THE STATUS AND DISTRIBUTION OF EUROPEAN STORM-PETRELS *HYDROBATES PELAGICUS* AND MANX SHEARWATERS *PUFFINUS PUFFINUS* ON THE ISLES OF SCILLY

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Heaney V., Ratcliffe, N., Brown, A., Robinson, P & Lock, L. 2002. The status and distribution of European Storm-petrels *Hydrobates pelagicus* and Manx Shearwaters *Puffinus puffinus* on the Isles of Scilly. Atlantic Seabirds 4(1): 1-16. *This paper describes the first comprehensive survey of the distribution and abundance of breeding European Storm-petrels and Manx Shearwaters on the Isles of Scilly. Diurnal tape playback of vocalisations was used to survey those islands in the archipelago on which birds had previously been reported breeding and to search others with suitable habitat. The total breeding population of Storm-petrels was 1475 Apparently Occupied Sites and of Manx Shearwaters 201 Apparently Occupied Burrows. These numbers are of regional importance for both species and the numbers of Storm-petrels are internationally important. Storm-petrel breeding distribution was restricted to rat-free outer islands, but some Manx Shearwater colonies were found on islands with rats and also feral cats. The role of eradication and control of mammalian predators in the conservation of petrels on the Scilly Isles is discussed.*

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INTRODUCTION

The European Storm-petrel *Hydrobates pelagicus* has a world breeding population that probably lies between 135 000 and 380 000 pairs, of which 65 to 75% nest on offshore islands along the western coasts of Britain and Ireland (Lloyd *et al*. 1991; Tucker & Heath 1994). The Manx Shearwater *Puffinus puffinus* has an estimated world breeding population of 300 000 pairs, of which over 90% nest in Britain and Ireland (Lloyd *et al*. 1991). Its distribution within Britain and Ireland is restricted to three main areas: Rum, off the west coast of Scotland, the Pembrokeshire Islands in Wales and the islands off south-west Ireland (Lloyd *et al*. 1991).

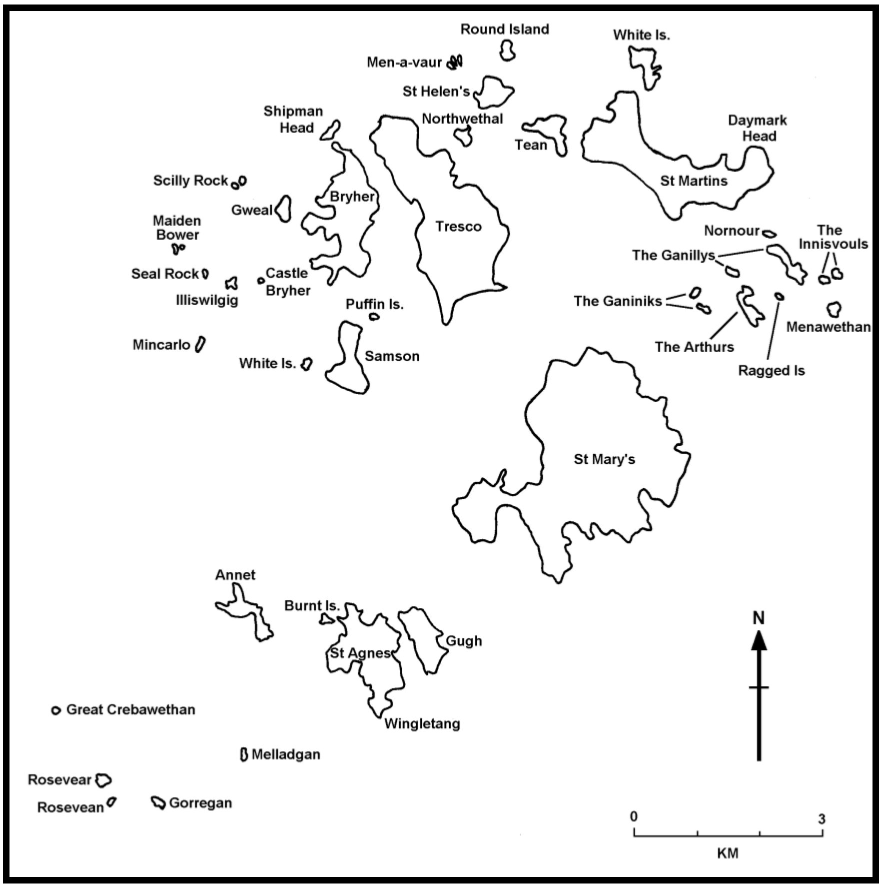

Figure 1. Map of the Scilly Isles, showing the locations of place names mentioned in the text and Tables.
Figuur 1. De Scilly Eilanden met de ligging van de plaatsnamen die in de tekst en tabellen worden genoemd.

Although the distribution and relative abundance of nocturnal petrels in Britain and Ireland are reasonably well documented, their population status and trends are poorly known owing to difficulties in censusing them (Lloyd *et al.* 1991). These are due to the nests being underground or in cavities among boulders, the presence of birds being evident only at night, and problems in accessing the remote islands on which they breed (Lloyd *et al.* 1991). Previous estimates of breeding population size have tended to rely on subjective impressions of bird abundance or on capturing birds in mist nets at night. The former method is non-repeatable and both methods are confounded by the presence and behaviour of large numbers of non-breeding birds, so resulting in overestimates of abundance (Scott 1970; Furness & Baillie 1981; Fowler *et al.* 1982, 1986; Brooke 1990; Fowler & Hounsome 1998). Establishing standardised methods for estimating population sizes and monitoring trends is an essential prerequisite for the conservation of the internationally important Manx Shearwater, European Storm-petrel and also Leach's Storm-petrel *Oceanodroma leucorhoa* populations in Britain.

Since the last comprehensive seabird census of Britain and Ireland in 1985-87, attention has focused on developing a simple and repeatable method for surveying nocturnal, burrow nesting petrels. James and Robertson (1985) first advocated the use of diurnal tape playback of vocalisations to elicit responses from incubating birds in their burrows. The advantages of the method are that it is simple, it can be conducted during the day (promoting safety, longer working hours, more accurate mapping and enhancing detection of nesting habitat or burrows) and it excludes non-breeders and other burrow-nesting species (James & Robertson 1985; Ratcliffe *et al.* 1998a). Accurate estimates depend upon the survey being timed to coincide with the peak of diurnal nest attendance and the likelihood of a bird responding (James & Robertson 1985). These parameters have been estimated for Manx Shearwater (Brooke 1978a,b), European Storm-petrel (Ratcliffe *et al.* 1998a) and Leach's Storm-petrel (Ellis *et al.* 1998), and used to formulate standardised survey methods (Gilbert *et al.* 1998a). Playback has been used to produce population estimates of petrel colonies in Scotland and Wales (Gibbons & Vaughan 1997; Mainwood *et al.* 1997; Wood 1997; Gilbert *et al.* 1998b; Ratcliffe *et al.* 1998b; Vaughan & Gibbons 1998; Mayhew *et al.* 2000) and is being used during Seabird 2000, the complete survey of all seabird colonies in Britain and Ireland during 1999-2001.

European Storm-petrels and Manx Shearwaters have been known to nest in the Isles of Scilly since the mid 1800s (Rodd 1880), and several attempts have been made to assess their status and distribution (Allen 1977; Harvey 1983; Robinson 1999). Estimates of population size and distribution vary widely depending on the methods used, but the regional importance of the Isles of Scilly is clear because they are the only breeding locality for European Storm-petrels and one of only two for Manx Shearwaters in England (Lloyd *et al.* 1991). It is also likely that the Storm-petrel population is internationally important (Lloyd *et al.* 1991). This paper presents the results of the first comprehensive playback survey of the distribution and status of breeding European Storm-petrels and Manx Shearwaters on the Isles of Scilly. The results form part of a national petrel survey that will allow recognition of the importance of the Isles of Scilly in a regional, national and international context and provide baseline population estimates for further monitoring. The conservation implications of the findings are discussed.

METHODS

Scope of survey The Isles of Scilly are situated 45 km off the south-west tip of Cornwall, at approximately 49^o 59' N 06^o 21' W. There are five inhabited islands and approximately 200 other islands, islets and rocks; not all of these could be surveyed with the resources available. Effort was therefore directed at surveying those sites where either Storm-petrels or Manx Shearwaters had previously been recorded (Rodd 1880; Bestwetherick 1968; Penhalluric 1969; Allen 1977; Harvey 1983; Robinson 1999).

Most inhabited islands (and those adjoining them at low tide), islets or rocks lacking suitable habitat, and those islands heavily infested with rats *Rattus norvegicus* would be very unlikely to support a breeding colony of Storm-petrels and so were not surveyed. Their omission was further justified given that many such sites were surveyed for the presence of Storm-petrels in 1999, but no breeding birds were found (Robinson 1999).

The presence or absence of petrels on 17 uninhabited islands at which they had not been recorded breeding previously was also assessed (Table 2). These were selected on the basis of being uninhabited islets not joined to inhabited islands at low tide, and also being adjacent to recorded colonies or having suitable areas of breeding habitat for Storm-petrels or Manx Shearwaters.

The locations of sites mentioned in the text and tables are shown in Figure 1. It is unlikely that any large colonies were omitted from the survey, but it is possible that some small ones were overlooked.

Survey coverage on islands Complete surveys of all suitable habitat were undertaken on each of the islands selected for survey except for Annet. Suitable habitat comprised boulder beach above the high tide mark, burrows, cracks in rocks, scree and dry-stone walls for Storm-petrels and burrows for Manx Shearwaters. Most of the islands were small enough to allow a complete survey of all these habitats. For Manx Shearwaters on the larger inhabited islands such as Bryher and St. Martins, the entire areas from which birds had been recorded previously were surveyed.

The extent of suitable habitat for both Storm-petrels and Manx Shearwaters on Annet precluded a full survey. In the case of Storm-petrels, all areas of boulder beach and rocky outcrops were surveyed fully, including any burrows along their landward edges. Burrows along the coast away from boulder beaches were sampled using a total of 27 10x10 m quadrats placed every 100 m along the coast. A total of 13 10x10 m quadrats was surveyed in the interior of the island.

The distribution of Manx Shearwaters on Annet was established by a low intensity survey of the whole island, which involved searching the island for burrows and directing playback into them. The main concentration was found to be along the east coast within 10 m of the coast, extending to within 30 m of the coast along the thrift *Armeria maritima* covered slopes at the north-east end of the island. Outside this area only a single response was obtained. The eastern coast was surveyed completely using a series of 76 contiguous 10x10 m quadrats, extending to 30x10 m (N = 22) transects in the thrift. Within the thrift area, burrrows under the overhang along the cliff to the north of the island could not be surveyed safely, but breeding birds were found in the few areas that could be accessed. The rest of the coast was surveyed using 27 10x10 m quadrats spaced every 100 m and seven were placed randomly in the interior of the island.

Survey methods Playback surveys of Storm-petrels were conducted between 24 June and 13 July 2000. The survey period was coincident with the peak of incubation for Storm-petrels according to data on laying phenology from colonies in Brittany collected during June 2000 (B. Cadiou, *unpubl. data*). For Manx Shearwater, the survey was conducted between 31 May and 15 June 2000, which coincides with the average peak of incubation for this species (Brooke 1978a). Nest attendance is highest during the peak of incubation in both species (Brooke 1978a; Scott 1970) so the likelihood of detecting breeding birds with playback was maximised by the timing of the survey.

The playback survey for both species followed the methods outlined in Gilbert *et al*. (1998a), using Sony TCM 50DV dictaphones played at full volume. For Storm-petrels, a male purr call recorded on Mousa, Shetland was used. In all habitats, the tape was played for 10 seconds and any responses elicited were noted. In boulder beach habitat, ropes were set 10 m apart along the width of the beach and the tape played at 2 m intervals along these ropes. The playback was therefore always within 1 m of any bird, and at this distance the response probability is similar to that if the playback were directly over the bird (Ratcliffe *et al*. 1998a). In all other habitats where potential nest entrances could be discerned (e.g. burrows or cracks in rocks) the tape was played at each of them. The unit of survey for Storm-petrel was the Apparently Occupied Site (AOS), defined as any cavity or burrow from which a bird responded to playback.

Manx Shearwaters were surveyed using a tape of both male and female calls (both contained within each 10 second loop) recorded on Skokholm. This was played for 10 seconds at all burrow entrances within the survey area and any responses were noted. The use of both male and female calls maximises the response likelihood as birds reply only to calls of the same sex (Brooke 1978b). The unit of survey for Manx Shearwaters was the Apparently Occupied Burrow (AOB), defined as any burrow from which a bird responded to playback.

Presence or absence of breeding Storm-petrels at sites where they had not been recorded previously was established using diurnal playback during incubation on the smaller islands. Larger islands with extensive tracts of habitat were visited overnight to listen for purr calls between 8 and 13 June when males are most vocal and few non-breeders are present (Scott 1970; Ratcliffe *et al.* 1998a). This is the most effective way to locate colonies on large islands quickly so they can be surveyed by playback later in the year (Ratcliffe *et al.* 1998b).

Data analysis Not all active nest sites are detected during a single playback survey because both parents may be absent or a bird may be present that fails to respond. Correction factors must therefore be applied to the total number of responses elicited to produce an accurate population estimate. The response rate on Annet in 1996 was 0.35 (Ratcliffe *et al.* 1998a) and the number of Storm-petrel responses were divided by this value in order to estimate population size. The lower and upper 95% confidence limits of the response rate were 0.252 and 0.448 respectively (derived from Ratcliffe *et al.* 1998a), and so the total number of responses were divided by these values in order to produce confidence intervals around the archipelago-wide population estimate. For Manx Shearwater the number of responses was multiplied by 1.08, which was derived from attendance data and response rates for both sexes presented in Brooke (1978a,b).

The surveys of both species on Annet were based on a combination of complete surveys of some areas or habitats and sampling of others. The number of AOSs or AOBs in areas or habitats that were surveyed completely were estimated by multiplying the total number of responses by the appropriate correction factor. Population estimates of the sampled areas necessitated estimating the mean and confidence limits of the densities within quadrats and extrapolating this to the total area of the habitat. The frequency distributions of AOS and AOB density were skewed, and this invalidated the arithmetic calculation of confidence intervals. Instead, a bootstrapping procedure (Westfall & Young 1993) was used to estimate the average density and the 95% confidence limits. The recorded density data were resampled with replacement (i.e. each value could be sampled more than once or not at all), until the sample size equalled that of the original data set, and the mean of these values then calculated. This procedure was repeated 999 times to produce a frequency distribution of resampled mean values. The average of these was taken as the measure of central tendency and the 2.5 and 97.5 percentiles as the lower and upper 95% confidence intervals respectively.

For the Storm-petrels in burrow habitats on Annet, only the strata along the coast contained AOSs. The length of the island perimeter in metres was calculated using a Geographical Information System, and the length of the boulder beaches subtracted because burrows behind those were surveyed completely. This length was multiplied by 10, as sampling was conducted within 10x10 m quadrats to determine the area of the strata sampled. This area was multiplied by the mean and confidence limits of the density estimate. These were corrected for response rate and added to the survey total for boulder beaches to arrive at the final population estimate for the island.

Densities of Manx Shearwater burrows on Annet were analysed and extrapolated in four strata: the east coast, which was surveyed completely; the cliff overhang along the east coast cliffs that was not safe to survey; the remainder of the coast; and the interior of the island. No AOBs were located in the island's interior, so this was excluded from further calculations. A single AOB was found in the 27 quadrats along the coastal strip away from the east coast and these data were bootstrapped to produce 999 resampled means. The number of AOBs along the cliff top was estimated by calculating 999 resampled mean densities from the 76 10x10 m quadrats surveyed along the east coast. Of the AOBs found here, 70% were in overhang habitats similar to those along the cliff top, and the remainder were in the flat ground within the 10x10 m quadrat. As the flat thrift-covered ground along the cliff top had already been surveyed, the resampled mean was multiplied by 0.70 to exclude birds that had already been counted.

To combine the overall mean and confidence limits, each of the 999 resampled means for the coastal strip was multiplied by the area of this stratum and those for the cliff top were multiplied by the length. Both were then divided by the response rate. The bootstrapped population means were then added together pairwise in random order. The mean and percentiles were calculated to produce a population estimate with confidence intervals and this was added to the total counted in the complete survey of the east coast.

RESULTS

Eleven Storm-petrel colonies were located in the Isles of Scilly; the estimated breeding population of each is shown in Table 1. A further 17 sites were surveyed but no evidence of breeding was found (Table 2). The main Storm-

Table 1. The number of European Storm-petrel diurnal playback responses and AOSs on each of the Isles of Scilly that held breeding birds.
Tabel 1. Het aantal antwoorden van Stormvogeltjes op overdag afgespeelde roep en het aantal blijkbaar bezette nestplaatsen (AOS) op de afzonderlijke Scilly Eilanden waar Stormvogeltjes broeden.

Island site	Number of responses	AOS
Illiswilgig	1	3
Scilly Rock	5	14
Castle Bryher	6	17
Mincarlo	6	17
Men-a-vaur	7	20
Rosevean	13	37
Gorregan	17	49
Rosevear	20	57
Melledgan	49	140
Round Island	64	183
Annet	281 (see text)	938

petrel nesting habitat at most of these colonies was cavities in boulder beaches, with a few birds nesting in deep cracks in rock behind boulder piles (most notably on Men-a-Vaur). On Round Island, boulder beach was absent; 10 of the responses came from old stone walls, two from holes in the walls of the lighthouse and the remainder were in scree.

The boulder beaches on Annet produced a total of 276 responses, equating to 789 AOSs. The few cairns and rocky outcrops produced a further six responses. No responses were obtained from any of the 13 sample quadrats searched in the interior as this was generally unsuitable as nesting habitat, being covered with either thick bracken that was occupied by breeding gulls or thrift containing very few burrows.

For the sample quadrats searched along the coast of Annet, the mean bootstrapped count of burrow-nesting Storm-petrels per 10x10 m square was 0.1889 (lower 95% confidence limit (LCI) = 0.0095, upper 95% confidence limit (UCI) = 0.3683). The island perimeter was estimated at 3 986 m, with 2 752 m of this free of boulder beach. Extrapolating the bootstrapped AOSs density to this 27 520 m^2 strip returns an estimated total of 149 AOS (LCI = 8, UCI = 290). Combining this with the corrected population count for boulder beaches gives an estimated breeding population on Annet of 938 AOSs (LCI = 797, UCI = 1 079). Adding this figure to the sum totals found at other islands returns an estimated population of 1 475 (LCI = 1 055, UCI = 2 268) Storm-petrel AOSs in the Isles of Scilly in 2000.

Table 2. Isles of Scilly surveyed in 2000 at which no European Storm-petrel AOSs were found (no birds heard calling at night or no response to diurnal playback of call).
Tabel 2. Scilly Eilanden die in 2000 zijn geïnventariseerd, waar geen blijkbaar bezette nestplaatsen (AOS) van Stormvogeltje werden gevonden (geen vogels gehoord die 's nachts riepen of antwoordden op overdag afgespeelde roep).

Island site	Past presence?	Day/Night?
Daymark, St. Martin’s	Yes[4]	Day
Great Crebawethan	Yes[3]	Day
Great Ganilly	Not checked	Day
Gugh	Yes[1] 1943, [2] 50 pairs 974	Night
Gweal	Not checked	Day
Innisvouls	Not checked	Day
Maiden Bower	Not checked	Day
Menawethan	No[4]	Day
Nornour	Not checked	Day
Norwethel	Not checked	Night
Puffin Island	Not checked	Day
Samson	Not checked	Night
Seal Rock	Not checked	Day
St. Agnes	Yes[2] 50 pairs 1974	Night
St. Helen’s	Not checked	Night
Tean	Not checked	Night
White Island, St. Martin’s	Not checked	Day

[1]Penhalluric (1969), [2]Allen (1977), [3]Lloyd *et al.* (1991), [4]Robinson (1999).

Manx Shearwaters were found to be nesting on six of the islands in the Scilly archipelago; the numbers found at each is presented in Table 3. Eight others were checked for presence of AOBs but none were found (Table 4). On St. Helens, St. Agnes, Gugh and Bryher, birds were discovered breeding in burrows among rocky outcrops and at the edges of grassy slopes along the coast. On Round Island AOBs were recorded only in three gullies that held sufficiently deep turf to allow burrow excavation.

A total of 72 responses was elicited along the east coast of Annet. These were all from burrows and the majority (70%) were in the eroded, bare soil under overhanging vegetation at the very edge of the island. The others were mostly under tussocks of thrift within 2 m of the coastline, although some burrows were located slightly further inland in both bracken and long grass. No responses were elicited from the plots in the interior of the island. The combined bootstrapped estimate within the 10 m wide coastal strip away from the east coast and under the cliff overhang was 41 AOBs (LCI = 26, UCI = 56).

Combining the data for all six colonies results in a total estimated breeding population of 201 Manx Shearwater AOBs (LCI = 185, UCI = 217).

DISCUSSION

The total breeding population of Storm-petrels was estimated to be 1 475 AOSs. Previous estimates of the breeding population on Scilly range from as low as 500 pairs in 1987 (Lloyd *et al*. 1991) to over 1 900 pairs in 1974 (Allen 1977) and as many as 9 811 – 17 390 pairs in 1999 (Robinson 1999). However, these counts are not comparable to this survey, as they were based on guesses and rates of trapping in mist nets, and cannot be used to interpret trends. The numbers of birds recorded in this survey represent 1-7% of the British breeding population based on the totals in Lloyd *et al*. (1991). However, the British totals are based largely on guesswork and so the true importance of the Storm-petrel in the Scilly Isles can be confidently established only once playback surveys from other colonies in Britain are analysed. However, the Isles of Scilly is certainly the only known breeding locality of the Storm-petrel in England and probably hold over 1% of the UK population of this Annex 1 listed species; if so, the islands would thus qualify for Special Protection Area status.

The total breeding population of the Manx Shearwater on Scilly was estimated to be 201 pairs. This is considerably lower than previous population estimates, which range from 900 pairs in 1974 (Allen 1977) to 375-530+ pairs in 1977 (Harvey 1983) and 500-700 pairs in 1999 (Robinson 1999). Harvey (1983) reported that the population on Annet declined from 800-900 to 350-500 pairs and suspected that predation by the expanding gull population was to blame. This survey recorded an even lower total of 123 AOBs. Harvey (1983) also recorded 224 occupied burrows on Round Island, which is far higher than the total of 32 AOBs recorded there in this study. Although these could represent population declines, the differences in methodology among surveys prevent this conclusion being drawn unequivocally. Previous surveys did not deploy playback to confirm burrow occupation by Manx Shearwaters and so empty burrows or those occupied by non-breeders, rabbits *Oryctolagus cuniculus* and Atlantic Puffins *Fratercula arctica* (both the latter occur on Annet, but neither on Round Island) could have been included in the totals. These would not have been included in this survey so this could explain the apparent decline in population size.

The estimated breeding population of Manx Shearwaters in Britain during 1985 was 220 000-250 000 pairs (Lloyd *et al*. 1991), so the numbers on Scilly are not of importance in a UK context. However, the Isles of Scilly are important for Manx Shearwaters in an English context since the only other

Table 3. Number of Manx Shearwater diurnal playback responses and AOBs at each of the Isles of Scilly that held breeding birds.
Tabel 3. Het aantal antwoorden van Noordse Pijlstormvogels op overdag afgespeelde roep en het aantal blijkbaar bezette holen (AOB) op de afzonderlijke Scilly Eilanden waar Noordse Pijlstormvogels broeden.

Island site	Number of responses	AOBs
St. Helen's	5	5
Wingletang, St. Agnes	5	5
Shipman Head, Bryher	11	12
Gugh	20	22
Round Island	32	34
Annet	73 (see text)	123

Table 4. Isles of Scilly checked with no evidence of Manx Shearwater breeding presence (no birds heard calling at night or no response to diurnal playback of call).
Tabel 4. Scilly Eilanden zonder bewijs van de aanwezigheid van broedende Noordse Pijlstormvogels (geen vogels gehoord die 's nachts riepen of antwoordden op overdag afgespeelde roep).

Island site	Past presence?	Day/Night?
Burnt Island, St. Agnes	Not checked	Day
Daymark, St. Martin's	Not checked	Day
White Island, St. Martin's	Not checked	Day
Tean	Not checked	Both
Samson	Not checked	Both
Tresco, North End	Yes[1] 1945	Day
Norwethel	Not checked	Both
Gweal	Not checked	Day

[1]Penhalluric (1969).

breeding locality in the country is Lundy, where an estimated 1 200 pairs breed (Taylor 1985).

The distributions of the breeding colonies of both Storm-petrels and Manx Shearwaters generally accorded with previous information for the archipelago. No previously undocumented Storm-petrel colonies were found, but several sites that were believed to hold breeding birds in the past did not during this survey (Allen 1977; Robinson 1999). Some of these sites were headlands of inhabited islands and the presence of breeding birds at most sites was inferred from captures of birds in mist nets using tape lures (Robinson 1999). It is well-known that this method captures birds at sites where none breed (Maguire *et al*. 1980; Furness and Baillie 1981; Fowler *et al*. 1982, 1986; Fowler & Okill 1988; Harris *et al*. 1998), and given that the headlands and islands were frequented by rats or feral cats it seems unlikely that Storm-petrel

Table 5. The status of mammalian predators on selected Isles of Scilly during the 1990s and action taken to remove them (D. Moore pers. comm*.).*
Tabel 5. De status van zoogdierpredatoren op geselecteerde Scilly Eilanden in de jaren negentig en actie die is ondernomen om deze zoogdieren te verwijderen (D. Moore pers. med*).*

Island Site	Predators	Action
Great & Little Arthur	Rats	None
Great & Little Ganinick	Rats	None
Grt. & Little Innisvouls	Rats	Failed eradication attempt
Grt. Ganilly & Nornour	Rats	None
Guther's Island	Rats	Apparently successful eradication
Gweal	Rats	Apparently successful eradication
Little Ganilly	Rats	None
Menawethan	Rats	None
Norwethel	Rats	Apparently successful eradication
Puffin Island	Rats	Apparently successful eradication
Ragged Island	Rats	Apparently successful eradication
Samson	Rats and cats	Successful eradication
White Island, Samson	Rats and cats	Successful eradication
St. Helen's	Rats	Apparently successful eradication
Tean	Rats	None

colonies would persist there. Great Crebawethan was the only predator-free site previously reported as a breeding site (Lloyd *et al*. 1991) and that was not occupied in this survey. The island is so low-lying that it can be inundated by large swells, so nesting attempts could be flooded and abandoned in some years. The only Manx Shearwater breeding locality previously reported that was not confirmed in this survey was the north end of Tresco, and it is possible that mammalian predation has extirpated this small colony. A previously undocumented colony was found on the island of St. Helen's.

No island in the archipelago supported both rats and Storm-petrels during this survey. Small petrels are known to be very vulnerable to rat predation (Moors & Atkinson 1984) and the introduction of rats to islands generally produces rapid extirpation of breeding Storm-petrels. The conservation of Storm-petrels on the Isles of Scilly is dependent on preventing rats colonising those islands where they currently breed. This is particularly true of Annet and Round Island, where most of the Scilly population nest, and which have sufficient alternative sources of food to support a viable rat population over winter. Monitoring the continued absence of rats at these colonies and developing plans for their eradication should they colonise are essential to prevent potentially large declines. The other main colonies on the Western and

Northern Rocks have little or no vegetation and are washed over by winter storms, and are unlikely to sustain a rat population even if they were colonised.

Rats may limit the population size of Storm-petrels on the Isles of Scilly by rendering large areas of nesting habitat unsuitable for breeding. Many of the islands that host rats have large areas of boulder beach that are apparently suitable for breeding Storm-petrels. The Isles of Scilly Environmental Trust and the Isles of Scilly Seabird Group have conducted eradication work on several islands in the archipelago (Table 5). The colonisation of restored islands could arise through redistribution rather than population increase, and so the success of these projects needs to be assessed at the scale of the whole archipelago rather than at individual islands.

Small colonies of Manx Shearwaters breed on islands where both rats and cats are present (St. Agnes, Gugh and Bryher). Predation on eggs and chicks could be causing a chronic, long-term decline at these colonies. Rat predation caused the extirpation of the large colony on the Calf of Man (Brooke 1990) and has been associated with reduced productivity and population declines on Rum and Canna (Thompson *et al.* 1997, 1998; Upton *et al.* 2000). Maintaining or increasing the populations of Manx Shearwaters at these sites will depend on targeted control programmes around the colony during the breeding season as eradication on large and inhabited islands is generally impractical.

Gull predation is a potential threat to both Storm-petrels and Manx Shearwaters breeding on the Isles of Scilly. Harvey (1983) suggested that declines in Manx Shearwater populations on Annet could be due to gull predation. The remains of Manx Shearwaters that had been killed by gulls and gull pellets containing Storm-petrel feathers and bones were found on Annet, but the population level effects of this predation are unknown. Bioenergetic studies of the number of petrels that are consumed by gulls annually (Phillips *et al.* 1999) and modelling of the likely effects of this on the population are advisable.

Another predator recently introduced to the Isles of Scilly is the hedgehog *Erinaceus europaeus*. The adverse impacts of introduced hedgehogs on ground nesting birds has been documented (Jackson & Green 2000), and it is possible that they could also prey on burrow nesting birds where these are accessible. At present, hedgehogs have been recorded only on St. Mary's and although it may be too daunting a task to eradicate them from such a large island, it is vital that they are prevented from establishing elsewhere in the archipelago.

STATUS EN VERSPREIDING VAN STORMVOGELTJE *HYDROBATES PELAGICUS* EN NOORDSE PIJLSTORMVOGEL *PUFFINUS PUFFINUS* OP DE SCILLY EILANDEN

Dit artikel beschrijft de resultaten van de eerste uitgebreide inventarisatie naar de verspreiding en aantallen van broedende Stormvogeltjes en Noordse Pijlstormvogels op de Scilly Eilanden. Eilanden, waar uit het verleden meldingen van broedende vogels bekend zijn, werden geïnventariseerd door het overdag afspelen van respectievelijk de mannelijke purr-roep van Stormvogeltje en de roep van mannelijke en vrouwelijke Noordse Pijlstormvogels. Deze methode werd tevens gebruikt om andere eilanden met geschikt habitat te inventariseren. De totale broedpopulatie van Stormvogeltje bedroeg 1475 blijkbaar bezette nestplaatsen (AOS, gedefinieerd als iedere holte waaruit gereageerd werd op het afspelen van de roep) in 11 kolonies (tabel 1). Op 17 onderzochte eilanden werden geen Stormvogeltjes aangetroffen (tabel 2). De totale broedpopulatie van Noordse Pijlstormvogel bedroeg 201 blijkbaar bezette nestholen (AOB) verdeeld over zes eilanden (tabel 3). Voor beide soorten zijn deze aantallen van regionaal belang. De aantallen Stormvogeltjes zijn bovendien van internationaal belang. De verspreiding van broedende Stormvogeltjes was beperkt tot de ratvrije eilanden. De bescherming van Stormvogeltje op de Scilly Eilanden is gericht op het voorkomen dat ratten deze ratvrije eilanden koloniseren. Sommige kolonies van Noordse Pijlstormvogel werden gevonden op eilanden met ratten en verwilderde katten. Bescherming van de Noordse Pijlstormvogel is gericht op het gedurende het broedseizoen binnen de perken houden van predatoren rond de kolonies. De invloed van predatie door meeuwen op de populaties van beide stormvogels zou in kaart gebracht moeten worden. De egel tenslotte is een recent geïntroduceerde predator, waarvan uitbreiding voorkomen dient te worden.

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REFERENCES

Allen R. 1977. Population trends of seabirds in Scilly. Unpublished report to Nature Conservancy Council.

Bestwetherick A.T. 1968. Bird notes from Round Island. Cornwall Birdwatching and Preservation Society Annual Report 31: 62-64.

Brooke M de L. 1978a. Some factors affecting the laying date, incubation and breeding success of the Manx Shearwater *Puffinus puffinus*. J. Anim. Ecol. 47: 477-495.

Brooke M de L. 1978b. Sexual differences in the voice and individual recognition in the Manx Shearwater (*Puffinus puffinus*). Anim. Behav. 26: 622-629.

Brooke M. de L. 1990. The Manx Shearwater. T. & A.D. Poyser, London.

Ellis P., Ratcliffe N. & Suddaby D. 1998. Seasonal variation in diurnal attendance and response to playback by Leach's Petrels *Oceanodroma leucorhoa* on Gruney, Shetland. Ibis 140: 336-339.

Fowler J.A. & Hounsome M.V. 1998. Migration and arrival of immature Storm-petrels *Hydrobates pelagicus* in Shetland. Ringing & Migration 19: 91-94.

Fowler J.A. & Okill J.D. 1988. Recaptures of Storm-petrels tape-lured in Shetland. Ringing & Migration 9: 49-50.

Fowler J.A., Hulbert M.E. & Smith G. 1986. Sex ratios in a sample of tape lured Storm-petrels *Hydrobates pelagicus* from Shetland, Scotland. Seabird 9: 15-19.

Fowler J.A., Okill J.D. & Marshall B. 1982. A retrap analysis of Storm-petrels tape lured in Shetland. Ringing & Migration 4: 1-7.

Furness R.W. & Baillie S.R. 1981. Factors affecting capture rate and biometrics of Storm-petrels on St. Kilda. Ringing & Migration 3: 137-148.

Gibbons D.W. & Vaughan D. 1997. The population size of Manx Shearwater *Puffinus puffinus* on 'the Neck' of Skomer Island: a comparison of methods. Seabird 20: 3-11.

Gilbert G., Gibbons D. W. & Evans J. 1998a. Bird Monitoring Methods. RSPB, Sandy, UK.

Gilbert G., Hemsley D. & Shepherd M. 1998b. A survey of Storm-petrels on the Treshnish Isles in 1996. Scott. Birds 19: 145-153.

Harris P., Fowler J.A. & Okill J.D. 1998. Initial results of Storm-petrel *Hydrobates pelagicus* ringing in Portugal. Ringing & Migration 14: 133-134.

Harvey P. 1983. Breeding seabird populations, Isles of Scilly. Nature Conservancy Council Report. Taunton.

Jackson D.B. & Green R.E. 2000. The importance of the introduced hedgehog *Erinaceus europaeus* as a predator of the eggs of waders (Charadrii) on machair in South Uist, Scotland. Biol. Cons. 93: 333-348.

James P.C. & Robertson H.A. 1985. The use of playback recordings to detect and census nocturnal burrowing seabirds. Seabird 8: 18 - 20.

Lloyd C., Tasker M.L. & Partridge K. 1991. The status of seabirds in Britain and Ireland. T. & A.D. Poyser, London.

Maguire E.J., Zonfrillo B., Clark H. & Wilkins M. 1980. Status of Storm-petrels in Clyde and Forth. Scott. Birds 11: 51-53.

Mainwood A.R., Ratcliffe N., Murray S. & Mudge G.P. 1997. Population status of Storm-petrels *Hydrobates pelagicus* on islands off north-west Scotland. Seabird 19: 22-30.

Mayhew P., Chisholm K., Insley H. & Ratcliffe N. 2000. A survey of Storm-petrels on Priest Island. Scott. Birds 21: 78-84.

Moors P.J. & Atkinson I.A.E. 1984. Predation of seabirds by introduced animals and factors affecting it's severity. In: Croxall, J.P, Evans P.G.H. & Schreiber R.W. (eds) Status and conservation of the world's seabirds. pp 667-690. ICBP Technical Publication No. 2, Cambridge.

Penhallurick R.D. 1969. Birds of the Cornish coast. Bradford Barton, Truro.

Phillips R.A., Thompson D.R. & Hamer K.C. 1999. The impact of Great Skua predation on seabird populations at St. Kilda: a bioenergetics model. J. Appl. Ecol. 36: 218-232.

Ratcliffe N, Vaughan D., Whyte C. & Shepherd M. 1998a. Development of playback census methods for Storm-petrels *Hydrobates pelagicus*. Bird Study 45: 302-312

Ratcliffe N., Vaughan D., Whyte C. & Shepherd M. 1998b. The status of Storm-petrels on Mousa, Shetland. Scott. Birds 19:154-159.

Robinson P.J. 1999. Distribution of European Storm-petrel *Hydrobates pelagicus* in the Isles of Scilly with probable abundance. English Nature Report, Peterborough, UK.

Rodd E.H. 1880. The birds of Cornwall and the Scilly Islands. Turner & Co., London.

Scott D.A. 1970. The breeding biology of the Storm Petrel *Hydrobates pelagicus*. Unpublished D. Phil thesis, University of Oxford.

Taylor A.M. 1985. Manx Shearwaters on Lundy: ringing information and other observations. Rep. Lundy Field Soc. 36: 23-24.

Thompson K.R., Brindley E. & Heubeck M. 1997. Seabird numbers and breeding success in Britain and Ireland, 1996. Joint Nature Conservation Committee. (UK Nature Conservation, No 21). Peterborough.

Thompson K.R., Brindley E. & Heubeck M. 1998. Seabird numbers and breeding success in Britain and Ireland, 1997. Joint Nature Conservation Committee. (UK Nature Conservation, No 22). Peterborough.

Tucker G.M. & Heath M.F. 1994. Birds in Europe: Their Conservation Status. BirdLife Conservation Series No. 3. BirdLife International, Cambridge.

Upton A.J., Pickerell G. & Heubeck M. 2000. Seabird numbers and breeding success in Britain and Ireland, 1999. Joint Nature Conservation Committee. (UK Nature Conservation, No 24). Peterborough.

Vaughan D. & Gibbons D.W. 1998. The status of breeding Storm-petrels *Hydrobates pelagicus* on Skokholm Island in 1995. Seabird 20: 12-21.

Westfall P. H. & Young S.S. 1993. Resampling based multiple testing. Wiley, New York.

Wood D. 1997. An estimate of the numbers of Storm-petrels, *Hydrobates pelagicus* breeding on Auskerry, Orkney. Seabird 19: 40-46.

COLONY SIZE, ADULT SURVIVAL RATES, PRODUCTIVITY AND POPULATION PROJECTIONS OF BLACK-LEGGED KITTIWAKES *RISSA TRIDACTYLA* ON FAIR ISLE

P. ROTHERY[1], M.P. HARRIS[2*], S. WANLESS[2] & D.N. SHAW[3]

Rothery P., Harris M.P., Wanless S., & Shaw D.N. 2002. Colony size, adult survival rates, productivity and population projections of Black-legged Kittiwakes *Rissa tridactyla* on Fair Isle. Atlantic Seabirds 4(1): 17-28. *The numbers of Kittiwakes on Fair Isle, Shetland, Scotland declined at a rate of approximately 6% per annum between 1987 and 1999. Breeding success over this period was extremely variable but averaged 0.81 young reared per completed nest. Average annual survival of adults between 1986 and 1996 was 86.0% with no significant annual differences. Survival in 1997 (51.6%) was significantly lower and preliminary estimates for 1998 suggested that survival was again low. Using our empirical data for adult survival and breeding success, we estimated a 20% survival from fledging to recruitment and an age of first breeding of 4-5 years. Incorporating these values into a simple population model indicated that the Fair Isle colony will decline by a further 13-48% over the next three seasons.*

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INTRODUCTION

Black-legged Kittiwakes *Rissa tridactyla* are an important component of the marine avifauna of the North Atlantic and North Pacific. They are small, surface-feeding gulls with high foraging costs and a high risk of predation of unattended eggs and chicks, characteristics that render them particularly sensitive to changes in prey abundance (Monaghan 1996; Furness & Tasker 2000). The Kittiwake has therefore often been used as a bio-indicator to monitor changes in the marine environment. The numbers of Kittiwakes nesting at many colonies in the North Sea have declined substantially over the last 10-15 years. Similar declines have been recorded at colonies in the far north-west of Britain, e.g. Handa and St. Kilda, whereas counts of nests at some colonies on Atlantic and Irish Sea coasts, e.g. Rathlin Island and the Cliffs of Moher, have increased substantially (details in Upton *et al.* 2000). Comparisons of temporal and spatial trends in demographic rates could assist in the diagnosis of the causes of declines but as yet there are too few published data on the survival of adults to allow this.

Most monitoring studies have concentrated on assessing changes in the size of the breeding population and breeding success (see listings in Upton *et al.* 2000; Byrd *et al.* 2000) but a few have also monitored the survival of adults between breeding seasons (Coulson & Strowger 1999; Danchin & Monnat 1992; Erikstad *et al.* 1998; Poole *et al.* 1998; Harris *et al.* 2000). At colonies where an integrated approach to monitoring has been adopted it is possible to carry out simple population modelling both retrospectively and to provide population projections. In this paper we use this approach for the colony of Kittiwakes on Fair Isle (59°32'N, 1°38'W) at the southern edge of the Shetland Islands where numbers, adult survival and productivity have been monitored annually since 1986 as part of the UK Joint Nature Conservation Committee's Integrated Seabird Monitoring Programme.

METHODS

Breeding population size and breeding success The number of occupied nests in 10 fixed plots dispersed through the colony were counted each year between 1987 and 1999 and the total used as an annual index of population size (JNCC contract reports; Heubeck *et al.* 1999). Complete censuses of all the nests on the island were made in 1988, 1992, 1997 and 2000. In these years, there was a high correlation between the whole-island counts and the plot totals (r = 0.99, P = 0.017), which suggests that changes in the numbers of nests in these plots were representative of the population as a whole. Data on breeding success (number of young fledged per completed nest) resulted from standardised checks of nests in these study plots throughout each breeding season in 1986 (five of the plots) and between 1987 and 1999 (all 10 plots; details of methodology in Harris 1987). Annual values were the means of the plot means.

Adult survival Starting in 1986, breeding Kittiwakes at South Gunnawark were caught and marked with unique combinations of three colour-rings. A total of 177 individuals were marked, and the average number of colour-ringed birds present in the colony at the end of each season between 1986 and 1990 was 109 (range 72-119). Searches were made for these birds each subsequent year up to 1994 and the data used to calculate annual survival rates between 1986 and 1991. In 1991, the study site was moved to Goorn where access was easier, thereby allowing more frequent checks to be made. A total of 91 birds was marked and the annual average number of colour-ringed birds present at the end of each season was 39 (range 24-48). Thorough searches were made for these birds each year up to 1999 and these observations were used to calculate survival rates from 1991 to 1997.

Annual survival and resighting probabilities were estimated using the program SURGE 4.2 (Pradel & Lebreton 1993). The analysis cannot separate mortality and permanent emigration. However, Kittiwakes normally exhibit high colony fidelity once they have bred (Aebischer & Coulson 1990; Fairweather & Coulson 1995; Golet *et al.* 1998; but see Danchin & Monnat 1992) and we had no reason to believe that permanent emigration was important in our study.

We followed Lebreton *et al.* (1992) in determining survival rates by fitting models of increasing complexity and using the Akaike Information Criterion for model selection. The goodness-of-fit of the Cormack-Jolly-Seber model with time-dependent survival and resighting probability was examined using TESTS 2 & 3 of the program RELEASE (Burnham *et al.* 1987). However, as the resighting probability was high most of the birds were resighted in the first year after release. In this case the component 3.Sm of TEST 3 and TEST 2 are not informative, and most of the information relating to goodness-of-fit resides in TEST 3.SR (Lebreton *et al.* 1992). The values are as follows: 1986-1992: $\chi^2_3 = 5.39$, $P = 0.15$; 1991-1999: $\chi^2_3 = 5.35$, $P = 0.15$; combined periods: $\chi^2_6 = 10.64$, $P = 0.10$. Thus, there was only weak evidence to suggest that subsequent survival depended on whether the bird had been previously resighted. Furthermore, the high resighting probability suggests that the survival estimates should be robust to heterogeneity in resighting.

During 1986-1991 the most parsimonious model had time-dependent survival of both resighting rates and survival, whereas in 1991-1998, there was again time-dependent resighting rate but a constant survival rate. Likelihood ratio tests were used to calculate the statistical significance of differences between years in survival and resighting probabilities. For convenience, survival between two years is referred to by the former year, i.e. 1995 survival refers to the survival between the 1995 and 1996 breeding seasons.

Population model We used a simple matrix model in which the size of the breeding population in a given year was expressed as the sum of the breeding pairs surviving from the previous year plus the number of recruits. For example, if birds breed for the first time as 4- or 5-year olds (the usual age of first breeding; Porter & Coulson 1987), the model is:

$$N_t = s_{At} N_{t-1} + s_{4t} Y_{t-4} + s_{5t} Y_{t-5}$$

where N_t denotes the number of breeding pairs in year t, Y_t denotes the number of young females fledged, s_{At} is adult survival, and s_{4t} and s_{5t} are the survival rates from fledge to recruitment into the breeding population as 4- and 5-year olds respectively. The model assumes a 50:50 sex ratio in young and a closed population in which adults breed annually. Average recruitment rates (s_4 and s_5) were estimated by a two-variable linear regression of N_t - s_{At} N_{t-1} against Y_{t-4} and Y_{t-5}, with no intercept, using the estimated pattern of adult survival and breeding success.

The fit of the model was examined using stepwise and free-running predictions. In the former, population size in a given year is predicted from the observed population size in the previous year. This effectively highlights particular years in which predictions are poor. In the free-running method, population size in a given year is predicted by repeatedly applying the model to the population size at the beginning of the series. This method was used to obtain long-term population projections.

RESULTS

Breeding population size and breeding success The total number of nests in the plots declined by almost 50% from 1 446 in 1987 to 751 in 1999. This represented a significant annual decrease averaging 6% (log-transformed counts, r = 0.98, n = 13 years, P < 0.001) (Fig. 1). Breeding success over the study period averaged 0.81 young per completed nest. However, productivity was extremely variable, ranging from total failure in 1990 to 1.33 in 1999. There was no obvious time trend in this parameter (Fig. 2).

Adult survival The average (± SE) annual survival during the study was 83.2 ± 3.4% (Table 1). There was no statistically significant trend over the period 1986-1996 (r = 0.40, P = 0.26) but the 1997 survival of 51.6 ± 17.4% was significantly lower than the mean for 1986-1996 (86.0 ± 2.0%; z = 1.97, P = 0.05). However, the 1997 survival rate was based on only 43 birds and should therefore be treated with caution. The program does not allow the calculation of a survival rate for 1998 but only 13 (54%) of the 24 birds present in 1998 had been resighted by the end of 2000, which suggests that survival had again been low. There was no evidence that overwinter survival was related to breeding success the previous season (r = 0.09, n = 12 years, ns) but there was a suggestion of a relationship between breeding success and survival over the previous winter such that poor survival was followed by low success (r = 0.87, n = 12 years, P = 0.09).

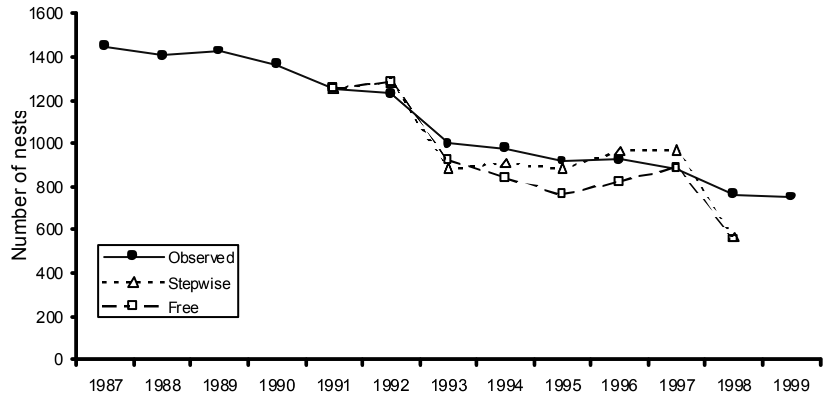

Figure 1. A comparison of the observed numbers of Black-legged Kittiwake nests in study plots on Fair Isle between 1987-1999 (filled circles, solid line) and those predicted using the stepwise method (triangles) and the free-running prediction (squares). Details of the models are given in the text.

Figuur 1. Een vergelijking tussen het waargenomen aantal nesten van Drieteenmeeuwen in studiegebieden op Fair Isle, 1987-1999 (stippen, doorgetrokken lijn) en de voorspelde aantallen volgens de getrapte methode (driehoeken) en de vrije voorspelling (vierkanten). Een beschrijving van de voorspellende modellen wordt in de tekst gegeven.

Population model Average (± SE) recruitment rates were estimated as follows: s_4 = 0.11 ± 0.12, t_5 = 1.11, P = 0.32, s_5 = 0.13 ± 0.11, t_5 = 1.11, P = 0.32. Assuming all birds breed for the first time at age 5 years resulted in s_5 = 0.20 ± 0.08, t_6 = 2.53, P = 0.045. For age 4 years, the corresponding value was s_4 = 0.18 ± 0.07, t_5 = 2.41, P = 0.047. The overall estimated recruitment rate was about 20% in each case, but the relative proportions of birds breeding for the first time at age 4 and 5 years are estimated rather imprecisely. In a stable population

1 – *adult survival = breeding success x survival to recruitment / 2*

Using our most realistic survival estimate of 0.86 (below) and an average breeding success of 0.81 chicks per pair, survival to recruitment would need to be 0.35 to maintain numbers. The predicted population trend for Fair Isle Kittiwakes using the stepwise method and values of age of first breeding at 5 years and survival from fledging to recruitment at 20%, accorded well with the observed changes in the monitoring plots (Fig. 1). The prediction was poorest in

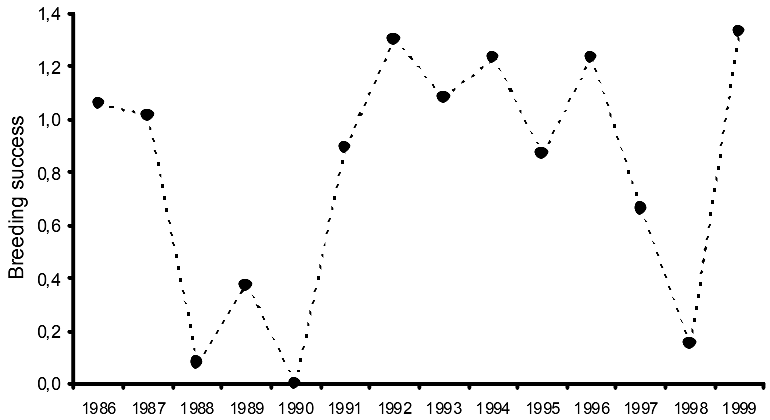

Figure 2. Breeding success (chicks fledged per completed nest) of Black-legged Kittiwakes on Fair Isle, 1986-1999.

Figuur 2. Broedsucces (aantal uitgevlogen jongen per nest) van Drieteenmeeuwen op Fair Isle, 1986-1999.

1998 with the observed population size being markedly higher than that predicted. Very similar results were obtained using the free-running model.

Population projections Fitting the models to the Kittiwake data produced sufficiently realistic estimates of the observed population changes to attempt to use the free-running model to predict population trends over the subsequent five seasons (2000-2004). We ran the model using a range of values for adult survival and recruitment rate (Table 2). With an adult survival rate of 86% (mean during the period 1986-1996, i.e. taking a realistic approach and omitting the low and imprecise estimate for 1997), a 20% survival from fledging to recruitment and age of first breeding of 5 years, the model predicted a population decline of 17% between 1999 and 2004 (Table 2). Substituting an age of first breeding of 4 years and keeping the same adult survival and recruitment rate, a 15% decline between 1999 and 2003 was predicted. We also took a more pessimistic approach and assumed that the recorded decrease in survival at the end of the study was a real effect, and ran the model assuming that survival was 75% (the geometric mean of the last three survival estimates). The corresponding projected declines for a 20% survival from fledging to recruitment and age of first breeding of 5 years and 4 years were 48% and 43% respectively.

Table 1. Estimates of annual survival and resighting probabilities for adult Black-legged Kittiwakes on Fair Isle, 1986-1998. Data were collected at different study sites, 1986-1991, and 1991-1998.

Tabel 1. Schattingen van de jaarlijkse overleving en van de kans op terugmelding voor volwassen Drieteenmeeuwen op Fair Isle, 1986-1998. De gegevens werden in verschillende studiegebieden verzameld: 1986-1991 en 1991-1998.

Period	Year	Survival % (SE)	Resighting % (SE)
1986-1992	1986	93.8 (2.5)	-
	1987	81.9 (3.6)	100 (0.0)
	1988	83.8 (3.7)	96.2 (2.1)
	1989	81.0 (3.6)	89.0 (3.5)
	1990	87.3 (3.3)	96.7 (1.9)
	1991	-	100 (0.0)
1991-1999	1991	90.8 (6.1)	-
	1992	70.8 (9.3)	94.4 (5.4)
	1993	85.9 (5.3)	100 (0)
	1994	90.5 (5.2)	91.7 (4.6)
	1995	93.0 (3.9)	82.1 (6.6)
	1996	87.5 (4.8)	91.3 (4.2)
	1997	51.6 (17.4)	100 (0)
	1998	-	42.2 (16.3)
	mean 1987-1997	83.2 (3.4)	90.3 (4.7)
	mean 1986-1996	86.0 (2.0)	94.7 (1.7)

We were able to make an initial check of the population projections from the model by comparing the plot total recorded in 2000 with the predicted values. A total of 716 nests (4.7% lower than the 1999 count) was recorded for the plots, 5.8% less than the number predicted using an adult survival rate of 0.86 and first breeding at four years (760 nests) and 11.4% more than the predicted value (643) using an adult survival of 0.75 and first breeding at 5 years (Table 2).

DISCUSSION

The average annual survival of Fair Isle Kittiwakes between 1986 and 1996 (86.0%) fell within the range of values reported elsewhere in the east Atlantic, e.g. 80% in north-east England (40 years of data; Coulson & Strowger 1999) and in northern Norway (6 years; Erikstad *et al.* 1998), 81% in Brittany (5 years; Danchin & Monnat 1992), 87% in south Wales (18 years; Poole *et al.* 1998) and 88% in south-east Scotland (11 years; Harris *et al.* 2000). Similarly

Table 2 Black-legged Kittiwake population projections for the monitoring plots on Fair Isle for 2000-2004 using a range of model scenarios. The number of recruits entering the population is based on the numbers of pairs in the plots and their nesting success 4 or 5 years earlier and a 20% survival from fledging to recruitment. The figure in brackets after the total is the geometric rate of change from the previous year.

Tabel 2. Voorspellingen van de populatie Drieteenmeeuwen in de studiegebieden voor de jaren 2000-2004 op basis van verschillende scenario's. Het aantal rekruten in de populatie is gebaseerd op het aantal broedparen in de studiegebieden en hun broedresultaten 4 of 5 jaar eerder, uitgaande van een 20% overlevingskans tussen uitvliegen en terugkeer in de kolonie. Het getal tussen haakjes is de geometrische verandering vergeleken met het voorafgaande jaar.

Adult survival rate	Year	Age of first breeding					
		4 years			5 years		
		Adults	Recruits	Total	Adults	Recruits	Total
	1999	-	-	751	-	-	751
	2000	646	114	760 (1.01)	646	80	726 (0.97)
86%	2001	654	58	712 (0.94)	624	114	738 (1.02)
	2002	612	11	623 (0.88)	635	58	692 (0.94)
	2003	536	99	635 (1.02)	596	11	607 (0.88)
	2004	-	-	-	522	99	621 (1.02)
	1999	-	-	751	-	-	751
	2000	563	114	677 (0.90)	563	80	643 (0.86)
75%	2001	508	58	566 (0.84)	482	114	596 (0.93)
	2002	425	11	436 (0.77)	447	58	505 (0.85)
	2003	327	99	426 (0.98)	379	11	390 (0.77)
	2004	-	-	-	293	99	392 (1.01)

the mean breeding success on Fair Isle (0.81 young per nest) was close to or slightly above that for most British colonies (Walsh *et al.* 1994; Upton *et al.* 2000; Heubeck *et al.* 1999). There was, however, considerable year-to-year variation in the estimates of both adult survival and breeding success on Fair Isle. Our study included the period at the end of the 1980s when breeding success of many Shetland seabirds was severely depressed (Heubeck 1989) and the run of poor breeding years is apparent in the Fair Isle data-set (Fig. 2).

These widespread breeding failures were attributed to poor recruitment of the lesser sandeel *Ammodytes marinus,* the only small shoaling fish available to Kittiwakes and other seabird species breeding in the area (Wright 1996). In accordance with this, there was a positive association between the breeding success of Kittiwakes on Fair Isle and the numbers of sandeels found in fishery

Nesting Black-legged Kittiwakes Nestelende Drieteenmeeuwen (S.C.V. Geelhoed)

tows around Shetland in June in nine years (data for 9 years from ICES 1997, 1998: numbers log-transformed, $r = 0.66$, $P = 0.052$).

Inspection of estimates of adult survival suggested that the largest changes occurred at the end of the study period. Indeed, the value for 1997 was one of the lowest recorded for the species. The available evidence suggests that this was not simply a one-off event since the preliminary estimate for 1998 was also low. Great Skuas *Stercorarius [Catharacta] skua* kill substantial numbers of adult Kittiwakes in Shetland, and in the late 1990s such predation was thought to be having a serious effect on the survival of Kittiwakes (Furness 1997, quoted by Heubeck 2000; Heubeck 2000). Predation of eggs and young and the associated disturbance of colonies could also have contributed to the desertion of some colonies (Hamer *et al* 1991; Heubeck *et al.* 1997).

We have no quantitative information on the effects of skuas on Fair Isle Kittiwakes during the period of our study; although some adults and young were killed on the island, the colonies where adult survival was being monitored were not particularly disturbed. Clearly the continued monitoring of adult survival at this colony is of high conservation priority. In contrast to breeding success, adult survival was not correlated with sandeel numbers during the summer. The only correlation that approached statistical significance (at the 10% level) was the tendency for poor over-winter survival to be followed by poor breeding success.

Average breeding success at the colony was not unusually low, which suggests that pre-breeding survival rates may have been low and/or chicks from Fair Isle were emigrating to other colonies. At present, there are insufficient data to investigate these two possibilities; many breeding populations in Shetland and Orkney are currently also declining so it is difficult to see where these missing recruits might have gone (Heubeck 2000; Thompson & Walsh 2000). Using the empirical data on adult survival and breeding success, the recruitment rate of Kittiwakes on Fair Isle was estimated to be about 20%. In an earlier study, Porter and Coulson (1987) calculated that about 40% of the chicks produced at a colony in north-east England, over a period when pairs normally fledged at least one chick each season, needed to return to nest to maintain a stable population. Currently a survival to recruitment of about 35% should enable the Kittiwake population on Fair Isle to remain stable; however, it appears to be well below this level. Our predicted population trends accorded well with observed changes in the monitoring plots and hence presumably in the colony. Predictions were markedly poorer in 1998 when the observed population size was substantially higher than predicted. The cause of this divergence appeared to stem from the very low (and imprecise) survival rate estimated for 1997. This further emphasises the need to clarify the later survival estimates.

Monitoring counts on Fair Isle indicate that the colony has been declining by about 6% *per annum* since 1987. Our results suggest that this decline will continue with the projected decrease over the next 3 years varying between 13 and 48% *per annum*. In general, adult survival rates and breeding success have not been particularly low, thus suggesting that poor recruitment was responsible for the decline. Monitoring programmes at many other colonies bordering the west coast of the North Sea have also shown that numbers have declined substantially over the last 10-15 years (Heubeck *et al.* 1999; Upton *et al.* 2000). A full analysis of the available data is now needed to see if these declines also might have been due to poor recruitment.

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KOLONIE-GROOTTE, OVERLEVING, VAN VOLWASSEN VOGELS, PRODUCTIE EN POPULATIEVOORSPELLINGEN VAN DRIETEENMEEUWEN OP FAIR ISLE

Het aantal broedparen van Drieteenmeeuwen op Fair Isle (Shetland Eilanden, Schotland) nam van 1987 tot 1999 af met 6% per jaar. Het broedsucces in deze periode was buitengewoon variabel maar bedroeg gemiddeld 0.81 jongen per afgebouwd nest. De overleving van volwassen vogels in de jaren 1986-1996 bedroeg 86% per jaar, zonder dat er significante verschillen tussen de jaren konden worden gevonden. De overleving in 1997 (51.6%) was echter significant lager en ook de voorlopige schattingen voor 1998 suggereren dat de overleving relatief laag was. Op grond van de in dit artikel gepresenteerde gegevens werd de overleving van jongen (tussen uitvliegen en recrutering in de broedpopulatie op een leeftijd van 4-5 jaar) geschat op 20%. Gebruik makend van een eenvoudig populatiemodel voorspellen de auteurs dat de kolonie op Fair Isle gedurende de komende drie seizoenen met nog eens 13-48% zal afnemen.

REFERENCES

Aebischer N.J. & Coulson J.C. 1990. Survival of the Kittiwake in relation to sex, year, breeding experience and position in the colony. Journal of Animal Ecology 59: 1063-1071.

Burnham K.P., Anderson D.R., White G.C., Brownie C. & Pollock K.H. 1987. Design and analysis methods for fish survival experiments based on release-recapture. American Fisheries Society, Monograph 5; Bethesda, Maryland, USA.

Byrd G.V., Dragoo D.E. & Irons D.B. 2000. Breeding status and population trends of seabirds in Alaska in 1999. U.S. Fish and Wildlife Service Report AMNWR 2000/02.

Coulson J.C. & Strowger J. 1999. The annual mortality rate of Black-legged Kittiwake in NE England from 1954 to 1998 and a recent exceptionally high mortality. Waterbirds 22: 3-13.

Danchin E. & Monnat J-Y. 1992. Population dynamics modelling of two neighbouring Kittiwake *Rissa tridactyla* colonies. Ardea 80: 171-180.

Erikstad K.E., Anker-Nilssen T., Barrett R.T. & Tveraa T. 1998. Demography and adult survival in some Norwegian seabird populations. NINA Oppdragsmelding 515: 1-15.

Fairweather J.A. & Coulson J.C. 1995. The influence of forced site change in the dispersal and breeding of the Black-legged Kittiwake *Rissa tridactyla*. Colonial Waterbirds 18: 30-40.

Furness R.W. 1997. The impact of predation by Great Skuas on other seabird species, with particular reference to Special Protection Areas in Shetland. Final Report on contract 5092AA, Scottish Natural Heritage, Lerwick.

Furness R.W. & Tasker M.L. 2000. Seabird fishery interactions: quantifying the sensitivity of seabirds to reductions in sandeel abundance, and identification of key area for sensitive seabirds in the North Sea. Marine Ecology Progress Series 202: 263-264.

Golet G.H., Irons D.B. & Estes J.A. 1998. Survival costs of chick rearing in black-legged kittiwakes. Journal of Animal Ecology 67: 827-841.

Hamer K.C., Furness R.W. &. Caldow R.W.G. 1991. The effects of changes in food availability on the breeding ecology of the Great Skuas *Catharacta skua* in Shetland. Journal of Zoology, London 223: 175-188.

Harris M.P. 1987. A low-input method of monitoring Kittiwake *Rissa tridactyla* breeding success. Biological Conservation 41: 1-10.

Harris M.P., Wanless S. & Rothery P. 2000. Adult survival rates of the Shag *Phalacrocorax aristotelis*, Guillemot *Uria aalge*, Razorbill *Alca torda*, Puffin *Fratercula arctica* and Kittiwake *Rissa tridactyla* on the Isle of May 1986-96. Atlantic Seabirds 2: 133-150.

Heubeck M. 1989. Breeding success of Shetland's seabirds: Arctic Skua, Kittiwake, Guillemot, Razorbill and Puffin. In: Heubeck, M (ed) Seabirds and sandeels: proceedings of a seminar held in Lerwick, Shetland, 15-16th October 1988: 11-18. Lerwick, Shetland.

Heubeck M. 2000. Population trends of Kittiwake *Rissa tridactyla,* Black Guillemot *Cepphus grylle* and Common Guillemot *Uria aalge* in Shetland, 1978-98. Atlantic Seabirds 2: 227-244.

Heubeck M., Mellor R.M. & Harvey P.V. 1997. Changes in the breeding distribution and numbers of Kittiwakes *Rissa tridactyla* around Unst, Shetland, and the presumed role of predation by Great Skuas *Stercorarius skua*. Seabird 19: 12-21.

Heubeck M., Mellor R.M., Harvey P.V., Mainwood A.R. & Riddington R. 1999. Estimating the population size and rate of decline of Kittiwakes *Rissa tridactyla* breeding in Shetland, 1981-97. Bird Study 46: 48-61.

ICES 1997 Report on the Working Group on the assessment of demersal stocks in the North Sea and Skaggerak. ICES CM1997/Assess:6. International Council for the Exploration of the Seas, Denmark.

ICES 1998 Report on the Working Group on the assessment of demersal stocks in the North Sea and Skaggerak. ICES CM1998/Assess:7. International Council for the Exploration of the Seas, Denmark.

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

Monaghan P. 1996. Relevance of the behaviour of seabirds to the conservation of marine environments. Oikos 77: 227-237.

Poole J., Smith S., Perrins C.M., Birkhead T.R. & Thompson K.R. 1998. Seabird monitoring on Skomer Island in 1996. JNCC Report No 277.

Porter J.M. & Coulson J.C. 1987 Long-term changes in recruitment to the breeding group, and the quality of recruits at a kittiwake *Rissa tridactyla* colony. Journal of Animal Ecology 56: 675-689.

Pradel R. & Lebreton J-D. 1993. User's manual for Program SURGE Version 4.2. Centre d'Ecologie Fonctionelle et Evolutive, C.N.R.S., Montpellier, France.

Thompson K.R. & Walsh P.M. 2000. Population trends and breeding success of cliff-nesting seabirds in Orkney, 1976-98. Atlantic Seabirds 2: 103-132.

Upton A..J., Pickerell G. & Heubeck M. 2000. Seabird numbers and breeding success in Britain and Ireland, 1999. Joint Nature Conservation Committee, Peterborough (UK Nature Conservation No. 24.)

Walsh P.M.., Brindley E. & Heubeck M. 1994. Seabird numbers and breeding success in Britain and Ireland, 1993. Peterborough, Joint Nature Conservation Committee. (UK Nature Conservation, No 17).

Wright P. 1996. Is there a conflict between sandeel fisheries and seabirds? A case study at Shetland. In: Greenstreet S.P.R. & Tasker M.L (eds). Aquatic predators and their prey: 154-165. Fishery News Books, Oxford.

EFFECT OF FOOD SHORTAGE ON THE REPRODUCTIVE OUTPUT OF COMMON EIDERS *SOMATERIA MOLLISSIMA* BREEDING AT GRIEND (WADDEN SEA)

RENÉ OOSTERHUIS[1] & KLAAS VAN DIJK[2]

Oosterhuis R. & Van Dijk K. 2002. Effect of food shortage on the reproductive output of Common Eiders *Somateria mollissima* breeding at Griend (Wadden Sea). Atlantic Seabirds 4(1): 29-38. *Following a food-induced major mortality of Common Eiders* Somateria mollissima *in the Dutch Wadden Sea in winter and spring 1999/2000, the reproductive output at Griend was evaluated in comparison with previous seasons. In 2000, the number of breeding pairs showed a decline of 38% and female Common Eiders commenced breeding some 2-3 weeks later than in three previous years. Mean clutch size in 2000 (4.6 eggs clutch^{-1}) was similar to 1999 (4.9 eggs clutch^{-1}), but the hatching probability declined from 0.41 in 1999 to 0.18 in 2000. In 2000, the majority of the nests were deserted prior to hatching, often a few weeks after incubating had started. The number of chicks hatched in 2000 was only a quarter of that in 1999. Observations elsewhere in the Dutch Wadden Sea indicated similar poor breeding results. We suggest that a food shortage in winter and spring was the principal cause for the low reproductive output of Common Eiders in 2000. Adult females probably failed to accumulate sufficient energy stores needed for their prolonged fast during laying and incubation.*

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INTRODUCTION

In winter and spring 1999/2000, an estimated 21 000 Common Eiders *Somateria mollissima* died in the Dutch Wadden Sea. The dead birds were in a very poor condition, and a shortage of the harvestable fraction of the food supply was suggested as the principal cause of the emaciation (Camphuysen *et al.* 2002; Van den Berk *et al.* 2000). Common Eiders are specialised feeders on large benthic invertebrates, mainly blue mussels *Mytilus edulis* and common cockles *Cerastoderma edule* (Swennen 1976). Recently, cut trough shells *Spisula subtruncata* in the adjacent North Sea coastal zone have been used as an alternative prey in some years (Leopold *et al.* 2001).

Adult female Common Eiders prepare for breeding by storing extensive body reserves which are subsequently utilised during egg-laying and incubation (Milne 1976; Parker & Holm 1990). Females fast during incubation

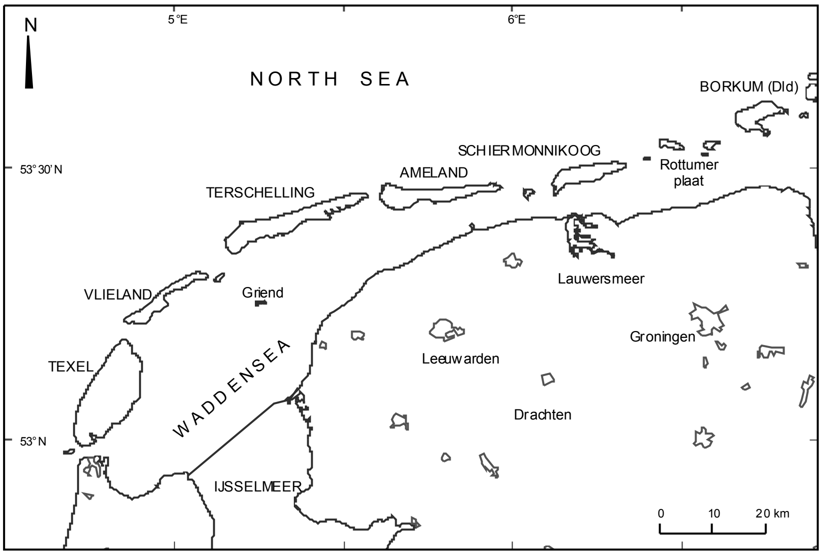

Figure 1. Map of the (Dutch part of the) Wadden Sea showing the location of Griend and other sites mentioned in the text.
Figuur 1. Kaart van de Waddenzee met de ligging van Griend.

and leave the nest only occasionally to preen or to drink some water (Parker & Holm 1990; Swennen *et al.* 1993). Hence, foraging conditions prior to the breeding season can be a critical factor influencing breeding success (Erikstad *et al.* 1993; Christensen 2000). The effect of the possible food shortage is evaluated in this paper by comparing reproductive output of Common Eiders nesting at the island of Griend prior to and following the major mortality.

METHODS

The study was conducted at Griend (53°15'N, 05°15'E; Fig. 1), a National Nature Reserve in the centre of the western Dutch Wadden Sea (Veen & Van de Kam 1988). The Vereniging Natuurmonumenten manages the island. It became a Ramsar site in 1980 and forms part of the Wadden Sea Special Protection Area, designated under the EC Wild Birds Directive in 1991 and the EC Habitats Directive in 1996. Griend is uninhabited and has no public access throughout the year. The vegetated part covers about 55 ha. Griend supports large numbers of colonial nesting gulls and terns, and their numbers and breeding success are monitored annually. Two wardens reside permanently on the island during the breeding season. In 1999 and 2000, special attention was paid to the breeding biology of the Common Eider (Oosterhuis 2000). The study periods were 21 April to 15 July 1999 and 11 April to 18 July 2000. Some additional data for 1997 and 1998 were derived from warden reports (Brenninkmeijer & Van Tienen 1997; Van Tienen & Baarspul 1998).

Systematic beached bird surveys were carried out each year in the first week after the arrival of the wardens. In 2000, the survey was also subsequently carried out on a weekly basis. All corpses were recorded and marked by clipping the primaries to avoid double counts during later surveys. We used some additional ringing data of Common Eiders found dead on the island before the wardens arrived.

The number of breeding pairs of Common Eiders was mainly assessed by recording all nests found. The construction of a new sand dyke in 1988 led to an increase in the surface of Griend and subsequently to a considerable increase in suitable breeding habitat. It became difficult to find all nests and since 1989 a varying percentage, depending on the amount of time spend in the field, has been added for undiscovered nests. In 1999 and 2000, the wardens searched two to three times per week for nests of scattered breeding species by visiting different parts of the island. All nests of Common Eiders were marked by an inconspicuous thin, 1 m long yellow stake placed 1 m north of the nest. Nests were visited 3-4 times (on average every 10 days) and to avoid disturbance, known nests were approached carefully. Only when there was no female present was the nest checked to determine whether the eggs had hatched, had been taken by predators, or if the nest had been deserted. Nest survival rates were estimated using the Mayfield method (Mayfield 1975). Nests were considered successful if at least one egg had hatched. We used a breeding period of 32 days (egg laying 5 d, incubation 27 d). Females with freshly hatched ducklings assemble along the beach, which was checked daily for Common Eiders with ducklings. The small size of Griend together with the excellent view from the wardens house made this check both easy and reliable.

RESULTS

In 1999, 23 dead Common Eiders were found in the first week after arrival, about the same number as in 1997 (27) and 1998 (31). About 10-20 dead individuals were subsequently recorded during the 1999 breeding season. In 2000, large numbers of dead Common Eiders were found on the beach when the wardens arrived on 11 April and 334 corpses were counted in the first weeks after arrival. Later that season, 52 new dead birds were found; 29 in the first half

Table 1. Ringing details of Common Eiders (all ringed as nestlings) found dead on Griend.
Tabel 1. Ringgegevens van doodgevonden Eiders (alle als kuiken geringd) op Griend.

Ring #	Ringing place	Ringed	Found dead	Sex
Arnhem 7.070.969	Vlieland, NL, 53°15'N; 04°56'E	1979	18 Feb 2000	m
Arnhem 7.073.841	Schierm.'oog, NL, 53°28'N; 0°613'E	1980	08 Mar 2000	m
Arnhem 7.074.174	Terschelling, NL, 53°24'N; 05°29'E	1980	27 Jan 2000	f
Arnhem 7.074.621	Terschelling, NL,53°24'N; 05°29'E	1981	02 May 1999	m
Arnhem 7.077.673	Terschelling, NL,53°24'N; 05°29'E	1984	13 Apr 2000	m
Arnhem 7.078.706	Terschelling, NL, 53°27'N; 05°25'E	1985	16 Apr 2000	m
Kalø 460830	Samsø, DK, 55°53'N; 10°37'E	1995	16 Apr 2000	m
Helsinki DX 017.168	Porvoo, Finl., 60°07'N; 25°25'E	1999	21 Feb 2000	f

of May, 15 in the second half of May, 4 in June and 4 in July. The majority of the birds found in May and June were freshly dead. None of about 300 dead Common Eiders examined had oil in their feathers. All checked corpses were severely emaciated with severe atrophy of the breast muscle. Seven ringed birds were found in 2000, while only one ringed bird had been found in 1999 (Table 1). All were ringed as nestlings, six at various colonies in the Dutch Wadden Sea, two were from Finnish and Danish breeding populations.

Common Eider nests were widely distributed all over the island. The construction of a new sand dyke in 1988 led to a considerable increase in the surface of Griend and subsequently to an increase in the number of breeding Common Eiders (Fig. 2). A maximum of 68 breeding pairs were recorded in 1999. In 2000, the number of breeding pairs had fallen to 42 pairs, a 38%

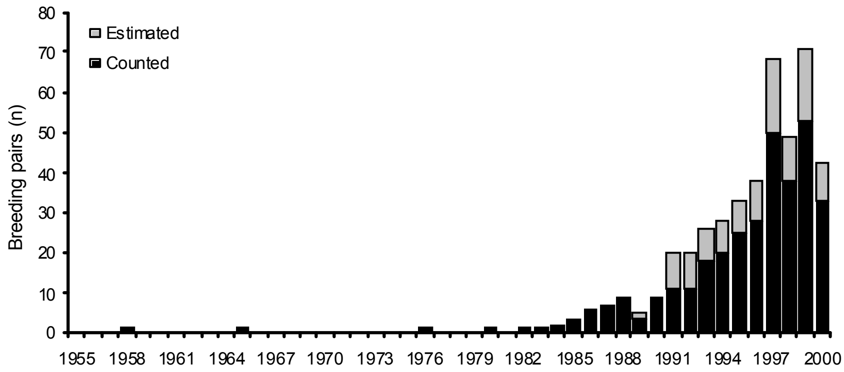

Figure 2. Number of breeding pairs of the Common Eider at Griend in 1955-2000.
Figuur 2. Aantal broedparen van de Eider op Griend tussen 1955 en 2000.

Table 2. Number of breeding pairs and reproduction data of Common Eiders at Griend. [1]calculated data.
Tabel 2. Aantal broedparen en reproductiegegevens van Eiders op Griend. [1]berekende gegevens.

Parameter	1999	2000
Number of breeding pairs	68	42
Laying date of first egg[1]	2 April	21 April
Date of first chicks	4 May	22 May
Mean clutch size (n = 25 / 21)	4.9	4.6
Range	3-6	3-6
n nests observed	21	31
n failed clutches	5	21
n nest days	176	376
Daily survival rate (p)	0.9724	0.9471
Hatching probability (%)	40.8	17.6
n chicks born[1]	136	34

decline. The first nests were found on 21 April 1999 and on 24 April 2000. In 1999, the wardens arrived on 21 April, so nests started earlier could not have been found in that year, while in 2000 the wardens arrived on 11 April. In 1999, the first female with four freshly hatched ducklings was observed on 4 May, a similar date to 1997 (1 May) and 1998 (28 April). In 2000, the first female with five freshly hatched ducklings was observed on 22 May suggesting that egg-laying commenced 2-3 weeks later than in three previous years.

The mean clutch size in 2000 (4.6 eggs clutch^{-1}) was slightly lower than in 1999 (4.9 eggs clutch^{-1}; Table 2). Clutch size varied between 4.8 and 4.2 eggs clutch^{-1} in 1997 (n = 40) and 1998 (n = 27) respectively. Hatching probability declined from 0.41 in 1999 to 0.18 in 2000 (Table 2), with nest desertion being the main reason for the difference between the two years. In 2000, at least 18 nests (out of 31 nests observed) were deserted prior to hatching, often a few weeks after incubating started. These nests were covered with down but the eggs were cold and wet. We estimated (number of breeding pairs x mean number of eggs x hatching probability) that about 136 ducklings hatched in 1999 and about 34 in 2000, a 75% decline. In 1999, no predation on Common Eider chicks was seen, while 14 ducklings were seen to be taken by Herring Gulls *Larus argentatus* in 2000.

DISCUSSION

Numbers of dead Common Eiders washed ashore at Griend in 2000 were about ten times higher than normal, indicating that birds staying around Griend also suffered a substantially increased mortality as recorded elsewhere in the Dutch Wadden Sea. At Griend, no oiled birds were found and all checked birds were severely emaciated. This corresponds with Werkman (2001) and Camphuysen *et al*. (2002), who found no toxicological, bacteriological, or virological explanations for the major mortality. We have no indications that these factors could explain the increased mortality at Griend.

Although our data are based on a small number of nests (albeit a large proportion of the local breeding population), there have been no detailed studies of breeding success elsewhere in the Wadden Sea following the major mortality in 1999/2000. A combination of all our results shows that the reproductive output of Common Eiders at Griend in 2000 was much lower than in previous years.

Dying Common Eider at Griend, June 1999 Stervende Eider op Griend, juni 1999 (*R. Oosterhuis*)

Earlier studies used hatching success as it was assessed in a traditional manner: the number of nests hatched divided by the number of nests found. We recalculated our Mayfield hatching success data to estimate traditional hatching success (0.76 in 1999, 0.32 in 2000), showing that the hatching success at Griend in 2000 was also very low in comparison to several earlier studies. Traditional hatching success at Vlieland averaged at 0.85 during 1959-1979 (range 0.66-0.96; *n* = 13 years of study; Swennen 1983). Swennen listed hatching success for Finland, Sweden, Germany and Scotland and most figures were similar to his data and to hatching success as assessed at Griend in 1999. Relatively low values were found at Vlieland during 1962-1968, when a high mortality of (nesting) adult female Common Eiders occurred, as a result of pollution of the Wadden Sea with pesticides. Swennen never recorded hatching success as low as at Griend in 2000, but cited one value that approached our 2000 results. This result (hatching success between 0.15 and 0.57) came from a study at Amrum (German Wadden Sea) in the 1950s, when nest desertion was mainly induced by predation and frequent human disturbance.

Observations elsewhere in the Dutch Wadden Sea in 2000 indicated similar poor breeding results. At Rottumerplaat, Lutterop & Kasemir (2001) reported that Common Eiders commenced breeding approximately 1-2 weeks later than normal and the first ducklings were observed on 20 May, while this normally occurred between 5 and 10 May. The number of breeding females found at Rottumerplaat was 39% lower than in 1999 and large numbers of non-breeding adult females were observed at and around the island, a phenomenon not seen in previous years. Lower than normal numbers of breeding females were reported from standard sampling areas on Vlieland, Terschelling, Ameland and Schiermonnikoog, and field workers in these areas had the impression that the nesting season was significantly delayed and that very few ducklings were successfully reared (Dijksen & Koks 2001). Moreover, counts of adult and sub-adult males in April and May at Vlieland and Schiermonnikoog (Dijksen & Koks 2001) and at Rottumerplaat (Lutterop & Kasemir 2001) showed a lesser decrease than was found using nest counts, which Dijksen & Koks (2001) suggested as indicating that a considerable number of adult females were in such a bad condition that they were not able to breed.

Other breeding bird species at Griend showed no indications of a poor breeding season. Common Shelduck *Tadorna tadorna*, Black-headed Gull *Larus ridibundus*, Sandwich Tern *Sterna sandvicensis* and Common Tern *S. hirundo* exploit other food sources than Common Eiders and all had an average to good breeding season in both years (Baarspul & Oosterhuis 1999; Oosterhuis & Heideveld 2000). This indicates that factors like weather, human disturbance and predation had no adverse effects on the breeding results in 2000. Eurasian Oystercatchers *Haematopus ostralegus* are common breeding birds at Griend with 440 breeding pairs in 1999 and 490 in 2000 and are specialised feeders on common cockles and blue mussels, as are Common Eiders. Eurasian Oystercatchers have experienced major mortality, reduced breeding success and population declines in and around the Dutch Wadden Sea as a result of a more or less chronic shortage of food since the early 1990s (Camphuysen *et al*. 1996; Nève & Van Noordwijk 1997; Smit *et al*. 1998; Smit *et al*. 2000), but a major mortality of this species did not occur in winter 1999/2000 and there was no reduction in reproductive output at Griend in 2000, with hatching probabilities of 0.70 in 1999 and 0.66 in 2000. However, the number of adult non-breeders waiting for a vacant territory has recently decreased sharply and the situation at Griend may have been less favourable than it seems (Van Dijk & Oosterhuis 2001).

De Vlas (1982) found evidence for a negative relationship between laying date of Common Eiders at Vlieland and common cockle resources in the Wadden Sea and indicated that common cockles are an important prey item in spring for pre-breeding females, which were supposed to quickly increase their body mass by foraging on an easily accessible prey such as common cockles of the right size. There is evidence (Swennen 1976) that Common Eiders prefer small-sized common cockles. The number of first year common cockles in the Dutch Wadden Sea was very low in the autumn of 1999 (Van den Berk *et al*. 2000), suggesting that pre-breeding females might not be able to find enough common cockles of their preferred size.

Although there may be another explanation, we believe the available information suggests a shortage of food in winter and early spring as the cause of the low reproductive output of the Common Eider at Griend in 2000. Many authors have argued that gross industrial shellfisheries in the Dutch Wadden Sea on common cockles and blue mussels and on cut trough shells in the adjacent North Sea has negative effects on the amount of food harvestable for molluscivorous birds in the Wadden Sea like the Common Eider and the Eurasian Oystercatcher (Camphuysen *et al*. 1996; Nève & Van Noordwijk 1997; Beukema *et al*. 1998; Smit *et al*. 1998; Piersma *et al*. 2001; Camphuysen *et al*. 2002). Our findings underline the need for a critical evaluation of the industrial shellfishery in the Dutch Wadden Sea, which is a National Nature Reserve, a Ramsar site, and a Special Protection Area under the EC Wild Birds Directive and under the EC Habitats Directive. Unfortunately, the breeding biology of Common Eiders in the Dutch Wadden Sea has not been studied in recent years so that the observations at Griend are rather unique. Our findings show the utmost importance of annual assessments of breeding performance and hatching success of the Common Eider, one of the most important molluscivorous bird in the Wadden Sea. Like our study, this monitoring project should follow the standard guidelines of the 'Nestkaartenproject' of SOVON Dutch Centre for Field Ornithology.

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GEVOLGEN VAN VOEDSELSCHAARSTE IN DE WADDENZEE OP DE BROEDPOPULATIE VAN DE EIDER *SOMATERIA MOLLISSIMA* OP GRIEND

Nadat in de winter van 1999/2000 ongeveer 21 000 Eiders Somateria mollissima *in de Nederlandse Waddenzee door verhongering waren gestorven, werd het broedsucces op Griend bepaald en vergeleken met gegevens uit eerdere jaren. In 2000 daalde het aantal broedparen ten opzichte van het piekjaar 1999 met 38%. De wijfjes begonnen 2 tot 3 weken later te broeden dan in normale jaren. De gemiddelde legselgrootte (4.6 eieren per nest) week niet af van die in voorgaande jaren (4.8 in 1997, 4.2 in 1998, 4.9 in 1999), maar de uitkomstkans van de legsels daalde van 41% in 1999 naar 18% in 2000. In 2000 werden veel nesten voortijdig verlaten, vaak enkele weken nadat met het broeden was begonnen. We denken dat de ongunstige voedselsituatie in de Waddenzee de meest waarschijnlijke verklaring is voor de gevonden verschillen. Helaas werd het broedsucces alleen in de kleine kolonie op Griend bepaald. Minder exacte gegevens uit andere delen van de Waddenzee laten overigens dezelfde trend zien. We pleiten voor het starten van een monitoringproject binnen het Nestkaartenproject van SOVON om het broedsucces op meerdere locaties jaarlijks vast te stellen.*

REFERENCES

Baarspul A.N.J. & Oosterhuis R. 1999. Griend, vogels en bewaking 1999. Warden report, Wageningen.

Beukema J.J., Cadée G.C. & Dekker R. 1998. How Two Large-Scale "Experiments" Illustrate the Importance of Enrichment and Fishery for the Functioning of the Wadden Sea Ecosystem. Senckenbergiana maritima 29: 37-44.

Brenninkmeijer A. & Van Tienen P.G.M. 1997. Griend, vogels en bewaking. Warden report, Arnhem.

Camphuysen C.J., Ens B.J., Heg D., Hulscher J.B., Van der Meer J.& Smit C.J. 1996. Oystercatcher *Haematopus ostralegus* winter mortality in the Netherlands: the effect of severe weather and food supply. Ardea 84A: 469-492.

Camphuysen C.J., Berrevoets C.M., Cremers H.J.W.M., Dekinga A., Dekker R., Ens B.J., Van der Have T.M., Kats R.K.H., Kuiken T., Leopold M.F., Van der Meer J. & Piersma T. 2002. Mass mortality of common eiders (*Somateria mollissima*) in the Dutch Wadden Sea, winter 1999/2000: starvation in a commercially exploited wetland of international importance. Biol. Conserv. *in press*.

Christensen T.K. 2000. Female pre-nesting foraging and male vigilance in Common Eider *Somateria mollissima*. Bird Study 47: 311-319.

De Vlas J. 1982. De effecten van de kokkelvisserij op de bodemfauna van Waddenzee en Oosterschelde. Report 82/19, RIN, Arnhem.

Dijksen L. & Koks B. 2001. The Breeding Season in 2000 for Common Eiders in the Dutch Wadden Region. Wadden Sea Newsletter 23: 11-13.

Erikstad K.E., Bustnes J.O. & Moum T. 1993. Clutch-size determination in precocial birds: a study of the Common Eider. Auk 110:623-628.

Leopold M.F., Kats R.K.H. & Ens B.J. 2001. Diet (preferences) of Common Eiders *Somateria mollissima*. Wadden Sea Newsletter 23: 25-31.

Lutterop D. & Kasemir G. 2001. Vogelwachtersverslag Rottumerplaat Broedseizoen 2000. Warden report, Staatsbosbeheer, Assen.

Mayfield H.F. 1975. Suggestions for calculating nest success. The Wilson Bulletin 87: 456-466.

Milne H. 1976. Body weights and carcass composition of the Common Eider. Wildfowl 27: 115-122.

Nève G. & Van Noordwijk A. 1997. Factors affecting Oystercatcher (*Haematopus ostralegus*) survival rate in the Dutch Wadden Sea area. Report, NIOO, Heteren.

Oosterhuis R. 2000. Broedsucces van Bergeend *Tadorna tadorna* en Eider *Somateria mollissima* op Griend. Broednieuws 11: 3-5.

Oosterhuis R. & Heideveld S. 2000. Griend, Vogels en Bewaking 2000. Warden report, Arnhem.

Parker H. & Holm H. 1990. Patterns of nutrient and energy expenditure in female Common Eiders nesting in the high Arctic. Auk 107: 660-668.

Piersma T., Koolhaas A., Dekinga A., Beukema J.J., Dekker R. & Essink K. 2001. Long-term indirect effects of mechanical cockle-dredging on intertidal bivalve stocks in the Wadden Sea. J. Appl. Ecol. 38: 976-990.

Smit C.J., Dankers N., Ens B.J. & Meijboom A. 1998. Birds, Mussels, Cockles and Shellfish Fishery in the Dutch Wadden Sea: How to Deal with Low Food Stocks for Eiders and Oystercatchers? Senckenbergiana maritima 29: 141-153.

Smit C.J., Ens B.J. & Koks B.J. 2000. Afnemende aantallen Scholeksters in de Waddenzee. SOVON Nieuws 13(3): 16-17.

Swennen C. 1976. Populatie-structuur en voedsel van de Eidereend *Somateria m. mollissima* in de Nederlandse Waddenzee. Ardea 64: 311-371.

Swennen C. 1983. Reproductive output of Eiders *Somateria mollissima* on the southern border of its breeding range. Ardea 71: 245-254.

Swennen C., Ursem J.C.H. & Duiven P. 1993. Determinate laying and egg attendance in Common Eiders. Orn. Scand. 24: 48-52.

Van den Berk V.M., Dirksen S. & Poot M.J.M. 2000. Sterfte onder eidereenden in de Waddenzee 1999 - 2000. Report W-186, Expertisecentrum LNV, Wageningen.

Van Dijk K. & Oosterhuis R. 2001. Scholeksters op Griend in het broedseizoen 1999 en 2000. Twirre 12: 11-16.

Van Tienen P.G.M. & Baarspul A.N.J. 1998. Griend, vogels en bewaking 1998. Warden report, Wageningen.

Veen J. & Van de Kam J. 1988. Griend vogeleiland in de Waddenzee. Natuurmonumenten, 's Graveland.

Werkman G. 2001. Water Quality and Common Eiders in the Dutch Wadden Sea - Is there a Relationship between Water Quality and the Mortality of Common Eiders in the Winter of 1999/2000? Wadden Sea Newsletter 23: 22-24.

News and notices

BOOK REVIEWS

CRESSWELL G. & WALKER D. 2001. *Whales & Dolphins of the European Atlantic, the Bay of Biscay and the English Channel.* Ocean Guides, Wild Guides Ltd, Old Basing. Price £ 8,=, 56pp, semi-hardback.

There is a new book, with an old title: Whales & Dolphins, how many books carry that name? But this, this is different. Here we have an incredible must-have for anyone sailing the North Atlantic every once in a while and for anyone with an interest in cetaceans. It is handy and small, it is informative and authoritative, it is glossy and colourful, it is a breakthrough using modern digital artwork, it is an extraordinary collection of photographs at the same time and, oh wonder, it's cheap too! Just buy this little identification guide, you will never regret it and you support this young organisation manning Bay of Biscay ferries with skilled observers, collecting highly valuable data. Don't hesitate, ignore the rest of this review for my sake, but make sure you place the order.

With nearly a metre of cetacean field guides on my bookshelf, I was never able to pick my choice when I went to sea again, being arrogant enough to know better than what was depicted or what was described. Photo collections were always incomplete, or better examples should have been chosen. Images showed the entire beast rather than what was actually visible 'in the field'. As a tourist guide, you had to hold your hand over 90% of the image to try and explain what was to be seen and how it looked in real life. Graeme Cresswell and Dylan Walker must have had the same experience, but better still, they did solve the problem by producing the ultimate little guide for their working area! With pages of photo's facing text pages, they did not accept the problem of having to pick one picture out of a selection to show what they wished to show. Instead, they manipulated the available images to such an extend that you now have for example "a pod" of blue whales at the surface, with all individuals in a different and highly informative pose: blowing, showing blowhole, showing the back, from the side and from an angle, and lobtailing. No suitable shot of Pygmy Sperm Whale available? Just make one! A shot of some calm sea, a little artwork implanted and there we are: perfectly comparable with a genuine shot of the Dwarf Sperm Whale in the same pose on the same page! I have only one wish, Graeme and Dylan: do give us the entire North Atlantic in a guide, including the (sub-)tropical whales and dolphins and with the Narwhal and Beluga. I can not wait to buy that one!!

Kees Camphuysen

YÉSOU, P. & SULTANA, J. 2000. *Monitoring and Conservation of Birds, Mammals and Sea Turtles of the Mediterranean and Black Seas. Proceeding of the 5th Medmaravis Symposium, Gozo, Malta.* Environment Protection Department, Malta. 320 pages. ISBN 99909-65-03-X. Price £20 or US$30 including p&p. Available from Birdlife Malta, 57/28 Marina Court, Abate Rigor Street, Ta' Xbiex MSD 11, Malta.

This volume contains a review of the legal protection given to Mediterranean and Black Sea seabirds, mammals and turtles, 13 papers on seabird distribution and biology, eight on seabird study techniques, six on important sites for seabirds, and 10 on sea turtles or marine mammals. All papers refer to populations or sites in the Mediterranean Sea or the Black Sea. The book can therefore be recommended to anyone interested in the seabirds, mammals or turtles of these regions as an important source of up to date information on status and biology. Several of the papers review knowledge of the rare endemic species of the region such as the Balearic Shearwater *Puffinus mauretanicus*, a species reduced to about 3000 breeding pairs, of which 75% nest on Formentera. Unlike most shearwaters, this species feeds extensively on fishery discards. In common with many other shearwaters, it is threatened by introduced alien mammals, loss of nesting habitat, oil pollution, long-line by-catch and possibly by overexploitation of clupeid stocks. Threats to the Great White Pelican *Pelecanus onocrotalus*, Dalmatian Pelican *Pelecanus crispus*, Pygmy Cormorant *Phalacrocorax pygmeus*, Armenian Gull *Larus armenicus*, Audouin's Gull *L. audouinii*, Mediterranean Gull *L. melanocephalus*, Slender-billed Gull *L. genei*, and Gull-billed Tern *Sterna nilotica*, and the population status of these species are reviewed in a series of chapters that complete the first, and largest, part of the book. There then follows a variety of techniques-based chapters, including studies of Mediterranean seabird distributions at sea, studies based on data loggers, and Mediterranean shearwater phylogeny based on mitochondrial DNA sequences. Sections on seabird sites and on turtles and marine mammals provide an eclectic range of papers, with information on many sites that will be little known to most Seabird Group members, but provide some mouth-watering facts and figures, such as the Audouin's Gull colony that also holds 37 breeding pairs of Eleonora's Falcon *Falco eleonorae* on only 40 ha of islands some 12 km from the mainland coast, despite the islands' being "continually disturbed [by] large numbers of tourists"!

I strongly encourage anyone interested in Mediterranean or Black Sea seabirds to buy this book. It contains a wealth of information on status, distribution and biology, with particular reference to threats and conservation action.

Bob Furness