THE STATUS OF BREEDING SEABIRDS IN MAINLAND NORWAY

ROBERT T. BARRETT[1], SVEIN-HÅKON LORENTSEN[2] & TYCHO ANKER-NILSSEN[2]

Barrett R.T., Lorentsen S-H. & Anker-Nilssen T. 2006. The status of breeding seabirds in mainland Norway. Atlantic Seabirds 8(3): 97-126. *Approximately 2.9 million pairs of 18 seabird species breed along the mainland coast of Norway. Of these, 1.4 million pairs breed along the Barents Sea coast and 1.3 million pairs along the Norwegian Sea coast. The commonest species are the Atlantic Puffin* Fratercula arctica *(1.7 million pairs), the Black-legged Kittiwake* Rissa tridactyla *(336,000 pairs) and Herring Gull* Larus argentatus *(233,000 pairs). Norway has a considerable responsibility for a large part of the world's seabirds as more than 10% of the total biogeographic population of all the present species breed on the Norwegian mainland, and seven of the twenty populations constitute (sometimes considerably) more than 25% of the world, Atlantic or European populations. While some species are increasing in numbers, those of the Atlantic Puffin, Black-legged Kittiwake, Common Guillemot* Uria aalge *and the northern subspecies of the Lesser Black-backed Gull* Larus fuscus fuscus *are all declining rapidly. It is feared that local populations of the Common Guillemot west of the North Cape will become extinct in the near future.*

[1]Department of Natural Science, Tromsø University Museum, NO-9037 Tromsø, Norway; [2]Norwegian Institute for Nature Research, NO-7485 Trondheim, Norway. E-mail: robb@tmu.uit.no

INTRODUCTION

The waters off the coast of Norway and in the Barents Sea are among the most productive in the world and are reputed to support *c*. 7.7 million pairs of breeding seabirds (or, when the non-breeding fraction is included, 26 million individuals) (Blindheim 1989; Sakshaug *et al*. 1994; Barrett *et al*. 2002). Previous studies reported that, of these, about 2.7 million pairs breed along the mainland coast of Norway north of the Arctic Circle and the remaining 5 million breed on Svalbard (which includes Bjørnøya), Franz Josef Land, Novaya Zemlya and the Russian mainland coast (Anker-Nilssen *et al*. 2000).

Marine production at all trophic levels is high and particularly favourable for seabirds off the Norwegian coast due to two major north-flowing ocean currents, the low saline Norwegian Coastal Current close to the shore and the North Atlantic Current, which transports warm, saline Atlantic water along the edge of the continental shelf. The two currents converge close to the coast of

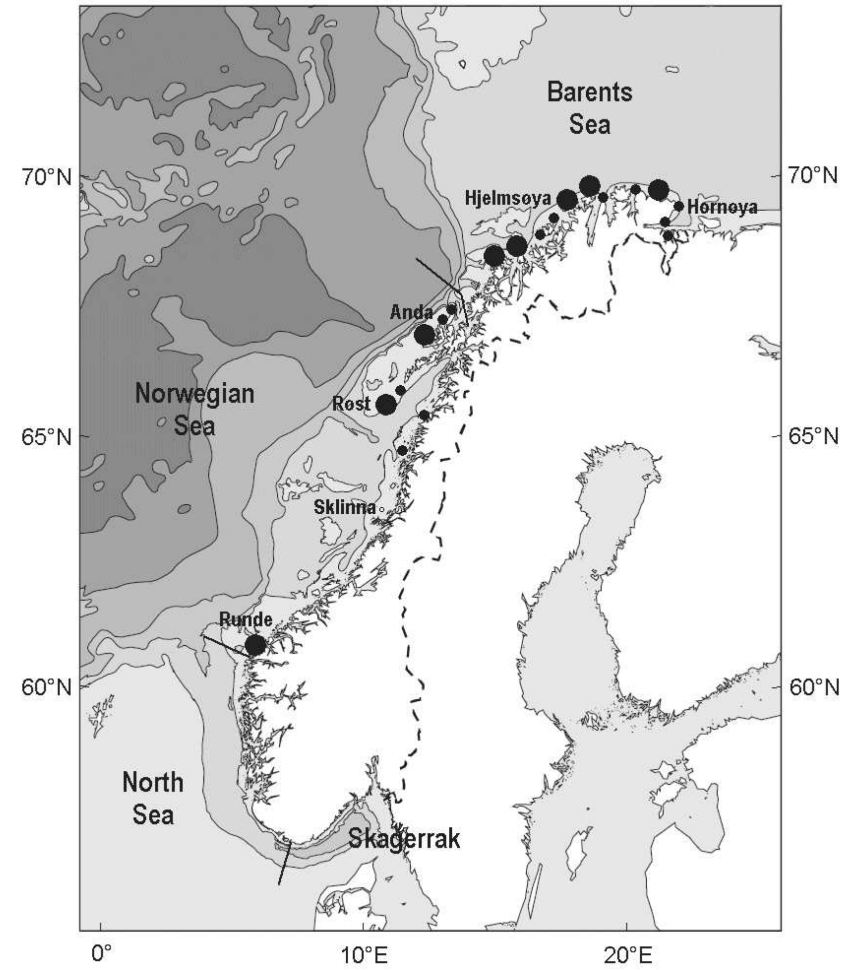


Figure 1. Map of Norway showing the positions of the largest seabird colonies (large symbols >100,000 pairs, small symbols 10,000 –100,000 pairs), the four major coastal ecoregions (Barents Sea, Norwegian Sea, North Sea and Skagerrak) and the six key seabird monitoring sites (Hornøya, Hjelmsøya, Anda, Røst, Sklinna and Runde). Bathymetric isolines are 200, 500, 1000, 2000 and 3000 m.

Figuur 1. Kaart van Noorwegen met de ligging van de grootste zeevogelkolonies (grote symbolen >100000 paar, kleine symbolen 10000–100000 paar), de vier belangrijkste ecoregio's (Barentszee, Noorse Zee, Noordzee en Skagerrak) en de zes belangrijke monitoringsgebieden voor zeevogels (Hornøya, Hjelmsøya, Anda, Røst, Sklinna en Runde). Bathymetrische isolijnen geven 200, 500, 1000, 2000 en 3000 m aan.

SW Norway but further north the mixing zone, where biological productivity is particularly high, follows the shelf edge relatively far offshore along most of the coast until it again approaches to within 10 km of the coast just north of the Lofoten Islands (Fig. 1). Nevertheless, where the continental shelf is relatively wide along the central Norwegian coast, deep channels and large, shallow banks cause gyres and mixing of water masses that increase retention time and boost productivity close to the shore (Rinde *et al*. 1998). Off the northernmost coast, production is further enhanced by increased tidal mixing and, in summer, continuous daylight.

Although Norwegian seabirds prey on a very wide variety of fish and invertebrates (Anker-Nilssen *et al*. 2000), two energy-rich, pelagic species have been highlighted as particularly important prey species - Norwegian spring-spawning Herring *Clupea harengus* (0- and 1-group, i.e. first- and second-year fish) and Capelin *Mallotus villosus* (all year classes) (Anker-Nilssen 1992; Barrett *et al*. 2002). Others, however, such as Saithe *Pollachius virens*, Haddock *Melanogrammus aeglefinus,* Cod *Gadus morhua,* and sandeels *Ammodytes* spp. may also comprise significant proportions of seabird diet but not always to the same degree and consistency as Herring and Capelin (Anker-Nilssen *et al*. 2000; Barrett 2002).

The mature stock of Norwegian spring-spawning herring resides primarily in the Norwegian Sea and the main part of the stock spawns in the southeastern Norwegian Sea in February-March. On hatching, very large numbers of larvae drift northwards with the coastal current reaching northern Nordland and Troms in late June or early July, by which time they have usually metamorphosed and reached an adequate size as prey for many seabirds. The young Herring drift onwards into the southern parts of the Barents Sea where they remain for 2-3 years before moving back into the Norwegian Sea to recruit into the spawning stock (Bakketeig *et al*. 2005). During their stay in the Barents Sea, the 1-group Herring (which are still small enough to be eaten by most seabird species) generally remain over coastal banks in the south and south-west and thus within the normal feeding range of breeding seabirds along the northeastern Norwegian coast (Loeng 1989).

The Capelin stock is restricted to the Barents Sea where it is the dominating pelagic species (Bakketeig *et al*. 2005). Because Capelin rarely grow longer than 14-15 cm, at which size they are mature, they are never too big to be eaten by seabirds. During the summer and autumn, the adult Capelin feed well offshore but move in late winter/early spring towards the coast of Finnmark to spawn. Here they become important food items during the early breeding period of seabirds. Some, however, also spawn in summer and are thus also available as food throughout most of the breeding season. Whereas in most years spawning occurs along much of the coast of Finnmark, it may in some years be restricted to isolated sites further west in Troms, or far to the east in Varanger and/or along the Kola Peninsula (Gjøsæter 1998).

Both the Herring and Capelin stocks have fluctuated greatly over the last 50-60 years (Fig. 2). The spawning stock of the herring declined rapidly in the 1950s and 1960s from more than 14 million tonnes in 1950 to near zero in the early 1970s but, after a fishing moratorium, recovered in the late 1980s reaching 6.7 million tonnes in 2004. The Capelin stock has also fluctuated greatly (Fig. 2) with minima in 1986/87, 1994/95 and 2003/04, and peaks in 1991/92 and 2000/01 (7.3 and 4.3 million tonnes respectively; Bakketeig *et al*. 2005).

Although Norway has long been recognized as being responsible for a significant part of the NE Atlantic seabird populations (Brun 1979), a comprehensive study of their numbers and population trends along the coast, the Norwegian Seabird Project, was not initiated at a national level until 1979 (Røv *et al*. 1984). Before this, the little knowledge concerning the population status and trends of Norwegian seabirds was based on total counts in a few selected colonies at irregular intervals. These were often limited in their accuracy, and their irregularity precluded detailed documentation of annual changes (Brun 1979). Large changes were, however, revealed, especially the overall decline of Common Guillemots *Uria aalge* at an alarming rate of approximately 5% *p.a.* between 1964 and 1974. At the largest colony, Hjelmsøya (Fig. 1), numbers decreased from 110,000 pairs in 1964 to 70,000 pairs in 1974 (Brun 1979).

The Norwegian Seabird Project ended in 1984 and some of the population data were summarized by Barrett & Vader (1984). It was, however, immediately followed by various mapping and monitoring projects, and much more detailed data concerning overall numbers, distribution and population trends have since been collected using international standards (e.g. Lorentsen 2005). Most of these data are now stored in The National Seabird Registry at the Norwegian Institute for Nature Research (NINA), Trondheim, from where all seabird monitoring is co-ordinated. The national monitoring programme for seabirds, which was established in 1988 and revised in 1996, now addresses population changes in 17 species of breeding seabirds along the coast, including the three key species (Atlantic Puffin *Fratercula arctica*, Black-legged Kittiwake *Rissa tridactyla* and Common Guillemot) and six key sites (Runde, Sklinna, Røst, Anda, Hjelmsøya and Hornøya, Fig. 1) (Røv *et al*. 1984; Anker-Nilssen *et al*. 1996; Lorentsen 2005). In 2005, the SEAPOP programme was launched (www.seapop.no). Its aim is to co-ordinate a long-term, comprehensive, standardised and cost-effective study of the most important aspects of seabird numbers, distribution, demography and ecology in Norwegian waters to satisfy the needs of the offshore industry, fisheries management, nature management, the scientific community and society at large in their

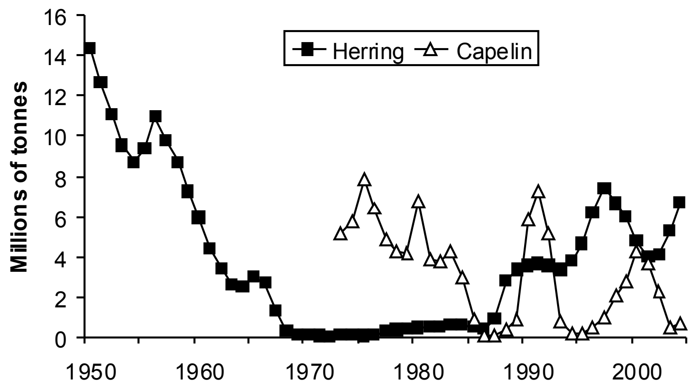


Figure 2.Changes in stock sizes of Capelin and mature spring-spawning Herring in the Norwegian and Barents Seas. (Source ICES 2005 a,b).
Figuur 2. Veranderingen in grootte van de 'voorraad' lodde en volwassen in het voorjaar paaiende haring in de Noorse Zee en de Barentszee (Bron: ICES 2005 a,b).

various roles as exploiters, protectors and researchers of the marine environment (Anker-Nilssen *et al.* 2005). The earlier established monitoring activities, which include the national programme and long-term studies of seabird ecology on Røst and Hornøya, will be continued as integrated parts of the SEAPOP programme (Anker-Nilssen *et al.* 2005).

Although documentation of numbers breeding in the north of the country (and the remainder of the Barents Sea) has recently been reviewed (Anker-Nilssen *et al.* 2000), and local and regional effects of large fluctuations in both the Herring and Capelin stocks on some seabird populations have been published (Barrett & Krasnov 1996; Anker-Nilssen *et al.* 1997; Durant *et al.* 2003), this paper presents the first overall synthesis of the status of breeding seabirds in mainland Norway since those published in 1984 (Røv *et al.* 1984; Barrett & Vader 1984).

METHODS

Species selection Although the scope of this review is restricted to marine birds "dependent on marine food items during most of the year", thereby including the Common Eider *Somateria mollissima*, other true seabirds such as the large gulls *Larus* spp. are also included. However, we exclude partially marine species such as the Red-breasted Merganser *Mergus serrator* and the Black-headed Gull

Larus ridibundus because they regularly occupy inland-breeding habitats. This review covers 18 species of seabird breeding along the Norwegian coast (Table 1), all but one (Black Guillemot *Cepphus grylle*) of which are included in the national monitoring programme for seabirds.

Population estimates Most population estimates are assessed from the latest data in the national seabird registry after taking into account the most recent population trends. Most of the data were collected during dedicated mapping projects, e.g. assessment studies in relation to petroleum activity, supplemented by *ad hoc* counts provided by researchers and others. Most counts were of apparently occupied nests (or, for Atlantic Puffins, burrows) and were considered equivalent to numbers of pairs, whereas guillemots were counted as numbers of individuals and early counts were converted to numbers of pairs using a conversion factor of 0.61 birds/pair (Bakken 1986). In general, the estimates are much more uncertain for the highly dispersed *Larus* and *Sterna* species (many of which also breed inland) than for the typical colony-nesting species (Northern Fulmar *Fulmarus glacialis*, Great Cormorant *Phalacrocorax carbo*, Black-legged Kittiwake and all auks except Black Guillemot).

In this review, we have divided the coastline into four ecoregions: the Barents Sea, the Norwegian Sea, the North Sea and Skagerrak (Fig. 1). The southwestern limit of the Barents Sea is defined as the continental slope which approaches the coast just north of Andøya (Blindheim 1989; Loeng 1989; Barrett *et al*. 2002; Skjoldal 2004). The boundaries between the Norwegian Sea, North Sea and Skagerrak are otherwise in accordance with Skjoldal (2004) and Moy *et al*.'s (2003) definition of Norwegian marine ecoregions. We have also allocated each species to a specific ecological group based on their primary feeding areas (pelagic or coastal) and behaviour (diving, plunge-diving, surface-feeding or benthic-feeding).

Population trends Population trends in colonies of different sizes were considered equally important when calculating overall trends for the different ecoregions, but colonies that contained less than 1% of the population within a specific region were omitted. The data were analysed in two steps using TRIM 3.4 for Windows (Pannekoek & van Strien 2005). TRIM (available at www.cbs.nl) is a program for analysis of time series with missing observations, and estimates the missing data based on site and year indices. In order to produce only one time series for each colony, TRIM was first used at the individual colony level to merge data for different study plots. The procedure was then repeated at the regional level by entering the different colony series to produce one time series for each ecoregion. At both levels, three models were run in TRIM: 1) No time effect (counts vary only across sites and not across time-points); 2) linear trend (site effect and a log-linear time effect); and 3) effects for each time-point (site effect and different effects for each time-point). Serial correlation between successive counts was assumed for all models. The

Table 1. Estimates of total numbers of pairs of seabirds breeding in four regions along the Norwegian coast in 2005. Ecological groups: P = Pelagic, C = Coastal, Su = Surface-feeding, Pd = Plunge diving, Di = Diving, Be = Benthic-feeding.
Tabel 1. Schattingen van aantal paar broedende zeevogels in vier regio's langs de Noorse kust in 2005. Ecologische groepen: P = pelagisch, C = kustgebonden, Su = oppervlakte foeragerend, Pd = stootduikend, Di = duikend, Be = benthos foeragerend.

	Estimated population size (pairs)						
						Total (rounded)	
	Ecological group	Barents Sea	Norwegian Sea	North Sea	Skagerrak	pairs	%
Northern Fulmar	P Su	100	7,500	1,500	20	9,000	0.3
Northern Gannet	P Pd	1,750	2,750	0	0	4,500	0.2
Great Cormorant[1]	C Di	10,000	20,000	0	0	30,000	1.0
Great Cormorant[2]	C Di	0	0	0	800	800	<0.1
European Shag	C Di	6,000	13,000	5,000	0	24,000	0.8
Common Eider	C Be	35,000	100,000	40,000	15,000	190,000	6.5
Great Skua	C Su	20	90	5	0	115	<0.1
Common Gull	C Su	10,000	75,000	30,000	20,000	135,000	4.6
Lesser Black-backed Gull[3]	C Su	< 300	*c.* 1,000	0	0	1,300	<0.1
Lesser Black-backed Gull[4]	C Su	0	*c.* 1,000	8,000	40,000	49,000	1.7
Herring Gull	C Su	100,000	100,000	13,000	20,000	233,000	8.0
Great Black-backed Gull	C Su	15,000	30,000	6,000	2,500	53,000	1.8
Black-legged Kittiwake	P Su	250,000	80,000	6,000	0	336,000	11.6
Common Tern	C Su	1,000	< 3,000	4,000	3,000	11,000	0.4
Arctic Tern	C Su	10,000	20,000	5,000	< 100	35,000	1.2
Common Guillemot	P Di	< 10,000	< 5,000	150	0	15,000	0.5
Brünnich's Guillemot	P Di	< 1,500	< 10	0	0	1,500	0.1
Razorbill	P Di	< 15,000	< 10,000	300	0	25,300	0.9
Black Guillemot	C Di	20,000	15,000	350	30	35,000	1.2
Atlantic Puffin	P Di	900,000	800,000	14,000	0	1,700,000	59.0
Total		1,385,670	1,283,330	133,305	101,450	2.9 mill.	

[1] *Phalacrocorax carbo carbo*, [2] *P. c. sinensis*, [3] *L. fuscus fuscus*, [4] *L. f. intermedius*

results from the model with the lowest AIC (Akaike's Information Criterion) value were selected. In order to treat all colonies equally, the TRIM indices for the best colony-specific models in the first run were scaled equally by setting the start year count at 100 before calculating the regional data series. Note that in a few colonies, both individuals and nests were counted with different results, for instance for common guillemot at Hjelmsøya where individuals (large decrease) and eggs (increase) are monitored on different plots (Lorentsen 2005). In such cases all count data were entered simultaneously into TRIM, in order to achieve a unified index for each colony.

Statistics for all trends were calculated by Monte-Carlo simulations. Using this method, the linear regression slope was estimated for the ln-transformed data set and compared with the corresponding slopes for 10,000 different randomised sequences of the same data values. The P value for a positive or negative trend was then computed as the fraction of the generated slopes that were greater or less than the real slope, respectively. Results of Monte Carlo simulations when n (here the number of census years) is small should be treated with great caution. When $n = 3$, only six permutations are possible and the lowest P value obtainable is 0.166 (1/6), while these numbers rise to 24 permutations and $P = 0.042$ when $n = 4$. Significance levels were chosen according to the recommendation by Anker-Nilssen *et al*. (1996) with $P < 0.1$, $P < 0.05$ and $P < 0.01$ indicating low, medium and high significance, respectively. The r^2 statistics were obtained by linear regression on ln-transformed data from TRIM, including the imputed values for missing years. Although fitting linear trends may mask short-term changes in numbers and no checks were made for density-dependent variation in trends between colonies (because most of the colonies of each species monitored were far apart and of similar magnitudes of size), we believe that the results offer an accurate representation of the status of each species.

RESULTS

Numbers and distribution An estimated 2.9 million pairs of seabirds breed along the coast of Norway with Atlantic puffins (1.7 million pairs) comprising nearly 60% of the total number (Table 1). Only four other species exceed 100,000 pairs: the Black-legged Kittiwake (336,000 pairs, *c*. 12% of total), Herring Gull *Larus argentatus* (233,000 pairs, 8%), Common Eider (190,000 pairs, 6%) and Common Gull *Larus canus* (135,000 pairs, 5%).

The distribution of breeding sites along the Norwegian coast is very uneven with >90% of the cliff-nesting species breeding north of the Arctic Circle, and >90% of all Norwegian seabirds breeding north of 62º N, i.e. along the coasts of the Norwegian and Barents Seas (Table 1). Furthermore, the

Barents Sea and Norwegian Sea seabird communities are dominated by pelagic-feeding species comprised mainly of diving (mostly Atlantic Puffins) and surface-feeding birds (mostly Black-legged Kittiwakes), whereas coastal species dominate the smaller communities in the south (Table 2). The proportions of coastal species within each region increase southwards, especially those of surface feeders (gulls and terns), which comprise only 10% of the Barents Sea seabird community but 84% of the Skagerrak community (Table 2). Coastal benthic feeders (represented by Common Eiders only) are also relatively important among the North Sea (30%) and Skagerrak (15%) seabird complements, although their absolute numbers are highest in the Norwegian Sea. Pelagic species are all but absent in Skagerrak except for *c*. 20 pairs of Northern Fulmar that breed near the border with the North Sea.

Table 2. The proportions (%) of seabirds breeding within each of the four marine ecoregions of the Norwegian coast according to feeding characteristics (area and behaviour).
Tabel 2. Het aandeel (%) broedende zeevogels in de vier mariene ecoregio's in Noorwegen verdeeld naar foerageerkarakteristieken (gebied en gedrag).

Feeding characteristic	Barents Sea	Norwegian Sea	North Sea	Skagerrak
Pelagic surface-feeding (P Su)	18	7	6	<1
Pelagic plunge diving (P Pd)	<1	<1	0	0
Pelagic diving (P Di)	67	64	11	0
Coastal surface-feeding (C Su)	10	18	50	84
Coastal diving (C Di)	3	4	4	<1
Coastal benthic-feeding (CBe)	2	8	30	15
Total	100	100	100	100

Population trends and species accounts Where available, summaries of monitoring effort and overall population trends from the start of the monitoring to 1995 and from 1996 to 2005 are presented for each ecoregion in Tables 3-6. Further details are given in the following species accounts.

SPECIES ACCOUNTS

Northern Fulmar Northern Fulmars first nested in Norway on Runde in the early 1900s, and the population has since spread to sites in Rogaland, Lofoten, Vesterålen and NW Finnmark. More than 95% of the Norwegian population breeds, however, along the Norwegian Sea and North Sea coasts, with

Table 3. Tabel 3.

Species	Monitored colonies	Monitoring period	Max years counted	Annual change (%) up to 1995	r^2	*P*- trend up to 1995	Annual change (%) 1996-2005	r^2	*P*- annual change
Northern Fulmar	1	1993-05	11				-2.5	0.13	n.s.
Northern Gannet	3	1961-05	23	16.8	0.92	< 0.01	12.8	0.98	< 0.01
Great Cormorant (*carbo*)	38	1983-05	21	4.2	0.42	< 0.1	6.6	0.86	< 0.01
European Shag	16*	1981-05	19	0.8	0.03	n.s.	9.9	0.85	< 0.05
Common Eider	BS/GS	2000-05	6				8	0.34	n.s.
Great Skua	1	1997-05	9				6.4	0.4	n.s.
Black-legged Kittiwake	49*	1980-05	24	-2.2	0.83	< 0.01	-6.4	0.94	< 0.01
Arctic Tern**	6-19	1989-05	17	-9.3	0.41	n.s.	-0.4	0.01	n.s.
Common Guillemot	2	1980-05	24	-14.8	0.7	< 0.05	10.3	0.99	< 0.01
Brünnich's Guillemot	2	1984-05	22	-14.3	0.35	< 0.1	-25.9	0.85	< 0.01
Razorbill	1	1996-05	8				-2.9	0.25	n.s
Atlantic Puffin	2	1980-05	23	2.4	0.88	< 0.01	1.9	0.48	< 0.05

* Including 12 shag and 47 kittiwake colonies in Sør-Varanger counted 10 times since 1966 (Barrett 2003), ** K.-B. Strann *pers. obs.*

Table 4. Tabel 4.

Species	Monitored colonies	Monitoring period	Max years counted	Annual change (%) up to 1995	r^2	*P*- trend up to 1995	Annual change (%) 1996-2005	r^2	*P*- annual change
Northern Fulmar	1	1997-05	9				-14.7	0.73	< 0.05
Northern Gannet	3	1946-05	30	9.8	0.84	< 0.01	-0.5	0.03	n.s.
Great Cormorant (*carbo*)	Many, AS	1974-05	23	2.4	0.68	< 0.01	-0.9	0.04	n.s.
European Shag	3	1975-05	22	1.6	0.32	< 0.1	9.9	0.68	< 0.01
Common Eider	GS+AS	1962-05	43	0.4	0.37	< 0.01	-0.5	0.03	n.s.
Great Skua	9	1998-05	5				10	0.88	< 0.05
Common Gull	24*	1989-05	10	17.4	0.45	< 0.1	14.2	0.52	n.s.
Lesser Black-backed Gull**	24*	1980-05	17	-5.4	0.81	< 0.01	-3.2	0.77	< 0.05
Herring Gull	21*	1989-05	10	-0.8	0	n.s.	15.7	0.77	< 0.05
Great Black-backed Gull	25*	1989-05	10	8.7	0.41	n.s.	0.8	0.27	n.s.
Black-legged Kittiwake	3	1979-05	24	-3.3	0.79	< 0.01	-7.8	0.9	< 0.01
Terns***	Many	2000-05	6				12.7	0.33	n.s.
Common Guillemot	2	1980-05	22	-5.7	0.73	< 0.01	-19.7	0.66	< 0.05
Razorbill	1	1997-05	7				-8.4	0.5	< 0.1
Atlantic Puffin	3	1979-05	27	-1.9	0.42	< 0.05	-2.1	0.74	< 0.05

* All relatively small colonies located in one restricted area (Sør-Helgeland)

**About half-and-half *Larus fuscus fuscus* and *L. f. intermedius*, but only colonies of *L. f. fuscus* monitored

***Common and Arctic tern

Opposite page: Table 3. Status of Norwegian seabirds breeding on the coast of the Barents Sea for which monitoring data exist. Population trends for the period from when the monitoring began and up to 1995 and for the last 10 years (1996-2005) are indicated. Boat surveys (BS) and ground level surveys (GS) cover relatively large areas.

Tegenoverliggende pagina: tabel 3. Status van Noorse zeevogels als broedvogel op de kust van de Barentszee. Populatie trends zijn aangegeven voor de periode vanaf het begin van de monitoring tot 1995, en voor de laatste 10 jaar (1996-2005). Scheepstellingen (BS) en inventarisaties vanaf land (GS) beslaan relatief grote oppervlakten.

Opposite page: Table 4. Status of Norwegian seabirds breeding on the coast of the Norwegian Sea and for which monitoring data exist. Population trends for the period from when the monitoring began and up to 1995 and for the last 10 years (1996-2005) are indicated. Aerial surveys (AS) and ground level surveys (GS) cover relatively large areas.

Tegenoverliggende pagina: tabel 4. Status van Noorse zeevogels als broedvogel op de kust van de Noorse Zee. Populatie trends zijn aangegeven voor de periode vanaf het begin van de monitoring tot 1995, en voor de laatste 10 jaar (1996-2005). Scheepstellingen (BS) en inventarisaties vanaf land (GS) beslaan relatief grote oppervlakten.

concentrations in Møre and Romsdal and Rogaland. The total population was estimated to be 1100 pairs in the early 1970s, 1850 pairs in 1982 and *c*. 7,000 pairs in the early 1990s (Brun 1979; Barrett & Vader 1984; Gjershaug *et al*. 1994). The present population is *c*. 9000 pairs, and Runde is still the largest colony (*c*. 5000 pairs, A.O. Folkestad *pers. comm.*). While there has been a steep decline in the one colony monitored on the Norwegian Sea coast (Røst; Table 4) and a recent decline at Runde and other small colonies nearby (A.O. Folkestad *pers. comm.*), there has been an expansion of the population further south resulting in large local increases in the small colonies at the south end of its range (mean 39% *p.a.*; Table 6). Why so few (<1000 pairs) Northern Fulmars breed in North Norway, where most of Norway's cliff-nesting seabirds breed, is perplexing, especially considering that well over 100,000 pairs breed on Svalbard (Anker-Nilssen *et al*. 2000).

Northern Gannet The Northern Gannet is also a relatively recent addition to the Norwegian seabird community. Since the establishment of the first Norwegian colony at Runde in 1947, numbers nesting in Norway have increased to *c*. 4100 pairs in 2005 with the establishment of colonies at 10 different sites in North Norway (Barrett & Folkestad 1996; RTB unpublished). However, due to a large turnover of colonies in the north in recent years, Norway has never hosted more than six extant gannetries at any given time.

Table 5. Tabel 5.

Species	Monitored colonies	Monitoring period	Max years counted	Annual change (%) up to 1995	r^2	*P*-trend up to 1995	Annual change (%) 1996-2005	r^2	*P*-annual change
Northern Fulmar	3-8	1973-98	23	20	0.9	< 0.01		0.41	
European Shag	18	1978-05	18	15.8	0.89	< 0.01	10.3	0.7	< 0.05
Common Eider	BS	2000-05	6				0.01	0	n.s.
Herring Gull	30	1978-05	7	-0.4	0.03	n.s.	-12.2	0.56	< 0.05
Terns*	22	1978-05	8	-9.3	0.96	< 0.01	-18.3	0.77	< 0.01

* Common and Arctic tern

Table 6. Tabel 6.

Species	Monitored colonies	Monitoring period	Max years counted	Annual change (%) up to 1995	r^2	*P*-trend up to 1995	Annual change (%) 1996-2005	r^2	*P*-annual change
Northern Fulmar	1	1995-05	11				38.7	0.74	< 0.05
Great Cormorant	8	1997-05	9				53.2	0.73	< 0.01
Common Eider	AS	1988-05	18	7.9	0.56	< 0.1	1	0.06	n.s.
Common Gull	26+ +M	1974-05	32	-6.1	0.66	< 0.01	-9.4	0.64	< 0.05
Lesser black-backed Gull*	24+	1974-05	32	9.8	0.69	< 0.01	-4.1	0.85	< 0.01
Herring Gull	26+	1974-05	32	7	0.8	< 0.01	-3.3	0.63	< 0.05
Great Black-backed Gull	28+ +M	1974-05	32	5.3	0.68	< 0.01	0.1	0	n.s.
Common Tern	24 +M	1974-05	32	-3.1	0.51	< 0.01	-12.5	0.7	<0.1

* *Larus fuscus intermedius*

Opposite page: Table 5. Status of Norwegian seabirds breeding on the coast of the North Sea and for which monitoring data exist. Population trends for the period from when the monitoring began and up to 1995 and for the last 10 years (1996-2005) are indicated. BS = boat surveys (covering large areas).

Tegenoverliggende pagina: tabel 5. Status van Noorse zeevogels als broedvogel op de kust van de Noordzee. Populatie trends zijn aangegeven voor de periode vanaf het begin van de monitoring tot 1995, en voor de laatste 10 jaar (1996-2005). Scheepstellingen (BS) beslaan relatief grote oppervlakten.

Opposite page: Table 6. Status of Norwegian seabirds breeding on the Skagerrak coast for which monitoring data exist. Population trends for the period from when the monitoring began and up to 1995 and for the last 10 years (1996-2005) are indicated. AS = aerial surveys (covering the whole coastline). M = many colonies surveyed within a defined coastal section.

Tegenoverliggende pagina: tabel 6. Status van Noorse zeevogels als broedvogel op de kust van het Skagerrak. Populatie trends zijn aangegeven voor de periode vanaf het begin van de monitoring tot 1995, en voor de laatste 10 jaar (1996-2005). AS = vliegtuigtellingen (gehele kustlijn). M = in een bepaalde kustsectie zijn vele kolonies geïnventariseerd.

While numbers have continued to increase at Runde (*c.* 2% *p.a.* since 1996, Lorentsen 2005), there have been contrasting population trends in colonies further north resulting in a stabilization of the population along the Norwegian Sea coast (Table 4). Two of the largest colonies in the Lofoten/Vesterålen region (Hovsflesa and Skarvklakken), for example, declined after the early 1990s and were abandoned in 2001 and 2003 respectively (RTB unpublished data). At the same time, two new colonies were established in neighbouring Great Cormorant colonies, partly as the result of movements of birds from Hovsflesa and Skarvklakken (Våge & Stenersen 2003). The abandonment of Hovsflesa and Skarvklakken has been attributed to increasing numbers of immature White-tailed Eagles *Haliaeetus albicilla* that have long been observed threatening and preying on both adult and young gannets (Barrett & Folkestad 1996; Våge & Stenersen 2003; RTB unpublished).

In the Barents Sea, numbers of Northern Gannets continued to increase (13% *p.a.*, Table 3) at least until 2005, partly due to the establishment of a new colony off the coast of Troms in 2001.

Great Cormorant Approximately 30,000 pairs of the nominate subspecies *P. c. carbo* breed in Norway, all north of 62º N and mostly along the Norwegian Sea coast (Table 1). This is a little more than 50% of the world population of *P. c. carbo* (57-58,000 pairs; Mitchell *et al.* 2004, Table 7) such that Norway is the most important area for this subspecies.

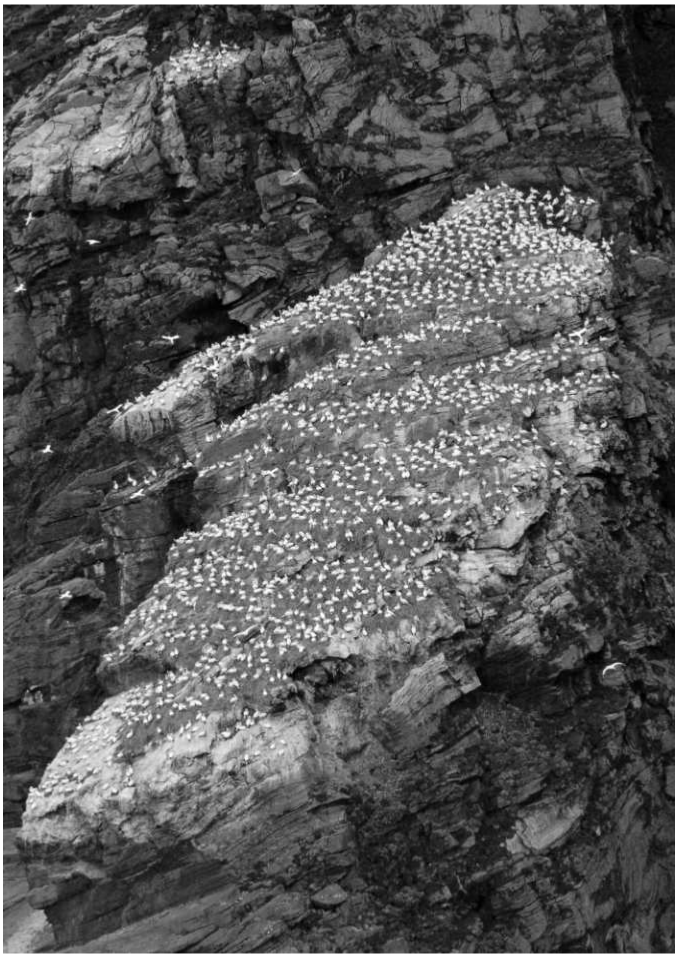
The Great Cormorant was described as being “quite rare” in Norway in the 1950s and 1960s (Gjershaug *et al.* 1994), but subsequent population estimates indicate a rapid increase in numbers. Barrett & Vader (1984) listed the Great Cormorant as a "declining species", but this was based on limited, local data from Troms. Røv & Strann (1987) published an estimate of 21,000 pairs based on counts made between 1982-1986, and drew attention to rapid increases in mid-Norway. Ten years later, Røv (1997) again reported an estimate of 21,400 pairs in 1995, followed by an estimated 26,650 pairs in 2003 (Røv *et al.* 2003). Since then, the Norwegian Sea population has remained stable (Table 4) while the Barents Sea population has continued to increase at an annual rate of 7% (Table 3).

The continental subspecies *P. c. sinensis* established a colony in Rogaland in 1996 and another in Østfold, east Skagerrak in 1997. The population in Østfold has since increased rapidly to *c.* 870 pairs in 2005 (though peaking at 990 pairs in 2004), distributed in seven colonies, and the same subspecies has also colonised Aust-Agder and Vest-Agder with some 200 pairs (Lorentsen 2005; C. Steel *pers. comm.*).

European Shag As for *P. c. carbo*, Norway is an important area for the European Shag with a population total of 24,000 pairs or *c.* 30% of the NE Atlantic population (*c.* 75,000 pairs; Table 7, Mitchell *et al.* 2004). The European Shag has a more southerly distribution than the Great Cormorant with 5,000 pairs breeding along the North Sea coast where no nominate Great Cormorants breed (Table 1).

Again, in common with the Great Cormorant (and again contrary to Barrett & Vader's (1984) pessimistic view), numbers of European Shags breeding in Norway have increased since the late 1970s, from estimates of *c.* 15,000 pairs in 1980-1984 (Røv *et al.* 1984) to the present 24,000 pairs.

While overall numbers increased by 10% *p.a.* between 1996 and 2005 in colonies monitored in the Barents, Norwegian and North Seas (Tables 3-5), there have been large inter-colony differences in population changes. For example, within the Norwegian Sea region, the population at Runde declined by *c.* 75% from *c.* 5000 pairs in 1975 to <1000 pairs in 2004 (Lorentsen 2005), but has since showed tendencies towards a recovery. At Sklinna, by contrast, numbers have increased by 8.3% *p.a.* between 1984 and 1995, and by 11.3% *p.a.* since 1996. The latter was partly due to the building of a new breakwater in 1999, but the trend was paralleled by a 14% *p.a.* increase at Ellefsnyken, Røst despite two or three seasons of presumed deferred breeding in the mid 1990s, when numbers declined from >450 breeding pairs in 1993 to 69 in 1995 and returning to *c.* 440 in 1996 (Lorentsen 2005; Anker-Nilssen & Aarvak 2006). Sklinna has now replaced Runde as one of Europe's largest colonies, with 2500 pairs in 2005.



The world's northernmost Northern Gannet Morus bassanus *colony at Storstappen, Gjesvær, Norway. The colony was established in 1987 and numbered 1250 pairs in 2005.* De meest noordelijk gelegen kolonie van Jan-van-gent ter wereld, in Storstappen, Gjesvær, Noorwegen. De kolonie is in 1987 gesticht en telde in 2005 1250 paar. *Rob Barrett, Tromsø University Museum.*

*Table 7. Importance of seabird populations in Norway as a proportion (%) of the relevant biogeographic areas to which they belong. Those areas marked * contain the world population, and those marked ** nearly the whole world population. Data for the biogeographic areas are from the species-specific tables of international importance in Mitchell* et al. *(2004), having corrected for the revised estimates for Norway. Data for Common Eider and Brünnich's Guillemot are from Anker-Nilssen* et al. *(2000).*

*Tabel 7. Belang van zeevogelpopulaties in Noorwegen, uitgedrukt als het aandeel (%) van de biogeografische regio's waartoe deze behoren. * deze regio's herbergen dewereldpopulatie; ** deze regio's herbergen vrijwel de gehele wereldpopulatie. Data voor de biogeografische regio's zijn afkomstig van de soortspecifieke tabellen 'of international importance' in Mitchell* et al. *(2004), gecorrigeerd voor de aangepaste schattingen voor Noorwegen. Data voor Eider en Dikbekzeekoet zijn afkomstig van Anker-Nilssen* et al. *(2000).*

	Norway	Biogeographic population		
	Pairs	%	Area	Pairs
Northern Fulmar	9,000	< 1	N. Atlantic	2.7-4.1 mill.
Northern Gannet	4,500	1	N. Atlantic*	400,000
Great Cormorant[1]	30,000	53-54	N. Atlantic*	57-58,000
Great Cormorant[2]	800	< 1	W. Palearctic	200-230,000
European Shag	24,000	31-32	N.E. Atlantic	75-77,000
Common Eider	190,000	10-13	Europe	1.5-2.0 mill.
Great Skua	155	< 1	N. Atlantic*	16,000
Common Gull	135,000	23-31	Europe**	430-590,000
Lesser Black-backed Gull[3]	1,300	8-9	Europe*	14-17,000
Lesser Black-backed Gull[4]	49,000	37-49	Europe*	100-131,000
Herring Gull	233,000	27-28	Europe	840-860,000
Great Black-backed Gull	53,500	46-49	Europe**	110-120,000
Black-legged Kittiwake	336,000	13-15	N. Atlantic	2.3-2.6 mill.
Common Tern	11,000	3-5	Europe	220-330,000
Arctic Tern	35,000	2-7	Europe+N. Atlantic	0.5-1.8 mill.
Common Guillemot	15,000	< 1	N. Atlantic	2.8-2.9 mill.
Brünnich's Guillemot	1,500	< 1	N. Atlantic	4.9-7.5 mill.
Razorbill	25,000	5	N. Atlantic*	530,000
Black Guillemot	35,000	9-13	N. Atlantic**	275-405,000
Atlantic Puffin	1,700,000	26-31	N. Atlantic*	5.5-6.6 mill.
Total	2.9 mill.	13-17		17-22 mill.

[1] *P.c. carbo*, [2] *P.c. sinensis*, [3] *L.f. fuscus*, [4] *L.f. intermedius*

In the Barents Sea region, there have also been considerable fluctuations in all colonies monitored, and the 10% *p.a.* increase since 1996 (Table 3) reflects primarily the rapid recovery of the population in one large colony (Lille

Kamøya), which collapsed between 1990 and 1994 (from 400 to four nests in the monitoring plots). In Sør-Varanger, eastern Finnmark, where 12 colonies have been recorded, there have also been large fluctuations but an overall decline from 110 pairs in 1975 to 15-20 pairs in 2002 (Barrett 2003).

Common Eider Common Eiders nest along the whole coast of Norway and the total population is roughly estimated to be 190,000 pairs. The only previous population estimate of 70,000-100,000 pairs was made in the early 1980s (Gjershaug *et al*. 1994), but it is difficult to determine if the present figure is due to a real population increase or simply better coverage of the population.

Surveys within the four ecoregions suggest that numbers have been more or less stable along the coast of the North and Norwegian Seas and possibly increasing along the Skagerrak and Barents Sea coasts. Common Eider data are, however, very sparse and numbers fluctuate greatly within given areas and colonies. In several smaller areas between Central Norway and the Lofoten area, local populations have declined severely since the early 1980's. This trend is observed both for breeding and wintering populations (Lorentsen & Nygård 2001; Lorentsen 2005). Common eiders were heavily hunted during the Second World War, and subsequent protective legislation and establishment of protected areas have enabled the population to recover. Interpretation of monitoring results is further complicated by recent declines in egg and down harvesting practices, changing levels of pollution within some monitoring areas, and the spread of feral American Mink *Mustela vison* to island colonies.

Great Skua The first mainland-breeding Great Skua nest was recorded in the Barents Sea region (at Loppa) in 1975 (Vader 1980) and the next breeding site was established at Runde in 1980 (Folkestad *et al*. 1980). The population has since spread to several sites near Runde, to Røst, Hjelmsøya, and Sværholt (Gjershaug *et al*. 1994), and was estimated to be *c*. 115 pairs in 2005. Most breed in at least eight sites along the northwestern coast of South Norway.

In common with the Northern Fulmar, it is interesting that few Great Skuas nest in North Norway (20-30 pairs) considering the large numbers breeding in the Northern Isles of Scotland, and the 500-1000 pairs breeding on Svalbard (Mitchell *et al*. 2004; H. Strøm *pers. comm*.).

Common Gull The Common Gull breeds over most of the country, often far inland (Lorentsen 1994) but only the coastal population is considered here. This is estimated to be approximately 135,000 breeding pairs, or about 25-30% of the European population (Table 7).

More than half of the Norwegian coastal population breeds within the Norwegian Sea ecoregion. Here the population, which is monitored only in a small area on the coast of Helgeland, Nordland, increased rapidly up to the mid-1990s, since when it has remained stable. Along the Skagerrak coast, however, the population has declined considerably since monitoring started in 1974, and in 2005 was only about 15% of its initial size. Numbers at the sites monitored along the Skagerrak coast have decreased by 9% *p.a.* since 1996, possibly due to movements to inland and urban areas (Lorentsen 1994).

Lesser Black-backed Gull Two subspecies of the Lesser Black-backed Gull breed in Norway. The nominate *fuscus* subspecies breeds in the north of the country, mainly along the Norwegian Sea coast (*c.* 1000 pairs), and *intermedius* breeds further south and in much larger numbers (*c.* 50,000 pairs) with 80% breeding in Skagerrak (Table 1).

Total numbers of *L. f. intermedius* in Norway have increased greatly since the 1960s when the population was estimated to be 5800 pairs (Barth 1968; Thingstad 1994). Monitoring has shown that despite large annual variations, this increase has continued at least in the Skagerrak at a rate of 1-5% *p.a.* since 1974. There seems, however, to have been a reversal of this trend in recent years with numbers falling again at a rate of 4% *p.a.* (Table 6). No monitoring data for *L. f. intermedius* exist for the North Sea and Norwegian Sea coastlines.

While there has been an overall increase in *L. f. intermedius*, the numbers of *L. f. fuscus*, have declined sharply over the last 40-50 years. Although no complete census was ever made, the population of this subspecies in the 1960s was probably at least 3000-4000 pairs (Haftorn 1971) but has since declined to *c.* 1300 pairs in 2005. In the colonies monitored in Nord-Trøndelag and Sør-Helgeland, the heart of their distribution, declines of 5-10% *p.a.* have been recorded since 1980 and up to the mid-1990s. Between 1996 and 2005, an increase in numbers has for the first time been recorded in Helgeland, which is considered to be the core area for this subspecies along the Norwegian coast. The situation for the subspecies is considered to be critical, and there is great need for expanding the monitoring of this subspecies into the northern part of its range and to include demographic parameters that might explain the status of the species.

Herring Gull Nearly a quarter of a million pairs of Herring Gulls are thought to breed in Norway more or less evenly distributed along the whole coastline (Table 1). This is similar to Brun's (1979) estimate of 260,000 pairs, but higher than Gjershaug *et al.*'s (1994) figure of 150,000-200,000 pairs. An estimate of 260,000 pairs constitutes the largest national population in western Europe; only Russia hosts larger numbers with *c.* 800,000 pairs (Mitchell *et al.* 2004).

Despite this large population, few Herring Gull colonies are monitored regularly, and little can be said about overall population trends. The few colonies that are monitored (in Skagerrak and on the Norwegian Sea coast) have increased in size over the last 16-30 years. They constitute, however, a very small part of the total population, and an expansion of the monitoring programme to increase the coverage should be considered.

Great Black-backed Gull In common with the Herring Gull, the Great Black-backed Gull breeds along the whole Norwegian coastline and the total population is estimated to be around 50,000-55,000 pairs (Table 1). This is slightly higher than Brun's (1979) and Gjershaug *et al*.'s (1994) figure of 40,000 pairs, and constitutes *c*. 50% of the European population and *c*. 30% of the world's population (170,000-180,000 pairs; Table 7, Mitchell *et al*. 2004). Current monitoring is limited to a few sites only, but it seems that numbers in the Norwegian Sea have been relatively stable since 1989, whereas Skagerrak numbers increased by 5% *p.a.* from the mid-1970s to the mid-1990s, and have since remained stable (Tables 4 and 6).

Black-legged Kittiwake The total Black-legged Kittiwake population in Norway is *c*. 336,000 pairs (13-15% of the North Atlantic population of 2.3-2.6 million pairs; Table 7, Mitchell *et al*. 2004), with almost all breeding along the coasts of the Norwegian and Barents Seas, and none in the Skagerrak. The largest colony is at Syltefjord, where the population was estimated to be 140,000 pairs in 1989 (Stougie *et al*. 1989). Unfortunately, very few colonies have been censused in more recent years.

Numbers of Black-legged Kittiwakes increased in North Norway at a rate of *c*. 1% *p.a.* in the 1960s and 1970s, and this increase continued into the early 1980s, at least in eastern Finnmark where the increase was as high as 4-8% *p.a.* in 1970-1983 (Brun 1979; Krasnov & Barrett 1995; Barrett 1985). There is also evidence of population increase in Troms and Vesterålen during the same time period (Bleiksøya 2000 pairs 1974, 5800 pairs in 1993, RTB unpubl. data). Since 1980, when the total Norwegian population was estimated to be about 500,000 pairs (Barrett & Vader 1984), numbers in all monitored colonies in Norway have declined significantly at rates varying between 1-5% *p.a.* Furthermore, there is evidence that the rate of decline has accelerated since the mid-1990s, up to 10-15% *p.a.* in some colonies (Barrett 2003; Lorentsen 2005) resulting in average decreases of 6% *p.a.* in the Barents Sea colonies and 8% *p.a.* in the Norwegian Sea colonies (Tables 3 and 4). Numbers of apparently occupied nests in monitoring plots on the key sites Runde, Vedøya (Røst), Hjelmsøya and Hornøya decreased by 75%, 50%, 75% and 50% respectively between the early 1980s and 2005 (Lorentsen 2005).

Little is known about the causes of this decline, but Brun (1979), Furness & Barrett (1985) and Krasnov & Barrett (1995) have reported Capelin to be the preferred food of Black-legged Kittiwakes breeding in East Finnmark, and have suggested that large Capelin stock fluctuations (including several collapses) in the Barents Sea (Gjøsæter 1998) may be having negative effects on the population (Barrett 2007). There is also evidence that increasing harassment from White-tailed Eagles in many colonies along the whole coastline has caused repeated local breeding failures and declines in Black-legged Kittiwake numbers (Barrett 2003; Anker-Nilssen & Aarvak 2006; *pers. obs.*). For example, on Bleiksøya, where White-tailed Eagles continually patrol the cliff face causing the Black-legged Kittiwakes to repeatedly fly out in panic, the kittiwake population has declined from 5800 pairs in 1993 to *c.* 600 pairs in 2005 (RTB *pers. obs.*).

Common and Arctic Tern Common and Arctic Terns breed over most of the country, also inland (Gjershaug *et al.* 1994), but only coastal populations are considered here. Both species breed along the whole mainland coast with most Common Terns (total population *c.* 11,000 pairs) in the south and most Arctic Terns (total population *c.* 35,000 pairs) in the north (Table 1). Both figures must be considered as very approximate estimates, but are in the same order of size as those suggested by Gjershaug *et al.* (1994; 10,000-20,000 pairs and 40,000 pairs respectively).

The population status of terns in Norway was recently summarized by Lorentsen (2006). Arctic Terns are monitored in 6-19 colonies within a small area in the Barents Sea where numbers were relatively stable in 1989-2005 (Table 3, K.-B. Strann *pers. comm.*). Along the Norwegian Sea coast, numbers (mostly of Arctic Terns) have been stable since 2000 (Table 4), whereas they have declined considerably in the North Sea (mostly Common Terns) and Skagerrak (Common Terns) at rates of 3-18% *p.a.* (Tables 5 and 6). The severe negative trends for terns in the North Sea and Skagerrak have persisted for the last 25 years and are probably a result of food shortage (Lorentsen 2006).

Common Guillemot The present population of Common Guillemots in mainland Norway is *c.* 15,000 pairs and thus less than 0.5% of the total North Atlantic population (2.8-2.9 million pairs; Mitchell *et al.* 2004). This is a very marked reduction since the first population estimate of 120,000-160,000 pairs, estimated in the 1960s (Brun 1969), and is most likely mainly the result of drowning in fishing gear, hunting and food shortages. West of the North Cape (Barents Sea), the annual drowning of breeding adults during the long-line and drift-net fisheries for Atlantic Salmon *Salmo salar* was probably the most significant single factor causing declines in what were once the largest colonies in Europe in the 1960s and 1970s (Brun 1979; Strann *et al.* 1991). These fisheries were banned in the early 1980s and 1989 respectively, but birds are still sometimes reported drowned in nets set for cod. Some colonies have declined so much that they may now be on the verge of extinction with seemingly too few pairs remaining for the colonies to be viable. Although drowning in fishing gear is now considered a minor threat to adult birds, numbers along the Norwegian Sea coast (Table 4) and at Hjelmsøya (one of the two colonies monitored in the Barents Sea) continue to fall steeply (98-99% declines at Vedøy in Røst and at Hjelmsøya between the early 1980s and 2005). While the breakdown of the social structure of the colonies (with single or very few birds on individual breeding shelves) is thought to contribute to the further decline, there is now compelling evidence that the present large population of White-tailed Eagles (which has gradually recovered since it was legally protected in 1968) is exacerbating the situation, resulting in some populations (e.g. Røst, Bleiksøya, Hjelmsøya) being forced to breeding under cover, for example in large cracks or stone screes, to avoid predation. Although still poorly covered by existing monitoring, birds breeding in such habitats are much more productive than those on exposed cliff ledges (*pers. obs.*).

While colonies west of the North Cape declined in the 1970s and 1980s, numbers seem to have increased during the same period further east (Vader *et al.* 1990; Krasnov & Barrett 1995). Between 1986 and 1987, however, very large (up to 80-85%) declines were recorded in all colonies in the Barents Sea and northern Norwegian Sea as a result of a collapse in the Capelin stocks. This caused a mass mortality of adult Guillemots during the winter of 1986/87 and a near total breeding failure in 1987 (Vader *et al.* 1987, 1990). Subsequent monitoring on Hornøya has, however, revealed a rapid recovery at 11% *p.a.* (Barrett 2001; Lorentsen 2005), perhaps partly due to both many years of high reproductive success and to immigration of birds from abroad, for example Shetland (RTB pers. obs). The recent increase at Hornøya outweighs the simultaneous decline at the second Barents Sea colony, Hjelmsøya, resulting in a (somewhat misleading) overall 10% *p.a.* increase for the region (Table 3).

The population development of the Common Guillemot in the Norwegian Sea is similar to that in the western Barents Sea (Table 4). Although three colonies (Runde, Sklinna and Vedøy in Røst) are monitored regularly, the results from Sklinna were not used in the present analysis because this is a relatively small and atypical colony that increased from only three pairs in 1983 to *c.* 50 pairs in 1999 and *c.* 400 pairs in 2005. This increase (an average 19% *p.a.* for the whole period 1980-2005, 35% *p.a.* in the period 1996-2005, Lorentsen 2005) is unlikely to have taken place without extensive immigration.

Brünnich's Guillemot Brünnich's Guillemots were first recorded breeding in Norway in 1964 (Brun 1965), but were almost certainly present in colonies before then. Today they breed in small numbers on at least Hjelmsøya, Gjesvær, Syltefjord and Hornøya/Reinøya, and the total population is in the order of 1500 pairs, with few or none south of the Barents Sea area (Table 1). Little is known about population trends, but it seems that after a steep decline in 1986/87 similar to that of Common Guillemots (Vader *et al.* 1990), numbers have further decreased west of the North Cape (on Hjelmsøya, Table 3), but are recovering east of the cape. This is certainly true for Hornøya, where the population doubled to *c.* 600 individuals between 1987 and 1996, since which it seems to have stabilized (RTB unpubl. data).

Razorbill Most of the Razorbills breed in the large seabird colonies in the Norwegian and Barents Sea regions and the total population is roughly estimated to be 25,000 pairs. Although this is only 5% of the world's population, Norway hosts an important proportion of the subspecies *A. t. torda*, with *c.* 30% of the 80,000 pairs in NW Europe (Mitchell *et al.* 2004). The Norwegian population estimate is, however, of doubtful accuracy, and present population trends are not well-known (Tables 3 and 4). There was, however, an increase on Hornøya between 1967 (65 pairs) and 1980 (*c.* 200 pairs), and this seems to have continued (Barrett & Vader 1984; RTB *pers. obs.*)

Black Guillemot An estimated 35,000 pairs of Black Guillemot breed along the whole coast of Norway, all but a few hundred of them breeding in the Norwegian and Barents Sea regions. Although there has been no recent monitoring of population trends, large local declines were reported in southwestern Norway in the 1970s and 1980s, and in North Norway (by >50%) since 1930, all probably due to the spread of feral American Mink (Barrett & Vader 1984; Gjershaug *et al.* 1994).

Atlantic Puffin Approximately 1.7 million pairs of Atlantic Puffin breed in Norway today (Anker-Nilssen 1991, updated for more recent trends), representing 25-30% of the world population (5.5-6.6 million pairs, Mitchell *et al.* 2004). Only a very small fraction (comprising 14,000 pairs) breeds on the North Sea coast and none in the Skagerrak. A number of estimates of the total breeding population of Atlantic Puffins in Norway have been made over the years, including figures of 1.25 million pairs in the mid-1970s and *c.* 2 million pairs in the early 1990s (Brun 1979; Anker-Nilssen 1991; Gjershaug *et al.* 1994). Since these two estimates, based on both direct and indirect counts (Brun 1979), much effort was directed towards conducting detailed, accurate counts of apparently occupied burrows in most of the large colonies, with large discrepancies being revealed. For example, at Gjesværstappan, a census conducted in 1991 resulted in an estimate of more than 400,000 pairs (TAN unpubl. data), whereas Brun's (1979) estimate in 1973 was only 18,000 pairs. In Vesterålen, counts made of four colonies (Fuglenyken, Måsnyken, Frugga and Bleiksøya) in 1988-1990 were all two to four times higher than estimates made 15-20 years earlier (RTB unpubl. data). Similarly, the population estimate at Røst increased from Brun's (1979) 700,000 pairs in 1964 to almost 1.5 million pairs in 1979 (Anker-Nilssen & Røstad 1993; Anker-Nilssen & Øyan 1995). Such differences are not considered to be a result of population increases but rather increases in survey effort.

Since 1980, the Norwegian Sea population of Atlantic Puffins has declined greatly (Table 4). This is largely because the Røst population, which, although still probably the largest seabird colony in mainland Europe, declined by 70% to an estimated 433,000 pairs in 2005 (Anker-Nilssen & Aarvak 2006). The decline has primarily been caused by repeated breeding failures, but also perhaps by reduced adult survival caused by the collapse in the spring-spawning herring stock in the late 1960s (e.g. Anker-Nilssen 1992; Anker-Nilssen *et al.* 2003; Durant *et al.* 2003; Anker-Nilssen & Aarvak 2006). Whereas numbers have decreased at Røst, the colony at Runde (*c.* 100,000 pairs), which is much less dependent on herring and more on sandeels and gadoids as prey (Barrett *et al.* 1987), increased by nearly 50% between 1980 and 1995, after which it decreased again by 1.6% *p.a.* over the next decade (Lorentsen 2005).

The Barents Sea colonies, where Atlantic Puffins have access to other prey such as Capelin and sandeels, have either been stable (at Gjesvær since 1997) or are increasing slightly (at Hornøya since 1980; Table 3). There are, however, indications that puffins in eastern Finnmark, where numbers have increased since the 1960s (Barrett & Vader 1984; Barrett 2001; Table 3), have recently suffered from a gradual deterioration in food conditions (Barrett 2002). If this continues, one might expect a reversal in the present positive population trend.

DISCUSSION

Internationally important populations In this review, about 2.9 million pairs of seabirds are estimated to breed along the coast of mainland Norway. Of these, 1.4 million (48%) and 1.3 million pairs (42%) breed within the Barents Sea and Norwegian Sea ecoregions respectively, confirming Anker-Nilssen *et al*'s (2000) earlier estimate of 2.7 million pairs. Norway, therefore, has a considerable responsibility for a large part of the world's seabird community (Table 7). More than 10% of the total biogeographic population of all the present species breed on the Norwegian mainland, and seven of the 20 populations considered in this review are very important in that they constitute (sometimes well) over 20% of the world, Atlantic or European populations (Table 7). The population of Great Cormorant (subsp. *carbo*) is the only one that exceeds 50% of the world population, whereas those of European Shag, Common Gull, Lesser Black-backed Gull (subsp. *intermedius*), Herring Gull, Great Black-backed Gull and Atlantic Puffin all comprise between 25 and 50% of their respective biogeographic populations and should thus be considered species of extra international responsibility (>25% of biogeographic population, DN 1999). Several other populations, including the Lesser Black-backed Gull (subsp. *fuscus*), Common Guillemot, and Black-legged Kittiwake, would probably also have met this criterion only a few decades ago, had they not declined so severely in recent years.

In recognition of this, Norwegian authorities have established many nature reserves and sites of special protection, encompassing of the largest colonies of cliff-nesting seabirds and islet colonies of Great Cormorants and Northern Gannets. Harvesting of eggs and down (from Common Eider and *Larus* spp.), and hunting (of Great Cormorant, European Shag, Common Eider, Great Black-backed, Herring and Common Gulls) are still permitted, but under strict restrictions that are subjected to regular revision (see www.lovdata.no/for/sf/md/td-20020211-0149-0.html#1 for the current regulations). There is, however, a need for improved population surveillance of Norwegian seabirds. Whereas most of the internationally important populations are included in the present monitoring project, almost nothing is known about the population trends of the Great Black-backed and Herring Gulls. Further priority should thus be accorded to monitoring these species, and to expanding the monitoring of the Lesser Black-backed Gull *intermedius* and Common Gull to include populations from the North Sea (both species) and Barents Sea (Common Gull) coasts.

Population trends The Northern Fulmar, Northern Gannet, Great Skua and *sinensis* subspecies of the Great Cormorant all established populations in Norway during the 1900s (1920, 1946, 1975, and 1996 respectively), and numbers were still increasing at the end of the century throughout much of their ranges. There are, however, indications that the increase in Northern Gannet numbers has reached a plateau, with very variable trends in the northern parts of the Norwegian Sea. Other populations with overall positive trends are the nominate subspecies *carbo* of the Great Cormorant, the European Shag and the *intermedius* subspecies of the Lesser Black-backed Gull. With the exception of the European Shag, which has declined recently in Britain and Ireland (Mitchell *et al.* 2004), these are all taxa that occur in large and increasing numbers and whose breeding range in western Europe is expanding. In several instances, this is a result of a general recovery after heavy persecution in the late 1800s and early 1900s through increased protection and increased access to non-natural food sources (e.g. garbage, offal and fish discards).

Although the Atlantic Puffin population has declined significantly in the centre of its distribution, it has increased at the southern and northern limits of its range in Norway. The increase in the south mirrors the 13% increase documented in Scotland and the near doubling of the English population between the late 1980s and 2000 (Mitchell *et al*. 2004). Whether the large differences between Brun's (1979) population estimates in the late 1960s and the more recent counts are due to real population increases or a simple refinement in counting methods (probably the latter) is unknown, but as the Herring stock collapsed more than a decade prior to the onset of the present monitoring schemes, a very large increase in puffin numbers seems unlikely, at least in colonies on the Norwegian Sea coast (Anker-Nilssen 1992). There are now indications that the recovery of the Norwegian spring-spawning herring stock (Fig. 2) has improved the feeding conditions and breeding success of Atlantic Puffins at Røst to an extent that might allow a slow recovery of this important population (Anker-Nilssen & Aarvak 2006).

The recent decline in the Black-legged Kittiwake population after a long period of steady population increase throughout its range is unexpected, and what may be an acceleration in this decline rate is disquieting. While deteriorating feeding conditions through a decrease in the availability of Capelin and Herring is a possible cause for the declines on Hornøya (Barents Sea) and on Røst (Norwegian Sea) respectively (Anker-Nilssen *et al*. 1997, Barrett 2007), little is known about possible causes elsewhere in Norway. Disturbance by White-tailed Eagles may have been a contributing factor over large areas of North-Norway in the last few decades (e.g. Anker-Nilssen & Aarvak 2006; *pers. obs.*), but the fact that numbers have decreased outside the range of eagles (e.g. in Britain and Ireland where numbers have declined by 23%, including a 69% decline in Shetland, in the same time period) suggests that other causes may be important. The decline in Britain has been attributed to changes in oceanographic conditions resulting in changes in the distribution and stocks of key prey fish species, and decreases in breeding success, body condition and survival of adult birds (Frederiksen *et al*. 2004).

Of considerable concern is the overall >95% decline in the Common Guillemot population in North Norway since the 1960s and the possible future extinction of what were important Common Guillemot colonies in the European context. There are signs of a recovery of the species east of the North Cape, but to the west some shelf-nesting colonies are close to extinction. This decline, originally a result of direct and indirect human pressure (fisheries), but recently a breakdown of social structure exacerbated by harassment from White-tailed eagles is in sharp contrast with the recent large, overall increase in numbers of Common Guillemots in Britain, a major breeding region with >30% of the North Atlantic population (Mitchell *et al.* 2004) where none of these negative factors has been a significant problem.

Similarly, the critical situation for the northern subspecies of the Lesser Black-backed Gull *L. f. fuscus* requires immediate investigation. Not only are numbers dropping in Norway, but the subspecies is now absent from the Kola Peninsula and has declined at a rate of 8% p.a. between 1986-2002 in Finland where the present population is around 5000-10,000 pairs (Hario *et al.* 1998; Anker-Nilssen *et al.* 2000). Only in a few colonies in Onezhski Bay in the southern White Sea has there been some respite with an increase at least until the early 1990s when the population was estimated to be c. 1600 pairs (Anker-Nilssen *et al.* 2000). The cause(s) of the decline are unknown but may be related to food shortages during the breeding season or, as proposed for the Finnish population, high chick mortality caused by elevated levels of DDE picked up in the wintering areas in East Africa (Strann & Vader 1992; Anker-Nilssen *et al.* 2000; Bakken *et al.* 2003; Hario *et al.* 2004).

There is also a general concern about deteriorating conditions for seabirds in the North Sea where breeding failures among a number of species have been recorded recently, the possible population effects of which have not yet been discerned. This may be exacerbated beyond national scales in the light of the far-reaching and, as yet, largely unknown physical and biological consequences of climate change at different marine trophic levels, including the biological and distributional responses of important seabird prey such as sandeels, Herring and Capelin (Arnott & Ruxton 2002; Rose 2005a,b). How and through which mechanisms climate change, the resulting (and other) changes in the fishing industry (with its direct and indirect effects on fish stocks), and other anthropogenic activities will affect Norwegian seabird populations cannot readily be predicted (e.g. Durant *et al.* 2004).

ACKNOWLEDGEMENTS

The nationally organised monitoring of seabirds in Norway in the study period was funded by the Directorate for Nature Management (and its predecessors), but some data series were collected by independent initiatives from different County Governor Administrations, local groups of the Norwegian Ornithological Society and private individuals. Data in 2005 were also collected as part of the Norwegian SEAPOP project (Anker-Nilssen et al. 2005, www.seapop.no). We thank all fieldworkers, far too numerous to mention by name, who for decades have spent days and weeks in the field, often under appalling conditions, and have then contributed data to the National Seabird Registry and the National Seabird Monitoring Programme. Without them, a description of the status of the Norwegian seabird populations would be complete guesswork. This review was financed by the Norwegian Institute for Nature Research and Tromsø University Museum.

STATUS VAN BROEDENDE ZEEVOGELS OP HET VASTELAND VAN NOORWEGEN

Langs de kust van het vasteland van Noorwegen broedt ongeveer 2,9 miljoen paar verdeeld over 18 soorten zeevogels. Hiervan broedt 1,4 miljoen paar langs de Barentszee en 1,3 miljoen paar langs de Noorse Zee. De algemeenste soorten zijn Papegaaiduiker *Fratercula arctica* (1,7 miljoen paar), Drieteenmeeuw *Rissa tridactyla* (336.000 paar) en Zilvermeeuw *Larus argentatus* (233.000 paar). Noorwegen heeft een grote verantwoordelijkheid voor een groot deel van 's werelds zeevogels, aangezien meer dan 10% van de totale biogeografische populatie van de aanwezige soorten op het Noorse vasteland broedt en zeven van de twintig populaties bedragen (soms zelfs aanzienlijk) meer dan 25% van de wereld-, Atlantische of Europese populatie. Sommige soorten nemen in aantal toe. Papegaaiduiker, Drieteenmeeuw, Zeekoet *Uria aalge* en de noordelijke (onder)soort van de Kleine Mantelmeeuw *Larus (fuscus) fuscus* nemen alle in rap tempo af. Gevreesd wordt dat lokale zeekoetpopulaties ten westen van de Noordkaap in de nabije toekomst zullen uitsterven.

REFERENCES

Anker-Nilssen T. 1991. Taksering av lunde i risikoområdet for midt-norsk sokkel. In: Børresen J.A. & Moe K.A. (eds.) AKUP Årsrapport 1990. Sect. I. Ministry of Oil and Energy, Oslo: pp. 13-18.

Anker-Nilssen T. 1992. Food supply as a determinant of reproduction and population development in Norwegian puffins *Fractercula arctica.* Dr. Scient thesis, Univ. Trondheim.

Anker-Nilssen T., Aarvak T. & Bangjord G. 2003. Mass mortality of Atlantic Puffins *Fratercula arctica* off Central Norway, spring 2002: causes and consequences. Atlantic Seabirds 5: 57-71.

Anker-Nilssen T. & Aarvak T. 2006. Tidsseriestudier av lunde og andre sjøfugler i Røst kommune, Nordland. Hekkesesongene 2004 og 2005. NINA Rapport 133, Trondheim.

Anker-Nilssen T., Bakken V., Strøm H., Golovkin A., Bianki V. & Tatarinkova I.P. (eds.). 2000. The Status of Marine Birds Breeding in the Barents Sea Region. Norsk Polarinst. Rapport Serie no. 113. Tromsø.

Anker-Nilssen T., Barrett RT., & Krasnov J.V. 1997. Long- and short-term responses of seabirds in the Norwegian and Barents seas to changes in stocks of prey fish. In: Anon. (ed). Proc. Forage Fishes in Marine Ecosytems. Proc Lowell Wakefield Fisheries Symp. University of Alaska Sea Grant College Program, Report No. 97-01: 683-698.

Anker-Nilssen T., Bustnes J.O., Erikstad K.E., Lorentsen S.-H., Tveraa T., Strøm H. & Barrett R.T. 2005. SEAPOP. Et nasjonalt sjøfuglprogram for styrket beslutningsstøtte i marine områder. NINA Rapport 1, Trondheim

Anker-Nilssen T., Erikstad K.E. & Lorentsen S.H. 1996. Aims and effort in seabird monitoring: an assessment based on Norwegian data. Wildl. Biol. 2: 17-26.

Anker-Nilssen T. & Røstad O.W. 1993. Census and monitoring of Puffins *Fratercula arctica* on Røst, N. Norway, 1979-1988. Ornis Scand. 24: 1-9.

Anker-Nilssen T. & Øyan H.S. 1995. Hekkebiologiske langtidsstudier av lunder på Røst. NINA Fagrapport 15, Trondheim.

Arnott S.A. & Ruxton, G.D. 2002. Sandeel recruitment in the North Sea: demographic, climatic and trophic effects. Mar. Ecol. Prog. Ser. 238: 199-210.

Bakken, V. 1986. A method for assessing changes in the breeding population of guillemots *Uria aalge* on Vedøy, Røst. Fauna norv. Ser. C, Cinclus 9: 25-34.

Bakken V., Runde O. & Tjørve E. 2003. Norsk Ringmerkingsatlas. Vol. 1. Stavanger Museum, Stavanger.

Bakketeig I., Dommasnes A., Føyn L., Haug T., Iversen S., Røttingen I., Svendsen E. & Torstensen E. (eds.). 2005. Havets ressurser og miljø 2005. Fisken og Havet, særnr. 1-2005, Inst. Mar. Res., Bergen.
Barrett R.T. 1985. Further changes in the breeding distribution and numbers of cliff-breeding seabirds in Sør-Varanger, N. Norway, 1975-1983. Fauna norv. Ser. C, Cinclus 8: 35-39
Barrett R.T. 2001. Monitoring the Atlantic puffin *Fratercula arctica*, common guillemot *Uria aalge* and black-legged kittiwake *Rissa tridactyla* breeding populations on Hornøya, northeast Norway, 1980-2000. Fauna norv. 21: 1-10.
Barrett R.T. 2002. Atlantic puffin *Fratercula arctica* and common guillemot *Uria aalge* chick diet and growth as indicators of fish stocks in the Barents Sea. Mar. Ecol. Prog. Ser. 230: 275-287.
Barrett R.T. 2003. The rise and fall of cliff-breeding seabirds in Sør-Varanger, NE Norway, 1970-2002. Fauna norv. 23: 35-41.
Barrett R.T. 2007. Food web interactions in the southwestern Barents Sea: black-legged kittiwakes *Rissa tridactyla* respond negatively to an increase in herring *Clupea harengus*. Mar. Ecol. Prog. Ser. 349: 269-276.
Barrett R.T., Anker-Nilssen T., Gabrielsen G.W. & Chapdelaine G. 2002. Food consumption by seabirds in Norwegian waters. ICES J. Mar. Sci. 59: 43-57.
Barrett R.T., Anker-Nilssen T., Rikardsen F., Valde K., Røv N. & Vader W. 1987. The food, growth and fledging success of Norwegian Puffin chicks *Fractercula arctica* in 1980-1983. Ornis Scand. 18: 73-83.
Barrett R.T. & Folkestad A.O. 1996. The status of the North Atlantic Gannet *Morus bassanus* after 50 years in Norway. Seabird 18: 30-37.
Barrett R.T. & Krasnov Y.V. 1996. Recent responses to changes in stocks of prey species by seabirds breeding in the southern Barents Sea. ICES J. Mar. Sci. 53: 713-722.
Barrett R.T. & Vader W. 1984. The status and conservation of breeding seabirds in Norway. ICBP Tech. Publ. 2: 323-333.
Barth E.K. 1968. The circumpolar systematics of *Larus argentatus* and *Larus fuscus* with special reference to the Norwegain populations. Nytt Mag. Zool. 15 (Suppl 1): 1-50.
Blindheim J. 1989. Ecological features of the Norwegian Sea. Proc. 6th Conf. Com. Arct. Int: 366-401.
Brun E. 1965. Polarlomvien *Uria lomvia* (L.) som rugefugl i Norge. Sterna 6: 229-250.
Brun E. 1969. Utbredelse og hekkebestand av lomvi (*Uria aalge*) i Norge. Sterna 8: 209-244.
Brun E. 1979. Present status and trends in populations of seabirds in Norway. In: Bartonek J.C. & Nettleship D.N. (eds) Conservation of Marine Birds of Northern North Americ. US Dep. Fish. Wildl. Ser. Wildl. Res. Rep. 11: 289-301.
DN. 1999. Nasjonal rødliste for truete arter i Norge 1998. Norwegian Red List 1998. DN-rapport 1999-3. Directorate for Nature Management, Trondheim
Durant J.M., Anker-Nilssen T. & Stenseth N.C. 2003. Trophic interactions under climate fluctuations: the Atlantic puffin as an example. Proc. Roy. Soc. London B 270: 1461-1466.
Durant J.M., Stenseth N.C., Anker-Nilssen T., Harris M.P., Thompson P.M. & Wanless S. 2004. Marine birds and climate fluctuation in the North Atlantic. In: Stenseth N.C., Ottersen G., Hurrell J.W. & Belgrano A. (eds.) Marine Ecosystems and Climate Variation – the North Atlantic. Oxford Univ. Press: pp. 95-105.
Folkestad A.O., Follestad A. & Valde K. 1980. Successful breeding of Great Skuas on Runde, W. Norway, in 1980. Fauna norv. Ser. C, Cinclus 3: 55.
Frederiksen M., Wanless S., Harris M.P., Rothery P. & Wilson L.J. 2004. The role of industrial fisheries and oceanographic change in the decline of North Sea black-legged kittiwakes. J. Appl. Ecol. 41: 1129-1139.
Furness R.W. & Barrett R.T. 1985. The food requirements and ecological relationships of a seabird community in North Norway. Ornis Scand. 16: 305-313.

Gjershaug J.O., Thingstad P.G., Eldøy S. & Byrkjeland S. (eds.). 1994. Norsk Fugleatlas. Norsk Ornitologisk Forening, Klæbu.

Gjøsæter H. 1998. The population biology and exploitation of capelin (*Mallotus villosus*) in the Barents Sea. Sarsia 83: 453-496.

Haftorn S. 1971. Norges Fugler. Universitetsforlaget, Oslo.

Hario M., Bianki V. & Zimin V. 1998. Larus fuscus fuscus. In: Kotiranta H., Uotila P., Sulkava S. & Peltonen S.-L. (eds.) Red Data Book of East Fennoscandia. Min. Environ. Finn. Environ. Inst. & Bot. Mus., Finnish Mus. Nat. Hist., Helsinki: pp. 247-249.

Hario M., Hirvi J.-P., Hollmén T. & Rudbäck E. 2004. Organochlorine concentrations in diseased vs. healthy gull chicks from the northern Baltic. Environ. Poll. 127: 411-423.

ICES. 2005a. Report of the arctic fisheries working group (AFWG). ICES Advisory Committee on Fishery Management ACFM:20.

ICES. 2005b. Report on the northern pelagic and blue whiting fisheries working group (WGNPBW). ICES Advisory Committee on Fishery Management CM 2006/ACFM:05.

Krasnov J.V. & Barrett R.T. 1995. Large-scale interactions among seabirds, their prey and humans in the southern Barents Sea. In: Skjoldal H.R., Hopkins C., Erikstad K.E. & Leinaas H.P. (eds) Ecology of Fjords and Coastal Waters. Elsevier Sci BV, Amsterdam: 443-456.

Loeng H. 1989. Ecological features of the Barents Sea. Proc 6th. Conf. Comité Actique Int.: 327-365.

Lorentsen, S.-H. 1994. Fiskemåke *Larus canus*. In: Gjershaug J.O., Thingstad P.G., Eldøy S. & Byrkjeland S. (eds.) Norsk Fugleatlas. Norsk Ornitologisk Forening, Klæbu: 234-235.

Lorentsen S.-H. 2005. Det nasjonale overvåkningsprogrammet for sjøfugl. Resultater til og med hekkesesongen 2005. NINA Rapport 97.

Lorentsen S.-H. 2006. Hvordan er utviklingen i de norske ternebestandene? Vår Fuglefauna 29: 22-26.

Lorentsen S.-H. & Nygård T. 2001. Det nasjonale overvåkingsprogrammet for sjøfugl. Resultater fra overvåkingen av overvintrende sjøfugl fram til 2000. NINA Oppdragsmelding 717.

Mitchell P.I., Newton S.F., Ratcliffe N. & Dunn T.E. (eds.). 2004. Seabird Populations of Britain and Ireland. Results of the Seabird 2000 census (1998-2002). T & AD Poyser, London.

Moy F., Bekkby T., Cochrane S., Rinde E. & Voegele B. 2003. Marin karakterisering. Typologi, system for å beskrive økologisk naturtilstand og forslag til referansenettverk. NIVA Rapport lnr. 4731-2003.

Pannekoek J. & van Strien A. 2005. TRIM 3 Manual (trends and indices for monitoring data). Statistics Netherlands.

Rinde E., Bjørge A., Eggereide A. & Tufteland G. (eds.). 1998. Kystøkologi, den ressursrike norskekysten. Universitetsforlaget, Oslo.

Rose G.A. 2005a. On distributional responses of North Atlantic fish to climate change. ICES J. Mar. Sci. 62: 1360-1374.

Rose G.A. 2005b. Capelin (*Mallotus villosus*) distribution and climate: a sea "canary" for marine ecosystem change. ICES J. Mar. Sci. 62: 1524-1530.

Røv N. 1997. Population trends and regulation of breeding numbers in the great cormorant in Norway. Suppl. Ric. Biol. Selvaggina XXVI: 5-9.

Røv N., Lorentsen S.-H. & Nygård T. 2003. Status and trends in the Great Cormorant *Phalacrocorax carbo carbo* in Norway and the Barents Sea Region. Vogelwelt 124, Suppl: 71-75.

Røv N. & Strann K.-B. 1987. The present status, breeding distribution, and colony size of the Cormorant *Phalacrocorax carbo carbo* in Norway. Fauna norv. Ser. C, Cinclus 10: 39-44.

Røv N., Thomassen J., Anker-Nilssen T., Barrett R.T., Folkestad A.O. & Runde O. 1984. Sjøfuglprosjektet 1979-1984. Viltrapp 35, DVF, Trondheim.

Sakshaug E., Bjørge A., Gulliksen B., Loeng H. & Mehlum F. 1994. Structure, biomass distribution, and energetics of the pelagic ecosystem in the Barents Sea: a synopsis. Polar Biol. 14: 405-411.

Skjoldal H.R. (ed.). 2004. The Norwegian Sea Ecosytem. Tapir, Trondheim.

Stougie L., den Ouden J.E. & Couperus A.S. 1989. [Survey of breeding seabirds at Syltefjordstauran (Finnmark, Northern Norway) in May 1989]. Sula 3: 63-67.

Strann K.-B. & Vader W. 1992. The nominate Lesser Black-backed Gull *Larus fuscus fuscus*, a gull with a tern-like feeding biology, and its recent decrease in northern Norway. Ardea 80: 133-142.

Strann K.-B., Vader W. & Barrett R.T. 1991. Auk mortality in fishing nets in north Norway. Seabird 13: 22-29.

Thingstad, P.G. 1994. Sildemåke *Larus fuscus*. In: Gjershaug J.O., Thingstad P.G., Eldøy S. & Byrkjeland S. (eds.) Norsk Fugleatlas. Norsk Ornitologisk Forening, Klæbu: 236-237.

Vader W. 1980. The Great Skua *Stercorarius skua* in Norway and the Spitsbergen area. Fauna norv. Ser. C, Cinclus 3: 49-55.

Vader W., Barrett R. & Strann K.-B. 1987. Sjøfuglhekking i Nord-Norge 1987, et svartår. Vår Fuglefauna 10: 144-147.

Vader W., Barrett R.T., Erikstad K.E. & Strann K.-B. 1990. Differential responses of Common and Thick-billed Murres to a crash in the Capelin stock in the southern Barents Sea. Studies in Avian Biology. 14: 175-180.

Våge H. & Stenersen J. 2003. Havsulene i Nordland - en statusrapport. Havørna. 14: 38-39.

THE BREEDING BIOLOGY OF TERNS ON THE WESTERN ISLES IN RELATION TO MINK ERADICATION

NORMAN RATCLIFFE[1], DAN HOUGHTON[2], ANDREW MAYO[2], TREVOR SMITH[3] AND MARTIN SCOTT[2]

Ratcliffe N., Houghton D., Mayo A., Smith T. & Scott M. 2006. The breeding biology of terns on the Western Isles in relation to mink eradication. Atlantic Seabirds 8(3): 127-135. *American Mink were introduced to Lewis in the 1950s and '60s, and their range expanded south to colonise North Uist by the late 1990s. Mink eradication was initiated in 1999 and the islands were almost entirely cleared by 2004. The breeding biology of terns on the Uists prior to colonisation by mink and after eradication was compared with that on Lewis where mink were present during the entire period. The results showed that nest survival was significantly higher on the Uists compared with Lewis in 2005, and this was largely explained by lower mammalian predation rates in the Uists. However, there was no significant additive effect of mink occupation on productivity across years. Productivity was mainly affected by year, with little evidence of differences between archipelagos within years. However, productivity was low in the only two years when good sample sizes were available in both archipelagos, probably due to poor food supply or inclement weather. In these situations, the effects of mink predation would be expected to be compensatory, since they were taking eggs and chicks that would probably have starved subsequently. Improved annual monitoring of colonies on both Uist and Lewis needs to be conducted in order to investigate the interactive effects of mink removal and food availability on tern productivity in the Western Isles.*

[1]RSPB Scotland, 10 Albyn Terrace, Aberdeen, AB10 1YP, Scotland, U.K.; [2]RSPB Scotland, Office 2, Clintons Yard, Rigs Road, Stornoway, Isle of Lewis, HS1 2RF, Scotland, U.K.;. [3]RSPB Scotland, Dunedin House, Ravelston Terrace, Edinburgh, EH4 3TP, Scotland, U.K. E-mail: Norman.Ratcliffe@rspb.org.uk

INTRODUCTION

American mink *Mustela vison* (hereafter "mink") feed on a wide variety of prey, including birds and their eggs and chicks (Dunstone 1993). They are amphibious and are able to reach islets within 2 km of the shore, and also those further offshore if linked by an island chain (Craik 1995). Incubating adults, eggs and chicks of small, ground-nesting seabirds (especially Black Guillemots *Cepphus grylle*, European Shags *Phalacrocorax aristotelis*, terns and small gulls) at such sites are vulnerable to direct mink predation (Folkestad 1982; Andersson 1992; Craik 1995, 1998, 2000; Kilpi 1995). The reduced productivity results in population declines through elevated adult mortality, low recruitment, and abandonment of affected sites (Kipli 1995; Craik 1997; Antolos *et al.* 2004).

Following colony abandonment, birds move to predator-free offshore sites (Folkestad 1982; Craik 1997; Nordström & Korpimäki 2004) or congregate in fewer, larger colonies (Clode & Macdonald 2002). Mink therefore reduce colony site availability and this may cause limitation of seabird populations even after direct predation has ceased. For example, productivity at offshore sites may be lower because of their remoteness from key foraging areas (Hall & Kress 2004), whereas that at large colonies may be depressed by elevated density-dependent competition (Birkhead & Furness 1985). However, the effects of mink on seabirds can be halted or reversed: their removal by trapping results in increased productivity, persistence of extant colonies, recolonisation of abandoned sites, and increased regional numbers (Craik 1998; Nordström *et al.* 2003; Nordström & Korpimäki 2004).

Mink were accidentally introduced to Lewis in the Western Isles of Scotland when they escaped from fur farms during the 1950s and 1960s (Dunstone 1993). Their range expanded throughout the island of Lewis and Harris (Hudson & Cox 1988), and by the late 1990s had spread further south to include the Uists island chain (Harrington *et al.* 1999; Roy 2006). The colonisation would undoubtedly have continued until all accessible parts of the archipelago were occupied but for the initiation of the Hebridean Mink Project in 1999, which removed mink from the Uists and South Harris with the aim of protecting nationally important ground-nesting seabird and wader populations (Moore *et al*. 2003). By 2004, few mink remained in the control areas and by the conclusion of the project in 2006, they had been successfully eradicated from them (Roy 2006). Meanwhile, the range and numbers of mink on North Harris and Lewis remained largely unchanged despite control measures, and recolonisation of South Harris and the Uists must be inevitable while this source population persists. Hence, the eradication programme was extended to North Harris and Lewis in September 2006, with the aim of protecting biodiversity there and preventing recolonisation of the Uists.

This paper describes the breeding biology of terns on the Uists in 2004 and 2005, and compares it with that on Lewis during 2005 in order to evaluate whether removal of mustelids has improved tern reproductive success. Data from previous published studies of tern breeding biology in the Western Isles are also included for comparison. The implications of the findings for mink management work in the Western Isles are discussed.

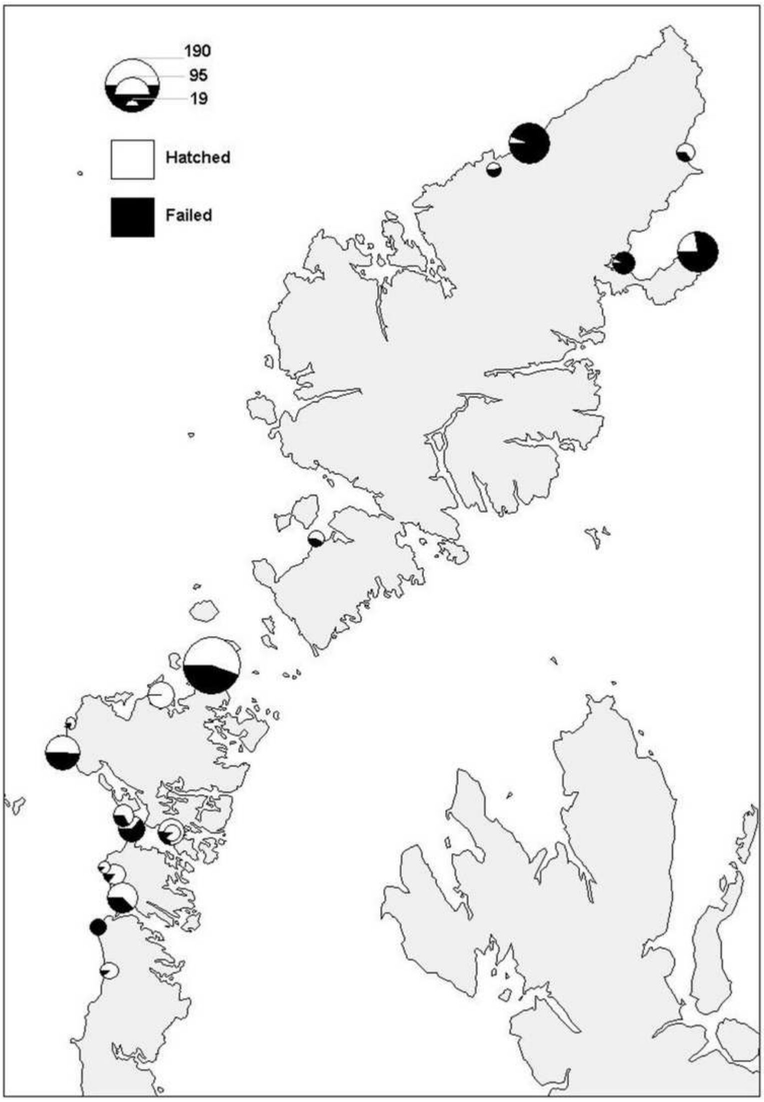


Figure 1. Locations, sizes and hatching success of tern colonies studied during 2004 and 2005 in the Western Isles. Pie sizes represent number of pairs (see key).
Figuur 1. Ligging, grootte en uitkomstsucces van sternkolonies die in 2004 en 2005 op de Western Isles bestudeerd werden. De grootte van de taartdiagrammen geeft het aantal paar aan (zie legenda).

METHODS

The breeding biology of terns was studied at selected sites in the Uists in 2004 and 2005 and on Lewis in 2005. The location of colonies studied is shown in Fig. 1. Colonies were visited every 3-5 days through May, June and July.

Nests were located and marked at each colony to determine their fate between repeat visits. Nests were classed as: "hatched" if chicks were present; "abandoned" if cold eggs were present; "eaten" if the nest was empty prior to the expected hatching date or shell remains indicating predation were present; "trampled" if the egg was crushed; "sandblown" if the egg was buried in sand, and "flooded" if the nest was empty or abandoned following a tidal flood. In cases where the fate of a nest was not certain (i.e. an empty scrape that could have been due to predation or chicks hatching and dispersing), the nest was classed as surviving up to the penultimate visit (Manolis *et al*. 2000). Daily nest survival rates (the probability of a nest surviving for one day) were estimated using a Generalised Linear Model (GLM) with a binomial error distribution and logit link, where the fate of the nest (survived or failed) was the response variable and number of days for which the nest was monitored was the binomial denominator (Crawley 1993). Year (2004 or 2005), species (Little Tern *Sterna albifrons* or Common *Sterna hirundo* and Arctic *Sterna paradisaea* combined), and mink presence for the region in which the colony was situated were defined as factors each with two levels. Variables were retained in the minimal adequate model if they explained a significant amount of the deviance, with model selection being conducted using *chi*-square tests and a maximum *alpha* of 0.05. Hatching success (the likelihood of a nest hatching at least one chick) was estimated by raising the daily nest survival rate to the power of the average tern incubation period (22 days; Cramp 1985). The asymmetrical lower and upper one standard error limits of the estimate are presented as LSE and USE.

Productivity (the number of chicks fledged per breeding pair) was estimated from peak counts of numbers of pairs and fledged chicks. Flush counts of adults and nest counts were made during the incubation period in order to determine colony size following Walsh *et al*. (1995). Flush counts were divided by 1.5 (Bullock & Gomersall 1981) to estimate the number of breeding pairs. The number of chicks fledged from the colony was estimated from either flush counts of fledglings or capture-mark recapture of near-fledged chicks (Walsh *et al*. 1995). Data on productivity and mink range were extracted from Clode & Macdonald (2002) and Rae (1999) for statistical comparison with current data. Productivity was estimated using a GLM with a Poisson error distribution and log link. The number of chicks fledged was the response variable and the number of pairs was defined as an offset (Crawley 1993). This procedure weights cases from each colony appropriately according to the sample size (i.e. number of pairs) and constrains predicted values to be greater than zero. Model selection was conducted as described for hatching success, except that the residual deviance was scaled (by the square root of the residual deviance divided by the residual degrees of freedom) in order to account for over-dispersion, and F-ratio tests were used to test the significance of terms (Crawley 1993).

RESULTS

Hatching success Average hatching success was 40.1% (LSE = 38.0, USE = 42.1) for all years and archipelagos combined. There were significant differences in hatching success between archipelagos, with that on the Uists (58.8%, LSE = 56.0, USE = 61.5) being significantly higher than that on Lewis and Harris (17.8%, LSE = 14.5, USE = 21.3; $\chi^2_1 = 110.2$, $P < 0.0001$). Spatial variation in hatching success between colonies is shown in Fig. 1. There were no significant effects of species or year once archipelago-dependent variation was explained.

Productivity Productivity varied between years ($F_{4,75} = 36.59$, $P < 0.001$, scale parameter 3.4). Productivity was highest in 1992, lowest in 2005 and intermediate in other years (Table 1). These overall annual variations were reflected by within-site trends, suggesting these fluctuations were not due solely to variations in the sites sampled between years. When controlling for year effects, productivity of Little Terns was significantly higher than that of Arctic Terns within years ($F_{1,74} = 4.65$, $P < 0.05$, scale parameter 3.32; Table 2). The difference between archipelagos was not significant ($F_{1,73} = 0.15$, $P > 0.6$, scale parameter = 3.34).

Causes of loss Of the 86 failed study nests at which cause of failure was established on Uists, 62% were depredated, 24% abandoned, 8% buried by windblown sand, 2% trampled by livestock and 2% flooded. Of the 190 failed nests on Lewis these figures were 74%, 12%, 0%, 2% and 12% respectively. Evidence of predation by mustelids was found mainly on Lewis, where a total of 21 eggs, 19 chicks and 29 adults were discovered in caches near four of the six colonies. Remains in one of these were more consistent with Otter *Lutra lutra* predation than with mink (C. Craik, pers. comm.). On the Uists, otter predation was evident on Berneray, in the far north of the archipelago, where six killed adults and a cache of *c*. 20 eggs were found, while at Aird a Machair in South Uist caches of eggs were found near a den site that from its size probably belonged to a Brown Rat *Rattus norvegicus*.

Table 1. Variation in sample sizes (colonies, pairs) and productivity (chicks per breeding pair) of terns in the Western Isles by species, year and archipelago. LSE and USE represent the asymmetrical standard error limits of the productivity estimates.
Tabel 1. Variatie in steekproefgrootte (kolonies, paar) en productie (kuikens per broedpaar) van sterns op de Western Isles per soort, jaar en archipel. LSE en USE geven de asymmetrische standaardfoutenmarge van de schattingen van de productie weer.

Species	Archipelago	Year	Colonies	Pairs	Productivity	LSE	USE
Arctic	Uists	1992	3	130	0.65	0.44	0.94
Arctic	Uists	1993	11	281	0.23	0.13	0.41
Arctic	Uists	2004	8	162	0.24	0.13	0.45
Arctic	Uists	2005	12	550	0.05	0.03	0.11
Arctic	Lewis	1992	2	840	1.04	0.70	1.55
Arctic	Lewis	1993	11	517	0.12	0.08	0.24
Arctic	Lewis	1999	21	1589	0.16	0.11	0.25
Arctic	Lewis	2005	6	1083	0.03	0.02	0.07
Little	Uists	2004	2	17	0.53	0.38	0.74
Little	Uists	2005	1	12	0.17	0.08	0.36
Little	Lewis	1999	2	15	0.20	0.10	0.39
Little	Lewis	2005	1	19	1.21	0.82	1.79

DISCUSSION

Hatching success was more than three times higher on the Uists than on Lewis and Harris. This could be explained by the fact that mink densities were much lower on the Uists than on Harris and Lewis because of the control programme in preceding years. This conclusion is supported by the higher mammalian predation rates on Lewis combined with evidence for mink presence at these sites in the form of caches, dens, spoor and scats. In contrast, on the Uists, severe mammalian predation was only noted at two sites, with gull predation, sandblow and abandonment causing most failures there. Hatching success on the Uists was still relatively low: in other studies; it generally exceeds 80% (for reviews see Hatch 2002, Nisbet 2002, Becker & Ludwigs 2004), indicating that conditions in 2004 and 2005 were unfavourable (see below).

Previous studies have shown that tern productivity is far lower in areas where mink occur than where they are absent (Craik 1998; Nordström *et al.* 2004), but this was not the case in Lewis compared with the Uists in 1993 and 2005; the only years when paired data were available. In 1993 and 2005, productivity was very low across both archipelagos, and this was probably due to inclement weather (Clode & Macdonald 2002) and reduced food availability (this study) respectively. Indeed, several species of seabird breeding in west Scotland experienced their worst year of productivity on record in 2005, with failures of auks, terns and Black-legged Kittiwakes *Rissa tridactyla* being noted throughout the inner and outer isles of western Scotland (Mavor *et al*. 2006). During such years, predation will be compensatory; with mink taking eggs and chicks that probably would have died subsequently from other causes such as exposure or starvation.

Predation would be expected to be additive in years when feeding and weather conditions are favourable, such that chicks that do not succumb to predation would survive to fledging (Newton 1998). In such years, the contrast in productivity between the Uists and Lewis would be expected to be evident, but this was not the case in 1992 when productivity was high on both archipelagos. Productivity was recorded at only two colonies on Lewis in 1992, and mink do not attack all colonies in suitable habitat within their range every year. For example, in south-west Scotland between 1990 and 2006, 58% of unprotected tern colonies were not mink-affected (J.C.A. Craik unpublished data). Hence, conclusions concerning the effects of mink on tern productivity cannot reliably be drawn from a small sample of colonies in areas where mink are present and absent.

Any benefits to terns of mink removal may be partially negated by recovery of feral Ferret *Mustela furo* numbers in some parts of the Uists. The numbers of ferrets were reduced incidentally in the Hebridean Mink Project, but numbers have begun to recover subsequently (Roy 2006). Ferrets in the Uists are distributed along the west coast (Roy 2006) where most of the Uists Arctic Tern colonies occur (Mitchell *et al*. 2004). Hence, Arctic Terns on the Uists may continue to suffer failures due to ferret predation, but Common Terns on the east coast, and both tern species on the Harris Sound Islands (between Harris and North Uist), will avoid this fate as they occur in sites that are unsuitable for, or inaccessible to, ferrets (Roy 2006).

Eradication of mink from Lewis and North Harris began in September 2006 and, if successful, will result in the whole of the Western Isles being free from mink. While our study provides little support for this initiative based on benefits for tern productivity, Rae (1999) reported that mink predation caused complete breeding failure at 13 of the 18 tern colonies present on Lewis in 1999, and reduced productivity at a further two. Furthermore, Clode & Macdonald (2002) found a reduction in the number of Arctic Tern colonies and extirpation of Common Terns, which they interpreted as a consequence of mink predation. Mink removal is therefore likely to improve productivity of terns on Lewis and may promote increases in their numbers and range; this, combined with putative benefits for other biodiversity and economic interests, provide justification for the project. We recommend further monitoring as part of the project so that the effects of mink management on tern productivity, numbers and range may be elucidated.

ACKNOWLEDGEMENTS

This work was conducted as part of the Hebridean Mink Project, funded by the EU Life programme, Scottish Natural Heritage, Scottish Executive, RSPB, Comhairle nan Eilean Siar, CSL and Western Isles Enterprise. Sugoto Roy of CSL kindly gave access to information on the mink control programme in the Western Isles.

BROEDBIOLOGIE VAN STERNS OP DE WESTERN ISLES IN RELATIE TOT VERDELGING VAN NERTSEN

De Amerikaanse nerts werd in de jaren '50 en '60 op Lewis geïntroduceerd en breidde zijn verspreidingsgebied zuidwaarts uit om North Uist in de late jaren '90 te koloniseren. Verdelging van nertsen begon in 1999 en de eilanden waren in 2004 vrijwel nertsloos. De broedbiologie van sterns op de Uist-eilandjes (de Uists) voor de kolonisatie door nertsen en na de verdelging van deze soort werd vergeleken met de broedbiologie op Lewis, waar nertsen de gehele periode aanwezig waren. De resultaten lieten in 2005 een significant hoger nestsucces op de Uists zien in vergelijking met Lewis, hetgeen grotendeels verklaard werd door lagere predatie op de Uists. Er was echter geen significant toegevoegd effect van nerts op broedsucces in de verschillende jaren. Broedsucces werd grotendeels beïnvloed door het jaar, met nauwelijks bewijs voor verschillen tussen eiland(groep)en in de verschillende jaren. In de enige twee jaren dat er goede steekproeven genomen konden worden, was het broedsucces echter laag; waarschijnlijk als gevolg van een slecht voedselaanbod of ongunstig weer. In deze situatie is te verwachten dat het effect van predatie door nertsen gering is, aangezien ze eieren en kuikens eten die anders waarschijnlijk verhongerd zouden zijn. Verbeterde jaarlijkse monitoring van de kolonies op de Uists en Lewis is wenselijk om het gecombineerde effect van nertsverdelging en voedselaanbod op het broedsucces van sterns op de Western Isles te onderzoeken.

REFERENCES

Andersson, Å., 1992. Development of waterbird populations in the Bullerö archipelago off Stockholm after colonisation by mink. Ornis Svecica 2: 107-118.

Antolos M., Roby, D.D., & Collis K. 2004. Breeding ecology of Caspian terns at colonies on the Columbia plateau. Northwest Science 78: 303-312.

Becker, P.H. & Ludwigs, J.-D. 2004. Sterna hirundo Common Tern. Pp 91-137 in: Parkin, D. (ed.). BWP update. The Journal of the Birds of the Western Palearctic. Vol. 6. Oxford University Press, UK.

Birkhead, T.R. & Furness, R.W. 1985. Regulation of seabird populations. Pp. 145-167 in: Sibley, R.M. & Smith, R.H. (eds.). Behavioural Ecology: Ecological Consequences of Adaptive Behaviour. Blackwell, London.

Bullock, I.D. & Gomersall, C.H. 1981. The breeding populations of terns in Orkney & Shetland in 1980. Bird Study 28: 187-200.

Clode, D. & Macdonald, D.W. 2002. Invasive predators and the conservation of island birds: the case of American Mink Mustela vison and terns Sterna spp. in the Western Isles, Scotland. Bird Study 49: 118-123.

Craik, J.C.A. 1995. Effects of North American Mink on the breeding success of terns and smaller gulls in west Scotland. Seabird 17: 3-11.

Craik, J.C.A. 1997. Long-term effects of North American Mink Mustela vison on seabirds in western Scotland. Bird Study 44: 303-309.

Craik, J.C.A. 1998. Recent mink-related declines in gulls and terns in west Scotland and the beneficial effects of mink control. Argyll Bird Report 14: 99-110.

Craik, J.C.A. 2000. Breeding success of common gulls Larus canus in west Scotland II. Comparison between colonies. Atlantic Seabird 2: 1-12.

Cramp, S. (ed) 1985. The Birds of the Western Palearctic. Volume IV: Terns to Woodpeckers. Oxford University Press, Oxford, UK.

Crawley, M.J. 1993. GLIM for ecologists. Blackwell, London.

Dunstone, N. 1993. The Mink. T and A.D. Poyser, London, UK.

Folkestad, A. 1982. The effect of mink predation on some seabird species. Viltrapport 21: 42-49.

Harrington, L.A., Grogan, A.L., Bonesi, L. & Macdonald, D.W. 1999. Survey of American mink Mustela vison, on North Uist. Unpublished report to Scottish Natural Heritage, Wildlife Conservation Research Unit, Oxford.

Hall, C.S. & Kress, S.W. 2004. Comparison of common tern reproductive performance at four restored colonies along the Maine coast, 1991-2002. Waterbirds 27: 424-433.

Hatch, J.J. 2002. Arctic Tern Sterna paradisaea. In: Poole, A. & Gills, F (eds). The Birds of North America, No. 707. The Birds of North America Inc., Philadelphia, USA.

Hudson, P. & Cox, R. 1988. Mink problems on the Outer Hebrides: a pilot study. Game Conservancy Report.

Kipli, M. 1995. Breeding success, predation and local dynamics of colonial common gulls Larus canus. Ann. Zool. Fenn. 32: 175-182.

Manolis, J.C., Andersen, D.E. & Cuthbert, F.L. 2000. Uncertain nest fates in songbird studies and variation in Mayfield estimation. The Auk 117: 615-626.

Mavor, R.A., Parsons, M., Heubeck, M. & Schmitt, S. 2006. Seabird numbers and breeding success in Britain and Ireland, 2005. Peterborough, Joint Nature Conservation Committee. (UK Nature Conservation, No. 30).

Mitchell, P.I., Newton, S., Ratcliffe, N., & Dunn, T. (eds) 2004. Seabird populations of Britain and Ireland. T & A.D. Poyser, London.

Moore, N.P., Roy, S.S. & Helyar, A. 2003. Mink (Mustela vison) eradication to protect ground-nesting birds in the Western Isles, Scotland, United Kingdom. New Zealand Journal of Zoology. 30: 443-452.

Newton, I. 1998. Population limitation in birds. Academic Press

Nisbet, I.C.T. 2002. Common Tern Sterna hirundo. In: Poole, A. & Gills, F (eds). The Birds of North America, No. 618. The Birds of North America Inc., Philadelphia, USA.

Nordström M., Högmander J., Laine, J., Nummelin, J., Laanetu, N. & Korpimäki, E. 2003. Effects of feral mink removal on seabirds, waders and passerines on small islands in the Baltic Sea. Biological Conservation 109: 359-368.

Nordström, M. & Korpimäki, E. 2004. Effects of island isolation and feral mink removal on bird communities on small islands in the Baltic Sea. Journal of Animal Ecology. 73: 424-433.

Rae, S. 1999. The effect of predation by mink on ground-nesting birds in the Outer Hebrides. Outer Hebrides (The Western Isles) Bird Report 1999: 105-113.

Roy, S.S. 2006. Mink control to protect important birds in SPAs in the Western Isles. Final Report to EU LIFE III – Nature, Brussels.

Walsh, P.M., Halley, D.J., Harris, M.P., del Nevo, A., Sim, I.M.W., & Tasker, M.L. 1995. Seabird monitoring handbook for Britain and Ireland. JNCC / RSPB / ITE / Seabird Group. Peterborough.

News and notices

BOOK REVIEWS

ANDY BROWN & PHIL GRICE 2005. *Birds in England.* T. & A.D. Poyser, London. ISBN 0-7136-6530-0, hardback, 694 pp. 50 colour photographs. Price £40.

This is the first book to be written on the birds of England, which is quite surprising given other country (Wales, Ireland, Scotland) avifaunas have been available for many years. It took some time to get used to thinking about England as opposed to Britain as an ornithological area but the authors have done very well to keep the focus on England throughout. This is a monumental book: 700 pages, 50 photographs and numerous line drawings throughout. The authors, Andy Brown and Phil Grice, who both hold senior posts at English Nature, the statutory nature conservation agency in England (the other UK countries have their own agencies), have done a tremendous job in digesting so much information into a well-written, concise and fact-filled book. Their rigorous scientific background and genuine enthusiasm for birds shines through in the introductory chapters and species accounts, and the reference list of over 50 pages is testimony to how well researched the book is.

Following a brief introduction, a chapter entitled 'The composition and character of the English avifauna' spans 29 pages, and covers subjects such as English birds in a global context, migration, weather and climate change. This is an impressive chapter with 34 tables summarising facts and figures relating to the English avifauna such as new breeding species, species that have gone extinct as breeders, subspecies restricted as breeding birds to England and adjacent areas (e.g. Razorbill *Alca torda islandica*), early and late migrant dates, exceptional visible migration counts, exceptional falls (e.g. Holme, Norfolk September 1993), and autumn influxes of Nearctic waders. For the seabird enthusiast, a table details 12 notable English seawatches to demonstrate the variation in species composition and numbers of seabirds that pass our coastal headlands. For example, 77,500 Kittiwakes passed Flamborough, Yorkshire on 21 August 1988, 20,000+ Gannets passed St Ives, Cornwall, on 3 September 1983 with 20-50,000 Manx Shearwaters on the same day are some of the most notable counts. Another fascinating table, and one which I will refer to regularly in my work, is of 'Unusual avian events in England 1900-2000', which includes the incident of hundreds of seabirds being killed during a hailstorm on 2 July 1914 in Teesmouth, Cleveland, the exceptional inland passage of Common and Arctic Terns in 1947, the influx of Grey Phalaropes and Sabine's Gulls in the storms of 1987, and the largest passage of Cory's Shearwaters in Cornwall in 1998.

The chapter on 'Bird habitats in England' is extremely thorough, supported by tables and maps that make this section very easy to dip into and glean information. Sections on sea cliffs (20% of England's coastline), rocky shores (10% of the coastline) and the open sea discuss our knowledge of these habitats and their characteristic fauna. Data sources such as Operation Seafarer, the Seabird Colony Register and Seabird 2000 are drawn upon here and throughout the species accounts. The conservation and future of these environments is discussed, highlighting the need for further research into temporal variation in the use of inshore waters, on the distribution and numbers of seaduck, divers and grebes, and on the feeding ecology of different age classes of seabirds and their interactions with fisheries. Problems of overfishing, pollution, oil spills and aggregate extraction are all covered in some depth.

A short chapter introduces the species accounts and outlines their scope – such as the species included, those species excluded, geographical coverage (excludes Channel Islands, Flatholm and Isle of Man), and survey and surveillance sources used. The species accounts cover the period up to the end of 2000, although Chapter 6 includes species new to England and the rarest of vagrants and exceptional events in the period 1 January 2001 to 31 August 2004 (for example, 2,674 Sooty Shearwaters passed Flamborough, Yorkshire on 22 September 2002). The main species accounts vary in length and the authors have dedicated more space to species of conservation concern or where there have been significant changes in populations or distribution. Each account starts with a review of the species' European and British status before discussing the numbers and distribution in England, simple ecology and any conservation issues. Sensibly, records of vagrants are summarised unless there are just a handful of records in which case they are all listed. No reference is made to the Hastings Rarities in these accounts. It is a shame that no distribution maps are included but this is completely understandable. The accounts for Lesser Black-backed Gull and Herring Gull are particularly interesting. Here, they document changing breeding status and habits, especially the recent expansion onto roof tops for nesting and migratory movements. I was a little disappointed that *Larus argentatus cachinnans* was simply described as a "scarce autumn and winter visitor to central and southeastern England", which is a rather simplified description. Many of the accounts have very useful tables documenting historical and recent status by county, for example Arctic Tern and Little Tern. For the Gannet, a table shows the long-term changes in status at Lundy and Bempton from 1919 to 1999, with information on world population estimates for those years.

For the conservationist or scientist, one of the most useful tables is the 'Annotated checklist of birds in England: population, conservation and legal status', which forms Appendix 1. The population status, breeding population estimate, global and European threat category, SPEC category, Red or Amber list status amongst other information, are listed for all species on the English list.

Overall, this is an excellent book and one that I will use over and over again. I would recommend it to anyone with an interest in birds in England. Congratulations must go to the authors who have produced a first-class book.

Dawn Balmer

ONLEY, D. & SCOFIELD, P. 2007. *Field Guide to the Albatrosses, Petrels and Shearwaters of the World*. Christopher Helm, London. ISBN 978-0-7136-4332-9, 240pp, many illustrations. £19.99.

In the last few years there have been dramatic changes in the classification of several groups within the procellariiforms, based primarily on new DNA sequence data. There are now 24 species of albatross rather than the 13 you will find in Peter Harrison's seabird identification guide and the Hoyo *et al*. Handbook of the Birds of the World, none of which is now called a wandering albatross; and there has been considerable rearrangement and splitting in other groups such as the little and Manx shearwaters. Apparently the little shearwaters I ringed in the Cape Verde Islands a few years ago are now Audoubon's shearwaters, while the Audoubon's shearwaters I saw in the Galapagos on an ecotourism trip are now Galapagos shearwaters. The little shearwaters I caught on Gough Island are now a separate new species, the sub-Antarctic little shearwater, while those I used to ring in the Azores are now Macaronesian shearwaters. Fortunately, the little shearwaters of New Zealand still count as little shearwaters. But I for one am quite confused by all these changes, particularly since the Audubon's shearwaters of the Cape Verde islands have blue legs, which Peter Harrison pointed out distinguishes the little shearwater from the pink-legged Audoubon's shearwater. Apparently the DNA tells us otherwise, however, and leg colour is no longer diagnostic for these taxa.

So Onley and Scofield's new field guide is a very useful book. It takes on the highly challenging task of illustrating and describing the diagnostic features of the currently accepted species of procellariiforms. In a few cases this is almost an impossible task, as plumages of some species really cannot be distinguished, despite the newly discovered clear differences in DNA. The resulting field guide has much that is praiseworthy. The illustrations are extremely fine, with a tendency to understate characteristic features of species rather than to emphasise them. This I think does help the observer. The layout of illustrations is also carefully thought out, and the range of upper and underside views, age variations and plumage morphs is well covered. The illustrations are all of birds in flight, the assumption being that the guide will be used at sea rather than at breeding colonies. The book is softback but printed on glossy paper that feels as if it would stand up well to splashes of seawater. How long it will remain a definitive guide may depend on the stability of procellariiform taxonomy in the next few years. I suspect that several groups may still be ripe for revisions and splitting, such as some of the storm-petrels in particular.

The species texts provide a detailed, accurate, and impressively up to date summary of taxonomy, distribution, plumage (including effects of moult and wear) and identification, with a very brief comment on behaviour (at sea). Each species account also contains a map, showing breeding colonies in orange and main at-sea distribution in green. Although many of these maps seem fine to me, quite a number contain inaccuracies, often even conflicting with the text on distribution. Perhaps some of the maps were prepared in haste, and certainly they were not thoroughly checked for accuracy. Dots to mark sub-Antarctic islands seem to have been added carelessly, so that Tristan da Cunha appears east of Bouvet in the southern giant petrel map, the dot for Kerguelen petrel seems to place Gough Island closer to Cape Town than to the correct position, and the dot for Antarctic petrel on Bouvet looks as if it indicates breeding on Gough Island. The northern fulmar map does not indicate breeding in Svalbard or Bear Island or northern Greenland. Antarctic prion is shown as breeding on Tristan or Gough, which it does not. White-chinned petrel is shown as breeding on Tristan, which it does not (because the Tristan population is now considered to be spectacled petrel, a new endemic species). Leach's storm-petrel is not indicated as breeding in Scotland, although the text does give a detailed list of colonies including those in Scotland. The poor quality of the maps detracts slightly from an otherwise very fine book.

I have rather few quibbles with the generally excellent text. Dickinson (2003) is cited early in the book as the authority for the species sequence followed, yet the reference is missing from the short reference list at the end. Several species of albatross are said not to follow fishing vessels, and these include the waved albatross, a species in rapid decline at present due to longline by-catch off Ecuador and Peru, and the short-tailed albatross, a species often found attending longline vessels off Alaska, so I'm not convinced that these species are not attracted to fishing boats. Various tables of measurements are included in the book, but without information on whether they were taken from live or museum specimens, what the sample sizes were, and when and how the birds were sampled. I realise that there is limited space for such details, but it makes it difficult to use these tables of data without such information. But overall, this book is very useful and timely, and should be high on the wish list for anyone with a strong interest in seabirds.

Bob Furness

LAST ATLANTIC SEABIRDS

This issue of *Atlantic Seabirds* is the last. The launch in 1999 of a journal published jointly by the Seabird Group and the Dutch Seabird Group (Nederlandse Zeevogelgroep) was ambitious and at the time it seemed the way forward for two organisations which had already forged close ties. The Executive Committees of both groups should be congratulated for establishing a venture which over the course of its relatively short lifespan nevertheless added up to more than the constituent parts from which it derived. Both *Sula* and *Seabird* were fine organs but *Atlantic Seabirds* expanded the geographical scope and scientific content of both. Ultimately, however, the challenge of maintaining a quarterly journal proved too demanding. *Atlantic Seabirds* was not alone in Europe in suffering for some time a low submission rate, and after much debate the two seabird groups have amicably decided to end the collaboration. Ties between the two groups will remain, however, not least through the active research collaborations that exist between their members.

Both the Seabird Group and the Dutch Seabird Group will revert to their old journals, *Seabird* and *Sula*, re-launching them as annual and four-monthly publications respectively. Martin Heubeck has taken over as Editor of *Seabird*, assisted by Linda Wilson as Publishing Editor, and a new Editorial Board has been established. Mardik Leopold, Kees Camphuysen, Steve Geelhoed, Guido Keijl and Martin Poot will start as Editors of *Sula*.

Submissions to *Seabird* should be made to either martinheubeck@btinternet.com or linda.wilson@jncc.gov.uk. Please see www.seabirdgroup.org.uk for more details and author guidelines. Submissions to *Sula* should be made to Mardik.Leopold@wur.nl. Please see www.zeevogelgroep.nl for more details and author guidelines.

Editors