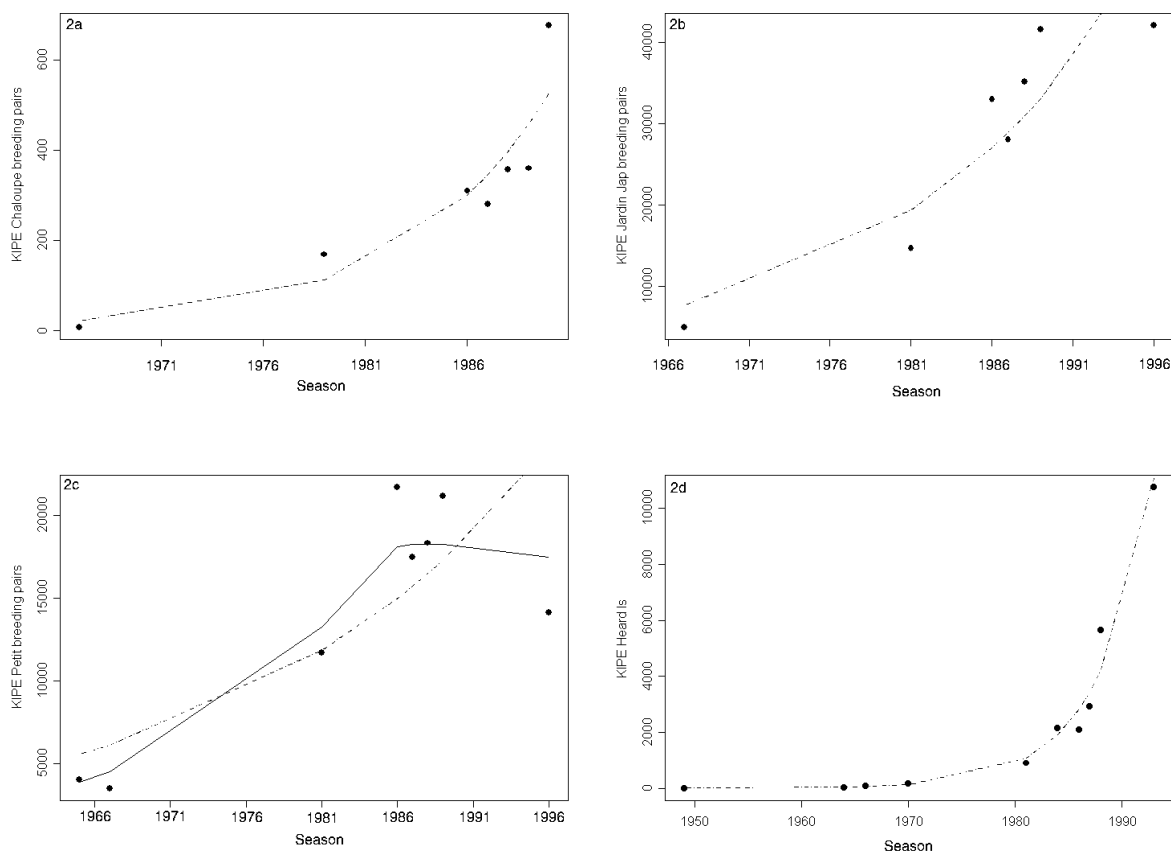
Figure 1. Plot of analyses of Emperor Penguin population data at Pointe Géologie. The continuous line is the GAM fit to the data, and the dashed line the linear regression fit.

Counts elsewhere did not meet the analysis criteria because of the inaccessibility of almost all Emperor Penguin colonies during the winter months. Exceptions are the winter censuses of incubating males at Auster and Taylor Glacier colonies (GG Robertson unpubl. data). Those censuses suggest a stable population from 1988 to 1999. Other accessible colonies where similar censuses could be accomplished are at Haswell I, Halley Bay and Cape Crozier. Several Ross Sea colonies have been

counted intermittently over the last 15 years (G Kooyman, unpubl. data). However, the counts have been of chicks only, and they were made only within a week of fledging; these counts are not comparable to the Pointe Géologie data.

King Penguin *Aptenodytes patagonicus*

Long-term data were only available from three colonies on Ile de la Possession (Iles Crozet) and from Heard I. Some data from Iles Kerguelen and Marion I were also examined (Table 1) but not plotted. All three colonies at Ile de la Possession have been surveyed intermittently since the early 1960s from oblique photographs. Surveys were conducted around 10 January, ie when the number of incubating birds approaches a maximum. Except for one large colony that was disturbed by the establishment of a base (Petite Manchotière) which shows no significant trend, all colonies have increased significantly, with larger colonies' populations possibly stabilising since the early 1990s. From the three undisturbed colonies with the highest number of counts, all show a linear increase (Chaloupe, Jardin Japonais, Petite Manchotière: +15.0%, +6.9% and +4.8%; with 88%, 83% and 66% VE, respectively) (Figs. 2a-c). The third colony (Petite Manchotière) may have stabilised in the early 1990s (Weimerskirch et al. 1992) (Fig. 2c). At Ile des Cochon (Iles Crozet), the colony of 500,000 pairs has shown similar trends to the colonies at Ile de la Possession (Guinet et al. 1995).



Figures 2a-d. Plots of analyses of King Penguin population data for a) Chaloupe, b) Jardin Japonaise, c) Petite Manchotière colonies at Ile de la Possession, Iles Crozet, and d) Heard I. The continuous line is the GAM fit to the data, and the dashed line the linear regression fit.

Populations on the eastern part of Iles Kerguelen have been counted from high- and low- altitude aerial photographs taken in early January. These populations are increasing at similar rates to those at

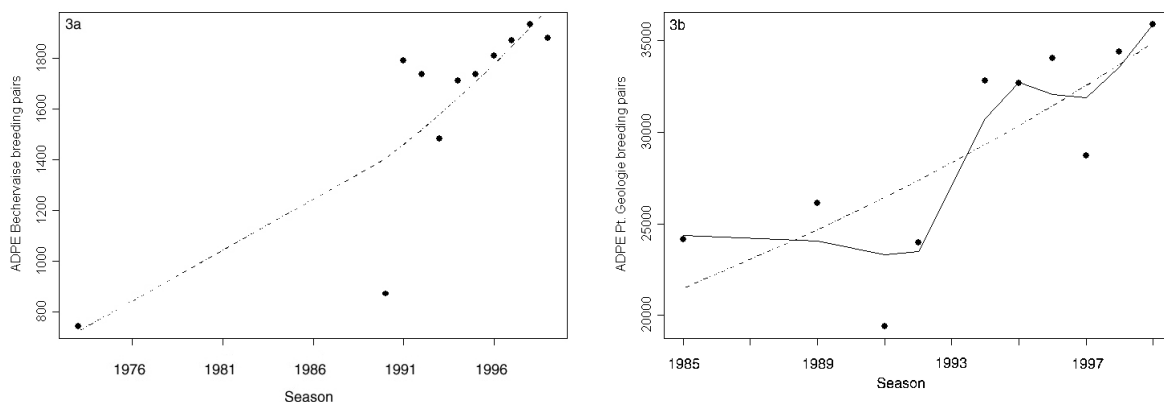
Iles Crozet, but the low number of counts prevent a determination of whether the population has stabilised in a manner similar to the Iles Crozet colonies (Weimerskirch et al. 1989). At Marion I, counts of chicks were carried out in September/October (Crawford et al. unpubl.). Due to large interannual variations in breeding success, no clear trend in population size could be determined in these data.

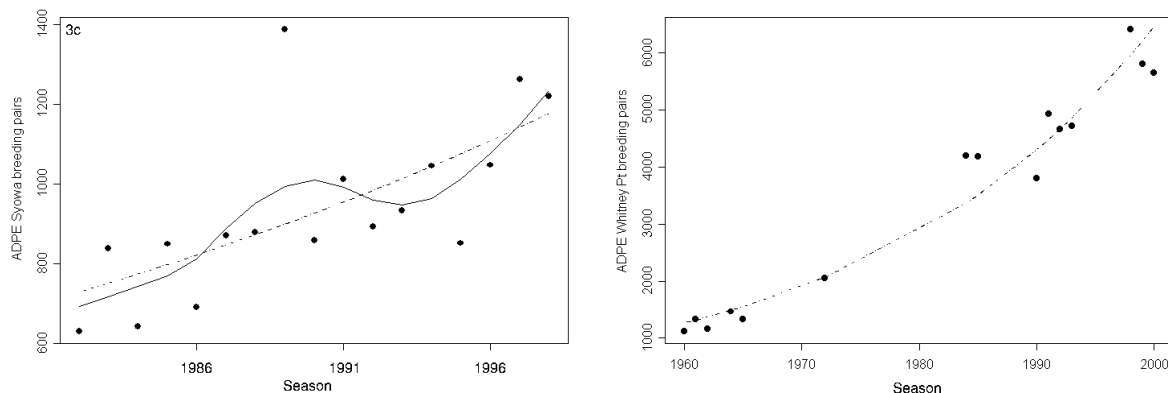
The King Penguin population on Heard I increased linearly between 1949 and 1993 (97.8% VE) (Fig 2d). There was no evidence of any nonlinearities in the data (Table 1). For other sites, only very patchy or anecdotal data are available; nevertheless almost all indicate continuing increases in populations (Woehler & Croxall 1997, Ellis et al. 1998). At present, there is no alternative hypothesis to that previously proposed, whereby these population increases are sustained by the enhanced availability of myctophids, the principal prey of king penguins at all breeding localities.

Adélie Penguin *Pygoscelis adeliae*

Long-term data for Adélie Penguins were available from localities in East Antarctica, the Antarctic Peninsula and the Ross Sea (Table 1). Four data sets for Adélie Penguins in East Antarctica were examined: Béchervaise I (Mawson), Pointe Géologie, Syowa, and Whitney Pt (Casey). The four data sets all showed persistent, long-term, statistically significant linear increases over the periods examined (Table 1 and Figs 3a-d). At Béchervaise I, the population has increased by 176% between 1973 and 1999 (71% VE), at an annual rate of 4%. At Pointe Géologie, the population has increased by 63% between 1982 and 1998 (53% VE), at an annual rate of 3.5% (Micol & Jouventin 2001). At Syowa, the population has increased by 62% between 1982 and 1998 (47% VE), at an annual rate of 3%, and at Whitney Point, the Adélie Penguin population increased linearly between 1960 and 2000 (95.7% VE).

Supplementary data were also available from the Mawson and Casey regions in East Antarctica. Breeding populations in the Mawson region increased between 1989 and 1994 (EJ Woehler unpubl. data), whereas the increases reported for the Windmill Is for the period 1959 - 1989 (Woehler et al. 1991) have continued (EJ Woehler, unpubl. data).





Figures 3a-d. Plots of analyses of Adélie Penguin population data at four localities in East Antarctica: a) Béchervaise I, Mawson, b) Pointe Géologie, c) Syowa, and d) Whitney Pt, Casey. The continuous line is the GAM fit to the data, and the dashed line the linear regression fit.

Within the Antarctic Peninsula region, the Admiralty Bay (King George I) Adélie Penguin population decreased significantly over the 1978 to 1998 period, with the linear model explaining 36% of the variance (Fig. 3e). Both non-linear models significantly improved the fit of the data (>58% VE), suggesting a pronounced nonlinearity to the time series. The PLR identified the breakpoint at 1990 (68% VE). The population increased by 16% between 1978 and 1989, then decreased between 1990 and 1995 by 5.7% PA, then stabilised again at a new population size with a mean 35% lower than the mean of the earlier series (WZ Trivelpiece unpubl. data).

At Palmer, the population decreased significantly from 1976 to 1996 (Fraser & Patterson 1997, Smith et al. 1999, WR Fraser unpubl. data), with the linear model accounting for 55% of the variance in the data; no non-linearities were apparent (Table 1). The 1988 datum influenced the fit of the models. Excluded, there was evidence of population stability during the 1980s and a decrease in the 1990s (Fig. 3f). Data more recent than 1996 were not analysed due to evidence that human disturbance may have had an effect in some of the colonies (WR Fraser unpubl. data).

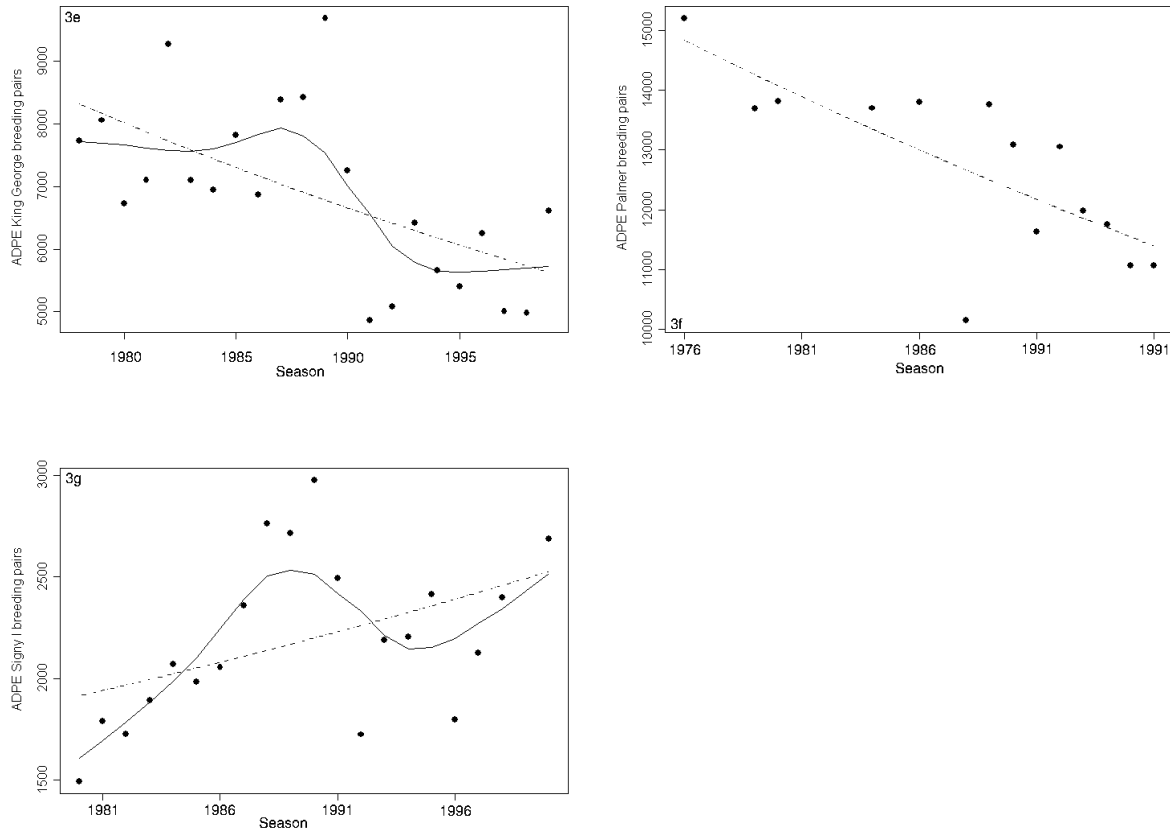
At Signy I, a significant linear increase over the 20-year data set explained only 21% of the observed variance. Both non-linear analyses suggested a period of increase from 1979 to 1989 (5% increase at an annual rate of approximately 0.6%) followed by pronounced fluctuations that have resulted in a subsequent decrease of about 18% (1.8% PA) (Fig. 3g). This analysis was on a slightly longer time-series than that available to Trathan et al. (1996) which investigated trends in relation to ice cover and proximity, providing broadly similar conclusions.

Analyses of populations in the Ross Sea were confined to three localities on Ross I (Figs 3h-j). There were significant non-linearities in the trend of the population at Cape Bird 1981-1997 (Table 1). Piece-wise regression indicated a breakpoint at 1984; the population increased up to 1984 and then decreased (27.9% VE). The best polynomial model to fit these data was a cubic polynomial (64% VE, Table 1). This model indicated an increase up to 1987, a decrease, and then an increase after 1994 (Fig. 3h). Further details on population trends are presented in Wilson et al. (2001).

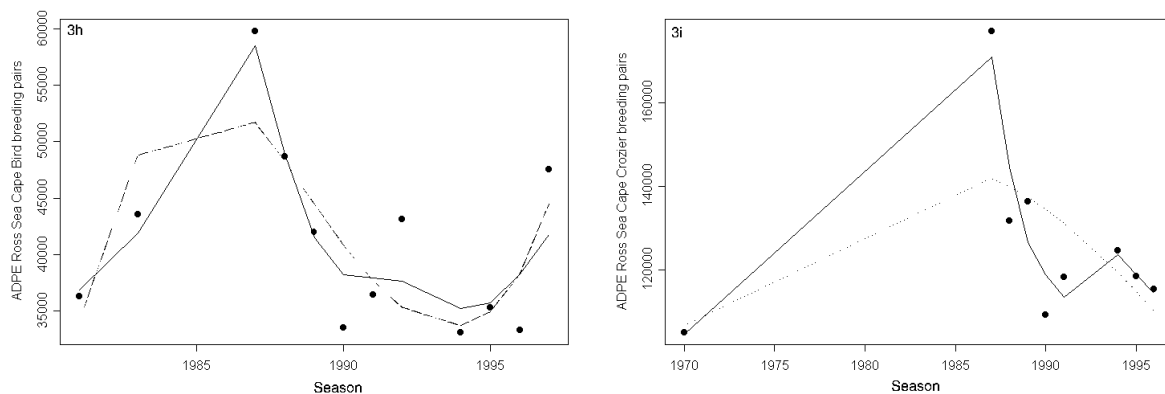
There were significant non-linearities in the trend of the population at Cape Crozier 1970-1996 (Fig. 3i, Table 1). Piece-wise regression indicated a breakpoint at 1986; the population increased up to 1986 and then decreased (50.6% VE). The non-linearity was driven by the data point at 1970. Without that datum, the population decreased linearly from 1987 to 1996 (43.2% VE) (Table 1). The population at Cape Royds increased linearly from 1959 to 1999 (80.5% VE) (Fig. 3j). There was no evidence for

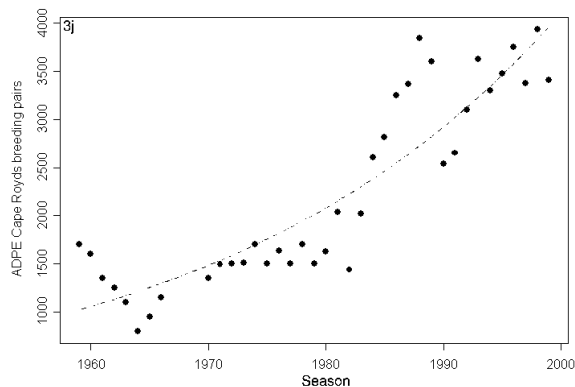
nonlinear trends (eg cycles) in the data (Table 1).

The three regional populations of Adélie Penguins examined exhibit strikingly different trends. Adélie Penguins in East Antarctica show a sustained increase, populations decreased or increased in the Antarctic Peninsula region, while populations showed no clear regional pattern in the Ross Sea.



Figures 3e-g. Plots of analyses of Adélie Penguin population data at three localities in the Antarctic Peninsula: e) Admiralty Bay King George I, f) Palmer, and g) Signy I. The continuous line is the GAM fit to the data, and the dashed line the linear regression fit.





Figures 3h-j. Plots of analyses of Adélie Penguin population data at three localities in the Ross Sea: h) Cape Bird, i) Cape Crozier, and j) Cape Royds. The continuous line is the GAM fit to the data, the dashed line the linear regression fit, and the dotted line the polynomial regression fit.

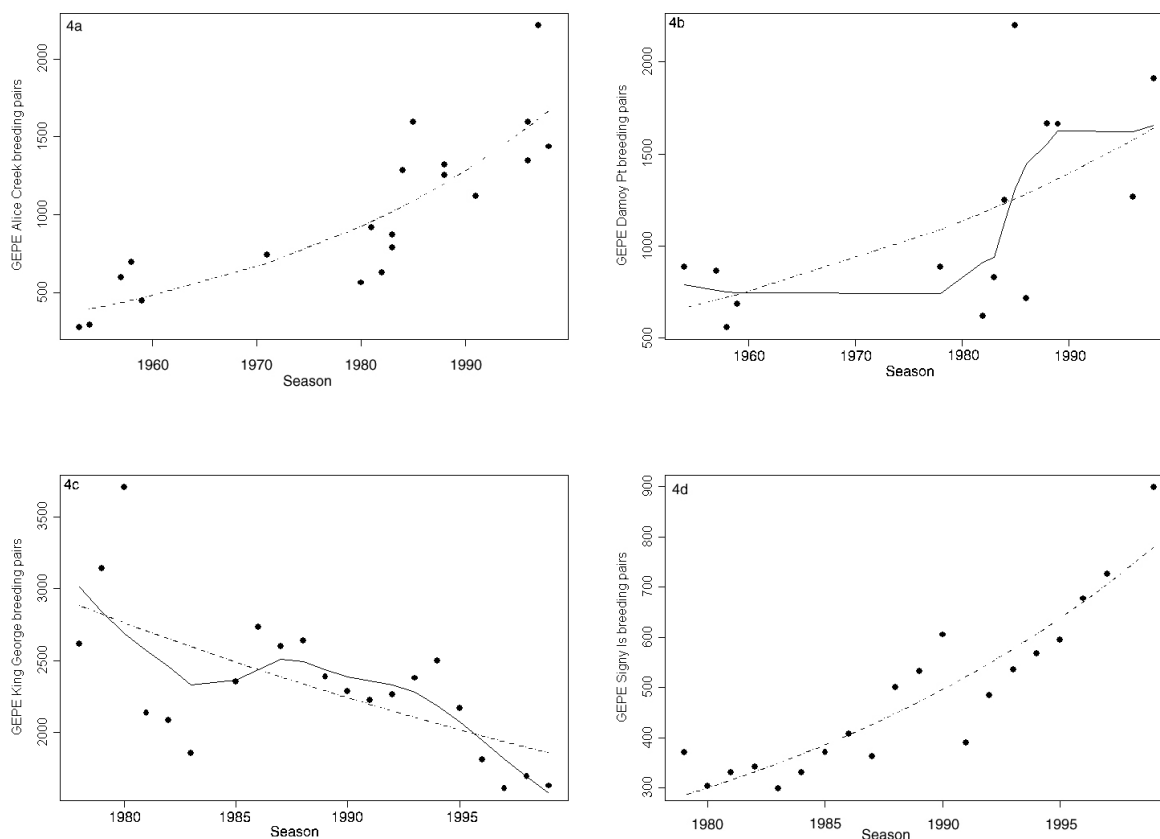
Gentoo Penguin *Pygoscelis papua*

Data sets examined were from the Antarctic Peninsula and Marion I. Populations at Alice Creek and Damoy Pt (Port Lockroy, Wienecke I) have shown significant linear increases (75% and 41% VE, respectively) since the 1980s (Figs 4a-b). The shape of the GAM fit to the Damoy Pt data (Fig. 4b) suggests that the population may have been generally stable (or the counts inconsistent in timing) before 1978; PLR indicates a change in slope at 1978 (Table 1). Several of the smaller data sets (eg Goudier I) suggest population increases in the last two decades.

At Admiralty Bay, South Shetland Is, however, the population has shown a significant decrease overall (Fig 4c). Although a linear decrease explains 42% of the variation, a better fit (61%) is achieved with the GAM (Table 1). This suggests some non-linearities in the data. Demographic data suggest that the underlying cause of this variability in population size may be due to infrequent, strong year classes that dominate the population over 10+ year periods (WZ Trivelpiece unpubl. data). These strong cohorts lead to rapid population increases followed by a 10-year period of stability. Following this stable period, the population again decreases, likely as a result of the strong cohort reaching senescence. Presently the Gentoo Penguin population is at a low point in the 20-year series, as the large 1984 cohort has expired (WZ Trivelpiece unpubl. data).

At Signy I there has been a significant linear increase (84% VE) of 173% at an annual rate of 5% over the 20-year time series (Fig 4d). There is some suggestion of different rates of increase (and/or population processes) before and after 1988 from the PLR. The Bird I, South Georgia, population shows a significant linear decrease of 67% at -4.4% PA over the 20-year time series, with a more consistent pattern of decrease since 1987 (Fig. 4e, Table 1).

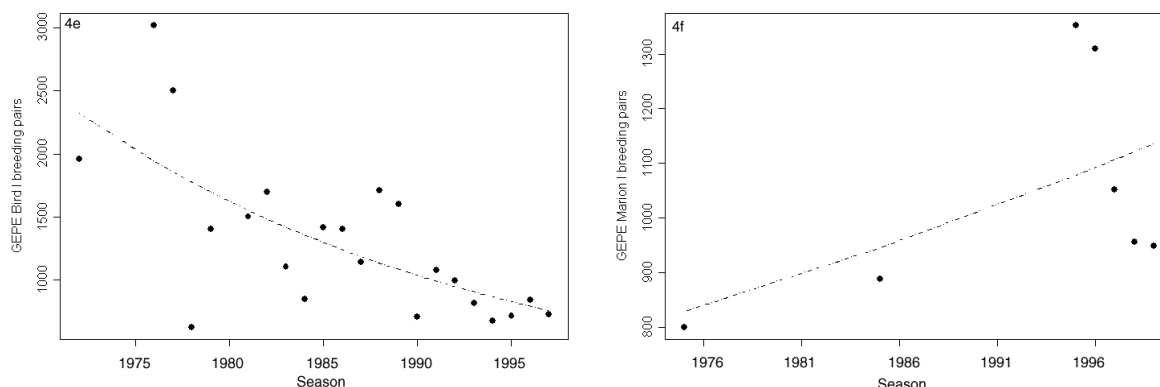
At Marion I, seven whole-island counts over the period 1975-1999 showed an overall increase (31% VE). A polynomial regression (44% VE) suggests there was a decrease after 1995. The initial increase to 1995 (based on only three counts) was followed by a decrease to 1999 using annual counts (Fig. 4f). The fluctuations in this inshore-foraging species at Marion I may be related to changes in food supply caused by shifts in the Subantarctic Front (Crawford et al. unpubl. data, Pakhomov et al. ms).



Figures 4a-d. Plots of analyses of Gentoo Penguin population data at four localities in the Antarctic Peninsula: a) Alice Creek (Port Lockroy), b) Damoy Pt (Port Lockroy), c) King George I and d) Signy I. The continuous line is the GAM fit to the data, and the dashed line a linear regression fit.

This species typically shows large inter-annual fluctuations in population size, usually reflecting some combination of deferred breeding and/or large variation in recruitment (Croxall & Rothery 1995). It shows less synchrony in timing of breeding within and among years than other penguin species, which may affect the accuracy of some counts. For the Antarctic Peninsula region (and associated islands) most data indicate significant recent population increases. The exception is at Admiralty Bay (King George I), where the population decrease appears due to exceptionally long intervals between major recruitment events.

At subantarctic islands, however, the main data indicate significant population decreases, over at least 15 years at South Georgia but probably only during the last five years at Marion I. Populations at the Falkland Is have also decreased over the last decade or more (Bingham 1998) though they have shown signs of recovery in a survey in 2000/01 (Ingham et al. 2001). No hypothesis has yet been proposed to explain the observed population trends in Gentoo Penguins.



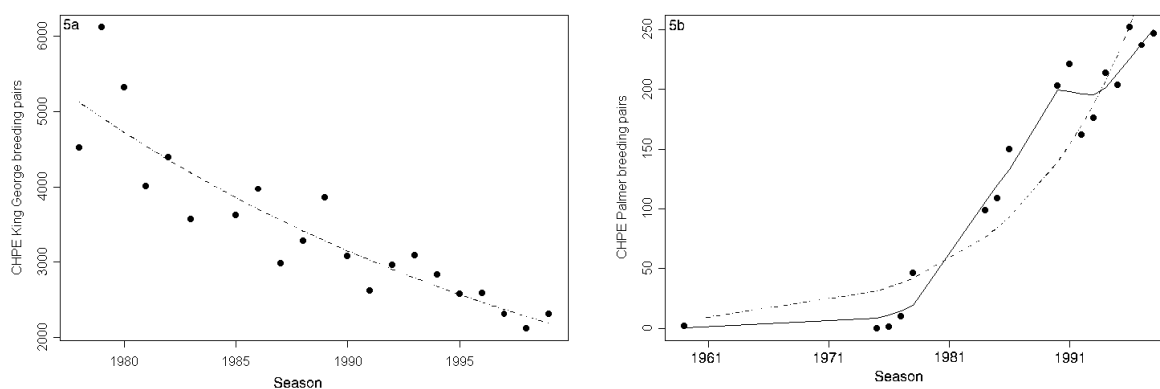
Figures 4e-f. Plots of analyses of Gentoo Penguin population data at e) Bird I South Georgia and f) Marion I. The dashed line is the linear regression fit to the data, and the dotted line the polynomial regression fit.

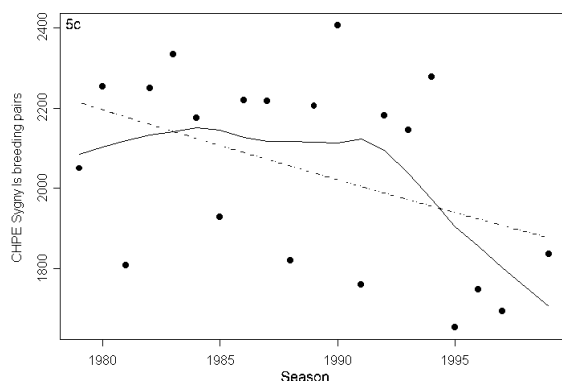
Chinstrap Penguin *Pygoscelis antarctica*

Three data sets from the Antarctic Peninsula were examined. The Admiralty Bay (King George I) population shows a significant decrease over the 1978 – 1999 period (84% VE) (Fig. 5a). There was no evidence of non-linearities in the Admiralty Bay data set (WZ Trivelpiece unpubl. data). At Palmer, the pattern during 1975-1998 shows a statistically significant increase (Smith et al. 1999, WR Fraser unpubl. data), with the linear model accounting for 81% of the variance. There may have been a plateau after 1990 and then a lower rate of population increase (Fig. 5 b, Table 1).

At Signy I, there was an overall significant linear decrease from 1979 to 1998; however this only explains 18% of the variance. The nonlinear analyses (PLR and GAM) suggest a relatively stable population from 1979 to about 1991 (the population increased by 5% at 0.5% pa), followed by a decrease of 15% (at 2% pa) thereafter (Fig. 5b, Table 1).

Populations of Chinstrap Penguins in the Antarctic Peninsula region exhibited different trends within the examined time periods. Whereas populations in the Palmer area have increased significantly since 1976, those on King George I and Signy I have decreased. Smith et al. (1999) proposed that the recent observed increase in Chinstrap Penguin populations in the western Antarctic Peninsula was correlated to regional warming since the 1950s.





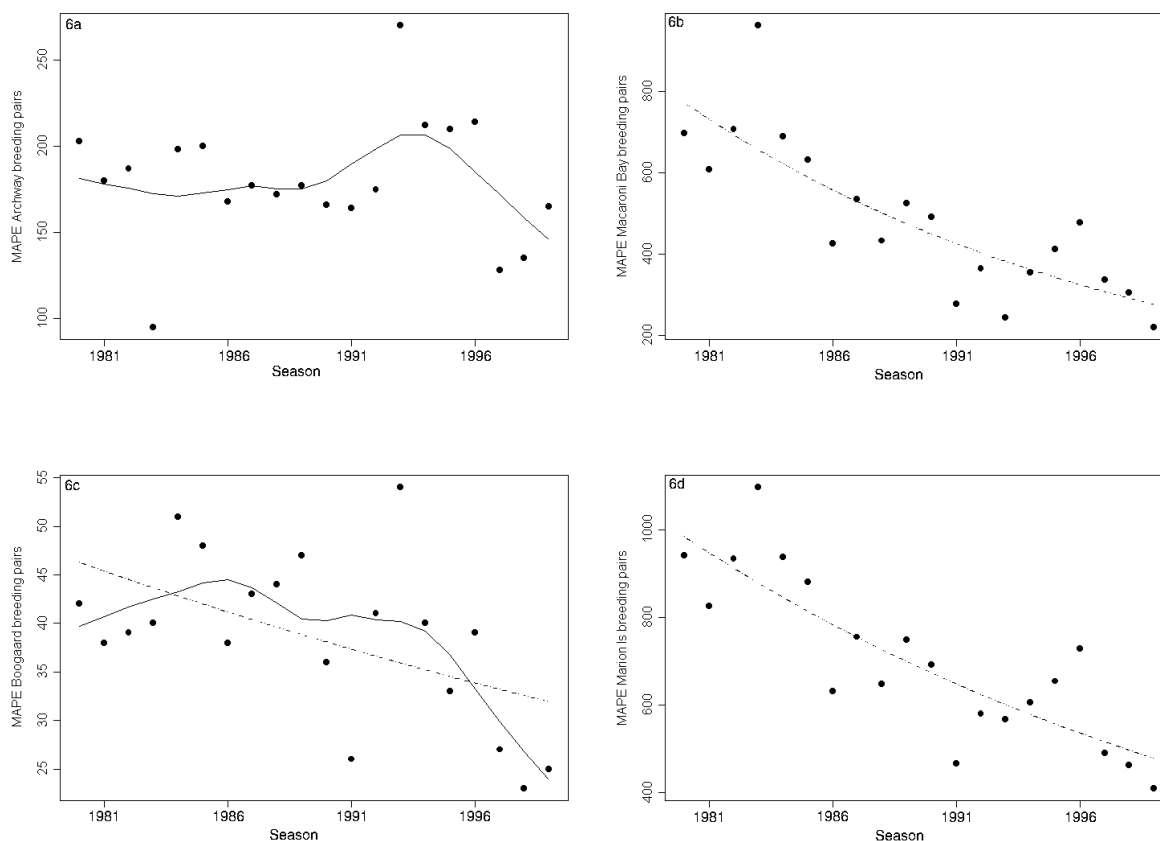
Figures 5a-c. Plots of analyses of Chinstrap Penguin population data at a) King George I, b) Palmer and c) Signy I. The continuous line is the GAM fit to the data, and the dashed line a linear regression fit.

Macaroni Penguin *Eudyptes chrysolophus*

Three data sets from Marion I were examined. The population at Archway Bay did not show a significant linear trend, but both non-linear models indicated a decrease in the last four years, with the breakpoint at 1990 (Table 1, Fig. 6a). The Macaroni Bay colony showed a linear decrease (64% VE) (Fig. 6b); there may have been a change in population trajectory after 1990 (Table 1). The van den Boogaard colony showed a linear decrease (26% VE) (Fig. 6c), with evidence of a breakpoint in 1989 (Table 1). Combined annual counts for these three study colonies in the eastern sector of Marion I from 1980-1999 (Cooper et al. 1997) showed an overall linear decrease of 3.7% PA (70% VE) (Fig. 6d). Six whole-island counts from 1977 to 1998 showed no clear trend, due to large error estimates, and therefore were not analysed. The total Macaroni Penguin population of Marion I has been considered stable (Crawford et al. ms), perhaps because the species' "intermediate" foraging range has been relatively little affected by assumed changes in food supply caused by shifts in the Subantarctic Front (E Pakhomov in litt.).

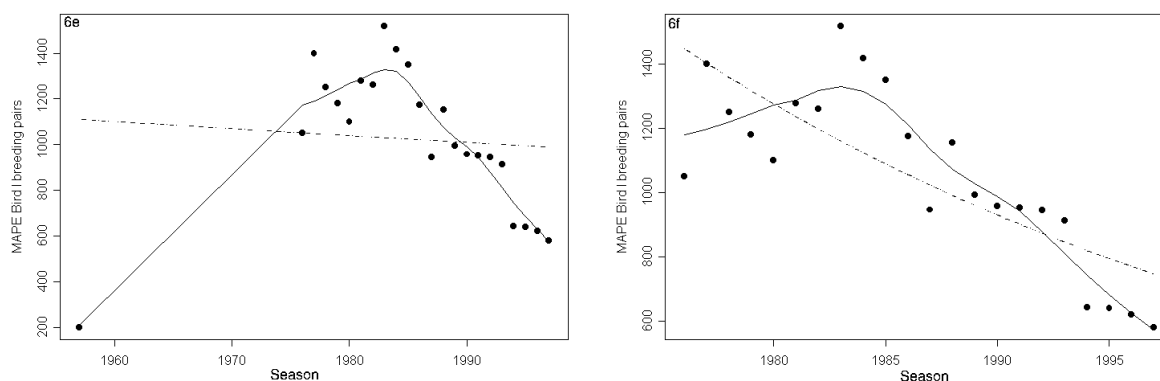
The Bird I, South Georgia study population increased from 200 pairs in 1958 to a maximum of 1400 pairs in 1983 (at an average annual rate of increase of 9.7%) (Figs 6e-f). Using data from 1977 on, there was a significant decrease overall (60% VE) (Fig. 6f). There was evidence of a breakpoint in 1987 (86% VE) with a population increase of approximately 20% between 1977 and 1986 (at a rate of about 0.5% PA), thereafter decreasing by 48% at an annual rate of about 5%. A much larger colony at Bird I, whose population size is estimated annually using two strip transects (see Croxall & Prince 1979) has decreased by 60% since 1977 (JP Croxall unpubl. data). These decreases are broadly reflected in other colonies at South Georgia, many of which have decreased by up to 50% over the last 20 years (Trathan et al. 1998).

At Iles Kerguelen, three counts obtained from aerial photographs at low and high altitudes of several large colonies (2000-40000 pairs) on the Courbet Peninsula in 1963, 1986 and 1999 indicate a slight increase of the colonies throughout the period (Weimerskirch et al. 1989, H Weimerskirch, unpubl. data).



Figures 6a-d. Plots of analyses of Macaroni Penguin population data at a) Archway Bay, Marion I, b) Macaroni Bay, Marion I, c) van den Boogaard, Marion I and d) Marion I study colonies total. The continuous line is the GAM fit to the data, and the dashed line a linear regression fit.

For the best-monitored populations at both South Georgia and Marion I, significant decreases are evident over the last 15-20 years. At South Georgia, similar decreases are evident on an island-wide scale, while island-wide counts on Marion I are less clear due to a high estimated error, caused by the topography. Elsewhere (eg Iles Kerguelen) populations may be stable but data are too few to assess population trends.



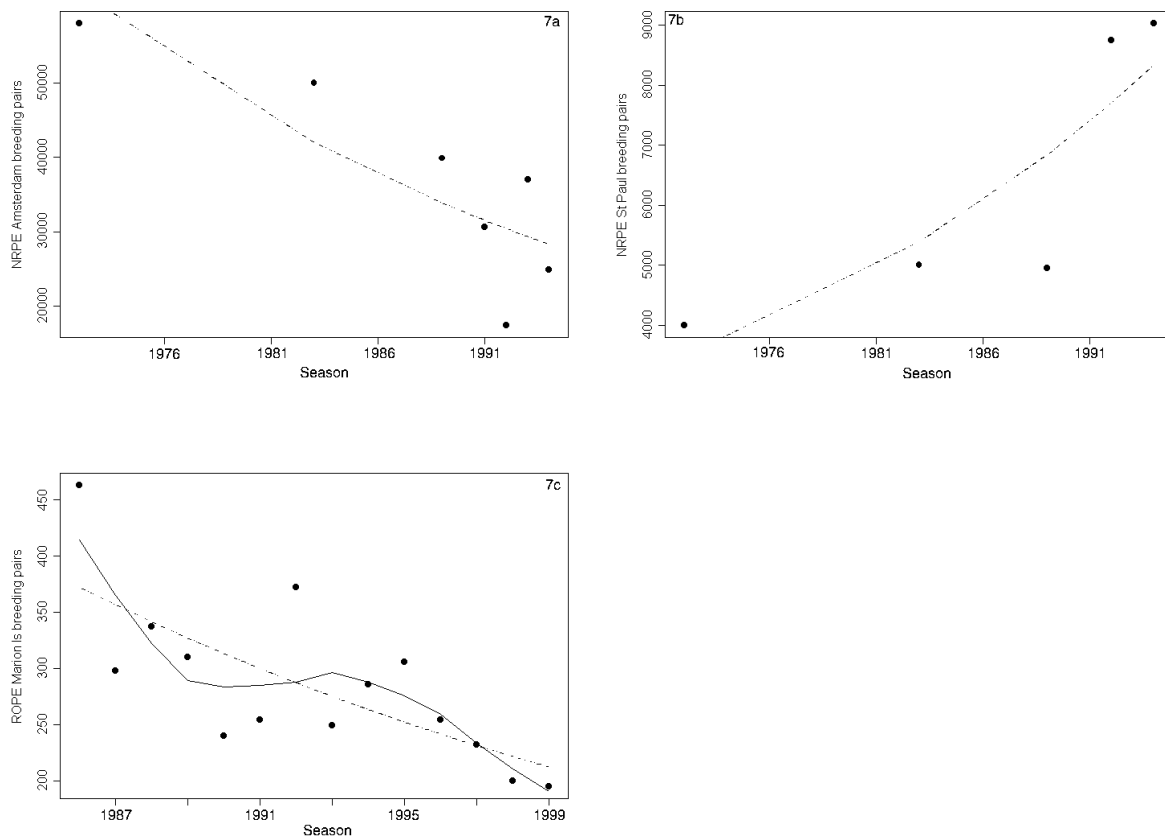
Figures 6e-f. Plots of analyses of Macaroni Penguin population data at e) Bird I and f) Bird I since 1977. The continuous line is the GAM fit to the data, and the dashed line a linear regression fit.

Northern Rockhopper Penguin *Eudyptes chrysocome moseleyi*

The population at Ile Amsterdam decreased significantly (65% VE) from 1972 to 1994 (seven counts) (Fig. 7a), whereas Northern Rockhopper Penguins on Ile St Paul increased significantly (75% VE), in the period 1972 to 1994 (five counts) Fig. 7b, Table 1. The reason(s) for these two diametrically opposed trends at two islands only 80km apart are unknown, although the Ile St Paul population may still be recovering from past exploitation as bait for a local crayfish (*Jasus* spp.) fishery. Conversely, the Rockhopper Penguins on Ile Amsterdam may suffer predation by the increasing Subantarctic Fur Seal *Arctocephalus tropicalis* population (Guinard et al. 1998).

Eastern Rockhopper Penguin *Eudyptes chrysocome filholi*

Combined annual counts from 1986-1999 for three small study colonies of Eastern Rockhopper Penguins close together in the eastern sector of Marion I (Cooper et al. 1997) showed an overall linear decrease of 4.3% PA (55% VE). The non-linear models indicated a breakpoint occurring in 1992 (70% VE) (Table 1, Fig. 7c). The trends for each colony were not analyzed separately due to their similarity. Five whole-island counts from 1974 to 1997 showed no clear trend due to large error estimates (Crawford et al. unpubl. data). Counts of sectors of the coastline that support a quarter of the overall population, and where the error on counts is thought to be low, show a decrease of 21% over three years between 1995 and 1998 (Crawford et al. unpubl. data). It has been suggested that the fluctuations in this relatively inshore-foraging species at Marion I may be related to changes in food supply caused by shifts in the Subantarctic Front (Crawford et al. unpubl. ms, Pakhomov et al. unpubl. data).



Figures 7a-c. Plots of analyses of Northern and Eastern Rockhopper Penguin population data at a) Ile Amsterdam, b) Ile St Paul and c) Marion I. The continuous line is the GAM fit to the data, and the dashed line a linear regression fit.

Wandering Albatross *Diomedea exulans*

Interannual fluctuations in population size are less pronounced in this species/species-group than in other biennially breeding albatross species; nevertheless assessing the utility/reliability of isolated counts may still be problematic.

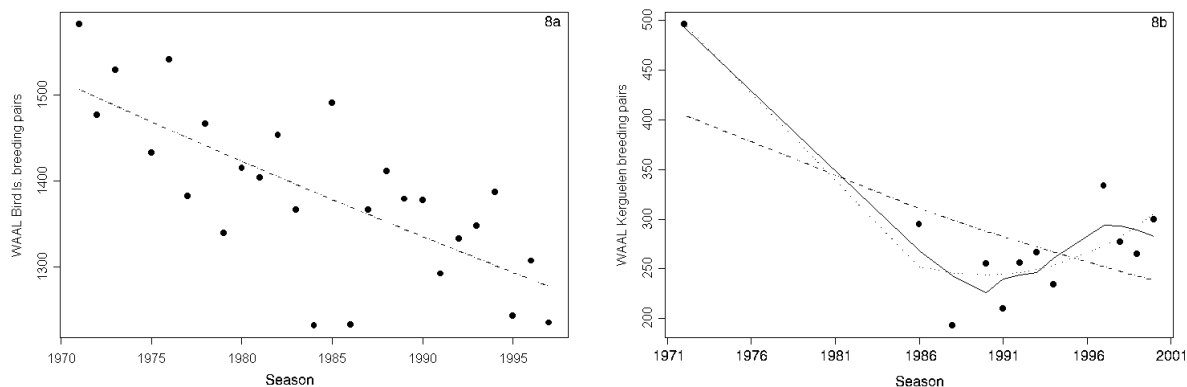
At Bird I, South Georgia the population (which is about half of the total South Georgia population) decreased significantly (15.3% at 1% PA) from 1972 to 1998 (52% VE) (Fig. 8a). This is very similar to the conclusions of Croxall et al. (1998), who incorporated data for 1963-1965 and showed that an exponential fit explained 65% of the variation in the data for 1963-1997. There is an indication from the PLR that there was a breakpoint in 1984 with decreases occurring on either side.

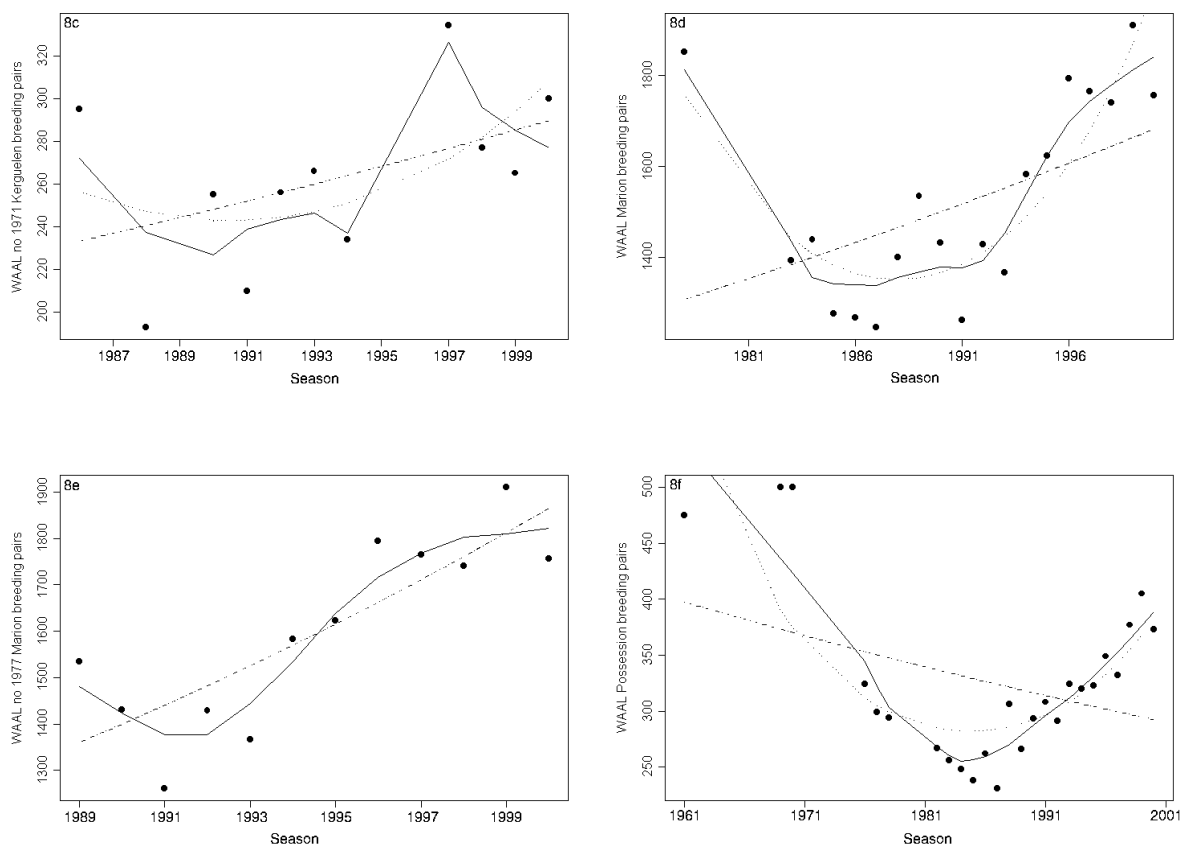
At Iles Kerguelen (Weimerskirch et al. 1997), using the complete data set (ie including 1972) there are non-linearities in the data with a significant overall decrease (Fig. 8b). There is a strong suggestion that the population decreased from 1972 to 1990 (PLR, Table 1) by 54% at 5.8% PA, subsequently increasing by 29% at 2.9% PA. Removing the 1972 datum results in detecting a significant increase over the period 1989 to 1999. The non-linear models indicate a breakpoint in 1991, with the population decreasing before 1991 and then increasing (Table 1, Fig. 8c).

At Marion I, whether or not the 1978 datum is included, the non-linear models gave the best fit of an overall significantly increasing population (Fig. 8d). Including the 1978 datum would suggest a probable decrease until about 1991, followed by an increase thereafter. Excluding 1978, the non-linear models suggest a population decrease in 1989-1993 and an increase thereafter (Fig 8e). The apparent plateau effect at the end of the time series is caused mainly by the 1999 datum; several more years data are needed to assess this.

At Ile de la Possession, there is a pronounced non-linearity in the data (Fig. 8f). This reflects the rapid population decrease of about 57% at 3.3% PA from the 1960s to mid 1980s, followed by a period of consistent increase (49% at 3.1% PA) from the mid 1980s to the present (Weimerskirch et al. 1997). The four data sets available indicate that in the Indian Ocean the three populations, after substantial decreases from at least the mid 1970s, have increased since about the end of the 1980s. In contrast, at South Georgia, the population has continued to decrease throughout the 1990s. Changes in the location of longline tuna fisheries are believed to have resulted in a decrease bycatch of albatrosses in the Indian Ocean (Weimerskirch et al. 1997).

Longline fisheries in the Southern Ocean are inferred to be the primary causes of these observed long-term population decreases (Woehler & Croxall 1997, Gales 1998, Tuck et al. in press). At-sea abundances of wandering albatrosses in the Prydz Bay region of East Antarctica have decreased significantly since 1980/81 (Woehler 1996, Woehler & Watts 2000).





Figures 8a-f. Plots of analyses of Wandering Albatross population data at a) Bird I, b) Iles Kerguelen, c) Iles Kerguelen excluding 1972, d) Marion I, e) Marion I excluding 1977 and f) Ile de la Possession. The continuous line is the GAM fit to the data, the dashed line the linear regression fit, and the dotted line the polynomial regression fit.

Amsterdam Albatross *Diomedea amsterdamensis*

The small population restricted to the plateau on Ile Amsterdam has been surveyed annually since 1984 through counts of incubating birds. The population has shown a significant linear increase throughout the period (70% VE) (Fig. 9) resulting from efforts to protect nesting areas on Ile Amsterdam (Weimerskirch et al. 1997, Inchausti & Weimerskirch ms). Bycatch in tuna longline fisheries off Australia have been reported (Gales 1998).

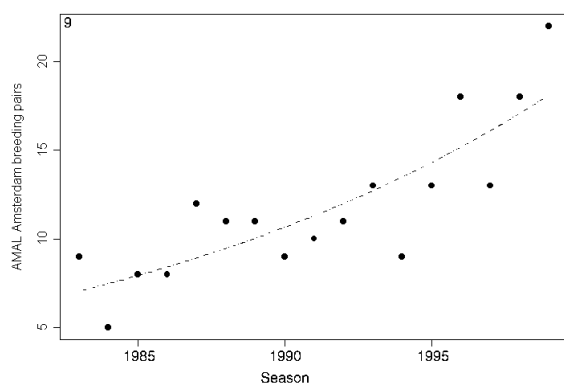


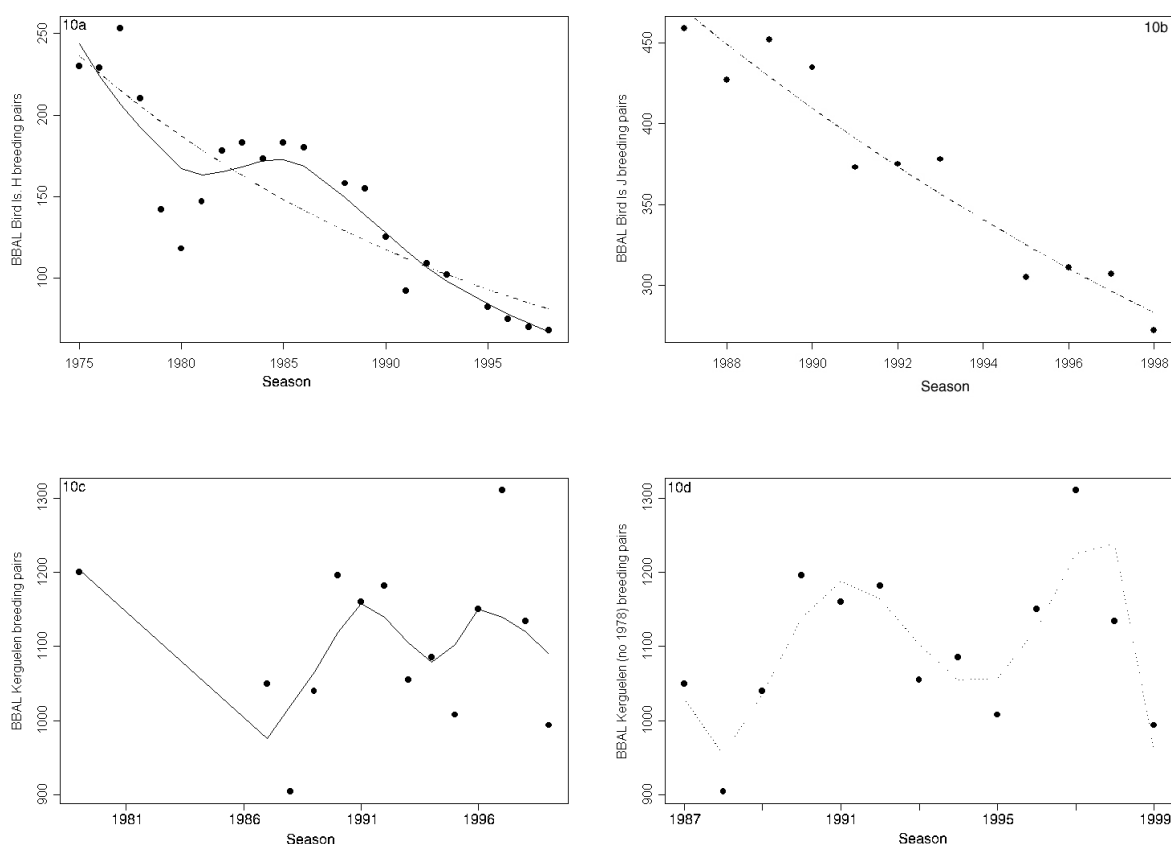
Figure 9. Plot of analysis of Amsterdam Albatross population data at Ile Amsterdam 1983-1999. The dotted line is the linear regression fit to the data.

Black-browed Albatross *Thalassarche melanophrys*

The main study colony at Bird I (Colony H) shows a significant linear decrease (66% at an annual rate of 4.5% from 1976 to 1999) overall (Fig. 10a). The non-linear models indicate a steeper population decrease between 1976 and 1981 and a partial recovery 1981-1986 (Table 1, Fig. 10a). From about 1986 (PLR, Table 1), the decrease has been very consistent (65% at 8.3% PA). This closely resembles the Colony J population (whose decrease averages 4.5% PA between 1989 and 1999), where the linear regression explains 93% of the variance (Fig. 10b). These conclusions (with two years' additional data) are very similar to those set out in Croxall et al. (1998) which also shows that broadly similar decreases have occurred in other Bird I colonies less frequently counted.

At Iles Kerguelen, there is no significant overall trend to the data (Weimerskirch & Jouventin 1998). The GAM suggests some quasi-cyclical fluctuations (46% VE) and a 5th order polynomial fits the data the best (Fig. 10d). This may reflect the links between this albatross and an important prey at Ile Kerguelen, the icefish *Champsocephalus gunnari*, in which strong cohorts appear every three to four years (Figs 10c & d), (Kock 1992).

As with wandering albatross, longline fisheries are inferred to be the primary cause for decreases in breeding populations of black-browed albatrosses. Differences among populations in the rates of decrease likely reflect the varying levels of interaction with longline fisheries (Gales 1998).



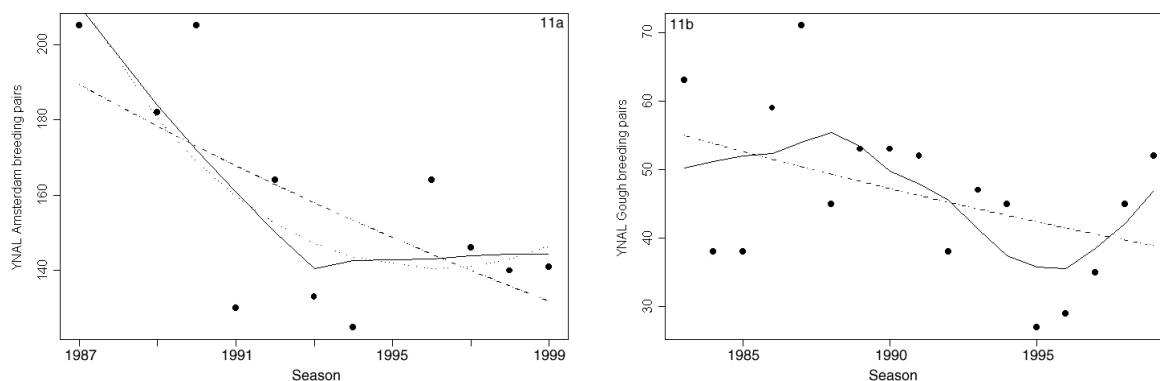
Figures 10a-d. Plots of analyses of Black-browed Albatross population data at a) Colony H Bird I, b) Colony J Bird I, c) Iles Kerguelen and d) Iles Kerguelen, excluding the 1979 datum. The continuous line is the GAM fit to the data, the dashed line the linear regression fit, and the dotted line the polynomial regression fit.

Indian Yellow-nosed Albatross *Thalassarche chlororhynchos*

Census data are available from Ile Amsterdam, where two types of data are available between 1982 and 1999: scattered counts of half of the colonies on the island, and yearly counts of three colonies (140 pairs in 1999). The latter data indicate a significant linear decrease (43% VE) with a suggestion from the non-linear models of stabilisation of numbers from the early 1990s (Fig. 11a). The former data also confirm this decrease (Weimerskirch & Jouventin 1998), Table 1.

Atlantic Yellow-nosed Albatrosses *Thalassarche carteri*

A small study colony (35-70 pairs) at Gough I was monitored for the period 1983-1999. There was an overall decrease in the population (19% VE). The non-linear models indicated a decrease in the population in the early 1990s with an increase after 1997 (Table 1, Fig. 11b). More data are needed to evaluate if this is a true increase. The population decrease is thought to be from bycatch in longline fisheries, especially off the Atlantic coast of South America (Olmos et al. 2000)



Figures 11a - b. Plots of analyses of Yellow-nosed Albatross population data from a) Ile Amsterdam, and b) Gough I. The continuous line is the GAM fit to the data, the dashed line the linear regression fit, and the dotted line the polynomial regression fit.

Grey-headed Albatross *Thalassarche chrysostoma*

Three long-term data sets exist from two populations; two from colonies on Bird I, South Georgia and one from total-island counts at Marion I. Considerable inter-annual variation (particularly in successive pairs of years) is characteristic of this biennially-breeding species.

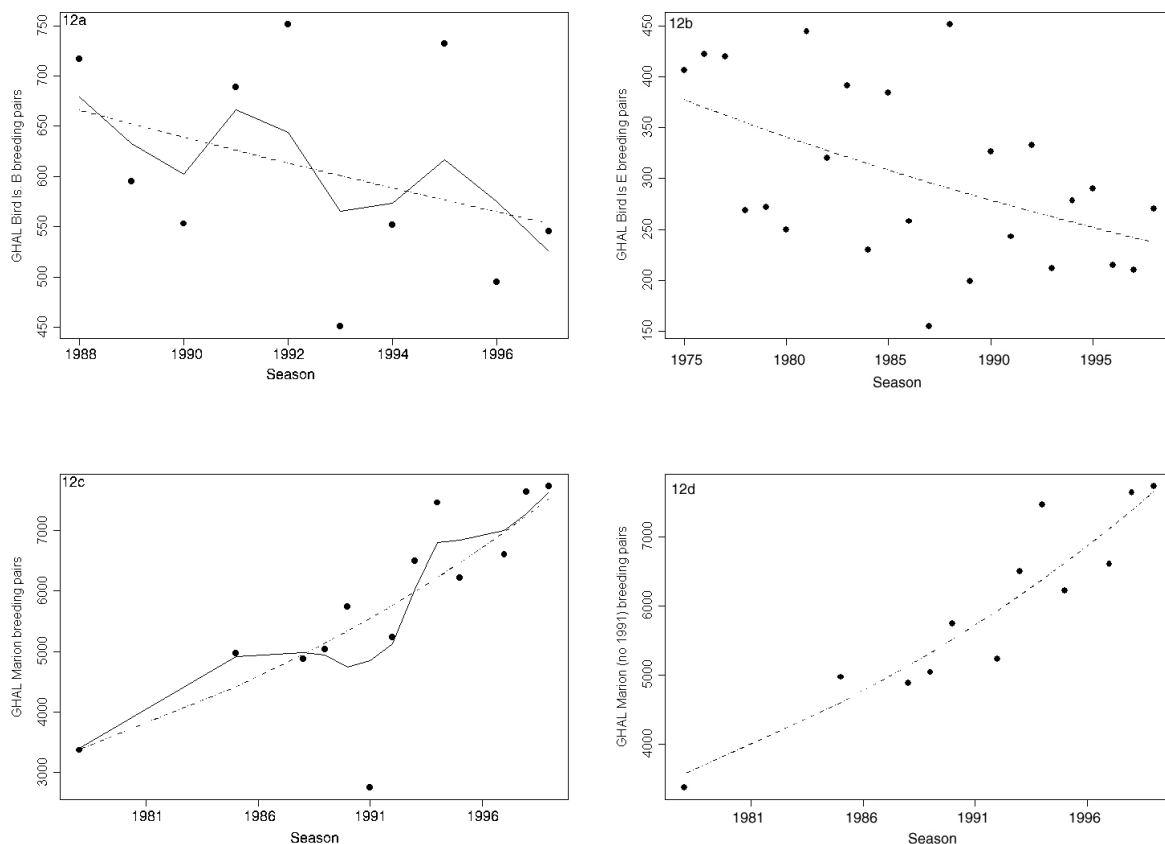
At Colony B, Bird I, South Georgia, the linear decrease is significant (17% at 2% PA from 1989 to 1998), but only explains 13% of the variation. A better fit is obtained by the non-linear models, but this is primarily reflecting the major disparities in breeding demi-populations in 1993 and 1996 (Fig. 12a). The breakpoints detected by PLR also reflect these disparities (Table 1). At the main Bird I study colony (Colony E), the population shows a significant linear population decrease (37% at 2% PA from 1976 to 1999, with 25% VE) (Fig. 12b). With two additional years data, this result is similar to that reported by Croxall et al. (1998) where a fitted exponential decrease explained 12% of the variance. The study colonies broadly reflect the trends at other, less frequently surveyed colonies at Bird I (Prince et al. 1994). Using data from Croxall et al. (1998), a linear regression detected a significant decrease (28% VE).

On Marion I, 13 whole-island counts were made between 1977 and 1999 (single counts in 1977 and 1985; annual counts between 1988 and 1999, except 1996). A significant linear increase was detected over the entire period (59% VE) (Fig. 12c). The non-linear models explain 75% of the variation

(Table 1); however, the non-linearity is mainly driven by the exceptionally low count in 1991. Removing this datum increases the VE by the linear model to 82%. A GAM does not improve the variation explained in the data. This suggests that the population has increased linearly with the exception of 1991 (Fig. 12d). The annual rate of increase was 3.7%.

A decrease in the breeding population at Campbell I, New Zealand, was reported for the period 1942 - 1996 (Waugh et al. 1999). However, the timing and nature of the decrease depends entirely on the accuracy of a single count conducted sometime between 1961 and 1967.

The populations of grey-headed albatross show different trends: the Bird I and Campbell I populations are decreasing whereas the Marion I population is increasing. The decrease in the Bird I population has been attributed to bycatch in longline fisheries (Croxall et al. 1998, Gales 1998), and bycatch has been implicated in the Campbell I population decrease (Waugh et al. 1999); no hypothesis has been proposed for the increase in the Marion I population.



Figures 12a-d. Plots of analyses of Grey-headed Albatross population data at a) Colony B Bird I, b) Colony E Bird I, c) Marion I and d) Marion I, excluding the 1991 datum. The continuous line is the GAM fit to the data, and the dashed line the linear regression fit.

Sooty Albatross *Phoebetria fusca*

Population data are only available from Ile de la Possession, Iles Crozet (Weimerskirch & Jouventin 1998), where approximately one third of the total island population of Sooty Albatrosses has been counted on eight occasions over the period 1981 to 1999. The population has shown a linear decrease throughout at an annual rate of 5.1% (85% VE) (Fig. 13). Longline fisheries are inferred to be the primary cause for the observed decrease (Weimerskirch & Jouventin 1997, Gales 1998).

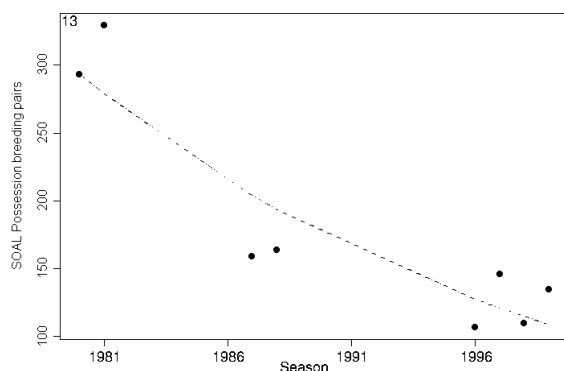


Figure 13. Plot of analysis of Sooty Albatross population data at Ile de la Possession, Iles Crozet. The dashed line is the linear regression fit to the data.

Light-mantled Sooty Albatross *Phoebastria palpebrata*

Data are only available from the Iles Crozet. Between 1981 and 1999, nine counts of the major part of the population of Ile de la Possession, Iles Crozet, were made (Weimerskirch & Jouventin 1998). Although linear models indicate a significant decrease, the variance explained is very low (7%) because of the scattered nature of censuses and the extensive inter-annual variation in numbers. The non-linear models fit better and suggest a decrease during the 1980s and a stabilisation or even increase afterwards, but this interpretation should be viewed cautiously due to low sample size (Table 1, Fig. 14). Longline fisheries are believed to constitute the greatest threat to Light-mantled Sooty Albatrosses (Gales 1998). At-sea abundances of light-mantled sooty albatrosses in the Prydz Bay region of East Antarctica have decreased significantly since 1980/81 (Woehler 1996, Woehler & Watts 2000). It is important that data include counts in successive years to account for interannual fluctuations characteristic of biennially breeding species such as Wandering, Light-mantled Sooty and Grey-headed Albatrosses.

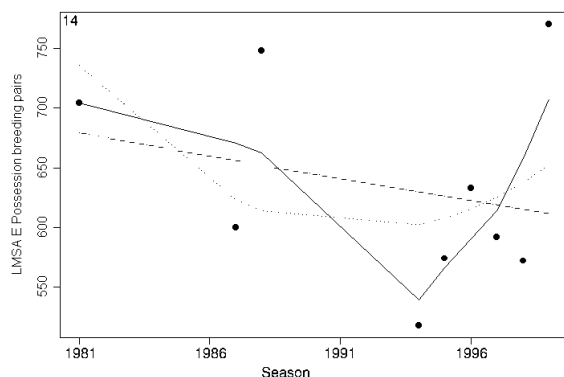


Figure 14. Plots of analyses of Light-mantled Sooty Albatross population data at Ile de la Possession, Iles Crozet. The continuous line is the GAM fit to the data, the dashed line the linear regression fit, and the dotted line the polynomial regression fit.

Southern Giant Petrel *Macronectes giganteus*

Seven data sets were examined. Long-term data sets spanning 25 years were available for four localities (Mawson, Davis, Casey and Palmer, Antarctic Peninsula); population data for three other localities examined spanned nearly 20 years (King George I, Marion I and Ile de la Possession).

Supplementary long-term data were available from Pointe Géologie; however, these data were not included in this analysis because the population decrease was due to station construction (Micol & Jouventin 2001). All census data have been tabulated by Patterson et al. (in press).

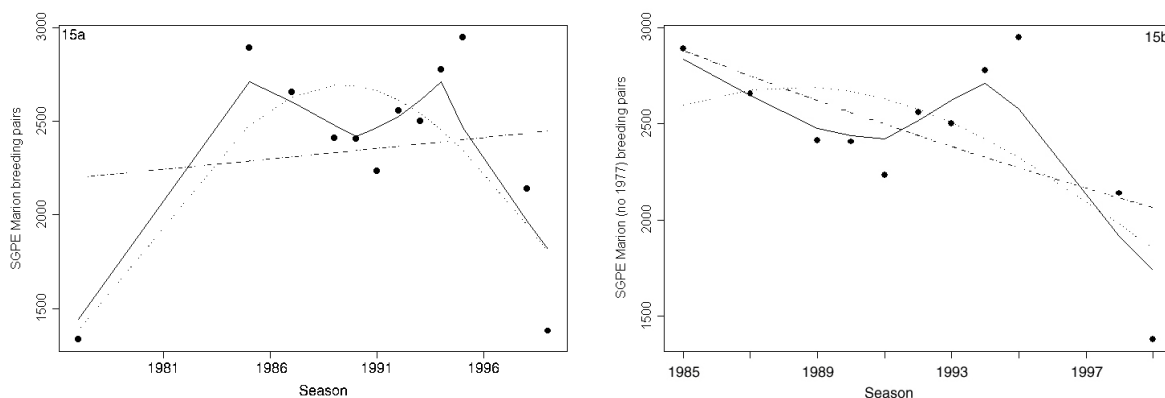
At Marion I, 12 whole-island counts have been made from 1977 to 1999 (single counts in 1977, 1985 and 1987 and annual counts from 1989 to 1999, except for 1996). No trend was detected by a linear model. There were strong nonlinearities in the data. The GAM showed a population increase from 1977 to 1986, followed by a decrease in the late 1980s, a second peak in 1995, and finally a steep decrease over the last four years (81% VE), (Fig. 15a). A polynomial model was consistent with the GAM, showing a peak in the late 1980s and early 1990s, followed by a steep decrease in the late 1990s (64% VE). The decrease in the early 1990s was indicated by the PLR also (Table 1). Excluding the 1977 datum, a significant decrease occurred over the period 1985 to 1999 (30% VE). A polynomial model once again picked up a decrease from 1985 to 1991, followed by a peak in 1995 and a subsequent decrease to 1999 (79% VE), Fig. 15b. In summary, the population at Marion I seems to have fluctuated during the late 1980s and early 1990s and is presently experiencing a substantial decrease over the last four years.

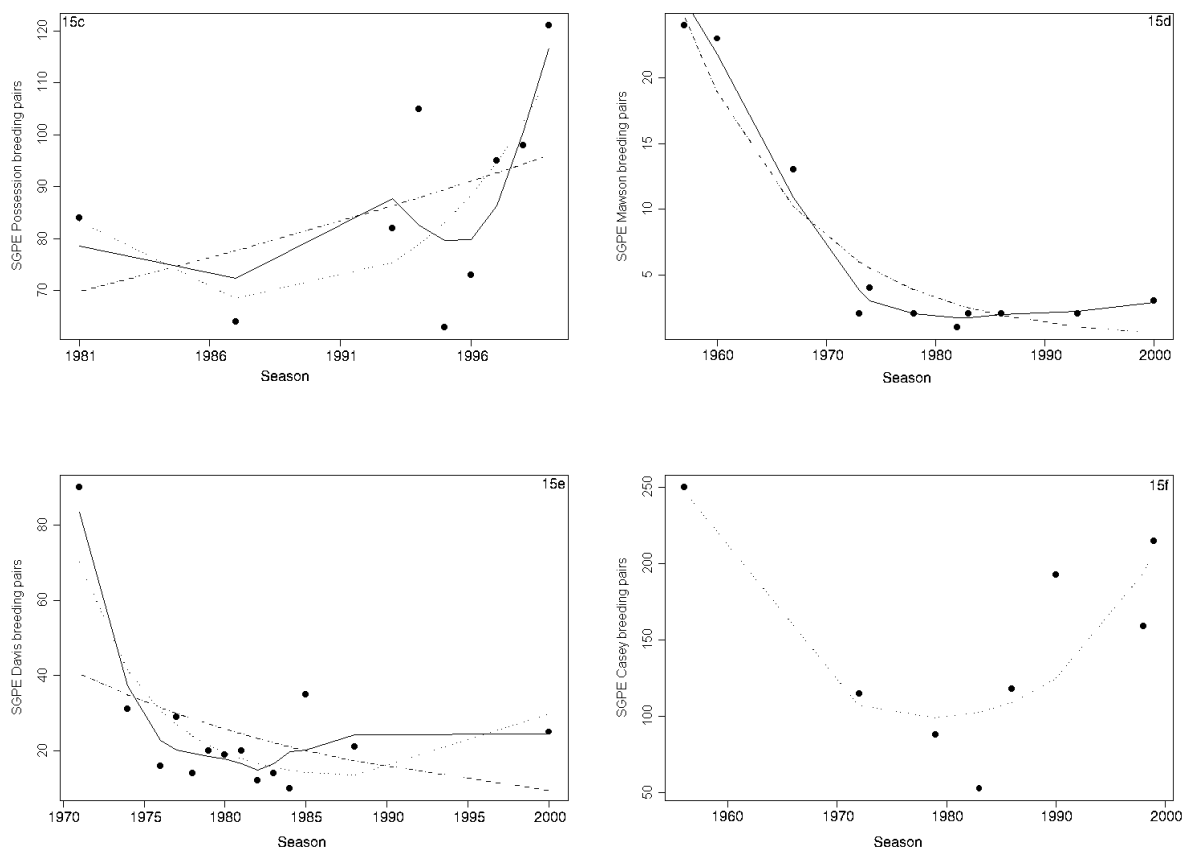
The breeding population at Ile de la Possession has been counted nine times between 1981 and 1999. The population appears to have increased significantly (LR: 20% VE). The non-linear models suggest that the population may have been slightly decreasing during the 1980s to 1993, and thereafter increased rapidly (Table 1, Fig. 15c).

The Southern Giant Petrel population at Mawson decreased linearly between 1957 and 2000 (84% VE) (Fig. 15d). There was some evidence that the population decreased from 1957 to 1980 and increased from 1981 to 2000 (PLR, Table 1).

The Southern Giant Petrel population at Davis decreased between 1971 and 1977 and then increased from 1978 to 2000 (PLR; 62.8% VE). This pattern was reflected in the significant polynomial regression (85.4% VE) and generalized additive models (82% VE) (Fig. 15e). A strictly linear model explained less of the variation (22% VE), but indicated a population decrease (Fig. 15e).

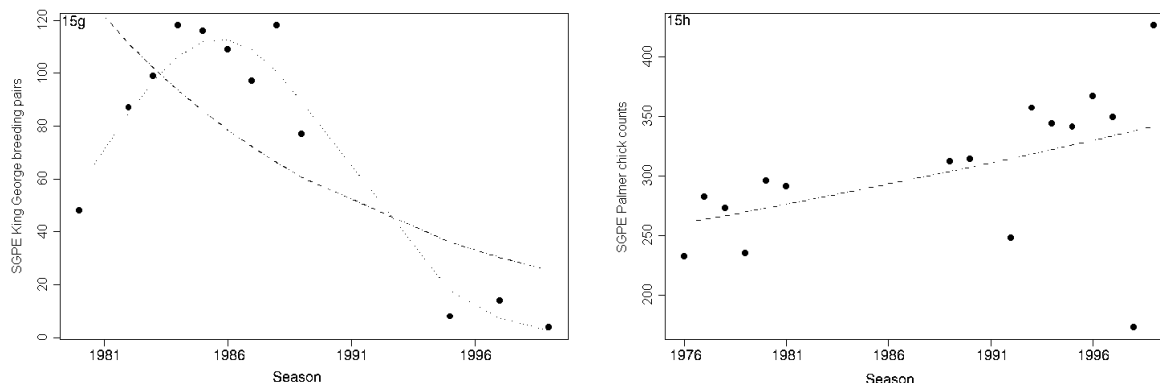
The Southern Giant Petrel population at Casey decreased between 1956 and 1982 and then increased from 1983 to 1999 (PLR; 74.6% VE). This pattern was reflected in the significant polynomial regression (68.3% VE) (Fig. 15f). No trend was detected using a strictly linear model (Table 1).





Figures 15a-f. Plots of analyses of Southern Giant Petrel population data at a) Marion I, b) Marion I excluding the 1977 datum, c) Ile de la Possession, d) Mawson, e) Davis and f) Casey. The continuous line is the GAM fit to the data, the dashed line the linear regression fit, and the dotted line the polynomial regression fit.

Breeding populations within the Antarctic Peninsula region show various trends. Populations on King George I (Fig 15g) decreased significantly within the time period examined (44% VE). Further analysis using non-linear models increased the proportion of variance explained to 95%, stressing the reversal in population trajectory from increasing to decreasing during the late 1980s. Within the King George I area, individual localities appear to have responded to anthropogenic influences (eg station construction, helicopter overflights, colony visitation). Local populations in the Nebles Pt area have decreased from over 100 pairs in 1986 to less than 10 in the 1999 season due to poor breeding success. In contrast to the Nebles Pt area, the Southern Giant Petrel breeding numbers within Fildes Strait have increased over the same period. The Fildes Strait breeding localities are relatively undisturbed and some evidence suggests that at least a small number of birds have relocated from the Nebles Pt breeding locality.



Figures 15g-h. Plots of analyses of Southern Giant Petrel population data at g) King George I and h) Palmer. The dashed line is the linear regression fit to the data, and the dotted line the polynomial regression fit.

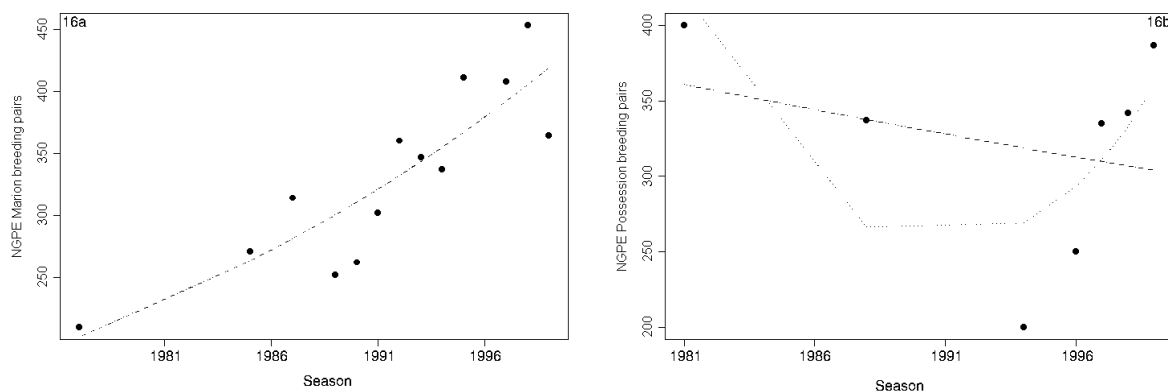
Chick production by Southern Giant Petrels in the Palmer area has increased at 2.7% PA since 1974 (22% VE) (Fig. 15h). The non-linear models suggested the presence of a breakpoint in 1988 (Table 1). The 1998 season showed a reduced level of breeding effort with a return to previous breeding population levels during the 1999 season. Removal of the 1998 productivity datum resulted in an increased VE in the linear model to 66%. In contrast with other Antarctic Peninsula breeding localities, the Palmer population is not exposed to high levels of human disturbance; however, recent and rapid increases in longline hook retrievals at nests indicate the most immediate threat to the local population.

At Bird I, South Georgia, a survey in 1996 (DR Briggs & R Humpidge, unpubl. data) for comparison with similar surveys in each of the years 1979 to 1981 (Hunter 1984) revealed an estimated population of 521 pairs, 7% lower than the average of 558 pairs for the earlier period (González-Solís et al. 2000).

Changes in breeding population of Southern Giant Petrels vary greatly among the three regions analysed. The Marion I breeding population has fluctuated and most recently has decreased sharply; probably due to mortality caused by the Patagonian toothfish *Dissostichus eleginoides* longline fishery, which is a recent development (J Cooper, unpubl. data). Within the Antarctic Peninsula region, populations to the north have significantly decreased with increased human activity (King George I), whereas breeding populations in the vicinity of Palmer may have doubled over the last 20 years, perhaps reflecting limited disturbance at nest sites (Woehler & Croxall 1997).

Northern Giant Petrel *Macronectes halli*

Two data sets, from Marion I and Ile de la Possession, were examined. At Marion I, 13 whole-island counts have been made since 1977 (a single count in 1977, and annual counts since 1985). These data show a significant linear increase over the entire time period (76% VE) (Fig. 16a) with an annual rate of increase of 3.4%. There was no evidence of non-linearities in the data (Table 1).



Figures 16a-b. Plots of analyses of Northern Giant Petrel population data at a) Marion I, and b) Ile de la Possession. The dashed line is the linear regression fit to the data, and the dotted line the polynomial regression fit.

At Ile de la Possession, seven counts have been made since 1981 (three counts between 1981 and 1994, and annual counts since 1996). A significant decrease was detected over the entire period; however, only 8% of the variance in the data was explained by a linear model (Fig. 16b). A polynomial model showed a decrease between 1981 and 1988, then a sharp increase between 1994 and 1999 (53% VE). The PLR analysis identified a breakpoint in 1993, with the population being stable before this and increasing thereafter (Table 1). We thus conclude that the population at Ile de la Possession was either stable or decreasing before 1993 and has since increased. The increase on Marion I may be linked to the increasing fur seal population there (J Cooper, in litt.).

At Bird I, South Georgia, a survey in 1996 (DR Briggs & R Humpidge, unpubl. data) for comparison with similar surveys in each of the years 1979 to 1981 (Hunter 1984) revealed an estimated population of 2062 pairs, some 60% greater than the average of 1288 pairs in the earlier period. The increased availability of carrion from the expanding population of Antarctic Fur Seals *A. gazella* is regarded as the most likely cause of the population increase (González-Solís et al. 2000).

At-sea abundances of northern giant petrels in the Prydz Bay region of East Antarctica have decreased significantly since 1980/81 (Woehler 1996, Woehler & Watts 2000).

Southern Fulmar *Fulmarus glacialis*

The small population at Pointe Géologie has been counted every year since 1964 (Micol & Jouventin 2001). There is very high interannual variability in the breeding population because of the effect of environmental parameters such as snow cover early in the season affecting the availability of nesting sites. Overall, there is a significant increase in the population (Fig. 17), with a relatively low variance explained (21%) due to the inter-annual variability. No significant additional non-linear trends were detected (Table 1).

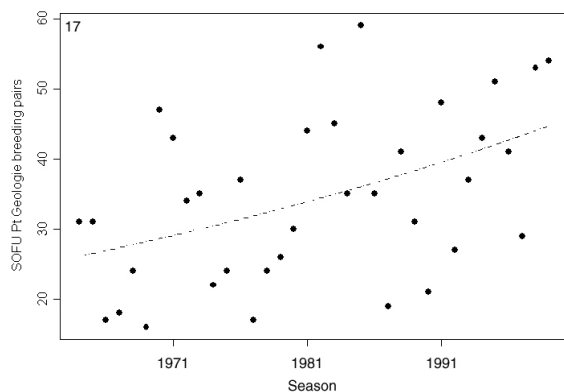


Figure 17. Plot of analyses of Southern Fulmar population data at Pointe Géologie. The dashed line is the linear regression fit to the data.

Cape Petrel *Daption capense*

Eight counts of the entire population breeding at Pointe Géologie have been made between 1985 and 1999 (Micol & Jouventin 2001). No significant linear trend was detected although the non-linear models suggest a consistent decrease during the 1980s and an increase thereafter (Fig. 18); a breakpoint was identified at 1989 (Table 1). The earlier decrease is related to the destruction of one breeding area during the construction of an airstrip in the late 1980s.

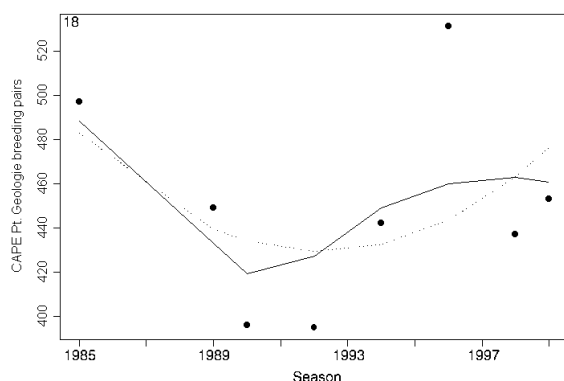


Figure 18. Plot of analyses of Cape Petrel population data at Pointe Géologie. The continuous line is the GAM fit to the data, and the dotted line the polynomial regression fit.

Snow Petrel *Pagodroma nivea*

Seven counts of the entire population of Pointe Géologie have been made between 1985 and 1999 (Micol & Jouventin 2001). Despite the destruction of several nesting sites in the late 1980s (due to airstrip construction), there has been no consistent decrease in the breeding population. As with Southern Fulmars, the availability of nesting sites varies extensively inter-annually, thus the low numbers of breeding birds during the two latest seasons should be interpreted with caution.

White-chinned Petrel *Procellaria aequinoctialis*

A paucity of population data precludes a statistical assessment for this species. At Bird I, South Georgia, counts of occupied burrows in 1981-1983 and 1996-1998 have revealed a significant decrease of 28% in the breeding population; very little of this decrease has been caused by destruction of

breeding habitat by Antarctic Fur Seals (Berrow et al. 2000a), suggesting that the population decrease is caused by factors operating away from the breeding colony. At-sea abundances of white-chinned petrels in the Prydz Bay region of East Antarctica have decreased significantly since 1980/81 (Woehler 1996, Woehler & Watts 2000). A monitoring programme for this species commenced at Marion I in 1997 (DC Nel unpubl. data).

White-chinned Petrels are one of the seabird species most commonly killed by longline fisheries for Patagonian toothfish (and ecologically similar species) in the Southern Ocean and adjacent regions (Barnes et al. 1997, Gales 1998). Their wide oceanic distribution during breeding (Weimerskirch et al. 1999, Berrow et al. 2000b) and their ability to feed during day and night renders them particularly vulnerable.

South Polar Skua *Catharacta maccormicki*

Few new data are available since the last summary. At Pointe Géologie, surveys spanning 33 years (28 surveys, Micol & Jouventin 2001) show no significant linear trend. The non-linear analyses indicate that the population increased from the late 1960s until the late 1970s when there was a decrease in the population (Fig. 19); a breakpoint occurred at 1984 (Table 1). The population increased linearly from 1966 to 1982 at 1.8% PA, and after 1984, the population increased at 3.7% PA. Elsewhere in East Antarctica, at Béchervaise I, there is a small population (two to four pairs) that may be increasing in association with the increasing Adélie Penguin population there (eight annual surveys, KR Kerry unpubl. data).

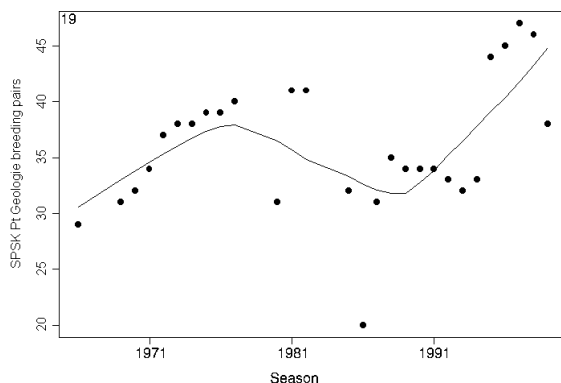


Figure 19. Plot of analyses of South Polar Skua population data at Pointe Géologie. The continuous line is the GAM fit to the data.

Populations at Cape Crozier (1962-1969, Wood 1971) and at Cape Bird (1987-1994) on Ross I were stable during their respective spans of data (Miller 1993, GD Miller unpubl. data). In the Antarctic Peninsula, the population at Cuverville I (KR Crosbie unpubl. data) was increasing until the last count in 1997. In the Palmer area, there has been a five-fold increase since the late 1970s, but due to recent reproductive failure the population around there may be levelling off or beginning to decrease (WR Fraser & DL Patterson unpubl. data). On King George I, the population around the Fildes Peninsula and Ardley I fluctuated through the 1980s but has increased for the past two counts in 1998 and 1999 (H.-U. Peter et al. unpubl. data). At Admiralty Bay, the population of South Polar Skuas which was established sometime after 1938, has seen a substantial increase between 1977-1999 (WZ Trivelpiece unpubl. data).

The populations of South Polar Skuas around East Antarctica (Pointe Géologie and Béchervaise I)

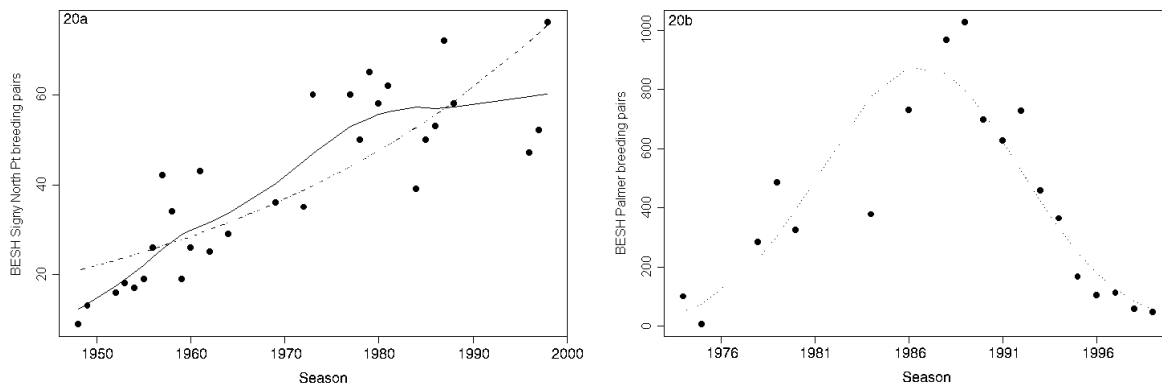
and at the most northerly locations in the Antarctic Peninsula region (Fildes Peninsula and King George I) are increasing. In contrast, the population around Palmer has grown substantially for the past 25 years but now appears to be levelling off. The most southerly populations for which we have data (Ross I) appear to have been stable for some time. There are no hypotheses for these differing population trends. Changes in the management of rubbish disposal at stations may have contributed to local changes.

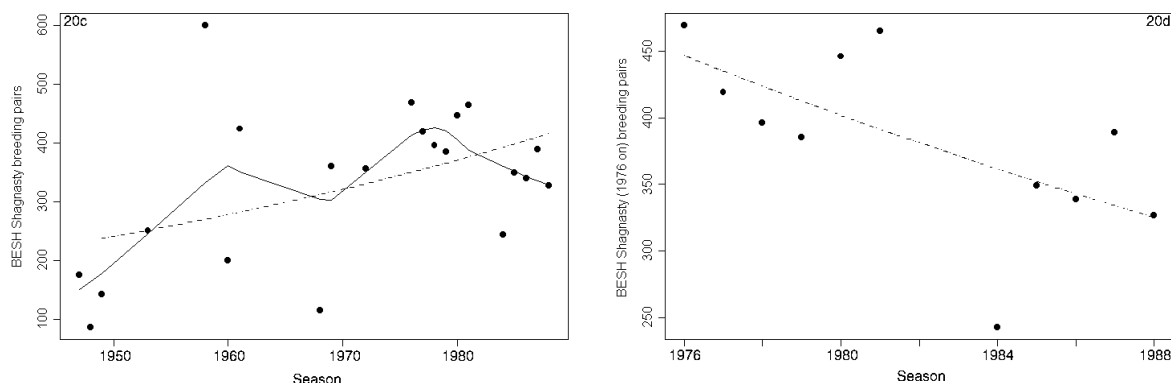
Subantarctic Skua *Catharacta antarctica*

The breeding population around Palmer has been stable for the last 20 years (WR Fraser & DL Patterson unpubl. data) and for the last 10 years at Admiralty Bay, King George I (WZ Trivelpiece unpubl. data). The population on Potter Peninsula has decreased by 30% over five years (7% PA, Hahn et al. 1998), whereas the population at Fildes Peninsula, King George I, has decreased by 30% over 19 years, (2% PA: H.-U. Peter et al. unpubl. data). Decreases in breeding populations at some stations may reflect improved treatment of waste, including the closing of outdoor rubbish dumps.

Blue-eyed Shag *Phalacrocorax atriceps*

Population data from four colonies, two on Signy I (North Point and Shagnasty), and two in the central Antarctic Peninsula area (Port Lockroy and Palmer) were examined. The North Point colony (Fig. 20a) shows a statistically significant linear increase (67% VE). Based on GAM (80% VE), the post-1981 data suggest that a plateau in the increase of this colony was reached in the mid-1990s and the population may now be stable or slowly decreasing (N Cobley unpubl. data). At Palmer, there were significant non-linear trends (Fig. 20b). The population increased until 1989 and decreased thereafter (83% VE) (Table 1). The post-1989 decrease in the population is correlated with the January 1989 *Bahia Paraiso* oil spill, but further analyses are needed to establish causality.





Figures 20a - d. Plots of analyses of Blue-eyed Shag population data at a) North Pt Signy I, b) Palmer, c) Shagnasty Signy I and d) Shagnasty excluding the 1976 datum. The continuous line is the GAM fit to the data, the dashed line the linear regression fit, and the dotted line the polynomial regression fit.

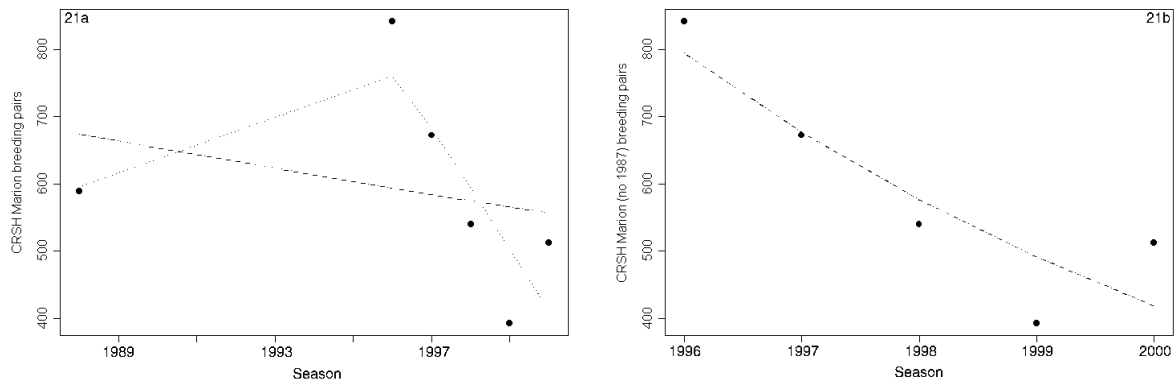
The Shagnasty colony also exhibited a statistically significant linear increase (21% VE) (Fig. 20c). The GAM model also indicated the presence of a cyclical component associated with the increase in this colony (Fig. 20c). Due to possible inconsistencies in the pre-1976 counts of the Shagnasty colony (N Cobley, unpubl. data), the data were also examined post-1976. This analysis indicates a statistically significant linear decrease in the population (39% VE) (Fig. 20d). The PLR analysis indicated a breakpoint between 1983 and 1984 (72% VE) (Table 1).

At Port Lockroy, a smaller data set indicated a statistically significant population decrease between 1981 and 1999 (N Cobley unpubl. data), but there are no data for this locality during the late 1980s to contrast with the nearby Palmer time series. The Port Lockroy time series, however, mirrors changes in the populations of this species reported for other Antarctic Peninsula colonies during the last decade (Woehler & Croxall 1997).

Populations of Blue-eyed Shags in the Antarctic Peninsula exhibit broad coherence over the region, with trends before the mid-1980s generally increasing and trends after this period decreasing. Additional anecdotal data confirm this pattern at other localities within the Peninsula area. Causal factors associated with these patterns have not been identified, but changes in abundance of juvenile year-classes of fish are likely to be involved.

Crozet Shag *Phalacrocorax atriceps melanogenis*

Six whole-island counts of Crozet Shags on Marion I have been made between 1988 and 1999 (a single count in 1988 and annual counts since 1995). A significant decrease was detected over the entire period, however the linear model only explained 8% of the variance. A polynomial model showed an increase between 1988 and 1995, followed by a steep decrease from 1995 to 1999 (68% VE), Fig. 21a. However, due to the variable breeding timetable of this species, the datum for 1988 and the first part of the polynomial curve should be treated with caution. The annual counts from 1995 to 1999 give a more reliable picture of recent trends, and these data show a significant decrease (77% VE), Fig. 21b. The rate of this decrease was 14.8% PA. Causal factors for the observed decrease are unknown.



Figures 21a-b. Plots of analyses of Crozet Shag population data at a) Marion I, b) Marion I excluding the 1988 datum. The dashed line is the linear regression fit to the data, and the dotted line the polynomial regression fit.

OTHER SOUTHERN OCEAN SEABIRDS FOR WHICH LONG-TERM DATA ARE ABSENT

Census data on a number of other species of Southern Ocean birds subsequent to the last review of populations and trends (Woehler & Croxall 1997) were reported to the Workshop. This included data from burrowing petrels, storm petrels, terns and sheathbills. None of these data allowed statistical analysis of population trends.

Wilson's Storm Petrel *Oceanites oceanicus*

New data for the colony at Three Brothers Hill, Potter Peninsula, King George I were estimated using different methods (Hahn et al. 1998). Populations of Wilson's Storm Petrels at this site are much higher than those reported by Araya & Arieta (1971) and Aguirre (1995) which may only illustrate the difficulties in obtaining population estimates, rather than a genuine population trend for storm petrels.

Burrowing petrels

Surveys of three species of burrowing petrels (Salvin's Prion *Pachyptila salvini*, Blue Petrel *Halobaena caerulea* and Great-winged Petrel *Pterodroma macroptera*) in four localities at Marion I in 1980 (when feral cats *Felis catus* were severely affecting these populations) and in 1997 and 1998 after the eradication of cats, suggest a further recovery in the population (Crawford et al. unpubl. data).

Antarctic Tern *Sterna vittata*

Antarctic Terns are very difficult to count. This species exhibits high variability in breeding pair numbers among seasons and colonies on King George I during the 1980s and 1990s (Peter et al. 1988, 1991, 1997).

Kelp Gull *Larus dominicanus*

Two recent censuses at Fildes Peninsula, King George I (Welcker & Peter 1999) showed no change to population estimates from the 1980s.

CONCLUSIONS AND RECOMMENDATIONS

Preliminary conclusions and recommendations from this review were as follows:

General

1. The review offers a good basis for appraising the availability of data for assessing population trends of Southern Ocean seabirds in relation to the distribution of the main populations of the principal species. Developing this overview is a high priority in order to advise on priorities for establishment of new monitoring programmes, in addition to help sustain the continuation of the main existing studies.
2. Workshop attendees recognised the value of maintaining the existing data set for similar assessments of seabird population data in the future. The Workshop welcomed the offer of the Australian Antarctic Data Centre's offer to continue hosting the password-protected database.

Population trends

3. Significant decreases in populations are evident for those species known to be caught on longline fisheries (albatrosses, Southern Giant Petrels and *Procellaria* spp.).
4. Substantial changes were noted for many of the penguin populations examined, but these varied in terms of degree and direction among species and geographical areas, so that no completely consistent overall pattern emerged. However, decreases in all penguins other than King Penguins in the Subantarctic was a feature of the data.
5. For some species of unfavourable conservation status (BirdLife International 2000) very limited, if any, data are available for assessment of population trends (eg Tristan Albatross *Diomedea dabbenena*, White-chinned Petrel, Spectacled Petrel *P. conspicillata* and Grey Petrel *P. cinerea*). Other species with notably inadequate data include Light-mantled and Atlantic Yellow-nosed Albatrosses.
6. Almost no data are available on trends in burrowing petrel species; however, obtaining such data present special difficulties. It would be optimistic to expect to detect other than very large-scale changes for these species.
7. No time-series population data appear to exist for Antarctic Petrels, a species that is endemic to the Antarctic continent and adjacent waters. Other characteristic Antarctic species for which there is little current population monitoring include Cape and Snow Petrels and Southern Fulmar.

Future Requirements

8. Population surveys would be enhanced by biological data (eg breeding success, recruitment and adult survival rates), which would improve our understanding of observed trends in populations. These supplementary data may also provide an early warning of changes to population trends.
9. There is a need to describe more explicitly (and refine where appropriate) the methods used to collect seabird population data. This is partly addressed in the CEMP protocols, but for species not included in the CEMP program, this is an issue of high priority. It was recognised that a number of species-specific protocols could usefully be developed (eg Emperor Penguins, all biennial-breeding species); the possible utility of a techniques and instruction manual was noted.
10. Time-series analytical methods should be used to examine long-term seabird data sets. Data holders are encouraged to undertake such analyses.
11. For the more complete and long-term datasets, investigation of potential interactions between population size and physical and biological environmental variables would be useful. Data holders are encouraged to undertake, and collaborate in, such investigations.
12. Comparisons of population trends, and timings of population change, across populations and species on regional bases might be a useful topic for future investigation.
13. The potential for short-term data sets (currently <10 years) to become useful for assessing long-term population trends (>10 years) was noted. Owners of such data sets were to be encouraged to provide these data provided data quality issues were addressed.
14. Attendees determined that ad-hoc, opportunistic collection of demographic data are of limited value for determining population sizes and of no value for investigating linkages between causality and population trends.

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Table 1. Results of Linear (LR), Piece-wise Linear Regression (PLR) and Generalized Additive Model (GAM) analyses for long-term population data for Southern Ocean seabirds.

Abbreviations used:

nf = not fitted, ss = restricted sample size so GAM not used, ns = not significant, ni = no improvement, (difference in proportion variance explained is less than 0.15).
 * = only results based on significant linear models are reported. All tests had $p < 0.05$ or less.

Species/Locality	Population trend from linear model	LR	Proportion of variance explained (VE) by analysis: PLR	PLR breakpt	GAM	poly	Comments
Emperor Penguin							
Pte Géologie	decrease	0.77	0.91	1979	0.91	nf	two levels of decrease
King Penguin							
Chaloupe	increase	0.88	ss		ss	nf	
Jardin Japonais	increase	0.83	ss		ss	nf	
Petite Manchoitière	increase	0.66	ss		ss	nf	exploratory nonlinear analyses indicate that there may be a leveling off or start of a decrease due to the last data point.
Heard	increase	0.98	ni		ni	nf	
Adélie Penguin							
Béchervaise I	increase	0.71	ni		ni	nf	looking at data after 1991 only, population is increasing though there are some nonlinear dynamics decrease and then stabilization
1990 onward	increase	0.40	0.90	1992	ss	nf	
Pte Géologie	increase	0.53	ni		0.85	nf	
Syowa	increase	0.47	0.70	1989	0.61	nf	increase, with high outlier 1989, then stable
Whitney Point	increase	0.96	ni		ni	nf	
King George	decrease	0.36	0.68	1990	0.58	nf	increase until 1990, then decrease
Palmer	decrease	0.55	ns		ni	nf	
excluding 1988	decrease	0.76	nf		0.88	nf	increase linear fit; some evidence of plateau in the 80s
Signy	increase	0.21	0.73	1988	0.67	nf	increase until 1988, then stable
Cape Bird	decrease	0.05	0.28	1984	0.82	0.64	cubic polynomial fit; cycle evident
Cape Crozier	increase	0.01	0.51	1986	ss	0.43	increase up to 1986 then decrease; nonlinearity driven by 1970 data point
excluding 1970	decrease	0.43	ni		ss	0.75	
Cape Royds	increase	0.81	ni		ni	ni	PLR (.89 VE) indicates a decrease from 1959 to 1964, and increase afterward; this decrease drives the GAM and cubic polynomial models (0.88 VE)

Table 1. Results of Linear (LR), Piece-wise Linear Regression (PLR) and Generalized Additive Model (GAM) analyses for long-term population data for Southern Ocean seabirds, continued.

Abbreviations used:

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 * = only results based on significant linear models are reported. All tests had $p < 0.05$ or less.

Species/Locality	Population trend from linear model	LR	Proportion of variance explained (VE) by analysis:			Comments
			PLR	PLR breakpt	GAM	poly
Gentoo Penguin						
Alice Creek	increase	0.75	ni		ni	nf
Signy	increase	0.84	ni		ni	nf
Damoy Pt	increase	0.41	0.81	1978	0.56	nf
King George	decrease	0.42	ns		0.61	nf
Bird Island	decrease	0.50	0.78	1986	ni	nf
Marion Island	increase	0.31	ns		ss	0.44
Chinstrap Penguin						
King George	decrease	0.84	ni		ni	nf
Palmer	increase	0.81	ni		0.94	nf
Signy	decrease	0.18	0.91	1991	0.31	nf
Macaroni Penguin						
Bird Island	decr (>1972 only)	0.60	0.86	1988	0.87	nf
Marion I:						
Archway Bay	no trend	ns	0.53	1990	0.30	nf
van de Boogaard	decrease	0.26	0.77	1989	0.53	nf
Macaroni Bay	decrease	0.64	0.81	1990	ni	nf
all 3 colonies	decrease	0.70	0.82	1990	ni	nf
Rockhopper Penguin						
Ile d'Amsterdam	decrease	0.65	0.84	1987	ni	nf
St Paul	increase	0.75	0.99	1986	ni	nf
Marion I	decrease	0.55	0.77	1992	0.70	nf

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Species/Locality	Population trend from linear model	LR	Proportion of variance explained (VE) by analysis:		Comments
		PLR	PLR breakpt	GAM	poly
Wandering Albatross					
Bird Island	decrease	0.52	0.72	1984	1984
Kerguelen	decrease	0.35	0.81	1991	1991
excluding 1972	increase	0.21	0.67	1991	1991
Marion Island	increase	0.05	ns		
excluding 1978	increase	0.69	ni		
Possession	decrease	0.13	ns		
Amsterdam Albatross					
Ile Amsterdam	increase	0.70	ni	ns	nf
Black-browed Albatross					
Bird Island:					
Colony H	decrease	0.76	0.89	1987	1987
Colony J	decrease	0.93	ni		
Kerguelen	no trend	ns	ns		
Yellow-nosed Albatross					
Ile Amsterdam	decrease	0.43	ns		
Gough I	decrease	0.19	0.35	1997	1997
Grey-headed Albatross					
Bird Island:					
Colony B	decrease	0.13	0.84	1993	1993
Colony E	decrease	0.25	0.75	1987	1987
Marion	increase	0.59	0.74	1990	1990
excluding 1990	increase	0.82	ni		
Sooty Albatross					
Possession	decrease	0.85	ni		

nf

1972 large residual

1978 large residual

5th order polynomial fit data best

results do not change significantly when the undercount in 1996 is excluded

stable, then decrease after 1993 decrease, then stabilizing after 1987 1991 large residual

Table 1. Results of Linear (LR), Piece-wise Linear Regression (PLR) and Generalized Additive Model (GAM) analyses for long-term population data for Southern Ocean seabirds, continued.

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Species/Locality	Population trend from linear model	LR	Proportion of variance explained (VE) by analysis: PLR breakpt	GAM	poly	Comments
Light-mantled Sooty Albatross						
Possession	decrease	0.07	0.75	1993	0.57	0.22 GAM is exploratory due to low sample size; span=0.75; indicates a decrease and then increase
Southern Giant Petrel						
Mawson	decrease	0.84	0.92	1981	ss	nf from PLR: decrease from 1957 to 1980; increase from 1981 to 2000; this pattern also detected in exploratory nonlinear analysis
Davis	decrease	0.22	0.63	1977	0.82	0.85 from PLR: decrease between 1971 and 1977; increase from 1978 to 2000; this pattern also seen in nonlinear models
Casey	no trend	ns	0.75	1982	ss	0.68 from PLR: decrease from 1956 to 1982; increase from 1983 to 1999. This pattern reflected in the polynomial model
Marion	no trend	ns	0.68	1990	0.81	0.64 span=0.75. Nonlinear models indicate an increase and then a decrease. GAM indicates a dip as well during the peak phase.
excluding 1977	decrease	0.30	0.72	1992	0.79	0.42 span=0.75
Possession	increase	0.20	0.76	1993	0.63	0.48 GAM is exploratory due to low sample size; span=0.75, decrease then increase but may be an overfit of model
King George	decrease	0.44	0.95	1988	0.95	0.95 GAM is exploratory due to low sample size; span=0.75; stable then declining after 1988
Palmer (chicks)	increase	0.22	0.83	1988	ni	nf 1998 is a very large residual; decrease then increase after 1988
Northern Giant Petrel						
Marion	increase	0.76	ni		ni	nf decrease and then increase
Possession	decrease	0.08	0.93	1993	ss	0.53
Southern Fulmar						
Pte Géologie	increase	0.21	0.73	1981	ni	nf high variability

Table 1. Results of Linear (LR), Piece-wise Linear Regression (PLR) and Generalized Additive Model (GAM) analyses for long-term population data for Southern Ocean seabirds, continued.

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* = only results based on significant linear models are reported. All tests had $p < 0.05$ or less.

Species/Locality	Population trend from linear model	LR	Proportion of variance explained (VE) by analysis: PLR	PLR breakpt	GAM	poly	Comments
Cape Petrel Pte Géologie	no trend	ns	0.42	1989	0.49	0.21	GAM is exploratory; span = 0.75; indication of decrease and then increase
South Polar Skua Pte Géologie	no trend	ns	0.65	1985	0.47	ns	PLR- increase more rapidly after breakpoint
Blue-eyed Shag Signy North Pt	increase	0.67	0.88	1971	0.81	nf	less pattern in the residuals with GAM; some evidence of plateau at the end of the data; PLR indicates increase until 1971, then stable
Shagnasty 1976 onward	increase	0.21	ns		0.51	nf	two up/down sections
Palmer	decrease	0.39	0.72	1983	ss	nf	stable until 1983, then decrease
	no trend	ns	0.77	1988	0.89	0.83	cycle in the data; peak in the mid-1980s
Crozet Shag Marion excluding 1987	decrease	0.08	ss		ss	0.68	
	decrease	0.77	ss		ss	ni	