Risks of decline and extinction of the endangered Amsterdam albatross and the projected impact of long-line fisheries

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Abstract

The Amsterdam albatross (*Diomedea amsterdamensis*) is one of the rarest bird species of world avifauna, consisting of a single population in the upland plateau of Amsterdam Island (SE Indian Ocean). All breeding birds of the population are today banded and a monitoring program involving mark-recapture procedures has been carried out continuously over the past 16 years. We present the first estimate of risk of decline for the Amsterdam albatross using a stochastic matrix population model, and evaluate the extent to which the measurement errors in demographic estimates may affect the baseline conservation assessment. We also estimate the potential effect that resumption of long-line fisheries in the vicinity of Amsterdam Island (one the alleged causes for its low numbers in the recent past) may have on the persistence of this population. Our results indicate that, in the absence of any impact of long-line fisheries, the Amsterdam albatross is unlikely to experience a decline larger than 20% of the current population abundance over the next 50 years. Our results point out the difficulty to assess with certainty the extinction risk of small populations despite the availability of long term data on their demography. They suggest that a very cautious approach should be taken for the preservation of small populations of long-lived species that cannot sustain any level of incidental by-catch. Any new long-line fishery resuming in the foraging range of the Amsterdam albatross, but especially close to Amsterdam Island, may rapidly put this species at risk of extinction. © 2001 Elsevier Science Ltd. All rights reserved.

Keywords: Amsterdam albatross; Procellariiformes; Seabirds; Long-line fisheries; Conservation; Extinction; Demographic models

1. Introduction

The Amsterdam albatross (*Diomedea amsterdamensis*) is one of the rarest bird species in the world. This recently described species (Roux et al., 1983) consists of a single population nesting in 400 ha of the upland plateau of Amsterdam Island (Jouventin et al., 1989) in the SE Indian Ocean. Amsterdam Island (37°50′S, 77°31′E) is one of the most isolated islands of the world, being located between Australia, Africa and Antarctica at > 3000 km away from any continent. The species' life history is typical of other Procellariiformes (Weimerskirch et al., 1987): it is large bodied (adults weigh between 4.8 and 8 kg) very long-lived (mean life expectancy 30–40 years), has a low fecundity (a single egg is laid at most every two years and a chick fledges after about 235 days) and very high adult annual survival

(Jouventin et al., 1989; Weimerskirch et al., 1997). Despite its isolation and small numbers, the demographic parameters of the Amsterdam albatross population are known for a longer time span and with more accuracy than for most endangered bird species (Perrins et al., 1991). Although this population has steadily grown since the mid 1980s (Fig. 1), its low numbers and low reproductive potential should be a cause of concern. The species is listed as Endangered by the IUCN (IUCN, 1996) and by the Bonn Convention (Convention on the Conservation of Migratory species of Wild Animals) under Appendix 1 and it is also classified as Critically Endangered by Birdlife International (Collar et al., 1994). Despite the obvious concern about the conservation status of the Amsterdam albatross, these classifications of threat have not been based on a quantitative estimation of its actual risk of extinction.

While the reasons for the present low numbers of the Amsterdam albatross are not well known, the extensive degradation of the island (Jouventin, 1995) and the impact of the long-lining fisheries operating in the

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southern Indian Ocean during the 1960s and 1970s (Weimerskirch et al., 1997) have been proposed as possible explanations for its rarity. Because the situation of the environment on the breeding grounds of the species has been improved as a result of an extensive conservation plan, the only potential threat for the population is at sea. Long-line fisheries are known to impact seabird populations and especially albatross populations in the southern ocean (Brothers, 1991; Murray et al., 1993; Klaer and Polacheck, 1997; Gales et al., 1998; Weimerskirch and Jouventin, 1987; Weimerskirch et al., 2001; they represent a major concern for the future of several species including the Amsterdam albatross (Gales, 1993; Alexander et al., 1997). Virtually all available information on by-catch of southern seabirds by long-line fisheries is either from Antarctic or sub-Antarctic waters (recently started fishery for Patagonian toothfish *Dissostichus eleginoides*: Ashford et al., 1995; Cherel et al., 1996; Weimerskirch et al., 2001), or from Australian and New Zealand waters (fishery for southern blue fin tuna *Tunnus accoyii*: Brothers, 1991; Murray et al., 1993; Gales et al., 1998).

Stochastic demographic models have become an important and increasingly used tool for assessing the degree of threat of natural populations (reviews in Boyce, 1992; Burgman et al., 1993; Caughley, 1994; Beissinger and Westphal, 1998). Under the generic name of Population Viability Analysis (PVA), these models offer the possibility of systematically exploring the effect of factors that place a population or a species at risk of extinction. Although population viability analyses can rarely identify the actual cause for population declines, they allow one to explore the relative importance that different factors have on the extinction risk including those having a systematic impact on a species' demography. Despite their increasing use for making management and conservation assessments (e.g. Beissinger and Westphal, 1998), PVA models should be viewed with caution since they generally require a large amount of age-specific demographic data that is rarely available for endangered species.

This paper has three main goals. First, we quantify the risk of extinction and of decline in numbers of the Amsterdam albatross using a stochastic matrix population model. Second, we estimate the potential effect that the resumption of long-line fisheries near Amsterdam Island may have on the persistence of this population. Finally, we evaluate the influence that the measurement errors in the parameter estimates may have on the assessment of the conservation status of the Amsterdam albatross.
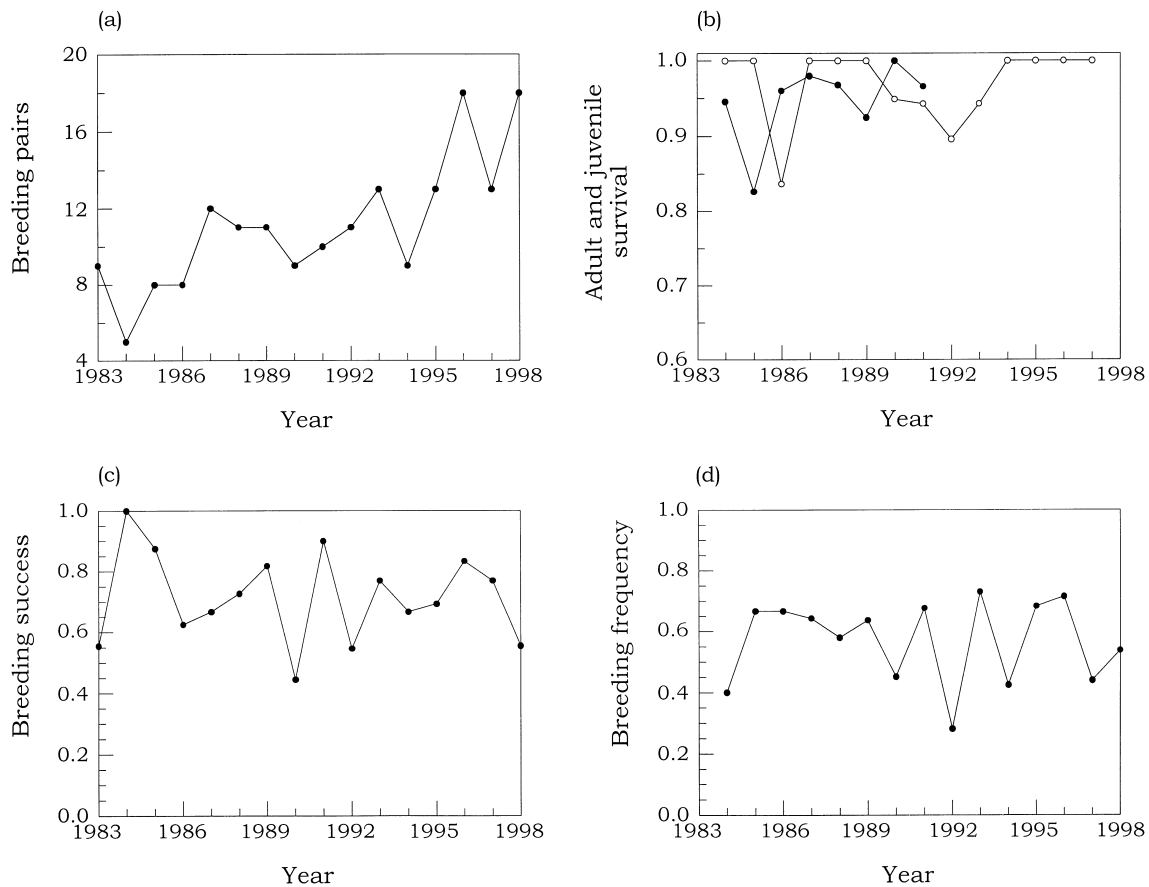

Fig. 1. Population abundance and main demographic parameters of Amsterdam albatross between 1983–1998. Data taken from Weimerskirch et al. (1997) and unpublished data. (a) Number of breeding pairs in Amsterdam Island. (b) Annual adult survival (open circles) and juvenile survival (filled circles). (c) Breeding success (proportion of eggs resulting in fledging). (d) Breeding frequency (proportion of adults breeding each year).

2. A population model for the Amsterdam albatross

A post-breeding approach was used for building of the age-transition model. The life cycle of the Amsterdam albatross population was subdivided into 24 age classes: chicks, juveniles (2- to 5-year olds), immatures (6-year olds), and adults (7- to 24+ year olds). For the older birds, age refers to the minimum age of marked individuals since they were breeding when the mark-recapture of the species was initiated in 1983. Juveniles disperse after fledging and remain scattered over large areas in the oceans, a common behaviour in southern oceanic birds (e.g. Warham, 1990). We differentiated juveniles from immatures the former being birds aged 1–5 years that remain at the sea and thus cannot be directly censused, while the latter return to the breeding grounds from the age of 5 years when they are re-captured for the first time after fledging. The age at first reproduction (7 years) was taken to be the minimum age of breeding individuals ever observed in Amsterdam Island.

An extensive census and mark-recapture program of the population over the last 16 years provided yearly estimates for breeding success, breeding frequency and survival (Fig. 1; Table 1) using the field methodology explained in Weimerskirch et al. (1997). The annual estimates of age-specific survival rates were assumed to be similar within juvenile and adult age classes, with the exception of the oldest individuals for which the mean survival was slightly lower than for other adult ages (Weimerskirch, 1992). Fecundity was calculated as the product of breeding frequency (proportion of breeding adults), breeding success (proportion of eggs resulting in fledglings), clutch size (equal to one) and the sex ratio (a 1:1 sex ratio was assumed) for each year. The age-specific fecundity rates were calculated as the product of the proportion of breeding adults of each age in Amsterdam Island and the estimate of adult fecundity calculated as explained above. We slightly decreased the mean fecundity of the oldest age class since fecundity declines at old ages for the wandering albatross (*Diomedea exulans*), a species whose demography and breeding biology is similar to those of the Amsterdam albatross (Weimerskirch, 1992; Weimerskirch et al., 1997). Since banding of adults started only in 1983, and the mean age at first breeding is 10 years, adults banded in 1983 were at least 24 years old in 1997. This is why we truncated the model at 24 age classes. We included a non-zero chance of remaining alive and reproducing after

Table 1
Age-specific annual survival and fecundity rates and 1997 initial age structure for Amsterdam albatross[a]

Age classes (years)	Initial age structure	Fecundity				Survival			
		No fishery		Predicted effect of fishery		No fishery		Predicted effect of fishery	
		Average	Variance	Low	Medium	Average	Variance	Low	Medium
Chicks 1	10					0.960	0.004		
Juvenile 2	14					0.960	0.004		
Juvenile 3	8					0.960	0.004		
Juvenile 4	5					0.960	0.004		
Juvenile 5	8					0.960	0.004		
Immature 6	5					0.974	0.003	0.920	0.785
Adult 7	7	0.038	0.00006	0.039	0.034	0.974	0.003	0.920	0.785
Adult 8	3	0.076	0.00025	0.078	0.068	0.974	0.003	0.920	0.785
Adult 9	6	0.114	0.001	0.118	0.103	0.974	0.003	0.920	0.785
Adult 10	4	0.167	0.001	0.173	0.144	0.974	0.003	0.920	0.785
Adult 11	5	0.187	0.002	0.194	0.162	0.974	0.003	0.920	0.785
Adult 12	4	0.187	0.002	0.194	0.162	0.974	0.003	0.920	0.785
Adult 13	5	0.187	0.002	0.194	0.162	0.974	0.003	0.920	0.785
Adult 14	2	0.187	0.002	0.194	0.162	0.974	0.003	0.920	0.785
Adult 15	2	0.187	0.002	0.194	0.162	0.974	0.003	0.920	0.785
Adult 16	1	0.187	0.002	0.194	0.162	0.974	0.003	0.920	0.785
Adult 17	4	0.187	0.002	0.194	0.162	0.974	0.003	0.920	0.785
Adult 18	2	0.187	0.002	0.194	0.162	0.974	0.003	0.920	0.785
Adult 19	1	0.187	0.002	0.194	0.162	0.974	0.003	0.920	0.785
Adult 20	1	0.187	0.002	0.194	0.162	0.974	0.003	0.920	0.785
Adult 21	2	0.187	0.002	0.194	0.162	0.974	0.003	0.920	0.785
Adult 22	3	0.187	0.002	0.194	0.162	0.974	0.003	0.920	0.785
Adult 23	2	0.187	0.002	0.194	0.162	0.974	0.003	0.920	0.785
Adult 24 and 24+	25	0.177	0.001	0.183	0.160	0.935	0.003	0.907	0.773

[a] The mean and variances were obtained from the yearly estimates of each demographic rate. The 'No Fishery' columns are the baseline estimates assuming no effect of fisheries and no measurement errors. The values in the columns 'Low' and 'Medium' are the average survival and fecundity values obtained assuming that long-line fisheries were to resume with the smallest and average extension of fisheries historically observed in the Amsterdam Sector of the Indian Ocean.

24 years of age to reflect the observed longevity of Procellariiformes. The latter is, however, not equivalent to assuming immortality since the chances of staying in the last age class diminish each year at a rate of $(1-S_{\text{Old adults}})$.

The initial age structure in 1997 (Table 1) was calculated from the number of breeding birds of different age classes captured in 1996 and 1997. This is because of the biennial breeding system of the species; birds raising a chick in 1996 stay at sea and thus cannot be recaptured in 1997. We assumed that all birds raising a chick in 1996 were still alive in 1997, a reasonable assumption given the high mean adult survival rate (0.969) of the species. For the birds aged 1–5 years that were at sea, the initial age distribution was calculated using the number of chicks fledged between 1991 and 1995. The mean juvenile survival for the ages 1–5 years estimated for earlier cohorts was used to estimate the values of juvenile survival between 1991 and 1997.

The demographic model was analysed by Monte Carlo simulation with the program RAMAS/Stage (Ferson, 1990) to estimate population abundance and chance of decline or extinction. Demographic stochasticity was incorporated by sampling the number of surviving individuals for each age class and the number of chicks produced by each age class from binomial distributions; the latter partly accounts for the effects that the potential variation of sex ratio could have on fecundity. Environmental stochasticity was included by using lognormal distributions to model the temporal variation of survival and fecundity rates, with age-specific means and variances of these survival and fecundity rates calculated from their yearly estimates (Table 1). Because the mean survival rates were generally higher than 0.9, we use a lognormal distribution for the mortality rates, and obtained the survival values by subtracting the mortalities from one. We implemented a truncation to assure that all survival values were < 1. The long chick-rearing period of the Amsterdam albatross makes it likely that breeding success and adult survival are correlated. We included this correlation in the routines used to generate the yearly values of fecundity and adult survival (see details in Burgman et al., 1993). We chose a time horizon for the projections of 50 time steps (years) and used 2500 replications for each set of parameter values.

2.1. Projected effect of the long-line fisheries on the degree of threat

Very little is known about seabird mortality in relation to pelagic fisheries, and to our knowledge there is no record of capture of an Amsterdam albatross by long-line ships. The Southern blue-fin tuna fishery overlapped extensively with the foraging range of Amsterdam albatrosses during the 1970s until the early 1980s (Fig. 2) when this fishery came to an end (Klaer and Polacheck, 1997; Tuck and Polacheck, 1997). Since the species was described in 1983, there is no data of survival and fecundity rates of the Amsterdam albatross during the period when long-line fisheries were active in the Amsterdam sector. We used the data documenting the effect of pelagic fisheries on the wandering albatross around Crozet Island (46°30′S, 52°E) to assess the effect that the resumption of fisheries can have on the Amsterdam albatross. The impact of long-line fishery on the wandering albatross was estimated by calculating separate linear regressions of its adult survival rate and the breeding success on the extent of the fishery as measured by the proportion of $5°\times5°$ squares of ocean with long-line ships (Weimerskirch et al., 1997; Weimerskirch and Jouventin, 1987).

Three degrees of impact (low, medium and high) of long-line fisheries were estimated using the minimum, mean and maximum extent of fisheries in the Amsterdam Sector (Polacheck and Tuck, 1995; Weimerskirch et al., 1997). These three sets of values of survival and fecundity (Table 1) were used in separate simulations to assess the impact that the resumption of fisheries may have on the viability of the Amsterdam albatross. For consistency, we used the values predicted by the regression equations, even though the mean fecundity values predicted using the lowest effect of long-line fisheries were slightly higher than the baseline values.

2.2. Effect of the measurement errors on the degree of threat

Measurement uncertainty reflects our inability to estimate accurately the values of the survival and fecundity rates at each time step. Since in reality each of the annual estimates of survival rates is inaccurately known, it follows that the mean obtained by averaging inaccurate values must also have an associated measurement uncertainty (Fig. 3). The temporal variation of survival rates reflects the vagaries of the environment, which is reflected in the spread of values around the mean survival (i.e. the variance of the distribution). In contrast, the measurement error is depicted as uncertainty in the location (rather than the spread) of the mean survival (Fig. 3).

The standard deviations of the adult survival (Table 2), calculated by SURGE (Lebreton et al., 1992) for the annual estimates that were different from one, were used as the yearly estimates of measurement error of this demographic parameter. The mean of these standard deviations (0.071), which represents 7.3% of the mean survival value, was taken as the magnitude of measurement error used to adjust the mean adult survival rates without exceeding the maximum allowable survival of one. The heterogeneity among the yearly values of the standard deviations of adult survival
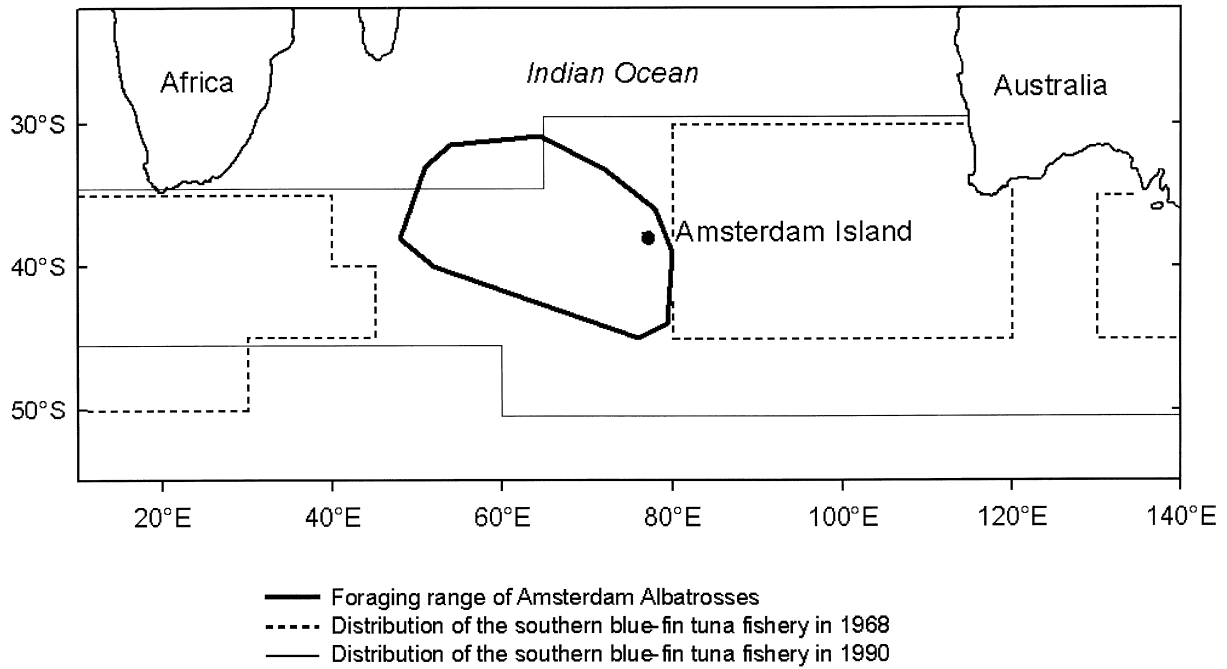
Fig. 2. Geographic extent of the blue-fin tuna fishery (data from Ardill, 1995) and the foraging area of the Amsterdam albatross, represented as a convex polygon obtained using all the satellite fixes during the breeding season (H. Weimerskirch, unpublished data).
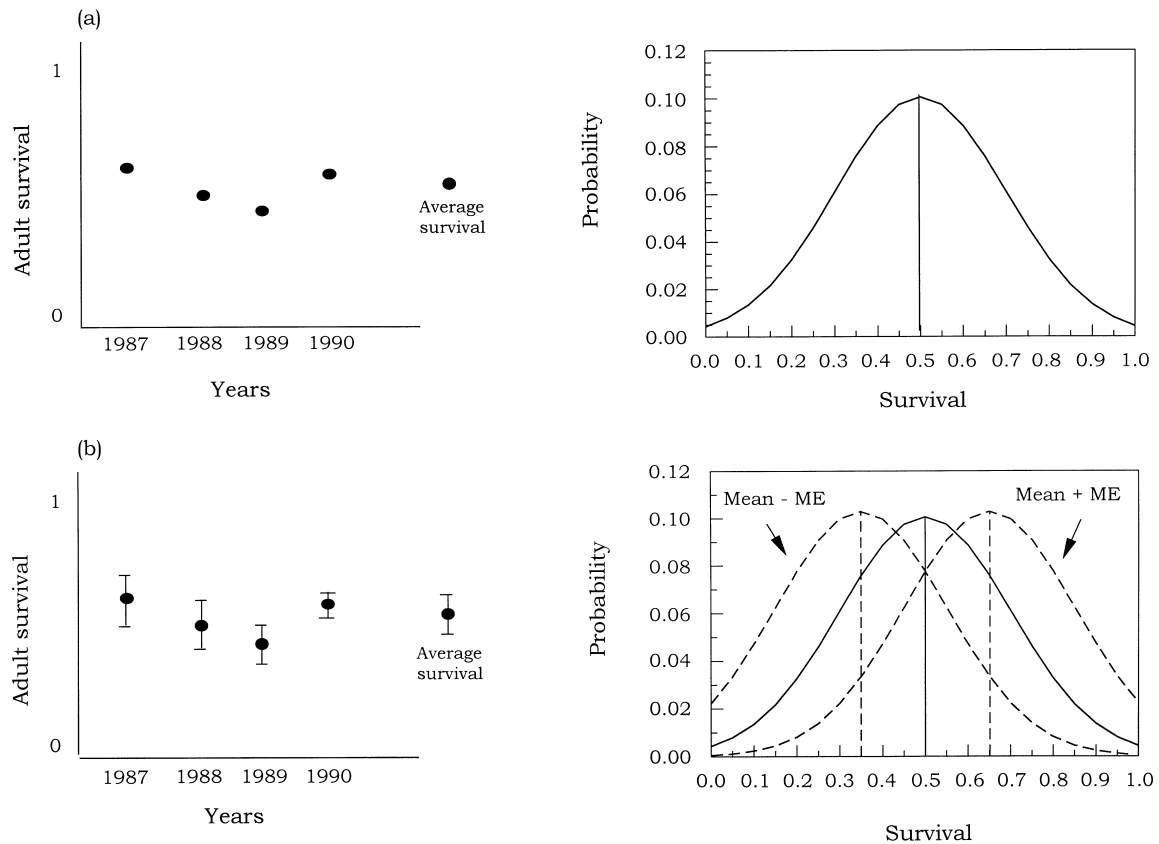

Fig. 3. Qualitative difference between environmental variation and measurement error. (a) Top panels: environmental stochasticity only. Left: yearly survival values measured without error. Right: probability distribution of adult survival. (b) Bottom panels: environmental stochasticity and measurement error (ME). Left: the same yearly survival values as above with ME. Right: probability distributions of adult survival based on above mean ± ME.

(Table 2) reflects the variability in the capturability and sampling effort between years. Three separate simulations were run to assess the influence of measurement error: 'no error', a decrease by 0.071, and an increase by 0.025 in the mean adult survival (the latter figure was the highest allowable value smaller than one). An additional simulation (decrease of 0.025 in the mean adult survival) was included to allow a direct comparison with the 'best case' scenario.

3. Results

The baseline model rendered a range of results for the abundance of the Amsterdam albatross, ranging from a decline of 1.2% per year to an increase of 5.2% per year, with a mean annual growth rate of 2.5% (Fig. 4a). The observed mean growth rate calculated using the counts of breeding birds in 1983–1998 of 4.7% per year (Fig. 1a), lies within the range of annual growth rates predicted by our model. In terms of relative decline, there is a 32% chance that the population could decline to less than one-quarter of the its current size (Fig. 4c), and this decline is most likely to occur after 15 years (Fig. 4d). We note that these baseline results assume that the survival and fecundity rates were measured without error and that long-line fishery is not operating and affecting the species.

Measurement uncertainty in the adult survival rates had a considerable effect on the quantitative assessment of the predicted degree of threat of the Amsterdam albatross. The baseline estimate of the chance that population abundance will experience a 25% loss during the next 50 years was 0.30, but measurement error could make this figure as low as 0.17 or as high as 0.50 (Fig. 4b). Similar statements can be made about the maximum and minimum predicted population abundance (Fig. 4a). Measurement error also affected the expected time until a loss of one-quarter of the current abundance: the probability of observing a 25% loss in 20 years was 0.13, with a range between 0.06 and 0.29 (Fig. 4c).

Table 2
Annual estimates and their standard deviations of the adult survival rate of the Amsterdam albatross[a]

	Estimate	Standard deviation
1983	1	–
1984	1	–
1985	0.836	0.098
1986	1	0
1987	1	–
1988	1	–
1989	0.949	0.064
1990	0.943	0.081
1991	0.896	0.065
1992	0.943	0.047
1993	1	–
1994	1	–
1995	1	–
1996	1	–
Average	0.969	0.071
Variance	0.003	

[a] These values were obtained using SURGE (Lebreton et al., 1992) using capture–recapture data. SURGE does not provide standard deviation of the estimates when the survival is equal to one. The heterogeneity among standard deviations of adult survival reflects the variability in the capturability and sampling effort between years.
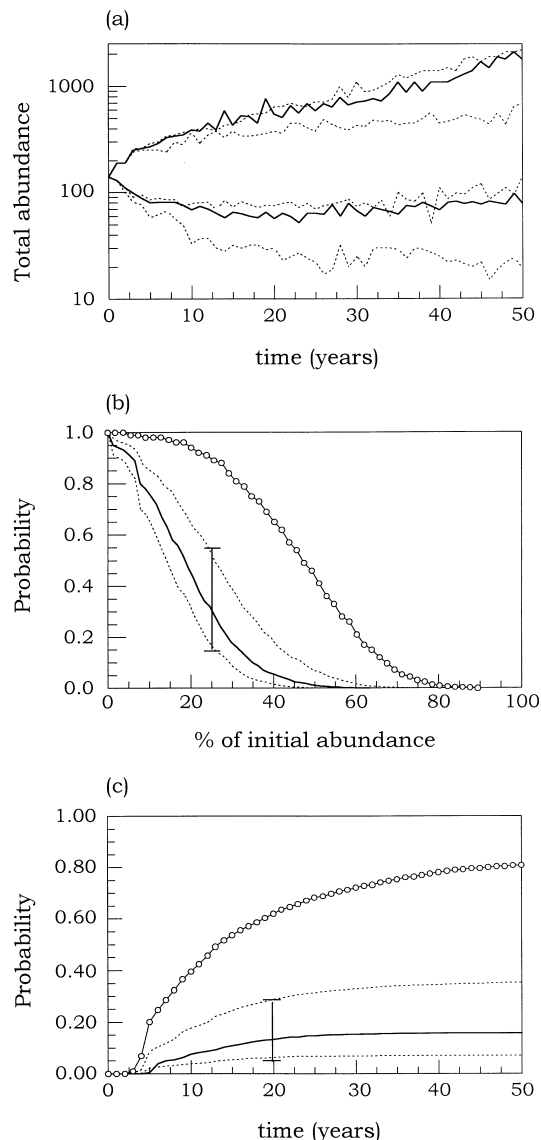

Fig. 4. (a). Predicted maximum and minimum total abundance with (dashed) and without (continuous line) the effect of measurement error. The magnitude of measurement error considered as mean adult survival ±0.025. (b) Probability that the population will decline a given percent of the initial abundance at least once during the next 50 years (lines as in panel a). Open symbols show the effect of measurement error on mean adult survival minus 0.071 (see main text for further details). (c) Probability of having observed a 25% decline of the initial abundance by any year (symbols and lines as in panel b).

Using the wandering albatross data from Weimerskirch et al. (1997; but excluding 1967 and 1969 values as outliers), we estimated a decline in adult Amsterdam albatross with extension of fisheries using the regression: $97.1 - 28.0 \times$ proportion of $5° \times 5°$ squares with long liners; S.E. of slope = 11.00; $n = 22$; $P = 0.02$; $R^2 = 0.20$). Similarly, breeding success (a component of fecundity) declined with the extension of fisheries (regression equation: $77.1 - 36.0 \times$ proportion of $5° \times 5°$ squares; $n = 25$ $P = 0.03$; $R^2 = 0.24$).

The projected effect of the resumption of the long-line fishery was a drastic impact on the predicted population abundance, on the chances of decline and in the time to observe a 25% population decline (Fig. 5). Compared to the baseline results, the resumption of long-line fisheries appreciably lowered the population abundance (Fig. 5a), and drastically increased the probability of observing a 25% decline in population abundance over the next 50 years (Fig. 5b). A projected decrease of 4.2% in the mean adult survival with the lowest projected effect of fisheries made it almost certain (probability = 0.98 compared to 0.32 with no fisheries) that there would be a 25% population decline during the next 50 years (Fig. 5b). In addition, the added mortality from long-line fisheries would greatly increase the chance of a rapid decline to 25% of the current abundance (Fig. 5c). The relatively small decrement in adult and immature survival that could be caused by the resumption of long-line fisheries is roughly equivalent to the deaths of six adult adults per year at the 1998 population size.

We estimated the abundance that the species could have had during the 1960s when long-line fisheries started to operate in this area. We used the values of the survival and fecundity rates predicted by the regression equations using a mean extent of long-line fisheries historically observed in the vicinity of Amsterdam island (data in Weimerskirch et al., 1997). This retro-estimation was carried out by trying different initial population abundances in 1965 until the final number of breeding adults at the end of the simulations approximately matched the number observed in 1983. The results (Fig. 6) suggest that long-line fisheries in the Amsterdam sector could have caused a 10-fold decrease of the Amsterdam albatross population in 18 years, and that its subsequent recovery is probably due to the absence of long-liners in the vicinity of Amsterdam Island.

4. Discussion

The conservation status of the Amsterdam albatross has been based on its low population numbers and its extremely restricted geographic range (IUCN, 1996). This is the first quantitative estimation of the actual risk of extinction of this species and the first on a rare, long-lived seabird species where demographic parameters are

actually known. Our findings that there is at present a relatively low risk of decline should not be interpreted as definite indication of the absolute risk faced by the species. Other features not included in our model such as density dependence, catastrophes and inbreeding effects, could affect the estimated risk of decline (Boyce, 1992; Burgman et al., 1993; Beissinger and Westphal, 1998). There are no indications of inbreeding depression or of density dependent effects during the period in which the Amsterdam albatross has been monitored. Ginzburg et al. (1990) showed that density independent
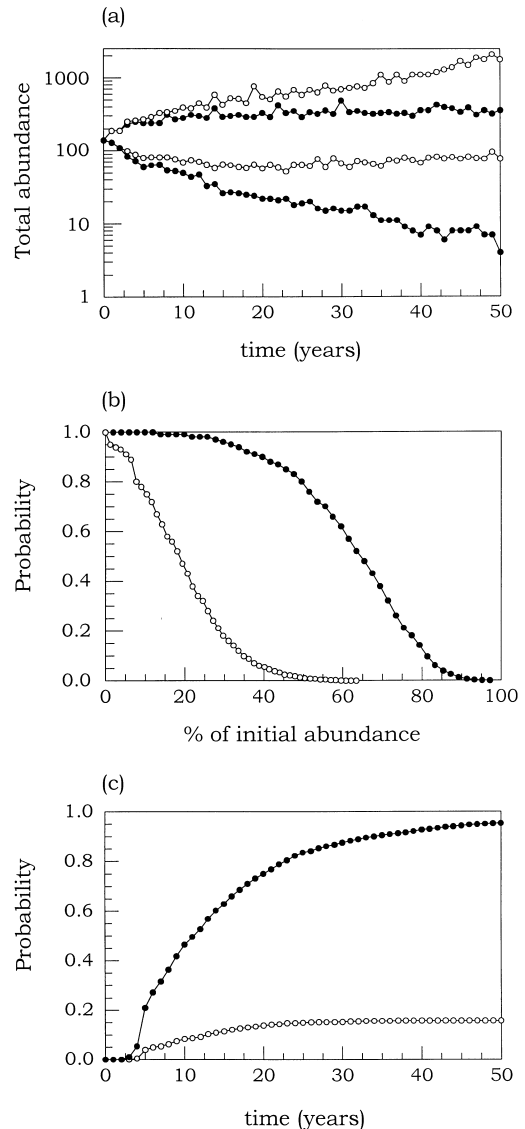

Fig. 5. Projected impact of the long-line fisheries on the abundance and viability of the Amsterdam Albatross. The effects are reflected in the values of the mean fecundity and adult survival rates (Table 1). Open circles represent baseline conditions (i.e. no fisheries); filled circles show the effect of the minimum extension of fisheries. (a) Maximum and minimum total abundance shown on logarithmic scale. (b) Probability that the population will decline a given percent of the initial abundance at least once during the next 50 years. (c) Probability of having observed a 25% decline of the initial abundance by any year.

models generally overestimate the extinction risk compared with density dependent models. Thus, our estimate of the chance of decline of the species based on a density independent model can be considered as an upper bound or conservative estimate of degree of threat face by this species. Catastrophes, i.e. sporadic events with large effects on the population abundance (Mangel and Tier, 1994), are difficult to quantify based on short-term demographic data.

We have shown herein that the Amsterdam albatross does not seem to face a high risk of decline over the next 50 years. This is provided that there is no impact of the long-line fisheries and that the future temporal variation in survival and fecundity rates is similar to that of the recent past. However long term changes in the southern ocean ecosystem are known to occur (Smith et al., 1999) and they might be an additional source of uncertainty for the long-term prospects of the species.

A realistic assessment of the viability of this (and of all) population(s) requires acknowledgement of the role of measurement uncertainty, and a quantification of its influence on the predicted degree of threat. Environmental variability and measurement uncertainties are two sources of uncertainty that should be treated separately because of their different origin and type of influence in model results (Ferson and Ginzburg, 1996). Theoretically, an increase in the sampling effort and/or improvements in field methods could lead to a quantifiable decrease in these measurement uncertainties, which in turn would render a more precise estimate of the risk of decline. In practice, it appears difficult to obtain better estimates of the demographic parameters for a small population of a long-lived seabird that spends most of
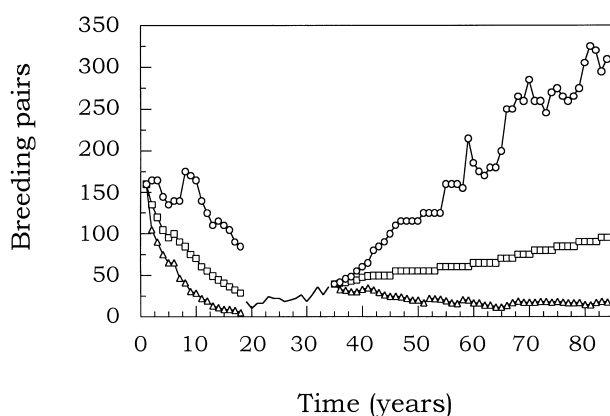

Fig. 6. Reconstructed (i.e. past), observed and predicted numbers of breeding pairs of Amsterdam albatross. Estimated maximum, mean and minimum trajectories shown by circles, squares and triangles, respectively. Reconstructed values obtained from survival and fecundity values assuming a mean extent of long-line fisheries in the Amsterdam sector. Observed values (continuous line) are the same data as shown in Fig. 1a. Predicted values obtained from baseline values of survival and fecundity rates assuming no measurement error and no effect of fisheries.

its time at sea. In the presence of these measurement uncertainties, one may wish to make a conservative prediction of risk of decline. This could be accomplished by adopting a higher value for the risk of decline that is equal to or higher than the 'best' estimate (assuming no measurement uncertainty) such that our assessment errs on the side of caution. We believe that adopting a conservative prediction of risk of decline is justified in this case given that there is only one extant population of the species.

6.1. Past, present and future of Amsterdam albatross

The historic effects of seabird mortality associated with long-line fisheries are difficult to demonstrate due to the paucity of seabird by-catch data and the absence of on-board observers (Weimerskirch et al., 1985, 1997). The Japanese blue fin tuna fishery was the main long-line fishery south of 40°S (Ardill, 1995) that could potentially affect seabirds from sub-Antarctic islands, and one of the few for which there is some by-catch and fishing effort data up to 1995 (Tuck and Polacheck, 1997). More recent by-catch and fishing effort data for the Japanese and Taiwanese fishing fleets (Tuck and Polacheck, 1997) are currently unavailable. The fishery for Patagonian toothfish operates mainly south of the Polar frontal zone, and therefore does not concern Amsterdam albatrosses, which are restricted to sub-tropical waters. The fishing effort for the southern blue-fin tuna is currently low in most of the foraging range of the Amsterdam albatross. This fishery, however, covered the entire foraging range of this species from the 1960s to the 1980s and it was not only intense but also particularly close to Amsterdam Island (Ardill, 1995; Polacheck and Tuck, 1995, for fishery statistics). Weimerskirch et al. (1997) showed that long-line fisheries operating very close to breeding grounds are likely to impact heavily on seabird populations compared to fisheries operating at a larger distances to the breeding areas. All evidence suggests that the Amsterdam albatross is likely to have been affected by long-line fisheries between the mid 1960s and mid 1980s by the fishery operating close to Amsterdam. Whilst extending the potential impact of long-line fisheries from one species to another should be interpreted with caution, we believe that the high degree of ecological similarity between the wandering and Amsterdam albatross justify the approach undertaken. We believe that the recent increase in numbers, concomitant with the shift and reduction of the fishing effort in the central Indian Ocean over the last two decades, has allowed the recovery of the population. Our back-calculation suggests that the breeding population could have been of ca. 320 individuals in the early 1960s.

We think that the Amsterdam albatross case clearly illustrates the need for a two-fold approach

of conservation as stressed by Caughley (1994), i.e. the need for empirical data based on long term monitoring studies together with a modelling approach. The re-establishment of long-line fisheries around Amsterdam Island is a very realistic possibility. The development of new fisheries for swordfish in sub-tropical waters, the deployment of an extensive Taiwanese fleet in the Indian Ocean, and the redeployment of the Japanese fishery for southern blue fin tuna after the closure of the Australian economic zone fishery, make this re-establishment very likely. In addition, we ignore if the long-line fishery for Albacore tuna (*Thunnus alalunga*), which already operates close to Amsterdam Island (Ardill, 1995) may represent a threat for seabirds similar to the southern blue-fin tuna fishery (Weimerskirch et al., 1997). Since these long-line fisheries in the open ocean are not regulated and have no observers on board, they represent a very realistic and worrisome threat for the persistence of the Amsterdam albatross.

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References

Alexander, K., Robertson, G., Gales, R., 1997. The Incidental Mortality of Albatrosses in Long-line Fisheries. Australian Antarctic Division, Tasmania.

Ardill, J., 1995. Atlas of Industrial Tuna Fisheries in the Indian Ocean. Indo-Pacific Tuna Fisheries Development Program, Colombo. IPTP/95/AT/3.

Ashford, J.R., Croxall, J.P., Rubilar, P.S., Moreno, C.A., 1995. Seabird interactions with longlining operations for *Dissostichus eleginoides* around South Georgia. CCAMLR Science 2, 111–121.

Beissinger, S., Westphal, 1998. On the use of demographic models of population viability in endangered species management. Journal of Wildlife Management 62, 821–841.

Brothers, N., 1991. Albatross mortality and associated bait loss in the Japanese long-line fishery in the Southern Ocean. Biological Conservation 55, 255–268.

Boyce, M., 1992. Population viability analysis. Annual Review of Ecology and Systematics 23, 481–506.

Burgman, M., Ferson, S., Akcakaya, H., 1993. Risk Assessment In Conservation Biology. Chapman and Hall, New York.

Caughley, G., 1994. Directions in conservation biology. Journal of Animal Ecology 63, 215–244.

Cherel, Y., Weimerskirch, H., Duhamel, G., 1996. Interactions between long-line vessels and seabirds in Kerguelen waters and a method to reduce seabird mortality. Biological Conservation 75, 63–70.

Collar, N., Crosby, M., Stattersfield, A., 1994. Birds to Watch 2: The World's List of Threatened Birds (Birdlife Conservation Series No. 4. Birdlife International, Cambridge.

Ferson, S., 1990. RAMAS-Stage: Generalized Stage-based Modelling for Population Dynamics. Applied Biomathematics. Setauket, New York.

Ferson, S., Ginzburg, L., 1996. Different methods are needed to propagate ignorance and variability. Reliability Engineering and Computing 54, 133–144.

Gales, R., Brothers, N., Reid, T., 1998. Seabird mortality in the Japanese longline fishery around Australia 1988–1995. Biological Conservation 86, 37–56.

Ginzburg, L., Ferson, S., Akcakaya, H., 1990. Reconstructability of density dependence and the conservative assessment of extinction risk. Conservation Biology 4, 63–70.

IUCN, 1996. International Union for the Conservation of Nature (Report 1996). Geneva, Switzerland.

Jouventin, P., 1995. Past, present and future of Amsterdam Island (Indian Ocean) and its avifauna. Bird Conservation Studies 1, 122–132.

Jouventin, P., Martinez, P., Roux, J., 1989. Breeding biology and current status of the Amsterdam Island albatross. Ibis 131, 171–189.

Klaer, N., Polacheck, T., 1997. By-catch of albatrosses and other seabirds by Japanese long-line fishing vessels in the Australian Fishing Zone from April 1992 to March 1995. Emu 97, 150–167.

Lebreton, J.D., Burnham, K.P., Clobert, J., Anderson, D.R., 1992. Modelling survival and testing biological hypotheses using marked animals: a unified approach with case studies. Ecological Monographs 62, 67–118.

Mangel, M., Tier, C., 1994. Four facts that every conservation biologist should know about persistence. Ecology 75, 607–614.

Murray, T.E., Bartle, J.A., Kalish, S.R., Taylor, P.R., 1993. Incidental capture of seabirds by Japanese southern bluefin tuna long-line vessels in New Zealand waters, 1988–1992. Bird Conservation 3, 181–210.

Perrins, C., Lebreton, J., Hirons, G., 1991. Bird Population Studies. Oxford University Press, Oxford.

Polacheck, T., Tuck, G., 1995. Trends in Tuna Long-line Fisheries in the Southern Oceans and Implications for Seabird By-catch (CCSBT-ERS/95). Division of Fisheries, CSIRO, Hobart, Australia.

Roux, G., Jouventin, P., Mouguin, J., Stahl, J., Weinmerskirch, H., 1983. Un nouvel albatros (*Diomedea amsterdamsis* nova species) découvert sur l'île Amsterdam (37°50′S, 77°35′E). Oiseau et R.F.O 53, 1–11.

Smith, R.C., Ainley, D., Baker, K., Doomack, E., Emslie, S., Fraser, B., Kennett, J., Leventer, A., Mosley-Thompson, E., Stammerjohn, S., Vernet, M., 1999. Marine ecosystem sensitivity to climate change. BioScience 49, 393–404.

Tuck, G., Polacheck, T., 1997. Trends in the tuna long line fisheries in the southern oceans and implications for seabird by-catch: 1997 update. Commission for the Conservation of Southern Bluefin tuna — Ecologically Related species/97/35.

Warham, J., 1990. The Petrels, their Ecology and Breeding Systems. Academic Press, London.

Weimerskirch, H., 1992. Reproductive effort in long-lived birds: age-specific patterns of condition, reproduction and survival in the wandering Albatross. Oikos 64, 464–473.

Weimerskirch, H., Jouventin, P., 1987. Population dynamics of the wandering albatross (*Diomedea exulans*) of the Crozet Islands: causes and consequences of the population decline. Oikos 49, 315–322.

Weimerskirch, H., Jouventin, P., Mougin, J.L., Stahl, J.C., Van Beveren, M., 1985. Banding recoveries and the dispersion of seabirds breeding in French Austral and Antarctic territories. Emu 85, 22–23.

Weimerskirch, H., Clobert, J., Jouventin, P., 1987. Survival in five southern albatross and its relation with their life history. Journal of Animal Ecology 56, 1043–1056.

Weimerskirch, H., Brothers, N., Jouventin, P., 1997. Population dynamics of the wandering albatross (*Diomedea exulans*) and Amsterdam albatross (*Diomedea amsterdamsis*) in the Indian Ocean and their relationship with long-line fisheries: conservation implications. Biological Conservation 79, 257–270.

Weimerskirch, H., Capdeville, D., Duhamel, G., 2001. Factors affecting the number and mortality of seabirds attending trawlers and long-liners in the Kerguelen area. Polar Biology (in press).