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Long term population changes of fur seals *Arctocephalus gazella* and *Arctocephalus tropicalis* on subantarctic (Crozet) and subtropical (St. Paul and Amsterdam) islands and their possible relationship to El Niño Southern Oscillation

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Abstract: The population trend over the last decade for subantarctic fur seals (*Arctocephalus tropicalis*) on Amsterdam and St. Paul islands and on Possession Island (Crozet Archipelago) and Antarctic fur seals (*A. gazella*) on Possession Island are analysed. At Amsterdam Island, based on pup counts, the subantarctic fur seal population appears to have stabilized after a period of rapid growth. At Possession Island subantarctic fur seal and Antarctic fur seal, with respective annual growth rates of 19.2 and 17.4%, are reaching the maximum growth rate for the genus *Arctocephalus.* Annual pup censuses at Possession Island since 1978 indicate important variations from year to year with pup production for *A. gazella* significantly lower the year after an El Niño Southern Oscillation (ENSO) event, but with no such relationship for *A. tropicalis.* Several other long term demographic studies of seabirds and marine mammals at different breeding locations in the Southern Ocean indicate that the breeding success of several of these predators appears to be widely affected in years which appear to be related to the ENSO events. To clarify this, it is necessary to analyse in more detail the demographic data obtained for the different subantarctic and Antarctic locations where long term monitoring programmes are conducted.

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Key words: fur seal, Crozet, Amsterdam, bio-monitoring, subantarctic, El Niño Southern Oscillation

Introduction

Both subantarctic (*Arctocephalus tropicalis*) and Antarctic fur seals (*A. gazella*) breed on Crozet Archipelago (Jouventin *et al.* 1982) whereas only subantarctic fur seals are found on St. Paul and Amsterdam Islands (Roux 1986, Fig. 1).

On Possession Island (Crozet Archipelago) the first breeding colony of subantarctic fur seals was found in 1978, while the first Antarctic fur seal pup was observed in 1981 (Jouventin *et al.*1982). On Amsterdam Island, the fur seal population was completely depleted at the end of the sealing period in the middle of the nineteenth century, with the last sealing activities recorded in 1875. By this time, and based on the fur seal counts in 1956, probably less than 100 adult fur seals were left (Paulian 1964). The population of subantarctic fur seals then recovered to a total of 30 500 adults in 1982 on Amsterdam Island, and 66 pups were counted on St. Paul during the 1984–85 breeding season (Roux 1987a). This paper aims to give the recent trends in population size of *A. tropicalis* and *A. gazella* in these localities, and to analyse these demographic trends in relation to the recovery of the populations and environmental factors.

Methods

On Possession Island, ground censuses of adult males, adult females, and pups were conducted every year from 1978. The original colonies, which are also the study colonies for both species, are situated in the same area at Pointe des Moînes, but the breeding biotopes of the two species are different. While Antarctic fur seals breed on a small pebble beach, the haul out area of subantarctic fur seals is composed of large blocks making their census difficult as many pups are hidden underneath them. During each breeding season two or three successive counts were made. The earliest census was conducted around 20 December (±4 days) when the Antarctic fur seal pups are born. This period corresponds to the beginning of the parturition of subantarctic fur seals. The second census was conducted around 15 January (±4 days), when pups of subantarctic fur seals are born and harems of Antarctic fur seals are dispersing. The last census was conducted by 30 January (±4 days). On Possession Island the ideal census dates, in terms of maximum female and pup numbers for *A. gazella* and *A. tropicalis,* would have been 20 December and 15 January respectively, but practically population trends were determined from later counts conducted around 15 January (±4 days) for Antarctic fur seals and 30 (±4 days) January for subantarctic fur seals. Counts were then easier to conduct and less disruptive as the males were less territorial with the dispersion of harems. Furthermore, pups were grouped in crêches and had not yet started to swim in the open sea. The less aggressive behaviour of males allowed investigation underneath large blocks for hidden pups and a more systematic count of subantarctic pups after the ideal census date, despite probable pup mortality during the previous weeks.

New small breeding colonies of subantarctic fur seals were discovered in 1989–1990 on beaches not regularly checked. The last complete census, where all the beaches were checked, was conducted during the 1990–1991 breeding season. For that reason the growth rate of *A. tropicalis* has been calculated between 1977–1978 and 1990–1991 while inter-annual fluctuations of pup production were calculated for the study colony from the 1977–1978 to the 1992–1993 breeding season.

On Possession Island, where both species breed sympatrically and where harems including females from both species have been reported (Jouventin *et al.* 1982), identification of the species status of males and pups was conducted using the criteria proposed by Condy (1978) and completed by Goldsworthy & Shaughnessy (1989). As most of the observers had difficulties in determining the species status of the females, analysis of interbreeding rate was not performed.

On Amsterdam and St. Paul islands, where only *A. tropicalis* breed, one count was conducted between 1–14 February 1993. Fur seals were counted on the 49 segments used by Roux (1986) (Fig. 2). For the analysis, five classes of sites were considered according to their accessibility for counts : 1) counts within the colony (n=11); 2) counts from the cliff on boulder beach (n=3); 3) counts using binoculars from high cliff on boulder beach (n=22); 4) counts from the cliff on coast line composed of large blocks (n=3); 5) non-census areas during the 1993 count (n=10).

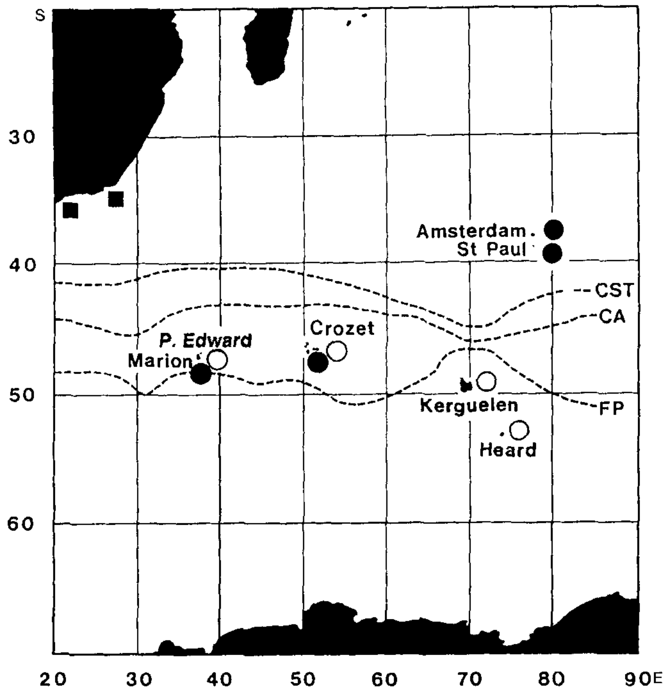

Fig. 1. South west of the Indian Ocean, with the main frontal zones. (CST: subtropical front; CA: subantarctic front, FP: Polar Front) and the breeding sites of *A. tropicalis* (●), *A. gazella* (○) and *A. pusillus* (■).

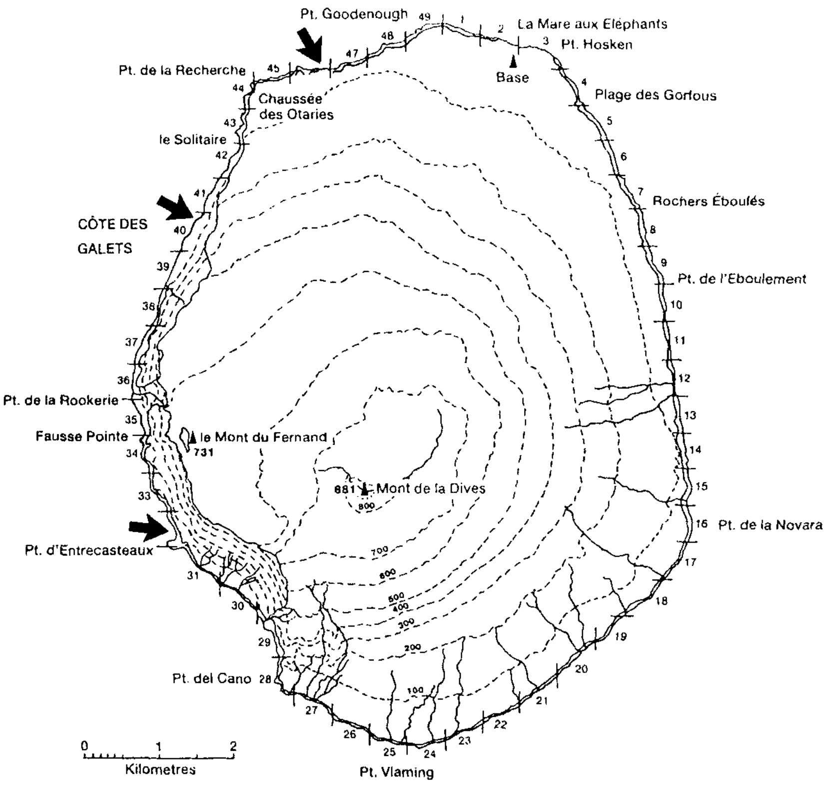

Fig. 2. The 49 census segments of Amsterdam Island defined by Roux (1986). The location of the three original colonies is indicated by the arrows.

Results

Population growth

Based on pups counted (Table I) the population size on Amsterdam Island in 1993 appeared to be almost identical to that of 1982: over these 11 years, the annual rate of increase was zero. However, if we consider only the 14 portions of the coast composed of the easiest beaches to census (classes 1 and 2) to minimize the risk of census errors, the annual rate of increase obtained was 0.4% between 1982 and 1993.

On St. Paul Island, 66 pups were counted in February 1985 (Roux 1986) and a total of 365 were present in February 1993. Thus, the exponential growth rate was 19% over the last nine years on St. Paul Island. Such a growth rate may result from intrinsic recruitment within St. Paul's colony. However, the tagging programme conducted by Roux (1986) showed that movement of juvenile fur seals was taking place between St. Paul and Amsterdam islands. If we consider fur seals breeding at both islands as a single population, the overall population growth rate calculated is estimated to be 0.5% when all the beaches censused are taken into account and 0.9% when only the easiest beaches to census (class 1 and 2) are taken into account (St. Paul Island shore with breeding fur seals belongs to classes 1 and 2).

On Possession Island, despite inter-annual differences in pup production, the populations of both species are growing rapidly (Table II). Based on total pups count, the rate of increase of *A. tropicalis* was 19.2% from 1978–1991 and 17.4% for *A. gazella* from 1983–1992. The new breeding sites of *A. tropicalis* found since 1988 on Possession Island are indicated in Fig. 3.

Table I. Pup production of *A. tropicalis* on the 49 coastal segments defined by Roux (1987b). The accessibility class are defined in the text. The overall growth rate is calculated for segments censused both in 1982 and 1993.

Census areas	accessibility class	Census 1982*	Census 1993	annual exponential rate of increase
1	1	296	554	5%
2	1	310	325	0%
3	1	325	287	-1%
4	2	247	196	-2%
5	2	238	173	-3%
6	3	175	212	2%
7	3	175	183	0%
8	3	44	48	1%
9	3	15	10	-3%
10	3	10	118	21%
11	4	10	0	-
12	4	10	0	-
13	4	10	0	-
14	3	10	0	-
15	3	29	0	-
16	3	29	0	-
17	3	15	0	-
18	3	0	0	-
19	3	0	0	-
20	3	0	0	-
21	3	0	0	-
22	3	0	0	-
23	3	0	0	-
24	3	0	0	-
25	3	0	3	-
26	3	15	4	-11%
27	3	141	3	-32%
28	3	126	8	-23%
29	?	233	N	-
30	3	0	29	-
31	1	165	600	11%
32	1	1179	615	-5%
33	?	941	N	-
34	?	0	N	-
35	1	485	570	1%
36	1	126	320	8%
37	?	126	N	-
38	?	0	N	-
39	?	315	N	-
40	?	1150	N	-
41	?	213	N	-
42	?	175	N	-
43	?	155	N	-
44	1	155	86	-5%
45	1	102	716	16%
46	1	805	733	-1%
47	1	543	257	-6%
48	2	272	132	-6%
49	2	272	232	-1%
Total pup production			9638	
Pup production on area census in 1982		6334	6414	0%

* Roux (1987b)= Non cenus areas in 1993. N = not censused.

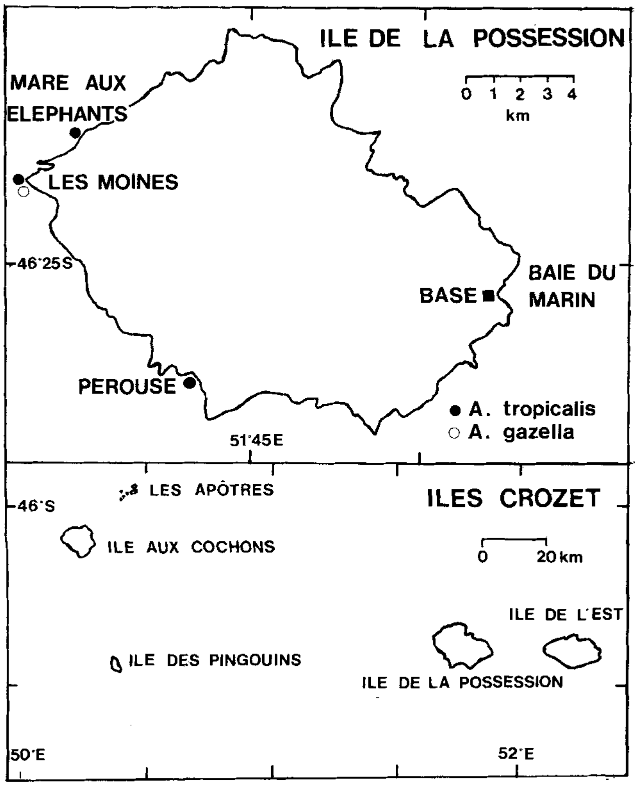

Fig. 3. Breeding sites colonized by *A. tropicalis* on Possession Island since the discovery of the original colony in 1978 at Les Moînes. New breeding sites have been colonized at La Mare aux Eléphants and la Pérouse during the 1988–1989 breeding season.

Inter-annual variability

On the study colonies of Possession Island the general trends of *A. tropicalis* and *A. gazella* pup production from 1983–1992 were positively correlated (r=0.86, n=10, P<0.001). However, the interannual fluctuations of population growth rate were not positively correlated (r_s=0.41, n=10, P>0.05), indicating that both species may respond differently to environmental conditions.

Counts from 1977–1978 to 1992–1993 breeding season, of pups and females of both species within the study colonies, showed strong inter-annual differences (Fig. 4). Females of both species counted ashore during the census, show three periods of decrease in numbers: from 1983–1985, from 1988 –1989 and from 1991–1993. *A. tropicalis* female numbers showed the strongest variations and pup production of *A. tropicalis* tended to be more depressed than *A. gazella* (Fig. 4).

Pup production was correlated with the number of females counted ashore both for *A. gazella* (r=0.89, n=11, P<0.001) and *A. tropicalis* (r=0.75, n=16, P<0.001). However, no correlation was found for *A. gazella* between the annual growth rate calculated from adult females counts and from pups counts (r=0.40, n=9, P>0.05), although a positive correlation was found for *A. tropicalis* (r=0.45, n=15, P<0.05).

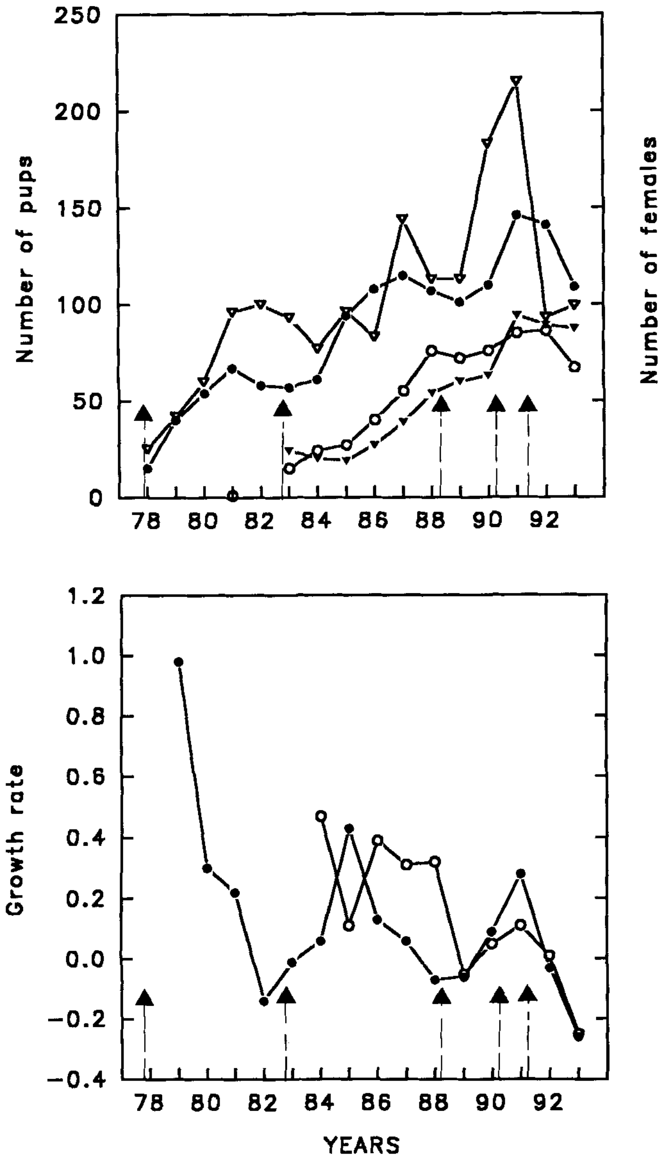

Fig. 4. Counts from 1977–1993 breeding seasons. **a.** number counted ashore at the study colonies since their discovery in 1978. **b.** Year to year variations of of the annual growth rate calculated from pup counts. Key: *A. tropicalis* pups (o), females (∇); *A. gazella* pups (•), and females (▾). Arrows indicate ENSO events.

Top predators and oceanographic change

From 1983 to 1992, the exponential annual growth rate of *A. gazella* (Table II) calculated from pup production, on Crozet Archipelago, indicated that pup production was depressed for the 1984–1985, 1988–1989, and 1991–1992 and 1992–1993 seasons. These periods followed the last three El Niño Southern Oscillation (ENSO) events. Considering the bimodal shape of the 1990–1991 ENSO (Fig. 5), the year following the ENSO events showed a significantly lower annual growth rate for *A. gazella* on Possession Island (U=2, N1=4, N2=6; P<0.02). *A. tropicalis*, despite depressed pup production following the 1987 and the 1990–1991 ENSO, did not present a different population growth rate compared to the non-ENSO years (P>0.05).

Discussion

On Amsterdam Island, the annual growth rate of *A. tropicalis* was 7.8% between 1956–1970 and then 15.5% between 1970 and 1982 (Roux 1986). The pup census conducted in early

Table II. Total pup production (alive and dead) of *A. tropicalis* and *A. gazella*.

Breeding season	*A. tropicalis*	*A. gazella*
77–78	15	-
78–79	40	-
79–80	54	-
80–81	67	1
81–82	58	-
82–83	57	15
83–84	61	24
84–85	94	27
85–86	108	40
86–87	129	55
87–88	117	76
88–89	-	72
89–90	141*	78
90–91	190*	87
91–92	-	86
92–93	-	67

* discovery of new breeding colony.

February 1993 suggests that, compared to January 1982, the population was almost stable over that period of time. Annual censuses of pups of *Arctocephalus* spp. on Crozet Archipelago (present study), but also of *A. gazella* at South Georgia (Lunn & Boyd 1993) have shown a high inter-annual variation in pup production. Such a large annual variation in pup production suggests that interpreting population trends from counts taken at 5- or 10-year interval may give misleading impressions on the actual status of the fur seals populations. For that reason the results obtained for Amsterdam and St. Paul islands should be taken with caution and the apparent stabilization observed on Amsterdam Island needs to be confirmed by future censuses. It may be argued that the late census on Amsterdam Island underestimated pup production. However, the bias is likely to be limited as data presented by Roux (1986) indicated that the number of pups counted ashore on the study beach remained at a comparable level from the end of December until the middle of March. In fact, 2–5% more pups were censused from mid-January until mid-February compared to a census counducted from the end of December until mid-January and later ones conducted from 15 February until the beginning of March. The differences between these three periods were not significant. We are unable to determine if the Amsterdam Island fur seal population has reached pre-exploitation level as it is not possible from historical documents to estimate the population size before exploitation began (Roux 1986). However, on St. Paul Island the recolonization only started in 1970 (Roux 1986) and the present population must be below the possible pre-exploitation level. The number of breeding females is now increasing rapidly but we are unable to determine the extent of emigration from Amsterdam Island.

At Crozet Archipelago, both *A. gazella* and *A. tropicalis* are reaching the maximum growth rate for the genus *Arctocephalus* (Roux 1986, Boyd *et al.* 1990). The 17.4 % exponential growth rate of *A. gazella* at Possession Island is similar to the rapid rate of increase observed at South Georgia between 1958–1972 (Boyd *et al.* 1990) and the 19.2 % growth rate of *A. tropicalis* is identical to the rate of increase observed at Gough and Amsterdam islands during the phase of exponential growth of these populations (Bester 1980, Hes & Roux 1983).

However, as the number of females ashore is highly variable from day to day (Boyd 1993), the growth rate calculated from female counts may be strongly affected by these daily variations. The presence ashore of the females can possibly be affected by the moon phase as shown by Trillmich & Mohren (1981) for Galapagos fur seals (*A. galapagoensis*) and *A. tropicalis* on Amsterdam Island by Roux (1986). Furthermore, if feeding conditions are poor reproductive females will tend to spend less time ashore and to have longer foraging trips as shown for California sea lions (*Zalophus californianus*) by Costa *et al.* (1991) and *A. galapagoensis* by Trillmich & Dellinger (1991), and as suggested by the data presented by Boyd (1993, fig. 2) for *A. gazella*. Thus, as pups are resident ashore for the first few weeks of their life and where an accurate pup census is possible, population trends should rather be calculated from these counts.

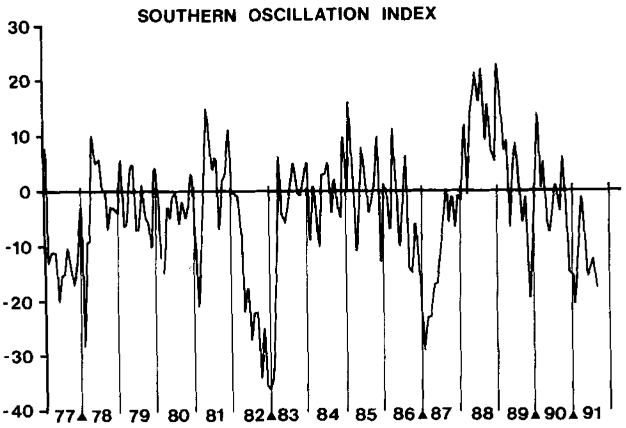

Fig. 5. The Southern Oscillation index from 1977–1991. Arrows indicate ENSO events.

The present study indicates that, despite the fact that the long term trend of female and pup numbers are correlated, year to year fluctuations of females and pups number tend to be independent. In fact, when the number of females was strongly depressed, as during the 1991-1992 breeding cycle, pup production was affected the following year. Lunn & Boyd (1993) have shown that the pup production of *A. gazella* at South Georgia is affected the year after the food shortage, while within the year of food shortage female will tend to spend more time at sea, and thus will be less present ashore. These authors suggest the year lag may result from a reduced implantation rate during the year of food shortage.

On Possession Island, no clear relationship could be established between fluctuation in pup numbers of *A. tropicalis* and the ENSO event, while the pup production of *A. gazella* was affected the year after such event. The differences in response between the two species is not understood but this may be related to a possible difference in foraging behaviour and/or to the differences in breeding cycle: *A. gazella* having a four-month lactation period while lactation lasts 11 months for *A. tropicalis*.

Chastel *et al.* (1993) have shown a similar relationship between the adult survival of snow petrels (*Pagodroma nivea*) and ENSO events. Identical demographic responses were also observed for blue petrels and Belsher prions at Kerguelen Island (Chastel *et al.* unpublished). Both of these oceanic petrels showed a lower body condition index, a lower reproduction rate and a lower breeding success during the 1987/1988 and 1991/1992 seasons (Chastel *et al.* unpublished). Croxall (1992), Boyd & Roberts (1992) and Boyd (1993) have also indicated that seabird, seal and whale populations off South Georgia may be affected by climatic and oceanographic factors having potentially important links with the Southern Oscillation. Testa *et al.* (1991) have shown that demographic fluctuations of Antarctic phocids were possibly related to the ENSO effects. Despite the fact that no proven relationships have yet been established between the ENSO events, the oceanographic conditions of the

Southern Ocean and demographic trends of top predators, these observations suggest that environmental (trophic) conditions in the Southern Indian Ocean, and possibly on a Southern Ocean scale, are probably affected on a very large scale in certain years in relation to the ENSO events. To determine if such a global effect can be detected on the Southern Ocean scale there is a urgent need to examine the long term data on demographic parameters obtained for seabirds and marine mammals for the different breeding locations where long term monitoring programmes are conducted. This examination should be made at the CCAMLR level but should also utilize demographic data for Antarctic and subantarctic species at present included in the CCAMLR monitoring programmes.

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