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Population trends in a community of large Procellariiforms of Indian Ocean: Potential effects of environment and fisheries interactions

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ABSTRACT

Despite the worrying conservation status of several albatross and petrel population, the long-term trends of many populations remain largely unknown and the causes of decline in many cases are known or very strongly suspected to be incidental mortality in fisheries. Here we combine long-term monitoring of population trends, breeding success and band recoveries to examine the past and current status of five species of albatrosses and giant petrels breeding at the same site: sooty albatross (*Phoebetria fusca*), light-mantled albatross (*Phoebetria palpebrata*), wandering albatross (*Diomedea exulans*), northern giant (*Macronectes halli*) and southern giant petrels (*Macronectes giganteus*) on Possession Island, Crozet archipelago. We identified three groups of trends over a 25-years period (1980–2005) suggesting common underlying causes for these species in relation to their bioclimatic foraging ranges. The Antarctic species – light-mantled albatross and southern giant petrel – appeared stable and increased recently, the Sub-Antarctic species – wandering albatross and northern giant petrel – declined with intermediate periods of increase, and finally the subtropical species – sooty albatross – declined all over the period. Breeding success, indicative of environmental conditions, showed two kinds of pattern (low and fluctuating versus high and/or increasing) which were consistent with oceanographic variations as found in a previous study. We present the analysis of fisheries-related recoveries, indicative of fisheries bycatch risks showing specific catch rates. No direct relationship between population trends and longline fishing effort was detected, probably because census data alone are not sufficient to capture the potentially complex response of demographic parameters of different life stages to environmental variation. This study highlights the contrasted changes of procellariiform species and the particularly worrying status of the subtropical sooty albatrosses, and in a lesser extent of Sub-Antarctic species.

1. Introduction

Seabirds and particularly albatrosses and large petrels, are becoming increasingly globally threatened at a faster rate than any other species-groups of birds (Butchart et al., 2004; BirdLife International, 2007). Seabirds face a variety of threats, both on land and at sea. Currently the most critical conservation problem facing seabirds is considered to be bycatch caused by mortality in commercial longline and trawl fisheries (Gales, 1998; Tasker et al., 2000; Weimerskirch et al.,

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2000; Woehler et al., 2001; BirdLife International, 2004; Delord et al., 2005; Sullivan et al., 2006). Longlining is a widely used technique for a range of fish stocks and the impacts on seabird populations is documented only for a limited number of fisheries (Weimerskirch et al., 1997; Tuck et al., 2001; Cuthbert et al., 2003; Nel et al., 2003; Baker and Wise, 2005; Petersen et al., 2007). It is estimated that over 100,000 birds – including tens of thousands of albatrosses – were killed annually by fishing vessels, especially illegal, in the Southern Ocean alone (SC-CAMLR, 2005; BirdLife International, 2007). Moreover, the extreme life history attributes (high survival, low fecundity) that are characteristic of albatrosses and large petrels (Warham, 1990) make them particularly vulnerable to any elevated levels of mortality. Consequently long-term monitoring of population numbers and trends analyses are essential to assess the conservation status of these species.

The increasing focus on the plights faced by albatrosses and petrels has resulted in renewed interest and support of long-term population monitoring studies. Information on population status remains scarce for many group of species such as sooty albatrosses (*Phoebetria sp*) and especially petrels, whereas for other groups information on population trends exist at least for one or even several breeding sites within the species range (e.g. large albatrosses). Recent surveys have detected an improvement in the status of some Procellariiform populations, whilst in some populations the rates of decline appear to have accelerated (ACAP, 2005; Crawford et al., 2003; Ryan et al., 2003; Reid and Huin, 2005; Woehler et al., 2001). The reasons for these contrasted situations are still poorly understood.

Therefore, it becomes increasingly important to have information on poorly known groups to be able to understand the present status of albatrosses and petrels communities to be able to assess whether conservation measures have to be taken and whether the existing ones are effective in the different sectors of the Southern Ocean.

In general, previous studies have examined the status of single species. Here, we provide an overview of population sizes and recent trends analyses of a community of five species: three albatross species and two large petrel species listed as "Near Threatened" to "Endangered" (BirdLife International, 2007) breeding on the same site. In addition, we present the trends in breeding success, a parameter that is generally influenced by environmental factors rather than by direct fishery mortality, of the five species as well as the fishing-related recoveries of birds. The aim of this study is to assess the conservation status of these species breeding on Possession Island, Crozet archipelago, identified as an Important Bird Area (IBA; Catard, 2001). We were particularly interested to examine whether within a community of large albatrosses and petrels the trends of five species with different ecological niches were similar or different according to habitats preferences. Finally we discuss whether population trends could be related to environmental factors or to risks incurred from longline fisheries.

2. Methods

The study was carried out at Possession Island (46°25′S, 51°45′E), Crozet archipelago. Ornithological field assistants conducted standardized annual counts of incubating birds of the five surface-nesting Procellariiform species: on the east coast of the island for light-mantled albatrosses LMSA (*Phoebetria palpebrata*), on the north coast for sooty albatrosses SOAL (*Phoebetria fusca*) and on entire island for northern giant petrels NOGP (*Macronectes halli*), southern giant petrels SOGP (*M. giganteus*) and wandering albatrosses WAAL (*Diomedea exulans*) (Fig. 1). Sectors monitored annually for both light-mantled and sooty albatrosses are those where more than 70% of the island population breeds. Several colonies or coastal sections were monitored for each species (see Table 1). Data were available prior to 1980 for WAAL (Weimerskirch and Jouventin, 1987). All the counts from the early 1980s were conducted at dates set shortly after the egg laying was complete. Unfortunately, previous data concerning giant petrels (Voisin, 1968) in the late 1960s could not be included in the analysis due to non exhaustive counts and differences in sections monitored.

Table 1 summarizes the available information for the study period.

Breeding success was calculated as the number of fledged chicks over incubating pairs counted for LMSA, SOAL, NOGP, SOGP and WAAL. Data were modelled with linear (LR) and polynomial (PR) regressions against time to detect trends, and the proportion of variance explained was used to summarize the overall fit of the models to the data.

To analyse the overall population trends, we combined the time-series (41 years for wandering albatross, 26 years for light-mantled albatross, 27 years for sooty albatross, 26 years for southern and northern giant petrels) with missing observations, and made a log-linear regression model with Poisson error terms using the program TRIM (Trends and Indices for Monitoring Data; Pannekoek and van Strien, 1996). To obtain the overall estimated breeding numbers on the monitored sites for each species, we used the population size estimates together with their standard errors obtained from the TRIM analysis. Because we were interested in identifying the changes in population trends across years, we started the analysis with a model with change points at each time-point, and used the stepwise selection procedure to identify change points with significant changes in slope based on Wald tests with a significance-level threshold value of 0.01 (Pannekoek and van Strien, 1996). We took into account over-dispersion and serial correlation since they can have important effects on standard errors, although they have usually only a small effect on the estimates of parameters (Pannekoek and van Strien, 1996). No covariate was used. Annual population rates of changes were calculated, for each species, using the relationship:

$$r = \ln \lambda = \ln \frac{N_{t+1}}{N_t}$$

where N_t and N_{t+1} are the number of pairs breeding in year t and $t+1$ respectively (taken to be the number of incubating birds counted in year t and $t+1$) and λ the population growth rate (Caughley, 1980). It was assumed that all the incubating birds were detected. N_{t+1}, N_t and λ where given by TRIM. All population size estimates are presented ±1 SE.

In biennial breeding species, pairs that raise a chick to fledging breed only every other year. Therefore, to calculate

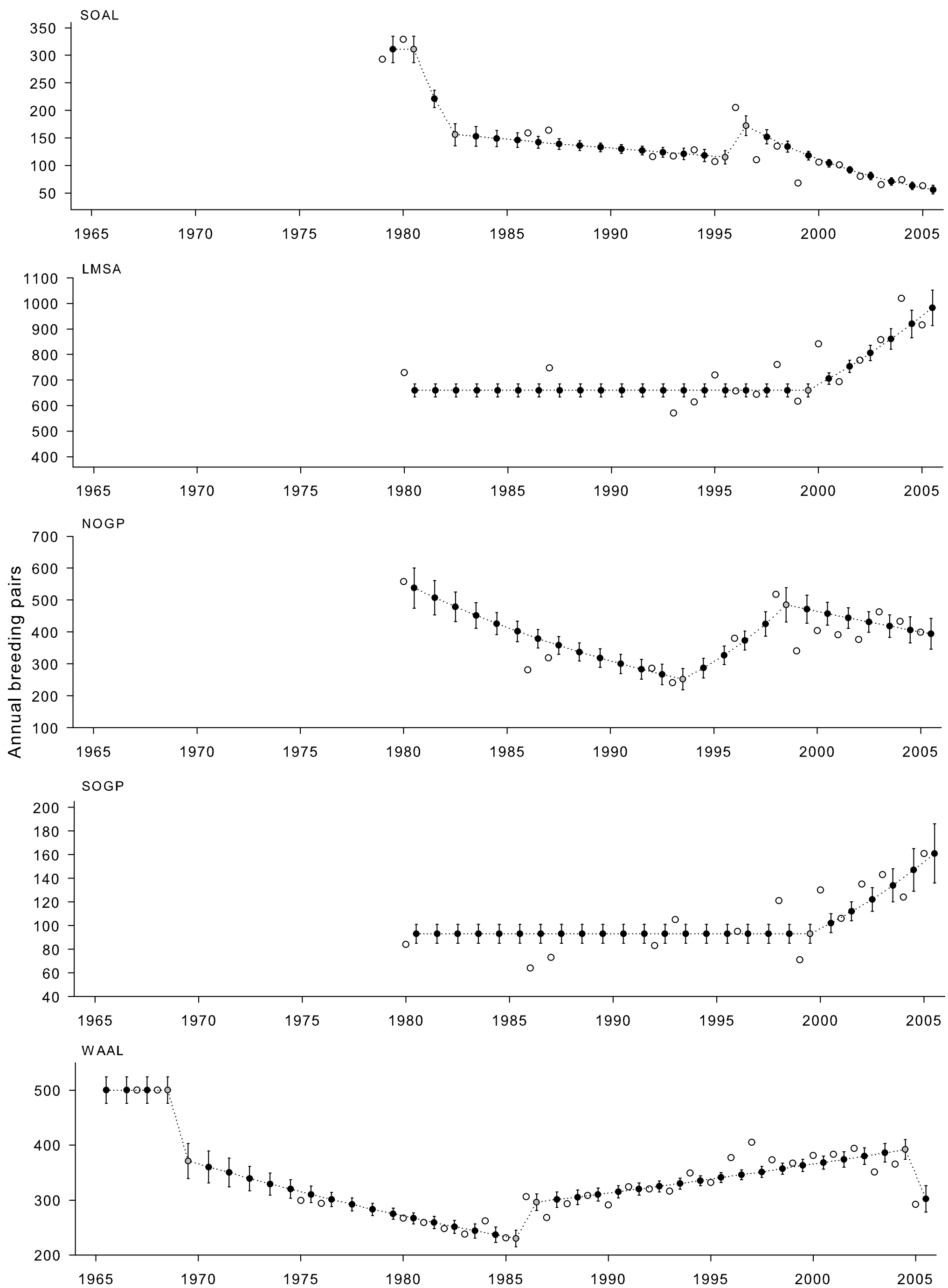

Fig. 1 – Estimates of annual breeding population size of sooty albatrosses (SOAL), light-mantled albatrosses (LMSA), northern giant petrels (NOGP), southern giant petrels (SOGP) and wandering albatrosses (WAAL) at Possession Island, Crozet archipelago, from 1979 (1965 for WAAL) to 2005. *Black dots* indicate the number of annual breeding pairs estimated from the time varying model with change points using program TRIM (see Section 2). *Grey dots* indicate the significant change points. Errors bars indicate ± SE. *White dots* indicate the number of annual breeding pairs counted in years where counts for all sectors were available.

the total breeding population, it is necessary to take into account the proportion of failed breeders that breed again the following year. The SOAL, LMSA and WAAL breed once every two years with an average of 89% (SOAL and LMSA) and 84% (WAAL) of failed breeders breeding again the following year (Jouventin and Weimerskirch, 1988). If we assume that on average about 34% (SOAL), 65% (LMSA) and 35% (WAAL) of breeders fail (Weimerskirch and Jouventin, 1998; Weimerskirch H., unpubl.data) it is possible to estimate the overall number of pairs breeding at Possession Island in years t and $t+1$ ($B_{t,t+1}$) as

Table 1 – Study period, breeding frequency, number of breeding localities, percentage of missing data, main foraging habitat and documented fisheries-related mortality of five species (considered in this paper) breeding on Possession Island, Crozet archipelago (Weimerskirch et al., 1985; Weimerskirch et al., 1987; Ridoux, 1994; Weimerskirch, unpubl. data; Stahl et al., 1996)

Species	Study period	Breeding frequency	Breeding localities	Number of colonies or coastal sections monitored	Percentage of missing data	Main foraging habitat	IUCN Red List Category	Bycatch concern with fisheries
Sooty albatross	1979–2005	Biennial	North coast	4	34%	Sub-Antarctic and Sub-Tropical waters	Endangered[c]	Yes[a]
Light-mantled albatross	1980–2005	Biennial	East coast	7	38%	Antarctic and Sub-Antarctic waters	Near Threatened[c]	Yes[a]
Northern giant petrel	1980–2005	Annual	Whole island	14	52%	Sub-Antarctic and Sub-Tropical waters	Near Threatened[c]	Yes[a]
Southern giant petrel	1980–2005	Annual	Whole island	6	43%	Antarctic and Sub-Antarctic waters	Near Threatened[c]	Yes[a]
Wandering albatross	1965–2005	Biennial	Whole island	3	20%	Sub-Antarctic waters	Vulnerable[c]	Yes[b]

a Gales et al. (1998).
b Weimerskirch et al. (1997).
c BirdLife International (2007).

$$\text{for } SOAL, B_{t,t+1} = N_t + N_{t+1} - 0.89^{*}0.34^{*}N_t, \quad (1)$$
$$\text{for } LMSA, B_{t,t+1} = N_t + N_{t+1} - 0.890.65^{*}N_t, \quad (2)$$
$$\text{for } WAAL, B_{t,t+1} = N_t + N_{t+1} - 0.84^{*}0.35^{*}N_t, \quad (3)$$

Thus we used the population size estimates from the TRIM analysis to obtain the overall estimated breeding numbers of SOAL, LMSA and WAAL. Note that this is a minimum estimate since the percentage of birds that lost their egg before counts and the percentage of birds that laid their egg after counts were not estimated, the latter being negligible. For biennial species the variance in population size was estimated using the delta method (Seber, 1982).

Since banding programmes have been carried out on Possession for LMSA, SOAL, NOGP, SOGP and WAAL we analysed the data of fisheries-related (longliners) recoveries of birds banded at Possession Island, outside their native (breeding) sites. The recoveries concerned the period 1984 to 1999. Previous recoveries are synthesised in Weimerskirch et al. (1985).The locations of recoveries are summarized in Fig. 4. Recovery rates were calculated as the number of banded birds recovered divided by the number of birds banded. Since we cannot quantify recovery effort, this analysis was dependent on those data made available to the authors, with inherent limitations such as the distribution of legal fishing effort (and the presence of on-board observers), since Illegal, Unreported and Unregulated (IUU) fisheries do not communicate any recovery data, or the proportion of birds killed that might be lost before hauling. Such limitations introduced unknown biases into the present analyses.

Breeding seabirds can be affected by several oceanographic factors such as sea surface temperature or sea ice extent (Croxall et al., 2002). The at sea distribution of the studied species excluded sea ice as a covariate. Because sea surface temperatures were measured only since 1981 we did not consider sea surface temperature as a suitable covariate to test for environmental effects on the variation of breeding population sizes. We used a large-scale climate index, the Southern Oscillation Index (SOI), which provides an index of oceanographic and climatic conditions associated with changes in marine food webs (Comiso et al., 1993) and population dynamics in some seabirds (Stenseth et al., 2002; Nevoux et al., 2007). The SOI was obtained from the Australian Bureau of Meteorology (http://www.bom.gov.au/climate/current). Negative values of SOI correspond to El Niño years whereas positive values correspond to La Niña years. According to band recoveries, ecological studies, radio-tracking and at sea observations from Possession Island (Weimerskirch et al., 1985; Ridoux, 1994; Stahl et al., 1996; Pinaud and Weimerskirch, 2007; H. Weimerskirch et al. unpubl. data) we restricted covariates to the geographical area frequented (30–75°E/30–50°S). We tested for effects of longliners targeting the Patagonian toothfish (*Dissosticus eleginoides*) – since early 1990s – and tuna (*Thunnus* spp.) – since late 1960s – on the subtropical high seas of the Indian Ocean. Regarding toothfish longline fishery, an illegal unreported and unregulated (IUU) fishing effort started since 1996–97 within the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR) statistical areas 58.5.1, 58.5.2, 58.6 and 58.7 (around Kerguelen, Heard, Crozet and Marion Islands). Estimates of IUU and legal fishing efforts were obtained from the CCAMLR annual reports. We then pooled annual legal and estimated IUU efforts. Tuna fishing effort were extracted from the Commission for the Conservation of the Southern Bluefin Tuna (CCSBT, http://www.ccsbt.org) and from the Indian Ocean Tuna Commission (IOTC, http://www.iotc.org) sources.

The effect of environmental covariates was not tested on breeding success data as this was done in a previous study (Inchausti et al., 2003). To test for environmental effects (SOI, toothfish and tuna longline fishing efforts) on the number of annual breeding pairs, we used generalized linear models with the glm function of software R (Faraway, 2006; R Development Core Team, 2005). We investigated the effects of the SOI of the current year and of fishing effort with a lag of 1 year. Fishing effort data were transformed using the equation $\log(x + 1)$, where x is the fishing effort in millions of hooks. There was no significant correlation between environmental covariates (all p's > 0.05). Not surprisingly, population time-series were significantly positively autocorrelated with a one year lag for the five species (autocorrelation function: all p's < 0.05). For each species, including biennials, we thus started with a model were the number of breeding pairs in year $t(N_t)$ estimated using TRIM was a function of the number of breeding pairs in year $t - 1(N_{t-1})$ to account for autocorrelation. We then tested for trends in breeding population sizes with linear, quadratic and cubic effects, and for effects of environmental covariates using a forward step by step variable selection. Model selection was performed using the Akaïke Information Criterion (AIC, Burnham and Anderson, 2002) and the model with the lowest AIC was retained at each step.

3. Results

3.1. Population trends

3.1.1. SOAL Sooty albatross

The number of SOAL breeding annually decreased by 82% between 1979 and 2005. The stepwise procedure for selection of change points indicated four significant change points (1980, 1982, 1995, 1996; all $p < 0.01$ for Wald tests) (Appendix A; Fig. 1). The long-term trend can be separated into two major periods of decline: (1) a dramatic decrease in early 1980s, followed by (2) a slower but significant decrease until 2005. A deceleration of this decline trend probably occurred around the 1982–1985 period, taking into account the biennial breeding frequency of the SOAL and the missing observations on the monitored sites.

The annual breeding population size estimates computed from this model indicated an average growth rate of −4.2% per year ($\lambda = 0.959 \pm 0.005$, CI 95% (0.948–0.969)) between 1979 and 2005. Between 1979 and 2005 the overall estimated number of total breeding pairs (see Eq. (1)) declined from 530 ± 42 to 100 ± 13.

3.1.2. LMSA light-mantled albatross

From 1980 to 2005 the breeding population increased (+49%) at an annual rate of +1.1% ($\lambda = 1.011 \pm 0.003$, CI 95% (1.005–1.017)). The stepwise procedure for selection of change points indicated one significant change point (1999; all $p < 0.01$ for Wald tests) (Appendix A; Fig. 1). The population size estimates computed from this model indicate a stable population from 1980 to 1999 with important inter-annual differences and a rate of +6.6% per year between 1999 and 2005.

The estimates of total breeding pairs (see Eq. (2)) increased from 938 ± 45 to 1370 ± 104 between 1980 and 2005.

3.1.3. NOGP Northern giant petrel

Although this species displayed important annual fluctuation in numbers of breeders (see Appendix B) with a decrease (−27%) between 1980 and 2005, no significant long-term trend was detected (average annual growth rate of −0.0004% ($\lambda = 1.000 \pm 0.006$, CI 95% (0.987–1.011))). The stepwise procedure for selection of change points indicated two significant change points (1993, 1998) (Appendix A; Fig. 1) indicative of (1) a decreasing phase at an average rate of −5.8% per year between 1980 and 1993, followed by (2) an increase at an average rate +13.1% per year between 1993 and 1998 and (3) a final period of decrease at an average rate −2.9% between 1998 and 2005.

3.1.4. SOGP Southern giant petrel

As for NOGP, we found important inter-annual variations of the breeding population (Appendix B). For the period 1980–2005, the breeding population increased (+73%) at an annual rate of +1.6% ($\lambda = 1.016 \pm 0.006$, CI 95% (1.003–1.028)). The stepwise procedure for selection of change points indicated one significant change point (1999) (Appendix A; Fig. 1). The population size estimates computed from this model indicate a stable population from 1980 to 1999 and a period of increase at an average rate of +9.2% per year between 1999 and 2005.

3.1.5. WAAL Wandering albatross

Even if the number of WAAL breeding annually decreased by 40% between 1965 and 2005, this overall trend was slightly not significant (average annual growth rate of −0.2% ($\lambda = 0.998 \pm 0.002$, CI 95% (0.994–1.001))). The stepwise procedure for selection of change points indicated five significant change points (1968, 1969, 1985, 1986, 2004) (Appendix A; Fig. 1). The annual population size estimates computed from this model with five change points indicate (1) a continuous decline period from the beginning of the monitoring period (1965) to 1985 with a dramatic decline probably between the late 1960s and early 1970s, followed by (2) a slight trend to increase from late 1980s to 2004 and a final period (3) of decrease since then. Between 1965 and 2005 the overall estimated number of total pairs breeding (see Eq. (3)) declined from 853 ± 42 to 579 ± 35.

3.1.6. Community trends

First, LMSA and SOGP had very similar trends, being stable and finally increasing during the last five years of the study period (Fig. 2a). Second, SOAL, NOGP and WAAL declined overall over the period but with periods of increase. The decreases took place for SOAL and NOGP in early 1980s, as did WAAL where the longer record indicates that the decrease started in late1960s or early 1970s (see Fig. 1).

3.2. Breeding success

The breeding success was highly variable except for WAAL from 1981 to 2005 (Table 3 Fig. 3). LMSA and SOGP had the

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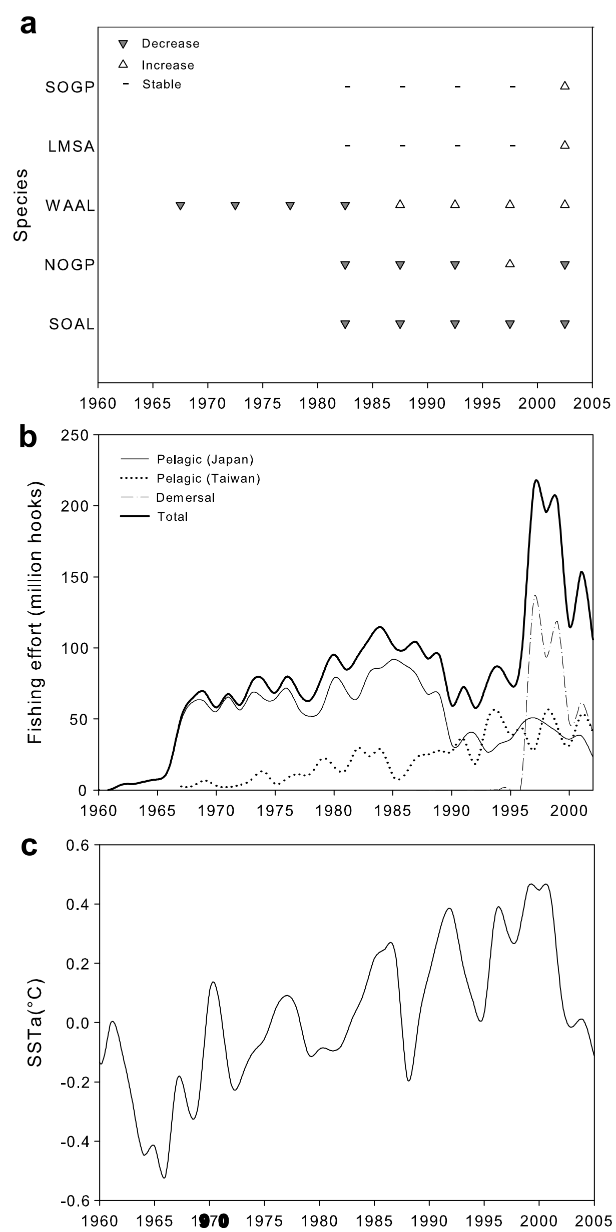

Fig. 2 – Population trends of annual breeding population size of sooty albatrosses (SOAL), light-mantled albatrosses (LMSA), northern giant petrels (NOGP), southern giant petrels (SOGP) and wandering albatrosses (WAAL) at Possession Island, Crozet archipelago, from 1979 to 2005. The trends estimated using program TRIM (see Appendix A) are summarised in decrease (*full triangle down*), increase (*open triangle up*) or stable (*horizontal mark*) trends for 5-year period (1960/1965, 1965/1970...); (b) Annual reported hooks from pelagic longline fisheries in the Southern Indian Ocean, south of 30°S (Japanese and Taiwanese) and from the demersal longline fisheries (legal and Illegal, Unreported and Unregulated) in the CCAMLR Crozet EEZ 58.6 (from Tuck et al., 2003); (c) Mean monthly variations of the sea surface temperature anomaly (SSTa) in the Southern Indian Ocean, south of 35°S (from NOAA, ERSST2 SST data (Smith and Reynolds, 2003; http://iridl.ldeo.columbia.edu/SOURCES/.NOAA/.NCDC/.ERSST/.version2)).

lowest average ± SE breeding success (respectively 40.9 ± 3.8% and 42.6 ± 4.9%) and WAAL the highest (73.6 ± 0.7%).

Breeding successes of the five species were not correlated ($p > 0.05$). Significant long-terms trends of the breeding success were found for SOAL and NOGP. The breeding success of the SOAL tended to increase up to 1995 (inflection point of the quadratic regression; Table 3) and then decreased (Fig. 3). The breeding success of the NOGP increased ($p = 0.014$) during all the study period (1986–2005) (Fig. 3). Although not significant at the 0.05 level (P = 0.106), the breeding success of LMSA tended to be higher in the 1980s (35 ± 6%) than in the late 1990s and early 2000s (47 ± 4%) (Fig. 3). The breeding success of WAAL (Fig. 3) and of SOGP (Fig. 3) showed no trend ($p > 0.05$) during the study period.

3.3. *Recovery rate and fisheries-related mortality*

Records of band recoveries were obtained from fisheries for four of the species (no recovery for LMSA). The recovery rate appeared to differ between the four species reported incidentally caught by longliners, with the NOGP showing the highest recovery rate (Table 4).

The analysis of the recoveries indicate that the proportion of birds recovered by longliners significantly differed between the four species ($\chi^2_4 = 13.99$, $p = 0.007$). The proportion of WAAL and NOGP reported in longline appeared to be higher than for other species (Table 4). Furthermore, the proportion of the recoveries related to bycatch in fisheries appeared smaller for SOGP than for WAAL ($\chi^2_1 = 3.57$, $p = 0.059$) and NOGP ($\chi^2_1 = 4.50$, $p = 0.034$). All the recoveries were reported south of 38°S, in an area between South East Atlantic to South Australia (Fig. 4) and mainly north from Crozet. For all species, the proportion of immature birds that were recovered (67.67%) tended to be slightly higher than for adults ($n = 11$; $\chi^2_1 = 2.64$, $p = 0.10$).

3.4. *Effects of environmental factors on breeding population sizes*

For LMSA, SOGP, and WAAL none of the environmental covariates was found to affect the estimated annual breeding pairs (Table 5). For NOGP and SOAL cubic effect of the fishing effort for toothfish was detected. However, we have to be careful in drawing conclusions since toothfish fishing effort presented a significant cubic trend over time (lowest AIC among models without trend, and with linear, quadratic and cubic trends). For SOAL, the lowest AIC model suggested a positive effect of SOI on annual breeding population size, although the 95% confidence interval of the slope parameter marginally included zero. Hence, during El Niño years the number of breeding pairs of SOAL decreased.

4. Discussion

4.1. *Contrasted trends*

Our results show that among the community, the five species show contrasted trends. In this community of large Procellariiforms we can distinguish three groups of species showing

Table 2 – Annual population size and status of five species (considered in this paper) breeding on Possession Island, Crozet archipelago

Species	Location	Breeding pairs (annual)	Study period	Population status	Source
Sooty albatross	Gough Island	<5000	1972–2000	Declining	Cuthbert and Sommer (2004)
	Tristan da Cunha Group	4125–5250	1972/74 and 1983/84	Unknown	See Gales et al. (1998) for review
	Prince Edward and Marion Is.	1564	1996–2001	Declining	Crawford et al. (2003), Ryan et al. (2003)
	Amsterdam Is.	470	1980s–2003	Unknown	H. Weimerskirch unpubl. data
	Crozet Is.				
	Possession Is.	114	1979–2005	Declining	This study
	Est Is.	1300	1982	Unknown	Jouventin et al. (1984)
	Cochons Is.	400–500	1976	Unknown	Jouventin et al. (1984)
	Pingouins Is.	250	1982	Unknown	Jouventin et al. (1984)
	Apôtres Is.	20–30	1982	Unknown	Jouventin et al. (1984)
Light-mantled albatross					
	South Georgia	5000	1983	Unknown	Thomas et al. (1983)
	Marion Is.	167	1980s–2005	Stable	Crawford et al. (2003)
	Prince Edward Is.	92	2002	Unknown	Ryan et al. (2003)
	Kerguelen Is.	3000–5000	1984–1987	Unknown	Weimerskirch et al. (1988)
	Heard and McDonald Is.	200–500	1954	Unknown	Gales (1998)
	Macquarie Is.	1100	1999	Stable	Terauds (2000)
	Auckland, Campbell and Antipodes Is.	6 800–6 900	1975and 1995/96	Unknown	Taylor (2000)
	Crozet Is.				
	Possession Is.	916	1980–2005	Increasing	This study
	Est Is.	>900	1982–1995	Unknown	Jouventin et al. (1984)
	Cochons Is.	50–100	1976	Unknown	Jouventin et al. (1984)
	Pingouins Is.	30	1982	Unknown	Jouventin et al. (1984)
	Apôtres Is.	150	1982	Unknown	Jouventin et al. (1984)
Northern giant petrel					
	South Georgia	2 062	1979–1996	Increasing	González-Solís et al. (2000)
	Marion Is.	350	1980s–2003	Declining	Crawford et al. (2003)
	Prince Edward Is.	300	2001	Unknown	Ryan et al. (2003)
	Kerguelen Is.	1 450–1 800	1984–1987	Unknown	Weimerskirch et al. (1988)
	Macquarie Is.	1 200	2000	Stable	Patterson et al. (in press), Terauds (2000)
	Crozet Is.				
	Possession Is.	399	1980–2005	Increasing	This study
	Est Is.	190	1982–1995	Unknown	Jouventin et al. (1984)
	Cochons Is.	250–300	1976	Unknown	Jouventin et al. (1984)
	Pingouins Is.	165	1982	Unknown	Jouventin et al. (1984)
	Apôtres Is.	150	1982	Unknown	Jouventin et al. (1984)
Southern giant petrel					
	Falkland Is.	19810	2004	Increasing	Reid and Huin (2005)
	South Georgia	4654	1958–1996	Stable	González-Solís et al. (2000)
	Marion Is.	1 759	1980s–2003	Fluctuating	Crawford and Cooper (2003), Nel et al. (2002) Q5
	Prince Edward Is.	1 000	2001	Unknown	Ryan et al. (2003)
	Heard Is.	3600	?	Unknown	Patterson et al. (in press), Baker et al. (2002)
	Macquarie Is.	2000–2200	2000	Stable	Terauds (2000)
	Gough Is.	225–245	2002	Increasing	Cuthbert and Sommer (2004)
	Eastern Antarctica				
	Frazier	248	1956–2001	Stable	Creuwels et al. (2005)
	Mawson	250	1957–2000	Declining	Woehler et al. (2003)
	Davis	25	1963–1999	Declining	Woehler et al. (2003)
	Casey	248	1955–2001	Increasing	Woehler et al. (2003)
	North Patagonia				
	San Jorge Is.	2300	2004	Increasing	Quintana et al. (2006)

(continued on next page)

Table 2 (*continued*)

Species	Location	Breeding pairs (annual)	Study period	Population status	Source
	Crozet Is.				
	Possession Is.	161	1980–2005	Increasing	This study
	Est Is.	323	1982–1995	Unknown	Jouventin et al. (1984)
	Cochons Is.	550–600	1976	Unknown	Jouventin et al. (1984)
	Pingouins Is.	50	1982	Unknown	Jouventin et al. (1984)
	Apôtres Is.	10	1982	Unknown	Jouventin et al. (1984)
Wandering albatross					
	South Georgia	2 480	1976–2004	Declining	Poncet et al. (2006)
	Marion Is.	1 436	1985–2004	Stable	Crawford et al. (2003), Nel et al. (2002)
	Prince Edward Is.	1 687	2001	Unknown	Ryan et al. (2003)
	Kerguelen Is.	1 455	1971–1992	Stable	Weimerskirch et al. (1988)
	Macquarie Is.	<10	2005	Stable	Terauds et al. (2006)
	Crozet Is.				
	Possession Is.	292	1965–2005	Increasing	This study
	Est Is.	325	1975–1982	Declining	Weimerskirch and Jouventin (1997)
	Cochons Is.	1 060	1975–1981	Declining	Weimerskirch and Jouventin (1997)

Table 3 – Breeding success (mean ± SE; coefficient of variation CV) and results of LR and PR of second and third order for long-term breeding success data for five species on Possession Island, Crozet archipelago, from 1981 to 2005. All tests had $P < 0.05$ or less

Species	Breeding success (%)		LR: r^2 adj	slope	PR: r^2 adj	Quadratic slope	Cubic slope
	Mean ± SE (min–max)	CV					
SOAL	65.4 ± 2.7 (29–86)	0.204	NS		0.250	−0.142	
LMSA	40.9 ± 3.8 (3–74)	0.422	NS		NS		NS
NOGP	53.3 ± 3.5 (32.7–82.2)	0.264	0.317	1.499	NS		NS
SOGP	42.6 ± 4.9 (1.8–74.3)	0.461	NS		NS		NS
WAAL	73.6 ± 0.7 (67–80.6)	0.047	NS		NS		NS

SOAL, Sooty albatross; LMSA, light-mantled albatross; NOGP, Northern giant petrel; SOGP, Southern giant petrel; WAAL, Wandering albatross; NS, not significant.

different trends over the period 1980–2005 (see Fig. 2). Firstly, the two species with an Antarctic/Sub-Antarctic distribution, LMSA and SOGP, had very similar trends. On Possession Island, our results suggest that the population of LMSA was stable and then increased recently. Although the model failed to find significant change point before late 1990s, probably because of the high inter-annual variation in numbers (see Appendix B), the inputed data appeared to confirm the decline of 13% reported by Weimerskirch and Jouventin (1997) for the period 1981–1995 (see Fig. 1). The SOGP showed the same pattern (+73%), with a breeding population stable followed by an increase initiated in late 1990s.

The trends observed for the second group of species (NOGP and WAAL which are Sub-Antarctic/Sub-Tropical species), were initially declining numbers of breeding pairs (in the late 1960s early 1970s for WAAL and in the early 1980s for NOGP) and then recovering since the late 1980s for WAAL and the early 1990s for NOGP but insufficiently to compensate for the decreases. Therefore, current breeding populations of these species remain lower than at the beginning of the monitoring period. For both species, there was no direct relationship between the numbers of breeding pairs and both fishing effort and SOI, making it difficult to understand the causes of trend reversals. This is surprising since previous studies suggested that bycatch mortality has been implicated in the population decline (Weimerskirch et al., 1997; Tuck et al., 2001). Because population fluctuations result from a balance between mortality, recruitment, immigration and emigration, more detailed studies on the relationships between fishing effort, climate and demographic parameters such as annual survival and recruitment are needed for a better understanding of the population dynamics of these species.

Finally, SOAL the only Sub-Tropical species showed a continuous decline over the entire study period with the most important decrease observed in the early 1980s. The successive reversal trends detected on two consecutive years

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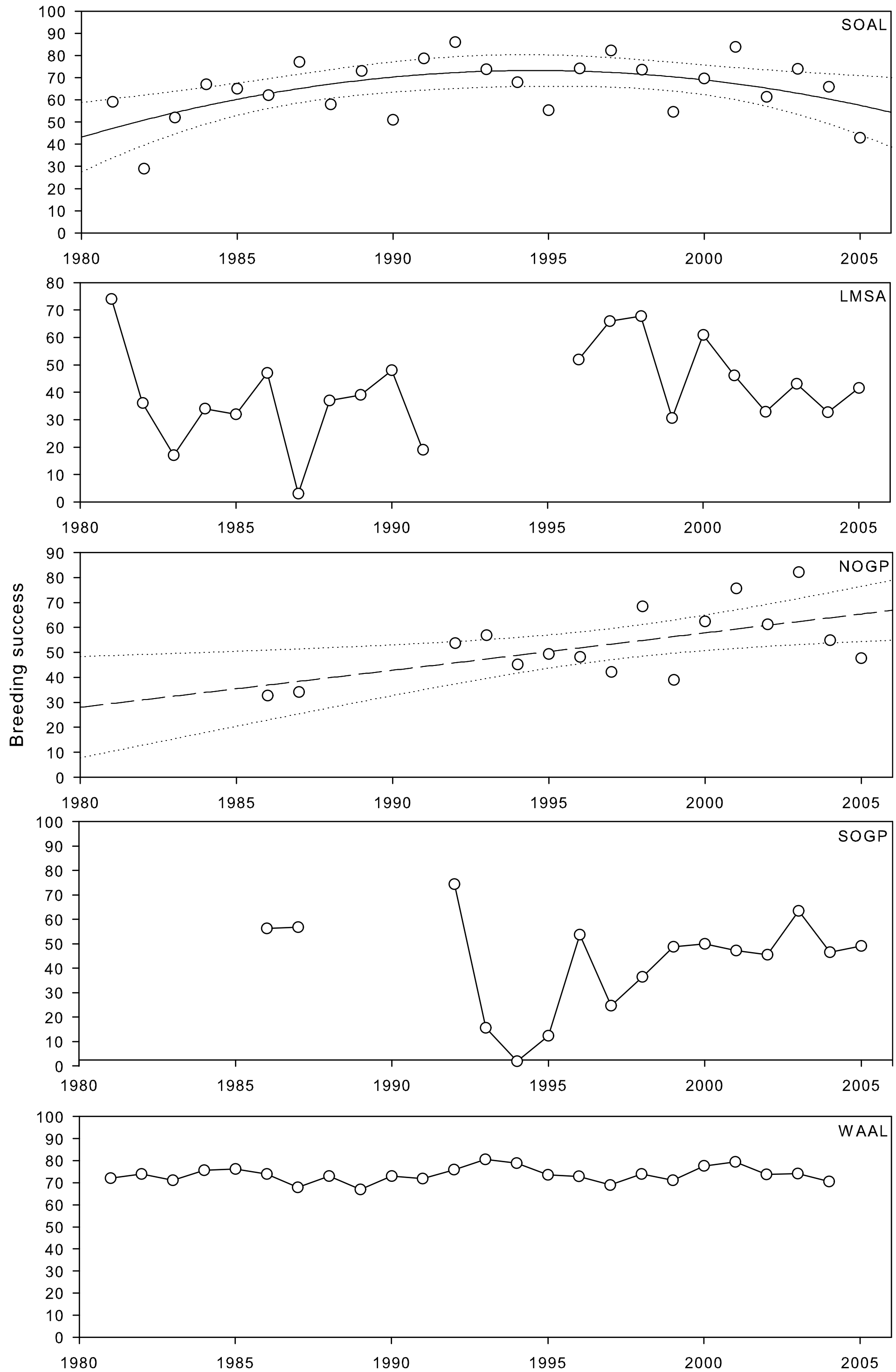

Fig. 3 – Comparison of inter-annual breeding success models fitted to data sets, 1980s–2000s, for sooty albatrosses (SOAL), light-mantled albatrosses (LMSA), northern giant petrels (NOGP), southern giant petrels (SOGP) and wandering albatrosses (WAAL) breeding on Possession Island, Crozet archipelago. Lines indicates predictions of LR model (NOGP) (*dashed line*), and PR models of second (SOAL) and third order (SOGP) (*solid line*) with the 95% CIs (*dotted line*).

(1995–1996) are also probably due to the biennial breeding. The decline of SOAL is the greatest among the species studied and is still occurring, although at a lower rate than during the 1980s.

4.1.1. *Global trends and status*

Interestingly the long-term trends we observed for the Possession community were very similar to the nearby population trends on Marion Island during a shorter period

Table 4 – Summary of fisheries-related recoveries compared to resightings data outside their native (or breeding colony) sites of five species of birds banded at Possession Island, Crozet archipelago

Species	Numbers banded		Total recoveries % (*n*)	Fisheries-related recoveries (mortality)			
	Adults	Fledglings		Fledglings/juveniles	Immature	Adults	Total % (*n*)
SOAL	1170	1548	0.67 (1)	0	1	0	0.04 (1)
LMSA	682	507	0.67 (1)	–	–	–	–
NOGP	710	2151	38.93 (58)	1	4	1	0.25 (7)
SOGP	711	962	8.05 (12)	0	1	0	0.08 (1)
WAAL	3350	6250	51.68 (77)	0	3	3	0.06 (6)
Total (*n*)	6623	11 418	149	1	9	4	15

SOAL, Sooty albatross; LMSA, light-mantled albatross; NOGP, Northern giant petrel; SOGP, Southern giant petrel; WAAL, Wandering albatross.

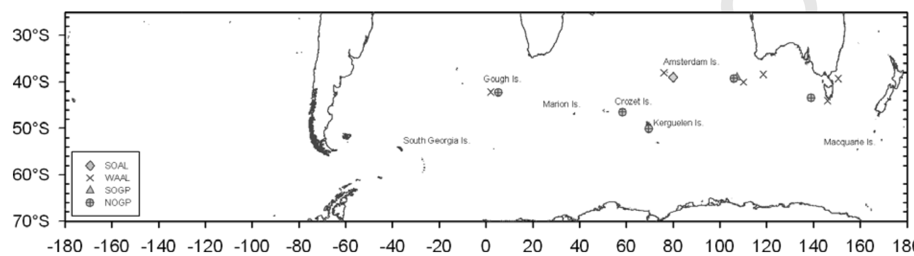

Fig. 4 – Localities of the fisheries-related (longliners) recoveries of birds sooty albatrosses (SOAL), light-mantled albatrosses (LMSA), northern giant petrels (NOGP), southern giant petrels (SOGP) and wandering albatrosses (WAAL) banded at Possession Island, Crozet outside their native (or breeding) sites.

Table 5 – Selected models of annual breeding pairs estimated by TRIM as a function of environmental covariates for five species on Possession Island, Crozet archipelago

Species	Model	Estimate (±SE)	*P*
SOAL	Intercept	+5.668 (0.412)	<0.001
	N_{t-1}	+0.001 (0.001)	NS
	+ T	−0.218 (0.072)	<0.01
	+ T^2	+0.016 (0.005)	<0.01
	+ T^3	−0.0004 (0.0001)	<0.01
	+ SOI	+0.029 (0.018)	NS
	+ T Log(Tooth + 1)	+1.609 (0.990)	NS
	+ T^2 Log(Tooth + 1)	−0.192 (0.114)	NS
	+ T^3 Log(Tooth + 1)	+0.006 (0.003)	NS
LMSA	Intercept	+5.351 (0.096)	<0.001
	N_{t-1}	+0.0017 (0.0001)	<0.001
NOGP	Intercept	+4.973 (0.060)	<0.001
	N_{t-1}	+0.0023 (0.0001)	<0.001
	+ T Log(Tooth + 1)	+1.284 (0.453)	<0.01
	+ T^2 Log(Tooth + 1)	−0.149 (0.052)	<0.01
	+ T^3 Log(Tooth + 1)	+0.004 (0.001)	<0.01
SOGP			
	Intercept	+3.449 (0.170)	<0.001
	N_{t-1}	+0.012 (0.002)	<0.001
WAAL			
	Intercept	+5.851 (0.270)	<0.001
	N_{t-1}	+0.0011 (0.0004)	<0.01
	+ T	−0.076 (0.021)	<0.001
	+ T^2	+0.003 (0.001)	<0.001
	+ T^3	−3.85E−05 (1.13E−05)	<0.001

(1985–2000) (Crawford et al., 2003; Nel et al., 2002) (Table 2). Populations of SOGP were stable or declining during the 1980s, followed by an increase during the early and mid-1990s. For SOGP, the trend at Possession corresponds with those observed at other Sub-Antarctic areas, but differs from the declining trends observed in Antarctic populations, partly attributed to increasing human disturbance linked to the vicinity of research stations (González-Solís et al., 2000; Micol and Jouventin, 2001). The trends of LMSA remain largely unknown, with available estimates of the tendencies for only 11% of the world's population (Table 2). Population changes for this species were only evaluated at two other breeding sites where LMSA appeared stable: Marion and Macquarie Islands.

WAAL represents the species for which the population estimates and trends are the best documented amongst all five species (Table 2). Population trends for Possession WAAL were very similar to those from other populations breeding in the Indian Ocean, at Marion and Kerguelen islands (Weimerskirch et al., 1997; Nel et al., 2002). All have experienced a rapid decline during nearly 20 years, initiated in the 1960s, followed by recovery during the 1980s. The South Atlantic population (South Georgia) represents an exception since no recovery trend was noticed and the population is still declining (ACAP, 2005).

The breeding populations of NOGP on other Sub-Antarctic islands appeared to fluctuate greatly, with alternating stable, increasing or decreasing periods. Nonetheless, the population at the most important breeding site (South Georgia), holding

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about 30% of the world's population, tended to increase since the 1960s (Hunter, 1984; González-Solís et al., 2000).

Declines of SOAL population sizes have been reported for Gough Island (−60% between 1972 and 2000, rate of −3.2% per year; Cuthbert and Sommer, 2004) and for Marion Island (rate of −2.6% per year; Crawford et al., 2003). At Possession we found the most important decline for the species at an annual rate of 4.2%. If this decline is representative of the Crozet archipelago (SOAL is reported breeding on the four others islands see Table 2 with 1950–2050 breeding pairs in 1984, see Catard, 2001) we may assume that only 680–780 pairs remain breeding annually. Crozet archipelago held between 14% and 21% of the total estimated breeding population of 12 500–19 000 pairs (BirdLife International, 2007) in the 1980s. Furthermore, the breeding population of Crozet archipelago represents more than 50% of the regional population (Amsterdam Is., Kerguelen Is. and Prince Edward–Marion Is.) and thus is of major conservation concern. In this context of global decline, SOAL has been recently uplisted to "Endangered" IUCN Red List category on the basis of a 75% decline over three generations (90 years) (BirdLife International, 2007). Furthermore, trend information from three sites suggests that this species could be classified as "Critically Endangered" if these trends are found to be more general at all breeding sites (Table 2).

4.1.2. *Environmental factors*

Similarities in long-term population trends of this Procellariiform community from the South Indian Ocean suggest common underlying causes, with the emergence of groups of species related to the habitats exploited. Nonetheless, there are probably complex interactions with local factors during the breeding season.

If we considered the breeding success, we have established two patterns amongst the Possession community. The first group of increasing species (LMSA, SOGP) shows relatively low and fluctuating breeding success, whereas the other stable or declining species (SOAL, NOGP, WAAL) exhibited relatively high and/or increasing mean breeding success with little variability. Since breeding success may be considered as an indicator of environmental conditions during breeding (Furness and Greenwood, 1993), this suggest relatively poor environmental conditions during breeding for LMSA and SOGP compared to the other species. Inchausti et al. (2003) found that warm sea surface temperatures positively (negatively) affected breeding success in SOAL and WAAL (LMSA). These findings fit with our results (increase in breeding success of SOAL, relatively low breeding success of LMSA, high breeding success of WAAL) and with an increase in sea surface temperatures south of 35°S since the early 1980s (Fig. 2c). In the two species of giant petrels, males are mainly carrion-dependent, feeding extensively on seals (elephant seals *Mirounga leonina*, fur seals *Arctocephalus* sp.) (Hunter, 1983) and penguins carcasses (Ridoux, 1994), while females forage principally at sea (González-Solís et al., 2000). At South Georgia changes in population may have been related to increase in fur seal population (Hunter, 1984). At Possession, the fur seal population has sharply increased and the elephant seal population declined until the late 1980s and remained stable thereafter (Guinet et al., 1994, 1992; Weimerskirch unpubl. data). However the size of fur seal population at Crozet is very small compared to South Georgia and changes in population of giant petrels is probably not related to access to seals carcasses or pups such as at South Georgia. At Possession, the main biomass available on land are king penguins (*Aptenodytes patagonicus*) that are preyed on by giant petrels (Delord et al., 2004; Descamps et al., 2005). Furthermore, Ridoux (1994) suggested that king penguins may represent an important part of the diet of giant petrels. Their populations have increased in parallel to NOGP at Crozet (Delord et al., 2004). SOGP exhibits more extensive pelagic foraging (Stahl et al., 1996), which may suggest that factors other than habitats exploited during the breeding period could be involved in driving SOGP population trends. Detailed studies of the foraging ecology and at sea distribution of giant petrels at Possession are needed to understand the causes of variation in their populations.

From a population dynamics point of view, the observed patterns suggest that the most recent population declines of SOAL, NOGP and WAAL are probably caused by declines in local survival and/or recruitment. Demographic studies of this WAAL population indicate that decline observed during the 1980s was mainly caused by an increase in adult mortality and decrease in recruitment rate caused by low survival of juvenile and immature birds, which seemed to hinder the recovery process (Weimerskirch and Jouventin, 1987; Weimerskirch et al., 1997; Weimerskirch unpubl. data). The decrease in survival was partly attributed to mortality caused by longline fisheries (Weimerskirch et al., 1997; Gales, 1998; Gales et al., 1998; Tuck et al., 2001; Tuck et al., 2003; see for a review Nel and Taylor, 2003), although climatic factors may also affect albatross survival (Nevoux et al., 2007; Rolland et al., 2007). Interestingly, the two species with declining breeding population have an increasing breeding success, which may suggest that density dependence factors might also be involved. Estimating and modelling demographic parameters of biennial species such as SOAL and WAAL is a particularly challenging task, but such studies are currently underway. Unfortunately, the absence of detailed demographic data based on individual mark-recapture data for NOGP, SOGP, and LMSA will not permit to disentangle the demographic processes driving the population dynamics of these species at Possession.

Our analysis of recoveries constitutes direct evidence of mortality associated with fishery activities. Here, we presented the first documented recovery of SOAL from Crozet in a longline fishery: a SOAL captured in a Taiwanese vessel (see Fig. 4). This subtropical species is supposed to largely overlap with increasingly important subtropical tuna fisheries (see Fig. 2b). However, the low number of birds reported killed by longliners probably does not reflect the real extent of mortality of Crozet birds, mainly because seabirds bycatch management remains incomplete over different fisheries and there are no observers aboard these pelagic fisheries. In this context, the differences in recovery rates may be representative of differences in mortality rates only if it is assumed that reporting rates were similar and evenly distributed, which is probably not the case. Research has also to be developed to examine whether changes in food availability could have occurred in subtropical waters.

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We found that WAAL and NOGP presented the highest fishing-related recovery rates. This coincides well with the ongoing identified threat for WAAL (Weimerskirch et al., 1997; Tuck et al., 2001; Weimerskirch et al., 2006) and with the less documented threat for NOGP (Delord et al., 2005; SC-CAMLR, 2005). In recent years NOGP was the third most common species caught by licensed vessels in the French EEZ (Delord et al., 2005; SC-CAMLR, 2005). Surprisingly, the average bycatch rates reported in longline fisheries were higher for SOGP than for NOGP (Nel et al., 2002; Nel and Taylor, 2003). This may be interpreted in relation to their foraging ecology, the SOGP being more susceptible to interactions with commercial fisheries. The trend of higher recovery rates of immature birds in fisheries may reflect a higher risk of being caught accidentally on longline hooks because of the high dispersal tendency of juvenile birds (Weimerskirch et al., 2006) and their lack of experience. However, care must be taken in these conclusions given the sampling biases mentioned above and that the rates for band recoveries are apparent rates.

There has been a substantial increase and expansion of longline fishing effort south of 30°S in the Indian Ocean since the mid-1960s (Fig. 2b), suggesting that it constitutes a major ongoing threat when considering the spatio-temporal overlap between seabird foraging areas and fishing effort (Barré et al., 1976; Weimerskirch and Jouventin, 1987; Weimerskirch et al., 1997; Gales et al., 1998; Weimerskirch, 1998; Brothers et al., 1999; Nel et al., 2002; Robertson et al., 2004; Kiyota and Minami, 2004; Delord et al., 2005; SC-CAMLR, 2005; Weimerskirch et al., 2006; BirdLife International, 2006). The fact that we did not detect any direct relationship between population trends and fishing effort does not imply that interactions with fisheries are not a serious threat for these species. In Procellariforms, bycatch in longline fisheries can differentially affect various categories (age specific, or status specific) of individuals at different periods of the year (Murray et al., 1993; Weimerskirch et al., 1997; Gales et al., 1998; Tuck et al., 2001; Véran et al., 2007; Rolland et al., 2007), which may in turn differentially affect the impact on breeding population sizes, with varying lags. The demographic rates of different life stages may respond to environmental variation in contrasting ways, and count data of breeding pairs do not capture this complexity. Consequently the effects of climate and/or fisheries may be diluted in analyses of such data. The dramatic increase of fishing effort does not coincide with declining trends for all the five species studied, suggesting an intricate interaction of factors.

Finally for most species (SOAL, LMSA, NOGP or SOGP) there is a lack of information on the degree of overlap between their at sea distribution and those of fisheries (Pinaud and Weimerskirch, 2007; Weimerskirch unpubl. Data, BirdLife International, 2004), as is the information on mortality levels in fisheries. This study underlines the importance of long-term monitoring programs of breeding populations and stresses the importance of the implementation of international observer programs (i.e. CCAMLR) to collect data on interaction and mortality rates at sea – especially pelagic fisheries in subtropical waters – by means of regional fisheries management organisations. (Indian Ocean Tuna Commission – IOTC, Commission for the Conservation of Southern Bluefin Tuna – CCSBT...)

5. Uncited references

Barbraud and Weimerskirch (2003),Lewison et al. (2004) and Pinaud and Weimerskirch (2002).

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Appendix A

Multiplicative slopes standing for the changes in annual breeding population size and their 95% CIs of five species on Possession Island, Crozet obtained from the model selected in the TRIM analysis

Species	Period	Trend	Multiplicative slope ± SE	95% CI
SOAL	1979–1980	Stable	1.0 ± 0.0	–
	1980–1982	Decrease	0.709 ± 0.053	0.605–0.813
	1982–1995	Decrease	0.977 ± 0.015	0.948–1.006
	1995–1996	Increase	1.492 ± 0.148	1.202–1.782
	1998–2005	Decrease	0.882 ± 0.020	0.843–0.921
LMSA	1980–1999	Stable	1.0 ± 0.0	–
	1999–2005	Increase	1.068 ± 0.015	1.039–1.097
NOGP	1980–1993	Decrease	0.943 ± 0.014	0.916–0.971
	1993–1998	Increase	1.140 ± 0.041	1.059–1.221
	1998–2005	Decrease	0.971 ± 0.025	0.921–1.021
SOGP	1980–1999	Stable	1.0 ± 0.0	–
	1999–2004	Increase	1.096 ± 0.034	1.029–1.163
WAAL	1965–1968	Stable	1.0 ± 0.0	–
	1968–1969	Decrease	0.742 ± 0.072	0.601–0.883
	1969–1985	Decrease	0.970 ± 0.008	0.954–0.986
	1985–1986	Increase	1.288 ± 0.094	1.104–1.472
	1986–2004	Increase	1.016 ± 0.005	1.006–1.026
	2004–2005	Decrease	0.769 ± 0.062	0.647–0.890

SOAL, Sooty albatross; LMSA, light-mantled albatross; NOGP, Northern giant petrel; SOGP, Southern giant petrel; WAAL, Wandering albatross.

Appendix B

Annual breeding pairs of sooty albatrosses (SOAL), light-mantled albatrosses (LMSA), northern giant petrels (NOGP), southern giant petrels (SOGP) and wandering albatrosses (WAAL) counted in sectors monitored at Possession Island, Crozet archipelago, from 1979 (1965 for WAAL) to 2005

Breeding season	Pointe Basse	Jardin Japonais
1979	158	135
1980	156	173
1981	–	–
1982	55	–
1983	–	–
1984	65	–
1985	–	–
1986	71	88
1987	68	96
1988	45	–
1989	33	–
1990	45	–
1991	55	–
1992	63	53
1993	51	56
1994	64	64
1995	47	60
1996	59	146
1997	43	67
1998	53	82
1999	26	42
2000	44	62
2001	43	58
2002	41	39
2003	31	34
2004	30	44
2005	29	34

The summer of 2005/2006 is referred to as 2005, hereafter, except for WAAL with the longest biennial breeding cycle, for which is referred to as 2006.

Breeding season	Petite Manchotière – Chivaud	Chivaud – Baie du Marin	Baie du Marin – Pointe Bougainville	Pointe Bougainville – Cap du Gauss
1980	22	265	242	200
1986	33	223	215	
1987	48	249	276	175
1993	43	187	176	165
1994	52	270	162	130
1995	51	296	245	128
1996	46	248	194	169
1997	42	232	173	197
1998	60	350	154	197
1999	34	269	176	138
2000	64	354	231	193
2001	81	234	238	140
2002	130	245	243	160
2003	67	328	296	167
2004	59	384	293	284
2005	94	266	262	294

Breeding season	Chivaud	Galliéni	Jardin Japonais	Trou du Diable	Petit Caporal	La Hébé	Petite Manchotière	Sphinx	Baie du Marin	Crique de Noël	La Pérouse	Les Moines	Mare aux Eléphants	Pointe Basse
1980	–	–	–	50	34	4	31	20	1	25	58	70	105	160
1986	–	27	–	4	10	1	9	17		8	32	38	59	76
1987	1	9	–	13	10	3	10	36	1	9	48	27	69	83
1992	–	–	1	–	22	2	17	33		0	70	–	8	133
1993	–	–	0	–	23	–	16	15	0	0	54	18	18	97
1994	–	–	0	–	–	–	0	17	–	–	–	8	31	147
1995	–	–	0	–	–	–	5	38	–	–	47	–	26	134
1996	–	–	3	–	5	12	0	20	2	–	70	28	21	219
1997	–	–	3	–	0	17	0	34	2	–	47	–	21	235
1998	17	–	5	–	–	19	0	50	3	–	58	13	48	305
1999	–	–	3	–	0	18	1	36	–	–	55	18	45	164
2000	–	–	6	–	–	21	1	25	–	–	69	17	61	204
2001	–	0	7	7	–	25	–	35	–	–	53	18	25	221
2002	0	0	4	0	0	26	0	28	0	0	58	21	25	214
2003	0	0	12	0	0	19	9	29	0	10	50	33	80	221
2004	2	3	8	–	0	18	5	34	1	2	52	33	77	198
2005	7	3	16	0	0	24	12	28	1	4	41	30	58	175

Breeding season	Jardin Japonais	La Hébé	Petite Manchotière	Sphinx	La Pérouse	Pointe Basse
1980	–	0	19	0	25	40
1986	–	0	15	1	19	29
1987	–	0	6	0	29	38
1992	6	0	3	0	27	47
1993	6	–	0	0	28	71
1994	9	–	0	0	–	54
1995	10	–	0	0	0	63
1996	5	0	18	0	29	43
1997	5	0	15	7	18	53
1998	4	–	14	–	33	70
1999	6	0	10	5	17	33
2000	17	1	23	–	35	54
2001	11	2	24	–	34	35
2002	29	1	23	–	35	47
2003	28	20	9	–	32	54
2004	36	1	8	1	28	50
2005	47	1	12	0	40	61

Breeding season	Pointe Basse	Est Baie du Marin	Baie du Marin
1965	–	194	54
1966	–	–	–
1967	250	205	45
1968	250	191	59
1969	–	194	54
1970	–	–	49
1971	–	–	58
1973	–	–	53
1974	203	–	51
1975	161	89	49
1976	162	84	48

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Breeding season	Pointe Basse	Est Baie du Marin	Baie du Marin
1978	130	–	–
1980	137	79	51
1981	133	66	60
1982	132	71	45
1983	124	65	49
1984	135	77	50
1985	115	74	42
1986	144	107	55
1987	133	81	54
1988	138	99	56
1989	152	103	53
1990	152	96	43
1991	162	95	67
1992	161	109	50
1993	162	93	61
1994	167	112	70
1995	155	111	66
1996	170	124	83
1997	186	139	80
1998	182	109	82
1999	181	120	66
2000	174	131	76
2001	185	129	69
2002	185	131	78
2003	177	97	77
2004	166	119	80
2005	160	74	58

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