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Executive summary

The Working Group on Seabird Ecology (WGSE) met for five days from 3–7 April 2006 in Texel, The Netherlands, and was attended by sixteen persons from ten countries (Annex 1). Thirteen were nominated members of the group and three were invited by the WG Chair to attend this year’s meeting. During the meeting WGSE was able to address all the terms of reference, though in varying detail and the results are reported here.

Identifying and delineating protected areas for seabirds have become an important issue during the past years. WGSE explored the different approaches taken by designating Special Protection Areas (SPAs), following the EU Birds Directive, and Important Birds Areas (IBAs) according to Birdlife International. Both concepts are presented in this chapter, as well as many scientific and practical issues involved. Further on, overviews of recent developments in nine countries are given.

Seabird reproductive performance was extremely poor in the northwestern North Sea in 2004 and partly so in 2005. WGSE gives an overview of which species were affected and to what extent, and also whether signals of worse breeding success were visible in other parts of the North Sea and Scotland. Furthermore, possible explanations for the decrease in reproductive output are explored.

Closely connected to the chapter above, and following on from discussions on the Ecological Quality Objectives for seabird populations at the 2004 WGSE meeting, recommendations were developed on which monitoring activities need to be carried out to understand seabird population trends. This includes definition of priorities, selection of species and parameters, and an overview of sampling design approaches. A brief account of current monitoring activities in the ICES area is presented, highlighting that a more thorough survey of current activities via a questionnaire will be useful.

Information on diet of seabirds has been essential not only for exploring many topics in seabird ecology but also for input into multispecies models. As a variety of methods exist that investigate diet qualitatively and quantitatively, WGSE developed an overview of the different methods and their respective advantages and disadvantages. Apart from the methods involved, the different details and formats of the results of such studies vary substantially and a start was made to develop recommendations on how these might be standardised. This will be completed at the next WGSE meeting.

In addition to these four main terms of reference, WGSE also dealt again with the EcoQO on plastic particles in fulmars. WGSE followed the suggestion to slightly alter the parameter for future monitoring activities. Furthermore, WGSE explored the option to publish another Cooperative Research Report and suggested alternative approaches to disseminate information gathered by this group.
2 **Introduction**

2.1 **Participation**

The following members of the Working Group on Seabird Ecology (WGSE) participated in the meeting (see Annex 1 for full information).

Tycho Anker-Nilssen Norway
Pep (J. M.) Arcos Spain
Rob Barrett Norway
Thierry Boulinier France
Kees Camphuysen The Netherlands
John Chardine Canada
Morten Frederiksen UK
Stefan Garthe (Chair) Germany
Ommo Hüppop Germany
Mardik Leopold The Netherlands
Manuela Nunes Portugal
Ib Krag Petersen Denmark
Iván Ramírez Portugal
Norman Ratcliffe UK
Jim Reid UK
Richard Veit USA

Thirteen persons were nominated members of the group; three persons were invited by the Working Group Chair to attend this year’s meeting. The possibility to appoint persons not yet nominated by national delegates was considered by the group to be an extremely useful tool.

2.2 **Terms of Reference**

The 2005 Statutory meeting of ICES gave the Working Group on Seabird Ecology the following terms of reference:

a) review the current approaches for identifying offshore seabird aggregations and delineating Important Bird Areas (IBAs) and Special Protection Areas (SPAs);

b) review the breeding success of seabirds in the northwestern North Sea in 2004 and 2005 and explore the reasons for the poor performance;

c) produce recommendations for a comprehensive monitoring programme for seabirds;

d) produce recommendations on how to sample diet and how to report results of dietary studies in seabirds, and provide diet information to SGMSNS for multispecies modelling work;

e) determine potential for a state of the art report as a Cooperative Research Report, and report to LRC.

Furthermore, a request from OSPAR on the EcoQO on plastic particles was passed on to WGSE to comment on a possible change in the metrics.

Because of the substantial work load related to the first four terms of reference we were not able in all cases to include all aspects as we wished. Thus, at the beginning of the respective chapters, information is given on how it was dealt with the respective terms of references.

2.3 **Note on bird names**

Throughout the text we provide common English names for bird species. A full list of both English and scientific names is given in Annex 4.
2.4 Acknowledgements

The Working Group wishes to thank the Royal Netherlands Institute for Sea Research for providing us with a meeting room, copying and computer facilities and for other valuable logistic support. The Group also thanks Kees Camphuysen and Mardik Leopold for making the local arrangements for our meeting. We furthermore thank the ICES Secretariat for information.
3 Current approaches for identifying offshore seabird aggregations and delineating Important Bird Areas (IBAs) and Special Protection Areas (SPAs)

The delineation of Important Bird Areas (IBAs) and Special Protection Areas (SPAs) has become an important tool for nature conservation in Europe and elsewhere in recent years. The application of these concepts to marine areas has started relatively late and is not straightforward due to various reasons. In this chapter, we outline the different issues that are most relevant for recognition of IBAs and SPAs at sea. Within the restricted time available at the WGSE meeting it was not possible to cover all aspects in as much detail as needed. Instead, this version should be treated as a discussion document outlining the most relevant topics. It awaits completion at the WGSE meeting in spring 2007.

3.1 Introduction

3.1.1 Definitions

There are several types of protected areas for a variety of marine system components. In the marine environment, the abbreviation MPA, if/when used here, merely denotes any marine protected area. Very often, however, a MPA would refer to an area at sea that is protected primarily for habitats or fish stocks. The principal instrument in the European Union for the classification of important habitats is the Habitats and Species Directive (EC, 1992). This piece of legislation provides not only for the designation of sites for important habitats for the whole range of organisms including birds, but also sites for most taxa. Such sites are known as Special Areas of Conservation (SAC). A related piece of legislation, the Birds Directive (EEC, 1979), provides for the protection of birds. So, under the Birds Directive, a Special Protected Area (SPA) is an area specifically classified for its important bird interest.

Together, SPAs and SACs are known as Natura 2000 sites. There may be political as well as practical conservation benefits to identifying sites that qualify as both SPAs and SACs.

An Important Bird Area (IBA) also denotes an area that hosts birds in significant concentrations. However, there are notable differences between the designation SPA and the accolade IBA.

Articles 4.1 and 4.2 of the Birds Directive require that Member States classify “the most suitable territories in number and size as special protection areas” for those bird species included in Annex I of the Directive and also for regularly occurring migratory species of bird, taking account of their protection requirements at sea as well as on land. The SPA concept therefore is a legally binding one: failure to classify SPAs for important bird populations by Member States of the EU carries the risk of legal proceedings being instituted against those Member States. SPAs are defined by applying criteria devised by state governments or their advisors.

The concept of an Important Bird Area applies globally rather than just within Europe. IBAs carry no legal weight; areas accorded such status are assessed using criteria compiled by non-government organisations, principally BirdLife International and its partners.

In practice, there may be a great deal of overlap between SPAs and IBAs, certainly in the broad areas so identified if not their exact boundaries (for example, RSPB 2005). Such an overlap is reinforced at a European level where a specific category within the IBA criteria was adapted to the EU bird populations (category C). Both these types of protected area aim to protect discrete concentrations of birds. What constitutes a discrete concentration as opposed to a widely dispersed pattern of birds is a question of scale, and ultimately the distinction to be made becomes a subjective judgement. However, that is not to say that concentrations may not...
be identified using “objective” and repeatable methods that result in operational definitions that can be applied in consistent ways (see below). The important point is that the protection requirements of species must be addressed. Depending on the nature of species dispersion it is sometimes important to fulfil these requirements via site designation and sometimes through wider conservation measures. In fact, in most cases it will be desirable to combine site-specific, species-specific and activity-specific approaches to guarantee the protection of the species.

3.1.2 Guidelines and criteria for identifying SPAs and IBAs

Compilation of a network of SPAs in the marine environment has reached only a relatively early stage across Europe; the focus over the past three decades has been firmly on terrestrial, freshwater and inter-tidal sites. In the absence of prescriptive advice from the EU, rules and guidelines for the identification and classification of SPAs across the European Union have varied across Member States. However, many states have adopted similar approaches. Given that the process is now fairly well established for terrestrial/freshwater/inter-tidal habitats over most of the EU, and also for the reasons of consistency, there has been a presumption that those guidelines formulated for non-marine environments should, as far as possible, be applied to the marine environment. Hence, little distinction has been made between the two environments, though simple transposition of the terrestrial model into marine areas is not without its difficulties (see Section 3.1.3).

Guidelines that have been applied to determining whether sites qualify as SPAs, certainly in those EU Member States that have made significant progress in compiling a suite of SPAs, tend to specify that specific proportions or numbers of relevant populations be represented within the sites. For example, areas that are used regularly by 1% or more of the national or biogeographical populations of species in any seas on, or that host particularly high densities, or contain more than 20,000 waterfowl or seabirds in any season, are deemed as meriting classification as SPAs. Notwithstanding this, there always remains scope for applying judgements based on other sorts of (reliable) information on species’ ecology and life histories. A more comprehensive outline of SPA qualification guidelines is contained in Stroud et al. (2001) and EU (in prep.).

In contrast to SPAs, the criteria used to assess whether a site qualifies as an IBA, are universally applicable. Again, these criteria are expressed in terms of 1% thresholds of relevant populations including flyways, and numbers of individuals (with 20,000 again being the preference). There is also a significant emphasis on the perceived threat or conservation status of the species. In common with the SPA issue, the IBA criteria were devised specifically with regard to bird populations inhabiting terrestrial, freshwater and intertidal habitats. Their application to marine ecosystems faces similar difficulties (see below). See BirdLife International (2005) and SPEA–SEO/BirdLife (2005) for full details of IBA criteria.

It will be clear that while the two concepts are very similar and share common purposes, SPAs and IBAs differ in their respective philosophical provenances. Both the IBA and SPA (or networks thereof) identification processes will be tempered by political considerations, on the one hand by non-government/lobbying organisations whose principal remit will be the strict protection of birds and their habitats, and on the other by state governments whose objectives might be influenced by wider political concerns. In any event, the elevation of sites to IBA or SPA status should as far as possible be done by applying sound scientific practices.

Inasmuch as no generic, prescriptive rules exist across the EU for the identification of SPAs. Governments may choose whether or not equate their IBAs with SPAs. While consistency of approach across the EU might be desirable for many reasons, it is perhaps not possible politically.
3.1.3 Problems with applying standard criteria to marine areas

Although the IBA and SPA criteria have been developed for all biomes, most aspects were derived from experiences on land and in the coastal zone, with the latter usually being viewed from a land perspective. For that reason there are specific problems when dealing with birds at sea.

1. Information on distribution and abundance of birds at sea is not as good as that for birds on land. Studies of seabirds at sea started late compared to most habitats and areas on land. Substantial progress has been made in the North Atlantic over the past three decades though, and many marine regions are now quite well known (e.g. North Sea). However, seasonal constraints tend to result in more bird surveys being conducted in summer than in winter months and coverage varies according to distance from shore. Many areas (at small scales of analysis) have been surveyed infrequently if at all, and therefore represent “snapshots” of marine bird distribution and abundance. This means it is difficult to determine whether sites are used consistently or ephemerally. Furthermore, sea areas are vast and remote from land and thus there are hardly any options to sufficiently cover such areas even once. The Macaronesian Sea and the northwest Atlantic are two such examples.

2. The activity range of seabirds – although very variable – tends to be much larger than on land, even if compared to birds of prey that have rather huge territories. Chiefly, the extent of spatial use of the seas is related to foraging activities and migration. In contrast to most birds feeding on land, foraging activities of breeding seabirds may reach as far as several hundred kilometres from the colonies as in the case of most procellariiforms (Shealer, 2002). Because of such ranges, it will be difficult to include such extensive foraging ranges into an IBA/SPA perspective. However, many species have much shorter foraging ranges, especially terns and auks (e.g. Pearson, 1968; Garthe, 1997). For wide-ranging species, most foraging areas may very likely be disconnected from the seabirds’ colonies, thus requiring independent protection.

3. Seabirds generally have dispersed distributions but some species in some circumstances do associate with habitat features (see Section 3.3.1). However, such habitats are often comparatively large-scale and the site boundaries rather indistinct and often also mobile compared to terrestrial, freshwater and intertidal areas. This results in relatively low spatial stability in seabird concentrations, which makes site-based conservation difficult, especially at smaller scales.

3.2 Methods of data collection

Recognition of IBAs and SPAs depends on identifying sites where birds occur and the numbers of birds associated with these, and this requires data. Data can be collected specifically to identify SPAs or IBAs or data on distribution and abundance from other sources (e.g. Environmental Impact Assessments for windfarm or oil developments) can be collated and used for this purpose. Specifically collected data has obvious advantages, since scales and coverage surveys can be designed to address site recognition. However, coverage and scales from other surveys can be adequate for this purpose, in which case lengthy and expensive survey work can be avoided. In some cases, gaps may exist in available data which need to be filled by dedicated survey, in which case the two sources of data are complementary.

There are two broad methods of collecting such data on seabird distribution and these are described in the following sections.

3.2.1 Transect survey

Open waters are too extensive to allow complete counts of birds inhabiting them, such that a sampling of densities followed by extrapolation has to be employed. Transect counts are the
most common means of sampling seabird density at sea. This provides data on the population scale, with the distribution of a large proportion of the population being described if the survey area is adequately wide and resolution sufficiently fine.

There are two main types of transect surveys. Strip transects use a fixed transect width, within which all individuals are supposed to be detected. A strip width of 300 m is the often used. Line transect surveys use observations from a wider transect width that is subdivided into distance bands to estimate densities. The decreasing detection probability with increased distance away from the survey track line is used to fit a detection function using Distance Sampling (Buckland et al., 2001). A crucial assumption is that all birds are detected on the innermost transect line. Transect data are usually computed as birds per km², and presented as “post”, density grids or contour plots.

Counts can be made from ships or aircraft, and evaluation of the two platforms can be found in Camphuysen et al. (2004). Aerial surveys allow rapid coverage of large survey areas and access to shallow areas or complex coastlines, whereas boat surveys are more suitable for offshore areas or restricted waters. Identification and detection of cryptic species (auks, storm-petrels) is more difficult from aircraft than from a boat, although easily flushed species (such as divers and seaduck) may flee from slow moving boats before they are counted whereas fast-moving planes are able to detect them as they take flight.

The advantages of transect surveys are that they are able to sample distribution of a large proportion of a population if coverage is sufficiently wide, and that estimates of numbers within areas can be tentatively calculated. The main problem with transect surveys is that they only provide information of distribution within the area covered, and where coverage is incomplete biases in assessment of distribution and relative importance of areas will result. Furthermore, the provenance of birds is unknown, which is problematic when assessing the importance of areas for birds from particular colonies or populations. Similarly, transect surveys fail to provide information on the age class (only for some species) and the breeding status of the birds observed. Finally, as the technique is visual, no data are obtained at night and distributions of some birds may exhibit diel variation.

3.2.2 Tracking individual birds

Tracking involves fitting devices to individual seabirds that store or transmit data that can be used to determine their locations at sea at varying time intervals. A general rule applies that the device should not be heavier than 5% of the bird’s body mass (Cochran, 1980) which restricts it use on small species. A variety of devices are available for this purpose and these are described below.

Radio-tracking involves fitting birds with a radio-transmitter, and the signals from this can then be detected by an antenna. This allows the location of a bird to be determined by triangulation from fixed points (Freeman et al., 1997; McSorley et al., 2005) or by following them by boat (Ostrand et al., 1998) or plane (Adams et al., 2004; Mañosa et al., 2004). Radio-tracking is relatively cheap, and transmitters can be as small as 1g, such that tags can be fitted to even the smallest seabirds. However, detection range is often limited to an order of tens of km, depending on transmitter size and height of the receiving antenna. As such, they are only suitable for determining foraging range of relatively inshore species unless individuals are followed by an aircraft, or rafting areas of pelagic species (McSorley et al., 2005) around colonies.

Platform Terminal Transmitters (PTTs, also known as satellite tags) transmit position data regularly to orbiting satellites and hence to the observer and so can be detected at any point of the globe without the need for retrieval, allowing wide-ranging, pelagic seabirds to be tracked. Accuracy is relatively high, with an error of usually few km at most (Wilson et al., 2002). Until recently, PTTs were heavy and bulky and so could only be fitted to large birds such as
albatrosses (Jouventin and Weimerskirch, 1990) and penguins (Davis and Miller, 1992). The size of these devices has decreased substantially in recent years (9g), which allows deployment on medium-sized species such as some shearwaters. However, PTTs are still too heavy for small species such as terns and storm-petrels.

GPS loggers calculate positions from to orbiting satellites and store these. They provide the highest accuracy available, but have to be retrieved to download the data. Size constrains have limited their use to large species such as albatrosses (Weimerskirch et al., 2002) and Gannets (Camphuysen, 2005), but improvements in power of storage and reduced size could make GPS loggers very useful tracking devices in the next few years. Combined GPS-PTTs merge the high accuracy of the GPS system and do not need to be recovered to obtain the data.

Global location loggers carry an internal clock and register light intensity, from which daylight duration and sunrise and sunset hours, and hence latitude and longitude, can be calculated. This technique is relatively cheap and easy to conduct, though it is necessary to recapture the tagged birds to download the stored information. Precision is low (tens of km), and it is best recommended to employ these loggers for wide-ranging species and also for wintering behaviour rather than foraging area identification while breeding.

Compass loggers have two or three compasses along with an internal clock. From the heading of the compasses and the flight duration a paths can be reconstructed. Flight routes of species exhibiting relatively straight flights and not showing too many changes in activity (e.g. gannets) are much easier to reconstruct than those from species turning very often and changing activity very frequently (e.g. kittiwakes). Short time intervals between data logging increase the accuracy of the flight route.

The advantage of tracking is that it can provide accurate data that covers a seabird’s global range that is not bounded by arbitrary survey areas (as transect surveys may be). Remote-sensing techniques also allow the provenance of birds in different areas to be assessed and reveal information on seabird movements during the night. The disadvantages are that tags are expensive and so the number of birds and colonies at which they can be deployed is limited. As such, distribution may not be representative of the population as a whole owing to variation in ranging behaviour according to colony, age, sex, breeding status, individual and season.

Transect methods and tracking can be complementary with both methods providing independent data on distribution, tracking revealing provenance of birds and transect counts providing estimates of numbers (Camphuysen et al., 2004).

3.3 Determining boundaries of areas

Boundary recognition is a fundamental process relating to IBAs and SPAs in order to determine the spatial limits of the site. There are several means of achieving this that depend on whether habitat or bird distributions are being used as the basis for site recognition, and the types of bird data available for analysis.

3.3.1 Use of habitat features

Marine systems may outwardly appear homogenous, but do contain various habitat features that birds associate with at elevated densities (e.g. Hunt and Schneider, 1987). Where birds associate strongly with a habitat feature, its limits may be used as an SPA or IBA boundary in the same way as in terrestrial, freshwater or intertidal systems.

Habitat features that may be important for boundary determinations are oceanographic features such as bathymetry, temperature and salinity.
Marine habitat features have been mainly used on the designation of MPAs worldwide (Hyrenbach et al., 2000), and also for SACs within Europe. These have potential to complement seabird conservation where important numbers occur within their boundaries. For example, the SAC Dutch Coastal Sea designated to protect shallow sandbanks encompasses important concentrations of seaduck (Lindeboom et al., 2005; Section 3.5.3), and that for the Friesian front post-breeding concentrations of guillemots (Leopold et al., in press; Section 3.5.3). It has also been recently proposed to use hydrographical clues to identify MPAs protecting the breeding foraging grounds of the Balearic Shearwater (Louzao et al., in press; Section 3.5.6).

MPAs centred on habitat features are often surrounded by buffer zones, which are intended to allow for dispersal of animals associated with it or, in the case of hydrographic features, uncertainty in the location of the feature itself (Hyrenbach et al., 2000). SPAs and IBAs do not include provision for buffer areas around sites.

### 3.3.2 Delimiting marine extensions to colony SPAs and assessment of foraging areas

Colonial nesting seabirds are central place foragers, and many species are highly aggregated in discrete colonies. As such, the distribution of breeding seabirds tends to be more clumped and spatially stable during the breeding season than at other times of year. Furthermore, many of the most important seabird colonies are already recognised as IBAs and designated as SPAs, but these generally only extend to the high water mark. Marine extensions to such colonies may therefore have merit for protecting bathing, resting or foraging areas upon which birds breeding at the colony depend for survival and successful reproduction, and various approaches exist to delimit the boundaries of these.

Bathing and resting birds often congregate around cliff colonies at high densities that can be incorporated into the colony SPA using generic species-dependent extensions. Ship-based surveys of seabird distribution revealed densities of auks declined markedly at 1km from the colony and those of northern fulmar and northern gannet at 2km from the colony, and these limits were used to define boundaries for marine extensions to colony SPAs (McSorley et al., 2004). The seaward boundary of Manx shearwater rafts around three UK colony SPAs was determined from fixes of radio-tagged birds (McSorley et al., 2005; Box 3–1). However, most species feed beyond the boundaries of resting areas and such extensions will not recognise or protect important foraging areas.

The boundaries of foraging areas for breeding seabirds can be delimited most simply using foraging radii (Birdlife International, 2000). These can be determined from empirical observations of foraging ranges using tracking devices or transect surveys from ships as described in Section 3.2. Alternatively, the distance travelled can be calculated from trip duration and flight speed (Pearson, 1968) or from provisioning energetics (Flint, 1991), although these tend to overestimate range (Birdlife International, 2000). Foraging ranges vary enormously among species (Birdlife International, 2000), and so need to be applied generically to each important colony at which a species occurs (Table 3–1). However, foraging ranges can vary among sites (Hamer et al., 2000) and years (Monaghan et al., 1994) and this may result in generic boundaries being inappropriate. The main problem with this approach is that seabirds generally use a small proportion of their potential foraging range, and so large areas that are seldom used by birds will be included in the IBA or SPA when radii approaches are employed (Birdlife International, 2000). This problem is most acute for species with large foraging ranges such as petrels and gannets, for which radius based methods are wholly inappropriate (Birdlife International, 2000). However, for those with short foraging ranges and low foraging habitat specificity such as terns, this approach is worthy of consideration (Birdlife International, 2000).
The following hierarchy of approaches is suggested to maximise the application of available data in order to apply appropriate radii:

1) Species x Site-specific data (either gathered from literature, or through current field based projects, bearing in mind potential density-dependence due to differences in colony size and other ecological considerations that may determine the size of the radii. In cases where multispecies colonies exist, the species with the largest foraging radius should be used to set the outer radius).

2) Species-specific data.

3) If data are not available to apply 1 or 2, use nearest neighbour or surrogate species.

Table 3–1: Recommended limits for foraging radii of seabird species breeding in the ICES area as given by SPEA-SEO/BirdLife (2005).

<table>
<thead>
<tr>
<th>Radius around colony (to include foraging and/or maintenance activities)</th>
<th>5 km</th>
<th>15 km</th>
<th>40 km</th>
<th>still unknown</th>
</tr>
</thead>
<tbody>
<tr>
<td>Stercorarius parasiticus</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Larus genei</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sterna albifrons</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sterna nilotica</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cepphus grylle</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Calonectris diomedea (rafts)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Puffinus puffinus (rafts)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Puffinus mauretanicus (rafts)</td>
<td></td>
<td></td>
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Box 3–1: Determination of possible SPAs for Manx shearwaters *Puffinus puffinus* using radio telemetry.

Radio transmitters were attached to breeding Manx shearwaters at three existing terrestrial breeding colony SPAs in the UK – Bardsey, west Wales, Skomer, south-west Wales, and Rum, west Scotland. These birds form dense flocks (“rafts”) on the sea before dusk before entering the breeding colony.

In order to identify likely boundaries for an extension of the existing SPAs into the marine environment, the results from radio-tracking birds in rafts were analysed using kernel analysis. This method aims to define the home range or area of greatest use of animals (Powell 2000).

![Map of shearwater locations and kernel cores](image)

Recommendations for seaward extensions to the SPA boundaries at these three colonies have been made based on the areas within which the birds appear to spend 95% of their time (McSorley *et al*., 2005). Although an arbitrary proportion 95% seemed a sensible one for three reasons:

a) there was little difference between the 90% and 95% kernels at all three colonies;

b) 95% is a useful analogy with statistical significance (though it should not be confused with that);

c) it accords with other studies.

### 3.3.3 Delimiting boundaries of bird concentrations from transect counts

Transect data are usually interpolated using kriging (Cressie, 1991), a statistical method that produces estimates of density in a fine-scale grid based on counts in adjacent squares and patterns of spatial autocorrelation. This generates a map of bird density over the entire study area. Boundaries around concentrations identified by transect surveys are generally determined by generating density contours (isolines) in GIS. In some studies (Skov *et al*., 1995, 2000), arbitrary contour intervals were chosen, which means that boundaries are to a certain extent determined by these predetermined limits as well as density. A more
quantitative approach is to estimate gradient of bird density change over space. This approach identifies the strongest gradient in spatial density and positions the isoline just outside this, which is then treated as the border of the concentration. In this way, the major part of the concentration is included in the selected area (Garthe, 2006; Garthe and Skov, under review). In other studies, an important concentration of birds may already be recognised and the challenge is to identify the boundary of this. In these cases, a contour that encloses a given percentage of the aggregation needs to be generated, as for Common Scoters in Camarthen Bay (Webb et al. 2004a; Box 3–2).

Box 3–2: Determining the boundary of Camarthen Bay SPA

Carmarthen Bay, south Wales, hosts internationally important concentrations of Common scoter *Melanitta nigra* outside the breeding season.

Aerial surveys, deploying a standardised methodology (Kahlert et al., 2000) were undertaken over the area in winter 2001/02. Sampled densities of scoter were converted into total population size using distance methodology (Buckland et al., 2001).

Scoter density was modelled throughout the survey area using kriging, a spatial interpolation method based on variography (Cressie, 1991).
Recommendations for the seaward boundary of the SPA were made, such that 95% of the modelled population was contained within the boundary. See Webb et al. (2004a) for full details of this case study. See also McSorley et al. (2004) and Webb et al. (2004b,c) for further details of the methods applied in identifying sites and boundaries for inshore concentrations of waterbirds outside the breeding season, including rules for the inclusion of satellite aggregations disjunct from the core aggregation.

### 3.3.4 Determining boundaries of bird concentrations from tracking data

For any bird or number of birds that have been instrumented, kernel density estimation provides a means of quantifying habitat use (Georges, 1997; Wood et al., 2000). To calculate kernels, data are first standardized by resampling the tracks followed by each bird at hourly intervals. This process gives the same density of locations per unit time. The resampled tracks are then summed to provide estimates of core areas used by the bird or birds. A smoothing parameter (“h”) has to be chosen that represents the spatial scale over which observations are averaged. This smoothing parameter represents the distance between each location and the nearest grid intersection on the surface to which the birds’ core range is to be plotted. The value of h chosen can significantly affect the results (Hemson et al., 2005), especially for animals that often return to a home point such as a nest or roost. This statistical problem can be minimized if a relatively large number of birds are tracked (BirdLife International, 2004). Concentric polygons are then drawn around areas encompassing some percentage of the total area visited by all birds (BirdLife International, 2004).

Kernel density estimates can be weighted by extrapolating to the total population within which individuals were selected for tracking. The weighted density estimates can then be plotted.

For some species such as petrels, which forage very far from their nesting locations, kernel density estimation could give misleading results, in that the area close to the breeding colony will emerge as important even if no feeding is done in its vicinity. Because of this, it may be advisable to remove commuting data points in order to highlight important foraging habitats. For migrating birds tracked by GLS, kernel estimators are not appropriate. Instead, one calculates mean latitude ± 1 SD within each 10 degree band of longitude and plots the results (BirdLife International, 2004).
3.4 Criteria for recognising SPAs and IBAs

3.4.1 Criteria within habitat features

Where seabirds occur within a habitat feature, their importance can be judged using standard IBA or SPA criteria (Section 3.1.2). This demands that the concentrations within the borders of the habitat are sufficiently important (>1% of the population of interest), that the density there is significantly higher than in the surrounding sea and that the extent of the site is not excessively large. In the latter case further refinements to criteria such as the MCC may be required (see Section 3.4.3).

3.4.2 Criteria for marine extensions to colony sites

In order to be eligible for marine extensions, the colony itself needs to be designated as an SPA or listed as an IBA. The colony extension then needs to be made in the context of the colony rather than in the context of the biogeographic population. This could be an area within which a desired percent of the activity of interest occurs: e.g. a radius or suite of patches that encompass 95% of the foraging activity from a colony SPA or IBA (McSorley et al., 2005). Assessing the importance of individual foraging patches at the biogeographic level is inappropriate because birds have already qualified at the colony site. Furthermore, since birds do not forage simultaneously and may be divided among several habitat patches, the likelihood of individual foraging patches qualifying at the biogeographic level will be excessively small.

3.4.3 Marine Classification Criterion

The 1% criterion is used as a qualifying threshold for both SPA and IBAs, and has gained wide acceptance despite its lack of strict biological basis. The maximum size of an area considered to be internationally important is, however, not defined. A larger site is by definition likely to contain more birds than a smaller site centred on the same location. The aim of SPAs and IBAs is to protect discrete concentrations of birds rather than extensive areas in which densities are low, so adjustments to the 1% criteria based on the area the concentration occupies may be necessary to avoid the latter occurring.

In order to compensate for area size, Skov et al. (2000) and Skov et al. (under review) developed the Marine Classification Criterion (MCC). The MCC combines the proportion of the total population and the degree of concentration. The proportion of the total population is defined as the estimated number of birds within an aggregation, divided by the total population, and then multiplied by 100. The MCC is met if not only 1% of the biogeographic population of a particular species are concentrated in an area (site), but if in addition, the average density at that site is at least 4 times the background density (in the investigated regional sea).

The MCC allows applying the 1% criterion for the identification of concentrations of seabirds of international importance (see also Skov et al., 2000) while minimising the risk of selecting unduly large areas of sea. Formerly, Skov et al. (2000) used an area of 3,000 km² as the reference to control for area size instead of the current density comparison. In spite of the fact that the different Marine Classification Criteria (MCC) uses different areas adjustment factors, the differences in the resulting selection of areas were minor. Most areas selected using the early version of the MCC were retained during the most recent analysis, and the boundaries were only modified slightly as an effect of adding or deleting the boundary of a few concentrations of moderate importance (Skov et al., under review).
3.4.4 Criteria applied to tracking data

The BirdLife International (2004) symposium volume stopped short of making specific recommendations about how to translate kernel density estimates to implementation of MPAs. However, they did state that “IBAs (for albatrosses) likely comprise three types of congregation: breeders around islands, breeders in oceanic areas and non-breeders”.

It will likely be necessary to combine tracking data with ship-based transect surveys in order to decide upon appropriate sites for seabird SPAs or IBAs. For example, a recommendation of BirdLife’s European Partnership is that a site known or thought to hold on a regular basis “>20,000 waterbirds or > 10,000 pairs of seabirds” would qualify as an IBA. Clearly, transect data are required to establish whether this criterion is met. Nevertheless, in the absence of transect data, some extrapolation from tracking data will need to be adopted to help identify such areas, especially for species with low and endangered populations. This could be based on areas that encompass a predetermined percentage of fixes, as adopted for rafting Manx shearwaters around UK colony SPAs (McSorley et al., 2005; Box 3–1).

3.5 National approaches

3.5.1 Denmark

In Denmark, an initial 111 areas were designated as SPAs in 1983, with preliminary boundary definitions. In 1994, precise delineation of these sites was finalised, the 111 SPA sites covering an area of 976,000 Ha. Of these, 47 SPAs were intertidal/near-shore areas or over shallow marine water. These constitute around 60% of the entire area covered by SPAs.

The Danish Forest and Nature Agency (Ministry of the Environment) is the administrative body responsible for classifying Danish SPAs; details can be found at http://www.skovognatur.dk/English/.

Site selection of coastal SPAs was made using available data. In terrestrial and intertidal areas, data were provided by the Danish Ornithological Society whereas those for marine areas were provided by a long-term monitoring programme on waterbirds in inner Danish waters.

For marine areas, designation was made primarily with reference to Article 4.2 of the EU Birds Directive; i.e. migratory species that occur in international important concentrations.

In 2002, the process of adding more marine areas to the SPA suite of sites was initiated, in response to the EU Commission. This led to the classification of another two SPAs in 2003. Of these, one was designated primarily for species foraging on benthos, while the other was entirely offshore and designated for red-throated diver and little gull. This added another 530,000 Ha to the total area of marine SPAs. In addition, two existing SPAs were enlarged in 2002, the aim being to improve inclusion of the interest species, common scoter and common eider.

A national monitoring programme has been established with the aim of assessing whether those species for which SPAs have been classified are of favourable status. As part of this programme, waterbirds are monitored every three winters (January/February) in all inner Danish waters, providing estimates of total numbers and distribution. Similarly, moulting diving ducks are monitored in selected areas every sixth summer (August).

3.5.2 Germany

In Germany, responsibility for designating marine protected areas is split between different administrations. Generally, nature conservation is the responsibility of the Federal States. As national territorial limits extend out to only 12 nm, responsibility for the German parts of the North Sea is now divided among the Federal State of Lower Saxony (within the 12 nm zone,
southern part), the Federal State of Schleswig-Holstein (within the 12 nm zone, northern part),
and the Central Government (for the EEZ). In the German Baltic, there are responsibilities
allocated to the Federal State of Schleswig-Holstein (within the 12 nm zone, western part), to
the Federal State of Mecklenburg-Vorpommern (within the 12 nm zone, eastern part), and the
Central Government (for the EEZ). This complicates the procedure of selecting suitable areas
because the process is not well coordinated between the different governmental units, and also
because these regions are not easily manageable (e.g. the EEZ in the Baltic is merely a small
strip in some German areas).

In order to meet legislative requirements, the Federal States began designating SPAs under the
Birds Directive a few years ago, the first area being announced in October 1997. Progress with
SPA classification developed differently in the German EEZ, which was declared in
November 1994. As this area is not German territory, designation depended for a long time on
resolution of international rules/laws/conventions because it was unclear whether SPAs could
be designated outside national territorial limits. Furthermore, it was not until April 2002 that it
was possible to designate SPAs within the EEZ because of national legislation. At that time,
the national law for Nature Conservation was changed and responsibilities for the designation
of marine protected areas were clarified. The Federal Agency for Nature Conservation is
responsible for selecting potential protected areas, whereas the Federal Environmental
Ministry is responsible for designation/submission to the European Commission.

Under politically induced time-pressure, proposals for SPAs in the EEZ were not addressed
until 2002 (see Box 3–3; Garthe, 2003, 2006; and Garthe and Skov, under review). Two areas
were proposed and designated in May 2004 with slightly altered geographical borders. The
Federal State of Schleswig-Holstein finally designated all SPAs in 2005; the Federal States of
Niedersachsen and Mecklenburg-Vorpommern are still in the process of designating the
remaining sites.

Box 3–3: Identifying marine SPAs in the German EEZ of the Baltic Sea.

Using standardised transect survey methods (Webb and Durinck, 1992) data on the
distribution of all birds at sea were collected between 1987 and 2002; more than 15,000 ship
km were travelled. Distance sampling analyses (Buckland et al., 2001) were applied to the
data for all EU Birds Directive Annex I (n=7) and migratory (n=13) species in order to
estimate total numbers of birds present. Ordinary kriging (Cressie, 1991), an interpolation
technique that uses the spatial autocorrelation in the raw dispersion data, enabled the
modelling of continuous dispersion throughout the survey area. The modelled distributional
data were projected onto a two-dimensional map for each species. For example, long-tailed
duck *Clangula hyemalis*:
Individual species maps were combined to depict areas of overall importance. Boundaries between high concentration areas were determined by analysing the gradient of modelled bird density change over space, thereby allowing the identification of potential SPAs.

Based on this procedure, a single large SPA of c. 2,000 km² in the German EEZ of the Baltic Sea has been classified (red shading). The dashed line indicates the limit of German territorial waters, the continuous line the limit of the German EEZ. This EEZ SPA is defined by overlapping concentrations of several species, primarily by the distribution and abundance of Slavonian grebe, long-tailed duck, common scoter, velvet scoter and black guillemot, and additionally red-throated diver, black-throated diver and red-necked grebe. This SPA complements those identified in inshore waters of the German Baltic Sea (blue shading).

For full details of this work see Garthe and Skov (under review).
3.5.3 The Netherlands

The Netherlands has designated two special protection zones under both the EU Birds Directive and the EU Habitats Directive. Four more “Areas with specific ecological values” are currently under review and might soon achieve similar status; another three are also being reviewed but will require further study (Lindeboom et al., 2005). The two designated areas (“Voordelta” and “Wadden Coast”) are special protection zones under both the EU Birds Directive and the EU Habitats Directive. These areas are in fact the southern and northern parts of a continuous strip of shallow (0–20 m deep) water, running along the entire length of the country. It is recognized that the mid-section of this area also potentially holds important bird numbers, but rather than designating all coastal waters, the Netherlands has opted for protection of the richest parts. Key bird species are red-throated diver, great crested grebe, common scoter, common eider, and all Larus and Sterna species breeding along the Dutch coast, migrating along the coast, or wintering in the nearshore waters (gulls only). The Netherlands has decided, in the “National Spatial Strategy” that more SPAs should be identified, particularly in the offshore parts of the EEZ, taking into consideration requirements both of the Birds and the Habitats Directives and the OSPAR Convention. Seabirds are therefore part of the equation, but other ecological values also play a significant role, both in identifying these areas and in defining their boundaries. Three such areas have been determined: “Dogger Bank”, “Central Oyster Grounds”, “Cleaver Bank” and “Frisian Front” (see Figure 3–1). The first two of these have no specific bird interest, but the Cleaver Bank was found to have a higher than average diversity of seabirds, and the Frisian Front was selected because of regularly occurring vulnerable concentrations of common guillemots with chicks in summer (Leopold et al., in press). The boundaries of these areas however, are largely determined by physical and benthic features, although in the case of the Frisian Front it was confirmed that the core area for the guillemots was well within these boundaries. Areas that will be considered in the future include an area with natural gas seeps (with little if any relevance for seabirds), a nearshore “reef” area at the Dutch/German border (mainly of interest because of the benthos, but possibly also important for divers), an area in the central Southern Bight of the North Sea, where large numbers of auks occur in late winter, and an area in the south-west with extensive shallow banks (which also might have high bird interest). In summary, while at least three areas have been specifically (co-)selected for seabirds, most proposed or studied areas have been or will be selected for more than one purpose, fulfilling Birds and Habitats Directives requirements, as well as OSPAR criteria.
3.5.4 Belgium

In 2001, a Royal Decree was issued in Belgium legally protecting all birds in Belgian marine waters. Belgium has not yet designated any SPAs in marine waters, but Haelters et al. (2004) has outlined the process for marine SPA classification in Belgium. This follows a formal assessment process. Those species of waterbird on Annex I of the EU Birds Directive that occur regularly in qualifying numbers in Belgian waters will have SPAs considered for their protection. However, only those regularly occurring migratory species not on Annex I that are already protected by another international convention(s) will potentially be accorded protection within SPAs. Hence, unless such species are already protected by some other instrument no SPAs will be classified for them. Haelters et al. (2004) propose three SPAs targeting the protection of seven species (three Annex I species and four migratory species). These areas were assessed firstly on the concentration of birds they host, resulting in the identification of the most suitable habitats for great skua and little tern. For the other five species (Sandwich tern, common tern, great crested grebe, common scoter, and little gull) the most suitable habitats in number and size were selected for each species and the areas overlaid to assess the final area for possible SPA status. From this analysis, three areas were identified, two of which extend from low water mark out to 6 nm offshore (off Koksidje and De Panne and off the coast from Middeelkerke to Bredene); the other area is focused around the harbour of Zeebrugge).

3.5.5 United Kingdom

Considerable progress has been made in the UK over the past few years in identifying marine SPAs. The general approach has been one of identifying important sites in three broad categories – a) marine extensions to existing breeding seabird colony SPAs; b) sites for inshore concentrations of waterbirds in the non-breeding seasons; and 3) sites for offshore aggregations of seabirds. To date, recommendations have been made and endorsed to extend existing SPAs for common guillemot, razorbill, and Atlantic puffin by 1 km into the marine environment, by 2 km for northern fulmar and northern gannet breeding SPAs, and by at least...
4 km for Manx shearwater SPAs (see Box 3–3). Recommendations have also been made not to extend, at least until further planned work has been completed, those SPAs for which storm-petrels, European shag, and terns are interest features; and recommendations have been made not to extend existing breeding site SPAs for which great cormorant, skuas, gulls, black-throated diver, great crested grebe, Slavonian grebe, common scoter, or red-necked phalarope are interest features.

Possible SPAs have been identified in two inshore sites, one for red-throated diver and one for both common scoter and red-throated diver, and one wholly marine SPA has been classified for common scoter (see Box 3–1). An aerial survey programme covering all the important inshore areas around the UK continues. Plans have been drafted to explore the possibility of identifying important concentrations of Balearic shearwater in the Channel, and of identifying possible SPAs for offshore aggregations of seabirds.

### 3.5.6 Spain

In Spain, the recognition of SPAs in the marine environment has yet to be addressed by the Birds Directive, although SACs have been designated under the Habitats Directive. As a first step in the process of establishing SACs, efforts have been focused on designation of Sites of Community Interest (SCI) in the Canary Islands, while only provisional SCIs have been identified in the Iberian area (Ambrosio et al., 2002). The main interest has been focused on sandbanks and sea-grass prairies, apart from inshore/coastal areas.

Spain comprises three biogeographical marine regions: Mediterranean, Atlantic and Macaronesia. These regions host a wide diversity of seabird species, many of them of conservation concern, and which face various human pressures (Madroño et al., 2004). Among these pressures, more or less effective protection exists on the land, particularly at colonies, but little effort has been addressed in the open sea, where pressures are not well understood but seem to pose a serious threat for some populations (Madroño et al., 2004).

The Mediterranean region holds significant populations of species of high conservation concern, which are subject to strong pressure by human activities such as fisheries (reviewed in Arcos et al. in press). Seabirds of interest include the whole breeding population of the critically endangered Balearic shearwater and about 90% of the breeding population of the threatened Audouin’s gull. The Macaronesian region (Canary Islands) holds significant populations of highly pelagic species, such as Bulwer’s petrel, little shearwater and Cory’s shearwater. The Atlantic region, although hosting only a few relevant breeding species, is a highly productive area of importance for migrating and wintering seabirds.

As a first step in the designation of marine SPAs, the Spanish Ornithological Society (SEO/BirdLife) is conducting a LIFE-Nature project to develop criteria for and identify IBAs at sea. The ultimate target of this project is to obtain a complete and detailed inventory, applying objective methodological criteria, of Important Bird Areas at sea for the seabird species listed in Annex I of the Bird Directive with populations in Spain. Species of conservation concern that do not breed in Spain will also be considered, but are not explicitly targeted by this project. It is expected to develop, along with a Portuguese sister project run by SPEA, a standard methodology for the identification and delimitation of IBAs at sea, which could be applicable to other countries.

The existing data on seabird distribution at sea for Spain will be considered, but most data are expected to come directly from transect surveys conducted within the framework of the project, as well as the tracking of breeding seabirds, using both radio-telemetry (Bulwer’s petrel, little shearwater, Cory’s shearwater, Madeiran storm-petrel and satellite telemetry (Cory’s shearwater, Mediterranean shag and Audouin’s gull). Environmental variables will be used to model seabird distribution patterns and the use of these to define IBAs boundaries, will be considered (see Louzao et al., in press; Box 3–4).
Box 3–4: Understanding the oceanic habitat of the critically endangered Balearic shearwater *Puffinus mauretanicus* to identify suitable areas for protection.

Data on Balearic Shearwaters distribution at sea were collected throughout the Iberian Mediterranean coast, using vessel-based transect survey methods, during the chick-rearing period of 1999–2000 and 2002 (Louzao et al., in press). The overall foraging range was first identified using presence/absence data, and corresponded to the frontal systems along the eastern Iberian continental shelf (<200 m depth). Main foraging grounds within that area, identified by high density aggregations, were characterised by elevated Chlorophyll *a* concentration.

It was proposed to establish a core-buffer protection model, in which the main foraging grounds (i.e., the core region – area of influence of the Ebro River discharge and Cape La Nao region) deserved elevated protection, while in the remaining foraging range (buffer region) more diffuse protection would be applied.

A predictive model based mainly on Chl *a* gradients properly identified the main foraging range for Balearic Shearwaters for each of the 3 years of study. The use of habitat variables to assess suitable areas among years and their overlap could help defining IBAs/SPAs boundaries.
3.5.7 Portugal

The oceanic area under Portuguese jurisdiction is eighteen times the size of the land area. The Portugal EEZ totals 1,727,408 km², making it the largest in Europe and the 11th largest in the world.

In Portugal, the extension of the Habitats and Birds Directives to the marine ecosystem is still at an early stage. The process has been initiated in Azores Archipelago with regards to habitats. The Azores archipelago has already listed offshore locations such as the SACs “Formigas Bank and Dollabarat” and “Dom João de Castro Bank”. In the mainland, it was so far classified as SPA a seaward extension of the Berlengas seabird colonies (SPA Berlengas and Farilhoses) and also an inshore area (up to 20m depth) for wintering scoters (SPA Aveiro). A SAC was proposed to protect the last sea-grass prairie (Zostera sp) in the marine environment in Portugal (SAC Arrabida/Espichel). In Madeira Archipelago, SPA designated for the main seabird breeding colonies of Desertas and Selvagens include a seaward extension of 100m to 200m depth. This is clearly insufficient to protect most breeding and wintering seabird species occurring in Portugal.

The scales at which seabird dispersion occurs in the Portuguese marine area render it quite difficult to define and characterise seabird densities. The possibility of identifying inshore-coastal aggregations of some wintering species, such as Balearic shearwater and scoters, may lead to the classification of a small number of inshore SPAs off continental Portugal, though this is data-dependent.

Terrestrial seabird colonies that have a coastal component and are already classified might be extended into the sea to protect feeding, resting and/or rafting aggregations of birds. This may include important coastal feeding areas in the breeding season for species such as terns, and species-specific seasonal concentrations, such as “rafting” Manx and Cory’s shearwaters during the breeding season. However, such seaward extensions will not integrate foraging/staging grounds of most threatened seabird species or those Annex I species for which Portugal has highest responsibilities. In this sense, offshore areas hosting concentrations of seabirds are probably Portugal’s main challenge for marine SPA designation. Data are very scarce and truly pelagic species such as Pterodroma madeira,
Pterodroma feae, Pelagodroma marina or Oceanodroma castro require large amounts of data and intense surveys to enable sensible proposals for future SPA classification. Their behaviour at sea is poorly known and the methodologies used to track them are still under development.

Between 2004 and 2008, the Sociedade Portuguesa para o Estudo das Aves (SPEA) is conducting a strategic LIFE-Nature Project to contribute to the implementation of the Birds Directive in the marine environment through a detailed inventory of the most significant marine areas for seabirds included in Annex I of the Birds Directive. This will aim also to formulate adequate methodological criteria for the identification and delimitation of IBAs off Portugal. Some of Europe’s rarest bird species will be addressed, such as the globally threatened Puffinus mauretanicus, Pterodroma madeira and Pterodroma feae, and many seabird species that are of conservation concern, e.g. Calonectris diomedea, Puffinus assimilis, Larus audouinii and Sterna sandvicensis.

In order to achieve this, the relationship between oceanographic variables such as temperature, productivity, currents etc. and the occurrence of seabirds will be studied. Monitoring of certain species will be carried out through data-logger and satellite tracking (Calonectris diomedea) and radio tracking (Bulweria bulwerii, Puffinus assimilis, Oceanodroma castro), survey of coastal waters, analysis and mapping of ringing recoveries in Portugal, and the creation of a database of beached birds. The identification and generic sampling of the most favourable areas is being carried out based on the oceanographic characteristics of the areas concerned.

### 3.5.8 USA

The official U.S. definition of marine protected area is: “Any area of the marine environment that has been reserved by federal, state, territorial, tribal or local laws or regulations to provide lasting protection to part or all of the natural or cultural resources therein” (Executive Order 13158). The USA initiated a program for the designation of Marine Protected Areas in 2003, and by 2006 there have been between 1500 and 2000 sites designated. Some of these are administered by five federal programs: the National Park Service, the US Fish & Wildlife Service, and three branches of NOAA – National Estuarine Research Reservation program, the National Marine Fisheries Service and the National Marine Sanctuary Program. Other sites are under the jurisdiction of 22 states and territories including Guam, the U.S. Virgin Islands, American Samoa, and the Mariana Islands.

Many of the sites had already been protected under existing regulations, so that listing them as Marine Protected Areas provided no additional protection. For example, the Monomoy National Wildlife Refuge on Cape Cod, Massachusetts, has recently been designated an MPA, but it already had the most restrictive regulations possible in effect due to its status as a National Wildlife Refuge.

Restrictions placed on each site vary according to what species are present and what sort of protection they need. Examples of reserves developed specifically for seabirds include Monomoy Refuge, Massachusetts “Momomoy provides habitat for hundreds of species of resting, feeding and migratory birds. The refuge is so important to migratory shorebirds, in 1999 the Monomoy Islands were designated a Western Hemisphere Shorebird Reserve Network (WHSRN) regional site. The refuge supports the largest nesting colony of common terns in the Gulf of Maine and the second largest on the Atlantic Seaboard with close top 8000 nesting pears in 2001.” And the Cordell Bank Marine Sanctuary, California “Cordell Bank provides important habitat for many species of groundfish, and is a feeding area for seabirds and marine mammals.”

### 3.5.9 Canada

1. Protected areas
Protected marine areas can be created in Canada under legislation administered by three federal agencies: Department of Fisheries and Oceans (DFO), Parks Canada and Environment Canada. DFO has been given the lead in Canada to develop a network of marine protected areas in the Pacific, Arctic and Atlantic Oceans.

DFO legislation specifically mentions the conservation and protection of marine species within Marine Protected Areas (MPAs), but here, the term “marine species” refers only to those administered by the department, i.e., fish, seals, cetaceans, turtles, and invertebrates. Conservation and protection of marine birds can be used only as supporting information in DFO’s MPAs, not as the sole reason for their establishment. Currently there are four MPAs in Atlantic Canada, one officially designated, and three announced (2005). None commands particular interest in relation to seabird conservation.

Parks Canada has a plan to establish one or more National Marine Conservation Areas (NMCA) in each ocean region of Canada. There are 13 designated marine regions broadly considered to be Atlantic (10 proper and 3 in high Arctic). To date there has been only one NMCA established in Canada; the Saguenay St. Lawrence Marine Park is located at the confluence of the Saguenay River and the St. Lawrence Estuary. This is an area of high productivity important for cetaceans, seabirds and other marine organisms.

The main mechanism to conserve and protect marine areas important to seabirds in Canada is under legislation administered by Environment Canada. These allow for the establishment of Migratory Bird Sanctuaries (MBSs), National Wildlife Areas (NWAs), which can be comprised of terrestrial and/or marine components. In addition, recent amendments allow for the establishment of Marine Wildlife Areas in the Exclusive Economic Zone (EEZ) outside of territorial waters. Even though the first MWA is still in the planning stages, Environment Canada’s existing system of MBSs and NWAs encompass more than 3 million ha of marine habitats making them currently the premier tool for protecting marine habitats in Canada, not just for seabirds, but for all marine species.

The Canadian provinces also have jurisdiction over nearshore waters and there are several examples of protected areas that include a marine component. For example, Newfoundland and Labrador protect five of the most important seabird breeding colonies in that province, under their Wilderness and Ecological Reserves Act. The Act allows for the protection of waters adjacent to the colonies out to a maximum of three nautical miles.

2. Identified important marine areas with no formal protection (Atlantic)

IBAs: Of the approximately 238 IBAs identified to date in provinces and territories bordering the Atlantic (from Nunavut in the north to the Maritimes in the south), 206 (87%) contain marine/open sea component. Almost all are coastal or nearshore and only four contain no land component. One of these is the Lancaster Sound Polynya, which is the only real offshore IBA in Atlantic Canada. The most important seabird colonies are contained within these IBAs and many also include a marine component in the vicinity of the colony.

Others: DFO has recently proposed an initiative to identify Ecologically and Biologically Significant Areas (EBSAs) in the marine environment. Consultation has been broad and has included consideration of areas important to marine birds. EBSAs can be located anywhere on the continental shelf from the coast out to the 200 mile EEZ.

3.6 References


Garthe S. and Skov H. under review. Selection of suitable sites for marine protected areas for seabirds: a case study with Special Protection Areas (SPAs) under the EU Birds Directive in the Baltic Sea.


RSPB. 2005. RSPB SPA Project 2005. [Comparison of IBAs and SPAs in the UK]


waterbirds using Liverpool Bay during the non-breeding season in support of possible SPA identification. Unpublished JNCC Report, JNCC, Peterborough.


4 Review of the extent and possible reasons for seabird breeding failures in the North Sea and NW Scotland in 2004 and 2005

4.1 Introduction

Total or partial breeding failures occurred in many UK seabird colonies along North Sea coasts in 2004 (Mavor et al., 2005), and this was reported widely in the popular media and elsewhere (e.g. Proffitt, 2004). While local breeding failures had been observed before, for example, in Shetland in the late 1980s (Hamer et al., 1993), reproductive success was depressed over a larger area and concerned more species in 2004 than previously seen in the UK. Most breeding seabird species in this region depend on lesser sandeels Ammodytes marinus for raising their chicks, and these species were particularly badly affected. Industrial sandeel fisheries in the North Sea have seen reduced landings since 2003, and in 2005 the fishery was closed in the entire North Sea (ICES, 2004; Anonymous, 2005). In the spring of 2005, it became clear that most seabirds were breeding very late, and concerns were raised about this leading to another bad season. On this basis, this ToR was included in the working programme for WGSE in 2006. However, the final outcome of the 2005 breeding season was different than expected. Along UK North Sea coasts, success improved for nearly all species compared to 2004, while colonies in NW Scotland experienced unusually low success for several of the same species (Harris et al., in press; Mavor et al., in press). There was thus considerable interest in determining the spatial scale of the problems experienced by breeding seabirds in these two years. It was therefore decided to extend this ToR to cover seabird reproductive success in the whole North Sea, as well as in NW Scotland, in 2004 and 2005. A summary is provided of the success or failure of sandeel-dependent seabird species, separately for the northern UK and other North Sea countries. Although it is still quite early to draw conclusions about which environmental factors caused the observed breeding failures, an attempt is made here to collate relevant information on recruitment, availability and nutritional quality of sandeels and other seabird prey in these years, and to evaluate whether these factors are likely to be causally related to the breeding failures.

4.2 Summary of the 2004 and 2005 seabird breeding seasons in the northern UK

Widespread breeding problems (either total failures or unusually low success) occurred along UK North Sea coasts from Shetland to NE England in 2004 for most sandeel-dependent seabirds (Proffitt, 2004; Mavor et al., 2005). These failures affected both surface-feeders and diving seabirds, and were thus spatially and ecologically considerably more wide-ranging than previously recorded events, including the ‘Shetland crisis’ in the late 1980s. In 2005, breeding was generally late in these same areas, and although success was still below average for many species, it was substantially higher than in 2004. Figure 4–1 illustrates this for seabirds breeding on the Isle of May, SE Scotland. However, many of the same species had very low reproductive success in NW Scotland in 2005. Table 4–1 summarises these patterns for the main sandeel-feeders based on results from the UK Seabird Monitoring Programme (Mavor et al., 2005, in press). The patterns were very consistent across species, the only notable exception being that northern gannets had normal breeding years in both 2004 and 2005. However, northern gannets have a large foraging range and a very broad foraging niche, and although sandeels form an important part of their diet in some areas, they can easily switch to other prey.
Figure 4-1: An index of reproductive performance of five sandeel-dependent seabird species on the Isle of May, SE Scotland. The index is the first principal component of seven original variables, explaining 57% of the total annual variation. Variables included are reproductive success of European shag, black-legged kittiwake, common guillemot, razorbill and Atlantic puffin, and mean fledging mass of common guillemot and Atlantic puffin chicks. From Frederiksen et al. (in prep).

Table 4-1: Reproductive success of sandeel-dependent seabirds in North Sea in 2004 and 2005, and in NW Scotland in 2005. General trends were very difficult to detect for Sandwich, roseate and common terns due to large local effects of predation and disturbance; these species are therefore not included in this table. Based on Mavor et al. (2005, in press).

<table>
<thead>
<tr>
<th>Species</th>
<th>2004</th>
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<tr>
<td></td>
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<td>NW Scotland</td>
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<tr>
<td>Northern gannet</td>
<td>Normal success</td>
<td>No data</td>
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<tr>
<td>European shag</td>
<td>Low success</td>
<td>High success</td>
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<td>Arctic skua</td>
<td>Very low success,</td>
<td>High success</td>
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<td></td>
<td>total failure in Shetland</td>
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<tr>
<td>Great skua</td>
<td>Very low success</td>
<td>High success</td>
</tr>
<tr>
<td>Black-legged Kittiwake</td>
<td>Very low success,</td>
<td>Normal success</td>
</tr>
<tr>
<td></td>
<td>total failure in Orkney and some Shetland colonies</td>
<td></td>
</tr>
<tr>
<td>Arctic tern</td>
<td>Very low breeding numbers in Shetland &amp; Orkney, very low success with some total failures</td>
<td>Normal success, very variable</td>
</tr>
<tr>
<td>Common guillemot</td>
<td>Very low success</td>
<td>Normal success</td>
</tr>
<tr>
<td>Razorbill</td>
<td>Low success, very variable</td>
<td>No data</td>
</tr>
<tr>
<td>Atlantic puffin</td>
<td>Very low success</td>
<td>No data</td>
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</tbody>
</table>
4.3 **Seabird breeding performance in other North Sea countries in 2004 and 2005**

No previous overview exists of seabird breeding performance in North Sea countries other than the UK. We have here compiled available data for black-legged kittiwake and three species of terns for Belgium, The Netherlands, Germany and Norway to put the 2004–2005 results into context. Three aspects were included in the analysis: (1) breeding numbers since 1990, (2) reproductive success since 1990, and (3) the importance of sandeels as a component of the diet in the breeding season.

**Black-legged kittiwake**

*Sandeel reliance.* Sandeels have been identified as ‘staple food’ (occurrence in >50% of samples studied) in breeding black-legged kittiwakes in the Farne islands (Pearson 1968), the Isle of May (Galbraith, 1983; Harris and Wanless, 1997) and on Shetland (Furness, 1982; Furness, 1990; Furness and Barrett 1991). On Helgoland, sandeels were listed as ‘common’ prey (25–50% of samples studied) by Prüter (1989). There is no information on the utilisation of sandeels for the small population in southern Norway (ICES diet database 2006).

*Breeding numbers and reproductive success.* Few black-legged kittiwakes breed in southern Norway, and population censuses have produced rather inconsistent results. In Einevarden, only 20 pairs were recorded for 2005, whereas in between 1760 and 2100 pairs were found between 1983 and 2000. Klovningen, Veststeinen and Utvær were all abandoned in recent years for which data were available (certainly in 2005), whereas small populations (<125 pairs in each) were known to occur in the early 1980s. Stallbrekka had 145 pairs in 2005, perhaps as much as 600 pairs in 2003, but small numbers in other years (1983–2000 32–50 pairs recorded). On Runde, breeding numbers have fallen from 1812 pairs in 1980 to 294 in 2004 and 458 in 2005 (Lorentsen, 2005). There is no information on reproductive success available.

The reproductive success of the only colony of black-legged kittiwakes in Germany, situated at the cliffs of Helgoland, has not been monitored in recent years. The population increased from 2340 pairs in 1980 to 8600 in 2001, but has since declined to 6500 in 2005 (Hüppop, in prep.). Anecdotal information suggests the reproductive success was ‘not abnormal’ in recent years, but there is no data to back this up.

In The Netherlands, the black-legged kittiwake established a small breeding population on two offshore installations in the late 1990s (53°38’N, 04°34’E). The largest of these colonies was probably established around 1999 (two nests) and was censured for the first time in 2005 (45 nests; Camphuysen and De Vreeze, 2005). The second site was found to hold at least two nests in 2005, but this platform was not visited by scientists. On the main colony, guano samples revealed that sandeels indeed constituted the main prey together with sprat *Sprattus sprattus*. There is no data on reproductive success, but mean clutch size in nests with eggs (1.65 ± 0.49 eggs) suggested that success would be modest at best (perhaps typical for a ‘young’ colony with presumably relatively inexperienced birds; Camphuysen and De Vreeze, 2005).

**Sandwich tern**

*Sandeel reliance.* Sandeels have been identified as ‘staple food’ (occurrence in >50% of samples studied) in breeding Sandwich terns at Farne islands (Pearson, 1968), in north-east Scotland (Fuchs, 1977), at Juist (Germany; Garthe and Kubetzki, 1998), and at Griend (Veen, 1977; Stienen and Breninkmeijer, 1998; Stienen, 2006). There is no information on the utilisation of sandeels in the Belgian colonies and for the terns nesting in the Dutch Delta area (ICES diet database).
Breeding numbers and reproductive success. As in the UK, general trends in breeding numbers and reproductive success of terns are highly dependent on local conditions and the effects of predation, disturbance and flooding. As a result, fluctuations in parameters such as total numbers breeding or reproductive success may have little to do with feeding conditions.

Sandwich terns nesting at Griend (Dutch Wadden Sea) preyed mainly upon sandeels in some years, but mostly upon clupeids in other years (Stienen, 2006). Population censuses were available for 16 years since 1990 (including 2004 and 2005), ranging from 5000–11275 pairs. Reproductive success was monitored within enclosures between 1990 and 2004, ranging from 0.01 (2004) to 1.0 chicks per pair (1990) (Figure 4–2; E.W.M. Stienen, unpubl. data, 2006). In 2005, 27 Sandwich tern pairs nesting in an enclosure at Griend experienced comparatively high mortality through predation, and these birds produced only 0.26 fledglings per pair. From chick ringing effort, however, it was estimated that the colony as a whole (10,560 pairs breeding) produced c. 0.45 fledglings per pair. Bad weather in the early phase of the breeding season was responsible for high chick mortality (Willems et al., 2005). 2004 may be considered a breeding failure, with the lowest ‘success’ ever reported, whereas the reproductive success in 2005 was slightly below average.

![Sandwich Tern Griend, Wadden Sea (Netherlands)](image)

Figure 4–2: Sandwich tern reproductive success at Griend (The Netherlands), from E.W.M. Stienen (unpubl. data) and Stienen (2006).

The stronghold for Sandwich terns in the Delta area (The Netherlands) is (or was) in the Grevelingen area. A colony in that area was established several decades ago (Meininger et al., in prep.). Between 1990 and 2004, the population fluctuated between 1575 (1992) and 4200 pairs (2003) (Figure 4–3), and reproductive success varied between 0.25 and 0.80 chicks per pair. There is some doubt regarding the accuracy of the estimates of reproductive success, and there were certainly no methodological procedures in place such as enclosures that would lead to a proper assessment (P.L. Meininger, pers. comm.). Reproductive success in 2004 was apparently lower than before, whereas the site was not used in 2005. Two new colonies were formed in the Delta area (2650 pairs), together not making up for the numbers just lost in Grevelingen. Numbers in nearby Zeebrugge were also lower than in 2004.
Sandwich terns nesting in Zeebrugge (Belgium) prey mainly upon sandeels in some years and mostly upon clupeids in other years. Population censuses were available all years since 1990 (including 2004 and 2005), ranging from 46 (2002) to 4067 pairs (2004). Enclosures were used in seven years since 1997 using from 30–110 clutches and leading to estimates for reproductive success in five years of measurements ranging from nil (2002) to 0.8 chicks per pair (2003) (Figure 4–4). The reproductive success was high in 2004, and rather low in 2005.

**Common tern**

*Sandeel reliance.* Sandeels have not been identified as ‘staple food’ (occurrence in >50% of samples studied) in breeding common terns at North Sea colonies, but were ‘common’ prey (25–50% of the samples) at Farne islands (Pearson, 1968) and Baltrum (Germany; Frank, 1992). There are probably few colonies that largely rely on sandeels, whereas clupeids (Germany, The Netherlands), sticklebacks (Germany), and saithe (Shetland) have been identified as staple foods in the various colonies (ICES diet database; to check).
Breeding numbers and reproductive success. For common terns nesting at Griend (Dutch Wadden Sea), population censuses were available for 15 years since 1991 (including 2004 and 2005), ranging from 1068–3200 clutches. Reproductive success monitored within enclosures in 15 years of measurements ranged from nil (2002) to 1.0 chicks per pair (2003) (Figure 4–5). Common terns were moderately successful in 2004 (0.5 chicks per pair) and 2005 (0.6 chicks per pair). Reproductive success in 25 other locations within the Wadden Sea in 2005 (colonies ranging from 5–934 pairs, 2663 breeding pairs in total) ranged from 0.0–1.3 chicks per pair (in all c. 1940 chicks produced, or 0.73 chicks per pair; Willems et al., 2005).

![Common Tern Griend (Netherlands)](image)

Figure 4–5: Common tern population and reproductive success at Griend (Wadden Sea, The Netherlands), from Willems et al. (2005).

For common terns breeding in the Delta area (The Netherlands), population censuses were available for 12 years since 1994 (including 2004 and 2005), ranging from 3733–7001 clutches. Reproductive success ranged from 0.07 (2002) to 1.04 chicks per pair (1995) (Figure 4–6). Common terns largely failed again in 2005 (0.1 chicks per pair and reproductive success declined overall; Meininger et al., in prep).
For common terns breeding at Zeebrugge (Belgium), population censuses were available in all years since 1990 (including 2004 and 2005), ranging from 300–3052 clutches. Reproductive success ranged from nil (2002) to 1.4 chicks per pair (2001) in the six years these data were available (from Dumortier et al., 2003; Figure 4–7). In 2004, overall reproductive success amounted to 0.7 chicks per pair, but with contrasting results in two subpopulations: the original site largely failed (0.1 chicks per pair), whereas birds nesting on a newly created peninsula for nesting terns reproduced well (1.1 chicks per pair). Depredation by (probably) ferrets in the original breeding site may have caused the breeding failure.
**Arctic tern**

*Sandeel reliance.* Sandeels have been identified as ‘staple food’ (occurrence in >50% of samples studied) in breeding Arctic terns at numerous UK North Sea colonies, including sites on Shetland (Furness, 1982; Ewins, 1985; Monaghan et al., 1989; Furness, 1990; Furness and Barrett, 1991; Uttley et al., 1994), at Farne Islands (Pearson, 1968) and on Coquet Island (Monaghan et al., 1989). Sandeels were found, but never ‘common’ prey (25–50% of the samples), at colonies such as Scharhörn/Nigelhörn, (Hartwig et al., 1990, Niedernostheide, 1996), and on Minsener Oldeoog (Frick and Becker, 1995). Quantitative information for the small Dutch population is unavailable (ICES diet database).

**Breeding numbers and reproductive success.** The breeding population of Arctic terns nesting within the Dutch Wadden Sea declined from c. 2250 pairs in 1998–1999 to exactly 1000 pairs in 2004 and 1073 pairs in 2005. This decline followed an increase between 1990 and 1993, followed by a fairly stable period with c. 2000 breeding pairs. Between 1975 and 1989, around 1000 Arctic terns nested in the Wadden Sea area. Currently, half the population is situated on the uninhabited island of Griend, where no chicks fledged in 2005. Reproductive success in nine other locations within the Wadden Sea in 2005 (colonies ranging from 2–159 pairs, 541 breeding pairs in total) ranged from 0.0–1.0 chicks per pair (in all c. 255 chicks produced, or 0.35 chicks per pair; Willems et al., 2005). In recent years, around 0.20–0.25 chicks per pair were produced, probably explaining why the population is currently declining.

**Common/Arctic terns**

In Sogn and Fjordane (Norway), Arctic and common tern populations have declined between 1983 and 2005 from hundreds of pairs to nearly none (Table 4–2). In Hordaland, between 1980 and 2005, the breeding population fell from c. 2600 pairs to less than 300. Even though these censuses are probably incomplete, the results suggest that the tern populations in southern Norway are in distress. In Hordaland, the decline was obvious also within nature reserves, perhaps suggesting that prey availability rather than disturbance or management/conservation issues have influenced the decline.

On the German Wadden Sea islands, tern reproductive success was low during the last five years, mainly due to flooding by high water. However, in 2004, terns started to breed about four weeks later than normal, and many chicks died from starvation (B. Hälterlein, pers. comm.).

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<tbody>
<tr>
<td>Sum of six colonies, Sand F</td>
<td>253</td>
<td>322</td>
<td>390</td>
<td>10</td>
<td>0</td>
<td></td>
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<tr>
<td>Hordaland total</td>
<td>2633</td>
<td>1523</td>
<td>703</td>
<td>274</td>
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<td>2419</td>
<td>753</td>
<td>480</td>
<td>108</td>
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</table>

**Common guillemot**

*Sandeel reliance.* In the NW North Sea, common guillemot chick diet consists mostly of 1+-group sandeel in most years, although clupeids (mostly sprat) are important in some years (Harris et al., 2005). Adult diet is much more variable (Wilson et al., 2004). Very similar patterns occur on Helgoland for both chick and adult diet (Grunsky-Schöneberg, 1998; Sonntag and Hüppop, 2005).
Breeding numbers and reproductive success. No long-term data exist on common guillemot reproductive success outside the UK. However, measurements of chick fledging mass on Helgoland (Germany) show the lowest value on record in 2005, reminiscent of the declining trend observed on the Isle of May (Figure 4–8), where fledging mass and reproductive success are highly correlated (Frederiksen et al., in prep.).

![Figure 4–8: Mean fledging mass of common guillemot chicks at Helgoland in the SE North Sea and the Isle of May in the NW North Sea, 1986–2005. No data were available from Helgoland in 2004. Data from O. Hüppop and M. P. Harris (unpubl. data).](image)

Summary

It is very difficult to draw general conclusions about reproductive success of sandeel-dependent seabirds in North Sea countries other than the UK in 2004 and 2005. While some colonies seem to show lowered reproductive success in these years, there are so many individual factors at work (predation, disturbance etc.) that no general picture emerges. It is thus not possible to confirm exactly the spatial extent of the problems experienced by seabirds, and thus the underlying low availability of pelagic fish prey.

4.4 Previous large-scale seabird breeding failures in Europe

There are several previous cases of relative large-scale breeding failures of seabirds in NW Europe in recent decades, and most of these have been linked to crashes in stocks of pelagic fish. The ‘Shetland crisis’ in the late 1980s is particularly well documented. Here, surface-feeders suffered more or less complete breeding failures for consecutive years (black-legged kittiwakes and Arctic terns from 1985–1990, Arctic skuas from 1987–1990 and great skuas from 1988–1990), whereas diving species such as common guillemots were much less affected (Figure 4–9; Hamer et al., 1993; Uttley et al., 1994), although some problems were noted for Atlantic puffins (Martin, 1989). The underlying reason was local failures in sandeel recruitment, most likely linked to changes in the Fair Isle current, which in most years brings large numbers of sandeel larvae spawned around Orkney to Shetland (Wright, 1996). Atlantic puffins suffered repeated breeding failures in the Norwegian Sea over a twenty-year period, due to the almost complete absence of young herring Clupea harengus following a fishery-induced collapse in the Norwegian spring-spawning herring stock (Anker-Nilssen et al.,...
A crash in capelin *Mallotus villosus* stocks and lack of alternative prey caused breeding failures of most seabirds in the Barents Sea in 1986 and 1987, widespread non-breeding of Brünnich’s guillemots and approx. 90% mortality of adult common guillemots (Anker-Nilssen et al., 1997).

![Diagram](image-url)

**Figure 4–9:** Reproductive success (chicks per pair) of black-legged kittiwakes and common guillemots at Sumburgh Head, Shetland, from 1989 to 2005. Data from M. Heubeck (unpubl.).

### 4.5 Possible reasons for the observed breeding failures in 2004 and 2005

From the outset, there was a general suspicion that the underlying cause of the 2004 breeding failures was low availability of sandeels. The fact that all sandeel specialists were affected, and that problems were so widespread, certainly seemed to point to this explanation. The available data also suggested that the proportion of sandeels in seabird diet was lower than usual in North Sea colonies in 2004 (Harris *et al*., 2005; Mavor *et al*., 2005), further supporting this suggestion. Detailed studies on the Isle of May showed that the nutritional quality of sandeels and sprats fed to seabird chicks was extremely low in 2004 (Wanless *et al*., 2005), with lipid contents near zero and energy values far below the level seen in previous years. Both a lack of sandeels and low quality of the few fish available may thus have contributed to the problems faced by seabirds.

At the North Sea level, sandeels as well as herring and Norway pout *Trisopterus esmarkii* have shown several years of low recruitment since 2001 (ICES, 2006). The reasons for the repeated recruitment failures of pelagic fish are not quite clear, but one of several suggestions is that they are linked to climate-driven changes in plankton community composition (e.g. Arnott and Ruxton, 2002). There seems to be stronger evidence for this explanation for sandeels than the other species (ICES, 2006). One of the most important food sources for pelagic fish, including sandeels, is copepods of the genus *Calanus*. Two species are common in the North Sea, the northern *C. finmarchicus* and the southern *C. helgolandicus*. Since the late 1980s, the total abundance of *Calanus* copepods has decreased, and at the same time the balance between the two species has shifted so that *C. helgolandicus* has become dominant (Beaugrand *et al*., 2002; Edwards *et al*., 2002; Edwards and Richardson, 2004). The shift in copepod communities is thought to be linked to increasing sea surface temperature in the
North Sea (Edwards et al., 2002). *C. helgolandicus* tends to occur later in the year than *C. finmarchicus*, and the shift between the two species thus means that less food is available to pelagic fish during their critical growth period in spring and early summer. *C. helgolandicus* is also smaller, and it has been suggested that it is an inferior food item for fish because of lower lipid content (ICES, 2006).

Seabird monitoring data from the NW North Sea provided strong indications that the 2005 year class of sandeels was considerably stronger than in the previous years. In Shetland and Orkney, after a very slow start to the breeding season, black-legged kittiwakes suddenly initiated nest-building in late May (Mavor et al., in press), around the time that 0-group sandeels normally appear in the diet. Likewise, on the Isle of May there were several indications that conditions improved late in the season 2005, and that 0-group sandeels were available in large numbers (Harris et al., in press). Unusually, 0-group was more common in European shag diet than older sandeels. Black-legged kittiwakes had average or high reproductive success in most North Sea colonies (Mavor et al., in press), consistent with this species mainly feeding its chicks 0-group sandeels (Lewis et al., 2001). In contrast, common guillemot chicks are very rarely fed 0-group sandeels (Wilson et al., 2004), and in 2005 the chick diet was dominated by clupeids, indicating very low availability of older sandeels (Harris et al., in press). There are not yet any fishery data available to confirm that the 2005 year class was stronger than seen for several years, and no suggestions have been forwarded for why this should be the case.

The data from NW Scotland are not sufficiently detailed to provide clear evidence for why breeding failed for several species in this region in 2005. Much less is known about seabird diet in this region, and it is e.g. not clear how dependent seabirds are on sandeels. However, the similarity of the observed patterns to the situation in the North Sea in 2004, in terms of which species experienced failures, suggests that the underlying reasons may be the same. Sandeel aggregations in the two regions are separate and their dynamics are in general not synchronised (Frederiksen et al., 2005), so it is not too surprising if sandeel availability was low and seabirds experienced problems in different years. Whether there is a common causation for the lack of sandeels in the two regions in consecutive years is not clear, as much less is known about possible changes in plankton communities in the area west of Scotland than in the North Sea. The general role of climatic changes in driving the observed seabird problems in 2004 and 2005 thus cannot be confirmed, although the available evidence does suggest that the North Sea failures in 2004 could be linked to increasing temperatures.

### 4.6 Conclusions and recommendations

It was clear that the breeding problems experienced by sandeel-dependent seabirds along UK North Sea coasts in 2004 were more widespread in terms of areas and species affected than previously seen in the UK. The entire western North Sea coast, from Shetland to E England, was affected, and all sandeel-dependent species were seriously hit. However, it was not possible to confirm whether these problems also affected other parts of the North Sea. This was due to multiple factors: some of the relevant seabird species not breeding in other North Sea countries, lack of monitoring of breeding performance and diet and/or results not yet being available, and the lack of regional consistency in tern reproductive success because of, for example, colony-specific predation and disturbance. WGSE therefore strongly recommends that, in addition to breeding numbers, reproductive success and diet are monitored annually at representative sites in all countries and that results are reported promptly, allowing determination of the extent and causes of future widespread breeding failures (see also Chapter 5). This recommendation also applies to monitoring programmes in Canada, the United States and other parts of the world.

It should also be noted that lack of suitable food may also have affected adult and immature survival of seabirds. There is an inevitable delay in reporting results from monitoring of
survival, but future analyses of data from the few sites where this is done should concentrate on evaluating the overall effect of the lack of sandeels on seabird demography.

Events in 2005 provided some clues to the underlying causes of the breeding failures. The season was very late in the North Sea, but when birds finally started to breed, they mostly had an average or good season. Young (0-group) sandeels were prominent in the diet of all species studied. Consistent with the reported recruitment failures in 2001–2004 (ICES, 2006), it appears that availability of older sandeels was very low, but that a strong 2005 year class made reasonably successful breeding possible. The low availability of sandeels in 2004 was exacerbated by very low nutritional quality of the remaining few fish in at least some areas (Wanless et al., 2005). Anecdotal observations also show that seabirds in many colonies were feeding their chicks on snake pipefish 

Entelurus aequoreus

, a non-traditional and probably unsuitable prey item because it is hard to swallow, very bony and has low lipid content (Harris et al., in prep). Whether the sudden appearance of snake pipefish in seabird diets was entirely due to a lack of other prey or whether the apparent recent increase in range and abundance of this fish also played a role is not clear, but the importance of these two factors probably varies regionally (Harris et al., in prep.).

It is not known exactly what caused the low recruitment, availability and nutritional quality of sandeels in the North Sea in recent years, but available evidence does suggest that climate-driven changes in plankton communities, particularly Calanus copepods, may have been important (Arnott and Ruxton, 2002; ICES, 2006). Further field, experimental and modelling studies are needed to test this hypothesis.

4.7 References


5 Recommendations for a comprehensive monitoring programme for seabirds

At its 2005 meeting, ICES (2005) recommended the group should perform a review of the variety of methods applied across the North Atlantic region to monitor the performance of seabirds and to assemble a set of standardised and cost-efficient guidelines that could make monitoring more amenable to broad-scale analysis across regions and national borders. Developing recommendations for a comprehensive monitoring programme for seabirds was therefore put up as a term of reference for the 2006 meeting. As this is no simple, straight-forward task, WGSE needs to elaborate on this ToR on future meetings. This chapter therefore only presents a preliminary set of notes and guidelines on how to design and implement adequate monitoring schemes for breeding seabirds within the ICES region. WGSE will also distribute a questionnaire (Annex 5) to all countries to get an overview of the current monitoring effort within the North Atlantic.

5.1 Introduction

Seabirds are identified as very valuable components of marine ecosystems, not only for their attractiveness and recreational value, but also because many of them have proven to be excellent indicators of important changes in the marine environment (Furness and Monaghan, 1987; Furness and Camphuysen, 1997). Thus, often, seabird data give early indications of fluctuations in fish stocks and oceanographic conditions (Montevecchi, 1993). Monitoring in its broad sense can be defined as the process of gathering information about some system state variables at different points in time for the purpose of assessing system state and drawing inferences about change in state over time (Yoccoz et al., 2001). In the case of monitoring programmes for seabirds, the systems of interest are typically seabird populations, and the state variables of interest include quantities like breeding population size, reproductive success, diet, pollutant concentration or adult survival.

Monitoring programmes for seabirds have been implemented in European and North American countries for many years. The scale, design and intensity of these programmes vary greatly. Some have been extensive, like for instance the monitoring of seabird numbers and breeding success in Britain and Ireland, which involves yearly surveys of hundreds of sites for several species (Mavor et al., 2005), and thus enables the coverage of large geographic areas. Alternatively, some seabird populations have been the subject of very intensive and long-term programmes conducted on a few locations (Wooller et al., 1992). Such monitoring programmes have allowed the detection of effects of various environmental changes on seabird populations at different scales, such as the effect of climate (e.g., Durant et al., 2003; Frederiksen et al., 2004) and oil spill pollution (Votier et al., 2005). Obviously, the extent and strength of the inferences drawn from these monitoring programmes varies with the design used.

In this context, recommendations for the development of comprehensive monitoring programmes for seabirds can build on the experience gained from these programmes, and can be done by considering three key questions that need to be addressed for any monitoring programme: why monitoring, what to monitor (which species and parameters), and how to monitor (including sampling frequency and how to estimate parameters while accounting for sources of error).

5.2 Why monitoring?

A critical step in the design of any monitoring programme is to identify the reason why the monitoring programme is to be implemented, as this will condition what parameters should be monitored and how. This is especially important as monitoring activities have often been
criticized for their lack of justification. Sound monitoring programmes can be implemented either for management purposes (e.g., to detect the need for potential management measures and/or to detect the effect of management measures to keep the system in a given state), or for pure scientific reasons (often it then relates to attempting to understand processes underlying changes in system state). As the understanding of underlying processes is often of direct or indirect relevance for knowing what management measures should be taken, comprehensive monitoring programs could ideally combine scientific and management objectives.

The main focus of the monitoring we consider here is to assess the health of seabird populations as suggested by the EcoQO proposed by ICES (2005). The overall purpose is to detect undesired trends in time to uncover the most important reasons, such that any mitigating actions can be identified and implemented in time to be effective. However, seabird monitoring also provides additional value as the performance of many seabirds indicate important changes in the marine environment.

It should be noted that an increased interest has become apparent in spatial issues linked with the response of seabird populations to environmental changes, which has direct implications for the development of monitoring programmes (Box 5–1). This is because such monitoring programmes should enable evaluation of the effect of large scale changes of the environment on seabird populations (e.g., see Chapter 3), but also because the meta-population structure of seabird populations, and their exploitation of large areas of the seas at different time of the year, make them exposed to various factors at different scales.

Box 5–1: Spatial scale in seabird monitoring.

One of the issues in the design of monitoring programmes is the selection of representative colonies. This requires some assumptions or preferably knowledge about the spatial scale of any geographical patterns in monitored parameters. At the extremes, if all colonies behave differently, it is not possible to select representative colonies, and if there is no spatial variation, purely practical considerations will determine which colonies to monitor. More realistically, there will be some spatial structuring in most or all monitored parameters, but little information about the most relevant scale to monitor will usually be available when a programme is being designed. However, existing knowledge about relevant aspects of the physical or biological environment can often provide useful pointers for the regional structure of the programme, together with current knowledge of the population biology of the species considered.

Spatial variation in demographic parameters is not well known for most seabirds, but some studies have been carried out on black-legged kittiwakes. On the wide-range scale, this species has extremely variable survival and reproductive success, with colonies in the N Pacific having higher survival and lower reproductive success than most Atlantic colonies (Frederiksen et al., 2005a). At the smallest scale, survival and reproductive success can vary between nearby colonies or parts of the same colony (Danchin and Monnat, 1992), and change in local numbers of breeders can be largely explained by differential dispersal and recruitment of individuals among colonies or sub-colonies (Danchin et al., 1998). On intermediate scales, possibly more relevant for the design of monitoring programmes, analyses of data from the UK Seabird Monitoring Programme have shown clear evidence of spatial structuring, with reproductive success being highly correlated between colonies within regions, but mostly uncorrelated between regions (Frederiksen et al., 2005b). Further analyses show that reproductive success at seven colonies within one such region, SE Scotland, all show similar temporal patterns and relationships with environmental parameters, and that one colony is particularly representative of the region as a whole, with a correlation of 0.96 between this colony and the regional mean (Frederiksen et al., in prep.). Further work is needed to confirm
whether these findings can be generalised to other regions and species, and thus whether it is
generally possible to designate biologically meaningful regions for monitoring as well as
representative colonies in each region. Further work on factors affecting dispersal and
recruitment at various hierarchical scales could be especially interesting in these respects, as it
could help quantifying how independent nearby colonies are in terms of breeding numbers.

5.3 Ongoing seabird monitoring in the ICES areas

Before discussing what to monitor and how to monitor things, it is worthwhile to describe the
extent of ongoing monitoring programme in the ICES areas. As an example, information
about the current UK Seabird Monitoring Programme within Scotland is provided in Box 5–2.
Existing monitoring programmes vary in many ways and in order to fully review the current
monitoring of seabirds in the ICES region, WGSE will distribute a questionnaire (Annex 5) to
all countries to map what their present monitoring schemes cover in terms of species, sites,
parameters and monitoring frequencies. We aim to present a summary and more detailed
review of this material in the report from our working group meeting in 2007. This will
provide important basic elements for evaluating the possibility of actually developing large
scale comprehensive programmes.

Box 5–2: Seabird monitoring in Scotland.

The status and trends of seabirds in Scotland have been monitored using two complementary
programmes since 1985. There have been two broadly comprehensive censuses during 1985–
1987 (Seabird Colony Register, SCR) and 1998–2002 (Seabird 2000). These give snapshot
estimates of status and long-term change without colony sampling bias. Between these
censuses, counts of whole sample colonies or counts of study plots within colonies have been
collated by the Seabird Monitoring Programme (SMP) with the aim of describing annual
patterns of change. The complete censuses were carefully designed and co-ordinated, but
counts contributed to the SMP were largely collected on an ad hoc basis by professional
biologists, reserve wardens and volunteers.

Since data collection for the SMP is not based on a stratified random design, they have many
biases that need to be overcome by design of the statistical analysis. Not all sites in all years
are counted, and so missing values have to be imputed to obtain likely trends. Some regions
are undersampled relative to the proportion of the national population they host, and so trends
have to be weighted accordingly in order to avoid bias. Large colonies are less likely to be
counted in their entirety than small ones owing to logistics, and this can result in
overestimation of population growth rates due to density dependence. Plot counts are often
available between complete counts for some species at large colonies, and in order to include
these data models have to have a hierarchical design.

Analysis of the data for selected species in Scotland between 1986 and 2004 was approached
using a Bayesian inference model (JNCC, in prep.). The model imputes missing counts based
on trends within and across colonies, has a hierarchical design which allows inclusion of both
whole colony and plot counts and weights trends by colony size and regional importance to
overcome the previously mentioned sampling biases. The approach is more flexible and there
is greater control over the assumptions that are made than in traditional chain indices or
generalised linear modelling methods. The hierarchical component of the model assumes that
plot counts are representative of those across the whole colony, though in some cases this was
clearly untrue (e.g. great skuas on Hoy, Orkney). Despite this problem, the model produced
accurate trends for most seabird species studied, with extrapolations of annual change from
the SCR census predicting status during the Seabird 2000 census reasonably well. The
predicted trends were more accurate than those produced by chain indices that suffered from biased sampling with respect to colony size and density-dependent growth. However, in the case of tern species and great cormorant, trends were inaccurate owing to site colonisation and extinction events that the model predictions were unable to track because of assumptions that were made. Although chain indices produced more accurate trends in these species, these remain unsatisfactory, and require further refinement to yield properly weighted estimate of trends with confidence limits.

The project has demonstrated that data collected on an ad hoc basis can yield reliable annual population trends, but that sophisticated analyses are required to overcome the inherent biases in such data. However, in some instances biases may be extreme and unquantifiable, and so careful design of sampling is advisable to ensure that estimated trends are accurate.

5.4 Monitoring priorities

5.4.1 Selection of species

A number of specific considerations should be made when selecting the target species for a reasonable effort of seabird monitoring. WGSE recommends that priority is given to those species that are:

- valuable indicators (in terms of their sensitivity, cost-efficiency and early-warning capacity) of ecosystem changes that are important for the well-being and management of other marine resources.

- monitored throughout most of their breeding range within the ICES region, in order to facilitate wide-scale analyses of population dynamics.

- of special conservation concern, either because they are listed on the national and/or European red lists, or they are considered as problem or key species.

- particularly vulnerable to impact factors that are expected to be of extra importance.

- of special international importance, i.e. the national population constitutes a large part (≥25%) of the biogeographical population (normally the European or NE Atlantic population) it belongs to.

Additionally, considerations should also be made with respect to:

- including representatives from each of the main ecological groups of seabirds present as defined by their main feeding areas and feeding ecology (Table 5−1).

- selecting species that have proven to be (or are expected to be) representative for several other seabird species that are more difficult to monitor.

- avoiding species and/or parameters that, for different reasons, are highly impractical or unfeasible to monitor properly (low cost-efficiency).
Table 5–1: The seabird species or seabird taxa that occur in significant numbers in European waters, grouped according to their main feeding areas (pelagic versus near-shore or deep versus shallow waters) and feeding behaviour (diving, plunge-diving or surface-feeding) in the breeding season.

<table>
<thead>
<tr>
<th></th>
<th>PELAGIC/DEEP WATER</th>
<th>NEAR-SHORE/SHALLOW WATER</th>
</tr>
</thead>
<tbody>
<tr>
<td>Diving</td>
<td><em>Uria</em> guillemots, Razorbill, Atlantic puffin, Little auk</td>
<td>Divers, grebes, cormorants, seaducks, Black guillemot</td>
</tr>
<tr>
<td>Plunge-diving</td>
<td>Gannets</td>
<td>Terns</td>
</tr>
</tbody>
</table>

5.4.2 Selection of monitoring parameters

ICES (2005) concluded that the huge variety of possible mechanisms underlying changes in seabird breeding numbers makes it necessary to also monitor different parameters of population dynamics for (at least some of) the key species, although these parameters are usually more labour intensive to monitor adequately. The great advantage of this approach is that it enables an immediate exploration of possible reasons for any population trends of special concern, without having to spend additional years to collect less adequate *a posteriori* information on the same parameters. As most seabirds are migratory outside the breeding season, trends in population numbers are also affected by environmental conditions far away from the breeding areas, in many cases outside the ICES areas in question. These factors are probably best reflected by changes in adult survival rates, whereas reproductive rates and chick diets (as well as other aspects of adults’ feeding ecology or their physical condition) are likely to be better indicators of local conditions within the breeding seasons.

Based on this, WGSE recommends that for all breeding species identified as important targets for monitoring, the monitoring should preferably produce series of annual data for the following key parameters:

- Population size
- Adult survival rate
- Reproductive success (specific parameters varying between species)
- Diet of breeding adults and/or chicks

Whereas the first three are needed to explore the essential dynamics of the target populations, data on diet are considered valuable as many environmental factors (man-induced as well as natural variation) affect seabirds indirectly through their food base. Except for the sampling of diets, which is treated in more detail in Chapter 6, references to standardised methods for sampling these parameters are given in Section 5.5.4. Due to the relatively high efforts needed to carry out state-of-the-art monitoring of survival rates (by capture-recapture techniques) and to sample diets throughout a significant part of the breeding period, these parameters are usually best monitored on a limited selection of sites, only.

Members of the WGSE have produced several reviews on the value and suitability of different population parameters for monitoring purposes (e.g. Becker and Chapdelaine, 2003; Furness *et al.*, 2003). Although the group did not have enough time at its 2006 meeting to assess (in relation to current and potential threats to seabirds in the ICES area) the value and suitability of monitoring other parameters than those above, we intend to revisit this task at our next meeting. For example, parallel to the collection of data on the key parameters listed above, additional and potentially valuable information might be collected with little extra effort,
including simple indices of foraging performance (e.g., feeding rates and breeding site attendance), breeding phenology and likelihood of breeding, as well as sampling tissues for various purposes (e.g., contaminant levels, parasites and pathogens, genetic and stable isotope analysis, sexing).

5.5 Monitoring methods

5.5.1 Hierarchical approach combining utility and practicality

Depending on the objectives of a given monitoring programme and the parameters to be monitored, some hierarchical approach combining an extensive survey of numbers of breeders, productivity and chick diet on samples of study plots at different locations and more intensive monitoring of annual survival in a limited number of locations could be suggested.

In addition to practical aspects of monitoring methods, sound monitoring has to consider potential sources of errors when estimating parameters for seabird populations. Of these, detection errors and spatial variability, are discussed in the next Section (5.5.2), and a more specific discussion of sampling and analytical design for population trend estimation is provided thereafter (Section 5.5.3).

5.5.2 Accounting for sources of error: detectability issues and spatial variability.

Two important sources of variability of parameter estimates of seabird populations are detection error and survey error in relation to spatial variability.

For breeding numbers and demographic parameters such as survival, the first source of error occurs because few survey methods permit the detection of all individual animals or breeding events in surveyed areas (Williams et al., 2001). For estimating parameters like annual survival rate, capture-mark-recapture methodology is now commonly used for seabird populations (Cam et al., 1998; Frederiksen et al., 2004; Oro et al., 2004; Sandvik et al., 2005) and is highly recommended. Classically, this approach involves the marking of individuals in the field, their recapture at later occasions and the use of probabilistic modelling to account for the fact that some individuals remain undetected in the field even when they are still alive (Lebreton et al., 1992). One important limit to this sort of approach is that individuals are considered dead if they leave the study area, which thus confounds permanent emigration with mortality (this is why the term “apparent survival rates” is used). The assumptions made in terms of homogeneity in detection probabilities among individuals are also important to consider and various approaches have been proposed to account for these issues, usually combining complementary sources of information.

The second source of error of population parameter estimation involves the inability to survey large areas entirely, and the resulting need to draw inference about large areas based on (usually non-random) samples of locations within those areas. What should be avoided is the focus on a few very subjectively chosen monitoring plots and the sampling design should provide a survey of a representative set of monitoring plots. As illustrated in Box 5–1, these issues can be especially important at some small spatial scales. The next section explains how the sampling and analytical design of seabird population trends can allow dealing with some of these issues.

5.5.3 Sampling and analytical design for population trend estimation

Ideally, a complete census of all colonies for species of interest would be conducted annually. This is achieved for some relatively rare or restricted range species (e.g. Roseate Tern), or species such as the northern gannet, which breed only in a restricted number of locations and are easy to monitor; for most species, the number and size of colonies are too great for this to
be practicable. In these instances, sampling through time and space is needed to estimate trends, and surveys and/or data analyses have to be carefully designed in order to avoid bias.

Complete censuses overcome spatial sampling bias and provide information and absolute status, distribution and relative importance of regions or colonies (Mitchell et al., 2004). Complete censuses are complementary to annual sample surveys as they facilitate assessment of potential sampling biases when designing data collection or analyses, and independent post-hoc assessment of trend accuracy (JNCC, in prep). Repeat censuses also provide information on population change but, owing to the long time intervals between these, distinguishing longer-term population trends from shorter-term fluctuations can be difficult or take several decades to become evident.

Annual population change, and estimation of trends within relatively short time periods, can be estimated by sampling counts at the whole colony or study plot scale on a year-to-year basis (e.g. Mavor et al., 2005). Ideally, the colonies and plots would be selected according to a random stratified design to avoid bias in trends, and each site would be counted every year. In most countries, however, annual sampling is on an ad hoc basis by a mixture of professional biologists, wardens and volunteers. These data have the potential to produce accurate trends for minimal cost, provided that inherent biases are overcome analytically.

In most ad hoc surveys, not all colonies are counted in every year, such that comparisons of summed annual counts reflects both changes in status and the pattern of missing counts (Ter Braak et al., 1994). Spatial bias is also common in ad hoc surveys, with those colonies or plots that are easy to access or observe being over-sampled. Bias can also occur relative to density. Density-dependence often results in large colonies growing more slowly than small ones, and in core areas of colonies changing more slowly than peripheral ones (e.g. Moss et al., 2002). Whole colony counts are often biased towards small colonies as they are easier to count annually, with the result that colony growth rates are overestimated. Sometimes plots are initially located in areas of the colony where birds are breeding and thus do not capture expansion into areas not used when the plots were set up, further underestimating rates of increase. Plot counts are often biased towards core areas where there are large numbers of birds available to count, such that rates of increase may be underestimated. Furthermore, plot counts are often discontinued when the number of birds becomes “too few to be worth counting”, which results in rates of decline being underestimated also.

Biases in ad hoc annual surveys can be overcome by modelling in some cases. Problems with missing values can be remedied using chain indices, in which only colonies counted in consecutive years are included in analysis, but this approach is wasteful of data and may exacerbate other biases (Ter Braak et al., 1994). GLMs such as TRIM (Pannekoek and van Strien, 2001) or Bayesian inference models (JNCC, in prep) are preferable to impute missing counts, and these also allow statistical significance of trends or year-to-year changes to be tested. Of these, the Bayesian method is the most flexible as it allows both whole colony and plot counts to be modelled in a hierarchical manner and makes weaker assumptions regarding synchronicity of trends across sites compared to GLMs (which reduces density-related biases). Spatial bias in trends can be overcome by weighting trends according to regional status as determined from complete census data where this is available (JNCC, in prep). An example from modelling seabird trends in Scotland is given in Box 5–2. Sophisticated analyses cannot be relied on to overcome shortcomings in all data however, and surveys should be designed to minimise the biases discussed above in order to attain the maximum accuracy.

The precision of trends as well as their accuracy is a consideration when designing surveys. Sampling has to be at an effort that allows trends of interest to be detected within an acceptable time-period. The statistical power to detect trends increases with sample size but declines with variance in trends among sampled units (Steidl and Thomas, 2001). Power analyses can be used to determine the minimum sample size needed to detect the desired rate
of change for an observed level of variance within an acceptable time-period and these can be useful to produce a parsimonious monitoring programme (Anker-Nilssen et al., 1996; Sims et al., in press).

When designing monitoring programmes the dynamic nature of seabird populations needs to be considered. Shifts in range or distribution may result in biases changing, and increases in variability of trends among units may result in loss of power. Monitoring programmes therefore need to be reviewed periodically to maintain the desired accuracy and precision of results.

5.5.4 References to descriptions of standardised monitoring methods

Standardised methods for monitoring of population size and reproductive performance are described for a variety of seabird species by Walsh et al. (1995). In relation to the challenges outlined in the above sections, we aim at reviewing these and supplementary methods in more detail at our next meeting.

Standard methods for designing capture-mark-recapture programme to estimate a key demographic parameter like annual survival rate are described in various outlets (Pollock et al., 1990; Lebreton et al., 1992; Williams et al., 2002), available together with software for data analyses (e.g., White and Burnham, 1999). As seen in Section 5.5.2, the application of such approaches to seabird populations has been developing greatly over the last 10 years, and the capture-mark-recapture of individuals is now a significant component of most colony-based monitoring programmes of seabird populations.

5.6 References


6 Recommendations on how to sample diet and how to report results of dietary studies in seabirds

The development of recommendations on how to sample seabird diet and report results in a standard manner is dependent on a thorough review of the methods being used today. A review of sampling methods was all but completed during the 2006 meeting, while that of reporting results was only briefly discussed. Because time ran out, the latter section and the final recommendations were deferred as a ToR in the 2007 working group meeting.

Data on seabird food consumption in the North Sea were provided to the Study Group on Multispecies Assessments in the North Sea in the latest available form at the Annual Meeting of SGMSNS in February in Copenhagen when S. Garthe and U. Kubetzki participated in that SG meeting.

6.1 Introduction

Many methods have been, and still are being used to study the diet of seabirds. Some are based on opportunism whereby samples (e.g., from watching food uptake directly or collecting dropped fish, regurgitated pellets, or faeces) are collected ad hoc, whereas others aim to be more systematic through regular collections or sightings made during a given time window. Techniques also vary greatly and range from the direct killing of birds to inspect their stomach contents through to totally non-disturbing and repeatable observations of fish-carrying birds, or the indirect methods including observations of feeding flocks, analyses of faeces, regurgitated food remains (pellets), and tissue collection for isotope or fatty-acid analyses. Unfortunately, all have biases of one kind or other (Duffy and Jackson, 1986; Rodway and Montevecchi, 1996; Carss, 1997; González-Solís et al., 1997; Andersen et al., 2004), and the vast majority are restricted to the short breeding season when birds are readily accessible on or near land.

As to the majority of the year when seabirds are spread along the coasts and over the open seas, there is no completely satisfactory non-destructive method for sampling diet. As a result, far too little is known about what and how much seabirds eat when they are at sea, outside the breeding season or for immature and non-breeding birds.

6.2 Stomach sampling/regurgitations

In order to determine the diet of seabirds, it is necessary to extract items from the digestive tract, sort and identify them and take measurements such as mass, linear dimensions and volume. Since some items extracted will be considerably damaged or even lost due to digestion, it is often necessary to make estimates of mass and dimensions of the original food item based on fragments. Food items may be found in the crop, proventriculus, oesophagus, gizzard or small intestine. Generally, the only items retained in the gizzard are hard parts such as bones, shells, exoskeletons, polychaete jaws and squid beaks. Everything from the proventriculus up to the mouth can be sampled by lavage without harm to the bird. Sampling the gizzard or intestine, however, is only possible from dead, dissected specimens.

Size of digested prey can often be estimated from measurements of undigested hard parts such as otoliths, bullae, bones, shells and squid beaks, but the accuracy depends greatly on the amount of digestion and wear of these items (see Section 4 Pellets).
6.2.1 Dead Birds

Shooting birds at sea is the surest way of obtaining data on diet, and is relatively free of bias resulting from differential attraction to any kind of trap (except for ship-followers”). Shooting has the obvious limitation of killing the bird, which tables the issue of ethical concerns and which may have significant consequences for long-lived and endangered species and will thus become increasingly unacceptable as a tool for sampling diet. However, birds shot for other reasons, e.g. for pollutant analyses, harvesting (such as the Newfoundland turre hunt) or shot as pests (at aquaculture sites – NB. these may provide very biased data), etc. have successfully been used for diet studies (e.g. Rowe et al., 2000). Other sources of dead birds are oiling incidents, bycatches in fishing gear and beached birds (e.g. Lorentsen and Anker-Nilssen, 1999; Ouwehand et al., 2004), although the latter, beached birds have often starved to death and thus may yield few or biased data. Dead birds may arrive on beaches in a trickle, e.g. as a result of chronic oil pollution, or may hit a coastline in masses, e.g. after a major oiling incident. Such large scale events should be seized for diet studies whenever possible, as they often provide large samples across a range of species from the same time and location (see e.g. Ouwehand et al., 2004), even though the logistics of such “sudden samples” are often hard to deal with.

The entire digestive track, from mouth through intestine, is usually removed from the bird as soon as possible after death and frozen or preserved in either ethanol or formaldehyde. A substantial fraction (often 30% or more) of birds shot at sea do not, however, contain any food items, other than bony fragments in the gizzard. Therefore substantial numbers need to be shot in order to obtain a sample large enough to ensure representation of all prey items taken. Care needs be taken for the differential digestion of food items in different portions of the digestive track. Items in the crop can be near intact, but the further an item progresses down the digestive track, the more it is digested and consequently the more difficult it may be to identify. Items in the gizzard may be retained for a long time; sometimes until they are forcibly regurgitated as a pellet. Squid beaks or polychaete jaws, for example, may be retained for a month or longer, and this retention needs to be taken into account when estimating dietary composition based on dissected dead birds. As the soft parts of squids or polychaetes are digested quickly, the beaks or jaws in the gizzard are often the only evidence of their importance in the diet. Nevertheless, using the number of these items in the gizzard will likely overestimate their proportional contribution due to retention.

6.2.2 Regurgitations

Some birds, especially nocturnal petrels, when attracted to lights at night become disoriented and land upon a ship’s deck or the ground. To lessen mass or as a panic response, they vomit the contents of the upper intestinal tract. Sampling this way can be especially valuable as it may be the only way to obtain dietary information from birds at sea and/or outside the breeding season. The problem with this technique of sampling is that it is entirely opportunistic and dependent on certain weather conditions, as birds are much more likely to be attracted to lights during foggy or rainy weather. Nevertheless, such sampling can produce valuable information on the food types available at prey patches at sea.

Other species like gannets, cormorants, gulls and terns on the nest or on their way to feed chicks may regurgitate food held in the proventriculus if disturbed. Chicks also spontaneously regurgitate in response to disturbance, or can be easily stimulated to regurgitate. Such samples may often be of little or only partly digested material which is readily identifiable in the field (e.g. gannets, cormorants) or on return to the laboratory (gulls, kittiwakes) such that the data may be fairly good for estimates of diet diversity. Another advantage is that it can be repeated (using different birds each time) throughout the breeding season. The hard body parts (otoliths, bones, etc.) are also often not worn, or digested (although there is a differentiation in
digestion rates between opaque and hyaline otoliths) thus allowing reliable determinations of prey size and hence energy content.

One limitation of this method is that the disturbance involved in some breeding colonies limits the numbers of visits possible. Furthermore, the proportion of ingested items in the regurgitations is variable, so one cannot use the amount regurgitated as an estimate of total crop contents or meal size.

6.2.3 Stomach lavage, emetics

If the bird does not regurgitate “voluntarily”, the upper intestinal tract can be safely sampled without harming the bird by flushing the contents out with water. This process, referred to as lavage, stomach flushing or water off-loading, involves pumping salt water through a tube inserted in the mouth of a bird and catching the regurgitated contents in a bag, sieve or bucket (Wilson, 1984; Ryan and Jackson, 1986). A latex tube is inserted deep into the bird’s oesophagus, and salt water pumped (using a syringe) in the other end of the tube. The bird is then inverted over a suitable receptacle into which the water and stomach contents are emptied. The process should be repeated to ensure complete emptying of the gastric system.

The limitations of lavage relate to how the birds are captured in the first place; as many birds vomit immediately upon being captured in a mist net or trap, so appear to be empty upon having their stomachs flushed. It has also proved difficult to use in some groups of seabird, e.g. auks, but see Wilson et al. (2004).

Birds do not always eject all contents of the upper gut tract during lavage, and may be induced to do so using an emetic (Ryan and Jackson, 1986) such as the Texel ferry coffee (Camphuysen, pers. comm.).

6.3 Faeces

Bird faeces have been used in various ways to reconstruct diets. Hard parts from prey, such as bones, scales, eggs or otoliths of fish, parts of the exoskeletons of crustaceans, jaws of squid and nereid worms, setae of nereid worms, or shell hinges of molluscs may all survive digestions and are often excreted with the faeces. If such parts can be identified and still bear a relationship with original prey size, these may be used to identify prey and reconstruct prey size. This method has been applied to many different piscivores, most notably on pinnipeds and otters (e.g. Pierce et al., 1991; Kingston et al., 1999; Andersen et al., 2004; Tollit et al., 2004). Seabirds that excrete such remains through their faeces are also candidates for such studies and many have been carried out on omnivorous gulls and skuas (e.g. Andersson and Götmark, 1980; Ambrose, 1986; Kubetzki et al., 1999; Kubetzki and Garthe, 2003), piscivorous ducks (Rodway and Cooke, 2002), mollusc-eating seaduck (Swennen, 1976; Nehls, 1989; Nehls and Ketzenberg, 2002; Leopold et al., submitted), benthos-feeding waders (e.g. Dekinga and Piersma, 1993; Scheiffarth, 2001) and other birds (e.g. Ormerod and Tyler, 1991; Taylor and O’Halloran, 1997). Relatively few studies have, however, been carried out on terns (e.g. Veen et al., 2003; Stienen et al., in prep.).

The advantage of the method is that it is non-invasive and low-tech. Furthermore, large sample sizes can be processed and time series can be built by repeated sampling schemes. Given that different methods often reveal different prey types, studying remains in faeces may uncover prey species previously unthought-of, such as the find of lots of Nereis jaws in sandwich tern faeces (Stienen et al., in prep.). Faecal or scat samples may be used to identify the sex of the predator involved, allowing sex-specific studies of diet (Reed et al., 1997). Being widely used and with samples being readily available, particularly in seals, the method has been rather extensively tested against other diet study methods (Prime and Hammond, 1987; Dellinger and Trillmich 1988; Crottrell et al., 1996).
These tests have, however, revealed that studies of faeces are unlikely to reveal all prey taken by the predator. Some prey do not survive digestion in a way that would allow finding traces in the faeces, while some birds also use other means to rid themselves of prey hard parts, e.g. through regurgitation of pellets (see below). Faeces are unlikely to be collected offshore, at sea, unless a suitable platform on which faeces are deposited are available for sampling (e.g. Camphuysen and de Vreeze, 2005). Processing faecal samples can be unpleasant although several washing methods have been developed (Bigg and Olesiuk, 1990; Brasseur and Janssen, in prep.). It is also time-consuming compared to e.g. measuring whole fish in a bird’s oesophagus, and good reference collections (e.g. Härkönen, 1986; Watt et al., 1997; Leopold et al., 2001) are required. Prey remains are bound to be worn after passage through the predator’s gut, and correction for wear and tear needed. Some parts survive better than others and some prey may be completely overlooked or greatly underestimated.

### 6.4 Pellets

Several seabirds eject indigestible prey remains in regurgitated pellets. These may be collected and the remains sorted out, using similar methods as described under “Faeces”. This method has been widely used on cormorants and shags (e.g. Kennedy and Greer, 1988; Grémillet and Argentin, 1998; Hald-Mortensen, 1995; Leopold et al., 1998; Olmos et al., 2000), pelicans (e.g. Derby and Lovvorn, 1997), gulls (Meijering, 1954; Spaans, 1971; Wietfeld, 1977; Kubetzki et al., 1999; Kubetzki and Garthe, 2003), terns (e.g. Granadeiro et al., 2002; Veen et al., 2003) and other birds such as waders, kingfishers, dippers (Swennen, 1971; Jost, 1975; Cairns, 1998). Being widely used and with samples being readily available, particularly in cormorants, the method has been extensively tested against other diet study methods (Brugger, 1993; Harris and Wanless, 1993; Russell et al., 1995; Trauttmansdorff and Wassermann, 1995; Zijlstra and van Eerden, 1995; Suter and Morel, 1996; Casaux et al., 1997, 1999).

Again, one big advantage of this method is that it is non-invasive and low-tech. Large sample sizes can be processed and time series can be built by repeated sampling schemes. The method is also quantitative, on the assumption that birds generally eject one pellet per day and that this pellet will contain the hard parts of all prey eaten. Although these assumptions are often violated, pellet studies do allow some quantification of diet. The method is, however, better for determination of diet composition rather than for quantification of consumption.

Many different prey types have been found in pellets, including some unexpected ones (e.g. Leopold and van Damme, 2003), suggesting that indeed most (but not all – see below) prey can be traced back in the pellets. Although finding pellets is often restricted to breeding colonies or roosts, in this case this is not a great problem as the birds concerned are largely feeding locally and inshore. Pellets can be collected from any dry surface where the target birds breed or roost, such as offshore lighthouses and platforms, or even especially designed floating pellet-collecting devices (Gagliardi et al., 2003).

Some comparative studies have, however, clearly indicated that particularly the hard parts of small prey, may not end up in the pellet but in the faeces (e.g. Veen et al., 2003). Furthermore, as in faecal studies, some prey do not leave hard parts in pellets and processing pellets and making reconstructions of number of prey and prey sizes is time consuming.

Further problems arise as a result of the possibility of secondary consumption of prey by the seabird, i.e. the pellet may contain remains of prey present in the digestive tract of the fish consumed by the seabird. For example, Johnson et al. (1997) suggested that the invertebrate prey found in the pellets of double-crested cormorants were prey of the fish consumed and not of the cormorants themselves. This source of error may also be relevant in faecal studies, those of regurgitated remains and in analyses of dead birds containing partly or completely digested material.
6.5 Archaeological: guano, middens and mummies

Old geological deposits, including guano layers in recent and abandoned seabird colonies (e.g. Rand, 1960) or archeological sites may contain information on diets of seabirds in the past. Perhaps even more spectacular, although of little relevance to modern diet studies of seabird, prey remains are sometimes found in fossil seabirds (e.g. Mayr, 2004).

6.6 Fish dropped in the colony

Fish dropped by adults returning to the colony or dropped by chicks during feeding are often found on the ground or on breeding ledges, and may be readily collected and identified. They are, however, poor indicators of food choice. In mixed colonies, the species that dropped the food is generally unknown, and those dropped by chicks may be unrepresentative of the fish normally eaten. For example, guillemot chicks may reject fish that are too large to swallow, or those dropped by displaying guillemots (often non-breeding birds) may not be representative of those caught by chick-feeding adults.

6.7 Observations and collection of food from fish-carrying species

Some seabird species bring whole fish (and sometimes but rarely other food items) carried openly cross- or lengthwise in the bill to their chicks. Some seaducks too bring large prey items to the surface before swallowing them. With a little practice and for species carrying single or few fish, e.g. terns, guillemots, razorbills, or black guillemots, it is generally easy to identify such fish from a distance using binoculars or telescope as the bird stands in the colony (Birkhead and Nettleship, 1987; Harris and Wanless, 1995). It is further possible to estimate fish size (e.g. small, medium, large) in relation to e.g. bill length. In some cases, identification and estimates of fish size can be controlled by subsequently catching the observed birds and collecting the fish (see below). For species carrying many small fish (e.g. puffins), species identification and quantification is also possible (and often used) but the possibilities of observation error are larger (Rodway and Montevecchi, 1996).

The main advantage of being able to make direct observations of fish is the possibility to collect large samples without any disturbance of the birds in question. If the species breed in dense colonies, e.g. guillemots, the simplicity of the method and the possibility to make many observations can be made over short periods of time also enable the documentation of short-term temporal and spatial (within or between colonies) variations in prey choice.

The main disadvantage is the possibility of misidentifying the prey with no possibility of later confirmation (unless the fish are photographed). This is even more of a problem for species carrying many small (even larval) fish, as numbers and sizes are very easily misjudged. As a result, it is often preferable to sample the fish directly by capturing the fish-carrying birds (see Rodway and Montevecchi, 1996).

In large colonies, fish-carrying common (and less often Brünnich’s) guillemots on their way to the nest site can be caught easily using a fleeg net, or with a noose pole once they have landed on or near the site (Birkhead and Nettleship, 1987; Montevecchi, pers. comm.). Fish-carrying puffins (and razorbills) can be caught in mist nets (Wanless et al., 2004) or again with fleeg nets as they arrive at or circle past the colony, or with a noose pole once they have landed. Because small fish, fry or larvae are easily lost in the undergrowth (or even over the cliff edge!), sampling sites should be chosen with care. Plastic sheeting under the mist net might reduce this problem of fish lost in the vegetation.

For burrow nesting species, a second method is to block the entrances of 20–30 burrows for 1–2 hours using a screen, e.g. of wire netting, placed a short distance inside the entrance (Sanger and Hatch, 1987; Finney et al., 2001), and to collect any dropped fish. One problem with this method is that the samples are sometimes damaged as the adult tries to get past the
successful trials have also been carried out in which the chick’s bill was sealed shut using a pipe-cleaner such that it can not pick up food dropped by the adult (Harding et al., 2002).

Both methods involve a limited disturbance of the adults, but the collection of fish has the great advantage of allowing accurate quantitative studies of prey composition (either by number, mass or energy content) in that the fish are whole and (often very) fresh when brought into the colony.

Either method (observation or collection) is, however, limited to the chick-rearing period which, for guillemots and razorbill colonies may last only 4–5 weeks between the hatching of the first egg and the fledging of the last chick. For terns and auks, however, the method may also be used to determine food choice and quality during the courtship period early in the breeding season, but how representative the fish fed to mates are of the general diet of the species is unknown.

6.8 Indirect biochemical assays

These indirect methods of determining seabird diets have several advantages over more traditional, direct methods. Direct diet sampling most often indicates what the individual seabird has just eaten and therefore may not reflect “average” or typical diet if temporal variability is high. In contrast, both stable isotope ratios and fatty acid signatures integrate diet information over space and time (ranging from days to months) (see Hobson et al., 1994). Biases associated with direct diet sampling can sometimes be large due to, for example, digestion of soft parts and preferential retention of hard parts (see above). These indirect methods do not suffer from the same problems.

6.8.1 Stable isotope analysis

This technique has been used extensively in avian feeding ecology studies over the past decade or more (e.g. Hobson and Welch, 1992; Hobson, 1993; Hobson et al., 1994; Sydeman et al., 1997). The method takes advantage of the fact that stable isotope ratios of nitrogen ($^{15}$N/$^{14}$N) and carbon ($^{13}$C/$^{12}$C) in tissues pass from prey to predator in a predictable manner. In the case of N, and to a lesser extent C, the ratio of the heavier (and rarer) isotope to the lighter (and more common) one increases at a rate of about 3–5 parts per thousand between each trophic level in marine systems. Therefore the method indicates trophic level of the predator, not the specific items in the diet (unless the diet is very simple). Although complex, stable isotope methodologies are now fairly routine and laboratories around the world offer this service at a reasonable cost.

Because the metabolic rates of various tissues differ, stable isotope ratios reflect trophic level at different time (and hence spatial) scales from days in the case of “fast” tissues (e.g. blood) to months in the case of “slow” ones (e.g. muscle, feathers) (Hobson et al., 1994; Bearhop et al., 1999). The method does, however, require voucher samples from hypothesised foraging areas. Stable isotope analysis of “slow” tissues provides the opportunity to assess diet during times of the year not normally covered with traditional diet sampling at seabird breeding colonies.

Although carbon isotope ratios change less between trophic levels than N, they are useful in providing a general idea of how far from shore the bird feeds. Carbon-13 is enriched in relation to $^{12}$C in nearshore compared to offshore waters.

6.8.2 Quantitative fatty acid signature analysis

A relatively new method to probe the diets of marine organisms takes advantage of the fact that 1) the fatty acid composition of prey species is diverse (between species) and
characteristic (within species), 2) long-chain (i.e. >14 units) fatty acids pass relatively undegraded to predators, and 3) the predator ultimately stores prey fatty acids in adipose tissue, which can be non-destructively sampled using biopsy (Iverson et al., 2004). As relatively few fatty acids are synthesised by the predators themselves, dietary versus intrinsic fatty acids can be distinguished. This technique has advantages over using stable isotopes because actual diet composition rather than just trophic level can be assessed.

A problem inherent in the technique is that predator diets usually contain more than one prey species such that the fatty acid signatures are often complex and cannot be examined just by eye. Furthermore, variability of fatty acid signatures between individuals of prey species, and intrinsic predator fatty acid production and metabolism sometimes need to be taken into account when interpreting the predator signatures. Iverson et al. (2004) outline a statistical modelling technique that was successful in estimating known diet composition of marine seals and mink. They suggest that the technique has wide application for other marine predators such as seabirds, and more recently, Iverson and Springer (in prep.) have confirmed the applicability of the technique to seabirds breeding in Alaska.

The technique, however, is demanding because a fatty acid database of all possible prey is needed to accurately interpret predator signatures. The database for seabird diets in the Atlantic will no doubt expand over the next few years (and is already doing so in Alaska, see Iverson and Springer in prep). The availability of software to perform the statistical modelling requirement of the method would aid its general applicability.

### 6.8.3 Serological methods

Serological methods also have the potential for detecting species-specific markers in digested prey items. The enzyme-linked immunosorbent assay (ELISA) has been used for identification of invertebrate tissue but it requires considerable laboratory effort to produce specific antisera to the range of potential prey species (Freeman and Smith, 1998). Trials to identify fish and molluscan prey of Jackass Penguins also noted problems with cross reactivity (Walter et al., 1986).

Pierce et al. (1990) tested the application of serological methods to the identification of fish prey in the diets of marine mammals. Antisera were raised to muscle protein extracts of three fish species. The antisera were tested for reactions with protein extracts from raw and in vitro digested fish muscle, stomach contents of captive bottlenose dolphins fed on known diets, digestive tract contents of grey and common seals and which contained hard remains of known prey species, and faeces of captive seals fed on known diets. The salmon antisera were shown to be sufficiently strong and specific to be used for identification of salmonid proteins in digestive tract contents of marine mammals, and were potentially applicable to screening seal faeces. Antisera raised for cod and herring were less successful, due to low specificity and low titre, respectively.

Due to the high number of prey species in most seabirds and the need for a reference database, serological methods presumably have a too high cost-effect ratio in most cases.

### 6.8.4 Gel electrophoresis and iso-electric focusing of proteins

Walter and O’Neill (1986) tested polyacrylamide gel electrophoresis to identify prey consumed by jackass penguins. They found that different prey species could be recognised up to 6 h after ingestion. Freeman and West (1998) used iso-electric focusing to identify fish tissue in Westland petrel diet samples. Forty-five percent of the samples from Westland petrel stomachs produced clear protein banding patterns and more than half of these were identified as species common in fisheries’ waste. Proteins in the other samples were presumably too digested for this technique. Despite Freeman and West (1998) claim that iso-electric focusing is a comparatively quick and inexpensive technique and is particularly useful for diet studies
where flesh eaten is likely to be relatively undigested at the time of sampling and despite the fact that the method is widely used in fisheries studies, the method was not established in seabird diet studies, neither was gel electrophoresis.

6.9 Food sampling under feeding birds

6.9.1 Fish/plankton hauls under seabird feeding frenzies.

When flocks of intensively feeding seabirds are encountered, sampling the sea for potential food items will provide direct information of the potential food species locally available. Such sampling can be done while conducting direct observations on the feeding birds, or sampling these birds in any other way. Food may be sampled by taking fish or plankton hauls at the spot, or acoustically. Fish hauls are often taken opportunistically, that is only when feeding frenzies of seabirds are encountered. It would be useful to also sample blanks, i.e. at similar locations away from the feeding frenzies. Both seabirds feeding on schools of fish and schools of plankton are eligible for this approach. Examples can be found in Grover and Olla (1983), Skov et al. (1989), Piatt (1990), Baars et al. (1990), Camphuysen (1999) and Frengen and Thingstad (2002).

6.9.2 Benthos sampling under flocks of seaduck

Flocks of seaducks that reside for a longer period at a certain location are likely to feed there, on benthic prey. Because benthic prey tends to stay the same place (possible exceptions being fish eggs, amphipods and other epi-benthos), such locations may be sampled with bottom grabs, dredges, nets or other devices to get an idea on available potential prey. In situations where one prey type is clearly dominantly present, and suitable as food, it may be inferred that this potential prey is also the actual prey taken by the ducks. Examples are given in Leopold et al. (1995), Kube (1996) and Degraer et al. (1999).

6.10 Application of data loggers

Data loggers have been applied successfully to study the timing of feeding and the amount of food ingested. Generally, all birds need to be caught first and the devices have to be deployed. Secondly, the birds need to be recaptured, so that data can be downloaded from the devices to a computer. So far, stomach temperature loggers have been most commonly applied. Their use is based on the principle that the ingestion of cold prey (fish, cephalopods etc.) by the warm-blooded seabirds leads to a drop in temperature (Wilson et al., 1992). From the magnitude of the temperature drop and the time it takes to re-warm the stomach and contents, the amount of food can be calculated (e.g. Wilson et al., 1995). This method has been successfully applied to a variety of seabirds including penguins, albatrosses, cormorants and gannets (e.g. Wilson et al., 1995; Grémillet and Plös, 1994; Garthe et al., 1999). It has however some limits. The major problem is that the detection works very well for single, large prey items but less well for multiple prey items and especially small items. In the worst case, lots of small fish such as sandeels or small clupeids cannot be detected at all after the stomach has partly filled so that both information on timing of feeding and amount of food might be masked (Wilson et al., 1995; Wanless et al., 2005). However, some studies were able to quantify prey consumption. In order to avoid the masking effect of prey lying in the stomach on top of the device, two other technological developments were developed that try to detect prey ingestion in the bird before the prey enters the stomach. Ancel et al. (1997), Charassin et al. (2000) and others applied sensors in the oesophagus that record prey ingestion while the prey moves from the beak of the birds towards the stomach. Wilson et al. (2002) have recently come up with a mandibular sensor that record changes in sensor voltage, proportional to magnetic field strength, and thus inter-mandibular angle. Captive feeding trials showed that prey mass could be determined with reasonable accuracy, and there was also some indication that prey type could be resolved if recording frequency were high enough (Wilson et al., 2002).
6.11 Differences in food between adults and chicks, breeders and non-breeders

When analysing and evaluating studies on seabird food, one has to be aware of possible bias that results from a non-representative sampling design. It is, for example, extremely difficult to sample diet of seabirds at sea. Hence it is not surprising that the vast majority of studies on seabird feeding ecology are restricted to the breeding sites and breeding times, i.e. only a small part of the whole year. Furthermore, even the comparatively few studies that have compared the diets of adults vs. chicks or of breeders vs. non-breeders have almost exclusively revealed substantial differences in diet.

Seabirds provisioning food to chicks face different constraints than when self-feeding and as a result, chick food normally differs from the food taken by adults (Ydenberg, 1994). Small chicks may physically be unable to ingest large prey (e.g. Shealer, 1998); parents flying with prey visible in their bills may be subject to kleptoparasitism (e.g. Veen, 1977; Furness, 1978; Burger and Gochfeld, 1991; Ratcliffe et al., 1997) or face aerodynamic or gravity constraints. Moreover, optimal prey for adults may be available only at distances from colonies that are too large for commuting (e.g. Weimerskirch, 1998). These constraints all lead to a shift away from prey optimal for chick rearing and hence accessible for diet studies in the breeding colonies. Optimal foraging theory or more precisely, central place foraging theory (Orians and Pearson, 1979) predicts that:

1) single prey loaders (such as guillemots or terns) should bring larger, and in energetical terms better, prey to their chicks than they swallow themselves (Wilson et al., 2004; Sonntag and Hüppop, 2005);
2) multiple prey loaders (such as many smaller auklets that hold several fish in their bill, Procellariiformes that convert prey to stomach oil, or seabirds that ferry multiple prey in their crop and/or stomach) should optimise their energy load per trip, particularly if trips are long, or few and far between (Ydenberg, 1994; Davoren and Burger, 1999).

Optimising energy load may be achieved by selecting fatty fish (such as clupeids, sandeels, capelin, mackerel) but also by selecting larger fish, as these generally contain more energy per item and per gram (Hislop et al., 1991; Lawson et al., 1998). Birds that need only to feed for themselves may satisfy their daily needs with small or lean prey, if these are easily available (low in feeding and transport costs), but parents that need to invest heavily in prey transport will benefit from being selective (Mehlum, 2001). Parents also need to sustain themselves and should thus attempt to optimally allocate their resources between themselves and their chicks. Optimal prey allocation may lead to letting the young starve if adult fitness is at risk in poor food years. Seabirds generally are long-lived birds that will rather desert their offspring when conditions turn bad, than putting their own survival and thus their further life time reproductive success at risk. Thus, they only invest in young (engage in provisioning) when resources are adequate. When young are being fed, the allocation of food between the parents and the chicks could, in theory, take the form of optimal sharing in single prey loaders (Leopold et al., 1996; Sonntag and Hüppop, 2005), i.e. a parent ingests all small prey, while flying off only with large prey, with the threshold being determined by their relative needs. Alternatively, parents could fulfill their own needs first, before switching to provisioning. Studies that simultaneously have looked at adult and chick diet are rare in seabirds (Brown and Ewins, 1996; Davoren and Burger, 1999; Dierschke and Hüppop, 2003).

Breeding birds without chicks could either be birds that still have eggs or birds that have lost their chicks. Studies generally showed that birds with chicks bring in food that is higher in energetic density than the food taken by birds without chicks (Keijl et al., 1986; Noordhuis and Spaans, 1992; Brown and Ewins, 1996). Mehlum (2001) showed that common guillemots and Brünnich’s guillemots that bring fish to their young can have much smaller prey, euphausiids, as their staple diet when self-feeding. Studies on the diets of seabirds in the non-
breeding season, i.e. away from the colonies and not connected to provisioning suggest that seabirds then take a larger variety of prey, including many species that are relatively low in energy density (e.g. Bradstreet and Brown, 1985).

Also the diet of adults may alter through the breeding period or may differ between sexes, reflecting changing demands e.g. for egg-production (Spaans, 1971; Pierotti and Annett, 1987, 1991; Pons, 1994), or even between individuals. For example, Niebuhr (1983) observed that female herring gulls in the pre-laying period preferred mussels, which provide calcium for egg-shell formation, whereas males fed on refuse. Despite the higher energetic value of refuse, mussel specialists produce more offspring, being larger at all developmental stages compared to refuse specialists (Pierotti and Annett, 1987). Things may additionally be complicated by individual feeding preferences (McCleery and Sibly, 1986) making large sample sizes necessary.

In seabird colonies, there are generally a high proportion of non-breeders present (e.g. Aebscher, 1986; Pons and Migot, 1995; Warham, 1996; Grunsky-Schönberg, 1998). These are birds that skip breeding for a year (or more), which might extenuate energetic constraints and hence increase lifetime reproduction and overall fitness (Calladine and Harris, 1997; Cam et al., 1998; Bradley et al., 2000). Again, due to deviating demands, their diet may be different from that of breeders.

Presumably the biggest issue in seabird diet studies is the food of birds at sea away from the colonies, i.e. outside the breeding season, in non-breeders or in immature birds. Most species of seabird spend the majority of their lives offshore, and most data on the diet of non-breeding seabirds is available from beached birds or from birds drowned in fishery nets. In general, feeding can be more opportunistic outside the breeding season since birds are not forced to stay in the vicinity of their breeding sites. Hence food composition is more varied outside the breeding season (e.g. Spaans, 1971; Halley et al., 1995; Ainley et al., 1996; Ouwehand et al., 2004; Ludynia et al., 2005).

### 6.12 Presentation of data

The large variety of data collection necessitates, besides standard methods of sampling, a unification of how to present the results. The main objectives of diet analyses generally are 1) to compare diet composition between species, times and sites and 2) to quantify the consumption rate of a predator on its prey on a species-level and, in fish-eating species, possibly also on a cohort-level. Hence the data have to be presented in a way to fulfil these aims and to allow inter-study comparisons. Duffy and Jackson (1986) gave a review on methods to analyse and present diet data, and this is still an excellent reference even twenty years after publication.

At present, this chapter is incomplete and needs further input at the WGSE meeting in 2007.

#### 6.12.1 Qualitative data

The minimum is a list of species found. For meta-analyses it is important to regard the taxonomic level to which species were identified, namely when comparing lists from different species or sites. For example, a category “unidentified polychaetes” may comprise only a single species, a few species or even some dozen species, which makes a big difference when comparing species numbers as an indicator of e.g. biodiversity. If different taxa were identified down to different levels (e.g. order, family, genus, species) comparisons have to take this into consideration. Also a possible bias caused by different stages of digestion for different taxa may cause severe deviations from reality. Ignoring unidentifiable diet components is likely to bias against more rapidly-digestible material (Duffy and Jackson, 1986). The same holds, of course, for quantitative analyses.
6.12.2 Quantitative data

Species lists should be extended to make at least some estimations of the abundance of the different taxa found. The easiest (and fastest) way is to note in how many “sample-units” the respective food-item occurred, i.e. in how many percent of all pellets, stomachs etc. which should be termed “frequency of occurrence”. However, this measure should only be applied when the prey items are of comparable size. Regarding the large differences in size of prey items in many seabird species (e.g. copepods vs. fish in fulmars, Furness and Todd, 1984, or amphipods and other small crustaceans vs. fish in Brünnich’s guillemot, Lønne and Gabrielsen, 1992) better measures to quantify food should be applied. Dietary data can be quantified e.g. in terms numbers of individuals per taxa (resulting in “numerical abundance”) or by biomass estimates per taxa if there are means to count individuals and to estimate volume or mass, respectively.

Several indices and methods to compare species or sites have been published. Day and Byrd (1989) developed an index of relative importance and Duffy and Jackson (1986) have listed a variety of diversity indices. Diet similarity (or overlap) among samples can be determined e.g. by using percent composition by mass and “Morisita’s Index of Diet Similarity”, which expressed similarity as a percent (Baltz and Morejohn, 1977). Other niche parameters that can be calculated if frequencies are available are niche breadth or niche overlap (Colwell and Futuyma, 1971; Mühlenberg, 1989). Also more sophisticated statistics such as cluster analysis or multidimensional scaling may be applied for categorizing diet data (Lønne and Gabrielsen, 1992; Kubetzki and Garthe, 2003). However, all these methods require that the original data are presented in a comparable manner (see above). Again it is extremely important to have the diet analysed down to same taxonomic level to achieve comparable data and to allow for e.g. differences in digestibility.

6.13 References


7 Further development of the EcoQO on plastic particles in stomachs of seabirds

OSPAR has asked ICES (formally at the HOD meeting in November 2005) to address the further development of the EcoQO on plastic particles in stomachs of seabirds.

In the light of the recommendations of the Save the North Sea (SNS) project, to provide advice on whether the formulation of the EcoQO on plastic particles in stomachs of seabirds would be more appropriately expressed in terms of mass of plastics rather than the number of pieces.

Currently, the EcoQO is currently formulated as follows:

“There should be less than 2% of northern fulmars (*Fulmarus glacialis*) having ten or more plastic particles in the stomach in samples of 50–100 beach-washed fulmars found in winter (November to April) from each of fifteen areas of the North Sea over a period of at least five years.”

The SNS project recommends that the formulation of the EcoQO should be revised to:

“There should be less than 10% of northern fulmars (*Fulmarus glacialis*) having more than 0.1g plastic particles in the stomach in samples of 50–100 beach-washed fulmars found in winter (November to April) from each of 4 to 5 areas of the North Sea over a period of at least five years.”

WGSE supports the work done by SNS and supports the suggestion by the SNS project that the new formulation would be an improvement on the current formulation. As a monitoring of this parameter has not yet been implemented in the North Sea countries outside the Netherlands on a regular scheme, the revised formulation would have no negative impact on monitoring activities. WGSE feels that the earlier 2% target should be maintained as the long-term target and that a date (year) should be connected to the currently proposed 10% target, e.g. 2015 or 2020.
8  WGSE plans for Cooperative Research Reports and other publications

We were given the term of reference to “determine potential for a state of the art report as an ICES Cooperative Research Report”. This term of reference led to a short general discussion about the main goals of further disseminating the results produced by this WG. WGSE found that previous Cooperative Research Report covered a wide range of issues that WGSE usually deals with. In so far many general aspects on seabird ecology are already well available to a wider audience. Furthermore, there are strong feelings by WGSE members that more attempts should be made to publish in prime scientific literature. Alternatively, web-based options such as mentioned in the 2006 Working Group on Marine Mammal Ecology (WGMME) report would be a possibility to provide information on seabirds. However, there does not seem to be any resources readily available by group members to compile and up-date such information. In conclusion, WGSE does not see the immediate need to produce another “ICES Cooperative Research Report” in the near future.
## Annex 1: List of participants

<table>
<thead>
<tr>
<th>Name</th>
<th>Address</th>
<th>Phone/Fax</th>
<th>Email</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tycho Anker-Nilssen</td>
<td>NINA, NO-7485 Teodheim, Norway</td>
<td>Tel +47 7380 1443 Fax +47 7380 1401</td>
<td><a href="mailto:tycho@nina.no">tycho@nina.no</a></td>
</tr>
<tr>
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</tr>
<tr>
<td>Rob Barrett</td>
<td>Tromsø University, Museum, Zoology Department, NO-9037 Tromsø, Norway</td>
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</tr>
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</tr>
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</tr>
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<td>Morten Frederiksen</td>
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</tr>
<tr>
<td>Stefan Garthe</td>
<td>FTZ, University of Kiel, Hafentöm 1, D-25761 Bösum, Germany</td>
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<td><a href="mailto:garthe@ftz-west.uni-kiel.de">garthe@ftz-west.uni-kiel.de</a></td>
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<tr>
<td>Ommo Hüppop</td>
<td>Institut für Vogelforschung ‘Vogelwarte Helgoland’, P.O. Box 1220, D-27494 Helgoland, Germany</td>
<td>Tel +49 4725 6402 11 Fax +49 4725 6402 29</td>
<td><a href="mailto:hueppop@vogelwarte-helgoland.de">hueppop@vogelwarte-helgoland.de</a></td>
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<tr>
<td>Mardik Leopold</td>
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<td>Tel +31 222 369744 Fax +31 222 319235</td>
<td><a href="mailto:Mardik.Leopold@wur.nl">Mardik.Leopold@wur.nl</a></td>
</tr>
<tr>
<td>Manuela Nunes</td>
<td>Nature Conservation Institute, Rua de Santa Marta 55, 1150-294 Lisboa, Portugal</td>
<td>Tel +351 21 3507900 Fax +351 21 3507984</td>
<td><a href="mailto:nunesm@icn.pt">nunesm@icn.pt</a></td>
</tr>
<tr>
<td>Ib Krag Petersen</td>
<td>National Environmental Research Institute, Department of Wildlife Ecology and Biodiversity, Grenaaevj 12, DK-8410 Roende, Denmark</td>
<td>Tel +45 89201518 Fax +45 89201514</td>
<td><a href="mailto:rkp@dmu.dk">rkp@dmu.dk</a></td>
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<tr>
<td>Iván Ramirez</td>
<td>SPEA, Rua de Vitória, 53 -3º Esq, P-1100-618 Lisboa, Portugal</td>
<td>Tel +351 21 322 0433 Fax +351 21 322 0439</td>
<td><a href="mailto:ivan.ramirez@spea.pt">ivan.ramirez@spea.pt</a></td>
</tr>
<tr>
<td>Norman Ratcliffe</td>
<td>RSPB, Conservation Science Department, East Scotland Regional Office, 10 Albyn Terrace, Aberdeen, AB10 1YP, United Kingdom</td>
<td>Tel +44 1224 627 852 Fax +44 1767 685071</td>
<td><a href="mailto:Norman.Ratcliffe@rspb.org.uk">Norman.Ratcliffe@rspb.org.uk</a></td>
</tr>
<tr>
<td>Jim Reid</td>
<td>Joint Nature Conservation Committee, Dunnet House, 7 Thistle Place, Aberdeen AB10 1UZ, United Kingdom</td>
<td>Tel +44 1224 655702 Fax +44 1224 621488</td>
<td><a href="mailto:jim.reid@jncc.gov.uk">jim.reid@jncc.gov.uk</a></td>
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<tr>
<td>Richard Veit</td>
<td>Biology Department, College of Staten Island, 2800 Victory Boulevard, Staten Island, NY 10314, USA</td>
<td>Tel +1 718 982 3853 Fax +1 718 982 3852</td>
<td><a href="mailto:veitr2003@yahoo.com">veitr2003@yahoo.com</a></td>
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Annex 2: WGSE Terms of Reference 2006

The Working Group on Seabird Ecology [WGSE] (Chair: S. Garthe, Germany) will meet in Barcelona, Spain from 19–23 March 2007 to:

a) finalize reviewing the current approaches for identifying offshore seabird aggregations and delineating Important Bird Areas (IBAs) and Special Protection Areas (SPAs);

b) continue developing recommendations for a comprehensive monitoring programme for seabirds;

c) finalise reviewing on how to sample diet and how to report results of dietary studies in seabirds, and develop recommendations for future field studies and analyses;

d) consider scientific ecological issues linked to the circulation of parasites and pathogens within seabird populations.

WGSE will report by 30 April 2007 to the attention of the Living Resources Committee.

Supporting Information

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<th>PRIORITY:</th>
<th>This is the only forum for work being carried out by ICES in relation to marine birds. If ICES wishes to maintain its profile in this area of work, then the activities of WGSE must be regarded as of high priority.</th>
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</tr>
<tr>
<td>Term of Reference b)</td>
<td>This comprehensive ToR could not be finalised at the 2006 WGSE meeting.</td>
</tr>
<tr>
<td>Term of Reference c)</td>
<td>This comprehensive ToR could not be finalised at the 2006 WGSE meeting.</td>
</tr>
<tr>
<td>Term of Reference d)</td>
<td>This issue is of relevance because of the current general interest in the role of wild birds in the epizootiology of some disease, and because seabirds represent interesting biological models for scientific investigations on the ecology of host-parasite interactions.</td>
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<td>RESOURCE REQUIREMENTS:</td>
<td>Facilities for WGSE to work in Barcelona are anticipated to be excellent.</td>
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<td>PARTICIPANTS:</td>
<td>The Group is normally attended by some 10–15 members and guests. The Working Group should be able to achieve most of the above objectives. However, some members may not be able to attend through lack of funding. Funding of these members from Member Countries would be very welcome.</td>
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<td>FINANCIAL:</td>
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<td>LINKAGES TO OTHER COMMITTEES OR GROUPS:</td>
<td>WGSE is keen to continue the process of integration of seabird ecology into ICES work.</td>
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<td>LINKAGES TO OTHER ORGANIZATIONS:</td>
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<td>SECRETARIAT MARGINAL COST SHARE:</td>
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Annex 3: Recommendations

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<td>1. WGSE strongly recommends that, in addition to breeding numbers, reproductive success and diet of seabirds are monitored annually at representative sites in all countries and that results are reported promptly, allowing determination of the extent and causes of future widespread breeding failures. This recommendation also applies to monitoring programmes in Canada, the United States and other parts of the world.</td>
<td>ICES</td>
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<td>EU</td>
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**Annex 4: English and scientific names of birds mentioned in this report**

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<td>Red-throated diver</td>
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<td>Black-throated diver</td>
<td>Gavia arctica</td>
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<td>Slavonian grebe</td>
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<td>Great crested grebe</td>
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<td>Red-necked grebe</td>
<td>Podiceps grisegegena</td>
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<td>Northern fulmar</td>
<td>Fulmarus glacialis</td>
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<td>Cory’s shearwater</td>
<td>Calonectris diomedea</td>
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<td>Westland petrel</td>
<td>Procellaria westlandica</td>
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<td>Little shearwater</td>
<td>Puffinus assimilis</td>
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<td>Northern gannet</td>
<td>Morus bassanus</td>
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<td>Great cormorant</td>
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<td>European shag</td>
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<td>Common eider</td>
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<td>Melanitta nigra</td>
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<tr>
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<td>Melanitta fusca</td>
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<td>Red-necked phalarope</td>
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<td>Stercorarius skua</td>
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<td>Little gull</td>
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<td>Audouin’s gull</td>
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<td>Herring gull</td>
<td>Larus argentatus</td>
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<td>Ivory gull</td>
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<td>Rissa tridactyla</td>
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<td>Common guillemot</td>
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<td>Razorbill</td>
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<td>Black guillemot</td>
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<td>Brünnich’s guillemot</td>
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<tr>
<td>Atlantic puffin</td>
<td>Fratercula arctica</td>
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Annex 5: Questionnaire

Questionnaire from the ICES Working Group for Seabird Ecology (ICES WGSE)

The ICES Working Group on Seabird Ecology [WGSE] met at Texel (The Netherlands) from 3–7 April 2006 to review the breeding success of seabirds in the North Sea in 2004 and 2005 and to explore the reasons for poor performance. Secondly, the WGSE was to produce recommendations for a comprehensive monitoring programme for seabirds.

Two important conclusions were drawn from the exercise: (1) it is difficult to obtain updated, high quality data on reproductive success of seabirds within the North Sea as well as beyond and, (2) long-term monitoring programmes were seemingly sparse and often on the brink of discontinuation. We are not confident that all available material was in fact traced and it was therefore considered useful to try to provide an overview of recent and current monitoring activities regarding seabird populations and breeding success within Europe. Our scope was soon extended to include the entire North Atlantic region, including the New World (USA, Canada, and Greenland).

In preparation for the 2007 meeting, the WGSE decided to send out a questionnaire to obtain an overview of current monitoring schemes in that large area. In a later phase, ICES/WGSE will contact the coordinators of existing schemes to provide an overview of the current status, population trends and demographics of seabirds within the entire North Atlantic region. Also, we may use the information to formulate concrete (methodological) recommendations, and to stimulate national or regional authorities to (re-)consider monitoring schemes to be established to fill in major gaps.

If this questionnaire is sent to the wrong address, or if you consider a colleague/other institute more/also appropriate to answer the following questions, please forward a copy of this document.

The first issue is breeding population size assessments

1. Which organisations are responsible for breeding seabird population censuses in your area/country?
2. Are population censuses conducted on a regular basis (indicate frequency)?
3. Are all breeding species included (indicate which are, which are not)?
4. What is the motivation for these censuses?
5. How and where are the results published or otherwise made accessible?
6. Are the ‘raw data’ (actual counts on colony level) accessible to third parties, and how could these be accessed (including under what conditions)?

The second issue is other demographic parameters such as reproductive success

7. Which organisations are involved in assessing
   a. Reproductive success?
   b. Annual survival rates?
c. Diet?
d. Other demographic parameters (please give details)?

(8) Are surveys conducted on a regular basis (indicate frequency)?
(9) Are all breeding species included (indicate which are, which are not)?
(10) What is the motivation for these surveys?
(11) How and where are the results published or otherwise made accessible?
(12) Are the ‘raw data’ (actual counts on colony level) accessible to third parties, and how could these be accessed (including under what conditions)?

The third issue is **wintering population assessments**, because breeding populations cannot always be monitored adequately (e.g. divers, grebes, seaduck)

(13) Which organisation is responsible for wintering seabird population surveys in your area/country?
(14) Are surveys conducted on a regular basis (indicate frequency)?
(15) Are all species included (indicate which are, which are not)?
(16) What is the motivation for these surveys?
(17) How and where are the results published or otherwise made accessible?
(18) Are the ‘raw data’ accessible to third parties, and how could these be accessed (including under what conditions)?

The fourth issue concerns **other parameters or elements that are monitored and that can possibly be of broad geographic relevance** (contaminant levels, parasites and pathogens, samples for genetic and isotopic analyses)

(19) What type of other parameter/element of is monitored and which organisation and/or people are responsible for the monitoring in your area/country?
(20) Are surveys conducted on a regular basis (indicate frequency)?
(21) Are all breeding species included (indicate which are, which are not)?
(22) What is the motivation for these surveys?
(23) How and where are the results published or otherwise made accessible?
(24) Are the ‘raw data’ or samples accessible to third parties, and how could these be accessed (including under what conditions)?

The last issue is: who should be contacted for each of these four issues / all species? Please fill in the first line (three boxes) for each group with contact details, fill the second line with comments and details regarding the monitoring programme(s).

*Indicate for which area your answer is meant (county, province, country, archipelago or whatever) and if all seabird species are considered in the reply.*
### Issue (1) Breeding Population Size

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### Issue (2) Demographic Parameters

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