

# SEABIRD 15



**SEABIRD GROUP – No. 15 : 1993**

# SEABIRD 15

*Edited by*  
S. Wanless

Produced by  
The Seabird Group

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# Population and productivity trends of Little Terns *Sterna albifrons* in Britain, 1969-89

*J. Sears and M.I. Avery*

## INTRODUCTION

The Little Tern *Sterna albifrons* is one of 79 birds regularly-breeding in Britain singled out for special conservation attention (Batten *et al.* 1990). The British Little Tern population breeds in internationally important numbers and is concentrated into a relatively small number of sites. Little Terns are legally protected through inclusion on Annex I of the European Commission Birds Directive and Schedule I of the UK Wildlife and Countryside Act, 1981. Considerable effort has been expended in wardening Little Tern colonies in Britain since the mid 1970s (Haddon & Knight 1983). The effect of such protection schemes has not been fully evaluated (Avery 1991).

A number of population estimates have suggested that Little Tern numbers are relatively stable but these analyses have simply summed the available records and regarded these as minimum estimates of total population size (Lloyd *et al.* 1975, Thomas 1982, Thomas *et al.* 1989). The aims of this study were to provide a more complete assessment of population trends of Little Terns in Britain from 1969-1989 and to present the first analysis of national trends in productivity. The distribution of birds between colonies and the relationship between colony size and productivity were examined to assess whether birds were becoming concentrated into fewer sites and if this was adversely affecting productivity.

## METHODS

### *Data collection*

Counts of Little Terns from colonies in England, Scotland and Wales have been collated by the Royal Society for the Protection of Birds (RSPB) since 1969. Complete national surveys of Little Terns were made in 1969-70 (Cramp *et al.* 1974) and 1985-87 (Lloyd *et al.* 1991). Only 12 colonies have been counted annually from 1970 to 1989; other colonies have been counted intermittently. A total of 106 colonies which were counted at least once during 1969-89 have been included in the RSPB database of numbers of breeding pairs and young fledged, which is the most comprehensive database available.

Data were collated from a wide variety of sources including reserve wardens, voluntary recorders, county bird reports and county bird recorders and are of variable and usually undefined precision and accuracy.

We have used reported estimates of numbers of breeding pairs, including zero counts (where no birds were breeding at the site). In most cases no details of the methods used are available to us. Where methods are known, many have counted peak numbers of nests, or the numbers of adult birds and used a correction factor (usually 1.5 birds:1 pair (Bullock & Gomersall 1981)) to estimate breeding pairs. Most Little Tern colonies are relatively small, with fewer than 20 pairs, and very few colonies are larger than 100 pairs (see below). Thus errors in counting nests or estimating adult numbers should be relatively small, but are generally unknown. However there are problems in using single nest counts to estimate population size for colonial breeders in that even peak counts usually underestimate the breeding population, especially when there is a large spread of laying dates and/or breeding success is poor (Green & Hirons 1988). Many colonies have been regularly counted by the same individuals or organisations which may mean that there is relatively high comparability of counts between years. Many of the sites which are regularly used by Little Terns are warded and therefore have a constant human presence.

Productivity is recorded as the total number of young fledged per colony. This has been estimated at varying times after hatching, leading to inconsistencies between colonies.

The strengths of this database are its relative completeness (a high proportion of the total population was surveyed each year) and the long run of years for which data on breeding numbers and productivity have been recorded.

### *Population Indices*

Since only a small number of colonies were surveyed annually an index was calculated to describe population changes between 1969 and 1989. The chain method (Marchant *et al.* 1990) was used, whereby colonies surveyed in one year were compared with the same colonies surveyed in the next year. The counts were summed for each of the two years and the percentage change calculated. A baseline figure of 100 was ascribed to a datum year and the percentage change related to this to give an index value. Further pairs of years were used to calculate the next percentage change which was applied to the previous year's index value.

Three chain indices were calculated, using an average of 87% of the colonies surveyed in each year. The first index used colonies counted in consecutive years using 1969 as the datum year. Thus all colonies surveyed in both 1969 and 1970 were analysed followed by all colonies surveyed in both 1970 and 1971, and so on. The second index used colonies counted in alternate years from the datum year of 1969. Thus 1969 and 1971 were compared, then 1971 and 1973, etc. The third index also used colonies counted in alternate years but took 1970 as the datum year. Thus each chain uses only part of the data. The three indices are not independent, since each colony may contribute to all three at different times. If the data matrix had been complete, the two indices for alternate years would intersect the annual index line every two years. The amount of deviation in the lines for alternate year indices from the annual index line gives a measure of the effect of colonies entering or leaving the matrix and therefore represents a simple form of sensitivity analysis.

### *Productivity trends*

Annual productivity was estimated by summing the number of young fledged for all colonies with available information and dividing by the total number of pairs in those colonies, to give the number of young per pair. Two estimates were compared; one including and the other excluding colonies which failed completely (ie. where pairs nested but no young fledged from the colony). Within this paper, unless otherwise stated, 'productivity' refers to estimates including complete colony failures.

The relationship between productivity and colony size was tested using all colonies with five or more productivity records. The smallest colony included had an average size of four pairs. The number of pairs and the annual productivity were analysed by Spearman's rank correlation for each colony.

If the matrix of years and colonies had been full then it would be straightforward to test for consistent trends in productivity between years across sites (year effects) and between sites over years (site effects). However, because the data matrix has as many missing values as full ones, subsets of the data were examined. Two subsets were chosen which provided full matrices; one of seven sites with productivity data for 14 years (1976-89)(subset A) and the other with 16 sites with five years of productivity records (subset B). Both subsets of data were analysed by Friedman two-way analysis of variance.

Statistical tests were performed using the SYSTAT statistical package (Wilkinson 1987). All means are given  $\pm 1$  standard deviation.

## RESULTS

### *Characteristics of the database*

One hundred and six Little Tern colonies monitored for 21 years gives a potential total of 2226 colony/years. The number of breeding pairs (including zero counts) was recorded for 1142 colony/years; 51% of the total. Twelve colonies have complete records of number of pairs for all 21 years from 1969-89 and 87 colonies were surveyed more than once.

The population indices might be biased if large colonies were counted more often than small ones if, for example, large colonies are more likely to be increasing in size. In order to test this, the relationship between colony size and number of records was examined. There was a significant positive correlation between the mean number of pairs and the number of years of records for each colony when all 106 colonies were analysed (Spearman's rank correlation  $r_s=0.22$ ,  $df=104$ ,  $P<0.05$ ). Taking only colonies with two or more counts, there was no significant correlation between colony size and number of records ( $r_s=0.148$ ,  $df=85$ ,  $P>0.05$ ). Most of the 19 colonies recorded once were counted in 1969, during a national survey (Cramp *et al.* 1974). All 19 were small when they were counted; only one had over 20 pairs (mean number of pairs =  $9.89 \pm 6.15$ ). Since they were not recorded in other years they could not be included in the chain indices and were excluded from the analyses. However, the fact that the correlation coefficient was positive suggests a tendency for large colonies to be recorded more often. It is likely that zero counts are underrepresented within the database, although, where possible, years of missing data were checked for unsubmitted zero counts.

The above test would be insensitive to observers ceasing to monitor previously large colonies when these colonies decline. We therefore carried out a further test. Colonies were divided into two groups, large and small ( $>20$  pairs and  $<20$  pairs, including zeros), on the basis of their size in a randomly chosen year, 1979. They were further categorised by whether the number of pairs increased or decreased in the following year, 1980. There were no significant differences between the mean number of records from 1980 onwards for each of the four categories (large increasing, large decreasing, small increasing, small decreasing;  $n=6, 14, 22, 11$ ; means = 9.8, 8.7, 8.2, 8.9 years; Kruskal-Wallis one-way analysis of variance,  $P>0.05$ ). This suggests that the number of times a colony is monitored is largely independent of its initial size and trend in numbers.

The numbers of colonies surveyed each year varied from 34-68 and the number of pairs from 792 to 1767 (Table I). The proportion of the British population represented by the indices can be estimated by comparing the number of pairs contributing to the indices with recent population estimates. Thomas *et al.* (1989) recorded 1528 pairs in 1984 in England (excluding the Isle of Man), Scotland and Wales and Lloyd *et al.* (1991) estimated that 2400 pairs bred in Britain in 1985-87. The 1984 count is an underestimate due to incomplete coverage. In 1985-87 an average of 1433 pairs contributed to the indices, which therefore represent around 60% of the estimated British Little Tern population.

The data were examined for changes in recording effort over the 21 years. The number of colonies known to have been surveyed increased significantly through the period 1969-89 (Kendall tau = 0.625,  $z=3.96$ ,  $P<0.0001$ ) but the number of occupied colonies recorded did not (Kendall tau = 0.236,  $z=1.5$ ,  $P=0.07$ ) (Table I). It is likely that the total number of colonies surveyed has increased because terns have occupied new colonies but recorders were continuing to visit previously occupied colonies.

### *Population trends*

The three indices of population change in breeding pairs show a relatively consistent pattern (Fig. 1). There was a rapid increase in the mid 1970s, levelling off from 1977 to 1982, followed by a slight decline in the mid 1980s. Numbers appeared to have partially recovered by 1987-89. We know of no appropriate statistical technique to test the significance of these overall changes, due to



TABLE I. NUMBERS OF LITTLE TERN PAIRS RECORDED AT COLONIES IN BRITAIN, 1969-89.

Year	69	70	71	72	73	74	75	76	77	78	79	80	81	82	83	84	85	86	87	88	89
No. colonies with records	58	34	40	34	38	37	41	51	57	53	62	65	58	59	60	61	62	59	55	67	68
No. occupied colonies with records	53	30	35	29	35	31	40	47	53	48	54	55	52	49	50	51	51	48	43	50	45
Total Count (No. of pairs)	1225	792	1033	957	823	900	1374	1474	1544	1467	1767	1689	1597	1651	1535	1475	1523	1333	1444	1506	1497

TABLE II. COLONY SIZE AND STRUCTURE, 1969-89.

Year	69	70	71	72	73	74	75	76	77	78	79	80	81	82	83	84	85	86	87	88	89
Mean colony size (No. of pairs)	23	26	30	33	24	29	34	31	29	31	33	31	31	34	31	29	30	28	34	30	33
% occupied colonies <20 pairs	66	67	51	52	63	55	60	55	53	56	54	53	50	51	48	55	55	52	51	56	47
% of total count in five largest colonies	43	61	49	56	51	52	45	42	43	41	41	38	36	36	42	40	36	38	39	44	41

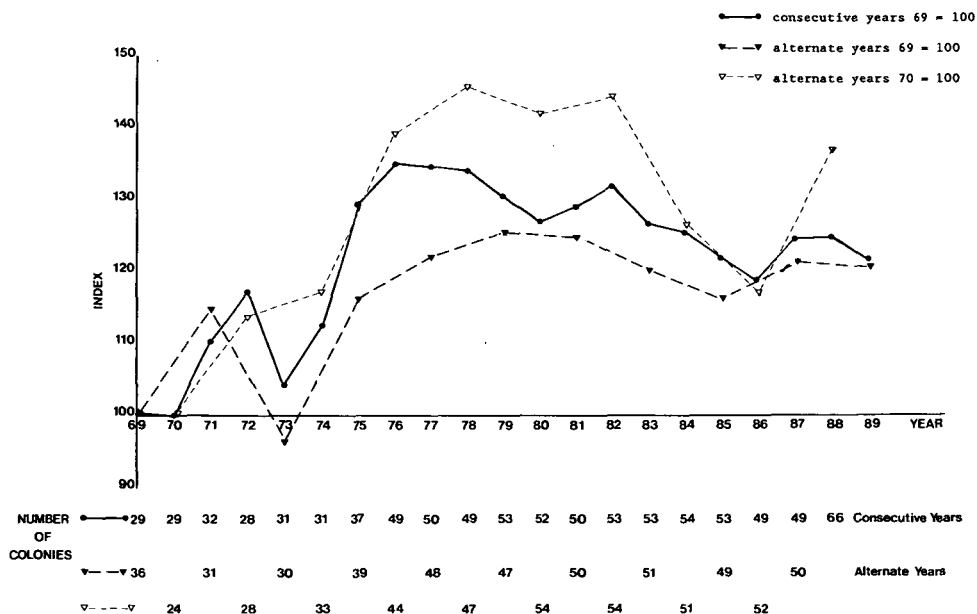


Figure 1. Population indices of breeding Little Terns in Britain, 1969-89, comparing three indices. The number of colonies surveyed in each pair of years is given beneath the figure.

the lack of independence between data points (Marchant *et al.* 1990). The fact that the three indices show broadly similar trends suggests that they reflect true changes in the population and that each index is not greatly biased by colonies entering and leaving the data set.

#### Colony size

There was relatively little variation in average colony size (Table II). Mean colony size ranged from 23.1 to 33.7 pairs (overall mean  $30.1 \pm 3.0$  pairs) and showed no significant tendency to change through the period (Kendall tau = 0.231,  $z=1.46$ ,  $P>0.07$ ). The percentage of occupied colonies under 20 pairs varied from 47 to 67% (mean  $54.76 \pm 5.33\%$ ) and decreased significantly over the period 1969 to 1989 (Kendall tau = -0.383,  $z=2.42$ ,  $P=0.008$ ).

The number of pairs in the five largest reported colonies in each year was compared with the total recorded (Table II) and accounted for a relatively constant proportion (mean  $43.5 \pm 6.74\%$ ).

Therefore, overall the British Little Tern population appears to have been quite stable since the mid 1970s. However, within this relatively stable pattern, there has been considerable variation at individual colonies. Over the 21 years, 19 different colonies were included in the largest five in each year, comprising 22% of the 87 colonies with multiple counts that have been analysed (Table III). Several colonies were only included in one or two years. Three colonies consistently featured: Blakeney Point, Norfolk in 20 years, Foulness/Maplin Bank, Essex in 16 and Langstone Harbour, Hampshire in each of the 10 years from 1980 to 1989. Some large colonies have declined in numbers, such as Foulness/Maplin Bank which held 360 pairs in 1983 and only 28 in 1989. Others have completely disappeared. For example at Ravenglass, Cumbria, which held 59 pairs in 1971, no Little Terns have bred since 1982. New colonies have been formed at sites such as Great

TABLE III. COLONIES INCLUDED IN THE LARGEST FIVE IN AT LEAST ONE YEAR, 1969-89.

<i>Colony</i>	<i>Number of years in largest five</i>	<i>Maximum size (no. of pairs)</i>	<i>Year</i>
Blakeney Point, Norfolk	20	216	1975
Foulness/Maplin Bank, Essex	16	360	1983
Langstone Harbour, Hampshire	10	171	1989
Scolt Head, Norfolk	9	110	1984
Tetney Marshes, Lincolnshire	8	120	1977
Chesil Beach, Dorset	7	125	1971
St. Cyrus, Kincardine & Deeside	4	106	1972
Winterton Dunes, Norfolk	4	90	1972
Keyhaven-Pitts Deep, Hampshire	4	88	1976
Holkham, Norfolk	3	120	1988
Walton-on-the-Naze, Essex	3	63	1981
Great Yarmouth, Norfolk	2	180	1989
Chichester Harbour, W.Sussex	2	102	1975
Pagham Harbour, W.Sussex	2	68	1981
Ravenglass, Cumbria	2	59	1971
Hurst-Pennington, Hampshire	1	80	1982
Rye Harbour, E.Sussex	1	76	1985
Colne Point, Essex	1	75	1986
Sands of Forvie, Gordon	1	74	1977

Yarmouth beach, Norfolk, where numbers have increased from nine pairs in 1982 to 180 pairs in 1989. Some interesting regional changes have occurred. In Wales in 1980, 65 pairs of Little Terns nested in six colonies. In 1985 the Welsh population of 50 pairs occupied only two colonies and by 1989, all 53 pairs were concentrated at one site, at Gronant.

#### *Productivity trends*

The number of young fledged was recorded at least once for 67 colonies and at least five times for 46 colonies. The number of productivity records per colony was significantly positively related to the mean colony size (Spearman's rank correlation coefficient,  $r_s=0.27$ ,  $df=85$ ,  $P=0.02$ ) indicating that productivity is monitored more often in large colonies than in small ones. This means that the data are biased towards larger colonies, although this is unlikely to affect the results since there is no significant relationship between colony size and productivity (see below). Productivity was assessed for about half of the total number of colonies surveyed and 60% of the total number of pairs recorded each year.

The average productivity for each year 1969-89 is plotted in Figure 2, including and excluding complete colony failures (where pairs nested but no young were fledged). There are wide variations in average productivity between years, with a four-fold range from 0.18 to 0.88 young per pair including complete failures and 0.23 to 0.90 when they are excluded. Despite the increase in sample size, from a mean of  $19.2 \pm 8.1$  colonies recorded in 1969-79 to  $36.0 \pm 4.9$  in 1980-89, the degree of variation between years in average productivity was only slightly reduced in the second period (1969-79: range=0.18-0.81; 1980-89: range=0.32-0.88 (inclusive of complete failures)) (Fig. 2).

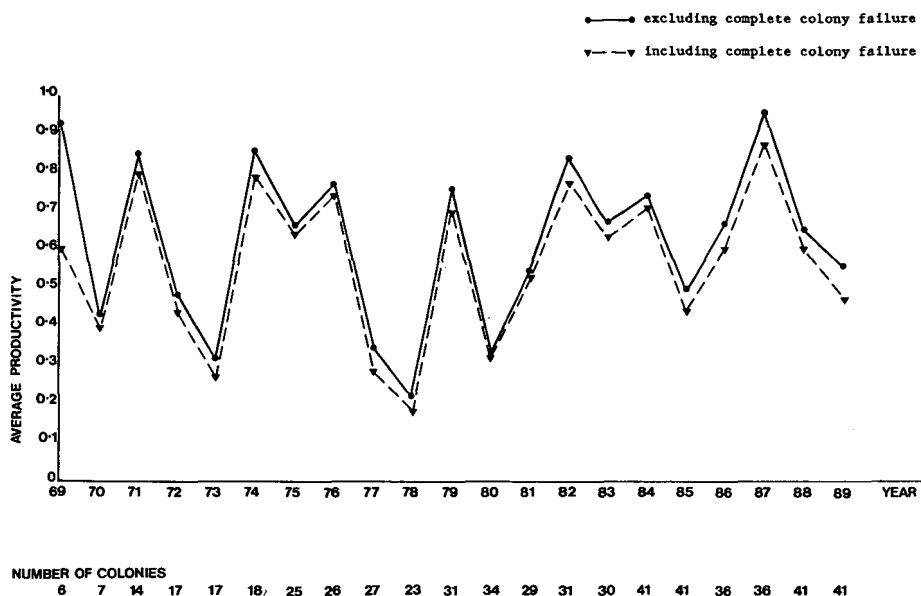


Figure 2. Average annual productivity (number of young per pair) of Little Tern colonies in 1969-89, including and excluding complete colony failures. The number of colonies surveyed each year is given beneath the figure.

The mean annual productivity for the 21 years, including complete colony failures, is 0.56 young/pair (0.62 young/pair excluding complete colony failures). There is no consistent trend in productivity over time (Kendall tau = 0.06,  $z=0.39$ ,  $P=0.34$ ) and no apparent relationship between productivity and population trends (Fig.1) (see population dynamics section of discussion).

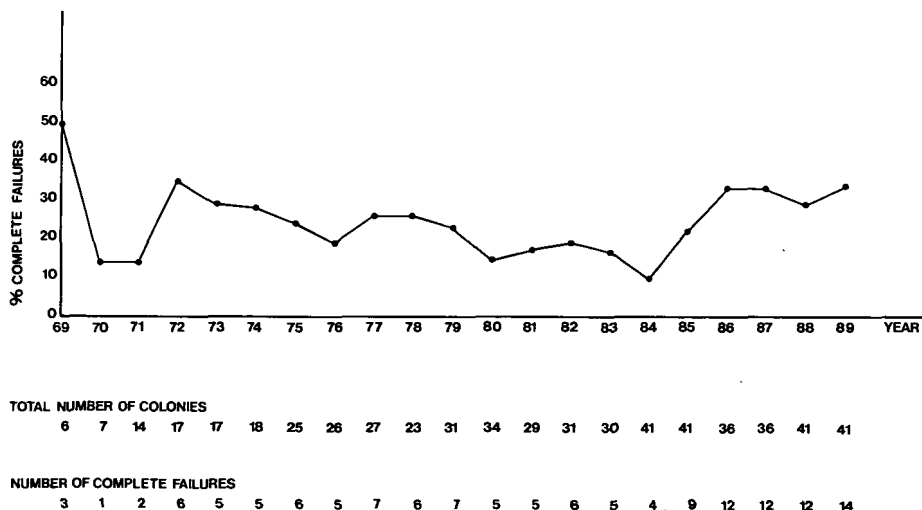
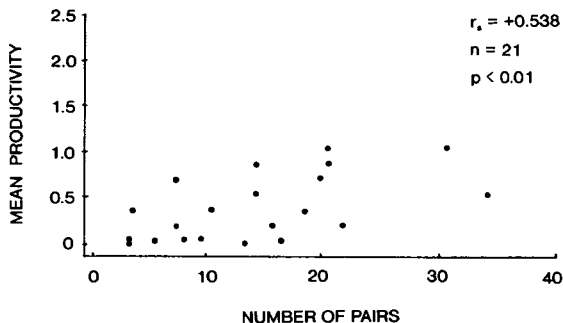


Figure 3. Proportion of complete failures amongst Little Tern colonies with productivity records, 1969-89.

The proportion of colonies which failed completely ranged from 10% to 34% of all colonies where productivity was recorded (average for 1969-89 =  $24.6 \pm 9.1\%$ ). The proportion remained relatively constant over the whole 21 years (Fig.3) and showed no trend over time (Kendall tau = 0.02,  $z=0.12$ ,  $P=0.45$ ).

### GIBRALTAR POINT



### RECULVER

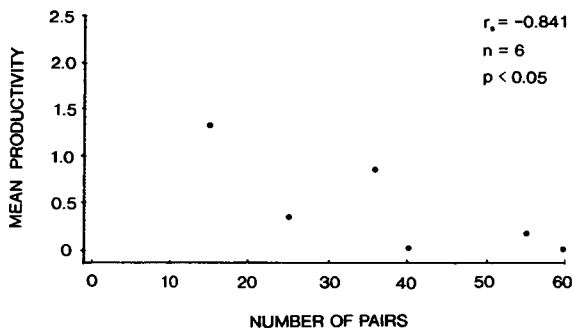


Figure 4. Examples of positive and negative correlations between productivity and colony size: two Little Tern colonies in England.

#### *Relationship between productivity and colony size*

For the 46 colonies with five or more productivity records (including complete failures) we investigated whether there was a relationship between breeding numbers and productivity. Twenty-three rank correlations were positive and 23 were negative. One of the negative and three of the positive correlations were significant. One of each is illustrated in Figure 4. There was no tendency for all the positive correlations to be large colonies and all the negative ones to be small colonies or vice versa (Chi-squared, comparing mean colony size  $>20$  and  $<20$  pairs, = 0.784,  $df=1$ ,  $P>0.05$ ). Thus there was no apparent relationship between colony size and productivity. However, because the breeding population is likely to be underestimated, especially when breeding success is low, there may be a stronger positive association than is apparent.

*Site and year effects on productivity*

In the analysis of the two subsets of data for site and year effects, subset A (7 sites with 14 years' data) showed no consistent variation between years (Friedman test statistic =16.804, df=13,  $P=0.208$ ) but a significant year effect was found for subset B (16 sites with 5 years' data) (Friedman test statistic =15.725, df=4,  $P=0.003$ ). This was largely due to the effect of one year, 1978, when five of the 16 colonies failed completely, one due to tidal flooding, one due to Kestrel *Falco tinnunculus* predation and three for unknown reasons.

A significant site effect was recorded for both subsets of data (subset A: Friedman =29.105, df=6,  $P=0.0001$ ; subset B: Friedman =31.434, df=15  $P=0.008$ ). This suggests that some sites tend to be consistently more or less productive than others.

Of the colonies with at least five productivity records, the ten most productive and the ten least productive are listed in Tables IV and V. There was little evidence that any part of the country was more or less productive than others, although three Lincolnshire colonies were among the least productive and two Norfolk colonies among the most productive. The remaining colonies were widely distributed.

TABLE IV. THE TEN MOST PRODUCTIVE COLONIES WITH AT LEAST FIVE PRODUCTIVITY RECORDS.

<i>Colony</i>	<i>Average productivity*</i>	<i>Number of years</i>	<i>Mean no. of pairs</i>	<i>Number of years</i>
Holy Island, Northumberland	1.43	19	22.05	19
Great Yarmouth, Norfolk	1.30	5	62.88	8
Shellness, Kent	1.04	8	13.70	10
Winterton Dunes, Norfolk	1.01	12	34.26	19
Gronant, Clwyd	0.98	19	29.35	20
Walton-on-the-Naze, Essex	0.92	6	34.80	20
Abermenai, Gwynedd	0.79	6	5.00	8
Easington Lagoons, N. Humberside	0.78	8	8.80	10
Ballantrae, Kyle & Carrick	0.74	10	5.94	17
Tywyn, Gwynedd	0.73	18	13.63	19

\* Total number of young/total number of pairs.

TABLE V. THE TEN LEAST PRODUCTIVE COLONIES WITH AT LEAST FIVE PRODUCTIVITY RECORDS.

<i>Colony</i>	<i>Average productivity*</i>	<i>Number. of years</i>	<i>Mean no. of pairs</i>	<i>Number of years</i>
Aberlady, Lothian	0.17	6	20.42	12
Sands of Forvie, Gordon	0.18	13	37.38	16
Point of Ayre, Clwyd	0.21	5	5.83	6
Dungeness, Kent	0.22	10	12.15	13
St. Cyrus, Kincardine & Deeside	0.24	15	37.50	18
Donna Nook, Lincolnshire	0.25	17	21.38	21
Saltfleetby/Theddlethorpe, Lincs	0.26	13	12.25	16
Chesil Beach, Dorset	0.30	14	62.65	20
Foulney, Cumbria	0.31	11	14.40	15
Saltfleet, Lincolnshire	0.31	6	9.07	14

\* Total number of young/total number of pairs.

## DISCUSSION

### *Data quality*

We would emphasise again that as the data have been collected by a large number of recorders and methods of data collection are often undefined, the accuracy and precision of the various counts which make up our database are unknown. However, for no other seabird breeding in Britain has such comprehensive monitoring of both numbers and productivity been undertaken on an annual basis. Most seabird monitoring is based on the results of detailed studies at a few, sometimes unrepresentative, colonies, whereas in this database numbers have been counted at a high proportion of colonies in each year, representing over half the population in the most recent years. On average, productivity has been assessed in 50% of the colonies surveyed in a given year. Although this database is not complete it has more in common with complete population surveys than with sampling schemes.

One problem of assessing tern numbers is that of duplicate counting of individuals (Lloyd *et al.* 1991) due to inter-colony movements and varying count dates between colonies. When birds attempt to nest at one colony, fail, and then move to another colony they may be counted twice. This problem is likely to apply not only to our population data but also to those of Cramp *et al.* 1974, Lloyd *et al.* 1975, Thomas 1982, Thomas *et al.* 1989 and Lloyd *et al.* 1991. One way this might be overcome would be to set a date for counting at all colonies. A high proportion of Little Terns nest at wardened sites where this should be feasible but it would be difficult to co-ordinate for the less accessible sites, such as Scottish islands, many of which are counted by volunteers in their spare time. There would also be problems in deciding a suitable date. It would have to vary annually due to the vagaries of weather and time of breeding. The best time may be during the peak of incubation, when the maximum number of birds are on nests. This would still miss birds which had failed early but not re-nested and any that had failed close to the count date and had not yet had time to re-nest. It would also miss any late breeders. The peak time would vary between colonies. There is thus no easy method for ensuring an accurate total count of terns.

### *Population and productivity monitoring: thresholds for action*

One of the functions of monitoring population numbers and productivity of species of conservation importance is to detect adverse changes in their status in time to be able to propose effective remedial action. This paper presents long-term data of a type which would be invaluable if productivity or numbers suddenly dropped dramatically. On the basis of our current knowledge we suggest that if either the year-on-year population index fell below 110 for two years in succession or if the national average productivity estimates fell below 0.25 young/pair for two years in succession then this might be cause for concern, since neither has happened at any time in the past 21 years.

### *Population dynamics*

The observed changes in the chain index of Little Terns nesting in Britain must be due to one (or a combination) of the following factors: counting error, variation in productivity, variation in survival, emigration and immigration. We can assess the extent to which our measures of productivity alone can account for the changes in the chain index which we have calculated. There are two aspects we can consider. First, could our productivity estimates, in combination with realistic assumptions about annual survival, account for the slight increase in Little Tern numbers over the 21-year period? Second, could the annual variations in productivity account for the variations in the index with time?

For a closed stable population the number of adults dying each year is balanced by recruitment of new breeders. The number of recruits is equal to the number of fledged chicks which survive to the age of first breeding. Taking any average productivity and the age at first breeding, one can

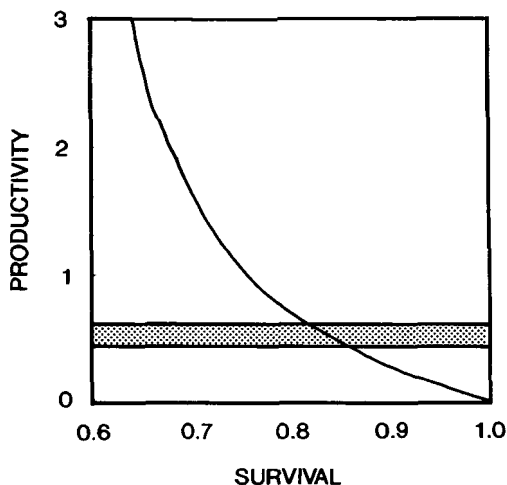


Figure 5. Combinations of survival and productivity and their consequences for population change. Areas above the curve represent combinations that lead to population increases; below the curve the population will decrease. The shaded area shows the 95% confidence intervals for the observed mean annual productivity of British Little Terns 1969-89. We assume that annual survival of breeding birds and birds younger than breeding age are identical (in practice we realise the latter is likely to be lower than the former; this would lead to an underestimate of adult survival). We also assumed that the age of first breeding is three years (Cramp 1985) although the survival estimate is relatively robust to this assumption.

calculate the necessary survival rate for population stability (Fig. 5). Given our estimate of average chick production this indicates that Little Terns must have greater than 80% survival to be at least stable in numbers. Is this survival biologically feasible? Unfortunately there have been no estimates of survival for Little Terns breeding in Britain, but Grosskopf (1964) estimated Little Tern survival at around 80% (in Glutz von Blotzheim & Bauer 1982). Survival for two other tern species is also estimated to be around this value (Common Tern *Sterna hirundo* 80%, Grosskopf 1964; Arctic Tern *S. paradisaea* 87%, Coulson & Horobin 1976). Thus, the estimate of survival which is needed to account for the gross observed change in population numbers is close to that found for this species in other areas, and for closely related species. From this evidence, we cannot dismiss the possibility that the British population of Little Terns is closed and self-sustaining.

How well do our productivity estimates reproduce the observed changes in the chain index? We chose the constant value (0.847) of annual survival which reproduced the observed average of the chain index over the period 1972-1989 (Fig. 6). It cannot be claimed that this best-fit model describes the observed index very well. Thus the number of breeding pairs has not fluctuated in relation to the variations in annual productivity. This may be because of: variations in mortality; errors in our estimates of numbers or productivity; emigration or immigration. Few data exist even to allow speculation on the likelihoods of these possibilities.

If we had found that our productivity estimates, combined with a constant and feasible survival value, could reproduce the year-to-year variations in the chain index then we would have grounds for believing that Little Tern population dynamics in Britain were very dependent on the mean national production of fledged young. This would be a good reason for further improving conservation actions which promote high nesting success. The results of this simple modelling exercise suggest that studies of movements and survival of Little Terns may be needed before the changes in Little Tern numbers in Britain can be understood.



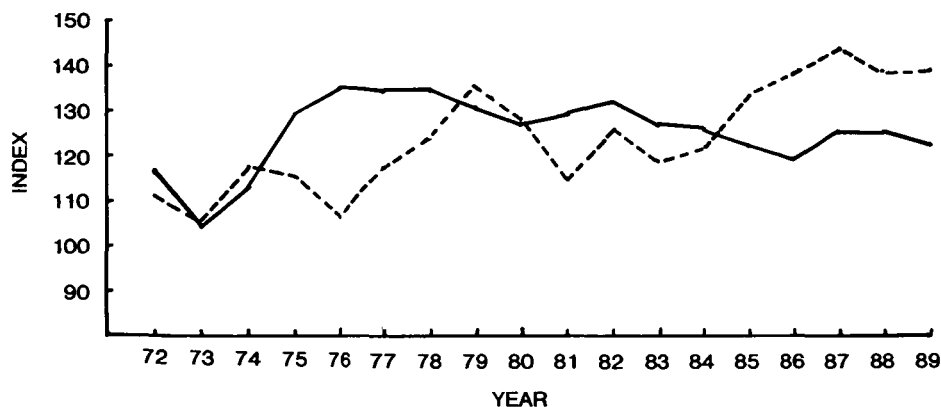


Figure 6. The observed change in chain index (solid line) and the predicted change (dotted line) based on the observed annual variation in productivity and taking a value of annual survival (0.847) which produces the same average chain index as the observed over the period 1972-89. We have assumed that the age of first breeding is three years, but taking values of either two or four do not improve the fit. To allow for recruitment of Little Terns fledged in 1969, we have modelled the population change from 1972 onwards.

#### Conservation measures

Tens of thousands of pounds, and thousands of hours of voluntary and professional effort, have been spent on wardening Little Tern colonies in the past 20 years (Haddon & Knight 1983, Avery 1991). Overall our data show no increase in numbers over the past ten years and no increase in productivity over the 21 years. It is difficult to assess whether conservation efforts have been effective at preventing a decline or ineffective at promoting an increase in numbers.

To examine the effect of protection schemes on breeding success one needs to compare numbers and productivity with and without protection. Ideally this comparison needs to be made within colonies, to allow for site factors that influence productivity. There are several problems associated with this. Few sites have changed their protection status recently and where this has occurred it has always been from unprotected to protected (thus the effect of increasing protection is confounded by any year effects which might exist). There are various degrees of protection, ranging from passive schemes (such as fencing and signs), to active wardening (such as predator-deterrence, disturbance control and nest-raising etc), which may be part- or full-time. In many cases the transition from unprotected to protected has been gradual, involving various types of schemes, and often the available information on what was done when is sparse. At many sites productivity data were not collected before wardening began so it is impossible to compare breeding success before and after protection. The second best option would be to compare protected and unprotected sites. This also meets the problems of varying degrees of protection and lack of available information, limiting comprehensive analysis of the effect of protection schemes.

Records of individual colonies show large fluctuations in annual productivity even after full-time wardening began. Wardening usually reduces human disturbance but it is difficult to prevent breeding failure due to flooding or predation. Recently, several large, fully warded colonies have been preyed upon by Foxes *Vulpes vulpes*, Hedgehogs *Erinaceus europaeus*, Kestrels and Crows *Corvus corone*, resulting in severe losses of nests and young. For example, at Great Yarmouth beach, Norfolk, where numbers have increased from nine pairs in 1982 to 277 pairs in 1991, and the site has been warded from 1986-1991, nesting success was initially high (with an average of 1.6 chicks/pair raised in 1986-88). In 1989 productivity dropped to 0.88 chicks/pair, largely due to predation of chicks by Kestrels. In 1990 only 15 chicks fledged from 201 nests (an

average productivity of 0.07 chicks per pair) and in 1991 only 12 chicks were fledged (0.04 young per pair). Predation, mainly by Kestrels and mammals, has been the cause of this greatly reduced breeding success in more recent years. Langstone Harbour, Hampshire has been a fully-wardened RSPB reserve since 1980 and Little Terns nesting there have increased from 27 pairs in 1979 to 171 in 1989. Breeding success was relatively high in 1986 and 1987, at over one chick per pair, but predation by Foxes and Kestrels reduced the success to 0.03 in 1988 and 0.18 in 1989. All 145 pairs failed completely due to Fox predation in 1990. Even chicken wire exclosures around individual nests, a measure used at Newcastle, County Wicklow, Ireland, only achieved limited success. Although ground predators were deterred, a Kestrel learned to enter the exclosures and caused complete failure in 1990 (O'Briain & Farrelly 1990).

On the whole, protection schemes are likely to have benefitted Little Terns since without them many sites might have been too disturbed to be occupied. However it is difficult to assess whether colony management has been effective in increasing productivity. Little Terns may be adapted to move site when confronted with predators or problems of flooding. There is therefore a need to maintain a number of alternative sites to enable colony movements to occur. Such sites may include currently unused but suitable locations which need to be kept free of disturbance.

#### ACKNOWLEDGEMENTS

These data have been collected by hundreds of people, many of them amateur bird-watchers, most of whom we have never met and some of whose names have probably never reached us; but we would like to thank them all. Frances Winder, Gareth Thomas and Lesley Underwood established and maintained the database; RSPB regional staff collated most of the data; RSPB, NCC, National Trust and local Naturalist Trust wardens and numerous volunteers collected data and protected colonies; many County Bird Recorders responded when we contacted them in 1987. Some of the counts used were collated and provided by the JNCC/Seabird Group Seabird Colony Register and we are grateful for their collaboration. Rhys Green, Fiona Hunter, Mike Harris, Mark Tasker, John Uttley and Paul Walsh kindly commented on earlier drafts of this paper.

#### SUMMARY

An index of numbers of Little Terns nesting in Britain has increased since 1969 but remained relatively stable since the late 1970s. Mean colony size has remained at around 30 pairs. Individual colonies have varied greatly in size. Of a total of 106 colonies, 19 have been among the five largest in Britain in at least one of the 21 years. Average annual productivity was 0.56 young/pair, but varied from 0.18-0.88 young/pair. There was no consistent trend in productivity over time and no apparent relationship between productivity and population trends. There was no consistent positive or negative relationship between colony size and productivity. Some sites are consistently more or less productive than others. Problems of predation at large colonies and the need to maintain alternative sites are briefly discussed.

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# The status of Arctic Terns *Sterna paradisaea* in Orkney and Shetland in 1989

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## INTRODUCTION

The Arctic Tern is one of the 79 regularly-breeding British bird species considered by Batten *et al.* (1990) to be of highest conservation importance and is listed on Annex 1 of the European Community Birds Directive. A full survey of Orkney and Shetland, conducted in 1980, estimated 64,863 pairs, about 85% of the British and Irish population (Bullock & Gomersall 1981).

Between 1984 and 1990, Arctic Tern chick production was very low in Shetland due to food shortage whereas on Orkney productivity was higher and appeared to have been relatively unaffected by food shortage (Monaghan *et al.* 1989, Walsh *et al.* 1990, 1991). Other seabirds, notably Kittiwakes *Rissa tridactyla*, Great Skuas *Stercorarius skua* and Puffins *Fratercula arctica* suffered reduced breeding success in Shetland over the same period, but are not known to have been so severely affected in Orkney (Heubeck 1989, Hamer *et al.* 1991, Martin 1989). Our survey aimed to assess changes in numbers of Arctic Terns in Orkney and Shetland between 1980 and 1989.

## METHODS

### *Timing of colony visits*

Colonies were visited once between 25 May and 12 July 1989. These dates are comparable to those of the 1980 survey and span the period between the laying of the first clutches and the early chick fledging period. Those colonies known to have been occupied in 1980 were randomly allocated to one of three groups to be visited once at either the beginning, middle or end of the survey period. This was done so that the three groups of colonies visited were similar in all respects so that comparisons could be made between counts made at different stages of the breeding season.

New colonies, those not recorded in 1980, fell into two groups; those located since 1980 but prior to 1989, and those found for the first time during the course of the present survey. For the purposes of the analysis both categories are regarded as the same. New colonies in Orkney were counted only once when found in the course of visiting the 1980 sites. In Shetland some colonies were visited several times. For these colonies, one count was selected at random.

### *Fieldwork*

Most of the fieldwork was carried out by DB in Orkney and NJD, MM and PME in Shetland, but additional volunteers counted terns at specific sites. As far as practically possible colonies were visited in the designated section of the season, although inevitably some constraints were imposed by the weather and the availability of transport. All the personnel involved were experienced at counting terns.

Birds were counted at the colonies by means of flushed counts as described by Bullock and Gomersall (1981). All results refer to numbers of adult birds and not pairs.

### *Coverage*

All known Arctic Tern colonies in Orkney and Shetland were visited and we believe we achieved practically complete coverage of possible breeding areas.

### Data analysis

This survey was thought to have achieved practically complete coverage of potentially suitable breeding habitat in Orkney and Shetland. Therefore, no consideration need be given to calculating confidence intervals for the estimates of population size since sampling errors are not involved in this study. The only sources of error in our estimates are counting errors. With large samples such as ours unbiased errors in counting terns in individual colonies will be expected to roughly cancel each other out. However, if there were biases in our counts (e.g. if we always underestimated large colonies) then this would affect our estimates and probably those of all previous studies. Counts of individual colonies on each island, or in the case of Shetland mainland, different parts of the island (Fig. 1), were summed for comparison with the 1980 counts. Statistical analysis was performed using SYSTAT (Wilkinson 1987).

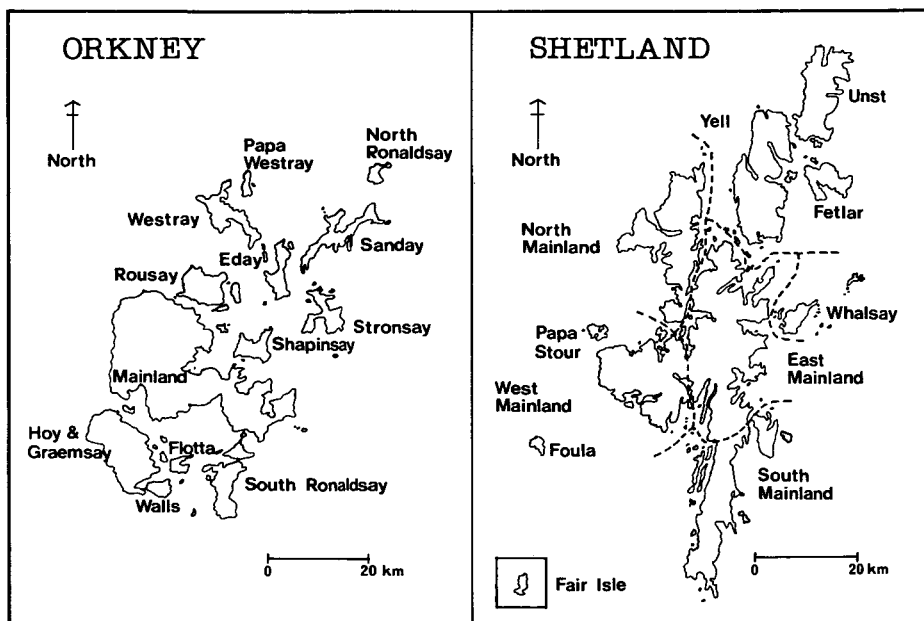


Figure 1. Maps of the Orkney and Shetlands Islands showing the islands and island divisions listed in Table II.

### RESULTS

We visited 800 Arctic Tern colonies in Orkney and Shetland. Colonies visited in the beginning, middle and end of the survey period were well matched in terms of three descriptive variables which were quantifiable; altitude, distance from coast and colony size in 1980 (Table I).

Shetland colonies held less than half the numbers of birds which had been present in 1980; a 59% decrease for those colonies visited in the first third of the study and a 51% decrease for those visited in the middle of the breeding season. These two figures suggest that the number of Arctic Terns present on the breeding grounds in Shetland, which may attempt to breed, was about 55% lower in 1989 than in 1980. In the last third of the season, numbers of birds dropped to only 18% of

TABLE I. MEAN NUMBERS OF ADULT ARCTIC TERNS ATTENDING COLONIES IN ORKNEY AND SHETLAND IN 1989, AND PHYSICAL ATTRIBUTES OF THOSE COLONIES, IN THREE DIFFERENT PARTS OF THE NESTING SEASON.

a) *Orkney*

	<i>Early</i>	<i>Middle</i>	<i>Late</i>	<i>P</i>
Altitude (m)	10	11	10	ns
Distance from sea (m)	435	370	463	ns
1980	276	77	88	ns
1989	159	44	97	ns
n	90	94	116	
% change	-43	-42	+10	

b) *Shetland*

	<i>Early</i>	<i>Middle</i>	<i>Late</i>	<i>P</i>
Altitude (m)	20	19	22	ns
Distance from sea (m)	251	263	310	ns
1980	56	65	93	ns
1989	23	32	17	<0.01
n	182	196	122	
% change	-59	-51	-82	

The colonies visited in early, mid- or late season were matched samples with similar altitudes, distances from the sea and 1980 tern populations. *P* refers to Kruskal-Wallis one-way analyses of variance.

the figure that had been present in 1980. Because the colonies visited in each part of the season were well-matched this late-season reduction must mean that numbers dropped throughout Shetland at this time, and it is likely that this was due to failed breeders leaving their colonies, and perhaps leaving Shetland completely.

Orcadian Arctic Tern numbers had declined by around 42% since 1980; again the estimates from the beginning and middle of the season are very similar. At the end of the breeding season numbers increased. The simultaneous drop in numbers in Shetland suggests that at least part of this increase is due to birds from Shetland passing through Orkney.

When the data from the whole season are pooled (Table II) areas of north and south Orkney show increases in numbers of terns whereas throughout Shetland, except for North Mainland, numbers declined.

In both Orkney and Shetland, most colonies changed in size between 1980 and 1989 (Table III). On both Orkney and Shetland some very large colonies (>500 birds) became extinct, and in Orkney, but not Shetland, some very large colonies were formed. Without data from intervening years these data are hard to interpret but they do illustrate the volatility of tern numbers between years.

## DISCUSSION

The numbers of Arctic Terns attending colonies in Orkney and Shetland declined between 1980 and 1989. Surveys alone cannot explain why this happened. Four possibilities exist. First, the birds may have moved elsewhere to breed. However, influxes of Arctic Terns have not been recorded

TABLE II. TOTALS OF ADULT ARCTIC TERNS RECORDED AT COLONIES ON DIFFERENT ISLANDS IN 1980 AND 1989.

a) Orkney	1980	1989	%change
Papa Westray	10678	6183	-40
Westray	3218	4447	+38
North Ronaldsay	1441	1818	+26
Sanday	4594	1626	-65
Eday	962	698	-27
Rousay	6663	1884	-72
Stronsay	3019	1611	-47
Shapinsay	231	26	-88
Mainland	1990	1172	-41
Hoy, Graemsay	1492	121	-91
Walls, Flotta	2117	2668	+26
South Ronaldsay	5966	7460	+25
Total	42371	29714	
b) Shetland			
Unst	1757	584	-66
Yell	6387	2828	-56
Fetlar	3451	235	-93
North Mainland	1544	3727	+141
Whalsay	3863	709	-82
P Stour	5932	514	-91
West Mainland	2463	1469	-40
East Mainland	2628	2063	-21
South Mainland	6478	2060	-68
Foula	6300	10	-99
Fair Isle	4	542	>>+100
Total	40807	14741	

elsewhere in Britain or Ireland (Walsh *et al.* 1990, 1991, Lloyd *et al.* 1991) and we are unaware of any data which could throw light on this question. Second, the incidence of non-breeding may have increased so that the apparent reduction in numbers is due to birds not attending their colonies. This possibility is difficult to test but seems quite likely to be at least a partial explanation for the decline in numbers. Third, the population may have declined due to lack of recruitment. Is the decline in Arctic Tern numbers as expected? Shetland Arctic Terns have suffered almost complete breeding failure in recent years. With no recruitment the breeding population would decline at the rate of adult mortality. Arctic Terns usually breed first at four years of age (Cramp 1985) and so between 1984 when breeding failures were first detected, and the year of our survey, there were only two years (1988 and 1989) when the lack of recruits would have fed back into the breeding population. Since adult annual mortality is estimated to be around 14% (Coulson & Horobin 1976) this means that if no chicks had been reared in Orkney and Shetland since 1984

TABLE III. CHANGES IN SIZES OF 800 ARCTIC TERN COLONIES IN ORKNEY AND SHETLAND BETWEEN 1980 AND 1989.

If all colonies stayed the same size then only the top left/bottom right diagonals of the tables would be filled. Colonies occurring below these diagonals are ones which declined, those above the diagonal increased in numbers.

a) Orkney		1989						
	0	<50	<100	<200	<500	>500	total	
1980	0	—	65	12	5	2	85	
	<50	59	<u>52</u>	9	5	2	130	
	<100	11	14	<u>3</u>	2	0	31	
	<200	7	9	4	<u>4</u>	0	24	
	<500	0	5	3	4	<u>2</u>	16	
	>500	4	1	2	2	0	<u>5</u> 14	
	total	81	146	33	22	6	12 300	

b) Shetland		1989						
	0	<50	<100	<200	<500	>500	total	
1980	0	-	92	11	5	3	111	
	<50	94	<u>121</u>	15	11	0	241	
	<100	30	31	<u>6</u>	4	0	71	
	<200	16	15	2	<u>3</u>	0	36	
	<500	11	8	3	5	<u>1</u>	28	
	>500	4	8	1	0	0	<u>0</u> 13	
	total	155	275	38	28	4	0 500	

then both populations should have declined by about 28%. Both populations have declined by much more than this amount (and breeding failure is not thought to have occurred this severely in Orkney). Thus the size of the apparent reduction in the breeding population is much bigger than that for which we can account simply by the recent breeding failures in Shetland. These calculations suggest that the decline in numbers is too great to be explained solely by the known breeding failures in Shetland since 1984. This may mean that low breeding success has occurred over a much longer period than has previously been realised, and/or that it has occurred over much greater geographical areas than has been appreciated. In the absence of more comprehensive monitoring of reproductive success it is not possible to say whether nesting success in Orkney in recent years has been good or poor; casual observations show only that nesting success has not been as disastrous as it has in Shetland. A fourth possibility is that adult mortality has increased in recent years and compounded the effects of recent breeding failure. Monaghan *et al.* (1989, 1991) showed that the weights of breeding Arctic Terns in Shetland were lighter than those in Orkney or Coquet Island where food shortage was not a problem, and Avery *et al.* (1992) found that in 1990 Arctic Terns were breeding at exceptionally low body-weights in Shetland. Light weight might indicate poor adult condition and an increased risk of death. All these factors, emigration, non-breeding, breeding failure and increased adult mortality, may have played a part in the decline in Arctic Tern numbers in Orkney and Shetland.

Before 1980 there are no wholly comparable and complete counts of Arctic Terns in Shetland. Some of the counts of Arctic Terns which contribute to the (1967-70) Operation Seafarer survey



(Cramp *et al.* 1974) were carried out too late in the breeding season for them to be used reliably to indicate population levels in the late 1960s (Avery 1991). However, the Operation Seafarer counts for Orkney do appear to be comparable with the 1980 survey (Bullock & Gomersall 1981) and with this study (Avery 1991). Numbers of Arctic Terns in Orkney in 1980 and 1969 appear to have been very similar (Avery 1991) and therefore the decline which we have found between 1980 and 1989 should be taken seriously.

Monaghan *et al.* (1989, 1991) found that Arctic Tern chicks in Shetland starved to death because of a lack of sandeels *Ammodytes marinus*. The cause of the lack of sandeels is unknown and is the subject of some controversy (Lloyd *et al.* 1991, Avery & Green 1989, 1990, Avery 1990ab, 1991). A lack of food is likely to underly the change in status of Arctic Terns, whatever the actual detail of the mechanism.

Why have Arctic Terns declined in Orkney as well as Shetland? First, little detailed information exists on nesting success of Arctic Terns in Orkney in the years 1980-89; we simply know that there have not been the same widespread dramatic failures that have occurred in Shetland and that some colonies have apparently been successful in some years. It may be that breeding success has been too low to maintain population levels in Orkney too. Second, the degree of mixing of the Orkney and Shetland breeding birds is poorly known. It is not, therefore, known to what extent a decline in breeding success in one area could affect the other. Obviously, to take an unlikely example, if Orkney and Shetland together formed a closed population, and there was random mixing of birds each year, then both populations would decline at the same rate, even if the cause of the decline were restricted to one island group, or one small part of one island group. Detailed studies of natal and breeding dispersal would be needed to evaluate the extent to which the decline in the Orkney population could be due to events occurring in Shetland.

Further work is necessary to investigate the breeding success of Arctic Terns throughout Orkney and Shetland. Ringing studies of nesting Arctic Terns should be carried out to establish the degree of mixing of the Orkney and Shetland populations and whether there are differences in adult survival between Orkney and Shetland.

#### ACKNOWLEDGEMENTS

We are grateful to staff of the former Nature Conservancy Council in Orkney and Shetland (particularly Dr Mike Richardson), RSPB wardens and many amateur ornithologists for numerous types of assistance during this survey. This work was part-funded by Shell-UK. We thank the many inhabitants of Orkney and Shetland who have allowed us access to their tern colonies. David Parkin, Jane Sears, Innes Sim, Adrian del Nevo and Eric Meek commented on a draft of this paper and the referees' comments improved this paper further.

#### SUMMARY

A survey of Arctic Terns in 1989 indicated that numbers had fallen by 55% in Shetland and 42% in Orkney since a comparable survey was carried out in 1980. Recent food shortages are likely to have caused these declines but simply from survey data it is impossible to quantify the contributions of emigration, non-breeding, lack of recruitment and increased adult mortality to the decline in breeding numbers.

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# Body weights of incubating Arctic Terns *Sterna paradisaea* on Orkney and Shetland in 1990 and 1991

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## INTRODUCTION

Arctic Terns have experienced very poor breeding success in Shetland since 1984 (Heubeck 1989, Monaghan *et al.* 1989a), and in 1990 only two chicks were known to have fledged in Shetland (Walsh *et al.* 1991). In Orkney the situation is less clear but some colonies have shown large scale failures in recent years. On Papa Westray only one chick was known to fledge from a colony of 7200 adults in 1990 (Booth *et al.* 1991). On Shetland, the major cause of low breeding success has been a lack of the main prey, sandeels *Ammodytes marinus* (Monaghan *et al.* 1989b). However, there are other factors which may also lower breeding success, such as predation by Herring Gulls *Larus argentatus*, Great Black-backed Gulls *Larus marinus*, Arctic Skuas *Stercorarius parasiticus*, Great Skuas *Catharacta skua*, and hedgehogs *Erinaceus europaeus*, and bad weather conditions, especially at hatching time. These factors may also be important in lowering breeding success on Orkney (Uttley, Dodman & Burns in press.)

In 1990 Arctic Terns on Shetland were found incubating at very low body weights (Avery *et al.* 1992), and breeding success was virtually nil (Walsh *et al.* 1991). The aim in 1991 was to test whether body weight would differ between Orkney (little evidence of food shortage) and Shetland (substantial evidence of food shortage), as previously found by Monaghan *et al.* (1992). Previous work was carried out at a very small number of colonies (Monaghan *et al.* 1992, Avery *et al.* 1992.). In this study several colonies were sampled on both Orkney and Shetland in 1991, to assess whether inter-colony variation occurred.

## METHODS

Birds were caught at the nest using funnel walk-in traps. All were trapped at least two weeks into incubation. All birds were measured by D.S. on Shetland and I.S. on Orkney. On capture maximum wing chord length (to the nearest 1mm) and weight (to the nearest 1g) were recorded, and the birds were ringed. At each nest clutch size, egg length, breadth (to the nearest 0.1mm) and weight (to the nearest 0.1g) were noted. For each egg the number of days to hatching was estimated using a relationship between density and days to hatching (Avery *et al.* 1992). At most nests only one adult was trapped but at two nests (one each on Orkney and Shetland) both were caught. Incubating adults, marked with indelible pen, were seen back on some study nests within 5 minutes of release and subsequent visits within a few days of trapping revealed birds sitting at most study nests. Several clutches belonging to trapped birds were known to have successfully hatched on Shetland in 1991. Hence trapping did not cause desertions.

## STUDY AREA

Nineteen widely-scattered colonies on Orkney and Shetland were sampled (Fig. 1). One colony on Shetland (1 - Nesbister) was studied in both 1990 and 1991. Supplementary data on egg volumes and clutch sizes were obtained from an additional 23 colonies in 1990 and 25 colonies in 1991.

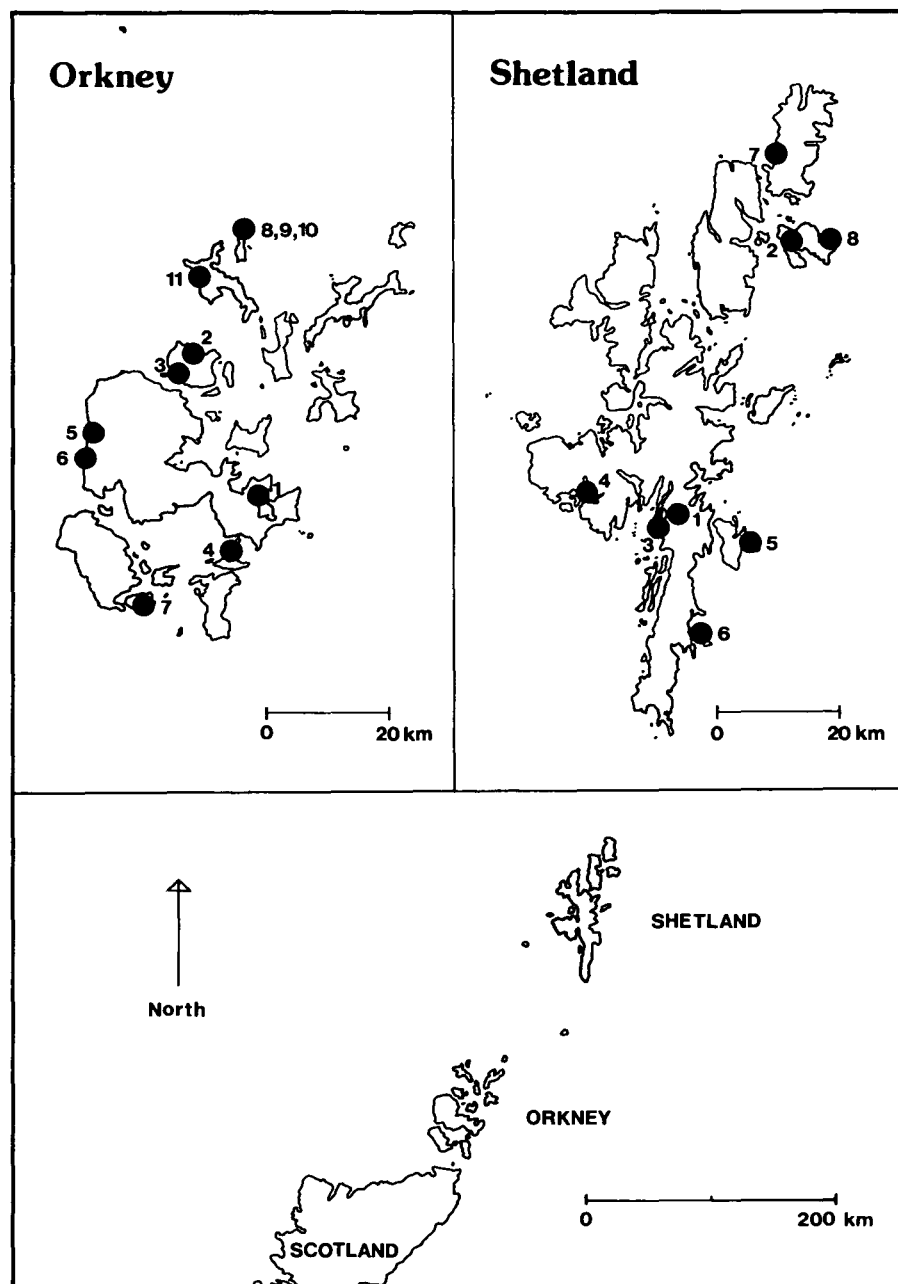


Figure 1. Location of study colonies on Orkney and Shetland in 1990 and 1991.

## RESULTS

*Weights of incubating adults*

Mean body weight of incubating Arctic Terns did not differ significantly between colonies within island groups (Shetland 1990: ANOVA  $F_{1,12} = 4.65$ ,  $p > 0.05$ ; Shetland 1991: ANOVA  $F_{6,71} = 2.00$ ,  $p > 0.05$ ; Orkney 1991: ANOVA  $F_{10,49} = 0.91$ ,  $p > 0.05$ ). The sample sizes on Shetland in 1990 were too small for meaningful conclusions on inter-colony differences to be drawn. For the 1991 data we calculated the minimum differences in mean body weight which we could have been 80% certain to have detected at a significance level of 0.05 with our sample sizes (Sokal & Rohlf 1981). These were 8% for Shetland and 14% for Orkney. Thus for both Orkney and Shetland our sample sizes were large enough to detect small differences in mean body weight. Birds caught in Shetland 1991 were significantly heavier than those of both Shetland 1990 and Orkney 1991 ( $t = 8.10$ ,  $df = 90$ ,  $p < 0.001$ ;  $t = -4.64$ ,  $df = 136$ ,  $p < 0.001$ ) (Table I).

TABLE I. MEAN BODY WEIGHTS OF INCUBATING ARCTIC TERNS ON ORKNEY AND SHETLAND IN 1990 AND 1991.

<i>Island Group</i>	<i>Year</i>	<i>Colony</i>	<i>Body weight (g) Mean <math>\pm</math> SE</i>	<i>n</i>
Shetland	1990	1	99.00 $\pm$ 2.10	13
		2	82.00 $\pm$ 0.00	1
		Overall	97.79 $\pm$ 2.30	14
Shetland	1991	1	110.30 $\pm$ 1.98	10
		3	108.86 $\pm$ 2.70	7
		4	114.62 $\pm$ 2.27	8
		5	117.15 $\pm$ 1.25	13
		6	114.73 $\pm$ 1.91	15
		7	115.11 $\pm$ 2.03	9
		8	113.94 $\pm$ 1.50	16
		Overall	113.91 $\pm$ 0.74	78
Orkney	1991	1	105.60 $\pm$ 1.56	5
		2	108.00 $\pm$ 2.97	5
		3	107.80 $\pm$ 3.65	5
		4	105.83 $\pm$ 3.58	6
		5	113.00 $\pm$ 5.55	5
		6	114.40 $\pm$ 5.49	5
		7	110.56 $\pm$ 1.86	9
		8	106.20 $\pm$ 3.06	5
		9	104.40 $\pm$ 0.61	5
		10	105.60 $\pm$ 1.43	5
		11	110.00 $\pm$ 3.31	5
		Overall	108.42 $\pm$ 1.01	60

*Other parameters*

Certain parameters which may have influenced body weight were examined, and we tested for differences between island groups (Table II).

TABLE II. PARAMETERS POTENTIALLY AFFECTING ARCTIC TERN BODY WEIGHTS ON ORKNEY AND SHETLAND IN 1990 AND 1991.

<i>Parameter</i>	<i>Shetland 1990</i>	<i>n</i>	<i>Shetland 1991</i>	<i>n</i>	<i>Orkney 1991</i>	<i>n</i>
Wing length (mm)	275.71 ± 1.66	14	276.50 ± 0.70	78	275.00 ± 0.82	60
Date of capture	51.07 ± 1.11	14	50.00 ± 0.16	78	51.25 ± 0.43	60
Clutch size	1.47 ± 0.02	419	1.98 ± 0.02	375	1.97 ± 0.03	210
Estimated days to hatch	8.21 ± 2.48	14	5.57 ± 0.67	77	6.54 ± 1.40	59
Egg volume (mm <sup>3</sup> )	33589 ± 106	419	34877 ± 102	375	34754 ± 128	210

Figures given are means ± SE.

Date of capture is day of season (Day 1 = 1st May).

Egg volume = length x breadth<sup>2</sup>

#### *Relationship between body weight and wing length*

Wing length did not vary significantly either within or between island groups, or between years. No significant correlation was found between body weight and wing length in either island group in either year. Combined data showed that body weight was significantly correlated with wing length, although only a very small proportion of the variance in body weight was explained (weight = 44.03 + [0.24 wing],  $r = 0.02$ ,  $p < 0.05$ ). Thus body size (as indicated by wing length) did not have a major influence on body weights of incubating Arctic Terns on Orkney and Shetland in 1990 and 1991, and does not explain the differences between them found above.

#### *Variations in date of capture, egg volume and days to hatching*

Date of capture was significantly later on Orkney than on Shetland in 1991 (Kruskal-Wallis analysis of variance = 13.51,  $df = 1$ ,  $p < 0.001$ ), but did not vary significantly between years on Shetland. In addition, egg volume was significantly larger on Shetland in 1991 than in 1990 (Kruskal-Wallis analysis of variance = 799.37,  $df = 1$ ,  $p < 0.001$ ), but did not vary significantly between Orkney and Shetland in 1991. Estimated days to hatch of clutches did not vary significantly either between island groups or between years.

#### *Distribution of clutch sizes*

On Shetland the proportion of different clutch sizes varied significantly between years, many more 2- and 3-egg clutches being laid in 1991 ( $X^2 = 167.96$ ,  $df = 2$ ,  $p < 0.001$ ). However, in 1991 there was no significant difference in the proportion of different clutch sizes between Orkney and Shetland ( $X^2 = 0.46$ ,  $df = 2$ ,  $p > 0.05$ ) (Table III).

TABLE III. THE PERCENTAGES OF DIFFERENT CLUTCH SIZES ON ORKNEY AND SHETLAND IN 1990 AND 1991.

	<i>Clutch size</i>			
	<i>1</i>	<i>2</i>	<i>3</i>	<i>n</i>
Shetland 1990	54	45	1	419
Shetland 1991	12	78	10	375
Orkney 1991	14	76	10	210

## DISCUSSION

The apparent dramatic increase in sandeel numbers in Shetland waters in 1991 (Wright & Bailey 1991), allowed us (fortuitously) to compare Arctic Tern body weights in years of good and bad food supply. Body weight, clutch size, egg volume and breeding success (Walsh *et al.* 1992) were all higher on Shetland in 1991 than in 1990. The 1990 data should be treated with caution since the sample size was low, and 13 of the 14 birds were trapped at the same colony. In 1991 sample sizes at individual colonies on Orkney and Shetland were low but variation in mean body weights was small. For example, on Orkney mean body weights at 9 of the 11 colonies lay between 104.4 and 110.6g. Our finding that body weight did not differ significantly between colonies on Shetland in 1991 supports the results of Monaghan *et al.* (1992), based on single-colony comparisons between Northumberland, Orkney and Shetland, that body weight is lower at 'poor' sites. The mean weight of birds trapped at colony 1 on Shetland (Nesbister) in 1990 was 99.0g but in 1991 it was 110.3g, an increase of 11.4 per cent. This includes a single bird trapped on June 18th in both years which weighed 101g in 1990 and 108g in 1991.

Previous work has shown that Arctic Terns lay smaller clutches but maintain normal egg volume under conditions of food shortage (Monaghan *et al.* 1991). Uttley (1991, 1992) showed that males increased courtship feeding effort when the food supply was poor, allowing females to lay clutches of normal size and quality. Nisbet (1977) found that clutch and egg size of Common Terns *Sterna hirundo* was related to courtship feeding rates, larger eggs and clutches being laid when courtship feeding rates were high. Both clutch size and egg volume were significantly smaller in 1990 than in 1991 on Shetland. Perhaps food shortage was so acute in 1990 that, despite the increased courtship feeding, males were unable to provision females to an extent where they could produce clutches of normal size or even eggs of normal volume. Some Arctic Terns found incubating at exceptionally low body weights on Shetland in 1990 (Avery *et al.* 1992) suggested that food was indeed extremely scarce in that year.

The finding that adult body weight was lower on Orkney than on Shetland in 1991 was unexpected. The differences were not due to variations in body size, as indicated by wing length. Date of capture was significantly later on Orkney than on Shetland in 1991 but the difference was only 1.25 days on average. In Shetland in 1988 Monaghan *et al.* (1992) found that the maximum rate of weight loss of Arctic Terns during incubation was about 1.4g/day. Assuming a similar rate of weight loss on Orkney in 1991, Arctic Terns would be expected to have lost only 1.75g in 1.25 days, an amount insufficient to explain the difference in body weights between Orkney and Shetland birds. The body weights recorded on Orkney were similar to those found on Shetland in 1987 and 1988 (109.8g and 104.3g respectively), years of food shortage and poor breeding success (Monaghan *et al.* 1992). However, neither clutch size nor egg volume was significantly lower on Orkney than on Shetland in 1991, and clutch size was higher on Orkney (1.97) than on Shetland in 1987 (1.90) and 1988 (1.76). These facts suggest that, if food shortage was causing low body weights on Orkney, it was insufficient to cause clutch depletion.

The body weights recorded on Orkney in 1991 were much lower than those found by Monaghan *et al.* (1991) there in 1988 and 1989. In contrast, Shetland birds were much heavier in 1991 than in 1987, 1988 and 1990. Monaghan *et al.* (1991) showed that Arctic Terns on Shetland were dependent on sandeels to maintain good body condition through incubation. Thus, in years of low sandeel availability (1987, 1988 and 1990), body weights were low but when sandeels were abundant (1991) body weights were higher. In 1988 and 1989, Arctic Terns on Orkney and Coquet were able to exploit alternative high quality food sources, such as rockling and clupeids, and were thus able to maintain better body condition than Shetland birds (Monaghan *et al.* 1991). The relatively low body weights found on Orkney in 1991 may indicate that there was a lack of not only sandeels, but also alternative high-quality prey.

Monitoring of breeding numbers, success and adult body weights of Arctic Terns on Orkney and Shetland will continue in 1992 to assess future trends.

## ACKNOWLEDGEMENTS

We would like to thank Ian Rendall and Craig Whyte (Orkney) and Pete Ellis (Shetland) for help with nest trapping. Eric Meek (Orkney) and Pete Ellis (Shetland) provided useful advice, and Pat Monaghan, John Uttley and Jane Sears helped improve an earlier draft of this paper.

## SUMMARY

Incubating Arctic Terns were heavier on Shetland in 1991 than in either Shetland in 1990 or Orkney in 1991. There were no significant differences in mean body weight between colonies within island groups. Variations in body weight between Orkney and Shetland were not due to body size differences, or to different dates of capture. Arctic Terns on Shetland laid significantly larger eggs and clutches in 1991 than in 1990. Although Orkney birds were significantly lighter than Shetland birds in 1991, egg volume and clutch size remained similar. These results are discussed with reference to breeding success and food availability.

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# The status of Roseate Terns *Sterna dougallii* and Common Terns *Sterna hirundo* in the Azores

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P. Akers, M.I. Avery and L. Monteiro

## INTRODUCTION

The Roseate Tern *Sterna dougallii* breeds in discrete, relatively small colonies in many parts of the world's oceans and is considered to be possibly threatened with extinction (Collar & Andrew 1988). The Atlantic race *S. d. dougallii* nests in Canada (where it is listed as threatened, Kirkham & Nettleship 1987), northeast USA (where it is listed as endangered, Andrews *et al.* 1988) and Florida and the West Indies (listed as threatened, Nisbet 1980). In the eastern Atlantic it nests within the European Community in Ireland, U.K. and France (Avery & Winder 1990; Avery 1991a). Small numbers of this race also nest in the Canaries (Grimmet & Jones 1989) and South Africa (Randall & Randall 1979). Bannerman & Bannerman (1966) commented that the Azores were likely to hold important populations of this species. In 1980, Le Grand estimated the Azorean Roseate Tern population at more than 300 pairs and suggested that Common Terns were no longer found in large colonies, particularly in the central and eastern islands (Le Grand *et al.* 1984). Here we report the results of two surveys of Roseate Terns and Common Terns *S. hirundo* in the Azores. The nine main islands of the Azores are of volcanic origin and lie on the mid-Atlantic ridge between latitudes 36° 55' N and 39° 43' N and longitudes 24° 26' W and 31° 16' W (Fig. 1). The archipelago is situated c. 1,500 km west of Lisbon.

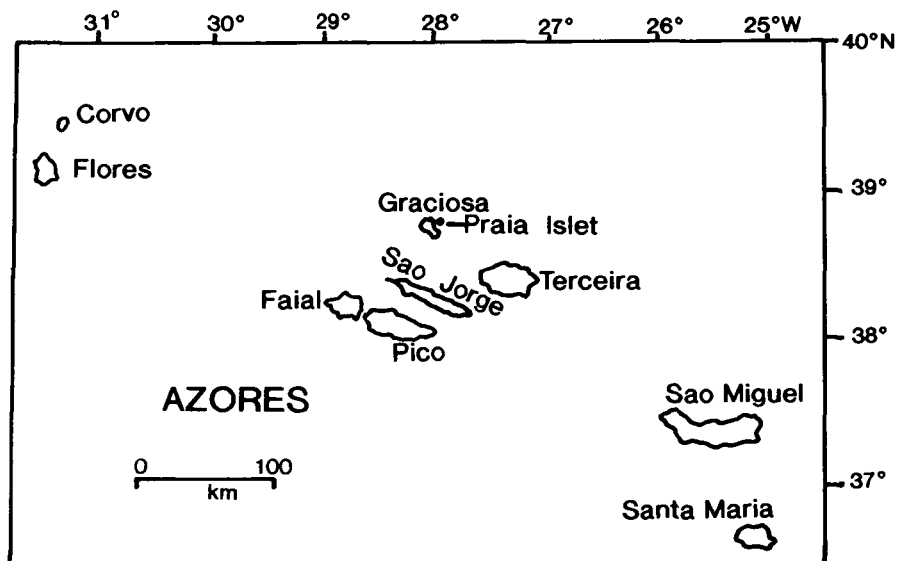


Fig. 1. The Azores archipelago.

## METHODS

### *Timing*

Terns were surveyed in 1984 (late May-early July) and 1989 (mid-May to late July). Colonies were located and surveyed using a mixture of land-based and sea-based transport. The 1984 survey started on the easternmost island of Santa Maria and proceeded westwards to São Miguel (where counts were made by G. Le Grand in the years just prior to the survey), part of the central group (Terceira and Graciosa) and then to the western group (Flores and Corvo), finally returning to the central group (Faial, Pico, and São Jorge). In 1989 Flores was visited at the beginning of the survey. Groups of observers then visited the eastern and central groups. In 1989 all islands, except Terceira and São Jorge, were revisited on several occasions and changes in tern numbers were monitored through the breeding season. During 1989 population estimates were taken from nest or flush counts recorded during the peak of laying (middle 80%, Gochfeld 1980) on each island.

### *Counting*

During both surveys observers attempted to record the number of clutches and/or broods of young to obtain the number of breeding pairs. Colonies were defined as five or more pairs of either species separated from other colonies by relatively wide areas of cliff or sea (100+ metres). While close circumnavigation and onshore searches ensured that the number of colonies overlooked was negligible (and none of significant size were missed) it was not possible (mainly through difficulty of access) to set foot in every colony. A 'colony visit' took place when a direct ground search was made to count eggs and young. Colony visits took place at 41% (26/63) of the colonies discovered in 1984 and 70% (75/107) of colonies in 1989. The relatively short survey period in 1984 allowed only one visit per colony and numbers are probably an underestimate as they do not include failed pairs or birds yet to lay. In 1989 a number of return visits were possible at most colonies.

In the absence of a colony visit, terns were flushed into the air (usually by sounding a horn just off-shore), individual birds were counted several times and the counts were averaged. In almost all cases it was possible to distinguish Roseate and Common Terns but on a few occasions proportions of each species were estimated. An assessment of the relationship between colony attendance and each of the following parameters: breeding status, stage of breeding, time of day, stage of season, and variation in species composition and subsequent population (breeding pairs) is presented elsewhere (del Nevo *et al.* in prep.) from which a correction factor of three flying birds to two breeding pairs was established. This gives a slight underestimate of the number of breeding pairs and thus totals should be considered as minima. An identical correction factor was obtained for Arctic Terns *Sterna paradisaea* and Common Terns in Shetland, U.K. (Bullock & Gomersall 1981). We present here the number of breeding pairs for each species on each of the nine islands. In retrospect it is clear that Santa Maria was probably surveyed too early in 1984 (before many birds arrived) and Flores may have been surveyed too late (after some birds finished breeding or failed). In 1989, with more time, personnel and initial knowledge, most islands were visited several times through the season. In order to compare the 1984 survey with the more comprehensive coverage in 1989 (when failed or late layers were less likely to be missed), we calculated separate population estimates for 1989, one based on visits during a comparable time period in 1984 (the single visit in 1984 compared with a first estimate in 1989), the other based on regular monitoring during several colony visits (a 1989 final estimate based on multiple visits). Since our population estimates are derived from complete surveys it is impossible to calculate confidence intervals for them; there is no sampling error. The sources of error in our estimates are thus threefold. First, we may have overlooked some colonies; we think this is unlikely. Second, some birds may have moved colonies within the season and been counted more than once; this is possible but difficult to assess. Third, our counts may be inaccurate or biased. In retrospect we should have designed our work to assess this problem but in practice we believe it to be a minor one.

### Coverage

In 1984 the whole coastline of the Azores was covered. In 1989 a small part (10km) of the northern coast of São Jorge was not surveyed due to adverse sea conditions. There was no evidence from either the 1984 survey or local fishermen that terns nested in this area and its omission is unlikely to have seriously affected the figures for either of the two tern species.

## RESULTS

### General

In 1984, 63 tern colonies were found; 1 held Roseates alone, 34 held only Common Terns and 28 contained both species. In 1989, there were no colonies containing only Roseate Terns, 87 colonies were solely occupied by Common Terns and twenty colonies contained both species (Table I).

There are few sandy beaches on the Azores and in 1989 most tern colonies were on mainland cliffs (37%) or steep-sided sea stacks (27%) with remaining sites roughly equally distributed between broken cliff/boulders, low lying islets, and relatively low land spurs (Ramos and del Nevo pers. obs.). Only Common Terns nest inland, mostly in crater lakes up to 5km from the sea.

TABLE I. NUMBERS OF COLONIES OF ROSEATE TERN, COMMON TERN AND MIXED SPECIES IN THE AZORES DURING 1984 AND 1989.

Island	Number of Roseate Tern colonies		Number of Common Tern colonies		Number of mixed Tern colonies	
	1984	1989	1984	1989	1984	1989
São Miguel	0	0	4	0	1	0
Santa Maria	0	0	3	6	3	5
Terceira	0	0	1	5	3	2
São Jorge	0	0	3	7	2	1
Pico	0	0	4	9	1	1
Graciosa	0	0	0	9	4	3
Faial	0	0	4	8	1	0
Corvo	0	0	3	9	0	0
Flores	1	0	12	34	13	8
Total	1	0	34	87	28	20

Note: 1. Total number of colonies (single species and mixed species combined) during 1984 and 1989 was 63 and 107 respectively.

### Roseate Tern

Roseate Terns bred on eight of the nine main islands in 1984 and on six in 1989 (Table I). Corvo lacked breeding Roseate Terns in both years. In 1984 colonies of 30 and 70 pairs were found on São Miguel and Faial respectively, but none were found on either island in 1989. In both years the largest totals, and largest individual colonies, were on Flores. Flores, Graciosa and Santa Maria between them held 445 pairs (69%) of the total in 1984 and 871 pairs (88%) in 1989.

An estimated 992 pairs of Roseate Terns were found in 1989 compared to 642 pairs in 1984 (Table II). This apparent large increase in numbers was probably not real. Roseate and Common Terns breeding on Flores and Corvo lay earlier than terns on the central group and markedly earlier

TABLE II. COMPARISON OF THE NUMBER OF BREEDING PAIRS OF ROSEATE TERNS FOR EACH ISLAND ON THE AZORES DURING 1984 AND 1989.

<i>Island</i>	<i>1984 Estimate<sup>2</sup></i>	<i>1989 first estimate<sup>3</sup></i>	<i>% difference 1984-89</i>	<i>1989 final estimate<sup>4</sup></i>	<i>% Europe</i>	<i>% difference between 1984 &amp; 1989</i>
Santa Maria	70	40	-43	116	7.7	+190
São Miguel	30	0	-100	0	0	0
Graciosa	112	265	+137	275	18.2	+3.8
Faial	70	0	-100	0	0	0
São Jorge	35	5	-86	5	0.3	0
Pico	20	19	-5	23	1.5	+21.1
Terceira	42	93	+120	93	6.1	0
Corvo	0	0	0	0	0	0
Flores	263	314	+19	480	31.7	+52.9
Total	642	736	+15	992	100	+34.8

Notes: 1. European population estimated at c.1500 prs. (Avery 1991a)

2. 1984 figure based on one population estimate per colony.

3. 1989 first estimate of the number of breeding pairs based on one visit to a colony during a comparable time period (i.e. within 10 days) of the 1984 observation date.

4. 1989 final estimate based on multiple visits from which a complete picture of the number of pairs (including failed and late birds) was obtained.

5. The percentage difference between the first and final estimate of the number of breeding pairs in 1989 was used to re-calculate the 1984 Roseate Tern breeding population, see text.

than those breeding on Santa Maria (Dunn 1989); the 1984 survey progressed westwards from Santa Maria to Corvo, this produced a minor bias which is likely to have underestimated numbers at both ends of the archipelago compared with the 1989 survey (see methods for further explanation).

For visits of comparable dates the change in Roseate Tern numbers between 1984 and 1989 was an increase of c. 15%. This is probably close to the accuracy of the counts themselves. However, it does suggest that Roseate Tern numbers in the Azores are either increasing or stable. The final population estimate (i.e. based on several colony visits) for Roseate Terns in 1989 is c. 35% higher than the first (single visit) estimate. This suggests that the 1984 survey may have underestimated the population by c. 35%. Adjusting this estimate accordingly yields a revised population estimate of c. 867 pairs of Roseate Terns in 1984.

### *Common Tern*

Common Terns numbered 4015 pairs in 1989 compared to 2006 pairs in 1984 (Table III). In both years Common Terns bred on all nine islands. In 1984, Flores held most (444 pairs, 22%) and in 1989 Santa Maria held the most (800 pairs, 20%). São Miguel, São Jorge and Pico have long coastlines but few terns, with 249 pairs (c. 6%), 232 pairs (c. 6%) and 249 pairs (c. 6%) respectively in 1989. The smallest island, Corvo, contained relatively high numbers (260 pairs, 6.5%) in 1989. The proportions of Common Terns on the different islands in 1984 and 1989 were similar (Table III).

TABLE III. COMPARISON OF THE NUMBER OF BREEDING PAIRS OF COMMON TERNS FOR EACH ISLAND ON THE AZORES 1984 AND 1989.

<i>Island</i>	<i>1984 Estimate<sup>2</sup> estimate<sup>3</sup></i>	<i>1989 first 84-89</i>	<i>% difference estimate<sup>4</sup></i>	<i>1989 final 1984 &amp; 1989</i>	<i>% Europe</i>	<i>% difference between</i>
Santa Maria	260	318	+22	800	0.8	+152
São Miguel	100	249	+149	249	0.2	0
Graciosa	231	429	+86	463	0.5	+8
Faial	320	556	+74	576	0.6	+4
São Jorge	230	232	+1.0	232	0.2	0
Pico	122	49	+122	249	0.2	0
Terceira	220	372	+152	418	0.4	+12
Corvo	89	260	+192	260	0.3	0
Flores	444	557	+11	768	0.8	+40
Total	2006	3222	+61	4015	4.0	+25

Notes: 1. European population estimated at c.100,000 prs. (Lloyd *et al.* 1991).

2. 1984 figure based on one population estimate per colony.

3. 1989 first estimate of the number of breeding pairs based on one visit to a colony during a comparable time period (i.e. within 10 days) of the 1984 observation date.

4. 1989 final estimate based on multiple visits from which a complete picture of the number of pairs (including failed and late birds) was obtained.

5. The percentage difference between the first and final estimate of the number of breeding pairs in 1989 was used to re-calculate the 1984 Common Tern breeding population, see text.

Comparing observations from similar dates in 1984 and 1989 suggests a population increase of c. 61% (Table III). The marked increase in numbers of Common Tern colonies between 1984 and 1989 (Table I) could be partly due to unintentional differences in definitions of the boundaries of colonies (often a rather subjective decision) but probably reflects a real increase in the breeding numbers.

We compared the first-visit population estimate in 1989 with a final estimate (based on several visits) in the same year. The final estimate differs from the one-off figure by an average of +25%. If we revise the 1984 population estimate by +25% then a recalculated total comes to c. 2500 pairs.

#### *Colony shifts*

Observations during 1989 and subsequent seasons (A.J. del Nevo 1990-91, pers. obs.) indicate that both within and between-island movement (of equal extent) of individuals and whole colonies takes place between years on the Azores. For example, in 1984, no Roseate Terns were recorded on Praia Islet, Graciosa. However, Bannerman & Bannerman (1966) observed between three hundred and four hundred individuals there, many of which were feeding young at the beginning of July 1963. In May 1988, 118 nests of Roseate Terns were present on the eastern end of Praia islet (M.I. Avery pers. obs.) but none were present in 1989. During 1990, 140 pairs of Roseates Terns and c. 90 pairs of Common Terns bred on Praia Islet (A.J. del Nevo pers. obs.). In the years when Roseate and Common Terns were not breeding on Praia Islet, comparable numbers of birds

TABLE IV: THE NUMBER OF NESTING PAIRS AT THE TEN LARGEST ROSEATE TERN *STERNA DOUGALLII* COLONIES IN 1984, THE NUMBER OF PAIRS AT THE SAME COLONIES IN 1989, AND THE TEN LARGEST COLONIES IN 1989.

1984 Island/ Colony <sup>1</sup>	Rank <sup>2</sup>	1984 nesting pairs	1989 nesting pairs	1989 Island/ Colony <sup>1</sup>	Rank <sup>3</sup>	1989 nesting pairs
Flores 1	1	126	160+	Flores 1	1	160+
Faial 1	2	70	0	Flores 2	2	150
Santa Maria 1	3	56	2	Graciosa 1	3	133
Graciosa 1	4	50	0	Graciosa 2	4	113
Flores 2	4	50	0	Santa Maria 1	5	103
Graciosa 2	6	40	0	Terceira 1	6	90
Terceira 1	7	28	3	Flores 3	7	62
Sao Jorge 1	8	20	0	Flores 4	8	43
Flores 3	8	20	0	Graciosa 3	9	29
Pico 1	8	20	21	Pico 1	10	21

- Notes: 1. Numbers (ranks) indicate the order of size of colonies for each island, e.g. Flores 1, Flores 2, Flores 3 give the three largest colonies for their island in descending order (rank) of magnitude in the year indicated.
2. Rank in descending order of the largest colonies in the Azores during 1984.
3. Rank in descending order of the largest colonies in the Azores during 1989.
4. Colony locations in del Nevo *et al.* 1990.

were recorded on a large islet c. 3km away. Subsequent observations of colour ringed birds has confirmed the movement of adult breeding birds between these two colonies in different years (A.J. del Nevo pers. obs.). An example of the changing status of some colonies is illustrated in Table IV. Here we present the number of nesting pairs at the ten largest Roseate Tern colonies in 1984, the number of nesting pairs at the same colony in 1989 and the ten largest colonies in 1989. Only at one colony (Pico 1) were the number of pairs between the two surveys similar. The increase in nesting pairs at other colonies suggests some inter-colony movement may take place. This is further supported by an observation in 1990 of two (1984) colour ringed birds nesting at a site which was not visited in 1984, in this case a minimum shift distance of 24km.

## DISCUSSION

### *International importance*

This work has established the international importance of the Azores for terns, particularly for Roseate Terns, and confirms the observation by Bannerman & Bannerman (1966) that the archipelago represents a stronghold for this species. The European population (including Azores) of Roseate Terns is estimated at c. 1500 pairs (Walsh *et al.* 1990; RSPB unpublished) using similar methods employed in this study. The Azores therefore contain about two-thirds of the European population, with Flores, Graciosa and Santa Maria holding 32%, 18% and 8% respectively. On the basis of surveys presented here, Roseate Terns did not decline between 1984 and 1989 on the Azores, an encouraging contrast to the serious decline in the rest of Europe (Avery 1991a). It is not known how current Roseate Tern numbers on the Azores compare with those of 20 years ago when the population in the rest of Europe was much higher. Grimmett & Jones (1989) listed several parts

of the Azores as important bird areas for Europe since they contain more than one per-cent of the European breeding population of Roseate Terns. Our surveys show that at least 11 Azorean colonies contained this percentage of the European total between 1984 and 1989.

The European population of Common Terns (including the Azores) in 1984 was estimated at c. 100,000 pairs (Avery 1991b) so the Azores hold c. 4% of the European total.

#### *Future research and recommendations*

In view of the serious decline in the numbers of Roseate Terns in north-west Europe, it is important that they are regularly monitored in the Azores and more discovered about their relationship with those in the north-east Atlantic. The possibility exists that Roseate Terns move regularly between the Azores and the rest of Europe, (to date, there has been one confirmed Azores-bred bird recorded in another European colony) or even accompanying Common Terns to and from America (Burson 1990). Continued ringing and colour-ringing of Roseate Terns in Ireland, UK and the Azores and an understanding of tern population genetics and gene flow might establish the extent to which the Azores birds are isolated from the rest of the *S.d. dougallii* population. Mixing does take place in the winter quarters at least (see below).

On the Azores, as elsewhere in Europe, it is not known whether Roseate Tern population levels are determined mostly by factors acting on the breeding grounds or those outside the breeding season. Sightings in Ghana during December 1985 (Everett *et al.* 1987) and in September-December 1988, 1989 and 1991 of respectively 2, 4 and 4 Roseate Terns ringed as pulli on the Azores in 1984, 1989 and 1991, and 4 adults individually colour-ringed in June 1991 (A.J. del Nevo pers. obs.) indicates that Roseate Terns breeding in northern Europe and those breeding in the Azores share common wintering grounds, or at least that the populations mix briefly during the winter months. Thus Roseate Terns nesting in the Azores and elsewhere in Europe will probably be subjected to the same pressures (for example trapping), during winter.

In order to guide conservation measures in the breeding season it is necessary to identify sites which are of long-term importance for terns on the Azores. It is already clear that numbers of terns at individual colonies may change dramatically from one year to the next. Monitoring dispersion for the whole archipelago forms part of our continuing studies.

We are also investigating the type and scale of potential threats to terns in the Azores. Preliminary observations suggest that introduced predators and human disturbance may have a serious impact on breeding terns at some colonies. At some, the control of rats, cats, goats and polecat-ferrets needs to be addressed. However, since the locations of important colonies appear to change from year to year, even when breeding success was high, it may be premature to recommend site-protection measures until it is established which, if any, sites consistently hold significant numbers of Roseate Terns. It appears that the most appropriate protection measure in the present state of knowledge would be a form of roving warden to carry out predator control, prevention of disturbance, and education work.

#### ACKNOWLEDGEMENTS

We gratefully acknowledge the support, assistance and kindness from the following organisations, institutions and individuals: Gulbenkian Calouste Foundation, International Council for Bird Preservation, British Ornithologists Union, Vincent Wildlife Trust, Fauna and Flora Preservation Society, Percy Sladen Memorial Fund, Russell Trust, Frank Chapman Memorial Fund, Gilchrist Educational Trust, University of St. Andrews, British Ecological Society, Seabird Group, Mr & Mrs Warren, J & L Cunha, M. Gomes, J. Doat, C.M. Perrins, R. Speir, J. Lemlin, Crew of 'Gaia Quest', Dr. Cardigo, J. Cordoso, M. Lajosa, F. Martins, R. Santos, Sr. Weber, R. Cordeiro, A. Lockley, I. Texeira, M. Martins, R. Cunha, T. Charlton, A. Pombo, P. Melo, Sr. Reiss, Camara do Municipal of S.Maria, Flores, Corvo, Faial, and Graciosa, the Capitania do Porto for S.Maria, Faial and Flores, Departamento de Ambiente, M.Paixao e Silva, Forestry Department of the Azores.

## SUMMARY

Two censuses of terns in the Azores archipelago yielded 642 pairs of Roseate Terns in 1984 and 992 pairs in 1989, for Common Terns 2006 pairs and 4015 pairs respectively. The apparent increase in Roseate Terns between 1984 and 1989 may be an artefact of the relatively short observation period in 1984. The 1984 survey took place before some pairs had bred at one island (Santa Maria) and after some had fledged at another (Flores), leading to an underestimate of both Roseate and Common Terns by c. 35% and c. 25% respectively. In both 1984 and 1989, Flores, Graciosa and Santa Maria were the most important islands for Roseate Terns, containing in 1989 c. 870 pairs, which represents c. 58% of the European population. It is only possible at present to describe the general dispersion of important breeding sites, due to the shifting nature of several colonies from year to year. The influence of this factor on possible conservation measures for the Roseate Tern is discussed. Outside the breeding season, there is evidence that the Azores population of Roseate Terns mixes with the rest of the European population, and is therefore subject to similar mortality factors.

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# Cormorants *Phalacrocorax carbo* at cage fish farms in Argyll, western Scotland

D.N. Carss

## INTRODUCTION

Cormorants (*Phalacrocoracidae*) are attracted to extensive aquaculture facilities such as ponds and farm dams, throughout the world. Moerbeek *et al.* (1987) report Common Cormorant *Phalacrocorax carbo* predation at Dutch pond farms holding carp *Cyprinus carpio* and grass carp *Ctenopharyngodon idella*. Barlow & Bock (1984) document predation by the Great Cormorant *P. carbo*, the Little Pied Cormorant *P. melanoleucos* and the Little Black Cormorant *P. sulcirostris* on stocks of native warm-water fishes held in fish dams in New South Wales, while the Double-crested Cormorant *P. auritus* is known to visit farm ponds containing channel catfish *Ictalurus punctatus* in the USA (Avault 1988).

Fish farming in Britain is dominated by intensive aquaculture systems, predominately floating cages holding Atlantic salmon *Salmo salar* and rainbow trout *Oncorhynchus mykiss* at high densities in an enclosed mesh bag supported by a floating collar within a larger body of either fresh or salt water. Cormorants (from here on taken as referring to *P. carbo*) have been recorded visiting such fish cages and attacking stock (Ranson & Beveridge 1983). Furthermore, the waters immediately adjacent to cages may also be attractive to Cormorants as they hold substantial concentrations of wild and escaped fishes (Carss 1990a).

This note gives details of Cormorant predation at two cage farms in Argyll, western Scotland and describes the seasonality, nature and scale of the attacks and the stomach contents of birds killed close to the farms.

The study had three aims: (1) to record Cormorant abundance and age structure within the Argyll study area, (2) to determine the circumstances of any stock losses to Cormorants at the cage farms, (3) to quantify such losses.

## STUDY AREA, SITES AND METHODS

The study was carried out between September 1985 and August 1987 in an area containing the sealochs Creran, Etive, Feochan and Melfort and Loch Awe, a large freshwater loch, Argyll, western Scotland (Carss 1989). Few Cormorants breed in Argyll and Bute, and none breed in the study area (Lloyd *et al.* 1991), therefore the Cormorants I recorded were dispersing from the natal/breeding colonies, over-wintering or passing through, and a few immature birds which remained over the summer.

An index of Cormorant abundance was obtained by transect counts on a standard 70 km route on land which included all the sealochs and several running and standing freshwater bodies. The index was the maximum of two counts made each month between November 1985 and August 1987. The presence of Cormorants on other freshwater bodies was recorded throughout the study, but no counts were made on the River Awe. Birds were categorized on plumage characteristics as immature (i.e. traces of brown plumage, up to and including birds three calendar years old) or adult (older) (Alstrom 1985). The monthly immature:adult ratio of birds was determined from records of all Cormorant sightings in the study area between July 1986 and June 1987.

Cormorant predation was investigated at two cage farms; one was a large rainbow trout farm on Loch Awe, the second a smaller farm rearing both rainbow trout and Atlantic salmon in Loch Etive. Cormorant damage to stock was studied by examining dead fish for characteristic beak marks (Carss 1990b). At the Loch Awe farm, cages were protected by suspending weighted sheets of netting from their walkways for two or three months when staff thought there might be

problems with diving birds. However, for most of the time cages were left unprotected. All dead fish from each cage were removed every week and I examined them for damage at least twice a month between October 1985 and March 1986, and the same period in the following year. A further 31 751 dead trout were examined between these periods.

At the Loch Etive farm, dead fish were examined for damage between October 1986 and April 1987. During this period the opportunity was taken to compare the numbers of damaged fish in two pairs of cages. The first contained *ca.* 4000 (A) and *ca.* 1320 (B) small (190mm - 300mm, *ca.* 200g) salmon, respectively, and data were collected between August 1986 and February 1987. Cage A was enclosed by an underwater anti-predator net of the 'bag type' which protected all four sides and the bottom of the cage net whilst cage B was 'unprotected' having no anti-predator netting. The second pair of cages (C and D) each contained *ca.* 400 rainbow trout (240mm - 300mm, *ca.* 250g) and data were collected during September and October 1986. Neither was protected by underwater netting; the cage net of C was clean whilst that of D was heavily fouled with algal growth.

All Cormorant-damaged fish collected at the Loch Awe farm were examined to quantify the position of their wounds; assigned to one of four positions - operculum, belly, dorsal fin region and tail.

Cormorants shot as a crop protection measure, or which drowned accidentally through entanglement in cage netting were collected. As well as fresh food, stomachs sometimes contained the indigestible remains of previous 'meals'. However, the distinction between recently eaten food and the retained items from previous meals was always clear and only fresh meals were included in the analysis. Fish were identified using a reference collection and Wheeler (1978), and their lengths recorded by direct measurement, comparison with freshly caught specimens or, in the case of salmonids, by measuring the width of caudal vertebrae (Wise 1980).

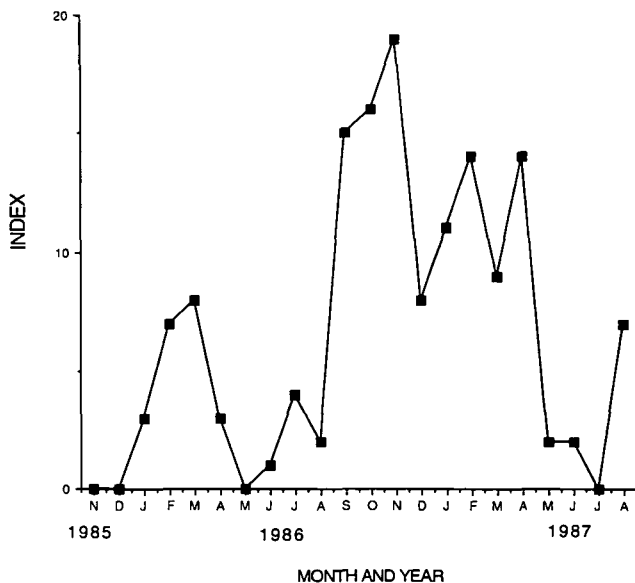


Figure 1. Changes in the index of Cormorant abundance on sealochs in the study area, November 1985 - August 1987.

## RESULTS

*Abundance and age ratios*

During transects, Cormorants were only recorded on the sealochs. However, at other times up to five birds were recorded on Loch Awe in winter and single birds were seen infrequently on other freshwater lochs and rivers.

The number of birds seen on sealochs varied widely between months and years (Fig. 1). More birds were seen between September-August in 1986/87 than in the previous year when no birds were seen in November and December. The peak in the winter of 1985/86 was less than half that of the following year. Small numbers were recorded in June, July and August; these were all immature. Adults began arriving in September and thereafter proportions rose steadily to a peak of 80% in February before falling once again (Fig. 2).

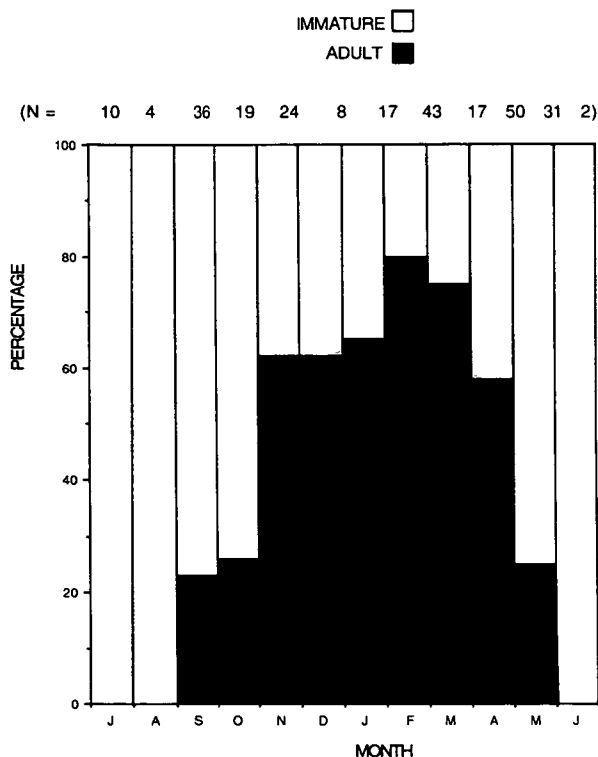


Figure 2. Age classification of all sightings ( $n = 261$ ) of Cormorants on sealochs in the study area, July 1986 - June 1987. Figures in parenthesis are monthly sample sizes.

*Cormorants at farms*

Cormorant visits to both farms were sporadic and, as a result, there were few observations of them foraging there. Nevertheless, some birds were seen diving alongside, and sometimes under, cages and all of the fish they manipulated on the surface were apparently alive.

At no time during the study were holes in nets which could be attributed to Cormorants reported at either farm, nor were there stock losses consistent with such net damage.

*Quantifying Cormorant damage to stock*

Trout showing signs indicating that they had been damaged by Cormorants were recorded at Loch Awe in three of the seven months between September 1985 and March 1986 (Table I). These damaged fish ( $n = 85$ ) ranged from 13 - 35cm and had a mean length ( $\pm$  SE) of 23 cm ( $\pm$  0.5 cm). The highest number was in March when 71 (4.7%) out of 1513 dead fish were damaged. However, over the whole period, only 0.4% of the 22 400 dead fish could be attributed to Cormorants. During the same period the following year, none of the *ca.* 20 000 dead trout examined at this site had been damaged nor had any of the 31 751 trout examined between these periods.

TABLE I THE NUMBER OF DEAD TROUT EXAMINED AT THE LOCH AWE FARM AND THE PROPORTIONS DAMAGED BY CORMORANTS, OCTOBER 1985 - MARCH 1986.

<i>Month</i>	<i>Examined</i>	<i>No.</i>	<i>(%)</i>
		<i>Damaged</i>	
October	7587	0	(0)
November	7649	3	(0.04)
December	1651	11	(0.7)
January	2500	0	(0)
February	1500	0	(0)
March	1513	71	(4.7)
Total	22400	85	(0.4)

At Loch Etive only 104 (0.6%) of the 17 302 dead fish examined bore the marks of Cormorants. Significantly fewer damaged fish were recorded in the cage with an underwater anti-predator net (A) than in an unprotected cage (B) ( $\chi^2 = 14.5$ ,  $df = 1$ ,  $P < 0.001$ ) (Table II). Similar proportions of damaged fish were recorded in cage B and another unprotected cage with a clean net (C) ( $\chi^2 = 0.007$ ,  $df = 1$ , NS). However, significantly more damaged fish were recovered from C than from a cage with a fouled net (D) from which no damaged fish were recorded ( $\chi^2 = 48.8$ ,  $df = 1$ ,  $P < 0.001$ ) (Table II).

TABLE II THE NUMBER OF DEAD FISH EXAMINED (N) AND THE NUMBER (n) AND PERCENTAGE DAMAGED BY CORMORANTS IN FOUR FISH CAGES AT THE LOCH ETIVE FARM BETWEEN AUGUST 1986 - FEBRUARY 1987.

<i>Month</i>	<i>CAGE</i>											
	<i>A</i>			<i>B</i>			<i>C</i>			<i>D</i>		
	<i>Damaged</i>			<i>Damaged</i>			<i>Damaged</i>			<i>Damaged</i>		
	<i>N</i>	<i>n</i>	<i>%</i>	<i>N</i>	<i>n</i>	<i>%</i>	<i>N</i>	<i>n</i>	<i>%</i>	<i>N</i>	<i>n</i>	<i>%</i>
August	0	0	—	0	0	—	0	0	—	0	0	—
September	1	1	100	18	11	61	20	11	55	18	0	0
October	7	7	100	11	10	91	38	35	92	11	0	0
November	3	1	33	36	35	97	0	0	—	0	0	—
December	0	0	—	0	0	—	0	0	—	0	0	—
January	4	0	0	3	0	0	0	0	—	0	0	—
February	9	0	0	2	0	0	0	0	—	0	0	—
Totals	24	9	37.5	70	56	80	58	46	79	29	0	0

### *Type and position of damage*

Eighty-five (75%) of the fish examined had a single deep triangular wound on one side and corresponding lower mandible marks on the other. Wounds were not evenly distributed over the body of the fish but were on the operculum (39.1%) or belly (32.8%) where they often penetrated into the pericardial cavity or damaged internal organs. Other fish were damaged near the dorsal fin (25.0%) or tail (3.1%). The remaining fish had multiple injuries ranging from single wounds on both sides to a fish with five wounds on one side and three on the other.

### *Cormorant stomach contents*

Of the 37 Cormorants collected from the Loch Awe site, 35 had been shot and two had been drowned in the underwater anti-predator netting. Twenty-five (67.6%) of the birds were immature.

Twenty stomachs (54.1%) contained 'fresh' food (Table III); all of the 26 items identified were fish. Salmonids were the most commonly recorded fishes (88.5%) and most of these (52.2%) were rainbow trout. The size of fish ranged from a 33 mm three-spined stickleback to a 315 mm rainbow trout. The identifiable rainbow trout ( $n = 12$ ) ranged from 15 - 31 cm and had a mean length of ( $\pm$  SE) of 21.5cm ( $\pm$  1.4cm). Regardless of the state of digestion, all 'fresh' fish were intact and had obviously been eaten whole.

Six of the Cormorants contained items of fishing tackle; hooks and monofilament line. One bird contained a freshly-eaten rainbow trout with a hook in its mouth attached to 830 mm of line. There was especially high angler pressure in the vicinity of cages at Loch Awe.

TABLE III CONTENTS OF 20 STOMACHS OF CORMORANTS COLLECTED FROM THE LOCH AWE FISH FARM WHICH CONTAINED UNDIGESTED FOOD SEPTEMBER 1985 - AUGUST 1987.

<i>Prey Species</i>	<i>% of Items (n = 26)</i>	<i>% Frequency in Stomachs (n = 20)</i>
Rainbow Trout	46.1	50
Brown Trout	3.8	40
Unidentified salmonid	38.5	40
Three-spined Stickleback	7.7	10
Unidentified fish	3.8	5

## DISCUSSION

Cormorants were seen infrequently at farms during counts and site visits. It was therefore not possible to determine the proportion of the study area 'population' which was associated with the fish farms nor to compare the age ratios of birds in the two habitats. However, attacks by Cormorants were only recorded in the winter (September - March), a time when their numbers in the study area were highest. Dunnet (1986) noted that immature Cormorants arrived to over-winter on the Ythan estuary in Aberdeenshire slightly earlier than did adults. The same was true in the present study, with adults being recorded in small numbers in September and October before a major influx in November.

Fish are regularly graded to maintain size uniformity within each cage and cage net mesh sizes are chosen to prevent the smallest fish from escaping. This also means that Cormorants are unable

to pull even the smallest fish in a cage through the mesh. Recommended mesh sizes for fish of ca. 13-26 cm is 13 mm square and for larger fish, 22 mm square. The fish brought to the surface by Cormorants foraging close to cages could therefore not have been pulled through the mesh of cage nets alive without tearing a hole in the net. Such holes would also have been large enough for stock to escape through. Neither torn nets nor stock losses consistent with such holes were recorded and the main problem caused by Cormorants at cage farms was therefore one of stock damage when the birds poked their beaks through the mesh and gripped the fish.

Damage to fish occurred sporadically throughout the annual period of Cormorant presence but although many fish were sometimes damaged over a short period, the overall losses were low and trivial compared with the losses reported for extensive aquaculture systems (e.g. Barlow & Bock 1984, Moerbeek *et al.* 1987).

Damage was reduced significantly, though not eliminated, by the use of commercially available underwater anti-predator netting. Stock in a protected cage suffered less than half the damage of an unprotected one. Nevertheless, it was clear that there was a need to improve existing underwater anti-predator net technology. In the absence of anti-predator nets, the condition of the cage net also influenced the levels of Cormorant predation; stock in a cage net fouled with algal growth suffered lower levels of predation than that in a cage with a clean net. Presumably, fouling reduced the bird's ability to see fish in the cage or possibly its ability to catch them.

Cormorant wounds on caged fish were similar to those inflicted on wild fish. Van Dobben (1952) found most fish regurgitated by Cormorants had been caught just behind the gills, while Takashima & Niima (1957) recorded that the wounds caused by Temminck's Cormorant *P. attirelis* were also centred behind the gills and speculated that this region was attacked because it contained the vital organs. Most caged fish probably die immediately from their injuries while those with minor wounds may succumb to secondary bacterial or fungal infections. Furthermore, fish with even minor wounds were likely to be unmarketable.

The stomach contents of Cormorants killed at Loch Awe fish farm supported the contention that the birds caught fish outside rather than inside the cages. The majority of prey items were rainbow trout but brown trout and three-spined stickleback were also taken. These species were regularly caught in seine net hauls outside the cages at the Loch Awe farm, 'feral' rainbow trout concentrating there in large numbers (Carss 1990a).

The rainbow trout in stomachs were of a similar size to those found damaged inside cages but they were also similar to those outside the cages (Carss 1990a). Furthermore, these trout had been eaten whole and there was no evidence that they had either been mutilated prior to ingestion or eaten in small pieces, as might be expected had they been extracted through intact cage netting.

The incidence of fishing tackle in Cormorant's guts further suggested that the birds were taking fish from outside the cages, rather than preying upon farmed stock.

#### ACKNOWLEDGEMENTS

I am very grateful to the various fish farms and landowners who allowed me free access to their properties. I am obliged to Mick Marquiss and Iain Taylor for their advice and to Mick and two anonymous referees for their helpful comments on the manuscript. This work was funded by the Natural Environment Research Council.

#### SUMMARY

Most Cormorants in the study area were recorded in the winter, presumably having dispersed from breeding colonies elsewhere. The proportion of adult birds increased from September to February before falling to zero in June. Cormorants apparently did not take fish from farm cages but attacked them through the netting and caused fatal wounds. Some individual attacks appeared severe but the resulting stock losses were small in relation to the overall recorded mortality. Such damage could be reduced, but not eliminated, by the use of appropriate underwater anti-predator netting. Unprotected cage nets fouled with algae appeared to give more protection than clean ones. Most of the fish eaten by Cormorants foraging near farms were wild or escaped fishes that concentrate in the waters immediately adjacent to cages.

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# Racial identity of Cormorants *Phalacrocorax carbo* breeding at the Abberton Reservoir colony, Essex

Robin M. Sellers

## INTRODUCTION

Two forms of the Cormorant *Phalacrocorax carbo* are usually recognised in western Europe, the nominate or North Atlantic race *P.c.carbo*, which breeds in Norway, the British Isles and the north coast of France, and the continental race *P.c.sinensis* (sometimes known as the Southern Cormorant) which breeds in Sweden, Denmark, East and West Germany, Poland, the Netherlands and more recently France (inland) and Italy (Cramp & Simmons 1977, Hansen 1984). The population of the continental subspecies has increased dramatically over the past two decades or so at an average rate of at least 20% pa (Gregersen 1982, Hansen 1984, van Eerden & Zijlstra 1985, Lok & Bakker 1988), whereas that of the nominate race has overall shown a more modest increase (Debout 1987, Lloyd *et al.* 1991, Macdonald 1987, Røv & Strann 1986, Sellers 1991). Within the British Isles there have been increases in Ireland and in SW and NE England but elsewhere, notably in northern Scotland, numbers have decreased.

In 1981 a new Cormorant colony was founded at Abberton Reservoir in Essex, numbers increasing from 9 pairs in this first season (Moore 1981, Ekins 1989) to 356 in 1990 (R.M. Sellers unpublished). Given that the nests are in trees (a habit often considered more typical of *sinensis*), and the relative proximity of the colony to the rapidly expanding Dutch population, it has been suggested that Abberton might represent the colonisation of Britain by the continental race. This paper describes an investigation into the provenance of the Abberton birds based on their plumage, morphology and ringing recoveries.

## METHODS

No simple criteria exist for the racial identification of individual Cormorants in the field. The two European races differ to a certain extent in a number of respects and for the purposes of this investigation information has been collected on the following: (a) gloss to body plumage, greenish in *sinensis* and bluish or bluish-purple in *carbo* (e.g. Witherby *et al.* 1941, Cramp & Simmons 1977, Alström 1985 but *cf.* Stokoe 1958 who questions the validity of the differences). Gloss is difficult to determine in poor light and can vary somewhat depending on the angle from which birds are viewed. To ensure consistent results all observations reported here were made in bright sunshine or strong diffuse light with the observer standing between the bird and the sun; (b) extent of white feathers on the head and (c) size of white thigh patches. Both adult *carbo* and *sinensis* acquire white head feathers (filoplumes) and white thigh patches at the commencement of the breeding season. These feathers are at their brightest and most extensive in the early part of the breeding cycle, and decline in intensity as breeding progresses. In general *sinensis* birds have whiter heads than *carbo* and these white feathers tend to be retained longer, although some *carbo*, notably older males, may attain heads as white as typical *sinensis* (Witherby *et al.* 1941, Stokoe 1958, Cramp & Simmons 1977, Marion 1983, Alström 1985). The presence or absence of white feathers cannot, therefore, be taken as indicative of one race or the other in individual birds, as often assumed. Observations were made mainly during the incubation phase (incubation/feeding young at the Dutch colonies) by which time the intensity of the white head feathers and thigh patches has begun to wane. However because *sinensis* keeps these features longer than *carbo* this is the stage at which the differences are likely to be most pronounced; (d) shape of the gular pouch. Alström (1985) has noted that in *sinensis* the gular pouch extends further back beyond the end of



the gape than it does in *carbo* and comparisons between skins in the British Museum (Natural History) collection suggest that this is an unusually helpful feature for differentiating between the races.

Breeding birds were examined through 8.5 x 44 binoculars and a 20-60x telescope from a range of 50m or less. For comparative purposes data were collected not only at Abberton but also at two *sinensis* colonies in the Netherlands, and at a *carbo* colony at St.Margaret's Island, Dyfed, all within a period of three weeks during the 1989 breeding season (Table I). Further observations were made at Abberton and St.Margaret's Island during the 1990 breeding season and at three other British colonies, two in Wales (Sheep Island and Llanddeiniol, both in Dyfed) and one in Scotland (Ceann Leathad in Caithness).

Information on the origins of Cormorants wintering in Essex is based on ringing recoveries made available by the British Trust for Ornithology, and sightings of colour-ringed birds collected

TABLE I. DATES OF COLONY VISITS, BREEDING STATUS AND WEATHER CONDITIONS.

<i>Date</i>	<i>Colony<sup>a</sup></i>	<i>Breeding Status</i>	<i>Weather Conditions</i>
<i>Main surveys</i>			
27 Apr 89	Oostvaardersplassen	some incubating, others feeding young	bright sun
28 Apr 89	Naardermeer	some incubating, many others feeding young, some well grown	bright sun
14 May 89	Abberton Reservoir	mainly incubating, but <i>ca.</i> 20 feeding young, some well grown	cloudy with bright diffuse light whilst surveying north sub-colony, bright sunlight for south sub-colony
20 May 89	St. Margaret's Island	<i>ca.</i> 90% incubating, remainder feeding small young, except 3 nests with well grown young	mostly bright sun
13 May 90	St. Margaret's Island	<i>ca.</i> 95% incubating, remainder feeding small young, except 1 nest with well grown young	bright sun
20 May 90	Abberton Reservoir	mainly incubating, but <i>ca.</i> 20% feeding young, some well grown	mainly strong diffuse light but periods of sunshine
<i>Other surveys</i>			
16 May 87	Ceann Leathad	incubating	bright sun
24 Jun 89	St. Margaret's Island	feeding young, almost all well grown (suitable for ringing)	bright sun
25 Jun 89	Sheep Island	feeding young, all well grown, about half already fledged	cloudy with bright diffuse light
1 Jun 90	Llanddeiniol	mainly incubating, <i>ca.</i> 10% with well grown chicks	raining, light poor
21 Jun 90	Sheep Island	most nests with large young, some already fledged, <i>ca.</i> 30% incubating or with small young	raining, light poor
23 Jun 90	St. Margaret's Island	most nests with medium sized young, some still with eggs	bright sun

<sup>a</sup> Oostvaardersplassen and Naardermeer birds refer to *P.c.sinensis*, St. Margaret's Island, and all those under Other Surveys to *P.c.carbo*.

by the author. During the 1980s 11 colour-ringing projects have been established in Western Europe, 4 in the British Isles (in respectively Dyfed, the Solway Firth, Co. Dublin and Co. Down) and 7 on the continent (one each in Sweden, Poland, West Germany, East Germany, Denmark, The Netherlands and France). Approximately 2000 birds were colour-ringed in the British Isles during this period compared with about 10,000 on the continent, the majority of them in Denmark.

## RESULTS

### *Plumage gloss*

At both the Dutch colonies all except one bird seen at close range had plumage with a clear greenish gloss, whereas at St. Margaret's Island and Caenn Leathad all birds had a deep bluish or bluish purple gloss (Table II). With just one exception (of 109 checked) all the Abberton birds had a bluish gloss.

TABLE II. PLUMAGE GLOSS OF BREEDING CORMORANTS AT ABBERTON RESERVOIR IN COMPARISON WITH OTHER COLONIES.

Colony	Date	n	Plumage gloss (%)	
			green	blue or bluish-purple
Oostvaardersplassen	27 Apr 89	30	97	3
Naardermeer	28 Apr 89	35	100	0
Abberton Reservoir				
north sub-colony	14 May 89	13	8	92
south sub-colony	“ “	68	0	100
“ “ “	20 May 90	28	0	100
St. Margaret's Island				
(sub-colony A)	20 May 89	57	0	100
“ “	13 May 90	54	0	100
Ceann Leathad	16 May 87	25	0	100

### *Whiteness of head and prominence of white thigh patches*

At both the Dutch colonies when visited in April 1989 the vast majority of birds had some white head feathers and many had markedly white heads (Table III). At Abberton, St. Margaret's Island, and the other British colonies visited, by contrast, no more than 5% had any white head feathers and even these were not very distinct, for all that these birds were less advanced in the breeding cycle than the Dutch birds. A similar trend was apparent in the intensity and size of the thigh patches – large and prominent in the Dutch birds, smaller and less obvious in the British ones (Table III).

### *Shape of the gular pouch*

In a small sample (based on photographs rather than direct field observation) all the Dutch birds corresponded to Alström's *sinensis* pattern and over 97% of those in Scotland and Wales had *carbo* style gular pouches (Table IV). At Abberton a total of 13 birds showed the *sinensis* pattern

TABLE III. WHITENESS OF HEAD AND PROMINENCE OF THIGH PATCHES IN BREEDING CORMORANTS AT ABBERTON RESERVOIR IN COMPARISON WITH OTHER COLONIES

Colony	Date	Approx. No. birds examined	Whiteness of head	White thigh patches
Oostvaardersplassen	27 Apr 89	100	Most birds with white head feathers, <i>ca.</i> 30 with very white heads	Mostly large and distinct
Naardermeer	28 Apr 89	150	Most birds with white head feathers, <i>ca.</i> 30% with very white heads	Mostly large and distinct
Abberton Reservoir	14 May 89	300	<i>ca.</i> 5% with white heads but none as distinct as Dutch birds	Most if not all birds with white thigh patches but smaller and less distinct than Dutch birds
„ „	20 May 90	300	none with distinctly white heads	most birds with white thigh patches, but smaller and less distinct than Dutch birds; some without any white thigh patches
St. Margaret's Island	20 May 89	200	Virtually no birds with white head feathers	All breeding birds with white thigh patches, but smaller and less distinct than Dutch birds
„ „	13 May 90	200	A few birds with some white head feathers, but most without any	Present but smaller and less distinct than Dutch birds
Sheep Island	25 Jun 89	20	No birds with white head feathers	Small and indistinct
Ceann Leathad	16 May 87	25	Few indistinct white head feathers only	Present, but less obvious than Dutch birds
Llanddeiniol	1 Jun 90	50	No birds with white head feathers	<i>ca.</i> 20% with white thigh patches but these smaller and less distinct than Dutch birds

and 87 the *carbo* pattern. Of the 13 (all of which were seen on 20 May 1990) 7 were in a small group in the southern sub-colony; all the 1989 data were collected in the northern sub-colony. Unfortunately weather and other conditions were never suitable to determine what colour plumage gloss the birds with the *sinensis* pattern gular pouches had.

#### *Evidence from ringing*

South-east England is an important wintering area for Cormorants (Feare 1988, Sellers 1991) and Abberton itself has been used as a night-time roost for at least 30 years, numbers increasing from about 100 in the 1950s to *ca.* 250 in the 1980s (Essex Bird Report 1955-1988, Cox 1984). In the non-breeding season there has been only one sighting of a colour-ringed Cormorant at Abberton itself, and this came from the Netherlands. Elsewhere in Essex colour-ringed birds have been seen from the Netherlands (1 at Hanningfield Reservoir, 30 km south-west of Abberton), from Denmark (1 at Holyfield Marsh G.P., 60 km west-south-west of Abberton) and from Wales (1 at West Thurrock, 55 km south-west of Abberton, and another at Walthamstow Reservoir, 66 km west-south-west of Abberton), whilst in Suffolk several have been reported at the Melton roost near Woodbridge, 38 km north-east of Abberton. These included three from Denmark, two from the Netherlands and three from Wales. In addition a bird colour-ringed at St. Margaret's Island has

TABLE IV. SHAPE OF GULAR POUCH IN CORMORANTS AT ABBERTON RESERVOIR IN COMPARISON WITH OTHER COLONIES.

Colony	Date	n	Shape of gular pouch	
			sinensis type (%)	carbo carbo type (%)
Naardermeer	28 Apr 89	17 <sup>a</sup>	100	0
Abberton Reservoir	14 May 89	34	0	100
" " "	20 May 90	66	20	80
St. Margaret's Island	20 May 89	48	4	96
" " "	13 May 90	10	0	100
Ceann Leathad	16 May 87	7 <sup>a</sup>	0	100

<sup>a</sup> Shape of gular pouch determined from photographs.

been found dead on the Blackwater Estuary, 10 km south-west of Abberton and another colour-ringed in Denmark at Hanningfield Reservoir. Sightings of continental birds are likely to be over-represented as approximately five times more have been colour-ringed there than in the British Isles (*cf.* Methods).

Nearly all ringed Cormorants recovered dead in Essex had been ringed as nestlings in the British Isles (Table V). In addition a number of fully-grown Cormorants have been ringed in Essex generating 22 further recoveries. Thirteen (59%) were in the same or subsequent winters in Essex, 4 (18%) in the winter months outside Essex (1 each in Sussex, Hampshire, Gloucestershire and Gwynedd) and 5 (23%) during the breeding season (2 in Dorset, and 1 each in Dyfed, Clwyd and the Netherlands).

TABLE V. RINGING LOCALITIES OF CORMORANTS FOUND DEAD IN ESSEX.

Ringing locality		n	(%)
France	Manche	1	1.4
England	Northumberland	13	18.3
Wales	Dyfed	24 <sup>a</sup>	33.8
	Gwynedd	21	29.6
Ireland	Wexford	1	1.4
Scotland	Dumfries & Galloway	4	5.6
	Strathclyde	2	2.8
	Lothian	3	4.2
	Highland	1	1.4
Denmark		1 <sup>b</sup>	1.4
Total		71	

<sup>a</sup> Includes one colour-ringed bird

<sup>b</sup> Colour-ringed bird *P.c.sinensis*; all others *P.c.carbo*.

## DISCUSSION

To judge by their appearance the majority of birds breeding at Abberton Reservoir in 1989/90 were more likely to be *carbo* than *sinensis*, and by implication, therefore, originated from within the British Isles rather than the continent. Cormorants of the N. Atlantic race usually breed on cliff ledges, rocky stacks and islets (Cramp & Simmons 1977) but tree-nesting is not unknown. Thus in Ireland, Macdonald (1987) recorded that 14% of colonies (11 of 77) surveyed in 1985 and 1986 were in trees and there were formerly tree-nesting colonies in Norfolk (Seago 1967), Suffolk (Payn 1962), Pembrokeshire (Lockey *et al.* 1949) and Scotland (Baxter & Rintoul 1953), as well as an unsuccessful attempt in Kent in 1947 (Gregory 1948).

The presence of Cormorants in the Abberton area in the winter months suggests a ready mechanism by which colonisation could have taken place and the evidence from ringing (Table V) is that most of these are of British origin. Continental Cormorants are mainly migratory, moving to winter on the shores of the Mediterranean, though many especially from Dutch colonies move only to northern France (van Eerden & Munsterman 1986). Few regularly occur in the British Isles and less than 2% of sightings of birds colour-ringed in Denmark have been in Britain (Gegersen & Halberg 1988), although these few are concentrated mainly on the east coast of Britain between Kent and Yorkshire. The movements of British-bred Cormorants are usually described as dispersive and are mostly directed to the south and east (Coulson & Brazendale 1968). Many move into southern Britain, numbers being highest in south-eastern England (Feare 1988, Sellers 1991). Significant numbers move to the west coast of France and some go as far south as Spain and even Portugal (Coulson & Brazendale 1968). It is not known how far Cormorants will move between natal and breeding colonies, but movements of several hundred kilometres must take place as shown by the recent colonisation of Italy (Baccetti 1989), Poland (Hansen 1984), East Germany (Zimmermann 1985) etc. Within the British Isles three nestlings colour-ringed on Lambay Island, Co. Dublin were observed as breeders on St. Margaret's Island, a straight line distance of about 210 km (R.M. Sellers unpublished). The nearest colonies to Abberton are on the Isle of Wight (*ca.* 190km) and in the Netherlands (*ca.* 250km). Colonisation by either race is clearly feasible both in terms of distance and given what is known of the birds' dispersal or movement patterns, but colonisation by *carbo* seems most likely on the evidence from ringing and is consistent with the conclusions from the plumage of the current breeding population. That there may be some *sinensis* birds breeding at Abberton cannot be ruled out and the data in Table IV suggest that as many as 13% may be *sinensis*. It is possible that the initial colonisation was by *sinensis* birds from the continent with wintering *carbo* attaching themselves to the colony in later seasons. The growth rate of the colony has been too great to be due solely to the initial colonising cohort, and it seems that recruitment from other colonies must have occurred in some, if not most, years since the colony was founded.

In a development closely paralleling that at Abberton, Cormorants in France have also recently begun nesting in trees. The French breeding population, which over the past 20 years has increased from *ca.* 300 pairs to *ca.* 1200 pairs (Debout 1987), was formerly strictly coastal and was identified as being of the N. Atlantic race (Yeatman 1976). The new colony of tree-nesting Cormorants was founded in 1981 (7 pairs) at the Lac de Grand-Lieu in Loire-Atlantique, some 23 km from the sea (Marion & Marion 1984). As with the Abberton birds the question arises as to the provenance of the birds. Marion (1983, 1991) concluded that they were *sinensis*, an assignment based primarily on the duration and intensity of the white head feathers, but also on nesting habitat. Birds of both races winter in France so, as at Abberton, colonisation by either is feasible. Marion (1983) has shown using ringing recoveries that there is some segregation of the races, most *carbo* wintering on coasts and estuaries, whilst the majority of *sinensis* are found on estuaries and inland waters. The birds wintering at the Lac de Grand-Lieu and its immediate area are, therefore, mostly *sinensis*.

## SUMMARY

The racial identity of Cormorants *Phalacrocorax carbo* breeding at a recently founded tree-nesting colony at Abberton Reservoir, Essex was assessed by comparison with birds at two breeding colonies in the Netherlands (birds of the continental race *P.c.sinensis*) and at several colonies in Wales and Scotland (birds of the nominate race *P.c.carbo*). The two races differ in several respects and based primarily on the gloss to the plumage, the whiteness of the head, the size of the white thigh patches and the shape of the gular pouch, together with evidence from ringing, it is concluded that the birds breeding at Abberton in 1989/90 were more likely to have originated from the British Isles than the continent.

## ACKNOWLEDGEMENTS

I am indebted to Dr. M. R. van Eerden for arranging access to the Dutch colonies (as part of the 2nd European Cormorant Workshop meeting in Lelystad, the Netherlands, April 1989), George Brown, Graham Ekins and the Essex Water Authority for access to the Abberton colony, the Dyfed Wildlife Trust and Steve Sutcliffe for access to the St. Margaret's Island colony, Mike Williams for getting me (safely) to St. Margaret's and Stuart Mackay for introducing me to the Ceann Leathad Cormorants. Thanks are due also to the British Trust for Ornithology for providing details of the Essex ringing recoveries and the British Museum (Natural History) for permission to consult their collection of skins. Nicholas Aebischer, Chris Feare, Loic Marion, Mick Marquiss, Richard Potts, John Sanders and Steve Sutcliffe kindly commented on earlier drafts of this paper.

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# Post-mortem examination of Great Northern Divers *Gavia immer* killed by oil pollution in Shetland, 1979

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## INTRODUCTION

Divers are highly vulnerable to floating oil and are regular casualties of both chronic oiling (Camphuysen 1989, Joensen 1972, Stowe 1982) and specific oil spill incidents, e.g. at least 175 were found oiled after the Amoco Cadiz tanker wreck off Brittany in 1978 (Monnat 1978) and c.400 were retrieved from Alaskan beaches in early 1989 after the Exxon Valdez grounding (Piatt *et al.* 1990). In the first three months of 1979, 195 oiled divers were found in Shetland, victims of the Esso Bernicia incident and subsequent, unattributed oil spills (Heubeck & Richardson 1980). This was one of the largest documented kills of divers in Europe and while all four Atlantic species were found, 93% of the total were Great Northern Divers *Gavia immer*. There has been relatively little study of Great Northern Divers wintering in British waters and a sample of the corpses from Shetland was examined in an attempt to increase our knowledge of the species' biology and determine the breeding origin of the birds.

## METHODS

A total of 68 oiled Great Northern Divers were collected from shorelines between late January and 23rd March 1979 and stored frozen. At the Royal Museum of Scotland\* (RMS), Edinburgh, the corpses were cleaned by massaging them thoroughly in a sequence of 3-5 baths of the industrial solvent 'Inhibisol' and dried using magnesium carbonate and air-blowers. The birds were then scanned by a Faxitron X-ray, weighed and measured. The maximum flattened chord of the left wing (carpal joint to the tip of the primaries) was measured with a rule to the nearest 1mm while the tail, culmen, gonys and tarsus were measured to the nearest 0.1mm using vernier calipers. Wing measurements were repeated on the curated specimens after 5 years to provide a measure of shrinkage. Some comparable biometric data were obtained from collections in the British Museum (Natural History), the Museum of Natural History, Reykjavik, the Zoology Museum, Copenhagen, the Museum of Natural Sciences, Ottawa, the American Museum of Natural History, New York and the National Museums of Scotland.\*

Corpses were skinned, examined for dermal and sub-dermal shot, sexed and prepared into study skins, the largest single collection of this species in Britain. Stomach contents and gizzard stones from 23 birds were examined. Samples of pectoral muscle and liver were taken from 12 arbitrarily selected birds and analysed for organochlorine and mercury residues.

## RESULTS

Of the 68 divers collected, 34 were females, 23 were males and 11 were unsexed. The sex ratio is not significantly different from parity ( $\chi^2=2.12$ ). The birds were aged according to plumage differences between first-year, second-year and older birds in January/February (Table I). Sixty-

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\*This former institution has been subsumed as part of the National Museums of Scotland (NMS). The Natural History collections are situated in the Royal Museum of Scotland, Chambers Street, Edinburgh.



TABLE I. AGE-RELATED CHARACTERISTICS OF GREAT NORTHERN DIVER PLUMAGE IN JANUARY AND FEBRUARY (Palmer 1962, Burn & Mather 1974, Cramp & Simmons 1977, Appleby *et al.* 1986).

	<i>First-winter</i>	<i>Second-winter</i>	<i>Third-winter and older</i>
Upperparts	Brownish	Dark grey-brown	Dark grey-brown
Mantle/rump feathers	Rounded with wide, pale margin	Square-ended with indistinct pale margin	Square-ended with indistinct pale margin
Upper-wing coverts	No white spotting	No white spotting	White spotting
Throat	White with some brown speckling	White with no speckling	White with no speckling
Under-tail coverts	Dusky ventral band	White	White
Tips of retrices	Pale brown	White	White

TABLE II. FRESH MEASUREMENTS OF ADULT GREAT NORTHERN DIVERS, SHETLAND, JANUARY-MARCH 1979.

	<i>Sex</i>	<i>Mean</i>	<i>S.D.</i>	<i>Range</i>	<i>n</i>	<i>t</i>	<i>P</i>
Weight (g)	M	3253.6	309.2	2740-4050	21		
	F	2774.5	444.6	2000-4120	32	4.71	0.001
Tarsus length (mm)	M	89.4	5.68	76.5-100.0	23		
	F	86.5	4.86	75.0-98.0	34	2.04	0.05
Tail length (mm)	M	68.5	3.48	61.2-75.0	22		
	F	65.7	3.68	57.8-75.0	34	2.88	0.01
Wing length (mm)	M	375.7	13.59	347-397	19		
	F	359.4	11.32	331-385	28	4.43	0.001
Culmen length (a) (mm)	M	105.1	8.73	87.1-115.5	19		
	F	100.7	6.16	91.0-116.0	27	1.90	0.1
Culmen length (b) (mm)	M	81.7	6.16	71.0-94.0	19		
	F	78.1	3.58	70.5-84.0	27	2.32	0.05
Gonys depth (mm)	M	21.9	1.33	19.5-24.0	23		
	F	20.3	1.25	18.5-23.0	33	4.79	0.001

Wing length: excluding 7 specimens (2 males, 5 females) in wing moult. The shortened wings ranged from 196-286mm.

Culmen length: (a) tip of upper mandible to base of skull, (b) tip of upper mandible to base of feathering.

*t* is the student's *t*-test statistic from the comparison of males and females and *P* is the significance level of each test.

six were in their third winter or older and 2 were in their second winter.

During this work we found variations from the plumage characteristics in Table I. The 8 first-year birds previously in the NMS collection showed the characters listed except that 2 had white instead of dusky-brown under-tail coverts and 2 birds lacked the throat speckling, which was indistinct on a third. Neither of the 2 second-winter birds from Shetland had spotted upper-wing coverts although one had acquired 2 white spots on the mantle. The ventral band was indistinct on both these specimens. The colour of the upperparts of one of them was similar to that of an adult, those of the other similar to a first-year bird. Of the 66 older birds from Shetland, all but 6 had white throats with no speckling.

Full wing and body moult begins during February in adult Great Northern Divers and is completed by early May (Cramp & Simmons 1977). All the Shetland birds were in winter plumage although only 5 (including one found on 6th March) had not begun moulting either rump or mantle feathers. All birds found during late January and February were full-winged but 7 of the 11 adults found between 1st-23rd March were in full wing moult and subsequently flightless. Of the 4 specimens already in the NMS collection obtained during February and March, only 1 (a second-winter bird taken on 18 February in Orkney) was in wing moult. None of the Shetland birds had lost tail feathers.

Males were, on average, 17% heavier than females but weight varied considerably, some birds weighing twice as much as others of the same sex (Table II). Of only 15 birds with noticeable subcutaneous fat, 7 had "slight", 4 had "moderate" and 4 (all shot) had thick (>3mm) deposits. Males were larger than females in all measurements (Table II). Mean shrinkage after 5 years was 1.49% which was significant ( $t = 3.90$ ,  $p < 0.001$ ) and comparable with that reported for other seabirds (Barth 1967, Harris 1980, Ewins 1985).

TABLE III. LEVELS OF DDE, PCBs (PPM/WET WEIGHT) AND MERCURY (PPM/DRY WEIGHT) IN PECTORAL MUSCLE AND LIVER TISSUE OF 12 GREAT NORTHERN DIVERS, SHETLAND, JANUARY-MARCH 1979. SUBDERMAL FAT WAS RECORDED ON ONLY 3 SPECIMENS: MALE 2 AND FEMALE 2 HAD "SCATTERED SPOTS" AND FEMALE 6 HAD "0.5MM OVER PECTORAL MUSCLES" AND "SOME" MESENTERIC FAT. NO BIRDS HAD SERIOUS GUNSHOT WOUNDS BUT SINGLE PIECES OF LEAD SHOT WERE FOUND IN MALES 5 AND 6, AND TWO PIECES WERE FOUND IN FEMALE 5.

		Weight (g)	DDE		PCBs		Mercury	
			Muscle	Liver	Muscle	Liver	Muscle	Liver
Males	1.	2900	0.90	2.20	0.9	2.2	0.7	1.8
	2.	3125	0.81	2.70	1.7	3.9	0.8	12.7
	3.	3400	3.10	3.08	2.3	4.8	—	7.4
	4.	3420	0.68	1.75	0.7	3.4	0.6	17.5
	5.	3760	0.59	0.68	1.1	1.2	0.6	18.9
	6.	—	0.38	0.20	<0.5	1.7	0.2	14.1
Females	1.	2180	0.93	3.30	1.0	4.7	0.7	8.4
	2.	2420	0.77	0.62	1.1	3.3	0.4	4.4
	3.	2540	0.50	1.04	0.6	3.0	0.8	17.7
	4.	2700	0.26	0.70	<0.5	3.9	0.8	70.0
	5.	2750	0.78	1.20	1.7	3.3	0.7	1.1
	6.	4120	0.39	0.10	<0.5	<0.5	0.4	6.3
Mean			0.84	1.46	1.05	2.99	0.61	15.03

Levels of DDE, PCBs and total mercury in pectoral muscle and liver tissue of 12 birds are shown in Table III. There was significant correlation between levels in muscle and liver for both DDE ( $r = 0.64$ ;  $p < 0.05$ ) and PCBs ( $r = 0.52$ ;  $p < 0.05$ ), but not mercury ( $r = 0.31$ ). There was no significant difference between the sexes in any of the contaminant levels.

Sixteen of the 68 birds X-rayed and examined sub-dermally had been shot, most by shotgun lead pellets but 2 by .22 bullets. Four birds had critical or fatal wounds and were presumably shot in attempts at humane destruction. The average number of pellets in the other 10 birds was only 2.3, mostly lodged in or just under the skin. These birds were probably the victims of distant range shooting and in some instances, including one bird with a .22 bullet in the elbow joint, the wounds had healed indicating that shooting had occurred a considerable time before death.

Stomach contents of the 23 birds examined consisted of little other than stones, suggesting that oiled birds had not been feeding. Three stomachs contained mollusc shell fragments and 3 contained fragments of Shore Crab *Carcinus maenas*. Worn otoliths, probably of 0-1 group Haddock *Melanogrammus aeglefinus* or Whiting *Pollachius virens* were also recorded (B.Blake pers.comm.). Gizzard stones were found in all 23 birds and of the total of 336 examined, most were of quartz/feldspar or granitic material of a wide range of geological types which all occur in Shetland, if in a few cases in restricted distribution. A single piece of lava was the only stone of recent volcanic origin.

## DISCUSSION

If the sample examined was representative, most of the Great Northern Divers killed by the Shetland oil spills in early 1979 were in their third winter or older. The absence of first-winter birds in the sample was surprising. Family parties of adults with juveniles are commonly seen in Shetland in early winter but the extent to which first-winter birds remain there throughout the winter is unknown. Enquiries could not determine whether breeding success was abnormally low in 1978 in either Iceland or Greenland, although that summer was colder and wetter than normal in West Greenland (K.Kampp & A.Petersen *in litt.*).

Organochlorine levels in the Shetland birds were low compared with analyses of 25 other divers (mostly Red-throated *Gavia stellata* and Great Northern Divers) found dead or dying in Britain, of which 52% had PCB and DDE levels of 10-100 ppm (Bourne 1976, NERC 1983, Mason & Macdonald 1988). However, contaminant levels can vary because of several factors, including age, physiological condition and methods of sample collection, preservation and analysis, as well as whether wintering areas are relatively clean or polluted (NERC 1983).

Other than the bird with 70 ppm, mercury levels in the livers of the Shetland Great Northern Divers were mostly lower than those found in 6 Red-throated Divers analysed by the Institute for Terrestrial Ecology (ITE) in the mid-1970s (mean 13.9 ppm, range 8.18-21.54) (M.Haas pers.comm.). Published values from Great Northern Divers are normally expressed as ppm wet weight but if a correction factor of dry weight = wet weight  $\times$  3.9 (derived from the ITE sample) is applied then 11 of the Shetland values were lower than Ontario figures for birds breeding on unpolluted lakes (Barr 1986) or healthy adults drowned in fish nets, with the single elevated Shetland level of 70 ppm comparable to those of emaciated birds drowned in nets (Frank *et al.* 1983).

Occurring further offshore and in deeper waters than other divers, Great Northern Divers are difficult to census during winter and regional estimates of numbers wintering in Europe may be underestimates. The Scottish wintering population may number 1,000-1,500 (Thom 1986) with a further 2,000-3,000 around the rest of the British Isles (Lack 1986). Perhaps 1,000 birds winter along the Norwegian coast (Nygard 1985) and with smaller numbers elsewhere along the coast of continental Europe, the north-west European wintering population clearly exceeds the Icelandic breeding population, variously estimated at 100-300 pairs (Cramp & Simmons 1977) and nearer 500 pairs (A.Petersen *in litt.*). A suggestion that birds from Greenland or the Canadian arctic

winter in European waters seems logical (Lack 1986). Based on the above estimates, oil pollution in Shetland in early 1979 may have killed c.14% of the Scottish, c.5% of the British and c.2-3% of the European wintering population.

Great Northern Divers vary in size geographically within a range Storer (1988) described as consisting of "an Old World population breeding in Greenland and Iceland and wintering on the coasts of Europe, and a New World population which is widespread in the United States and Canada". We attempted to determine the breeding origins of the Shetland birds by comparing their wing lengths with those of birds collected during summer on their breeding grounds. The Shetland birds were longer-winged than those breeding in Ontario or in Quebec, Labrador and New Brunswick (Fig.1), which probably winter along the east coast of North America (Powers & Cherry 1983). Palmer (1962) suggested that the Baffin Island population also wintered in North America but the similar wing lengths of available specimens from Baffin Island, Greenland and Iceland made it difficult to speculate as to which of these areas the Shetland birds came from.

Other factors hampered this comparison. The Shetland measurements were of birds prior to their

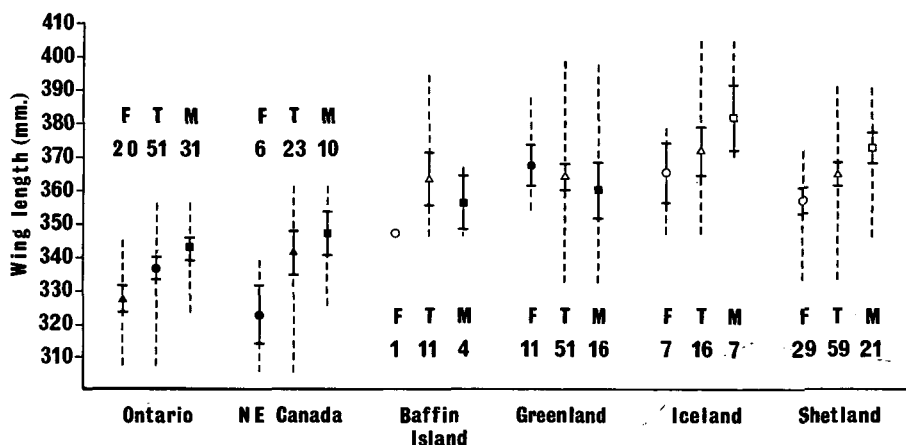


Figure 1. Comparison of wing lengths of Great Northern Divers killed in Shetland during winter with adults collected on breeding grounds between May and August. Northeastern Canada = New Brunswick, Quebec and Labrador. The mean, range and 95% confidence limits are given for females (F), males (M) and all birds (T) in each area and sample sizes are shown. Samples whose mean differed significantly (student's t-test,  $p < 0.05$ ) from their Shetland counterparts are indicated by solid symbols. All measurements were taken in the 1980s on specimens curated for at least 5 years, apart from 6 unsexed specimens from Baffin Island measured by Rand (1947).

annual wing moult and some had markedly abraded primaries, whereas those collected on breeding grounds would have been in relatively fresh plumage. Secondly, the measurements of birds in the five museum collections were all taken by different persons and while all were experienced museum staff, people with experience of measuring wings can consistently measure "short" or "long" (Ewins 1985). Among the divers measured in the National Museums of Canada collection in 1981, we identified 22 birds from different regions for which Rand (1947) had previously published biometrics (Table IV). Within regions with more than one specimen, it was not possible to identify which of Rand's measurements referred to which bird but the average

recent measurements were all shorter, by 15mm and 10mm on the two birds which could be identified. Additional sources of variation could have included the duration of storage (those from Greenland mostly dating from 1880 to 1920, those from Iceland mostly from 1950 to 1970 and those from Ontario mostly from 1976) and possible differences in curation methods (Herreman 1985).

TABLE IV. MEAN WING LENGTHS OF 24 GREAT NORTHERN DIVERS IN THE MUSEUM OF NATURAL SCIENCES, OTTAWA, COLLECTION PUBLISHED BY RAND (1947) AND REMEASURED INDEPENDENTLY IN 1981.

<i>Region</i>	<i>Number</i>	<i>Years collected</i>	<i>1947</i>	<i>1981</i>	<i>Difference</i>
Greenland	2 males	1926, 1943	346.0	336.0	-2.9%
Baffin Island	4 males	3 in 1925, 1 in 1937	373.2	356.6	-4.5%
	1 female	1925	362	347	-4.1%
Yukon	5 males	4 in 1912, 1 in 1944	356.8	345.2	-3.2%
	4 females	1912	345.5	331.7	-4.0%
Manitoba	1 male	1918	352	342	-2.7%
	3 females	1930, 1937, 1945	327.3	310.3	-5.2%
Quebec	2 females	1926, 1928	329.0	316.4	-3.8%
SE Ontario	2 females	1915, 1918	326.0	319.9	-1.9%

An enquiry found no apparent change in the Icelandic Great Northern Diver breeding population in the summer of 1979, although information received was limited (A. Petersen *in litt.*). Some Great Northern Divers winter in south-west Greenland (Salomonsen 1967) but it is not improbable that birds breeding in Greenland and/or the Canadian arctic winter in Europe. Gizzard grit examination showed that the Great Northern Divers wintering in Iceland include birds from elsewhere (Gudmundsson 1972) and a bird found in the Faroes last century contained an Inuit arrowhead (Salomonsen 1935). The discovery of healed gunshot wounds in a proportion of the Shetland birds suggested they spent part of the year in an area such as Greenland, where the risk of being shot at was high. Outwith Greenland, the five ringing recoveries of Red-throated Divers breeding there have been in winter in Norway, Britain and France (Salomonsen 1967, K. Kampp *in litt.*) but in the absence of ringing recoveries of Great Northern Divers there is still no firm evidence on the breeding areas of the European wintering population.

#### ACKNOWLEDGEMENTS

We are grateful to members of the Shetland Bird Club, NCC and RSPB for help with the collection of corpses and to British Petroleum (SVT) for their storage and transport to Edinburgh, where they were cleaned and curated by Martin Abramson, Robert Clarke, Callan Duck, Stephanie Hall and Peter Summers. The NMS's geology section examined gizzard stones, the chemistry section of the Agricultural Scientific Services, DAFS, Edinburgh performed tissue analyses. For provision of biometric data we thank Dr. Henri Quellet and Michel Gosselin (Museum of Natural Sciences, Ottawa), Dr. Aevor Petersen (Reykjavik Museum), Mr. Kaj Kampp (Zoology Museum, Copenhagen), Mr. Allison Victor Andors (American Museum of Natural History, New York) and Mr. J.P. Angle (National Museum of Natural History, Washington). Bill Bourne, Jeremy Greenwood, Mike Harris, Chris Mead and John Parrack kindly improved earlier drafts of this manuscript.

## SUMMARY

Examination of 68 Great Northern Divers killed by oil spills around Shetland in early 1979 found that 66 were in their third year or older and two were in their second year. No first-year birds were recorded. Tissue levels of organochlorines and mercury were within the ranges recorded from divers in Europe and North America though one specimen showed abnormally high mercury residues. Corpses were examined by X-ray and 24% were found to have been shot, some probably fatally, though others displayed healed wounds. The number of divers wintering in Europe is considered to be greater than the Icelandic breeding population alone. Wing lengths of the Shetland birds were greater than those breeding in continental north-east Canada but similar to those breeding in Baffin Island, Greenland and Iceland.

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# Recent changes in Fair Isle seabird populations

Nick Riddiford

## INTRODUCTION

Fair Isle has one of the largest concentrations of seabirds in northern Britain. The numbers of most species have been counted at regular intervals since 1959 (Cramp, Bourne & Saunders 1974; Lloyd, Tasker & Partridge 1991; Fair Isle Bird Observatory Reports). Major recent changes have occurred in the numbers and breeding success of several species of seabird further north in the rest of Shetland (Heubeck 1989a; Walsh, Avery & Heubeck 1990; Mead 1989). The present paper reports on the population fluctuations of sixteen Fair Isle seabird species both during and prior to the recent period of breeding failures.

## METHODS

Complete counts of all seabirds nesting on Fair Isle except Storm Petrel and Eider (see Results for scientific names) were made several times from 1959 to 1990. Much of the counting was done from land, but hidden cliffs and stacks were censused from a boat. A very few cliffs could not be seen and these appeared to have remained the same from count to count. The units counted were 'apparently occupied nest site' for Fulmar, Shag, Kittiwake, Common Gull, Common Tern and Arctic Tern; 'apparently occupied territory' for Herring Gull, Great Black-backed Gull and Lesser Black-backed Gull; and 'individuals' for Guillemot, Razorbill, Puffin and Black Guillemot. Census methodology for these species is described more fully in Lloyd *et al.* (1991). The unit of count for Gannet was initially 'occupied nests' but was changed to 'apparently occupied nest sites' from 1982 (Wanless 1987). Thus later counts included a small number of birds attending part-nests which were excluded from early counts. Arctic Skua and Great Skua counts were of apparently occupied territories except in 1972-78 and 1979-80 respectively when they were nest counts. Puffin totals up to 1969 were given as "pairs", based on June counts of individuals with the total divided by two to achieve a figure of "pairs". Two later counts were of individuals present offshore and at colonies in late April. For this study, counts of pairs have been multiplied by two to give a comparative figure of individuals. For species censused by 'apparently occupied nest site' totals were of apparently incubating birds counted by telescope when presence of egg or chick was not ascertained. Tern counts in 1989-90 were also based on apparently occupied nest site but 1974-88 counts were based on counts of scrapes with and without eggs.

## RESULTS

### Storm Petrel *Hydrobates pelagicus*

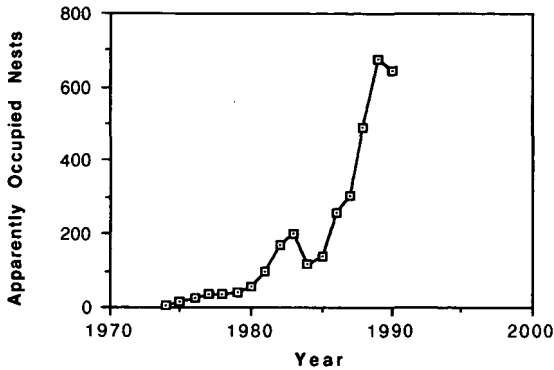
It was not possible to census the species because sites were inaccessible or too vulnerable to disturbance. The only observed change during the last 30 years has been the loss of breeding sites in old stone walls, though most of these walls are still present.

### Fulmar *Fulmarus glacialis*

The population increased four-fold between 1959 and 1975 but there was little subsequent change (Table I).

### Gannet *Sula bassana*

Colonisation began with 3 nests at Dronger in 1974. Numbers increased slowly but steadily to 1980 and more rapidly in 1981-83 when additional cliff and stack sites were colonised (Fig. 1).

**Figure 1: Gannet Counts**

There was an interruption to population growth in 1984 when a 40% decrease in apparently occupied nests occurred but this was followed in 1985-89 by the period of strongest growth at a rate of 48.8% per annum.

#### Shag *Phalacrocorax aristotelis*

The counts varied from 1,000 to 1,530 nests (Table I). Decreases between 1959 and 1966, rather higher numbers in 1969 and 1975 and a return to 1959-66 levels in 1986 suggested rather erratic changes in breeding numbers.

#### Arctic Skua *Stercorarius parasiticus*

The population was relatively stable in 1959-65 but then increased rapidly to 180 occupied territories in 1969 (Fig. 2). Numbers then declined with a slight recovery in the late 1970s. Between 1977 and 1989 the population declined by 1.3% per annum.

#### Great Skua *Stercorarius skua*

Small increases from 1959-63 were followed by decreases in the next two years and the population remained very low into the 1970s (Fig. 2). Increases from 1974 were slow at first but in the 1980s the population increased rather erratically to a 1989 level 3.5 times higher than in the early 1970s.

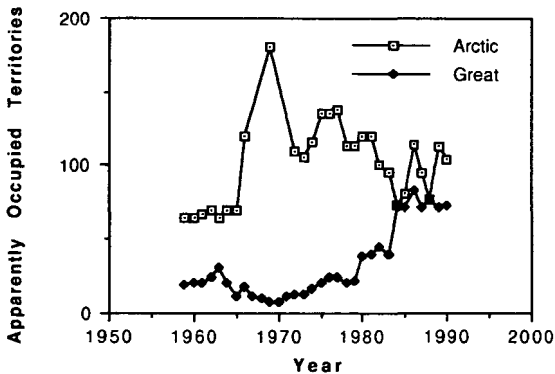
**Figure 2: Skua Counts**



TABLE I. CENSUS COUNTS OF 10 FAIR ISLE SEABIRD SPECIES, 1959-90 (details of units for counts are given in the text).

<i>Species</i>	<i>YEAR</i>						
	1959	1964	1965	1966	1969	1970	1975
Fulmar	5000				17000		26000
Shag	1200			1000	1530		1491
Herring Gull	140			315	315		486
Great Black-backed Gull	40			55	55		141
Lesser Black-backed Gull	20	30	30	37		20	46
Kittiwake	2746				12121		17000
Guillemot	3500			11280	14064		19200
Razorbill	1750				2200		2500
Black Guillemot							360
Puffin	40000			31000	30000		

<i>Species</i>	<i>YEAR</i>						
	1980	1982	1986	1987	1988	1989	1990
Fulmar			26995				
Shag			1099				1073
Herring Gull			262				
Great Black-backed Gull			113				
Lesser Black-backed Gull			22				
Kittiwake			19072		19340		
Guillemot	30000		33607	32437		32321	
Razorbill			3882		3436		
Black Guillemot		360	367			345	
Puffin			20224			23050	

*Common Gull Larus canus*

Single pairs bred in 1966 and 1973. The maximum population was 12 pairs in 1986 (Fig. 3).

*Lesser Black-backed Gull Larus fuscus*

The population fluctuated between 20 and 48 pairs (Table I).

*Herring Gull Larus argentatus*

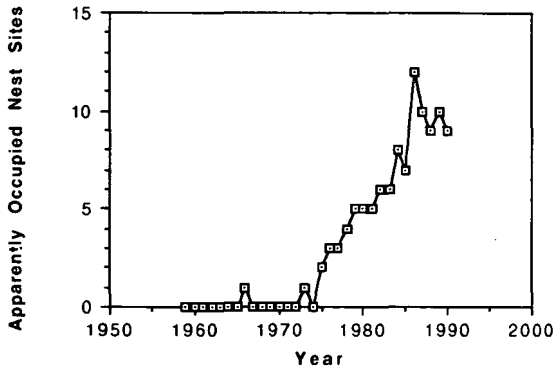
The population more than doubled between 1959 and 1966 and increased another 50% by 1975 (Table I). The 1986 census indicated a return to the 1960s level.

*Great Black-backed Gull Larus marinus*

Counts indicated small population growth in 1959-69 followed by a nearly three-fold increase in 1969-75. By 1986 the population had declined again to a level 20% below the 1975 count.

*Kittiwake Rissa tridactyla*

Between 1959 and 1975 the population increased at 12.1% per annum but there has been little change since (Table I).

**Figure 3: Common Gull Counts**

#### Common Tern *Sterna hirundo*

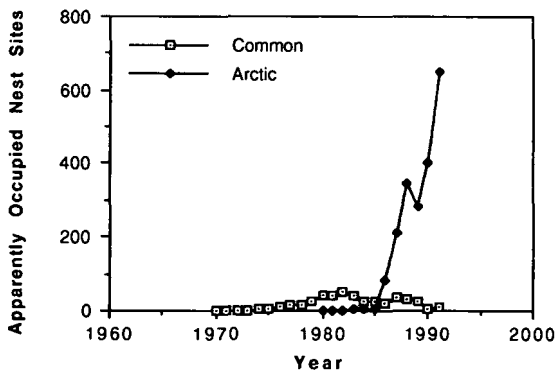
Population growth after colonisation in 1970 was very slow at first but accelerated to a peak of 50 nests in 1982 (Fig. 4). The population then began to decrease, a trend interrupted only briefly in 1987. The Stack o North Haven was the sole colony, though solitary pairs bred in Arctic Tern colonies at Bunes and South Light in some years in the 1980s.

#### Arctic Tern *Sterna paradisaea*

Two pairs nested among Common Terns at Stack o North Haven in 1980. In 1981 there were single pairs there and at Bunes. Breeding numbers remained low until 1986 when a ten-fold increase brought the total to 83 apparently occupied nest sites (Fig. 4), all on Bunes. Numbers then increased annually to a peak of 345 in 1988 before decreasing to 283 in 1989. In 1990 the count was 400 apparently occupied sites. Features of the marked increases after 1985 were late influxes to the breeding sites (peak numbers often not arriving until mid June) and extremely low breeding success. Several new breeding localities were established.

#### Guillemot *Uria aalge*

There was a large increase from 3,500 individuals in 1959 to 33,607 in 1986, since when there has been little change (Table I).

**Figure 4: Tern Counts**

*Razorbill Alca torda*

There was a gradual increase over the period from 1,750 individuals in 1959 to c3,500 in 1986-88 (Table I).

*Black Guillemot Cepphus grylle*

Four counts between 1975 and 1989 were all in the range 345 to 367 individuals (Table I).

*Puffin Fratercula arctica*

There were consecutive decreases in numbers of Puffins counted in 1965, 1966 and 1969 (Table I). The 1986 count (done in April instead of June and thus not strictly comparable) produced a figure only half that of 1959. An April count in 1989 recorded a small (14%) increase over the 1986 count.

## DISCUSSION

The counts suggest a number of trends for Fair Isle seabirds. Fulmar, Gannet, Great Skua, Common Gull, Kittiwake, Common and Arctic Terns, Guillemot and Razorbill were all species with populations apparently increasing to at least the 1980s. The sandeel *Ammodytes marinus* is, or has been, the basic food of chicks of eight of these species at Fair Isle or other parts of Shetland (Harris & Riddiford 1989; Furness & Todd 1984; Furness 1989; Martin 1989). We have no information for Common Gull. However, sandeels have been recorded in the diet of coastal birds at Murmansk (Cramp & Simmons 1982), and may also have been an item of diet for Fair Isle Common Gull chicks as their parents predominantly foraged at sea (pers. obs.).

A study of sandeel abundance in Shetland waters showed that the stock increased strongly from 1974 to a peak in 1982-83 (Kunzlik 1989). Spawning stock biomass remained high in 1984-86, but 0-group (juveniles in their first year) and total stock biomass declined to below 1974 levels, with 0-group biomass particularly low (Kunzlik 1989). Sandeels are of relatively high energy value, and are considered good food for seabird chicks (Harris & Hislop 1978). Thus the marked population growth and, in the case of Gannet, Common Gull, Common and Arctic Terns, colonisation of Fair Isle, may have been a direct response to an increased availability of sandeels contributing to high breeding productivity.

Only Gannet and Arctic Tern maintained a high rate of population growth throughout the 1980s. The Arctic Tern increases, however, mainly involved late influxes, and virtually no young were raised. Fair Isle had no tradition as an Arctic Tern colony before 1980, but Shetland did, with over 40% of the British population in 1980 (Bullock & Gomersall 1981). From 1983, however, virtually no young fledged from Shetland colonies (Monaghan & Uttley 1989) and recruitment to the small Fair Isle tern colony may have occurred after early failure at these or other more traditional sites. Gannet population growth was accompanied by moderately good breeding success, ranging from 0.48 to 0.78 chicks per monitored site in 1986-89 (Harvey, Silcocks & Howlett 1989). A study elsewhere in Shetland demonstrated that Gannets were able to respond to changes in prey availability by replacing sandeels as a major item of food for their chicks with herring *Clupea harengus* and mackerel *Scomber scombrus* (Martin 1989). Food collected in 1989 confirmed that Fair Isle Gannet chicks were also being fed herring and mackerel (Harvey, Silcocks & Howlett 1989). For other species population growth either slowed considerably (Fulmar, Kittiwake), stopped (Great Skua, Common Gull, Guillemot, Razorbill) or declined (Common Tern). This suggested that the population fortunes of most of these species may have been linked with sandeel stocks. However, the extent to which other factors may also have affected population trends is unknown. A recent study based partly on Fair Isle Guillemot ringing recoveries suggested that increased adult mortality in winter and possibly lower recruitment due to higher mortality of immatures may also have contributed to the end of population growth at Fair Isle in the late 1980s (Heubeck, Harvey & Okill 1991).

Three species, Arctic Skua, Herring Gull and Great Black-backed Gull had population increases into the 1970s followed by declines. The development of the Shetland herring fishery from 1965 (Goodlad 1989) was probably an important factor in the population growth of both Herring and Great Black-backed Gulls as both species foraged substantially for trawler discards (pers. obs.). The collapse of the herring fishery in 1974 prompted Shetland trawlers to turn increasingly to catching sandeels from which there is no by-catch and little waste (Goodlad 1989). Unlike their mainland counterparts, Fair Isle breeding Herring and Great Black-backed Gulls cannot visit refuse tips or similar alternatives, and the fall in population levels suggested a natural readjustment to the requirements of finding food without assistance from man.

The changes for Arctic Skua are more difficult to interpret because the population was historically maintained at an artificially low level by human persecution (Furness 1987). The presence of researchers undertaking intensive studies may have afforded greater protection to the species during part of the period of increases before renewed persecution reduced the population to its late 1970s levels. There was a change in human attitudes in the 1980s and the decline was no longer attributable to persecution (pers. obs.). Fair Isle Arctic Skuas parasitised Kittiwakes and other smaller seabirds specialising on sandeels. Thus, the species may more correctly be allied to those whose populations increased into the 1980s. Breeding success remained relatively good to 1988 (Riddiford & Silcocks 1988) but was poor and declining in Shetland in 1986-88 (Heubeck 1989b) and poor at Fair Isle in 1989 and 1990 (Harvey, Silcocks & Howlett 1989; Harvey, Proctor & Donald 1990).

There was no discernible pattern for Shag and Lesser Black-backed Gull, though population size fluctuated for each. Sandeels are a substantial prey item of Fair Isle and Shetland Shags (Harris & Riddiford 1989; Martin 1989; Harvey *et al.* 1989). However, the Shag is difficult to census fully at Fair Isle and the apparently erratic population fluctuations may have been an artefact of inefficient counting methods or other unknown factors. Lesser Black-backed Gull is a long distance migrant and exceptional mortality on migration or in winter may have contributed to low numbers in 1984 when c50% of adults failed to return to Fair Isle (Riddiford 1984). The species is less of a scavenger than some other large gulls and is known to catch its own fish at sea (Cramp & Simmons 1982).

Census counts suggested that the Black Guillemot population remained stable, at least from 1975. The species includes sandeels in the chicks' diet in Shetland (Martin 1989). At Fair Isle, in studies in 1970 and 1972 (Slater & Slater 1972) and 1987-89 (Harris & Riddiford 1989; Harvey *et al.* 1989), sandeels were not the main prey item brought to chicks and none was recorded in 1988-89 (Harris & Riddiford 1989; Harvey *et al.* 1989). Butterfish *Pholis gunnellus* was the commonest food item in all years (Harris & Riddiford 1989; Harvey *et al.* 1989; Slater & Slater 1972) and a variety of other prey suggested that the species was an opportunistic feeder, with sandeels included in the diet as and when encountered. Generalist feeding and consistent availability of main prey items such as butterfish may have contributed to the apparent stability of the Fair Isle population.

The Puffin is notoriously difficult to census accurately (Harris 1984). The main aim of Fair Isle seabird censuses was to establish breeding population trends, so to eliminate the involvement of non breeders the 1986 census was carried out on late April evenings, immediately before the start of the breeding season and at a time when non breeders were considered to be few or absent (Harris 1984). The 1989 census was also done at that time, and this showed a slight increase. With different methodologies employed we were unable to draw any conclusions from the 1959-89 counts. However, we observed full occupation of traditional breeding sites by the mid 1980s and some expansion into new sites. Expansion may have been partly due to abandonment of heavily eroded traditional sites, but our overall impression was that the species was probably as numerous in 1986 as at any time in the last twenty years and previous census totals may have been inflated by non breeders. Sandeels normally form a large proportion of the diet of Puffin chicks on Fair Isle (Harris & Riddiford 1989) and, unless other factors outside the breeding season were affecting

survival rates, the species may have experienced population trends similar to those of other species feeding their chicks predominantly on sandeels.

Long-term monitoring of seabirds breeding at Fair Isle has demonstrated major population changes for a number of species. The balance of population levels between 1959 and 1990 was largely one of increases and species gains. However, in 1988-90 a number of Fair Isle species (e.g. Kittiwake, Arctic Tern) had very low breeding success, in common with Shetland where failures have been more widespread and long-term (Heubeck 1989a). The size and long-term survival of seabird populations is probably heavily dependent on the influence of man and/or environmental pressures on their respective prey species. Continued monitoring of seabird populations is therefore important and Fair Isle, with a long run of data already in place and with a large number and range of seabird species, is in a very useful position both environmentally and geographically to study the future development of seabird populations in northern Britain.

#### SUMMARY

This paper summarises the results of 32 years of seabird census at Fair Isle, Shetland. Populations of Fulmar, Gannet, Great Skua, Common Gull, Kittiwake, Common and Arctic Terns, Guillemot and Razorbill showed a trend of apparent increases to at least the 1980s. Arctic Skua, Herring Gull and Great Black-backed Gull populations increased to the 1970s then declined. Shag and Lesser Black-backed Gull counts fluctuated but with no discernible pattern. Black Guillemot counts suggested a stable population size. The Puffin showed a trend of decrease, but this may have been an artefact of inefficient census methods. Trends are discussed in relation to methodology and food requirements. The counts suggested a relationship between food availability and population trends, particularly for those species including sandeels in their chicks' diet. The paper demonstrates the usefulness of long-term census work in monitoring the environmental well-being of seabird populations.

#### ACKNOWLEDGEMENTS

Mike Harris, Mark Tasker, Paul Harvey and Andy Gosler greatly improved drafts of this manuscript. The study comprises part of the work of Fair Isle Bird Observatory, supported in 1986-90 by a contract funded by the Nature Conservancy Council's Seabird Monitoring Programme. The Carnegie Trust (UK) assisted with the provision of computer facilities to Fair Isle Bird Observatory. My grateful thanks to them all.

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# Development of head moult of Black-headed Gulls *Larus ridibundus* in southern Spain

A. M. Paterson

## INTRODUCTION

Anyone who watches gulls regularly will have observed examples with abnormal plumage for the season. In the Black-headed Gull *Larus ridibundus* this most frequently manifests itself in occasional adults showing the dark brown hood of breeding plumage very early in December (eg. Cowley 1989), whilst the majority of adults in Britain attain this head plumage between the middle of February and the end of March (Grant 1986, Cowley 1989, Smith 1989). It was with the aim of investigating more thoroughly the head moult of Black-headed Gulls towards the southern limit of the common wintering range in northern Morocco (Isenmann 1976; 1978), that a study of the timing of head moult patterns was carried out in Málaga Bay, southern Spain.

## METHODS

The study was carried out along the shore at three points between Torremolinos and Fuengirola (Málaga province) between 1st July 1990 and 31st March 1991. The dates being determined by the numbers of Black-headed Gulls present and available for censusing. The census site was changed each time in an attempt to avoid multiple censusing of the same bird, given that there is probably some attachment to a specific area, as has been seen in Lesser Black-backed Gull *L. fuscus* in the same region (*pers. obs.*). Censusing took place 4 to 6 times in each half monthly period for about an hour on each occasion. Birds censused were in both resting parties on the beach and 'fly-by' birds, care being taken to avoid repetition counting. Determination of the state of head moult was carried out using 10 x 40 binoculars at ranges less than 100 metres. The following scheme was devised to determine and categorise the state of head moult:

- A: Completely white except for grey ear spot (adult winter plumage)
- B: With one or two dark bars across the crown of the head.
- C: Brown on about 50% of the head.
- D: Only a few white feathers remaining on the forehead on a basically brown head.
- E: Head totally brown (adult breeding plumage).

## RESULTS

A total of 5314 birds were censused in the period, of which 91.5% (4862) were adults and the remaining 8.5% (452) first-year birds, using the plumage criteria for separation given in Grant (1986). Any separation of second-year birds was not deemed possible given the variability of plumage factors (Grant 1986). Age group distributions were similar to those obtained in previous years (Paterson 1990), with only about 2% fewer first-year birds than usual.

Figure 1 shows the timing of the devised moult sequences of adult birds, expressed in half-monthly units as a percentage of the total for each age group. Data for the first half of July should be treated with caution, given the low numbers of birds present and censused. Figure 2 shows the timing of the moult sequences for first-year birds.

The moult of adults towards winter plumage (plumage E) begins at the start of July and there were no records of adults in this plumage after the end of the month. Feather loss in post-breeding moult follows a development which is inverse to that of pre-breeding moult, beginning with the presence of white feathers in the forehead whilst the last part of the head to show brown feathers is the outer edge of the hood, usually in the form of a partial or complete darker ring. The last adult to show any vestige of breeding plumage was on 8th September and from this date until 20th October all showed typical winter plumage (category A).

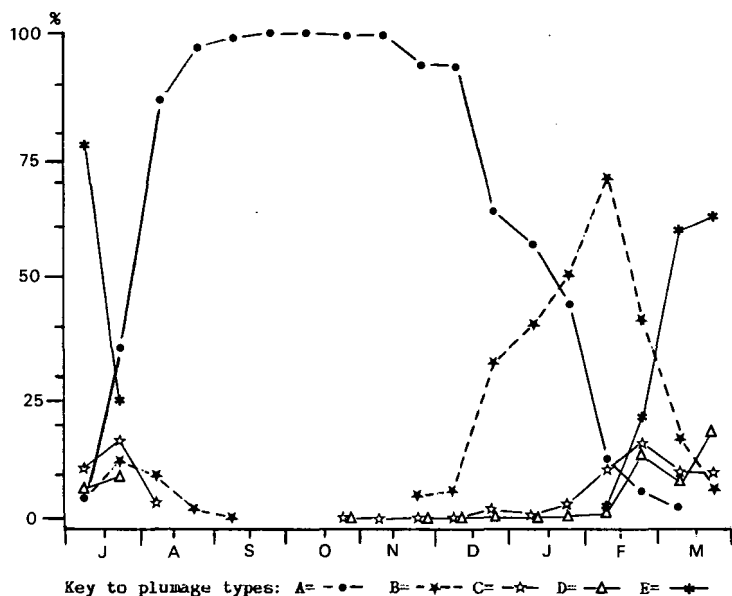


Figure 1. Distribution of categories of head moult in adult *Larus ridibundus* (n=4862) in half monthly units. See text for further explanation.

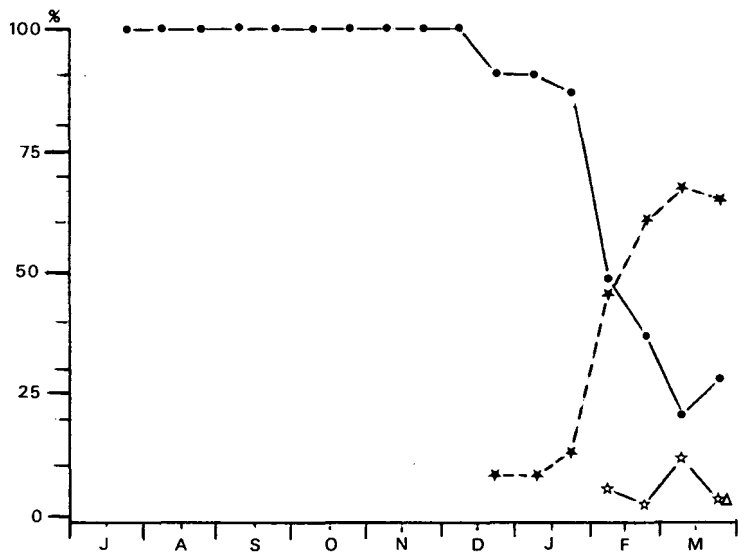


Figure 2. Distribution of categories of head moult in first year *Larus ridibundus* (n=452) in half monthly units. Key to plumage types as for Figure 1. See text for further explanation.



The first adults to show any sign of moult towards breeding plumage were as follows: One in plumage category D on 20th October and another in plumage category E on 23rd October, this second bird showing brown on approximately 75% of the head with a pure white forehead. Between these dates and the end of November a small number of birds in categories C and D were recorded, including one on 27th November which had a completely dark grey head with two black bars.

No adult birds in full breeding plumage (category E) were registered within the census area until 19th January (a casual observation, not during a census), although one was seen in Gibraltar on 12th January (C. Pérez *in litt.*) and in previous years occasional birds have been seen in full breeding plumage in the second half of December and early January in Málaga Bay (*pers. obs.*). No adults were recorded in full breeding plumage during a census until the first half of February and after this time the moult to breeding plumage (category E) showed an increase until finally reaching a fortnightly mean maximum of 62.5% of the adult population in March (maximum individual count 70.3%).

Head plumage development of first-year birds was rather as anticipated from previous observations (Paterson 1990), being some 6 weeks behind that of adults initially and rather slower overall. No birds in plumage category B were recorded until the second half of December, the first ones in category C were seen at the beginning of February and the first ones in category D in the second half of March. No first-year birds were recorded in full breeding plumage during the study period, although occasional oversummering birds are recorded annually in late May and early June (*pers. obs.*).

## DISCUSSION

The progress of the head moult of both adults and immatures was within the limits of what was expected from several years of observation in the region, but there are several significant differences from the data obtained in February and March in Plymouth, SW England, by Smith (1989). Smith censused 10.5% adults in breeding plumage on 2nd February, whilst in Málaga the mean for the first half of February was only 2.4%. Moult to full breeding plumage (category E) figures from Málaga closely equalled those of Plymouth only in the first half of March when 50% of the Plymouth birds were in this plumage and the half monthly mean from Málaga was 48.3% (all ages), 60% of adults, with maxima of 68.9% on 3rd March and 70.3% on 10th. After the middle of March the differences between SW England and Spain in terms of adults in full breeding plumage widened, and in Málaga did not exceed the 62.5% half monthly mean obtained in the second half of March, compared with the 90% recorded by Smith on 22nd March.

It is suggested that the differences in the phenology of head moult categories, especially that of full breeding plumage, obtained between SW England and southern Spain are due to two factors, acting either separately or together. Firstly, adults which are in a relatively advanced endocrinal state towards breeding show earlier development of breeding plumage and also depart earlier from the wintering grounds. Adults which are less advanced, including plumage development, remain further south until they reach a state of development (hormonal and plumage) comparable with the initiation of migration. There is a parallel in the later timing of moult and migration of first-year birds (Paterson 1990). The second possibility is the existence of several distinct populations wintering in the Málaga region and further south (Cramp & Simmons 1983), each possibly initiating its northerly migration at a different time. In either case, the result could be asynchronous plumage development which lowers the maximum proportion of adults in full breeding plumage in Málaga.

It is felt that the suggestion regarding the possible effect of temperature differences between southern Spain and SW England during different seasons of observation may be discounted in the moult phenology. Temperature variation is known to influence migration, accelerating it in low temperatures and retarding it in warmer weather (eg. Dorst 1962) but it is not known to affect moult.

## ACKNOWLEDGEMENTS

I am most grateful to the two anonymous referees whose comments and suggestions have greatly improved this paper.

## SUMMARY

A study of the timing of head moult in Black-backed Gulls *Larus ridibundus* was carried out in Málaga Bay, S. Spain. The pattern of head moult in adults (91.5% of the population) was more or less as anticipated until February. After the beginning of February there were very clear differences between data obtained on the moult of adult birds in SW England and those of Málaga, the numbers of adults in full breeding plumage being much inferior (60% of the adult population) to the 90% achieved in England. It is suggested that the differences are due to one or both of the following factors applying to Málaga: (a) adults which have not attained full breeding plumage remain south until later, this being a reflection of their state of endocrine development; (b) the involvement of different populations with different migration timings. Head plumage development of first-year birds was later than that of adults.

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# Failure to detect blood parasites in seabirds from the Pitcairn Islands

M.A. Peirce and M. de L. Brooke

## INTRODUCTION

Fieldwork for the Sir Peter Scott Commemorative Expedition to the Pitcairn Islands was undertaken between January 1991 and March 1992. The primary aim of this multinational multidisciplinary venture was to inventory the biota of these islands at the extreme south-east of the Indo-Pacific biogeographical province (Paulay 1989). Although the research effort was concentrated on the raised coral island of Henderson, a World Heritage Site, visits were also made at intervals to the low atolls of Ducie and Oeno. Seabirds on these small atolls were generally accessible and presented the opportunity to take blood smears for parasitological examination.

There have been only a few surveys of seabirds on oceanic islands for the presence of haematozoa, and only two have so far produced positive results (Lowery 1971, Peirce & Feare 1978). This paper reports on the samples taken from the Pitcairn Islands and the apparent absence of any haematozoa in the population.

## STUDY AREA AND METHODS

Ducie (24° 39'S, 124° 48'W), one of the world's most isolated atolls is only visited once or twice a year. Of the four islets fringing a central lagoon, the largest is Acadia Island where all sampling was done. The island is composed of coarse coral rubble. The vegetation is a stand of *Tournefortia argentea*. Only one other vascular plant, a single *Pemphis acidula* bush, is known.

Oeno (23° 56'S, 130° 45'W) is also only rarely visited. There is a single island, centrally situated in the lagoon. Compared to Ducie the vegetation is more diverse comprising some 13 vascular species (Fosberg *et al.* 1989), mostly widespread Indo-Pacific species such as *Pisonia grandis*, *Suriana maritima* and *Tournefortia argentea*.

Seabirds were sampled on Oeno on 2 and 3 September 1991 and on Ducie on 26 October 1991. Thin blood smears were prepared, stored and subsequently stained with Giemsa according to the methods adopted and described by Peirce & Prince (1980). Ectoparasites were also collected from birds and around nests.

## RESULTS

A total of 97 birds of 8 species was sampled. These included both adults and chicks as detailed in Table I. No blood parasites were found in any of the smears.

Ectoparasites collected included the argasid tick *Ornithodoros capensis* (not found on birds but from nests and other areas) on Ducie but not on Oeno, mallophaga and hippoboscids.

## DISCUSSION

There are very few records of haematozoa from seabirds. This may in part be directly attributable to the low numbers examined. Apart from *Hepatozoon albatrossi* described from albatrosses in the Antarctic (Peirce & Prince 1980) the only other blood parasites recorded from free-living seabirds are *Haemoproteus* sp from the Great Frigatebird *Fregata minor*, Lesser Frigatebird *F. ariel* and Brown Noddy *Anous stolidus* on Aldabra (Lowery 1971), and *Babesia* sp from Masked Booby *Sula dactylatra melanops* from Desnoeufs in the Amirantes (Peirce & Feare 1978). These latter two records are both from the western Indian Ocean.

TABLE I. NUMBERS AND SPECIES OF SEABIRDS FROM WHICH BLOOD SAMPLES WERE OBTAINED ON DUCIE AND OENO ATOLLS

	<i>Ducie</i>	<i>Oeno</i>
Herald Petrel <i>Pterodroma arminjoniana heraldica</i>	9 chicks: 3 adults	—
Kermadec Petrel <i>P. neglecta</i>	10 chicks: 2 adults	—
Murphy's Petrel <i>P. ultima</i>	10 chicks	20 adults
Red-tailed Tropicbird <i>Phaethon rubricauda</i>	—	20 chicks#
Red-footed Booby <i>Sula sula</i>	—	15 chicks#
Sooty Tern <i>Sterna fuscata</i>	6 chicks#	—
White Tern <i>Gygis alba</i>	—	1 chick
Brown Noddy <i>Anous stolidus</i>	—	1 adult

Note: <1% of the population was sampled except where indicated by # when the figure was between 1-10%.

Species of *Haemoproteus* are transmitted either by ceratopogonids or hippoboscids but since the former are usually absent from small oceanic islands vectors for seabirds are assumed to be the latter. Lowery (1971) found the hippoboscids *Olfersia spinifera* and *O.aenescens* on birds on Aldabra although only the former was found on birds infected with *Haemoproteus*. *O.aenescens* has also been recorded from the Pitcairn Group (Ducie and Henderson), as too has *Ornithoica pusilla* (Henderson) (Mathis 1989). Thus potential vectors of *Haemoproteus* are present.

The vectors of *Hepatozoon* have, until recently, been considered to be arthropods, usually ixodid ticks and mites. In the Antarctic such ectoparasites have been implicated as the possible vectors of *H.albatrossi* (Peirce & Prince 1980). More recently, studies on *H.atticorae* in Cliff Swallows, *Hirundo spilodera* in South Africa, have suggested an argasid tick *Ornithodoros peringueyi* and a flea as intermediate hosts (Bennett *et al.* 1992). *Ornithodoros* ticks were found during the present survey. Therefore a potential vector of *Hepatozoon* is present.

The ixodid tick *Amblyomma loculosum* is considered to be the vector of the *Babesia* sp found in Masked Booby chicks in the Amirantes (Peirce & Feare 1978). This tick, unusually for an ixodid, is a rapid feeder (Peirce & Feare 1978) and is therefore rarely found on its avian host. Like *Ornithodoros*, *A.loculosum* feeds on both adults and chicks at the nest during the breeding season. Failure to detect any ixodid ticks on birds during the present survey does not necessarily preclude their presence.

Since birds are breeding throughout the year, any seasonal trend in parasite prevalence associated with this activity, can probably be discounted. Thus whilst potential vectors for two and possibly three genera of avian haematozoa are present in the Pitcairn Group, the failure to demonstrate the presence of any blood parasites suggests either the sample of birds was too low to confirm a low prevalence of infection, or the seabird populations are genuinely free of blood parasites.

## ACKNOWLEDGEMENTS

We are grateful to Dr. G. F. Bennett, International Reference Centre for Avian Haematozoa for kindly supplying the microscope slides. MdeLB wishes to express his gratitude to the Royal Society, BOU, ICBP (British Section), UNESCO and the Foreign and Commonwealth Office for their financial support and the many individuals whose willing help made the Expedition possible. This is paper number 5 from the Pitcairn Islands Scientific Expedition.

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## BOOK REVIEWS

AINLEY, DAVID G. & BOEKELHEIDE, ROBERT J. (Eds) 1990. *Seabirds of the Farallon Islands. Ecology, dynamics, and structure of an upwelling-system community.* 450pp. \$60.00. Stanford University Press, Stanford. ISBN 0-8047-1530-0.

The Farallon Islands lie in the Pacific, 35km west of San Francisco. Perched at the edge of the continental shelf in an area influenced both by the cold southward flowing California Current and by coastal upwelling, the rocky islands provide ideal breeding conditions for many thousands of seabirds and sea mammals. Two species of storm petrels, three cormorants, the Western Gull and five species of auks nest in only about 46 hectares. The islands were once even more densely populated but exploitation in the last century by sealers seeking fur and oil, and by the Farallon Egg Company, set up to meet demand for eggs during the Californian Gold Rush, led to a decimation of the mammal and bird numbers from which they have never fully recovered.

Since 1971, the Farallones and their seabirds have been the subject of a long-term study by a team from Point Reyes Bird Observatory and numerous volunteers, of which I was one during 1984. This book presents the data gathered up to 1983 in twelve chapters. A general introduction, a description of the seasonal and annual variations in the marine environment near the Farallones, and a discussion of the birds' feeding ecology are followed by accounts of each species' breeding biology. Patterns which emerged within the community are discussed in the final chapter with particular reference to the timing of breeding and its success, chick growth, and the regulation of breeding population size. There are 22 pages of black and white photographs, numerous tables and figures in the text, and 34 appendices.

Two main features make this study of special interest. First it shows the value of true long-term research in seabirds and of study focused on a whole community level rather than on individual species. In addition, the Farallones seabirds provide useful evidence of the effects of extreme variation in the marine environment upon the birds' breeding ecology. Although many seabird studies, including those in Britain, ably demonstrate differences in breeding parameters between years, the underlying cause is rarely so clear-cut. The book will also be of special interest to those wishing to use data on the breeding biology of the Farallon species, provided in the detailed appendices, for comparative purposes. This research is compared by the authors to the classic review of the seabird bazaars of Novaya Zemlya and the Barents Sea published in Russian over 30 years ago. The Farallon Islands study provides a similar source of reference for anyone interested in seabird ecology.

Clare Lloyd

GASTON, A.J. 1992. *The Ancient Murrelet: a natural history in the Queen Charlotte Islands.* 249 pp. £17.00. T & A.D. Poyser, London. ISBN 0 85661 070 4.

Ever since its publication was announced I have been looking forward to reading Tony Gaston's monograph on the Ancient Murrelet. Having just finished it I can thoroughly recommend it, either to read cover to cover or just to dip into.

The book is divided into two parts, the first entitled rather grandiosely "A global glimpse of the Ancient Murrelet" provides a brief history of how the species became known to science and describes its present day distribution, status and breeding habitat. Although the habits of the Ancient Murrelet must have been familiar to the Aleuts, Haida and other native peoples of the area, until the last twenty years almost everything known about the species' biology came from the observations of Steller and Kraschenninikof and the accounts of a few American ornithologists. These early naturalists discovered that the adult birds visited their breeding burrows only at night and stayed out at sea during the day where they fed on small fish, molluscs and crabs. In 1894, Littlejohn described the most dramatic feature of Ancient Murrelet biology - the precocial departure of the chicks at night.

The second, and larger part of the book is devoted to Tony Gaston's own work on the species which was carried out on Reef Island, one of the Queen Charlotte Islands off the coast of British Columbia between 1984 and 1989. The text vividly conveys the trials and tribulations of getting to grips with this intriguing but easily disturbed seabird. Despite some initial setbacks, Tony Gaston and his co-workers have managed to amass a wealth of information on the breeding biology and behaviour of Ancient Murrelets and this is presented in seven highly readable chapters. As someone who never ceases to be

thrilled by the sight and sound of a big fledging night of Common Guillemots, I was fascinated by the section dealing with the chicks' journeys from the burrow, through the forest to their eventual reunion with their parents on the sea.

One of the main objectives of the study was the determination of the population dynamics of Ancient Murrelets on Reef Island and this involved a large scale ringing programme. The annual survival of breeding birds was found to be only 75%, much lower than that of any other alcid or pelagic seabird. Very few ringed chicks were recaptured back at the colony suggesting that philopatry was low. However, as the author acknowledges, these findings are based on only six years work and must be regarded as preliminary. It will take many more years before some aspects of inter-year variation and inter-island movement, become clear.

The Ancient Murrelet is one of only four species of seabird which produce young that are not fed at all in the nest but instead accompany their parents to the feeding area soon after hatching. In the penultimate and thought-provoking chapter, Tony Gaston uses the results from his study to evaluate some hypotheses for the evolution of this unusual departure strategy. He finds the argument that precocity evolved because of unpredictable feeding conditions unconvincing and instead proposes that the vulnerability of breeders, and probably chicks, to predation in the colony is a key factor. The book ends with a short and quite optimistic chapter on the conservation of the species. In 1990, the task of monitoring the Ancient Murrelet populations passed from Tony Gaston and his team to the Laskeek Bay Conservation Society, a group of local people committed to advancing conservation and environmental education. This monograph will surely be an invaluable source of inspiration and reference for this group.

The volume maintains the high standard of production expected of the Poyser series. It is enhanced by the fine jacket illustration by Ian Lewington and numerous delightful vignettes in the text by Ian Jones.

Sarah Wanless

## THE SEABIRD GROUP 1993

*The Seabird Group* was founded in 1966 to circulate news of work in progress on seabirds and to promote research. It is run by an elected Executive Committee and maintains close links with the three major British national ornithological bodies – the British Ornithologist's Union, the British Trust for Ornithology, and the Royal Society for the Protection of Birds. Membership (£10 per annum, £9 if paid by banker's order, £5 for students) is open to all with an interest in seabirds; for details please contact the Membership Secretary (address below) – payment by banker's order helps the Group.

*Current Executive Committee* The present Committee comprises: Chairman K. Taylor, Secretary M. Heubeck, Treasurer R.W. Furness, Membership Secretary S. Russell, Editor of *Seabird* S. Wanless, Newsletter Editor M. Tasker, also M.P. Harris and J. Sears.

*Newsletters and Meetings* Three Newsletters are circulated to members each year. They contain all sorts of items including reports on seabird conservation issues and research projects, news from seabird groups in other countries, book reviews, details of meetings, etc. The Newsletter Editor (address below) welcomes contributions from members. The usual venue for the Group's annual meeting is the BTO Ringing and Migration Conference at Swanwick, except when the Group holds its own conference, in which case the meeting is combined with that. Our conferences draw seabird workers from many countries to join in a forum of topical interest. In keeping with our desire to promote work in the field, practical manuals and guidelines evolve from the workshop sessions which cater for specialist topics within the conference theme.

*Seabird Group Grants* Each year the Group has some money available to help fund research conducted by members. All grant applications should be submitted to the Secretary by the end of February, and will be considered by the Executive Committee by the end of March. Successful applicants are required to submit a typed report, not exceeding 500 words, by the end of October of the same year for inclusion in the Newsletter. A full typed report (in triplicate) must be submitted by the end of the year.

*Seabird Colony Register* The Seabird Group has always sought to organise and implement national schemes involving the active participation of its membership, now standing at 350 members. The Group membership played a major role in the national Operation Seafarer survey whose results were published in 'The Seabirds of Britain and Ireland' (1974). The Group completed the Seabird Colony Register fieldwork in 1988, in cooperation with the Nature Conservancy Council, and the results were published in the book: 'The Status of Seabirds in Britain and Ireland' in 1991. This register was begun in 1985 to gather together all existing data on breeding seabird numbers in the British Isles, to bring our knowledge of their status up to date by detailed field surveys and to establish a computerised database which can be easily updated in the future. Although this round of survey work has been completed, it is important to continue monitoring of seabird breeding numbers: anyone eager to conduct counts on a regular basis should contact Paul Walsh, JNCC, 17 Rubislaw Terrace, Aberdeen AB1 1XE.

*Seabird Journal* In November 1984 the Group launched its new-look journal *Seabird*, numbered 7 in deference to its pedigree of Seabird Group Reports 1-6. Our priority is to maintain a high volume and quality of content and the current editor, Sarah Wanless, welcomes offers of papers (see Notice to Contributors, and address below). Members of the Seabird Group receive *Seabird* free of charge; additional copies to members, and any copies to non-members are £10 + 50p postage within the British Isles, £10 + £1 postage overseas. Postage overseas is by surface mail, unless the recipient can make prior provision for air mail. The subscription to Libraries is £15 per copy. To help reduce costs, overseas subscribers are kindly asked to make payment by international money order rather than by cheque. Back issues of *Seabird* 11, 12, 13 and 14 are



available at £5 + 50p per copy. There are no cost concessions for multiple orders of *Seabird* and postal charges are additive.

*Who to write to* While the Seabird Group maintains an accommodation address (c/o RSPB, The Lodge, Sandy, Bedfordshire SG19 2DL, England, UK), the following can be contacted directly, as appropriate. Please help the Group by enclosing a stamped addressed envelope for reply.

*Secretary* (general enquiries about the Group, seabird conservation matters, grants, etc.): Martin Heubeck, Mansefield, Dunrossness, Shetland ZE2 9JH, Scotland, UK.

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#### EDITORIAL ACKNOWLEDGEMENTS

The editor wishes to thank the following colleagues who have refereed manuscripts or provided help and advice. Apologies to anyone missed.

K. Camphuysen, E.K. Dunn, P.J. Ewins, C.J. Feare, A. Gosler, M.P. Harris, M. Heubeck, C.S. Lloyd, M. Marquiss, G.P. Mudge, P.A. Nuttall, M. Schaeffer-Pounds, M.L. Tasker, J.D. Uttley, P. Walsh and C. Wernham.

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On first mention a bird species should be designated by an English vernacular name drawn from *The Status of Birds in Britain and Ireland*, or from an authoritative faunistic work treating the appropriate region, followed by systematic binomial; author and date need be cited only in taxonomic papers. Thereafter only one name should be used, preferably the English one. Capitals should be used for the initial letters of all single words or hyphenated vernacular names (e.g. Great Black-backed Gull, White-eyed Gull) but not in a group name (e.g. gulls, terns). Trinomials should be used only if the subspecific nomenclature is relevant to the topic under discussion. Underlining is used for all words of foreign languages, including Latin, other than those which have been adopted into English. Underlining should also be used for phonetic renderings of bird vocalizations. Underlining is not needed for emphasis.

Measurements should be given in SI (International system of units), but if the original measurements were made in non-SI units, the actual values and units should be given, with SI equivalents inserted in parentheses at appropriate points. Measurements may be given in cm.

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COULSON, J.C. and WOOLER, R.D. 1976. Differential survival rates among breeding Kittiwake Gulls *Rissa tridactyla* (L.). *J. Anim. Ecol.* 45: 205-213.

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