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## **Seabirds as Monitors of the Marine Environment**

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# 1 Introduction

*Mark L. Tasker and Robert W. Furness*

This is the third *ICES Cooperative Research Report* produced by the Working Group on Seabird Ecology, following on from Reports on seabird/fish interactions (*ICES Cooperative Research Report* No. 216) and the diets of seabirds and the consequences of changes in food supply (*ICES Cooperative Research Report* No. 232). This *ICES Cooperative Research Report* focuses on the use that might be made of seabirds as monitors of the marine environment.

Section 2 examines the possibilities of using seabirds to monitor marine pollution, and recommends that they be used in monitoring a variety of substances. These recommendations are further developed in Section 5. Subsequent to this work, several of these recommendations have been developed for possible use as Ecological Quality Objectives (EcoQOs) under OSPAR and the North Sea Conference process. Ministers from around the North Sea adopted an objective in relation to the proportion of oiled common guillemots found dead or dying on beaches that “the proportion of such birds should be 10% or less of the total found dead or dying, in all areas of the North Sea (Bergen Declaration). In addition, Ministers requested that work continue towards defining EcoQOs in relation to mercury concentrations in seabird eggs and feathers, organochlorine concentrations in seabird eggs and plastic particles in the stomachs of seabirds. These decisions demonstrate the usefulness of seabirds in this area.

Section 3 describes the sensitivity of seabird populations to changes in life history parameters. Understanding in this area is necessary in order to fully appreciate the nature of threats that apply at only one part of seabird’s life style and is important in designing monitoring programmes. This is further discussed in Section 5.3.

Fishing is arguably the greatest human influence on the marine environment. Fishing can have both direct and indirect effects on seabird populations (Tasker *et al.*, 2000). The cumulative effects of these effects will show at the seabird community level and are examined in section 4.

The design of monitoring programmes is further explored in Section 5, while Section 6 describes the relatively new topic of the interaction between aquaculture and birds in the ICES area.

As with all work within the ICES Working Group structure, many individuals contribute to the group’s products. All those who participated in the relevant meetings of the Working Group on Seabird Ecology are listed overleaf.

## 1.1 References

- Bergen Declaration. Ministerial Declaration of the Fifth International Conference on the Protection of the North Sea, 20–21 March 2002, Bergen, Norway. Ministry of the Environment, Oslo. 50 pp.
- Tasker, M. L., Camphuysen, C. J., Cooper, J., Garthe, S., Montevecchi, W. A. and Blaber, S. J. M. 2000. The impacts of fishing on marine birds. *ICES Journal of Marine Science*, 57: 531–547.



## 1.2 Participants

The following nationally appointed members of the Working Group on Seabird Ecology participated in the meetings from which this Report is derived. As is usual with Working Group reports, all will have commented in order to help improve aspects of the papers in this Report.

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## 2 Seabirds as monitors of marine pollution

Peter H. Becker, Robert W. Furness, and Mark L. Tasker

### 2.1 Introduction

The use of seabirds as monitors of marine pollution has been advocated many times (Chapdelaine *et al.*, 1987; Furness, 1987, 1993; Gilbertson *et al.*, 1987; Becker, 1989, 1991; Walsh, 1990; Furness *et al.*, 1995; Monteiro and Furness, 1995; Barrett *et al.*, 1996; Elliott *et al.*, 1996; Becker *et al.*, 1998). In this review, the usefulness of seabirds as a means of monitoring contaminants in marine ecosystems is considered. Particular situations where the monitoring of contaminants in seabirds is highly desirable as a cost-effective and informative procedure are described.

Seabird eggs are useful in contaminant monitoring because eggs offer a stable, well-defined matrix with a high fat content, where many contaminants, especially organic substances, accumulate (ICES, 1995). Thus, eggs can be used for detecting spatial as well as temporal variations. Also, as birds are top predators, with a high rate of bioaccumulation, the monitoring of contaminants in birds can be combined with studies of biological effects on populations. Potential choices of seabird species in the Northeast Atlantic and Baltic Sea are given in ICES (1995). Some examples of estimates of variance components in the analysis of several contaminants in five species of seabirds are given in ICES (1999).

### 2.2 Seabirds as biomonitors of marine pollution

#### 2.2.1 Oil pollution

The use of seabirds as monitors of oil pollution at sea has been reviewed in a number of recent publications (Camphuysen and van Franeker, 1992; Dahlmann *et al.*, 1994; Camphuysen, 1995, 1998; Wiens *et al.*, 1996; Furness and Camphuysen, 1997). Beached bird surveys carried out predominantly by amateurs with organisation and data interpretation by professional staff (usually from non-governmental organisations) provide clear evidence of long-term trends in oiling rates of seabirds (Figure 2.1) and differences in oil impacts among regions (Figure 2.2). There is evidence to show that oiling indices based on the proportion of beached seabirds with oil give a reasonable measure of the number of oil slicks at sea, although factors such as wind direction and numbers of seabirds dying from starvation or disease can confound the picture (Stowe, 1982). Recent developments in fingerprinting oil from carcasses permit identification of the source of oil on birds and can be used in prosecutions for the discharge of oil at sea (Dahlmann *et al.*, 1994). Toxic effects of oil ingested by seabirds have been reviewed several times (e.g. Briggs *et al.*, 1996), but are unlikely to provide a useful monitor of

oil impact on seabirds. Seabirds as monitors of oil pollution are not considered further in this review, but readers interested in this topic are referred to the papers listed above.

#### 2.2.2 Plastic particle pollution

The use of seabirds as monitors of plastic particle pollution on the ocean surface has been suggested by a number of authors (Furness, 1985, 1993; Ryan, 1987; Spear *et al.*, 1995; Blight and Burger, 1997) because some seabirds, especially petrels, accumulate large numbers of plastic particles in their gizzard, and may suffer harmful effects in their ability to process food (Ryan, 1988). Sampling seabirds to measure quantities of ingested plastic requires obtaining dead birds or killing birds since the stomach contents must be obtained. Procedures that offload the proventriculus contents by 'stomach-pumping' or 'wet-offloading' do not extract material from the gizzard, which is where the vast majority of plastic is stored. Thus, sampling proventriculus contents from live seabirds provides very little data on plastic ingestion and is unlikely to be an effective means of monitoring plastic ingestion. Killing seabirds as a means of monitoring pollution is undesirable. However, there are two possible sources of seabird gizzards that do not require the killing of healthy birds. It may be possible to monitor plastic ingestion by sampling from petrels obtained from beached bird surveys (e.g., northern fulmars on the European beaches). However, this runs the risk that the birds washed up on beaches do not represent the population as a whole. Beached birds are likely to be predominantly juveniles rather than adults, and birds that have died slowly may have ingested plastic more or less than birds that are healthy and feeding well. Such biases may be difficult to quantify. Another possible source of gizzards from petrels may be from colonies where birds are killed in numbers by predators or accidents. At some colonies, petrels form the main prey of skuas and gizzards may be obtained from some of the birds killed by skuas before they have been eaten. At other colonies, birds may be killed by cats or rats or due to specific hazards. For example, shearwater fledglings at some colonies die when attracted to lights; gizzards at some colonies become tangled in vegetation. Sampling on a regular basis over years at such sites might provide indications of long-term trends in plastic particle pollution. There is evidence that plastic particle burdens in pelagic seabirds are increasing and that this is a problem that needs further attention but this topic will also not be considered further here, since it is assumed that the main focus of interest is in contamination by chemicals.

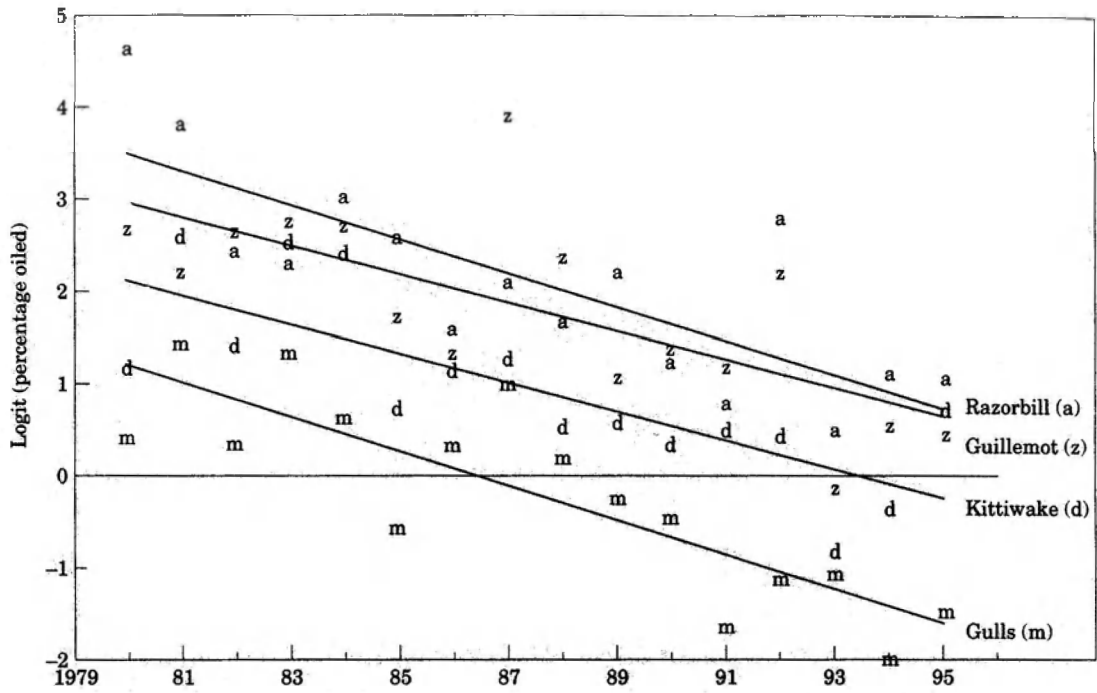


Figure 2.1. Trends in oil rates of razorbills, common guillemots, black-legged kittiwakes, and *Larus* gulls stranded at the mainland coast in the Netherlands, 1979–1995. Data from Camphuysen (1995).

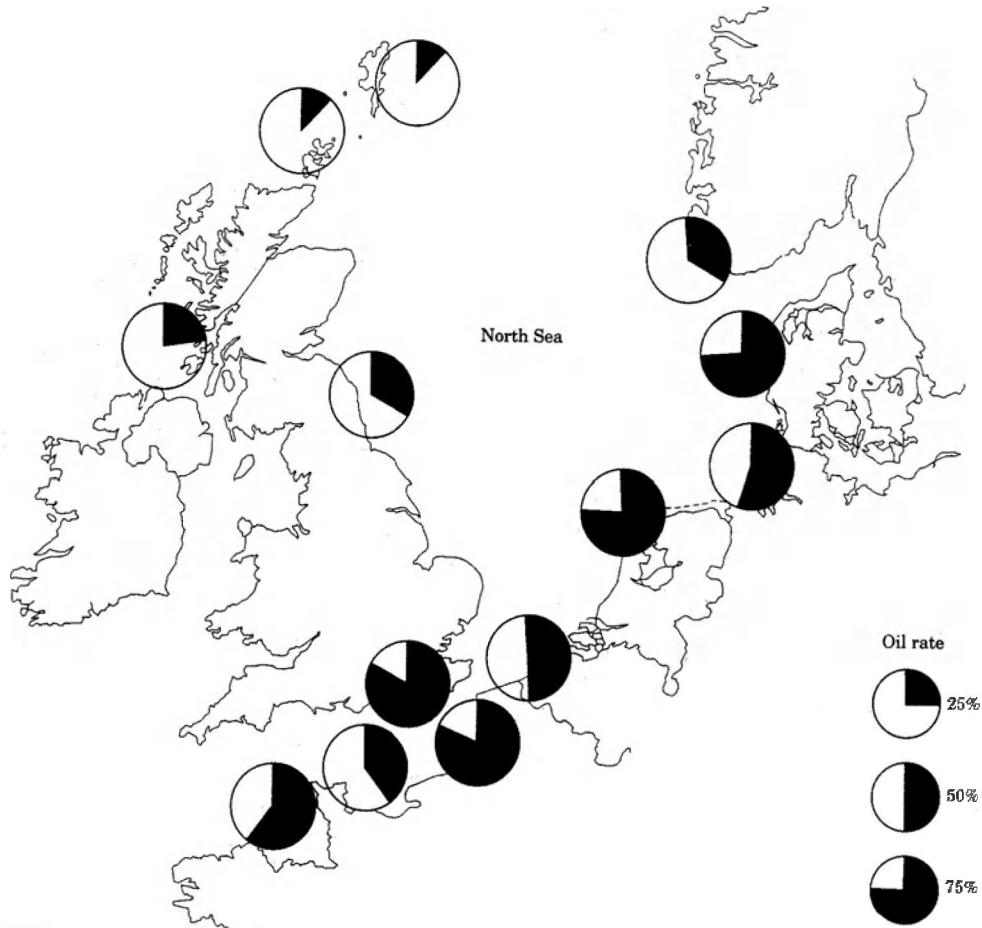


Figure 2.2. Oil rates of common guillemots at various locations in western Europe. Data from Camphuysen (1995).

### 2.2.3 Organochlorines

Organochlorine concentrations have been measured in the physical environment (e.g., Bignert *et al.*, 1998), in marine invertebrates (e.g., Johansen *et al.*, 1996; Mattig *et al.*, 1997), in fish (e.g., von Westernhagen, 1994; Jones and Franklin, 1997; Kennish and Ruppel, 1998; Mattig *et al.*, 1997), and in various populations of marine mammals (e.g., Addison *et al.*, 1984; Jarman *et al.*, 1997) and seabirds (e.g., Mehlum and Daelemans, 1995; Savinova *et al.*, 1995; Focardi *et al.*, 1996; Jones *et al.*,

1996; Joiris *et al.*, 1997; van den Brink, 1997; van den Brink *et al.* 1998).

Being lipid-soluble, organochlorines tend to accumulate in the lipid-rich tissues of animals, and biomagnify through the food chain, so that animals high in marine food chains tend to carry the largest body burdens and have the highest tissue concentrations (Figure 2.3, Tables 2.1 and 2.2). The variation of organochlorines within seabird samples, however, is similar or even lower compared with that of their food (Table 2.2).

Table 2.1. Mean concentrations ( $\mu\text{g g}^{-1}$  wet weight) and coefficients of variation for DDE and PCBs in marine organisms.

Species	Site	Age	Tissue	n	PCBs		DDE		Ref.
					mean	CV	mean	CV	
Herring	Nova Scotia	4 years	muscle	29	0.25	108%	0.06	490%	1
Herring	Gulf of St. Lawrence	–	muscle	26	0.44	98%	0.09	506%	1
Herring	Wadden Sea	2–3 years	whole	7	0.03	25%	0.005 <sup>1)</sup>	29%	2
Sandeel	Wadden Sea	2–3 years	whole	8	0.04	40%	0.002 <sup>1)</sup>	83%	2
Plaice	Wadden Sea	2 years	whole	7	0.02	30%	0.002 <sup>1)</sup>	28%	2
Flounder	Norway	–	liver	10	0.03	63%	0.06	116%	1
Cod	Norway	–	liver	18	0.45	57%	0.70	79%	1
Cod	Nova Scotia	5 years	liver	38	1.81	49%	0.28	50%	1
Cod	Nova Scotia	2–9 years	liver	100	1.71	53%	0.28	54%	1
Grey seal	Nova Scotia	immature	blubber	8	5.00	70%	1.50	40%	1
Grey seal	Nova Scotia	adult	blubber	8	15.7	37%	2.50	32%	1
Leach's storm petrel	Newfoundland, 1984	eggs	egg	5	1.16	24%	0.40	28%	1
Leach's storm petrel	Bay of Fundy, 1984	eggs	egg	5	3.44	36%	1.05	39%	1
Atlantic puffin	Newfoundland, 1984	eggs	egg	5	0.99	12%	0.30	19%	1
Atlantic puffin	Bay of Fundy, 1984	eggs	egg	5	3.20	20%	0.74	24%	1
Common tern	Wadden Sea	eggs	egg	10	4.65	33%	0.39	60%	3
Herring gull	Wadden Sea	eggs	egg	10	1.45	40%	0.39	60%	3
Eurasian oystercatcher	Wadden Sea	eggs	egg	10	2.66	48%	0.11	46%	3

<sup>1)</sup> EDDT; Ref. = Reference; 1 = Gilbertson *et al.* (1987); 2 = Mattig *et al.* (1996) (year 1992); 3 = Becker *et al.* (1991) (year 1987: Mellum, Minsener, and Oldeog islands).

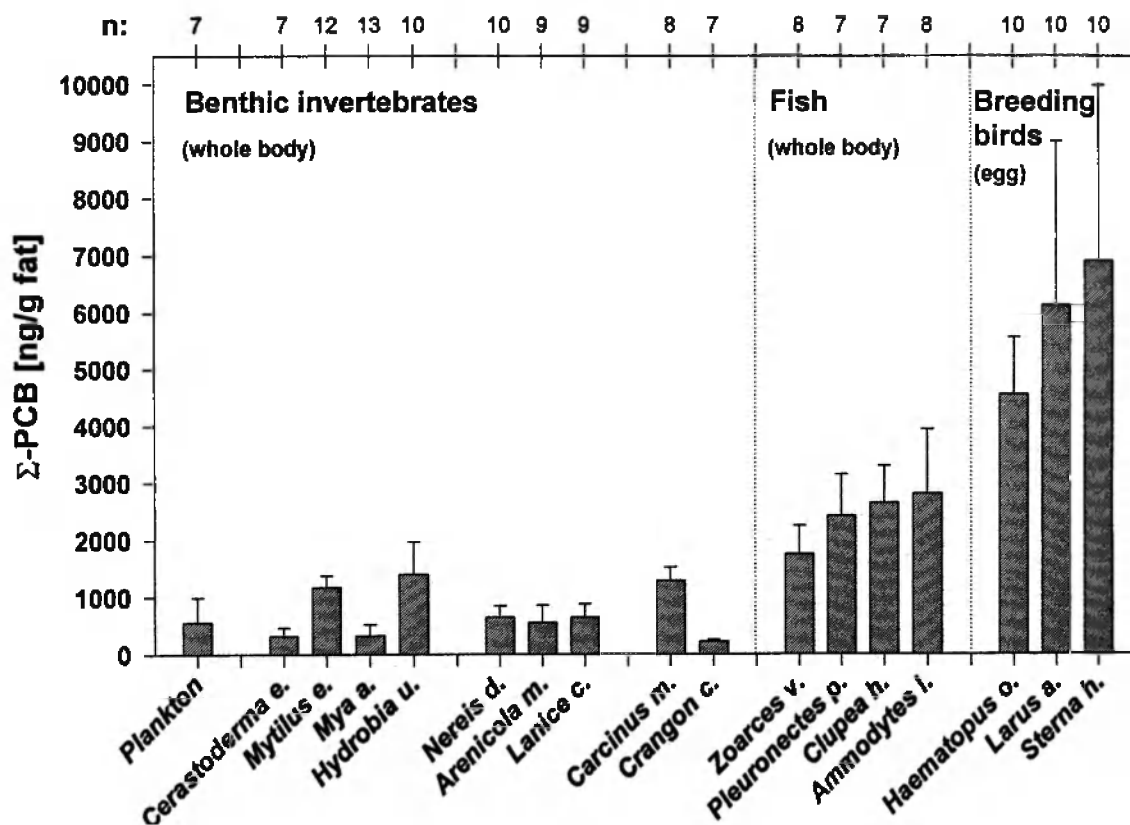


Figure 2.3. Biomagnification of PCBs in the food web of the Wadden Sea. Sums of the concentrations of eight PCB congeners on a fat weight basis are presented for plankton, nine benthic invertebrate species, four fish (juvenile stages) and three seabird species. Means + 1 standard deviation (Mattig *et al.*, 1996).

Table 2.2. Biomagnification factors between organochlorines in food and eggs of Eurasian oystercatcher, herring gull and common tern from Spiekeroog, German Wadden Sea in 1993 (Mattig *et al.*, 1996). See Figure 2.3 for PCB concentrations.

Biomagnification factors				
Species	Food	PCBs	DDTs	HCB
Eurasian oystercatcher	Benthic animals <sup>1)</sup>	4–14	3–23	12–21
Herring gull	Benthic animals <sup>1)</sup>	5–19	6–46	17–30
	Fish <sup>2)</sup>	2–3	2	2–3
Common tern	Fish <sup>2)</sup>	3	3	3–5

<sup>1)</sup> *Cardium, Mytilus*; <sup>2)</sup> Herring, plaice.

PCBs are industrial chemicals and consist of 209 congeners. The degree of metabolisation of the PCB mixture in seabirds depends on the species, on the level of PCB contamination in the environment, as well as on the length of the time during the year the birds spend in that environment (Beyerbach *et al.*, 1993; Denker *et al.*, 1994).

Different organochlorines, and even different congeners of PCBs, differ considerably in toxicity (Niimi, 1996) and toxic effects vary considerably between different

groups of animals. As a broad generalisation, seabirds tend to be less sensitive to organochlorine toxicity than marine mammals or terrestrial birds (Beyer *et al.*, 1996). Almost all sampling of adult birds and mammals for monitoring organochlorine concentrations has used samples of liver or muscle tissues, so that animals had to be killed or samples taken opportunistically from drowned or wrecked birds or mammals. These latter samples may have introduced bias due to starvation and consequent mobilisation of lipid reserves and, thus, high concentrations of organochlorines. High tissue (especially liver) concentrations of organochlorines can indicate tissue wastage rather than large intakes of the contaminants.

Seabird eggs have been sampled to provide more reliable monitoring of organochlorines, and this permits an assessment of geographical patterns of organochlorine contamination (Becker *et al.*, 1998) (Figure 2.4) as well as long-term trend analysis (Chapdelaine *et al.*, 1987; Elliott *et al.*, 1988, 1996; Bignert *et al.*, 1995; Becker *et al.*, 1998) (Figures 2.5–2.7). The change in load of organochlorines in the aquatic environment, e.g. of PCBs in the Elbe River, is reflected immediately in the seabird egg concentrations (Figure 2.6). While levels of many organochlorines such as DDE, dieldrin, HCB, and PCBs have been shown to have decreased since the 1970s (e.g., Figures 2.5–2.7), levels of some other

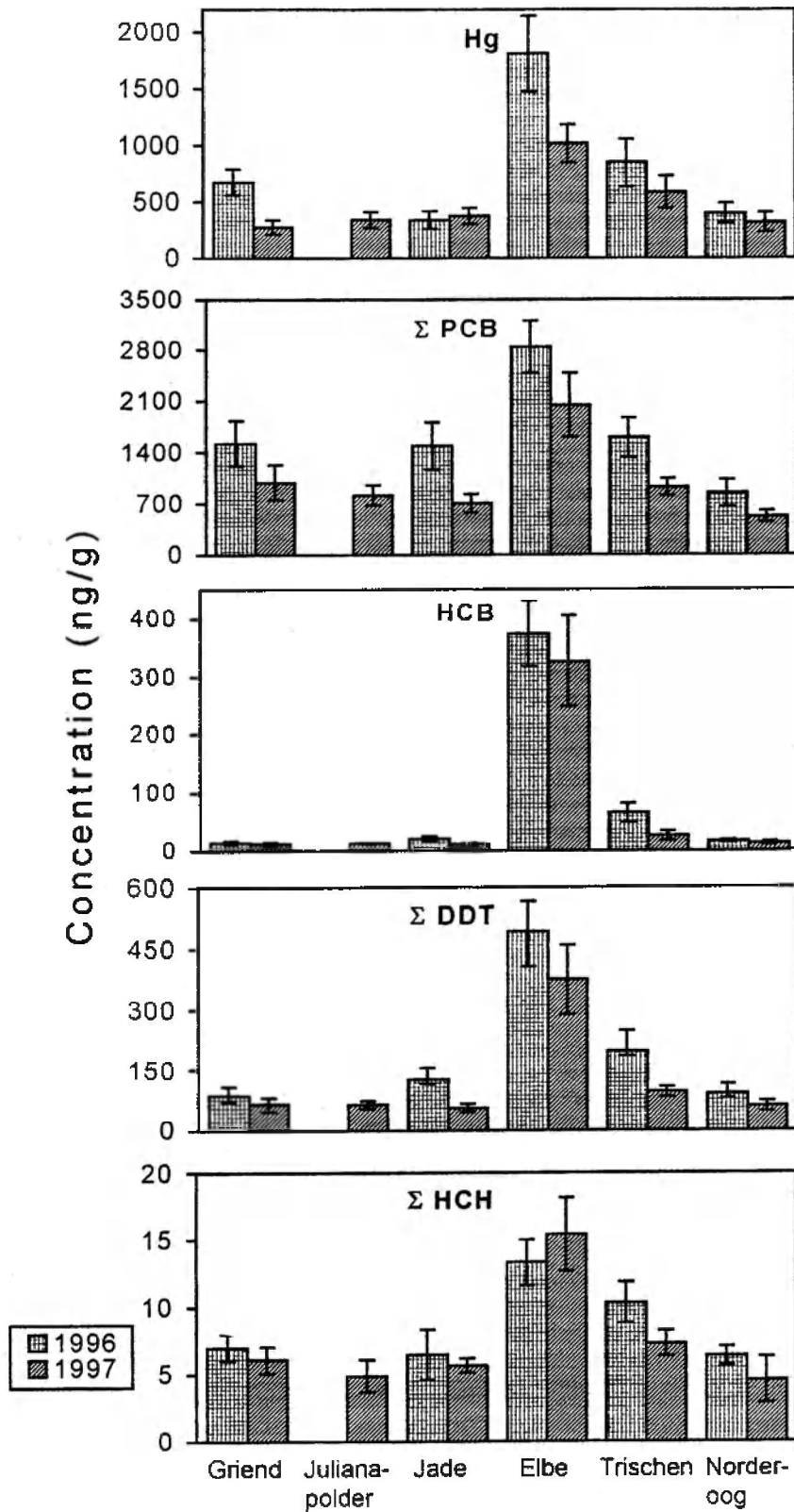


Figure 2.4. Geographical patterns of mercury and organochlorine contamination of common tern eggs in the Wadden Sea in 1996 and 1997 (six sites; from Becker *et al.*, 1998). Mean values  $\pm$  95% coefficients of variation,  $n = 10$  eggs each.

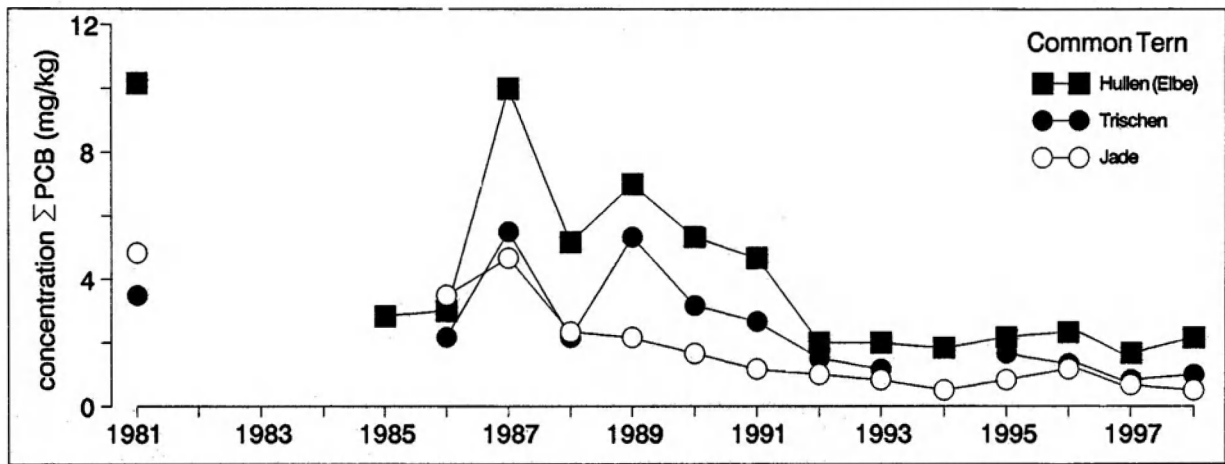


Figure 2.5. Temporal trends of PCB concentrations in eggs of common terns from three selected breeding sites in the Wadden Sea from 1981–1997 (Becker *et al.*, 1998).

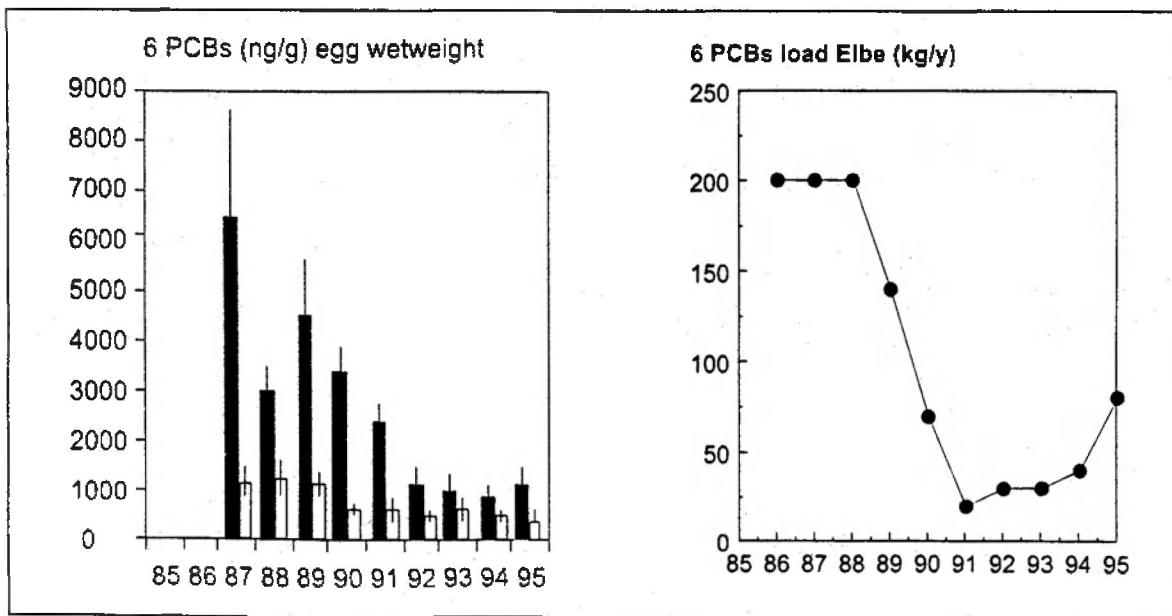


Figure 2.6. Temporal trend in PCB loads of the Elbe River compared to the trends in PCB concentrations in eggs of common terns (black columns) and Eurasian oystercatchers (white columns) breeding at the Elbe estuary (Bakker *et al.*, 1997).

Contaminant levels in eggs, measured in parts per million (wet weight)

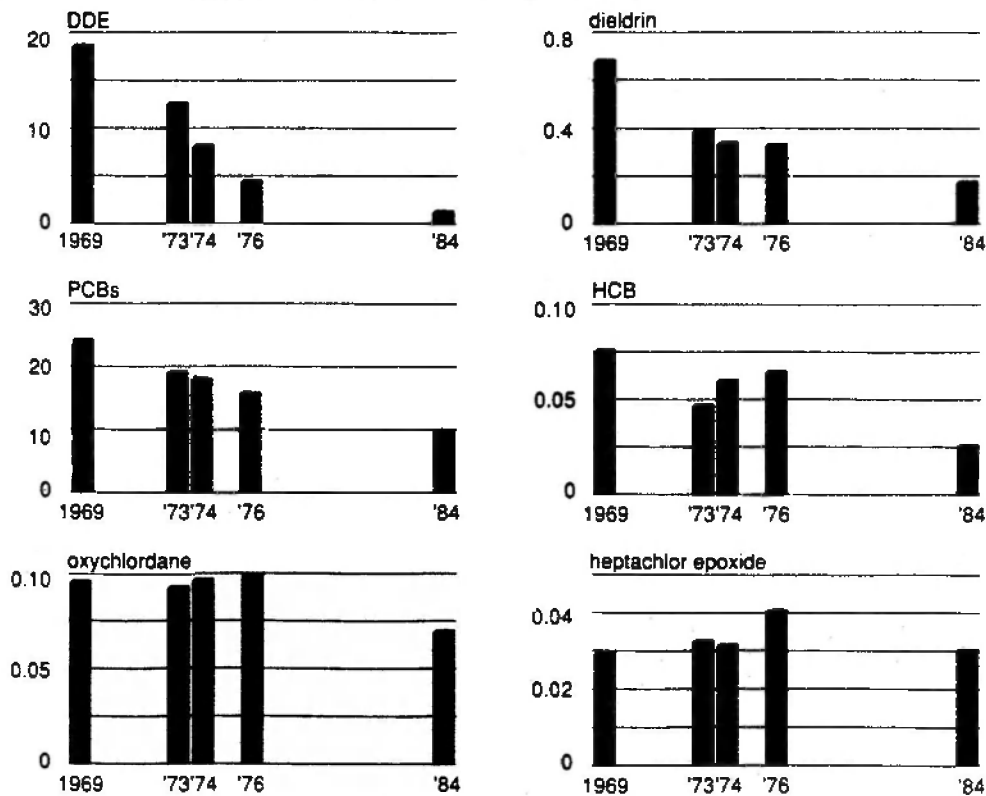


Figure 2.7. Temporal trends in organochlorine levels in northern gannet eggs from Quebec, Canada, from 1969–1984 (Chapdelaine *et al.*, 1987; Elliott *et al.*, 1988).

organochlorines have shown no clear trend (Figure 2.7) and a few have increased (e.g., Becker *et al.*, 1998). Analyses of organochlorines in seabirds resident at high latitudes (Arctic and Antarctic) have provided evidence of the global transport of these contaminants, although concentrations in resident seabird species tend to be low, and less than in seabirds that breed in these regions but migrate to lower latitudes in winter (Lønne *et al.*, 1997; van den Brink, 1998).

While eggs tend to be sampled at a very clearly defined and consistent time of the year, avoiding problems of interseasonal variation, long-term trends in organochlorine concentrations in tissues of adult seabirds can be obscured where there are pronounced seasonal variations, and sampling across years is not limited to a short period (Joiris *et al.*, 1997).

It is worth emphasizing that the organochlorine concentrations in seabird eggs reflect local contamination in the vicinity of each breeding colony, even in seabird species that are transequatorial migrants such as the common tern (see Figure 2.4). After arrival in the breeding area, terns and other seabirds need large amounts of supplementary food presented by the males during the short prelaying period, enabling females to raise weight and produce eggs (common tern: up to 50% weight increase during ten days, Wendeln and Becker,

1996). Thus, eggs provide a measure of contamination on a scale set by the foraging range of birds from their breeding colony. Exposure to these contaminants in the wintering area apparently has little or no influence on the amounts put into the egg. This may not be true for all seabird species, but it does appear to be a general pattern.

#### 2.2.4 Mercury

Mercury is the heavy metal most likely to present a toxic hazard in marine foods. Particularly high concentrations occur in long-lived predatory and deep-sea fish. It is readily converted by bacteria from inorganic forms into methylmercury in low-oxygen environments (deep water or in anoxic sediments). Methylmercury is not only much more toxic to vertebrates, but is also lipid-soluble so it tends to biomagnify through food chains and is accumulated in lipid-rich tissues of vertebrates in a similar way to organochlorines (Tables 2.3–2.5). As in the case of organochlorines, the within-seabird-sample variation is in the same order as the variation in fish, the main prey (Tables 2.3 and 2.4). Furthermore, while the assimilation efficiency of inorganic mercury from digested food is very low, the assimilation efficiency of methylmercury is around 95%. Thus, most of the mercury taken into the tissues of fish, marine mammals, and seabirds is methylmercury. However, at least some



Table 2.3. Mean concentrations ( $\mu\text{g g}^{-1}$  wet weight) and coefficients of variation for mercury in marine organisms in Shetland. Median values are given for each group.

Group	Species	Tissue	n	Mean ( $\mu\text{g g}^{-1}$ wet weight)	CV	Ref.	
Fish	Sandeel	whole	18	0.04	25%	1	
	Cod	whole	79	0.05	40%	1	
	Whiting	whole	20	0.07	29%	1	
	Plaice	whole	25	0.03	33%	1	
			<b>Medians</b>		<b>0.045</b>	<b>31%</b>	
Seabirds	Common guillemot	chick down	29	1.24	22%	2	
	Black-legged kittiwake	chick down	12	1.43	26%	2	
	Arctic tern	chick down	24	2.03	32%	2	
	Arctic skua	chick down	36	2.00	45%	2	
	Great skua	chick down	58	4.15	34%	2	
			<b>Medians</b>		<b>2.0</b>	<b>32%</b>	
		Black-legged kittiwake	chick feathers	26	0.37	32%	2
		Arctic tern	chick feathers	15	0.69	20%	2
		Arctic skua	chick feathers	30	0.46	47%	2
		Great skua	chick feathers	28	1.22	31%	2
			<b>Medians</b>		<b>0.55</b>	<b>32%</b>	
		Northern fulmar	adult body feathers	12	1.1	27%	1
		Black-legged kittiwake	adult body feathers	14	2.4	25%	1
		Black-legged kittiwake	adult body feathers	21	3.31	36%	2
		Razorbill	adult body feathers	16	2.1	14%	1
		Common guillemot	adult body feathers	17	1.5	27%	1
		Common guillemot	adult body feathers	34	0.99	34%	2
		Atlantic puffin	adult body feathers	10	5.2	52%	1
		Great skua	adult body feathers	197	7.0	73%	1
		Great skua	adult body feathers	54	6.34	41%	2
		Arctic tern	adult body feathers	23	0.86	27%	2
		Arctic skua	adult body feathers	28	2.52	88%	2
			<b>Medians</b>		<b>2.4</b>	<b>34%</b>	

Ref. = Reference: 1 = Thompson *et al.* (1992); 2 = Stewart *et al.* (1997).

Table 2.4. Mean concentrations ( $\mu\text{g g}^{-1}$  wet weight) and coefficients of variation for mercury in marine organisms in the Azores. Median values are given for each group.

Group	Species	Tissue	n	Mean ( $\mu\text{g g}^{-1}$ wet weight)	CV	Ref.	
Fish	<i>Macroramphosus scolopax</i>	Whole	42	0.16	34%	1	
	<i>Scomber japonicus</i>	Whole	4	0.27	26%	1	
	<i>Capros aper</i>	Whole	19	0.44	71%	1	
	<i>Trachurus picturatus</i>	Whole	20	0.45	81%	1	
	<i>Maurolicus muelleri</i>	Whole	11	1.03	22%	1	
	<i>Electrona rissoi</i>	Whole	10	0.97	44%	1	
	<i>Myctophum punctatum</i>	Whole	6	0.96	27%	1	
	<i>Ceratoscopelus maderensis</i>	Whole	14	1.20	9%	1	
			<b>Medians</b>		<b>0.7</b>	<b>30%</b>	
Seabirds	Bulwer's petrel	Egg	16	1.60	41%	3	
	Cory's shearwater	Egg	23	0.51	21%	3	
	Common tern	Egg	20	0.32	50%	3	
	Roseate tern	Egg	17	0.45	43%	3	
			<b>Medians</b>		<b>0.5</b>	<b>42%</b>	
	Cory's shearwater	Chick feathers	7	4.2	18%	2	
	Yellow-legged gull	Chick feathers	5	4.0	38%	2	
	Common tern	Chick feathers	42	1.6	41%	2	
	Roseate tern	Chick feathers	13	1.2	47%	2	
			<b>Medians</b>		<b>2.5</b>	<b>40%</b>	
	Bulwer's petrel	Adult body feathers	24	22.4	22%	2	
	Cory's shearwater	Adult body feathers	40	6.3	33%	2	
	Little shearwater	Adult body feathers	4	2.4	48%	2	
	Madeiran storm petrel (June breeders)	Adult body feathers	25	9.5	23%	2	
	Madeiran storm petrel (November breeders)	Adult body feathers	27	16.0	27%	2	
Common tern	Adult body feathers	28	2.4	28%	2		
Roseate tern	Adult body feathers	21	2.2	36%	2		
		<b>Medians</b>		<b>6.3</b>	<b>28%</b>		

Ref. = Reference: 1 = Monteiro *et al.* (1996); 2 = Monteiro *et al.* (1995); 3 = Monteiro *et al.* (1998).

Table 2.5. Biomagnification factors between mercury in food and in seabird feathers for populations in the Azores (from Monteiro *et al.* 1998).

Species	Mercury in food ( $\mu\text{g g}^{-1}$ dry weight)	Mercury in adult body feathers ( $\mu\text{g g}^{-1}$ fresh weight)	Biomagnification factor (fresh weight to fresh weight)
Bulwer's petrel	0.318	22.3	225
Madeiran storm petrel (hot season)	0.243	11.1	146
Madeiran storm petrel (cool season)	0.432	17.4	129
Cory's shearwater	0.131	5.4	132
Little shearwater	0.72	3.1	138
Common tern	0.54	2.1	125

marine mammals and probably some seabirds can demethylate methylmercury in order to store it in a relatively non-toxic, and non-labile, inorganic form in the liver (Thompson and Furness, 1989b).

Mercury concentrations increase with age in fish and in marine mammals, but not in seabirds (Furness *et al.*, 1990; Furness, 1993). Seabirds lose their mercury into growing feathers. All mercury in feathers is methylmercury (Thompson and Furness, 1989a), even in seabirds where the bulk of the mercury in the liver is inorganic (Thompson and Furness, 1989b). Concentrations of mercury in feathers vary according to the moult pattern. Feathers renewed first in the major autumn moult have the highest concentrations, and concentrations decrease as the moult progresses and the body pool of mercury is depleted (Furness *et al.*, 1986; Braune, 1987; Braune and Gaskin, 1987). A sample of several small body feathers provides a good measure of the mercury level in an individual bird, and is the most appropriate way of using feathers from live birds or museum skins for mercury monitoring. Female seabirds may have slightly lower concentrations of mercury than those found in males because they put some mercury into the eggs (Becker, 1992), but differences between the sexes tend to be small (Lewis *et al.*, 1993). Both laboratory experiments and oral dosing of wild birds in the field with mercury have shown that mercury concentrations in the blood and internal organs are closely related to the ingested dose, and that concentrations in feathers are dependent on the mercury level in the blood during feather growth, which itself is a function of mercury level in the diet.

Mercury concentrations in seabirds can be related to mercury levels in their prey (Monteiro *et al.*, 1995, 1998; Figure 2.8) and, in particular, show that seabirds feeding on mesopelagic prey have much higher mercury burdens than seabirds feeding in other food chains (Tables 2.4 and 2.5). This reflects a trend for mercury levels to be higher in fish from deeper water (Monteiro *et al.*, 1996), presumably due to methylation of mercury in deep low-oxygen water, thereby permitting greater assimilation of mercury into biota.

Geographical variations in mercury contamination can be seen from sampling seabird feathers, chick down, or eggs from different colonies. For example, Renzoni *et al.* (1986) showed higher levels of mercury in Cory's shearwaters in Mediterranean colonies than in Atlantic colonies. Becker *et al.* (1993) showed that feathers from tern and gull chicks indicated local patterns of mercury pollution of the German North Sea coast attributable to river inputs of mercury into the southern North Sea. Also eggs clearly indicate spatial variation (Figure 2.4) and temporal trends of mercury in the marine environment (Becker *et al.*, 1998).

Joiris *et al.* (1997) found a strong increase in the mercury levels in common guillemots in the southern North Sea through the winter, as these birds spend the summer in areas with much lower mercury exposure (the northwestern North Sea) (Figure 2.9).

Body feathers from adult seabirds can be used to show long-term trends in mercury contamination, as museums hold material dating back to the 1850s. Inorganic mercury-contaminated study skins can be separated from the methylmercury put into feathers by the birds by making a simple biochemical fractionation, so that only the mercury of biological relevance is measured. Such studies have shown approximately 400% increases in mercury in seabirds from the UK coast (Thompson *et al.*, 1992; Figure 2.10) and in the Azores region (Monteiro and Furness, 1997; Figure 2.11), but not in southern hemisphere seabirds (Thompson *et al.*, 1993b). These patterns match closely with predictions from modelling of atmospheric transport of mercury from industrial sources (Mason *et al.*, 1994; Fitzgerald, 1995), which predict a four-fold increase in mercury in northern hemisphere ecosystems but little increase in the southern hemisphere. On the southern North Sea coast, the pattern of mercury levels in herring gull feathers from 1880–1990 showed about a 300% increase during the Second World War and a second wave of increase during the 1960s and 1970s owing to the industrial development in central Europe (Thompson *et al.*, 1993a). It appears that only seabird feathers permit retrospective monitoring of mercury contamination in marine food webs over the past 150 years (Swain *et al.*, 1992).

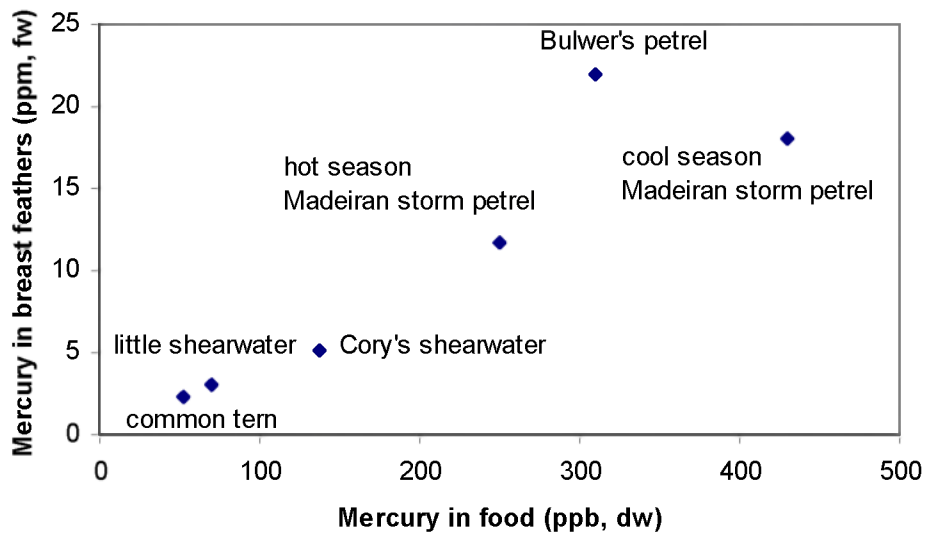


Figure 2.8. Relationship between mean mercury concentrations in breast feathers of seabirds from the Azores and in the food of these birds sampled during the breeding season at the colony. From Monteiro *et al.* (1998).

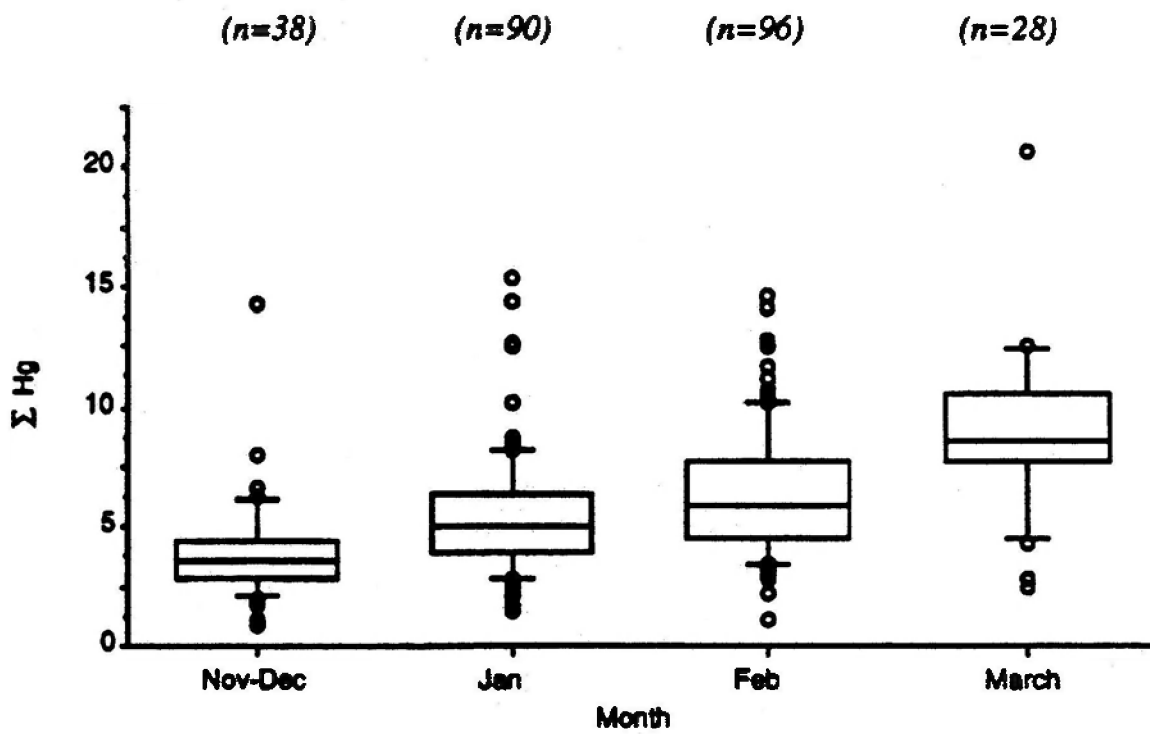


Figure 2.9. Mercury in common guillemot liver samples as a function of collection date ( $\mu\text{g g}^{-1} \text{ dw}$ ; Joiris *et al.*, 1997).

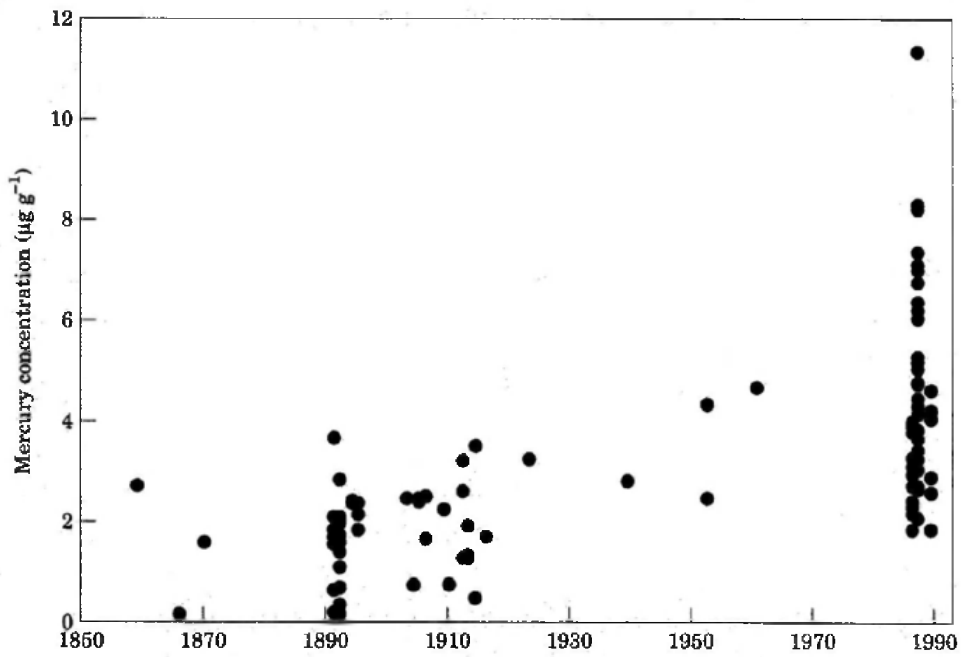


Figure 2.10. Mercury concentrations in body feathers of Atlantic puffins from southwest Britain and Ireland from 1850 to 1990 (from Thompson *et al.*, 1992).

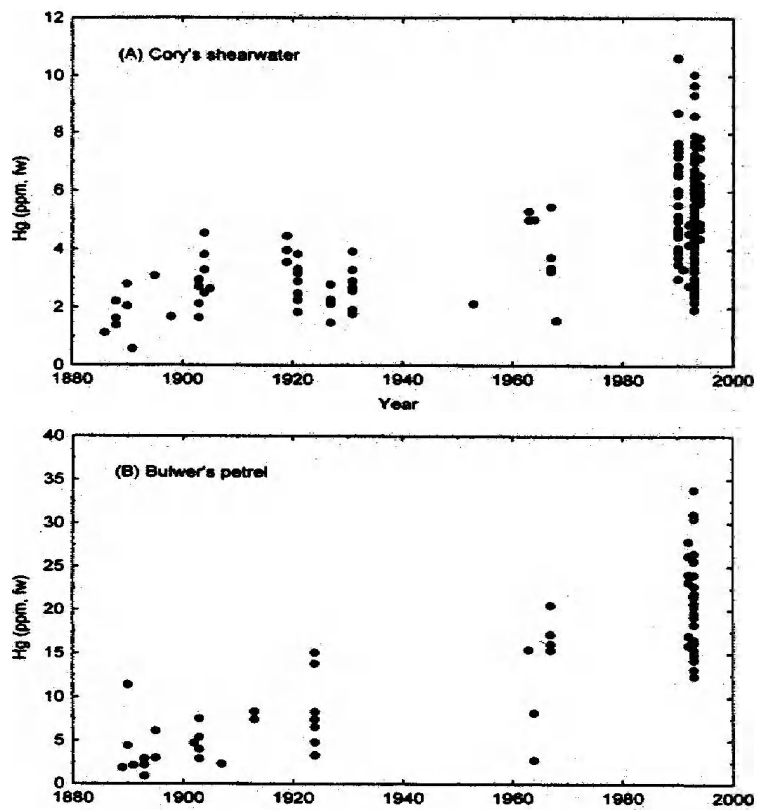


Figure 2.11. Mercury concentrations in body feathers of (A) Cory's shearwaters (which feed on epipelagic fish and squids) and (B) Bulwer's petrels (which feed on mesopelagic fish) in the Azores, from 1880 to 1995 (from Monteiro and Furness, 1997).

Whereas feathers reflect the bird's mercury body burden only during the time of feather growth, bird blood can be used indicating the present-day mercury contamination throughout the year (Kahle and Becker, 1999).

### 2.2.5 Organotin

Although organotins (mainly tributyltin, TBT) have been mostly monitored in molluscs (Morcillo *et al.*, 1997; Harino *et al.*, 1998) and, with particular regard to imposex in whelks and developmental abnormalities in oysters, there has recently been increased interest in the pronounced bioaccumulation (especially in the liver) of organotins in marine mammals (Lee, 1991; Iwata *et al.*, 1997; Tanabe *et al.*, 1998) and in seabirds (Guruge *et al.*, 1997a; Kannan *et al.*, 1998). Factors resulting in high concentrations of organotins in particular species or populations of seabirds are not yet known and there is a need for further work on the patterns of accumulation of organotins by seabirds and the toxic implications of the accumulation. It is as yet unclear whether organotin contamination can be monitored by sampling seabirds. So far, organotin levels have been measured in liver and kidney tissues of seabirds and marine mammals, requiring sampling of dead animals or the killing of animals. Guruge *et al.* (1996, 1997b), however, show elevated TBT levels in feathers suggesting that birds excrete TBT, as they do mercury during moult, and that feathers could be used as an indicator of TBT contamination in wild birds.

### 2.2.6 Other metals

Cadmium is concentrated at high levels in the kidney of birds and mammals. Lead is concentrated particularly in the bones, but can also be measured in the blood. These elements enter eggs or feathers from the blood only in minute amounts, but cadmium and lead can be deposited onto feather surfaces from the atmosphere (Hahn, 1991); thus, using feathers to monitor amounts of these metals in the food chains of birds is confounded by problems of low concentrations and a high likelihood of external contamination. Nevertheless, feathers have been used to assess contamination by a wide range of elements (Burger, 1993). New techniques permitting the location of atoms within or on the surface of feathers may permit this practice to be developed with greater reliability. Several papers published in the 1970s and 1980s contain measurements of improbably high levels of metals in seabird feathers or eggs that must now be considered unreliable data.

Several papers provide details of concentrations of a range of elements in seabird tissues (e.g., Honda *et al.*, 1990; Elliott *et al.*, 1992; Wenzel and Gabrielsen, 1995; Kim *et al.*, 1996, 1998). This requires either the killing of birds to obtain samples or the use of chance sampling opportunities. Wenzel and Adelung (1996) examined the possible use of oiled birds as a means of sampling for heavy metal monitoring.

### 2.2.7 Radionuclides

Seabirds are probably not very useful in radionuclide monitoring because levels do not tend to increase up food chains and the assimilation efficiency of most radionuclides through the digestive system of seabirds is poor (Brisbin, 1993). Matishov *et al.* (1996) reported on caesium-137 in seabirds in the Barents Sea, but very few data on radionuclide levels in seabirds are available. One might anticipate that levels in mollusc-eating shorebirds and seabirds could be elevated in areas such as the Cumbrian coast, but this does not seem to have been investigated.

## 2.3 Advantages of seabirds as biomonitors of pollution

The advantages that seabirds can provide as a tool for monitoring particular marine contaminants are considered below. Disadvantages are covered separately in Section 4.

### 2.3.1 Taxonomy and biology

The phylogeny of seabirds has been the subject of very detailed research. There still remain some minor uncertainties, such as the numbers of taxa within certain groups. For example, molecular data suggest that some albatross species could be split into several closely related but genetically distinct species (Robertson and Gales, 1998). The Madeiran storm petrel *Oceanodroma castro* may consist of sibling species with seasonally distinct breeding (Monteiro and Furness, 1998). Nevertheless, such examples are aspects of detail, and it is unlikely that significant changes to the phylogeny of seabirds will arise as a result of further research. Thus, studies of contaminant levels can be based on a stable and well-described phylogeny. Furthermore, the huge amount of research on the biology of seabirds means that the migration patterns, seasonal distribution, feeding ecology, breeding biology, and physiology of seabirds are very well known. Of course, the amount of detail known about populations varies. There are some seabirds that have been little studied, whereas there are very large numbers of publications on the biology of some species such as herring gull, common guillemot, black-legged kittiwake, and common tern. It is likely that widespread and well-studied species would also be most suitable as biomonitors of contaminants because a prerequisite for a biomonitor would be availability of satisfactory sample sizes and ease of sampling. The detailed knowledge of seabird ecology provides a good background for the interpretation of patterns of contaminant levels in seabirds, whereas for many other groups of marine animals, too little is known of the ecology of the organisms to permit such interpretation. Thus, for example, knowledge of the seasonal pattern of moult permits the selection of particular feathers to assess mercury contamination in different food webs in which the same individual bird is feeding at different times of the year (Thompson *et al.*, 1998a).

### 2.3.2 Tendency to accumulate high concentrations

Seabirds tend to feed at high trophic levels in marine food chains, so contaminants that accumulate up food chains will be well represented in seabirds. These include organochlorines and methylmercury, which are lipid-soluble contaminants with low solubility in water. Seabirds thus provide a potentially good biomonitor for lipid-soluble organic contaminants, since concentrations in seabirds are likely to be relatively high, and the careful selection of seabird species and sampling tissue should allow an appropriate spatial and temporal scale integration of the contaminant signal as well as giving an indication of the likely risk of toxic effects to animals high in the food chain (including man through harvesting of finfish and shellfish stocks). In contrast, many water-soluble contaminants with low lipid affinities, such as inorganic metals, show no trend of increased concentration with trophic level. In such cases, concentrations may be lower in seabirds than in some biota at low trophic levels. This is particularly true for radionuclides such as uranium and plutonium, where molluscs or algae provide a more appropriate biomonitor than do seabirds (Brisbin, 1993). Nevertheless, certain metals accumulate to high concentrations in particular avian tissues. For example, cadmium concentrations are particularly high in seabird kidneys (Stewart *et al.*, 1996).

As biomagnification factors not only increase with the food a species chooses, but also with the environmental burden from a contaminant, seabirds indicate inter-site or inter-year differences in contamination more distinctly than other animals (e.g., Figure 2.4).

### 2.3.3 Ease of sampling

Almost all seabird species are colonial breeders, thus sampling large numbers of birds is often possible at selected colonies. Most seabirds breed at traditional sites every year, with the same adults usually nesting in the same territory each year, so that locations where seabird samples can be conveniently collected are highly predictable. When incubating, many adult seabirds are relatively easy to catch, and adults of some species are easy to catch while rearing chicks; but catching adults away from the nest and at times of the year when the birds are not breeding can be difficult. Eggs and chicks can be sampled at appropriate dates during the breeding season. Breeding tends to be consistent from year to year, so that optimal dates for sampling are predictable. The behaviour of seabirds at the colony in response to human activity is highly variable from species to species. Birds of certain species panic and human disturbance can cause mortality of eggs or chicks, whereas other species are highly tolerant of disturbances. The choice of monitoring species and sites should take this into account. For example, great cormorants tend to lose eggs or chicks when people enter their colony and they are not ideal as a choice of monitoring species for this reason. In contrast, black-legged kittiwakes tend to remain on the

nest and egg or chick losses due to human disturbances are very rare. Responses can also vary between colonies. Adult northern gannets on the Bass Rock are easily caught at the nest with little disturbance, whereas northern gannets at Grassholm or St Kilda tend to panic when a human approaches. At the latter sites, northern gannets very rarely see humans at their colony, whereas at the Bass Rock human visitors are regular and numerous and the birds have learned to adapt to them.

Responses of particular species of seabirds to human intrusions are well known, as are the locations and accessibility of colonies, so it is very easily possible to plan, timetable, and cost a sampling programme.

By contrast, marine mammals are very difficult to sample, and most studies of contaminants in marine mammals have been based on small sample sizes of animals found stranded or entrapped in fishing gear (Addison *et al.*, 1984; Lee, 1991; Kannan *et al.*, 1994; Becker *et al.*, 1997; Fossi *et al.*, 1997; Iwata *et al.*, 1997; Krahn *et al.*, 1997; Moessner and Ballschmiter, 1997; Tanabe *et al.*, 1998). The concentrations of contaminants in marine mammals may be of interest because they give an indication of exposure that humans would experience from a marine diet, or from consumption of marine mammal meat (Weihe *et al.*, 1996), and they may reach levels that are toxic to marine mammals and so are of concern in terms of marine mammal conservation (Fossi *et al.*, 1997). However, in terms of monitoring marine ecosystems, sampling of marine mammals is difficult to achieve regularly and reliably. Seabirds are much more amenable to sampling in sufficient quantity.

### 2.3.4 Choice of sampling tissues

The ideal tissues to sample depend on the contaminant of interest, but the selection of several sampling tissues can often provide a much greater depth of information than taking a single monitoring tissue. For example, mercury concentrations vary among feathers of an individual seabird in a way that reflects the pattern of moult (Furness *et al.*, 1986; Braune, 1987; Braune and Gaskin, 1987), such that sampling feathers grown at different times of the year can indicate seasonal variations in mercury burdens of seabirds and, hence, seasonal patterns of mercury assimilation.

Adult seabirds tend to be long-lived and may range widely between breeding seasons, so that they may be exposed to contaminants far distant from the breeding colony. Migrations and diets may vary between individuals, between sexes, and between age classes, so that contaminant exposures may be highly variable between birds of differing status. With knowledge of these patterns, sampling can be planned to minimize the variance due to differences within populations. As a broad generalization, contaminant concentrations in chicks tend to be much less variable than those in adults. In part, this reflects the fact that contaminant burdens in chicks are largely derived from food fed to chicks during their growth, in addition to the amounts of contaminants received from their mothers prior to hatching (Becker *et*

*al.*, 1993). Chick diets tend to be rich in energy-dense food, whereas adults may take a more varied diet and with greater variation in diet among individuals. Secondly, the food for the chicks is taken from a relatively small area over which parents forage during the chick-rearing period. In contrast, adults may range over much longer distances from the colony during the pre-laying and incubation periods, and may carry stored contaminants that they assimilated from food eaten during the non-breeding period when they were widely dispersed away from their colony. Sampling chicks can therefore provide indications of the level of contamination within the defined foraging range of the parents during chick-rearing and permit comparisons between colonies so that geographical variations in contaminant concentrations can be determined.

For some contaminants, samples of chick feathers or down (mercury) or blood (mercury, organochlorines, butyltins) provide satisfactory monitoring information. For others (e.g., cadmium), the concentrations in feathers or blood may be too low to quantify, since the contaminant is strongly bound in a particular tissue and is not free to circulate in the blood. Monitoring of such contaminants might require killing the chicks to obtain the necessary tissue (e.g., kidney for cadmium analysis).

Since killing chicks may be unacceptable, sampling eggs has its attractions, particularly as many species of seabirds can replace a clutch that is removed. Taking a single egg from a clutch of several may have very little effect on breeding success as the survival of the chicks from the remaining eggs may be increased due to the reduction in sibling competition. Sampling eggs takes little time and eggs are easy to handle and to store. Contaminant burdens in the eggs of a specific area and year tend to reflect the contaminant uptake of the female (healthy and reproductive members of the population) in the period shortly prior to egg laying (Becker, 1989; Dietrich *et al.*, 1997). Eggs can be used to measure contamination of the food web in the area around the colony over which food is gathered in the pre-laying period. Being restricted to the breeding season, the seasonal variability in levels of contaminants is reduced. Compared to tissues, the egg matrix has a consistent composition with high lipid contents that accumulate persistent compounds to high concentrations, simplifying chemical analyses.

### 2.3.5 Known foraging range

Although details of the foraging range of breeding seabirds are not known for every species, the information is available for many, and can be inferred for others from knowledge of closely related species or from other aspects of breeding biology (e.g., duration of the alternating periods spent on and off the egg during incubation). Foraging ranges may vary between different colonies or according to food abundance, but there is enough information known to permit sampling seabirds at breeding colonies selected to provide fairly accurate estimation of the geographical variation in contaminant concentrations. Foraging ranges of breeding seabirds

vary from a few kilometres in the case of terns and shags, to thousands of kilometres, in the case of albatrosses and some shearwaters and petrels. Knowing the scale of foraging ranges of particular species may assist in selecting species that would provide the appropriate scale for a study. Terns, for example, can provide evidence of differences in contaminant levels resulting from local river discharges, whereas certain large albatrosses may provide data from the entire Southern Ocean.

### 2.3.6 Diet

Contaminant uptake will vary to some extent depending on the variability of the diet, both between individuals and across years. Selection of appropriate seabird species with narrow and consistent diets can avoid the noise that might otherwise be introduced by such variations in diet. For example, common guillemots and European shags have diets of fish that vary relatively little, whereas herring gulls are opportunists that may switch between highly differing diets. On this basis, herring gulls may be less suitable as biomonitors than common guillemots or European shags. However, there are methods that can be used to investigate diet so that the effects of changes in diet can be assessed. These include both the conventional sampling of food regurgitates, fish observed to be carried into the colony, contents of pellets regurgitated by adults, samples offloaded from chicks by 'stomach-pumping', or indirect methods of diet assessment such as analysis of stable isotope ratios. Stable isotopes of carbon and of nitrogen have been widely used as indirect measures of diet, and especially of trophic status; they have recently been used in combination with analysis of contaminants in the same samples to aid the interpretation of differences in contaminant levels between samples (Hobson *et al.*, 1994; Jarman *et al.*, 1996, 1997; Atwell *et al.*, 1998). Since stable isotope analysis is based on the analysis of protein, it has the advantage that it can be used with feather material from study skins in museums so that even historical samples can be examined for dietary variation detectable by isotope analysis (Thompson and Furness, 1995; Thompson *et al.*, 1998a).

### 2.3.7 Historical samples

Although there are few, and only rather recent, tissue banks that can provide materials for examining temporal trends in contaminant burdens in biota (Elliott, 1985; Schladot *et al.*, 1993; Becker *et al.*, 1997; Krahn *et al.*, 1997), museum materials can be of use as a means of examining long-term trends. Eggshells in museum collections provide clear evidence of the effects of DDT poisoning through eggshell thinning effects. Eggshells may also provide an opportunity to investigate trends in contaminant levels through chemical analysis of the shell or membrane composition, especially for some heavy metals (Burger, 1994; Burger and Gochfeld, 1996). Similarly, skeletal material might be used to examine trends in contamination, particularly for lead but possibly also for other contaminants. Feathers from study skins can be used to measure methylmercury contamination.



Mercury concentrations in feathers reflect mercury levels in the blood at the time of feather growth and these in turn correlate with the amount of mercury in the diet (Monteiro and Furness, 1995; Monteiro *et al.*, 1998). All of the mercury assimilated by seabirds and subsequently excreted into growing feathers is in the form of methylmercury, so any later contamination of the feathers with inorganic mercury from dust or preservatives can be removed by a biochemical separation (Thompson and Furness, 1989a). As a result, it has been possible to quantify the increases in mercury contamination of marine food webs over the past 150 years by analysis of mercury concentrations in selected feathers from seabird skins (Thompson *et al.*, 1992, 1993a, 1993b, 1998b; Monteiro and Furness, 1997). Such an analysis is not possible for fish since fish are stored in preservative solutions that can affect concentrations of metals in the tissues. It might be possible to investigate long-term trends in concentrations of other heavy metals in seabird feathers (Burger, 1993), but this would require an analytical facility that can discriminate between metal incorporated into the feather structure from the bird's blood and metal that has been deposited onto the feather surface, either during the life of the bird or during storage.

### 2.3.8 Low variance within population

Contaminant concentrations in samples of seabird chicks may be less variable than in other biota, such that the sample sizes required to detect a particular magnitude of increase would be less using seabirds than using other biota (Gilbertson *et al.*, 1987; Fryer and Nicholson, 1993). Similarly, selecting adult seabirds, or the chicks of seabirds with larger foraging ranges, may permit more cost-effective monitoring of long-term trends where spatial resolution of contaminant variation is not the objective. Where the aim is to examine spatial variation, seabirds are not as useful in providing the small-scale resolution that could be obtained using sedentary animals such as blue mussels; however, for large spatial scales (> 10 km), seabirds may integrate spatial variation that would be noise in an analysis based on sedentary animals and may provide a better means of assessing large spatial scale patterns.

### 2.3.9 High public interest

The fact that birds are of considerable public interest can be very helpful in a monitoring programme. Many amateur ornithologists, reserve wardens, and conservation staff are able to provide data or collect samples within a monitoring programme, and coordination of such work can be achieved through existing specialist groups such as the Seabird Group, Royal Society for the Protection of Birds (RSPB), scientific institutions or through wardens in nature reserves or national parks, and others. For example, in the UK, the Joint Nature Conservation Committee administers the seabird populations and productivity-monitoring programme, with assistance from the Seabird Group, RSPB, English Nature, Scottish Natural Heritage,

Countryside Council for Wales and others, to which over 100 people contribute data in a standardized format throughout the British Isles (Thompson *et al.*, 1997).

### 2.3.10 Resistance to toxic effects

As a broad generalization, seabirds appear to be more resistant to toxic effects of most contaminants than are mammals or terrestrial birds (Beyer *et al.*, 1996). High concentrations of mercury in apparently healthy breeding birds are found in many seabird species, well above levels that would cause toxic effects in terrestrial or freshwater birds (Thompson, 1996). PCBs can occur in concentrations in apparently healthy seabirds that would certainly have toxic effects at the same concentrations in mammals (Barron *et al.*, 1995; Guruge and Tanabe, 1997). On the other hand, during ontogeny seabirds are vulnerable to toxic chemicals like PCBs (e.g., Becker *et al.*, 1993). There is no evidence that TBT at moderately high levels has harmful effects in seabirds (Guruge *et al.*, 1997a; Kannan *et al.*, 1998).

The tendency for seabirds to be able to carry high concentrations of contaminants without displaying impaired reproduction or survival means that concentrations of contaminants in samples of seabirds should be a true reflection of exposure and not one that is biased by loss from the population of individuals carrying toxic doses. Choosing the egg as the sampling matrix avoids such problems, as eggs originate from the healthy, reproductive part of the population.

## 2.4 Drawbacks of using seabirds as biomonitors of contamination

### 2.4.1 Complex physiology

The complex physiology of vertebrates, in some respects, makes them less suitable than invertebrates as biomonitors. Seabirds regulate tissue concentrations of essential metals and partly regulate the concentrations of some non-essential metals. For example, the coefficient of variation for cadmium in seabirds is much less than for mercury. For monitoring purposes, seabirds would be of little or no use as monitors of iron, zinc or copper contamination, and may be less suitable than invertebrates as monitors of cadmium. Moulting, seasonal variation in organ size, adaptation to season, fat deposition, anorexia, and other processes can affect contaminant concentrations in tissues, making changes between samples difficult to interpret as due to changes in pollution load rather than changes in physiology (van den Brink *et al.*, 1998).

Sex differences can occur, especially where females can excrete a contaminant into developing eggs, but such sex differences in contaminant burdens tend to be small (Furness *et al.*, 1990; Lewis *et al.*, 1993; Stewart *et al.*, 1994, 1997; Burger, 1995).

Eggs may be formed directly from recently assimilated food, or from materials drawn from stores within tissues.

The relative importance of these two sources of material may vary between species, and often varies within a female through the laying sequence of egg production. As a result, since body stores and current diet may differ in contaminant content, egg mercury concentrations in terns and gulls decline with laying sequence (Becker 1992). Levels of organochlorines also vary systematically with egg laying sequence (Mineau, 1982; Becker and Sperveslage, 1989).

#### **2.4.2 Uncertain provenance**

Seabirds sampled at sea are unlikely to be all from a single local population but may include birds of differing status from a variety of breeding areas. Seabirds sampled at a colony may be more homogeneous, but their previous movements during the non-breeding season may have exposed them to various different sources of contamination and these may continue to be represented in the body burden through to the breeding season, possibly causing confusion.

#### **2.4.3 Need to avoid killing birds**

In most countries, a licence is required to kill adult birds or chicks, to collect samples of eggs or to draw blood samples, or indeed to catch and handle live seabirds. However, such requirements should not hinder contaminant monitoring that is based on non-lethal sampling. In many countries in Europe, sampling by killing birds for contaminant analyses would now be considered to be unacceptable in most situations. The trend towards greater protection for wildlife is likely to continue. This could especially affect any programme of contaminant monitoring in countries where killing is currently an acceptable approach but may not be in the future, and any new programme of contaminant monitoring should be designed with such trends in public attitude and legislative control in mind.

#### **2.4.4 Diet switching and diet specialisation**

Many seabirds will switch their diet according to the relative abundance of preferred prey, and such diet switching can affect contaminant burdens in seabirds, especially if the switch is between vertebrate and invertebrate prey types, or between prey at different trophic levels. Within populations, diet specialization can lead to increased variance of contaminant levels among individuals. Although seabird species can be selected as biomonitors on the basis of their having a stenophagous diet, even the most specialized seabirds may switch from one prey species to another if food availability changes enough. Such changes can be detected by monitoring the diet or by analysis of stable isotopes as indicators, but this adds to the cost and complexity of a monitoring programme.

#### **2.4.5 Difficulties in monitoring toxicity**

Since seabirds tend to show a higher tolerance of many contaminants than do other animal groups, the

opportunity to use breeding performance as a means of monitoring contaminant levels is of limited value. However, this possibility should not be discounted. Fox and Weseloh (1987) and Fox *et al.* (1991) suggested that breeding performance of gulls on the Great Lakes may be a useful indicator of contaminant exposure in this highly polluted region. They suggest the possibility that low gull breeding success might indicate toxic effects of a complex mixture of chemical contaminants that it would be extremely difficult and expensive to monitor by chemical analysis of gull samples or of other biota or physical samples. Thyen *et al.* (1998) proposed a programme for monitoring breeding success of coastal birds in the Wadden Sea, among other aims also to indicate contamination. However, seabird breeding failures can be caused by a wide variety of environmental factors, including food shortages, adverse weather, predators and human disturbances, so one should be cautious about the possibility of detecting a relatively weak signal due to contaminants from the considerable and often unpredictable noise caused by a wide range of other factors. One reproductive parameter which may indicate shell thinning or embryotoxicity by chemicals is hatching success, in case of its reduction after external causes have been excluded (Becker *et al.*, 1993).

It is possible that specific biomarkers of toxicity might be useful, either at the biochemical level (Peakall 1992) or at the level of specific effects on reproduction. Sex ratio distortion as a result of feminization of genetically male embryos, for example, provides a fairly specific effect of oestrogenically active contaminants that is unlikely to be mimicked by other environmental influences. Similarly, teratogenic effects of particular contaminants may be evident in embryos or recently hatched chicks, while in uncontaminated populations, even those stressed by various natural environmental factors, such abnormalities are exceedingly rare.

### **2.5 Criteria for selecting seabird biomonitors**

A good candidate for a seabird biomonitor of contaminants should have the following attributes:

- a) accumulation of the contaminant to high concentrations;
- b) resistance to toxic effects due to the contaminant (unless these are what is being monitored);
- c) known, or preferably no, migratory habits;
- d) a foraging range consistent with the spatial scale over which the contaminant is to be monitored;
- e) a large population size with known breeding biology and ecology, and with large numbers of colonies throughout the area where contaminant monitoring is required;
- f) be easy to collect without major disturbance to the breeding colony, and have easily identifiable life-stages if a particular category is to be sampled;

- g) have known physiology;
- h) have a narrowly defined and consistent diet;
- i) feed predominantly or exclusively on prey in the food web under investigation.

Based implicitly or explicitly on criteria similar to these, the Institute of Terrestrial Ecology sampled the eggs of common guillemots and northern gannets to monitor organochlorine contamination of marine ecosystems around the United Kingdom. Gilbertson *et al.* (1987) proposed the monitoring of contaminants in the North Atlantic by the sampling of eggs of Atlantic puffin and/or common guillemot (pelagic fish food web), and Leach's storm-petrel (plankton food web, west Atlantic). For the Wadden Sea, Becker (1989) and Becker *et al.* (1991) proposed common tern (fish) and Eurasian oystercatcher (benthic animals) as monitoring species of the parameter 'contaminants in coastal bird eggs'. This parameter, of high priority within the Trilateral Monitoring and Assessment Program (TMAP), has been studied in the Wadden Sea every year since 1996 (Becker *et al.*, 1998). Choosing more than one seabird species feeding on different prey can indicate contamination of different parts of the food web.

In practice, most published studies reporting contaminant concentrations in seabirds, or even reporting spatial or temporal patterns of contaminant levels in seabirds, have been based on data apparently collected adventitiously rather than as a planned monitoring programme.

## **2.6 Recommendations for monitoring pollutants using seabirds**

### **2.6.1 Oil**

Ongoing programmes of monitoring the proportions of dead seabirds found on shorelines ('Beached Bird Surveys') should be encouraged as a cost-effective means (most are carried out by amateurs at no cost and are organized by NGOs) of determining long-term trends and geographical patterns of oil pollution at sea. Such monitoring is of greater interest to seabird conservation than to fisheries.

### **2.6.2 Plastic particles**

Given that this type of pollution appears to be increasing, there is a need for monitoring the amounts of plastic ingested by seabirds, especially petrels. Whether ingestion of plastic pellets by fish results in harm to the fish is unclear. The evidence from seabirds suggests that the plastic is directly ingested by seabirds and not obtained indirectly inside prey that they consume. Such monitoring is of greater interest to seabird conservation than to fisheries, but would help to increase public awareness and concern about plastic pollution of the seas.

### **2.6.3 Organochlorines**

Sampling of seabird eggs as a means of monitoring local contamination by organochlorines (for advantages, see Section 2.3) should be developed into integrated marine contaminant monitoring programmes, with the selection of appropriate locally common and internationally widespread monitoring species. Table 2.6 lists seabird species suggested as monitors. In the Wadden Sea, the benthivorous Eurasian oystercatcher and the common tern were selected as monitoring species within the regional Trilateral Monitoring and Assessment Program, which was implemented in 1996.

### **2.6.4 Mercury**

Developed methods to monitor mercury contamination in marine food chains by sampling chick down or feathers from chicks or adults, or from blood or egg samples, should be applied in areas where there is concern about possible contamination of marine food chains by mercury. Sampling eggs from colonies located near rivers carrying mercury contamination can be used as a means of monitoring trends in riverine mercury loadings reaching the sea. Such monitoring provides useful evidence of the successful reduction in mercury contamination where technical measures have been implemented to reduce discharges. The same monitoring species used to measure organochlorine levels are proposed for integration in monitoring programmes for mercury (Table 2.6).

Table 2.6. Seabird species suggested as monitors of marine contamination by organochlorines and mercury in the Northeast Atlantic and adjacent seas. Information on population size and trends in Europe, clutch size, diets, feeding range as well as distribution is presented.

Species	Eggs	Population	Feeding ecology and food	Distribution
Northern fulmar	1	3 million pairs; populations increasing	wide-ranging pelagic: zooplankton, offal, discards, fish, squid	Norway, Iceland, Faeroe, UK, Ireland, France; North America, Greenland
Northern gannet	1	230,000 pairs; populations increasing	wide-ranging: fish, sandeel, sprat, herring, mackerel, discards	Norway, Iceland, Faeroe, UK, Ireland, France; North America
European shag	3–4	86,000 pairs; populations mostly stable	coastal, short range: sandeel, sprat	Norway, Iceland, Faeroe, UK, Ireland, France, Mediterranean countries
Black-legged kittiwake	2	2–3 million pairs; populations mostly increasing or stable	wide ranging: small fish, zooplankton	Norway, Iceland, Faeroe, UK (all coasts), Ireland, France, Helgoland, North America, Greenland
Common tern	2–3	208,000 pairs; some populations increasing, some stable some decreasing	coastal: small fish	all European coasts (except Ireland and Faeroe); North America
Common guillemot	1	2 million pairs; most populations increasing	inshore: fish, especially sandeel, sprat	Norway, Iceland, Faeroe, UK (all coasts), Ireland, France, Sweden, Helgoland, North America

### 2.6.5 Organotin

There is a need for research into organotin levels in seabirds to determine whether they may have toxic effects on seabirds, and whether seabirds may be used as a means of monitoring organotin contamination on larger scales.

### 2.6.6 Other metals

There is a need for research into the possible use of eggshells, egg contents or feathers for monitoring cadmium, lead, and other elemental concentrations to avoid the need to kill birds for liver, kidney or bone samples. In particular, if methods can be developed to measure elemental concentrations within feather keratin separately from contaminants on feather surfaces, this would permit retrospective monitoring of long-term trends in elemental contamination of marine food chains, which has been done successfully for mercury.

### 2.6.7 Radionuclides

Seabirds are not preferred organisms to monitor radionuclide contamination.

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### 3 Review of the sensitivity of seabird populations to changes in life history parameters

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#### 3.1 Introduction

Seabird numbers in a population change as a consequence of births, deaths, immigration and emigration (Figure 3.1). Much effort has been put into surveillance of the size of seabird populations, and almost all of this effort has been directed at the census or sample monitoring of breeding numbers. This is largely because it is very much easier to make accurate counts of breeding numbers, or nests, than it is to count all birds in seabird populations. For example, in many seabird species some immature birds visit colonies but some remain at sea far from the colonies and mix with immatures from other populations. However, changes in breeding numbers may reflect not only births, deaths, immigration and emigration, but also changes in age of recruitment (defined here as first breeding), changes in the proportion of established breeders that choose not to breed in a particular year (non-breeding), or changes in timing or synchrony of breeding that can be confounded with changes in breeding numbers if, as is often the case, census counts of nests or breeders are made on a particular date in the breeding season.

Life history theory envisages trade-offs between components of fitness, such as survival and reproductive investment (Roff, 1992; Stearns, 1992; McNamara and Houston, 1996). Long-lived birds, such as most seabirds (Furness and Monaghan, 1987), may be expected to respond to adverse environmental conditions (such as reductions in food abundance, increased predation or bad weather) by reducing their investment in current reproduction in order to increase residual reproductive value and maximise lifetime reproductive success (Monaghan *et al.*, 1989; Montevecchi, 1993; Sæther *et al.*, 1993; Pyle *et al.*, 1997; Oro and Pradel, 2000). Based on such life history considerations, Cairns (1987) proposed a model of seabird responses to reduced food supply in which slight reductions affect breeding adult activity budgets and diet selection but not breeding success or adult survival rate. In his model, moderate reductions would also affect chick growth and breeding success, but only severe reductions in food supply would cause a reduction in adult survival rates, by which time effects on breeding success would be catastrophic. Monaghan *et al.* (1992) proposed that arctic terns followed such a pattern of reproductive investment, abandoning current breeding effort when body condition fell below a threshold that might begin to affect survival prospects of the parents. Similarly, Wernham and Bryant (1998) showed inter-year reproductive costs for Atlantic puffins, supporting the hypothesis that such long-lived birds reduce the 'quality' of their offspring or abandon a breeding attempt rather than compromise their survival

and future opportunities to reproduce. A few studies have validated a corollary of this model, showing that the demography of long-lived birds may tend to be driven by variations in reproductive output across years or by the balance between immigration and emigration, since adult survival rates would be expected to remain rather constant and not vary with food supply (Pons and Migot 1995; Oro *et al.*, 1999). However, in contrast to predictions of life history theory, several recent experimental studies in which reproductive effort of adults was manipulated by adjustment of brood sizes have shown that in black-legged kittiwakes, adult survival rate is considerably reduced in birds that invest more in current reproduction (Erikstad *et al.*, 1995; Jakobsen *et al.*, 1995; Golet *et al.*, 1998). Thus the framework of life history theory leads us to anticipate that seabird breeding numbers (equivalent to estimated 'population size') may vary as a consequence of life history trade off by seabirds.

Monitoring seabird breeding numbers alone, will not only fail to provide any indication as to the cause of a measured change in numbers, but also may present a less than optimal approach to detecting change, since specific life history parameters may vary more strongly in response to changes in food supply than would breeding numbers. For example, breeding success may show strong relationships with food supply but despite this there may be little consequent impact on breeding numbers, as a result of various forms of buffering of breeding numbers. Also, while change in food abundance may affect breeding success, small and often undetected changes to adult survival might have a more significant influence on population change. There is therefore a need to evaluate the strength of influence of different life history parameters on seabird population dynamics, with a view to considering the best approach to monitoring seabird populations to identify influences of changes in food availability that may result from changes in fisheries or fish stocks.

There is good evidence that changes in food availability can affect the reproductive output of seabirds and can correlate with overall changes in population size (Croxtall and Rothery, 1991). The locations of seabird colonies and numbers of breeding pair numbers are relatively well documented in the ICES area. However, owing to the life history characteristics of seabirds – k-strategy with high annual survival rates and low reproductive outcome – short-term changes in numbers or reproductive outcome caused by reduction of the food supply are difficult to detect by estimating population size at breeding sites. In particular, due to the delayed maturity of seabirds, changes may only become obvious many years later in

the breeding pair numbers, while current monitoring programmes usually do not include counts of the non-breeding part of the population, immature (=prebreeders, =subadults) and non-breeders (=established breeding adults that choose not to breed in a particular year).

Reproductive success, on the other hand, is a parameter indicating the actual environmental situation at the breeding site during a respective year. Productivity is not necessarily a very sensitive indicator of reproductive performance (Hunt *et al.*, 1986) and is not only influenced by food supply. Many other factors may affect it, such as predation, weather, flooding and others (e.g., Becker, 1998). Another limitation is that the indicative value of productivity is restricted mainly to the environmental situation during the reproductive period, excluding other periods of the year.

In consequence, to use seabirds as indicators of food supply, other life history parameters should be selected in addition. In this chapter we briefly review life history traits with respect to their desirability, feasibility and practicability as characteristics for monitoring seabird population dynamics as response to changes in food supply.

### 3.2 Population characteristics in seabirds

In general, seabirds represent extreme K-selected species. Adult survival is generally high, and annual reproductive output low. Many species delay first breeding until several years old (Table 3.1). In consequence, the non-breeding part of the population is high.

The regulation of a seabird population is shown schematically in Figure 3.1. Most important parameters are adult survival, reproductive success, subadult survival to recruitment, as well as immigration and emigration. The two last parameters are difficult to study; therefore in population models immigration and emigration are usually considered to be equal (emigration – immigration = 0).

For conservation as well as in order to use seabird populations as indicators for the marine environment it is important to know what parameters are especially sensitive and important in determining trends in the size of the population. A simulation modelling analysis enables us to make assessment of the theoretical sensitivity of seabird populations to changes in the demographic parameters. To demonstrate this we analysed the effects of a 5 per cent negative change in adult survival, subadult survival to breeding and reproductive success on a flourishing population, as well as an increase in the age of first breeding. We performed the calculations on the basis of parameters in actual populations of three species with different life styles (Table 3.2). To indicate one extreme of life history strategy we selected the especially K-selected northern fulmar. Towards the opposite extreme, with relatively low adult survival but high breeding output we selected the example of the European shag, and the common tern was selected as an example of an intermediate species (Table 3.2).

The model is a modification based on Croxall and Rothery (1991):

$$R_{t+k} = N_t \times b_t \times s_j \times (sa)^{k-1} \times s_a$$

Table 3.1. Some mean demographic and biological characteristics of the main families of seabirds. From Croxall and Rothery (1991).

Group	Age (years) at first breeding	Adult annual survival rate (%)	Clutch size	Chick-rearing period (days)
<i>Sphenisciformes</i>				
Spheniscidae (penguins)	4–8	75–85 (-95)a	1–2	50–80 (-350)b
<i>Procellariiformes</i>				
Diomedidae (albatrosses)	7–13	92–97	1	116–150 (-280)c
Procellariidae (petrels)	4–10	90–96	1	42–120
Hydrobatidae (storm petrels)	c4–5	c90+	1	55–70
Pelacanoididae (diving petrels)	2–3	75–80	1	45–55
<i>Pelacaniformes</i>				
Pelacanidae (pelicans)	3–4	c85	2–3	55–60
Sulidae (gannets, boobies)	3–5	90–95	1–2 (-3)d	90–120 (-170)e
Phaethontidae (tropic birds)	?	?	1	60–90
Fregatidae (frigate birds)	?c9–10	?	1	140–170+
Phalacrocoracidae (cormorants)	4–5	85–90	2–3	60–90
<i>Charadriiformes</i>				
Alcidae (auks)	2–5	80–93	1–2	15–20
<i>Lariiformes</i>				
Laridae (gulls)	2–5	80–85	2–3	25–50
Sternidae (terns)	2–5	77–90	1–3	25–40
Stercorariidae (skuas)	3–8	c93	2	25–40

a) emperor penguin; b) emperor penguin 170 days, king penguin 350 days; c) great albatrosses; d) boobies; e) red-footed booby to 140 days, Abbott's booby to 170+ days.

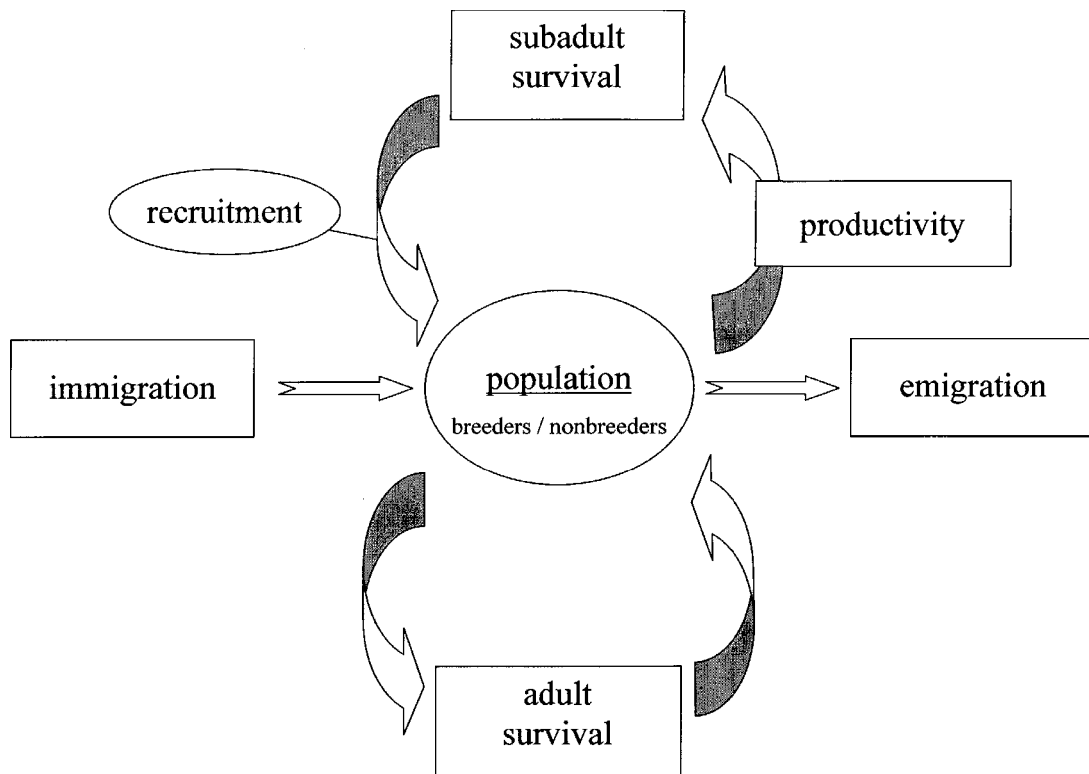


Figure 3.1. Scheme of the most important parameters regulating seabird populations.

where

$R_{t+k}$  = size of a cohort of potential recruits available in a population

$N_t$  = number of breeding pairs  $k$  years previously

$b_t$  = number of chicks fledged per pair  $k$  years previously

$s_j$  = the first-year survival rate

$(sa)^{k-1}$  = the annual survival rate from then until recruitment

$k$  = age at recruitment

$s_a$  = annual adult survival rate

The modelling output reveals that in northern fulmar and common tern, adult survival rate was the most sensitive parameter (Table 3.3). A reduction in adult survival by 5% causes a stronger response of population size than a change in the other parameters. In the common tern a reduction of subadult survival has an intermediate effect. In both species a shift of breeding success and age of first breeding have lesser effects on population change (Table 3.3). In the European shag, however, the change of each parameter has a similar effect on the population,

respectively. Owing to low adult survival a delay in the age of first breeding causes the strongest effect on population, in contrast to the other species.

The examples demonstrate the specific importance of adult survival and also of subadult survival on population size in seabirds. They indicate also the interspecific differences owing to various life history strategies. To understand the dynamics of a population in question it is necessary to gather information on these important population parameters.

### 3.3 Interactions between life history parameters

Life history theory predicts that selection should favour allocation of time and resources toward the stage of the life history at which survival or reproduction is least variable, all other things being equal (Stearns, 1992). In long-lived birds such as seabirds, survival of offspring is commonly variable and unpredictable, and selection should weight adult survival more heavily than fecundity (e.g., Goodman, 1974; Charlesworth, 1980). Several studies have shown that adult survival in many species of seabirds, including black-legged kittiwakes, is high and rather constant (e.g., Weimerskirch *et al.*, 1987;

Table 3.2. Population parameters of three species of seabirds leading to a population increase. Data from Birkhead and Furness, (1985; northern fulmar and European shag) and from Wendeln and Becker (1998) and Becker (unpubl.)

	Northern fulmar	European shag	Common tern
Adult survival	0.97	0.82	0.90
Sub-adult survival to breeding	0.63	0.39	0.34
Breeding success	0.49	1.84	1.20
Chick pair <sup>-1</sup> year <sup>-1</sup>			
Age at first breeding	8	3	3
Population change	+ 12%	+ 18%	+ 10%

Table 3.3. Population change in %, if population parameters are reduced by 5% (age of first breeding increased by 1 year) in three seabird species. The data are calculated using a simple population model and are based on the sources of Table 3.1.

Population parameters	Northern fulmar	European shag	Common tern
Adult survival	-5%	-5%	-5%
Subadult survival to breeding	-1%	-5%	-3%
Breeding success	-1%	-4%	-1%
Age at first breeding	0%	-7%	-1%

Aebischer and Coulson, 1990; Hatch *et al.*, 1993; Erikstad *et al.*, 1995; Pugsek *et al.*, 1995; Cam *et al.*, 1998; Frederiksen and Petersen, 1999; Wendeln and Becker, 1999), although adverse environmental conditions can decrease survival in specific years (e.g., Rattiste and Lilleleht, 1995; Harris *et al.*, 1997; Oro *et al.*, 1999).

Strong inferences show that intraspecific density-dependent competition for food has the potential to act as a regulating mechanism, especially during the breeding season. In several species reproductive performance or chick growth and quality are reported to be lower in highly populated colonies (Birkhead and Furness, 1985; Croxall and Rothery, 1991).

Body condition is an important parameter related to reproductive performance and output as found in wandering albatross (Weimerskirch, 1992; Weimerskirch *et al.*, 1997), blue petrel (Chastel *et al.*, 1995) and common tern (Frank and Becker, 1992; Wendeln and Becker, 1999). Impaired condition during reproduction can lead to an interruption of breeding or to brood reduction in order to avoid an increased risk of mortality (blue petrel Chaurand and Weimerskirch, 1994a, 1994b; king penguin Olsson, 1997; arctic terns, Monaghan *et al.*, 1989, 1992).

In common guillemots on Skomer, Lindner *et al.* (2000) showed a positive correlation of survival between adults and subadults of a year class, indicating that both age groups are influenced by the same factors outside the breeding season. The same tendency was found by Becker (unpubl.) in common terns (adult survival and subadult survival to age 2:  $rs=0.46$  ( $n=6$ , n.s.)). The two year classes 1992 and 1993 had lesser subadult return rates (0.24 on average) than the 4 following year classes (0.45). In addition, a lower percentage of the former

returned already at age 2 (67%/92%), and they recruited later (3 year recruits 38%/57%). These data indicate poor wintering conditions for the year classes 1992 and 1993 as well as for the adults wintering in Africa in these years.

The effects of egg size on chick survival and recruitment to the breeding population is unclear. For instance, Hipfner (2000) showed that common guillemot chicks hatching from replacement eggs with 25% less albumen had similar hatching success, fledging mass and survival to breeding age (5 years old) as first eggs. Alternatively, the opposite result was found in another study on common guillemots (Van Pelt and Monaghan, 2000). The discrepancy may be due to divergent foraging conditions among colonies. Nonetheless, egg size appears to directly affect the size of chicks upon hatching. Chicks that hatch from a larger egg have larger skeletal features (Hipfner and Gaston, 1999); however, chick growth also depends on the provisioning rates of parents.

These examples may show the dynamic interactions between the life history traits, and they underline the necessity to look to many of the relevant population parameters to be able to find the causes for population changes and influences of changes in food supply.

### 3.4 Monitoring of important life history parameters in seabirds

In Table 3.4 we have listed the life history parameters most relevant for integrated population studies in seabirds. We distinguish between different degrees of desirability to measure a specific characteristic, of the feasibility and practicability to record it (financial and/or logistic reasons; man power).

Table 3.4. Monitoring of important life history parameters.

Parameter	Desirability	Feasibility	Practicability
Population			
- population size (breeders)	+++	+++	+++
- incidence of non-breeding	+	+	-
- adult survival	+++	++	+
- subadult survival to breeding	+++	±	-
- recruitment			
% of fledglings	++	±	-
% recruits in population	+	±	±
- recruitment age	+	±	±
- immigration rate	+	-	-
- emigration rate	+	-	-
- sex ratio of the breeding population	±	-	-
- mean age of breeders	++	±	-
Reproduction and food provisioning			
- reproductive success	++	++	++
- laying date	+	++	++
- clutch size	+	+++	++
- egg size	+	+++	++
- chick growth rate	+	++	+
- mass of fledglings	+	++	+
adult nest and brood attendance; provisioning rate	++	++	±
- degree of kleptoparasitism	+	++	±
Body condition of breeders			
	+	+	+

+++ highest; ++ high; + moderate; ± more or less; - low

In this report we concentrate on the parameters of highest desirability, adult survival and subadult survival to breeding. The review of the techniques to study all the parameters listed, however, would require more time and space than is available here.

### 3.4.1 Adult survival

#### 3.4.1.1 Evidence that adult survival can vary as a function of food abundance

Adult survival rate of breeding black-legged kittiwakes in Shetland shows a high variation, from 0.53 to 0.98 for both males and females (Oro and Furness, 2002), even higher than that recorded for the snow petrel, a seabird that breeds in extreme environmental conditions and shows high annual variability in survival and fecundity (Chastel *et al.*, 1993). That result contradicts the concept of the 'typical' seabird as having consistently high adult survival and strong buffering against environmental stress through variation in reproductive effort (Lindén and Møller, 1989; Monaghan *et al.*, 1989; Sæther *et al.*, 1993; Oro *et al.*, 1999). In fact, models indicate that much of the variation in Foula black-legged kittiwake adult survival rates can be attributed to measured environmental factors. Adult survival rate was higher in years when adult body mass towards the end of the breeding season was high, when 0-group sandeel abundance was high, and when breeding success of great skuas was low (Oro and Furness, 2002). Similarly, adult survival of great skuas was lower at Foula, Shetland

during years of sandeel scarcity than it was when sandeels were in abundant supply (Catry *et al.*, 1998).

It is well known that when availability of food during breeding and especially during chick rearing is low kittiwakes may fail to breed successfully because they are beyond the limits of their buffering capacity (Walsh *et al.*, 1991; Danchin, 1992; Hamer *et al.*, 1993). However, accurate measures of food availability and their inter-year variations are very difficult to obtain in natural conditions, and the influence of this environmental factor on adult survival is difficult to test. Of the two published studies on the effects of food availability on gull adult survival (Pons and Migot, 1995; Oro *et al.*, 1999), neither reported any relationship between the two parameters. However, adult survival of black-legged kittiwakes in Shetland was significantly correlated with the abundance of 0-group sandeels, although it was independent of 1+ group sandeel abundance (Oro and Furness, 2002). There is a negative correlation between 0-group recruitment and 1+ group abundance of sandeels in the large North Sea stock, but no significant correlation between 0-group recruitment and 1+ abundance in the Shetland stock (Furness, 1999). Black-legged kittiwake breeding success in Shetland correlates with 1+ sandeel abundance and is independent of 0-group abundance, the opposite of the relationship for adult survival. However, these contrasts are not unexpected. The 0-group sandeels do not recruit until midsummer, but remain available to black-legged kittiwakes during July and August when 1+ sandeels

tend to remain buried in the sand and so have become unavailable to surface-feeding seabirds. Results suggest that 0-group sandeels probably play an important role for adult black-legged kittiwakes to replenish body reserves depleted during breeding. Thus, adult survival would depend not only on the environmental conditions during breeding and the reproductive effort associated (see above), but also on those occurring just after breeding. In fact, several studies showed that adult mortality of black-legged kittiwakes and of gulls in general probably occur mostly after the breeding season (Coulson and Wooller, 1976; Pugsek, 1987; Reid, 1987; Rattiste and Lilleleht, 1995; Pugsek *et al.*, 1995). Conversely, abundance of 1+ sandeels, which are the main food of breeding black-legged kittiwakes from April to June, affects breeding success of black-legged kittiwakes but 0-group sandeels apparently recruit too late to do so even though they are fed to chicks (Galbraith, 1983; Harris and Wanless, 1990, 1997; Furness, 1999).

#### 3.4.1.2 Adult body condition and survival

It has been suggested that a primary cause of adult mortality in gulls may be the loss of body mass (body condition) as the breeding season progresses (Coulson *et al.*, 1983; Pugsek, 1987, 1990). However, few studies have assessed properly the costs of reproduction associated to adult survival in gulls and their results are controversial (Reid, 1987; Pugsek and Diem, 1990; Cam *et al.*, 1998; Golet *et al.*, 1998). Golet *et al.* (1998) and Golet and Irons (1999) showed that chick rearing caused a reduction in body condition and fat stores of breeding black-legged kittiwakes and led to a lower adult survival rate than in birds in which reproductive effort was experimentally reduced by removing their chicks. However, individuals which had produced a replacement clutch and raised the second brood successfully, no negative effects on survival rates were detected in Brunnich's guillemots (Hipfner, 2000) and in common terns (Wendeln *et al.*, 2000).

#### 3.4.1.3 Effect of predation on adult survival rate

Black-legged kittiwake adult survival rate at Foula was reduced in years when great skua breeding success was high (Oro and Furness, 2002). Most killing of kittiwakes by great skuas occurs when great skuas have chicks to feed (Furness, 1987a; Hamer *et al.*, 1991; Phillips *et al.*, 1999), and it is therefore not surprising that the survival rate of adult black-legged kittiwakes was reduced in years when great skuas have more chicks. During years of great skua reproductive failure, the food requirements of the skua population will be much smaller as some birds that failed early will have dispersed, others will remain relatively inactive in the territory. In contrast, in years when chick survival is high despite low sandeel abundance, great skuas will have to work particularly hard in order to meet their chicks' food requirements and it is known that this involves increased killing of black-legged kittiwakes (Heubeck *et al.*, 1997, 1999; Ratcliffe *et al.*, 1998). There is a strong correlation between sandeel (%) in great skua chick regurgitates and productivity of great skuas (Spearman rank correlation  $r_s$

= 0.915,  $n = 12$ ,  $P < 0.001$ ). Sandeel availability has been low for great skuas during the whole study period, but when it was extremely low, breeding failure would mean that the food requirements of the great skua population would be much smaller as some birds that failed early will have dispersed, others will remain relatively inactive in the territory. In contrast, when breeding success of great skuas was higher because sandeel abundance was moderate, the extent of predation on black-legged kittiwakes would have been increased to meet the energy demand. If sandeel abundance were to return to the high levels prevailing in the 1970s, we would anticipate a low level of predation by great skuas on black-legged kittiwakes despite high great skua breeding success since the skuas would be able to meet their requirements easily by feeding predominantly on sandeels. This was the situation prevailing in the 1970s (Furness, 1987).

#### 3.4.1.4 Practicability

To calculate adult survival, local recordings (resighting, retrapping) are necessary year by year. Very large samples are required to detect a significant change in the survival rate. For instance, in the wandering albatross for a sample of 1000 birds the standard error of an annual survival estimate, assuming a 100 per cent recapture rate, is 0.8 per cent per annum (Croxall and Rothery, 1991). For such long-lived birds a consistent reduction in survival of 1–2 per cent per year can be highly significant for the population. Also accurate estimates of survival in one year cannot be made until a further 2 to 3 years have elapsed, because of the extent to which birds are known to show intermittent breeding.

#### 3.4.2 Subadult survival to breeding

Estimation of juvenile survival also needs large samples of birds. In species with long-deferred sexual maturity there will be a considerable delay in obtaining results (e.g., Harris *et al.*, 1992). For some species, this may be short-circuited by obtaining estimates of survival from the birds that attend the colony in the years prior to that of recruitment.

Variation in the survival of subadults is much greater than in that of adults. In the common tern, adult survival was 0.90 (0.87–0.97,  $n=7$  years, coefficient of variation: 3.8%), subadult survival 0.38 (0.22–0.48,  $n=6$  year classes, coefficient of variation=37.5%; measured as subadult return rates, Wendeln and Becker, 1998 and unpubl.). The difference suggests higher susceptibility of immature seabirds for environmental impacts like food shortage outside the breeding season, with strong influences on the population change.

A problem in estimating subadult survival is that only local recruits can be covered. That means that return rate is measured; the percentage of emigrated birds cannot be recorded. Another problem is that subadults normally cannot be recorded with high probability before they are breeders.

### 3.4.3 Reproduction and food provisioning

Estimation of breeding success is usually very straightforward; requiring only estimates of the breeding population size, mean clutch size, and the number of chicks fledging. However, as indicated by Hunt *et al.* (1986), breeding success is not necessarily a sensitive indicator of reproductive performance, and other indices, especially those relating to provisioning rate, might be preferable. The extent to which breeding success reflects food abundance seems to vary among seabird species, and this variation is somewhat consistent. For example, northern fulmars and common guillemots tend to show about the same breeding success in all colonies under almost all conditions of food abundance, weather and other environmental variation. At the other extreme, tern breeding success is highly variable from colony to colony and from year to year, being very severely affected by food abundance, but also by weather, disturbance, and predation impact (e.g., Becker, 1998). It may be possible to select those seabird species in which breeding success relates to food abundance but is relatively unaffected by weather or predation. For example, in the kittiwake, selection of colonies that are not affected by predators permits monitoring of breeding success as a function of food abundance (Furness, 1999).

#### 3.4.3.1 Laying date

Food availability can lead within populations to variation in the date of egg laying in a population among years (Birkhead and Harris, 1985; common tern, Becker, 1996). Environmental conditions that influence food abundance or accessibility, such as the extent of sea ice or stormy weather, may also be negatively correlated with lay date.

#### 3.4.3.2 Clutch size

The number of eggs laid depends on food availability in the breeding season (terns: Monaghan *et al.*, 1992, Frick and Becker, 1995; Becker, 1998). For instance, the mean clutch size of herring gulls declined from 2.8–2.9 eggs per clutch in years of high food availability to 2.7 in a low food year (Pons, 1992). The sex ratio of the clutch also varies with food supply in sexually dimorphic species. For instance, females of lower body condition tend to bias the sex ratio of their clutch towards the sex that has a smaller body mass or lower growth rates (Nager *et al.*, 2000; Kalmbach *et al.*, 2000). This presumably increases the number of offspring parents can raise successfully (reproductive success). There is also a balance between the number of eggs produced and the size and/or quality of each egg (Birkhead and Harris, 1985). Species that lay a single-egg clutch cannot vary the size of their clutch when food availability varies but instead vary egg size.

#### 3.4.3.3 Egg size and quality

Besides clutch size, egg size can also decline under extremely low food availability (Pons, 1992). In a year of

low food availability, lesser black-backed gulls given additional fish had larger clutch sizes and larger eggs than control birds (Bolton *et al.*, 1992). Species that lay a single-egg clutch can regulate their reproductive effort by varying egg size or the amount of resources allocated to the egg (Birkhead and Harris, 1985). Considerable variation in egg size (25–30%) occurs among and within auk populations (Birkhead and Harris, 1985). This reflects variation in female mass or body condition (Birkhead and Harris, 1985) and, thus, egg size is indirectly affected by food availability through adult body condition.

#### 3.4.3.4 Adult nest and brood attendance, and provisioning rates

The rate of provisioning depends on food availability and again varies considerably among and within populations and years (e.g., Gaston, 1985). Some seabirds have flexible time and energy budgets during the breeding season and are thus able to maintain provisioning rates under moderate declines in foraging conditions (e.g., Burger and Piatt, 1990). When foraging conditions decline further, however, parents become unable to maintain provisioning rates (Uttley, 1992; Uttley *et al.*, 1994). Some species can also alter the resting time at sea while breeding (e.g., Monaghan, *et al.*, 1994) but this does not allow adults to maintain provisioning rates. This indicates the importance of non-foraging time in other critical activities (e.g., digestion).

Many species of seabirds are unable to alter time budgets either at the colony or at sea and, thus, lower food availability results in reduced chick growth. Whether a species is capable of time-buffering is shown by cross-fostering and twinning experiments, or experimentally increasing the clutch size (Birkhead and Harris, 1985).

#### 3.4.3.5 Growth rates of chicks

Growth rates depend on food availability during breeding (Gaston, 1985; Mlody and Becker, 1991; Klaassen *et al.*, 1992). In common terns breeding in the Wadden Sea, there is a close relationship between the supply of young herring and the growth rate (Greenstreet *et al.*, 1999). This occurs because parents rank their own survival over the survival of their chicks in any given year and, thereby avoid jeopardising their own future reproductive potential.

Chick growth will reflect foraging conditions especially in species that cannot alter their time budgets (some procellariiformes, terns). Many procellariiformes alternate between foraging trips to maintain their own body condition and provisioning trips (Chaurand and Weimerskirch, 1994b; Weimerskirch *et al.*, 1997; Weimerskirch, 1998). The parent's body mass appears to play a central role in such foraging/provisioning decisions.

#### 3.4.3.6 Mass of fledglings

Fledging mass depends on food availability during the breeding season, varies between years and reflects the amount of energy reserves of fledgling and, thus, how long a bird can withstand starvation. Therefore, mass at fledgling should be positively correlated with the probability of recruitment to the breeding population. Becker (1999) and Becker *et al.* (2000) showed for common terns and Hipfner (2000) for common guillemots that higher body mass at fledging resulted in an earlier age of recruitment into the breeding population and a higher probability of being resighted at the colony. Davoren and Montevecchi (2000) also suggested that a common guillemot colony with a stable population had lower chick condition (mass/wing length) than a colony with an increasing population growth. Other studies on common guillemots that employed ringing techniques, however, found no relationship between fledging mass and the probably of being resighted at the colony as prospectors or recruits (Hedgren, 1981; Harris *et al.*, 1992). This discrepancy has led researchers to suggest that characteristics of fledglings other than mass (e.g., wing length, wing loading) are important to the survival of common guillemots as they depart the colony and in their first year at sea (Hipfner and Gaston, 1999).

#### 3.4.3.7 Degree of kleptoparasitism

Kleptoparasitism (or predation) at the colony may determine provisioning rates of chicks depending on the extent and consistency (Veen 1977, Furness 1987b; Gorke, 1990; Ludwigs, 1998). Marked intraspecific kleptoparasitism in a colony indicates food shortage (Ludwigs, 1998). Finney *et al.* (2000) showed that provisioning rates of Atlantic puffin parents were lower in areas where gulls were present versus areas where gulls were absent; however, this did not affect the growth rates of chicks or breeding success.

#### 3.4.3.8 Reproductive success

Reproductive success appears to be the least sensitive parameter in reflecting prey availability or foraging conditions because it is influenced by other factors like predation, weather or flooding. Additionally, parents are able to buffer provisioning rates, chick growth and mass of fledging. Reproductive success is positively correlated with the quality and age/experience of the breeders (e.g., Coulson and Thomas, 1985a, 1985b; Wendeln and Becker, 1999; Weimerskirch, 1992). In years of severe prey scarcity (e.g., off Peru during ENSO events), however, seabirds may abandon breeding attempts.

#### 3.4.4 Body condition of breeders

Body condition of adults is an important parameter linked with other life history characteristics such as reproductive performance, adult survival (see Sections 2.3 and 2.4.1) or with recruitment (e.g., Porter and Coulson, 1987; Becker, 1999). Body mass can be measured easily after catching the bird, mainly possible

during incubation or chick rearing. However, the stage of the breeding cycle and of the incubation shift of the weighed bird should be known and taken into account, as they influence body mass (e.g., Frank and Becker, 1992; Wendeln and Becker, 1996; Golet *et al.*, 1998; Golet and Irons, 1999). In some studies adult weight has been recorded remotely and automatically (Sibly and McCleery, 1980; Monaghan *et al.*, 1989; Frank and Becker 1992; Wendeln and Becker, 1996).

### 3.5 Possibilities of integrated population monitoring in seabirds

The use of seabirds as monitors of the marine environment is becoming increasingly widespread. Monitoring of parameters other than population size, breeding success and performance is usually regarded as necessary information for interpreting changes. As is shown above, the relevance of the various population parameters will depend on the life-style of the species concerned. For extreme K-selected species the order of parameters measured may be adult survival, subadult survival and breeding success.

The different population parameters integrate environmental effects over different time spans. Different lag times before the effects can be measured and different accuracies with which effects are likely to be detected have to be considered (Croxall *et al.*, 1988).

Logistically costly, long-term monitoring of seabird populations is rare (Nisbet, 1989; Wooller *et al.*, 1992). Nevertheless, there are some good examples of integrated long term studies: wandering albatross (Weimerskirch and Jouventin, 1987; Croxall *et al.*, 1992; Weimerskirch, 1999), short-tailed shearwater (Wooller *et al.*, 1989; Bradley *et al.*, 1991, 1999), black-legged kittiwake (Coulson and Thomas, 1985b; Porter and Coulson, 1987; Thomas and Coulson, 1988); Atlantic puffin (Harris and Wanless, 1991; Harris *et al.*, 1997); great skua (Furness, 1987a); common tern (Becker and Wendeln, 1997; Becker *et al.*, 2000). Such studies require special sites which allow good access, a long term perspective, an engaged team of researchers, the necessary financial basis and effective data management (Bradley *et al.*, 1991). In addition the use of new field techniques, e.g., field readable rings or remote and automatic recording of subadults and adults by passive transponders (Becker and Wendeln, 1997; Becker *et al.*, 2000) may help to increase the efficiency of integrated population monitoring in seabirds.

### 3.6 Conclusion

This is not an exhaustive review of the sensitivity of seabird populations to changes in life history parameters, even with regard only to influences of food abundance and hence to possible effects of fisheries. However, it is clear that different species of seabirds respond differently depending on their particular life history strategies, and that the ideal programme of seabird monitoring would consider not only breeding numbers and breeding



success, but would include attention to other aspects of seabird life history parameters, especially adult survival and recruitment rate. There is a need for further review of literature, demographic models and field techniques available in order to develop a better understanding of the ways in which seabird monitoring programmes might be improved by taking these parameters into account.

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## 4 The alteration of the composition of seabird communities by fisheries

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### 4.1 Introduction

There are direct and indirect effects of fisheries on seabirds. Most direct effects involve killing of seabirds by fishing gear or by culling, while indirect effects mostly work through the alteration in food supplies of birds (Tasker *et al.*, 2000). Seabird mortality in long-line fisheries or in other fishing gear can lead to drastic population declines and may bring certain vulnerable species to the brink of extinction, but even the (persistent) disturbance of birds due to some fishing activities, such as some aquacultural pursuits, may negatively affect seabird numbers (Davidson and Rothwell, 1993). On the other hand, seabirds may also benefit from fisheries, because many fishing activities increase the food supply or enhance the availability of prey for seabirds. The practice of discarding unwanted fractions of a commercial catch is clearly beneficial for scavenging seabirds (e.g., Camphuysen *et al.*, 1995). Furthermore, major shifts in fish stock composition, for example due to overfishing of large predatory fish, have led to a (relative) increase in smaller fish, suitable for consumption by seabirds. The most prominent fishery effects have recently been summarised by Tasker *et al.* (2000). This review concentrates on alterations in seabird communities that may have been influenced by fisheries.

Proving the scale of fisheries effects can be difficult due to confounding and interacting combinations with other anthropogenic effects such as pollution, culling, hunting, and disturbance or with more natural oceanographic factors that can influence prey availability. In fact, fishery effects can be masked completely in seabird populations that are subject to major shifts due to these and other factors. Moreover, the life history patterns of seabirds can also buffer them to some extent from anthropogenic influences associated with fisheries. The present review, shows examples of major shifts in numbers of seabirds breeding around the North Sea and in the species composition of certain breeding areas and try to explain these shifts. Examples are also provided of clear-cut fishery effects that are obvious and (directly) affect large numbers of birds, but that for some reason do not show up on the population level. First, what is meant by “the composition of seabird communities” is defined.

### 4.2 Seabird communities

Two main types of seabird community, can be distinguished. Seabirds may compete for or share breeding sites (*nesting community*) because of overlapping nesting habitat requirements or because they share a feeding area. Seabirds that share the same nesting area may however forage in widely separated areas and have a completely different prey spectrum, and some seabirds that nest far apart may interact while foraging in

overlapping areas and perhaps even directly compete for prey (*feeding community*). While competition for nesting sites and for prey are perhaps the most thoroughly studied aspects of inter-specific interactions between seabirds, the activities of one species may also enhance the feeding opportunities of another. As a result, a shift in a seabird community through a change in the abundance of one species (for example influenced by fisheries) may affect another species simultaneously, even though that fishery plays no obvious role in the feeding ecology of the latter. For example, social feeding pursuit diving common guillemots and razorbills significantly enhance the feeding opportunities of surface plunging black-legged kittiwakes (Camphuysen and Webb, 2000). Hence, a mass mortality of auks in gill-nets could indirectly have a negative impact on black-legged kittiwakes even although they themselves do not get entangled. Also, seabirds sharing the same breeding community (e.g., terns and gulls or northern gannets and common guillemots), but differing in feeding ecology, may compete for nesting space, so that the foraging success of the one species may indirectly affect the nesting space of the other (e.g., Howes and Montevecchi, 1993).

So, while species in a given area may share a certain nesting community, a feeding community, both or neither, any aspects affecting a given seabird species may work through community interactions to affect other species indirectly. As the effects of fisheries on seabirds are usually ambiguous, this study is limited to the examination of clear trends in population levels, downward or upward, on a moderately large spatial scale (e.g., SE North Sea, Shetland area, British Isles, etc.) for groups of species that share particular prey and foraging techniques. Such changes should lead also to shifts in relative abundance and species composition within seabird communities.

### 4.3 The difficulty of detecting fishery effects

From the above it is clear that fisheries effects may act directly on a species or indirectly through the wax or wane of either a competitor or a “co-operator” in mixed feeding systems. All other factors being equal, the numbers of seabirds breeding or feeding in a given area should reflect the carrying capacity of that region in terms of amount of food available. Prey availability is not the same as the size of prey stocks present, for several factors influence the accessibility of prey for seabirds and the profitability (in terms of intake rates achievable) of a given area. Prey availability may fluctuate independently of prey stocks and prey availability is also different for species using different foraging and feeding techniques.

Fishery effects can be strong, so that populations grow or decline, but fishery effects are often indirect and may be subtle, for instance the reproductive output, activity patterns or time budgets of birds may alter. Fishery-induced increases in food supply often result in an increase in 'secondary prey', that is not preferred when 'normal' prey is sufficiently abundant (e.g., Furness and Hislop, 1981). A clear-cut negative effect of fisheries, such as the bycatch of large numbers of seaduck, may be very hard to quantify, because a complete census of birds that 'disappear' to breed in vast Russian and Scandinavian forests is simply not feasible. Perhaps most important, however, is that fishery-induced changes in fish stocks are often very difficult to distinguish from natural variation or environmental impacts on fish, so that the next step, an effect on fish predators (such as birds), will become even more obscure.

#### 4.4 Direct effects on communities: seabirds drowning in gill nets

##### 4.4.1 Northern Norway

Among seabirds in the Northeast Atlantic, the pursuit-diving auks, common guillemots, razorbills and Atlantic puffins are the most vulnerable to entrapment in gillnets and other fixed fishing gear. Gillnets set for cod off

northern Norway killed very large numbers of Brunnich's guillemots and common guillemots between at least 1965 and 1985 (Strann *et al.*, 1991). In early spring 1985 the estimated kill of both species totalled was greater than 100 000 birds. In this same area, summer driftnet fisheries for salmon used to regularly drown thousands of local breeding birds (Vader and Barrett, 1982; Strann *et al.*, 1991). Numbers of common guillemots at Hjelmsøy declined from 220 000 in 1965 to 10 000 individuals by 1985. Brunnich's guillemots at this colony declined from >2000 to 220 in the same period (Vader *et al.*, 1990). The fishery has been closed since 1989 to conserve salmon stocks, but the auk populations had reached a critically low level and have so far not shown any signs of recovery (Lorentsen, 1999). Common guillemot populations on Kharlov, Hornøya, and Syltefjord were not affected by the gill-net fisheries, because the foraging areas were located beyond the sea area with gill nets, and two of these populations increased steadily between the 1970s and mid 1980s (Figure 4.1). Recoveries of common guillemots ringed as chicks on Helgoland since 1912 revealed an increase of the proportion killed in fishing gear from less than 4% until 1969 to 42% in the period 1989–1994. However, the mortality rate remained virtually stable (Hüppop, 1996).

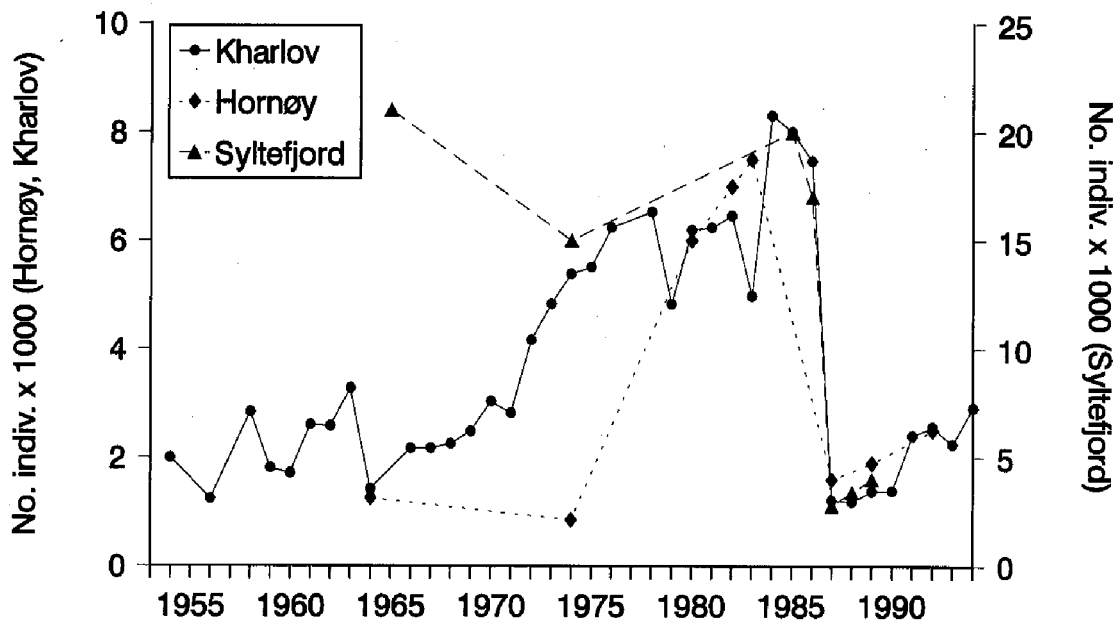


Figure 4.1. Changes in the total numbers of breeding common guillemots *Uria aalge* (individuals) counted on Kharlov, Hornøya, and Syltefjord since 1955 (Krasnov and Barrett, 1995).

#### 4.4.2 Baltic

In the Baltic Sea, seaduck such as common eider, black scoter and long-tailed duck, divers, grebes and auks (mainly common guillemot and razorbill) are affected mostly by fixed nets in coastal or shallow offshore waters. Certainly in the case of the seaducks, very large numbers can be affected and locally, this may affect substantial proportions of wintering seaduck communities (Kirchhoff, 1982; Kies and Tomek, 1990; Meissner, 1992; Stempniewicz, 1994). With a total of over 7 500 000 seaduck wintering in the Baltic (Durinck *et al.*, 1994), however, the overall seaduck community effects may be relatively small but are most certainly very difficult to detect.

#### 4.4.3 UK and Ireland

Seabird bycatch impacts around UK and Ireland tend to be of a localised nature, diluting any possible population effect (Tasker *et al.*, 2000). Studies around Wales (Thomas, 1992) and Scotland (Murray, 1993; Murray *et al.*, 1994) found no evidence of widespread impact, with at best 'hot spots' in mortality where nets were set immediately beside seabird colonies.

#### 4.5 Indirect effects on seabird communities

##### 4.5.1 The numbers of breeding *Larus* gulls and terns at the southeastern North Sea

In the southeastern North Sea, most European species of gulls *Larus* spp. have greatly increased in numbers and/or established new breeding colonies during most of the 20th century (Spaans 1998a, 1998b; Garthe *et al.*, 2000). In several of these species, most notably the black-headed, herring and lesser black-backed gulls, population increases have at times been very drastic. Such 'explosions' in the population have however only occurred in The Netherlands when control measures relaxed or when e.g., systematic eggging in dune areas had ceased. It has been widely assumed that the (human influenced) food availability of these birds has improved over time (e.g., discards, rubbish tips, littered beaches). Owing to control measures, gull numbers have been kept artificially low for most of the last century so that the carrying capacity of the area may have been much greater than has so far been reflected in numbers of nesting gulls. At present, several breeding populations of gulls are under pressure from predation by mammals and/or disturbance due to increased tourism. The main aspects influencing the increase and decrease of nesting gulls in the southeastern North Sea are summarised in Table 4.1.

Table 4.1. Main factors influencing the increase and decrease of nesting gulls in the SE North Sea.

Factor		Effect on population	Scale
Exploitation	eggging	negative	widespread, mainly early 20th century
Persecution	culling	negative	until late 1970s in Germany, late 1960s in Netherlands
	killing adults	strongly negative	Localised
Predation	clutches, chicks	negative	widespread on mainland since late 1970s
	adults	strongly negative	Occasional
Prey availability	rubbish tips	positive	widespread, mainly 1960s–1980s, most covered up since
	littered beaches	positive	widespread since 1970s, mainly common, black-headed and herring gulls
	discards in commercial fisheries	positive	progressive increase throughout 20th century, steep increase since 1960s, perhaps decline since late 1980s.
	eutrophication of coastal waters	positive	increase after 1945 and significantly in 1970s, phosphate decreased in 1980s, but this did not decrease N/P ratio, nitrogen levels remained high.
	agricultural development	positive	gradual change throughout the 20th century, particularly strong since the 1960s
	overfishing of small fish	negative	increase throughout 20th century, shortly interrupted during 2nd World War leading to recovery of fish stocks
	overfishing of predatory fish	positive	increase throughout 20th century, shortly interrupted during 2nd World War leading to recovery of fish stocks
Breeding habitat	disturbance	negative	increased tourism, particularly since 1960s since
	national parks	positive	since mid-1980s in Germany
Chemical pollution	adults, clutches, chicks	negative	1950s–1980s

Table 4.2 is an attempt to quantify the possible impact of each of these factors on gulls in the southeastern North Sea. It appears that gull populations must have been under severe pressure during the first half of the 20<sup>th</sup> century, mainly as a result of exploitation and persecution. The drastic increases in numbers could be anticipated from the 1960s onwards, on, in response to the relaxation of persecution and increased food supplies from commercial fisheries, rubbish tips and agriculture. This situation is apparently continues, although predation and disturbance are increasingly important as factors suppressing these trends.

The overall population trends of gulls observed in Germany and The Netherlands do in fact match up quite well with this pattern, although there are differences between species that require further explanation (Figures 4.2–4.5). Black-headed and herring gulls have dominated the area during most of the 20th century, with colonisation of new areas by common and lesser black-backed gulls and a very few great black-backed gulls in recent years.

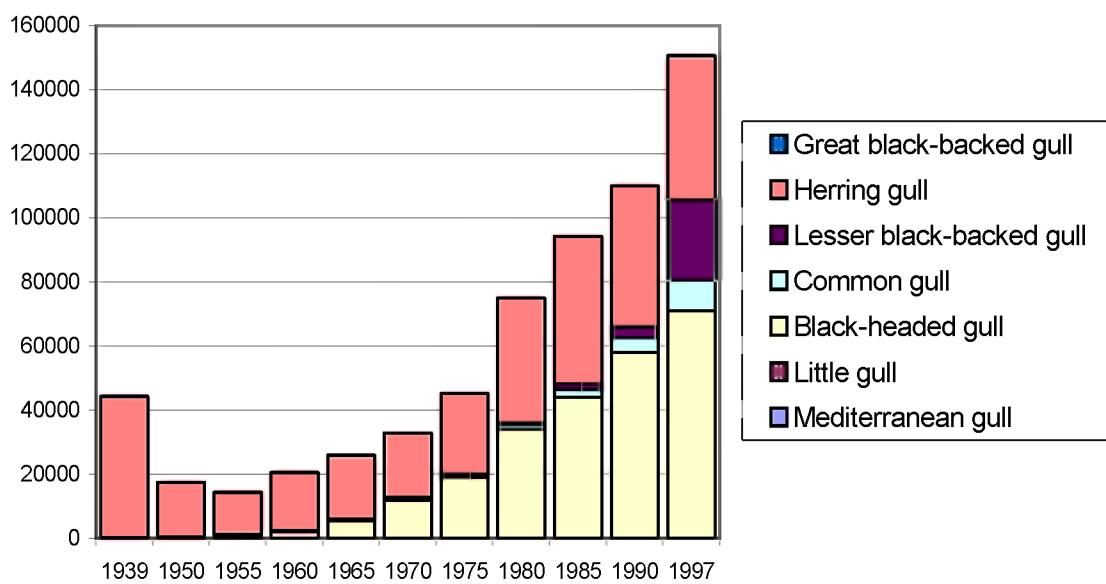


Figure 4.2. Population trend and species composition of gulls nesting in coastal areas of the North Sea in Germany (data after Schulz, 1947, and Garthe *et al.*, 2000). Y-axis represents numbers of breeding pairs.

Table 4.2. Possible impacts of factors listed in Table 4.1 on gulls in SE North Sea.

Decade	Exploitation		Persecution		Predation		Prey availability		
	Egging	Culling	Killing adults	Eggs Chicks	Adults	Rubbish tips	Littered Beaches	Discards	Eutrophication
1900	-	--							-
1910	-	--							-
1920	-	--							-
1930	-	--	-						-
1940	--	--	-						+
1950		--	-					+	+
1960		--	-			++	+	+	+
1970		-	-	-	-	++	+	++	++
1980		-		--	-	+	+	++	++
1990				--	-		+	++	++

Decade	Overfishing			Breeding grounds			Overall trend
	Agri-culture	Prey species	Predatory fish	Distur-bance	Nature Reserve	Pollution	
1900							---
1910		-	+				---
1920		-	+				---
1930	+	-	+				---
1940	+	-	+			-	----
1950	+					-	-
1960	+	--	++	-		--	+
1970	++	--	++	-		-	++
1980	++	--	++	-	+	-	++
1990	++	-	+	-	+		+



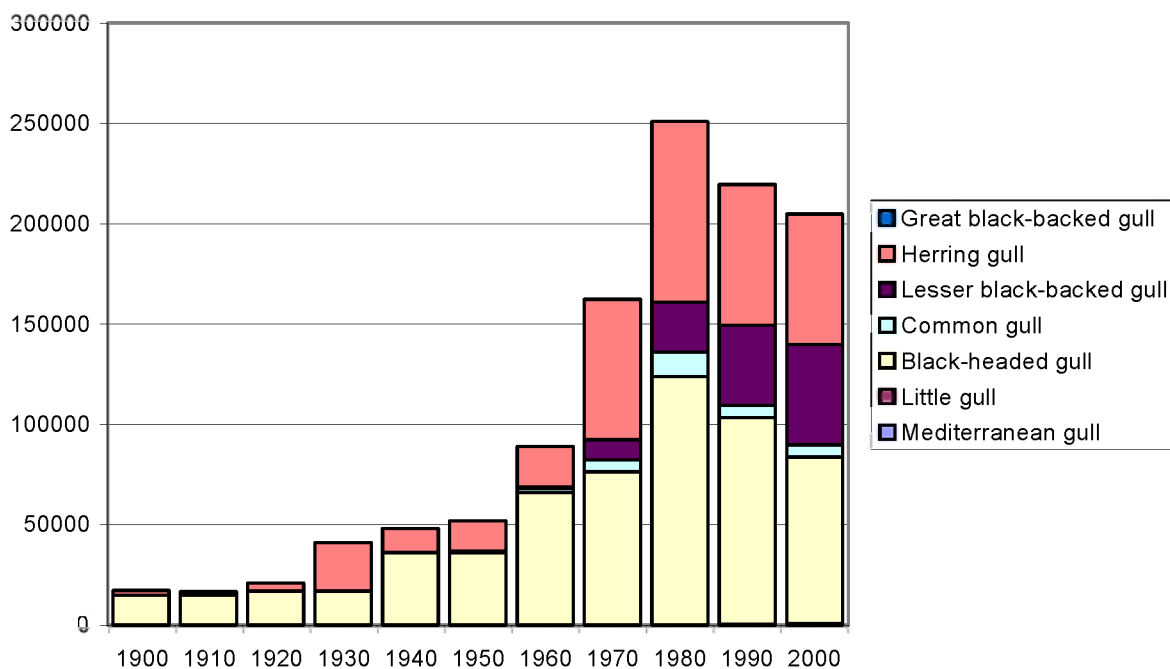


Figure 4.3. Population trends and species composition of gulls nesting in coastal areas in the Netherlands (data after Meininger and Flamant, 1998; Koks, 1998; van Dijk, 1998; Keijl and Arts, 1998; Spaans, 1998 a, and 1998b). Y-axis represents numbers of breeding pairs.

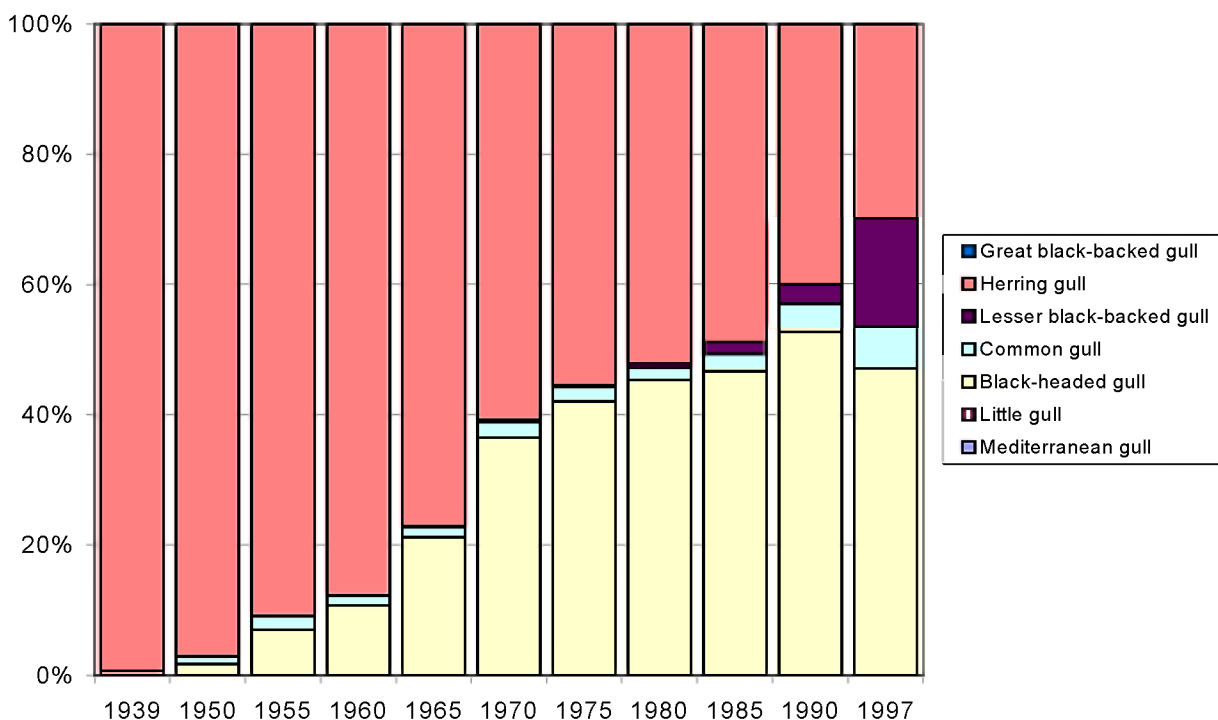


Figure 4.4. Species composition (% of all coastal nesting individuals) of gull at the North Sea coast of Germany (references as in Figure 4.2.).

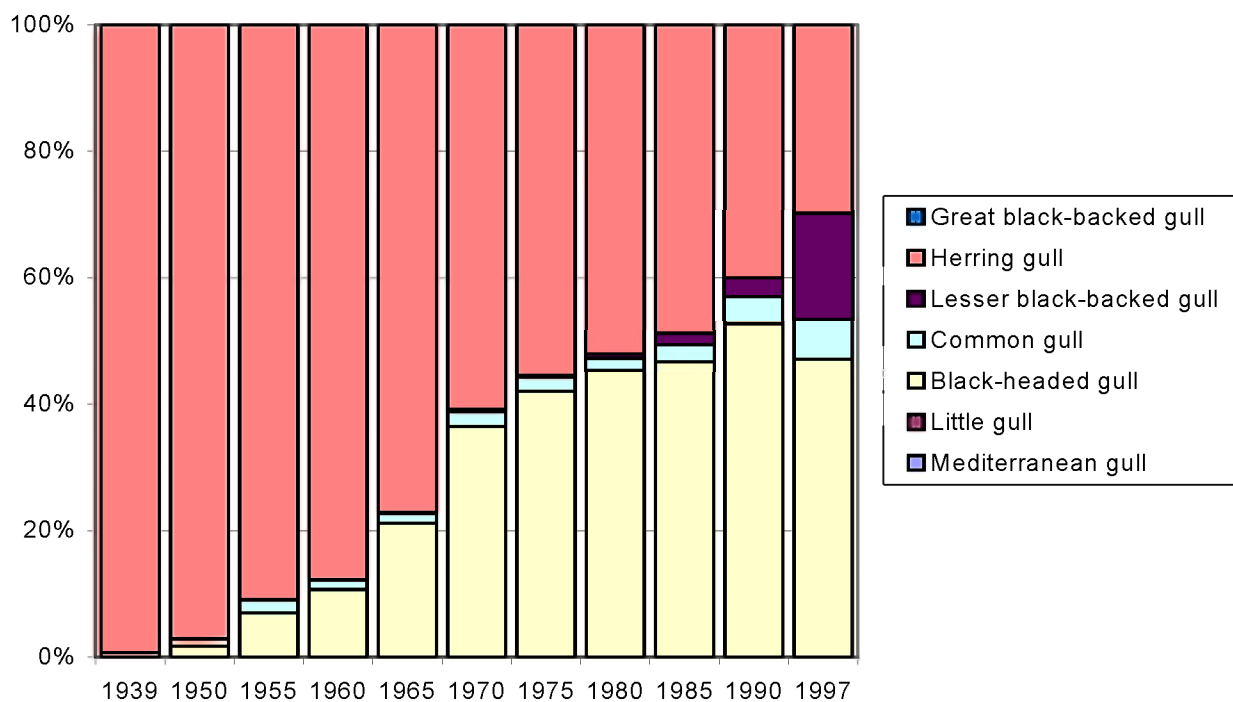


Figure 4.5. Species composition (% of all coastal nesting individuals) of gull in The Netherlands (references as in Figure 4.3.).

Perhaps the most important fishery induced impact on gull populations has been caused by the production of discards from commercial fisheries. The amounts of discards and offal in the North Sea offshore fisheries were estimated by Garthe *et al.* (1996) from published data and from unpublished statistics. In addition, Garthe *et al.* (1999) quantified discards in the coastal shrimp fisheries on the basis of calculations by Walter (1997). In total, approximately 1 million tonnes of biota is discarded every year by commercial fisheries in the North Sea. The fisheries in which discards are discharged at the highest rate are beam trawl fisheries (Camphuysen *et al.*, 1995; Garthe and Damm, 1997). The species of seabirds profiting most from discards and offal of these beamtrawl fisheries are several species of gull, most notably herring and lesser black-backed gulls (Camphuysen *et al.*, 1995). Several of the smaller species such as black-headed and common gulls are more numerous in inshore waters, notably in shallow areas such as the Wadden Sea (Garthe, 1997). Most of these gulls, however, produce more offspring in seasons when their natural prey is available and are consumed in abundance than in years when discards form the greater part of their diet (e.g., Spaans *et al.*, 1994).

Reducing the amount of discards available to scavenging seabirds might increase the predation pressure on other, non-scavenging, seabird species (Furness, 1992). On the island of Trischen (Germany) the predation and kleptoparasitism by black-headed gulls on terns is considerably higher in times when discards from the local shrimp fishery are not available (Hälterlein, 1996). Substantial effects of discard availability were observed

at least in herring and great black-backed gulls wintering on the island of Helgoland. When cod fishery discards were available close to the island, 83 to 87% of the gulls' pellets contained discard remains (70 to 73% of all pellets consisted even exclusively of these). At times when no nearby trawling took place, numbers of herring gulls dropped by up to 86% and numbers of great black-backed gulls by up to 80%. Adults of both gull species had a significantly lower body mass during periods of no fishing around the island (13 and 24% less, respectively). Effects on gull winter mortality and thus on population dynamics can be assumed (Hüppop and Wurm, 2000).

In conclusion, gulls in the southeastern North Sea have obviously benefited considerably from fishery induced changes in food supply, but the effects coincided with several important aspects either stimulating or suppressing further growth of the population. In the absence of detailed studies and with the impracticability of controlled experiments, the extent to which fishing activity has contributed to these trends are very difficult to judge. The species composition in the southeastern North Sea gull breeding community has changed considerably over time, with a tendency to much higher numbers and increased diversity in the last decades. During the 1990s in The Netherlands, numbers reached a plateau in some species, and levelled off in others, while new species colonised the country and lesser black-backed gulls were still increasing rapidly (Spaans, 1998a, 1998b). In Germany, there have so far been only signs of such a levelling off in the population of herring gulls in Niedersachsen (bordering The Netherlands; 13% decline between 1993 and 1997; Garthe *et al.*, 2000). It is

important to note that the increase in some species of gull is certainly not due to fisheries. This is best illustrated by the increase in Western Europe of Mediterranean gulls, a species with strictly terrestrial feeding habits, certainly in the breeding season, and which spends the winter in areas away from the main fisheries in southwestern Europe (Meininger and Flamant, 1998). Furthermore, most waders breeding in the Wadden Sea have increased in numbers simultaneously (e.g., Hälterlein, 1996), although they do not feed on fish and discards.

Since the tern species breeding on the Wadden Sea coast are very different in their food and foraging habits, they might offer an opportunity to use their breeding numbers as indicators of food availability. However, as with gulls, their population development is influenced by a variety of factors at a time. Both in The Netherlands and in

Germany, numbers of common and Arctic terns decreased considerably in the 1950s and 1960s (Figures 4.6–4.9.). The same holds true for the Sandwich tern in The Netherlands, whereas its numbers at the German North Sea even increased throughout this period (e.g., Glutz von Blotzheim and Bauer, 1982; Becker and Erdelen, 1987). The dramatic decrease in the 1950s and 1960s was most probably caused by contamination of the southern North Sea by chlorinated hydrocarbons from a Dutch factory (Becker and Erdelen, 1987; Südbeck *et al.*, 1998). Although the populations of common and Arctic terns recovered quickly in the early 1980s, numbers did not reach the level before World War II. In contrast, the numbers of breeding Sandwich terns in Germany did not show the pollution related decrease but instead doubled from 1939 to the late 1990s.

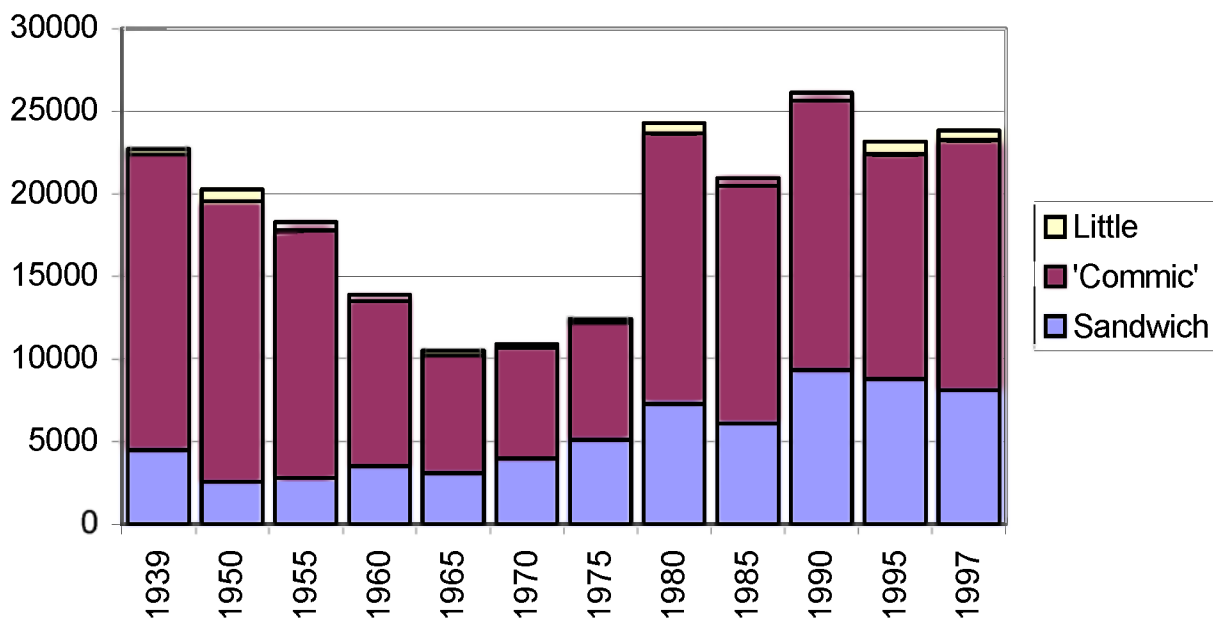


Figure 4.6. Population trends and species composition of terns nesting in coastal areas of the North Sea in Germany (data after Schulz, 1947; Becker and Erdelen, 1987; Behm-Berkelmann and Heckenroth, 1991; Hälterlein, 1996; Südbeck and Hälterlein, 1997, 1999, Südbeck *et al.*, 1998). (Commic = common/Arctic).

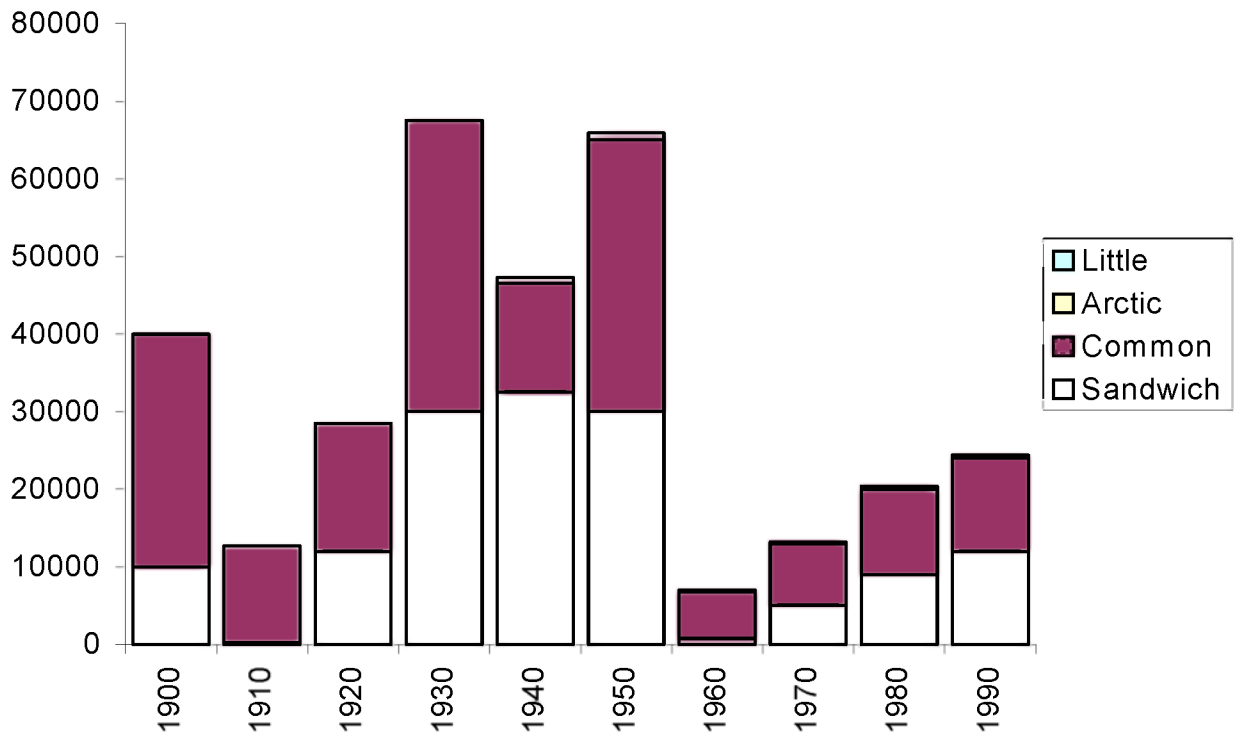


Figure 4.7. Population trends and species composition of terns nesting in coastal areas of the Netherlands (data after Brenninkmeijer and Stienen, 1992; and Stienen, and Brenninkmeijer, 1992). Y-axis represents numbers of breeding pairs.

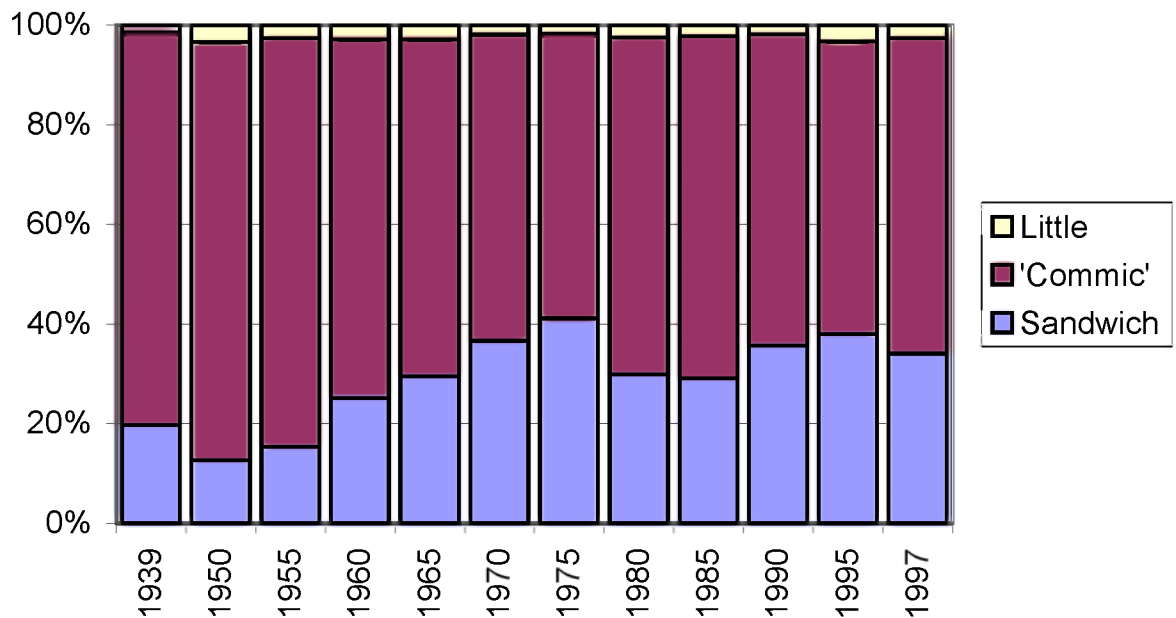


Figure 4.8. Species composition (% of all coastal nesting individuals) of tern at the North Sea coast of Germany (references as in Figure 4.6.). Y-axis represents numbers of breeding pairs. (Commic = common/Arctic).

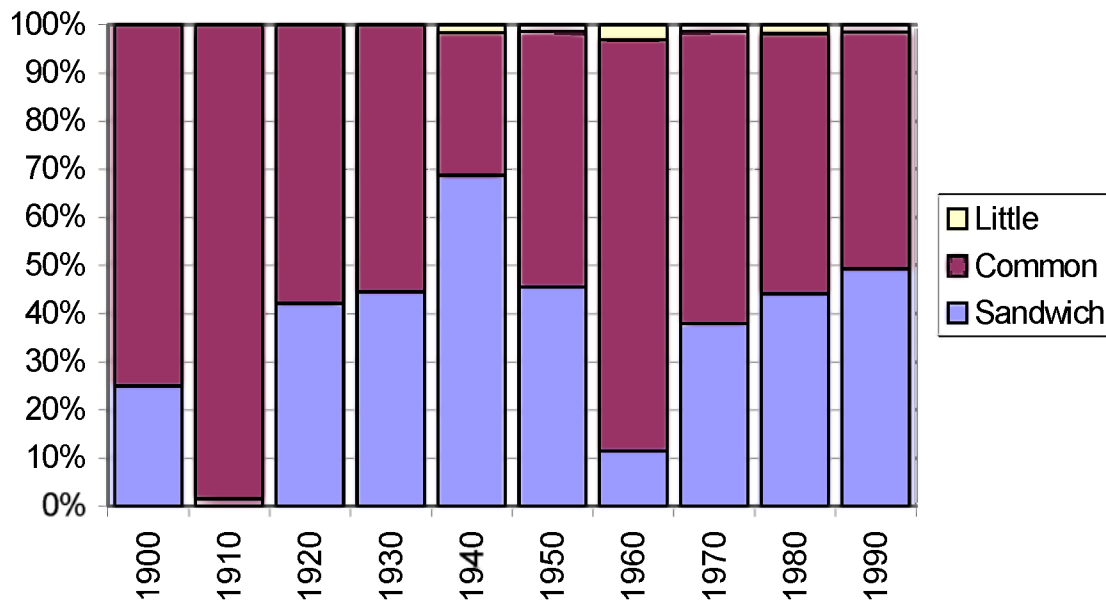


Figure 4.9. Species composition (% of all coastal nesting individuals) of tern in coastal areas of The Netherlands (references as in Figure 4.7.).

The steep increase in numbers of the common/Arctic terns at the beginning of the 1980s coincides with the rehabilitation of herring stocks in the North Sea (Dornheim and Wegner, 1998). Despite a continued improvement in herring stocks and despite the fact that the IBTS herring index is correlated with the nesting success of common terns in the German Wadden Sea (Greenstreet *et al.*, 1999), surprisingly the common/Arctic terns showed no further increase. However, the decrease in the herring index in the early 1990s was followed by a slight decrease in numbers of nesting of common terns. This supports the assumption that a reduction in food availability is responsible for the slight recent decrease (Südbeck *et al.*, 1998). In contrast, the numbers of Arctic terns breeding on the German North Sea coast remained fairly stable. This tern species has a broader food spectrum, namely it takes more crustaceans (Glutz von Blotzheim and Bauer, 1982; Niedernostheide, 1996). Thus it is likely that it was less affected by a decreasing availability of herring.

In contrast to common and Arctic terns, the German North Sea population of the Sandwich tern was virtually unaffected by changes in the herring stock, since the massive increase in Sandwich terns started about 10 years earlier than the recovering of the herring (e.g., Südbeck and Hälterlein, 1997). In the southern North Sea, this tern species almost exclusively preys on clupeids and sandeels (e.g., Glutz von Blotzheim and Bauer, 1982; Garthe and Kubezki, 1998). Hence, the increase in numbers could be related to a possible increase in sandeel stocks in the southeastern North Sea that might be caused by e.g., a reduction of the predatory fish species or by eutrophication. Once again it is impossible to prove any effects of fisheries on community structure.

#### 4.5.2 Prey stock depletion

More than one million pairs of Atlantic puffins breed in the Lofoten area (NW Norway) and along the coast adjacent to the Norwegian coastal current north of the main spawning areas of the Norwegian spring-spawning herring stock. First-year herring was an important food for these birds in the 1950s (Myrberget, 1962). In the late 1960s, this herring stock collapsed completely, as a combined effect of overfishing and poor recruitment (Jakobsson, 1985). For the next twenty years, the stock remained at a very low level. Although the very strong year class of 1983 led to a substantial increase in 1988 (Hamre, 1994), the stock was not fully recovered until the late 1990s when several good year classes recruited into the spawning population (Röttingen, 1998). During the first three decades following the collapse, the breeding success of Atlantic puffins in the large colonies at Røst in the Lofoten Islands was successful in seven season only, while in the other years most or all nestlings died from starvation (Lid, 1981; Anker-Nilssen, 1987; 1992, 1998). Atlantic puffins, as with most other seabirds, are long-lived and adapted to withstand years with poor feeding conditions, however, it is not surprising that this exceptional series of production failures have affected the number of breeding birds in the area. During the 1980s, the population of breeding Atlantic puffins at Røst declined at 13.7% per annum and levelled off at only about 35% of its previous level and although it has since remained relatively stable, a further decrease is now expected due to the annual breeding failures in 1993–1998 (Anker-Nilssen, 1998). Jensen *et al.* (1994) studied the overlaps between the distributions of fish (sandeel and sprat), three auk species (common guillemot, razorbill, Atlantic puffin) and the fisheries for

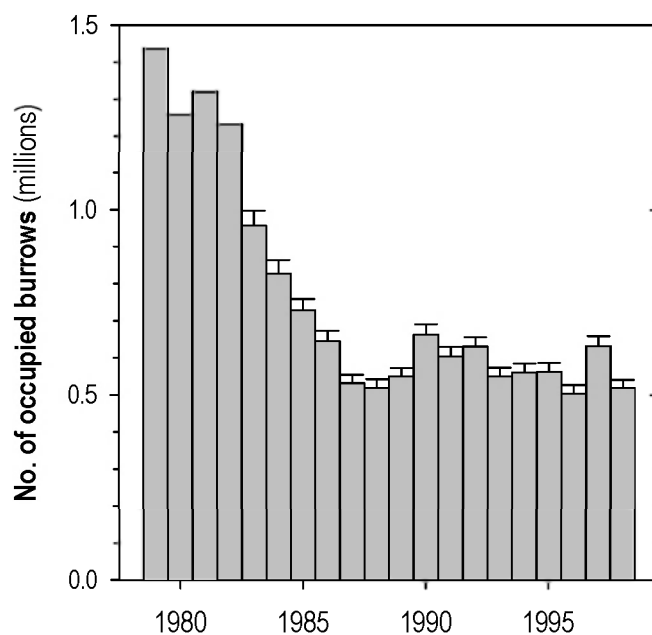


Figure 4.10. Population development (number of apparently occupied nest burrows  $\pm$  1 SE) of Atlantic puffins at Røst, North Norway in 1979–1998. Anker-Nilssen (1998).

these fish at a relatively large scale in the North Sea. There were significant positive correlations between bird and fish distribution in the third quarter of the year, with most of the association being in the northwestern North Sea. There were some associations between birds and the fisheries that indicated that at a large scale some fisheries and some birds were exploiting the same fish species in the same place, but at different times of year. An example of this was that fisheries for sprat in the southern North Sea took place mostly in the autumn, but there was greater spatial overlap between the two in winter. This study illustrated also the difficulty of comparing data collected at different scales for different purposes, and therefore highlighted a problem in demonstrating the competitive impact of fisheries on seabirds. However, fishery impacts at these harvest levels could be cumulative, and potential effects on seabird populations might be lagged in time.

In the most intensively studied seabird-sandeel interaction, certain seabirds, particularly surface feeding Arctic terns and black-legged kittiwakes, suffered a series of years with very poor breeding in Shetland in the 1980s. Birds in this area are entirely reliant on sandeels during the breeding season. This decline was again coincident with an increase in catch from local sandeel grounds. Research however indicated that fisheries were unlikely to be the cause of the decline in sandeel abundance. There was considerable fluctuation in recruitment of sandeels following a closure of the local fishery. A more likely candidate in this case was connected with the recruitment mechanisms of sandeels occurring in the area (Wright and Bailey, 1993; Wright, 1996). This case highlighted the importance of fluctuations in year class strength and in understanding the prey population structure in an area before any potential effects of fisheries can be understood.

#### 4.5.3 Depletion of shellfish stocks

In the early 1990s, the Dutch sector of the Wadden Sea was cleared of old mussel banks and 100% of the high-density cockle banks were removed (Beukema and Cadée, 1996). Common eiders and Eurasian oystercatchers suffered extra mortality in 1991, due to acute food shortages (Camphuysen, 1997). Wintering common eiders have sought refuge in later years in the coastal North Sea waters, mixing with black scoters over *Spisula* banks and competing for prey in these waters (Leopold and Dankers, 1997). Eurasian oystercatcher populations in the eastern Wadden Sea declined, largely as a result of recruitment failures of post-1990 cohorts into the breeding population (D. Heg, personal communication).

#### 4.5.4 Increases in prey stocks as a consequence of depletion of stocks of larger fish and marine mammals

An effect of the reduction in stocks of large fish by fisheries has been to reduce competition for the prey of some seabirds. Populations of diving seabirds in the North Sea and around the United Kingdom and Ireland increased dramatically during the 20th century (Lloyd *et al.*, 1991; Table 4.3). Some of this, as mentioned before, may have been a response to the cessation of seabird hunting at the end of the 19th century, but the duration of this increase indicates other causes. The increased proportion of small fish in the North Sea in comparison with the Faroe Bank (Pope and Knights, 1982) and Georges Bank (Pope *et al.*, 1988) was attributed to a possible direct consequence of the more intense harvesting in the North Sea. Sherman *et al.* (1981) considered that sandeel stocks had increased in the North

Sea and in the western Atlantic shelf seas as a response to reduced competition with herring and mackerel. In the Barents Sea, an increase in capelin may have occurred through over-fishing of their main predator, herring (Hamre, 1988, 1991; Gjosæter, 1998). It is possible that the increases in black-legged kittiwakes and common guillemots in the southern Barents Sea between 1960 and

1990 can be attributed to this superabundance of capelin in the region (Figures 4.10 and 4.11). However, there is no compelling evidence for this relationship in the northwest Atlantic (Carscadden and Nakashima, 1997), perhaps due to the complexity of direct and indirect effects in marine food webs (Lavigne, 1996) or to oceanographic effects (Frank *et al.*, 1996).

Table 4.3. Changes in numbers and relative abundance of seabirds breeding in the United Kingdom and Ireland between the early 1970s and the mid 1980s (Lloyd *et al.*, 1991). Only those species with relatively reliable census figures are included.

Species	1969–1971 population	1969–1971 % of all birds	1985–1987 population	1985–1987 % of all birds	population change
Northern fulmar	309 000	14.2	571 000	20.5	+
Northern gannet	138 000	6.3	188 000	6.8	+
Great cormorant	8 000	0.4	10 000	0.4	+
European shag	34 000	1.6	47 000	1.7	+
Arctic skua	1 000	>0.1	3 000	0.1	+
Great skua	3 000	0.1	8 000	0.2	+
Black-headed gull	75 000	3.4	84 000	3.0	+
Common gull	13 000	0.6	16 000	0.6	+
Lesser black-backed gull	50 000	2.3	64 000	2.3	+
Herring gull	335 000	15.4	191 000	6.9	-
Great black-backed gull	23 000	1.0	23 000	0.8	±
Black-legged kittiwake	447 000	20.6	544 000	19.6	+
Sandwich tern	12 000	0.5	18 000	0.7	+
Roseate tern	2 500	0.1	500	0.0	-
Common tern	15 000	0.7	15 000	0.5	±
Arctic tern	52 000	2.4	80 000	2.9	+
Little tern	2 000	0.1	3 000	0.1	+
Common guillemot	550 000	25.3	806 000	29.0	+
Razorbill	103 000	4.7	182 000	4.4	+

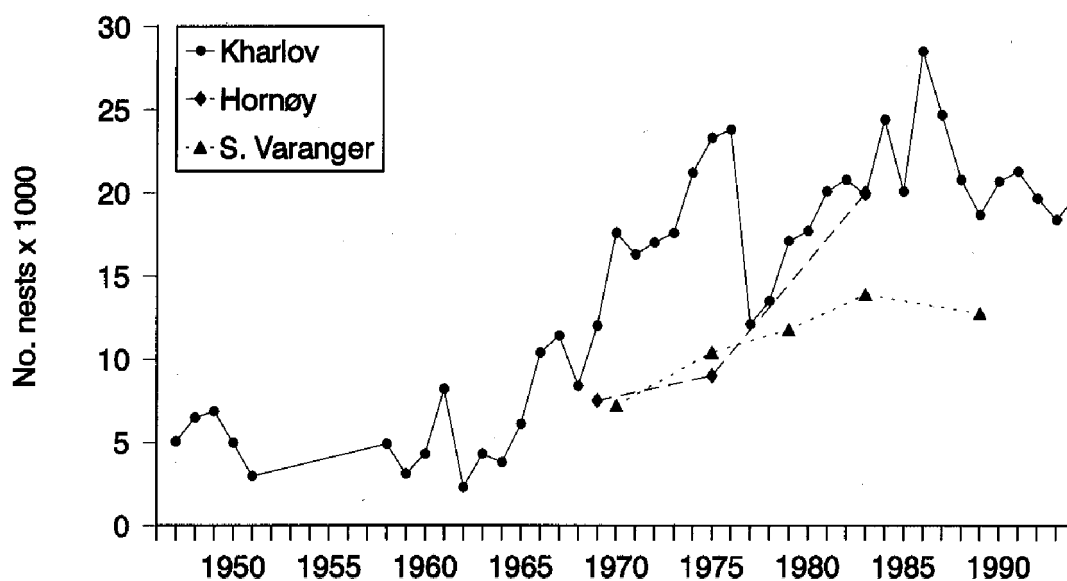


Figure 4.11. Changes in the total numbers of breeding black-legged kittiwakes (apparently occupied nests) on Hornøy, Kharlov, and on 35 colonies in Sør Varsanger (Krasnov and Barrett, 1995).

## 4.5.5 Discussion

Both negative and positive impacts of fisheries on seabirds can occur at multiple spatial and temporal scales. Commonly, effects on population abundance are difficult to demonstrate, even if they are very likely to exist. To make things more complicated, a single fishery can potentially have simultaneous positive and negative impacts on a species of seabird. The potential effects of shifts in age and size-structure of fish in the North Sea as a result of overfishing large predatory fish are very difficult to quantify, but are likely to be or have been substantial. Discards and offal as an extra source of food were probably most significantly positive effects for birds like the northern fulmar and several species of gulls.

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## 5 Further development of seabird monitoring

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### 5.1 Introduction

As long-lived top predators foraging over long distances at sea, seabirds can integrate marine environmental conditions over wide spatial and temporal scales, and respond with changes of life history parameters to various fluctuations in the marine environment. Aspects of their breeding and feeding ecology reflect seasonal and inter-annual changes in the productivity of oceans (Anderson *et al.*, 1982; Cairns, 1987; Croxall *et al.*, 1988; Furness and Nettleship, 1990). Because some seabirds feed on commercial species, their monitoring should provide information on the status of prey stocks, for instance when there is a large drop in the stocks (Montevicchi and Berruti, 1990). Furthermore, parameters of reproductive performance of seabirds may be indicators at meso- or macro-scale (Hunt and Schneider, 1987) of short and long-term changes in oceanographic conditions (Boersma, 1978; Schreiber and Schreiber, 1984; Croxall *et al.*, 1988). Certain oceanographic changes may be so significant that they can be quickly detected at the upper food level. In addition, various species of seabirds may have the potential to reflect large-scale oceanographic changes in their breeding, although their life history parameters may respond differently. Long-term studies in North Sea provided evidence that demographic parameters of seabirds may be correlated with changes in physical factors (Aebisher *et al.*, 1990). Several of these parameters such as population size, breeding success, duration of foraging trips, changes in body mass, or chick growth rate, have shown to be responsive to changes in environmental conditions.

As top predators in the marine food webs, seabirds accumulate environmental contaminants that can easily be measured in eggs or feathers. Consequently, seabirds have been used as monitors of ocean pollution by heavy metals and xenobiotics (e.g., Furness, 1993; Becker *et al.*, 1998) or by oil (Camphuysen and Heubeck, 2001).

These characteristics qualify seabirds as biomonitors, and they are accepted and already in use as indicators of various aspects of the marine environment (e.g., Furness and Greenwood, 1993) either as sensitive indicators or as accumulative indicators of pollutants. They have become parts of national and international monitoring programmes (see examples below). However their use as indicators of the marine environments could be further increased because of their value and advantages as biomonitors. By monitoring parameters additional to those presently used, and by establishing monitoring schemes at wider geographical and international scales, the value of seabirds as monitors of marine environmental change could be much improved.

In this chapter, we review a variety of parameters measured in seabirds and assess their usefulness as indicators of environmental quality and change to detect short-term and long-term ecosystem effects. In addition we make recommendations for designing monitoring programmes using seabirds.

### 5.2 Demands for the parameters and seabird species selected for monitoring

Main aims of marine monitoring with seabirds is to use them:

- as sensitive indicators of environmental change, by studying demographic parameters;
- as indicators of food supply and availability, by studying diets and provisioning of young;
- as accumulative indicators of marine pollution.

What are the preconditions of successful monitoring?

- The monitoring aims should be clearly defined: human effects on the marine ecosystem are the focus of interest, e.g., by fishing or by pollution with oil or environmental chemicals.
- Sensitivity: The species and parameters selected should be sensitive to the specific change in the marine environment.
- High signal to noise ratio: The response of the parameter to a specific environmental change should be distinct from sources of other variation (“noise”) and accessible to investigation by scientific methods.
- Practicability: The parameters should be easy to measure; an advanced and accepted methodology should exist, described in guidelines; the monitoring should be cost effective (favourable, accessible sites, man power, costs for field work, data management and evaluation).
- To be used as early warning of the environmental state, data evaluation has to be carried out continuously, that means by scientific staff.

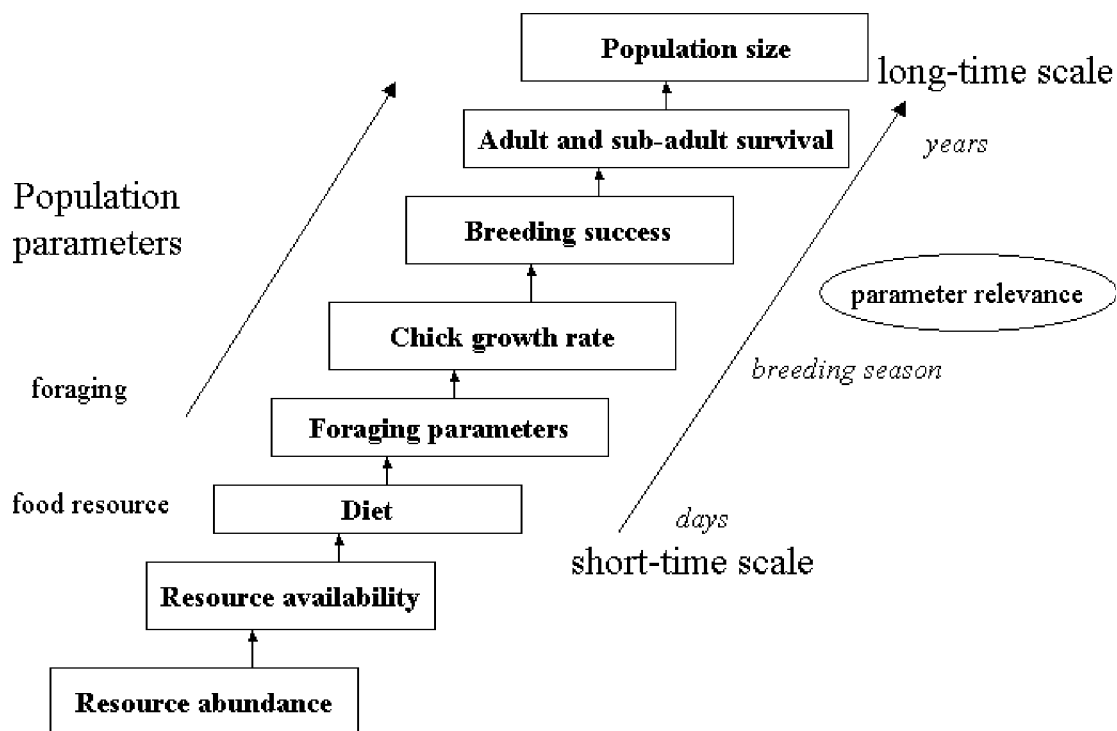


Figure 5.1 Scheme of the most important parameters as a function of time scale integrating marine resource abundance and availability in seabirds.

The parameters of life history are indicative of potential changes at different temporal and spatial scales (Figure 5.1). Most of the breeding parameters integrate the environmental situation during the breeding season and the sea area around the breeding colonies, whereas survival rates and population size reflect environmental influences over the whole year, over breeding as well as migrating areas and with long-term consequences for the population.

The selected indicator species should be common and widespread to be able to cover various sea areas, prey stocks and to detect spatial trends within one species.

### 5.3 Seabirds as sensitive indicators for change in the marine environment: monitoring characteristics of seabird life history besides population size

In general, seabirds are extreme long-lived species with high adult survival and low annual reproductive output. Many species delay first breeding until several years old. In consequence, the non-breeding proportion of the population is high.

Seabird numbers in a population change as a consequence of births, deaths, immigration and

emigration. Much effort has been put into surveillance of the size of seabird populations, and almost all of this effort has been directed at the census or sample monitoring of breeding numbers. This is largely because it is much easier to make accurate counts of breeding birds, or nests, than it is to count all birds in seabird populations. However, changes in breeding numbers may reflect not only births, deaths, immigration and emigration, but also changes in age of recruitment (first breeding), changes in the proportion of mature birds that choose not to breed in a particular year (deferred breeding), or changes in timing or synchrony of breeding that can be confounded with changes in breeding numbers if, as is often the case, census counts of nests or breeders are made on a particular date in the breeding season.

Life history theory envisages trade-offs between components of fitness, such as survival and reproductive investment (Roff, 1992; Stearns, 1992; McNamara and Houston, 1996), and leads us to anticipate that seabird breeding numbers (equivalent to estimated 'population size') may vary as a consequence of such trade offs by seabirds. Monitoring seabird breeding numbers alone, will not only fail to provide any indication as to the cause of a measured change in numbers, but also may present a less than optimal approach to detecting change, since specific life history parameters may vary more strongly in response to changes in food supply than would

breeding numbers (e.g., Cairns, 1987). For example, breeding success may show strong relationships with food supply but despite this there may be little consequent impact on breeding numbers, as a result of various forms of buffering of breeding numbers. Even if this were not the case, changes would only be manifested some years later owing to deferred breeding. Also, while change in food abundance may affect breeding success, small and often undetected changes to adult survival might have a more significant influence on population change (see Section 3).

Dynamic interactions between the life history traits exist, underlining the necessity to look to many of the relevant population parameters to be able to find the causes for population changes and influences of changes in food supply or pollutants. In consequence, to use seabirds as indicators of change in the marine environments, other life history parameters should be selected in addition to population size.

Section 3 reviews life history traits with respect to their desirability, feasibility and practicability as characteristics for monitoring seabird population dynamics in response to changes in food supply (Table 5.1). The results of this review are presented below. In addition, population parameters that are important for population regulation and worth taking into consideration for future seabird monitoring.

### 5.3.1 Monitoring important life history parameters in seabirds

Table 5.1 presents the life history parameters most relevant for integrated population studies in seabirds. We distinguish between different degrees of desirability to measure a specific characteristic, of the feasibility and practicability to record it (financial and/or logistic reasons; manpower). In this report the parameters of highest rates of desirability, feasibility and/or practicability, are emboldened in Table 5.1.

### 5.3.2 Survival of adults and subadults

#### 5.3.2.1 Adult survival

Sensitivity modelling reveals that among the demographic parameters adult survival and also subadult survival have the most decisive influence on population size in seabirds, but indicates also interspecific differences owing to various life history strategies (e.g., Croxall and Rothery, 1991).

To calculate adult survival, local recordings (resighting, retrapping) are necessary over many years. Very large samples are required to detect a significant change in the survival rate. For instance, in the wandering albatross for a sample of 1000 birds the standard error of an annual

Table 5.1. Monitoring of important life history parameters of seabirds. The most important parameters are emboldened.

Parameter	Desirability	Feasibility	Practicability
Population			
<b>- population size (breeders)</b>	+++	+++	+++
- incidence of non-breeding	+	+	-
<b>- adult survival</b>	+++	++	+
<b>- subadult survival to breeding</b>	+++	±	-
<b>- recruitment</b>			
<b>% of fledglings</b>	++	±	-
% recruits in population	+	±	±
- recruitment age	+	+	±
- immigration rate	+	-	-
- emigration rate	+	-	-
- sex ratio of the breeding population	±	-	-
- age structure of the population	++	±	-
Reproduction and food provisioning			
<b>- reproductive success</b>	++	++	++
<b>- laying date</b>	+	++	++
<b>- clutch size</b>	+	+++	++
<b>- egg size</b>	+	+++	++
<b>- chick growth rate</b>	+	++	+
<b>- mass of fledglings</b>	+	++	+
- adult nest and brood attendance; provisioning rate	++	++	±
- degree of kleptoparasitism	+	++	±
-attendance of pre- and non- breeders	+	±	±
Body condition of breeders	+	+	+

+++ highest; ++ high; + moderate; ± more or less; - low

survival estimate, assuming a 100 per cent recapture rate, is 0.8 per cent per annum (Croxall and Rothery, 1991). For such long-lived birds a consistent reduction in survival of 1–2 per cent per year can be highly significant for the population. Also accurate estimates of survival in one year cannot be made until a further 2 years have elapsed, because of the need to account for the probability of detecting individuals (Lebreton *et al.*, 1992).

To use survival rates as parameter at a medium time scale its year-by-year calculation is necessary. This implies intensive annual recapture or resighting activities at the study site that is possible only in intensive integrated monitoring studies.

The use of ring recoveries to analyse survival requires many previous years of intensive ringing and large spatial scales, and only allows retrospective insight in survival over long time scales. Therefore, ringing recoveries cannot be used as indicator of recent changes in survival, which may be needed as an early warning of the health of the population or an important environmental change. Also, the estimates tend to be less precise and more biased than resightings.

#### **5.3.2.2 Subadult survival to breeding**

Estimation of juvenile survival also needs large samples of marked birds. In species with long-deferred sexual maturity there will be a considerable delay in obtaining results from live resighting studies (e.g., Harris *et al.*, 1992). For some species, this may be short-circuited by obtaining estimates of survival from the birds that attend the colony in the years prior to that of recruitment. Variation in the survival of subadults is much greater than in that of adults (see Section 3).

A problem in estimating subadult survival is that, without very widespread monitoring; only local recruits can be covered. That means that local return rate is measured; the percentage of emigrated birds cannot be recorded. Another problem is that subadults normally cannot be recorded with high probability before they are breeders and it likely that there is a strong heterogeneity in the probability of detecting individuals. Analysis of dead recoveries circumvents this, but other problems exist.

Recently, capture-mark-recapture models have been applied to estimate the proportion of individuals of different ages that have previously bred (e.g., Oro and Pradel, 2000).

#### **5.3.2.3 Recruitment to the breeding population**

This parameter has also a high desirability, but even lower practicability than subadult survival. It can only be estimated in integrated population studies by resighting studies of marked birds recruiting to the natal colony. One innovative method to be able to monitor recruitment is marking birds with transponders and checking the breeding and non-breeding adults present at the colony

site automatically year by year (Becker *et al.*, 2001). The problem of measuring only natal colony recruitment remains an issue here.

### **5.3.3 Reproductive performance and success**

Reproductive performance indicates the actual environmental situation at the breeding site and the adjacent sea areas during a respective year. For some species, productivity is not necessarily a very sensitive indicator of reproductive performance (Hunt *et al.*, 1986) and may be influenced by factors other than food supply. These factors include predation, weather, flooding, pollution and disturbance (Becker, 1998). Another limitation is that the indicative value of productivity is restricted mainly to the environmental situation during the reproductive period, excluding other periods of the year.

Estimation of breeding success of a population is usually very straightforward, requiring only estimates of the number of chicks fledging per pair. The extent to which breeding success reflects food abundance seems to vary among seabird species, and this variation is somewhat consistent. For example, northern fulmars and common guillemots tend to show about the same breeding success in all colonies under almost all conditions of food abundance, weather and other environmental variation. At the other extreme, breeding success of terns is highly variable among sites and years, being very severely affected by food abundance, but also by weather, disturbance, and predation (Becker, 1998).

Further aspects of reproductive performance in seabirds may be easily measured. These parameters are important to consider when trying to determine the causes for variations in breeding success and are usually regarded as necessary information for interpreting changes. The following is a brief description of these other parameters.

#### **5.3.3.1 Laying date**

Food availability can lead to variation in the date of egg laying within populations between years (Birkhead and Harris, 1985; Becker, 1996). Environmental conditions that influence food abundance or accessibility, such as the extent of sea ice or stormy weather, may also be negatively correlated with laying date.

#### **5.3.3.2 Clutch size**

The number of eggs laid may depend on food availability in the breeding season (e.g., Monaghan *et al.*, 1992; Pons, 1992). There is also a balance between the number of eggs produced and the size and/or quality of each egg (Birkhead and Harris, 1985). Species that lay a single-egg clutch cannot vary the size of their clutch.

### 5.3.3.3 Egg size and quality

Egg size can decline under conditions of low food availability (Pons, 1992, Bolton *et al.*, 1992). Species that lay a single-egg clutch can regulate their reproductive effort by varying egg size or the amount of resources allocated to the egg (Birkhead and Harris, 1985). Considerable variation in egg size (25–30%) occurs among and within auk populations (Birkhead and Harris, 1985). This reflects variation in female mass or body condition and, thus, egg size is indirectly affected by food availability through adult body condition. But egg size depends on other confounding factors such as age of the bird (and averages therefore vary with the age structure of the population). Egg quality may respond in a non-linear fashion to variations in food supply.

### 5.3.3.4 Chick provisioning rates

See Section 5.4

### 5.3.3.5 Causes of egg and chick mortality

The separation of external and intrinsic factors causing egg mortality may give information on possible pollution problems by toxic contaminants, the percentage of infertile eggs and on egg predation. Investigation of the fate of the chicks can help in differentiating between various causes of mortality (e.g., Thyen *et al.*, 1998).

### 5.3.3.6 Growth rates of chicks

Growth rates may depend on food availability during the chick rearing period (Gaston, 1985; Mlody and Becker, 1991; Klaassen *et al.*, 1992). There is a close relationship between the supply of young herring and the growth rate of chicks that fledge in common terns breeding in the Wadden Sea (Greenstreet *et al.*, 1999). Chick growth will reflect foraging conditions especially in species that cannot alter their time budgets (some Procellariiformes, terns). However, in many seabirds chick growth occurs at the maximum rate possible and is insensitive to food abundance unless this is severely reduced.

### 5.3.3.7 Mass of fledglings

Fledging mass depends on food availability during the breeding season, varies between years and reflects the amount of energy reserves of fledgling and, thus, how long a bird can withstand starvation. Therefore, mass at fledgling should be positively correlated with the probability of recruitment to the breeding population (Becker *et al.*, 2001; Hipfner, 2000). This has been shown to be the case in several, but not all, seasons for a number of species. The rate of provisioning depends also on food availability and again varies considerably among and within populations and years (Gaston, 1985) so it should be positively correlated with fledging mass. But some seabirds have flexible time and energy budgets during the breeding season and are thus able to maintain provisioning rates under moderate declines in foraging conditions (Burger and Piatt, 1990).

### 5.3.3.8 Reproductive success

Reproductive success as the final outcome of the reproductive performance in a season may be the least sensitive parameter in reflecting prey availability or foraging conditions because it is influenced by other factors like predation, weather or flooding. However, these can often be recorded and controlled for.

## 5.3.4 Possibilities for integrated seabird population monitoring

The use of seabirds as monitors of the marine environment is becoming increasingly widespread. Monitoring of parameters other than population size is usually regarded as necessary information for interpreting changes. As is shown above, the relevance of the various population parameters will depend on the life history of the species concerned. For extreme long-lived species the order of parameters measured may be adult survival, subadult survival and breeding success.

The different population parameters integrate environmental effects over different time spans. Different lag times before the effects can be measured and different accuracies with which effects are likely to be detected have to be considered (Croxall *et al.*, 1988).

Long-term integrated monitoring of seabird populations is costly and thus rare (Nisbet, 1989; Wooller *et al.*, 1992). Nevertheless, there are some good examples of integrated long term studies at “constant effort sites”: wandering albatross (Weimerskirch *et al.*, 1987; Croxall *et al.*, 1992; Weimerskirch, 1999), short-tailed shearwater (Wooller *et al.*, 1989; Bradley *et al.*, 1991, 1999), black-legged kittiwake (Coulson and Thomas, 1985; Porter and Coulson, 1987; Thomas and Coulson, 1988; Danchin *et al.*, 1998; Cam *et al.*, 1998); Atlantic puffin (Harris and Wanless, 1991; Harris *et al.*, 1997); great skua (Furness, 1987); common tern (Becker and Wendeln, 1997; Becker *et al.*, 2001). Such studies require special sites which allow good access, a long term perspective, an committed team of researchers, the necessary financial basis and effective data management (Bradley *et al.*, 1991). In addition the use of new field techniques, e.g., field readable rings or remote and automatic recording of subadults and adults by passive transponders (Becker and Wendeln, 1997; Becker *et al.*, 2001) may help to increase the efficiency of integrated population monitoring in seabirds.

It is clear that different species of seabirds respond differently to environmental change, depending on their particular life history strategies, and that the ideal programme of seabird monitoring would consider not only breeding numbers and breeding success, but would include attention to other aspects of seabird life history parameters, especially adult survival, subadult survival and recruitment rate, as well as parameters of reproductive performance.

## 5.4 Seabird diets and provisioning as monitors of fish stocks and food availability

Dietary composition of seabirds can often be sampled easily and has been used to monitor prey stocks (Montevecchi, 1993). Some studies have demonstrated statistically significant correlations between commercially harvested fish stocks and their relative abundance in seabird diets. The proportion of sandeel in the diet of great skua chicks was positively correlated with the number of sandeels recruited to the Shetland population in the previous year (Hamer *et al.*, 1991). Coincidental collapses have been reported in the proportion of squid in northern gannet diet and in the squid fishery off Canada (Montevecchi, 1993). Sampling of regurgitated pellets at colonies or roost sites provides otoliths from these fish that can be identified to species, measured to give fish size and sectioned to count annual layers to determine fish age (Barrett *et al.*, 1990). This approach has not addressed the problem that such measures represent relative availability of prey only. Then, it is necessary to compare harvest prey by seabirds and fisheries. The relative abundance of fish taken by birds and fisheries could correspond closely but the spatial and temporal scales over which acoustic surveys conducted by fisheries scientists and seabird foraging may be incorrectly matched. The proportion of a particular species in seabird diet may closely reflect absolute abundances at low levels of biomass but may not be sensitive when prey availability is high (Becker *et al.*, 1987). The development of robust indices requires the inclusion of other seabird parameters that, considered together, are sensitive to change in prey over a large range of absolute abundances (Cairns, 1987).

Variation in parental food loads and in chick provisioning rates could also be obtained in species that carry food in their bills like terns or Atlantic puffins. Among multi-prey loaders, inverse relationships between number of items carried per trip and total masses of food loads suggest that the number of items carried may yield a crude index of prey conditions, i.e., fewer, larger items when food supplies are good (Barrett *et al.*, 1987). The chick-provisioning rates can be ranked according to good, intermediate and poor prey conditions.

## 5.5 Recommendations for further monitoring marine pollution using seabirds as accumulative indicators

### 5.5.1 Background

Pollution may affect seabirds in many ways as seabirds are marine top predators that biomagnify contaminants to high concentrations. Contaminants have the potential to affect life history parameters, especially during the sensitive phase of reproduction, and parameters such as egg shell thickness, embryonic survival, hatching success, chick growth or reproductive success have been

found to be impaired in several studies. The high biomagnification rates qualify marine birds also as monitors of pollutants (ICES, 1999). The potential influence of contaminants on seabird life history parameters means that any integrated seabird monitoring programme should take full account of them.

In some parts of the North Sea monitoring pollutants by seabirds as accumulative indicators occurs already (see below) but should be expanded to other areas of the North East Atlantic, using more species as monitors.

### 5.5.2 Oil

Ongoing programmes of monitoring of the proportions of dead and oiled seabirds found on shorelines ('Beached Bird Surveys' - BBS) should be encouraged as a cost effective means (most are carried out by amateurs at no cost and are organised by NGOs) of determining long-term trends and geographical patterns of oil pollution at sea (Camphuysen and Heubeck, 2001). Such monitoring is of greater interest to seabird conservation than to fisheries. BBS is carried out along all the North Sea coasts, and implemented in the TMAP (Trilateral Monitoring and Assessment Programme of the Wadden Sea).

### 5.5.3 Plastic particles

Numbers of plastic pellets in the oceans appear to be continuing to increase and there is a need to monitor the amounts of plastic ingested by seabirds, especially petrels. Some effects of ingested plastics on seabirds have been noted. Seabirds may provide a useful indicator of this form of pollution.

### 5.5.4 Organochlorines

Sampling of seabird eggs as a means of monitoring local contamination with organochlorines should be included in integrated marine pollution monitoring programmes, with the selection of appropriate locally common and internationally widespread monitoring species (ICES, 1999). In the Wadden Sea, within TMAP, the monitoring of organochlorines as well as of mercury in eggs has occurred since 1996 (Becker *et al.*, 1998, Thyen and Becker, 2000). In the Swedish Marine Monitoring Programme common guillemot eggs have been used as indicators of Baltic Sea pollution since the 1960s (Bignert *et al.*, 1998). Seabird eggs are already included in the JAMP guidelines for monitoring organochlorine contaminants in biota (OSPAR, 1997).

### 5.5.5 Mercury

Methods to monitor mercury contamination in marine food chains by sampling chick down or feathers from chicks or adults, or from blood samples or egg samples, have been developed and are standardised. Monitoring of mercury levels using these methods should be carried out in areas where there is concern about possible mercury contamination of marine food chains. Eggs sampled from



colonies located close to rivers contaminated with mercury can be used as a means of monitoring trends in river mercury loadings reaching the sea. Such monitoring can also provide useful evidence of the success of programmes to reduce mercury. Seabird eggs are included in the JAMP guidelines for monitoring mercury in biota (OSPAR, 1997).

### 5.5.6 Organotin and other metals

There is a need for research into organotin levels in seabirds to determine whether these may have toxic effects on seabirds, and whether seabirds may be used as a means of monitoring organotin pollution on large scales.

There is also a need for research into the possible use of eggshells, egg contents or feathers for monitoring cadmium, lead and other elemental concentrations. At present, monitoring of levels of these pollutants involves killing of birds to sample liver, kidney or bones. In particular, if methods can be developed to measure elemental concentrations within feather keratins separately from contaminants on feather surfaces, this would permit retrospective monitoring, using museum specimens, of long term trends in elemental contamination of marine food chains as has been done successfully for mercury.

## 5.6 Further aspects of designing monitoring programmes with seabirds

An important issue is also the spatial scale at which the processes of interest are occurring and thus the spatial scale at which monitoring needs to be carried out. Seabirds can breed in large discrete colonies. Good sample sizes are therefore possible for monitoring local parameters on series of discrete sub-units of the population, but it has led in the past to the concentration of efforts on a very limited number of "key" sites, usually where long-term studies have been taking place. When considering which parameters could be good to monitor, special effort should thus be put in:

- 1) consideration of the representativeness of the monitored sites,
- 2) integration of the monitoring of the parameters on series of sites at different scales (e.g., choice of colonies at a regional scale and choice of study plots within a colony), and
- 3) combining various indicative parameters into an integrated monitoring project.

However, for those species breeding on cliffs the choice of sites is constrained by ease of access.

For logistical reasons there is an understandable concentration of monitoring at "key" sites, but also additional and powerful analyses become possible if the key sites are supplemented by a wider net of sites

(Anker-Nilssen *et al.*, 1996). This may be especially important as this allows questions about the spatial dependence and scale of change in the parameters monitored to be addressed. For instance, variability in some seabird population parameters may suggest independence in variability among fish stocks of different areas (Furness *et al.*, 1996). Monitoring parameters at different locations in an integrated fashion is also crucial because of the potential importance of movements of individuals between sites and the effect that this can have on demography. This has classically been neglected due to the high natal philopatry and breeding philopatry of individuals of most species and the difficulty of quantifying the rates of movement among locations (Boulinier and Lemel, 1996). There is evidence that dispersal of individuals may be dependent on local conditions and can affect strongly local numbers of breeders (e.g., dispersal of breeding individuals among black-legged kittiwake colonies situated few kilometres apart (Danchin *et al.*, 1998). Further, statistical modelling tools are now available to estimate rate of dispersal between locations while accounting for different probability of individuals being sighted or dying in the different locations (Spendelov *et al.*, 1995). Finally, the availability of time series of comparable data from several locations can allow stronger inference to be made. This is because an analysis carried out on the time series analyses from only one site may suffer the weakness of the potential existence of several alternative factors driving the observed pattern. Monitoring interactions of trends in a parameter among several sites in relation to external parameters allows more powerful diagnosis of factors responsible for changes (e.g., Anker-Nilssen *et al.*, 1996; Thyen *et al.*, 1998). Study colonies should, where possible, be selected on the basis of the likelihood of changes in those variables that can affect seabirds.

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## 6 Review of the interactions between aquaculture and birds in the ICES Area

Robert W. Furness

### 6.1 Introduction

This section provides a brief review from the perspective of seabird ecologists of the interactions between aquaculture and birds.

There has been a huge growth in aquaculture in the last few decades, with consequent concerns for a variety of environmental impacts (Naylor *et al.*, 2000). In temperate and higher latitudes, this has particularly involved marine cage culture of high value finfish, such as Atlantic salmon, and various forms of marine culture of shellfish.

Aquaculture can provide new feeding opportunities for some birds, and so can create a local increase in bird numbers. Increases in bird numbers at fish farms may lead to problems for fish farmers. Birds may eat or damage stock, may represent a nuisance or a vector of disease. These problems may cause significant costs to fish farmers both in direct financial terms, and in terms of time spent in trying to reduce bird problems. The impacts of birds on fish farming can lead to farmers taking action against perceived bird problems, while farming itself may have some negative impacts on birds. Direct negative effects of aquaculture on seabirds (defined here as marine birds and shorebirds) have been thought to arise mainly from loss of habitat, harvest of young marine animals for culture that may represent natural food of some seabirds, disturbance of birds by fish farm workers, and killing of birds (either deliberately or accidentally). It should be noted that enhancing food supply to wildlife is not necessarily a positive effect, since increases in some animal populations can have negative impacts on others, and can lead to populations becoming dependent on artificial feeding opportunities. Aquaculture may also have indirect effects on seabird populations, since most fish farms in ICES areas are dependent on the use of aquafeeds derived largely from fish meal and fish oils (Thomson, 1990; Naylor *et al.*, 2000). This raises the indirect effects of industrial fishing on seabirds as an issue related to the practice of mariculture.

In this review the topics are considered in the following sequence: Enhancing food supplies to wildlife: common eider; shorebirds; gulls; seals, cormorants, herons and other birds; attraction of birds to aquaculture and consequent impacts on fish farming profitability; disturbance of birds by aquacultural activities; persecution of birds by fish farmers; impacts on birds of the harvest of mussel spat; impacts of industrial fishing on seabirds. Certain aspects that may be relevant have not been reviewed here due to lack of appropriate expertise, and lack of access to literature on the subject

area; these include wild birds as vectors of diseases affecting cultured fish and shellfish (Erwin, 1995; and see for example Moravec *et al.*, 1997 who report parasite infections in cultivated fish where birds host the adult parasite).

### 6.2 Enhancing food supplies to wildlife

There are several cases where aquaculture provides food for wild animals, to an extent that certainly influences the pattern of distribution of the animals and may in some cases increase their population size. Some of the better-documented examples are outlined below.

#### 6.2.1 Common eider

Common eiders are widely distributed seabirds that inhabit coasts around the globe in sub-polar and boreal regions. They specialise in feeding on mussels, which they locate by diving in shallow water, pull off the substrate and swallow whole, then grind these up in their muscular gizzard. Farmed mussels grown on suspended rope cultures are particularly attractive to common eiders because these tend to be thin shelled and have a high energy content compared with wild mussels growing intertidally. They are also grown at high densities and at shallow depths, so that energy costs of feeding are minimised for common eiders allowed to feed undisturbed at mussel farms. Common eiders, and to a smaller extent also some other seabirds such as long-tailed ducks and scoters, soon develop the habit of flocking at unprotected mussel farms where they can rapidly deplete the standing stock of cultivated mussels (Milne and Galbraith, 1986; Ross and Furness, 2000). Although experienced farmers tend to take steps to deter ducks from stealing cultivated mussels, many farms do not protect their stock very effectively, and common eider numbers feeding on farms can represent a high proportion of the local population. This is particularly the case in spring, when female common eiders need to feed intensively in order to build up reserves for egg production and for their fast throughput incubation. Somewhat unexpectedly, winter surveys of common eiders feeding on natural habitat, at mussel farms and at salmon cages in the west of Scotland (Figure 6.1) found that common eiders fed in very large numbers at many salmon farms as well as at mussel farms. In general, salmon farmers do not scare common eiders away and the ducks evidently learn this. They often roost in close proximity to salmon farms, where human disturbance may be reduced compared to other stretches of coastline, due to salmon farms preventing human access to reduce risks of disease transmission. It is not clear whether common eiders feeding around salmon cages are simply stripping off mussels that foul the cages, nets and ropes,

or whether common eiders also scavenge lost fish pellet food. Certainly, captive common eiders will happily feed on salmon feed pellets. Whichever is the case, there is no doubt that both mussel farms and salmon farms attract common eiders, and in the west of Scotland they have a strong influence on their local distribution from autumn to spring. Despite this strong, and relatively new, feeding association, there is no convincing evidence from detailed and accurate census data that the opportunities to feed at salmon and mussel farms has influenced population sizes of common eiders in Scotland.

### 6.2.2 Shorebirds

Many species of shorebirds spend the winter feeding on intertidal benthic invertebrates on estuaries and sheltered coasts. Where aquaculture is established on intertidal areas, high densities of cultivated animals may present unusually good feeding opportunities for some shorebirds. In particular, Eurasian oystercatchers may benefit from being able to feed on mussels that have been artificially set on intertidal areas. Such opportunities are probably rather limited, as costs of disturbance will often outweigh the benefits from enhanced food stocks.

### 6.2.3 Gulls

Many species of gulls are highly opportunistic feeders, and exploit a wide range of habitats as well as taking many kinds of food. Where aquaculture permits access to gulls, they will readily scavenge on remains of fish and invertebrates, steal aquafeed pellets from feeders or by tearing open carelessly stored bags of feed, and take the growing product from accessible areas on farms. This is especially true of herring gulls. Gulls do not need to spend much time feeding each day, but can often obtain most of their daily energy needs in a few minutes of intense feeding at a place where food is abundant. It is very common to see flocks of several hundred gulls hanging around aquaculture facilities waiting for occasional opportunities to feed at the farmers' expense. There is no doubt that the local distribution of gulls can be affected by the availability of such feeding opportunities, but populations of gulls use aquaculture as only a small part of their overall energy intake, and it is unlikely that this has much effect on gull population sizes. It is possible that the local aggregations of large scavenging gulls at aquaculture sites may drive away smaller birds that can be the target of robbery or predation by gulls.

**Feeding sites used by common eiders in various areas of west Scotland**

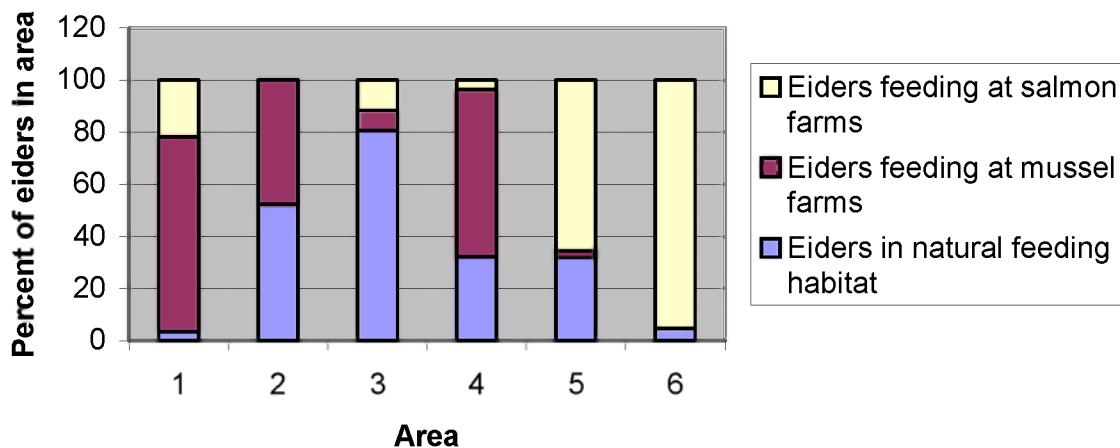


Figure 6.1. Percentages of common eiders in six areas in the west of Scotland that were in feeding flocks at salmon farms, at mussel farms and at natural habitat in surveys in autumn-winter 1998 and 1999. Large differences between areas reflect differences in numbers of aquaculture establishments in different areas as well as variations in common eider habits (based on data in Ross and Furness, 2000).

#### 6.2.4 Seals, cormorants, herons and other birds

Unprotected fish farms can attract seals, and a variety of piscivorous birds, especially some species of cormorants (Nettleship and Duffy, 1995) and herons (Kushlan, 1997). The fact that these animals gather at fish farms implies that they find that these sites provide an attractive feeding opportunity (Price and Nickum, 1995). Often the animals attracted to fish farms are predominantly young ones, suggesting that adult animals perceive the hazards of feeding at fish farms to outweigh the benefits of access to easy pickings. Young animals are generally less efficient at foraging and are often displaced from the best feeding sites by older, dominant, animals. The inferior abilities of young animals may make fish farms relatively more attractive to them because they present a chance for rapid food intake, albeit at a high risk. Young animals at increased risk of starvation may find that feeding at fish farms increases their chances of overwinter survival providing they are not subject to high risks of being shot by farmers.

Fleury and Sherry (1995) reported that winter populations of wading birds (mostly herons) increased dramatically in Louisiana between 1966 and 1989, and that increased acreage devoted to crayfish *Procambarus* aquaculture appeared to be the main factor explaining this increase. Wading bird numbers correlated over years with crayfish pond acreage, and birds used this feeding habitat extensively, especially during pond drawdowns that took place during the birds' breeding season and may have increased reproductive output. In addition, the greatest population increases were in the bird species making most use of crayfish as food.

### 6.3 Attraction of birds to aquaculture and consequent impacts on fish farming profitability

Where aquaculture attracts large numbers of birds, the activities of the birds may represent a serious impact on the profitability of farming, as a result of birds taking product or interfering with the functioning of the farm. The main perceived impacts on farming are depredations of fish by cormorants and herons, and depredations of mussels by seaducks (especially common eiders). Few studies in the ICES area have quantified costs, so we provide examples from studies elsewhere in the world that are relevant and instructive. Many of these refer to freshwater aquaculture sites, where quantifying damage or losses is easier.

Glahn *et al.* (1999a) conducted on-site interviews of fish farmers in the north-eastern United States, and made brief bird counts at these sites during summer visits (not necessarily the season when most birds feed at fish farms). They reported that 80% of fish farmers reported bird predation to be a problem at their site, great blue herons being the main concern, and trout farmers the most affected group. Nearly a quarter of trout farms were thought to be losing stock worth more than US\$10,000

per year to bird predation. Great blue herons were seen at 90% of aquaculture sites they visited. Further investigation involved shooting herons at these sites to investigate stomach contents. Herons showed crepuscular feeding habits, ingesting about three trout per day per bird, mainly taking fish in the 12–38 cm length range (Glahn *et al.*, 1999b). Trout losses varied between sites, being negligible at some sites not visited by herons, but up to 40% of trout stock was taken at other sites where herons were frequent visitors.

Avery *et al.* (1999) studied fish loss due to bird predation (mainly by herons and egrets) at aquaculture ponds in central Florida. Losses from ponds from which birds were excluded by netting averaged 11% while losses from ponds without netting averaged 38% of fish stock. These translated to economic losses of US\$589 per netted pond and US\$1360 per un-netted pond, making anti-predator netting highly cost effective even though not fully successful in preventing predation.

Costa-Pierce (1998) reported that bird predation was responsible for loss of about 50% of tilapia stocked in combined wastewater treatment-fish culture ponds in Los Angeles County. Pitt and Conover (1996) reported that herons (great blue herons and black-crowned night herons), ospreys and California gulls were the main predators of fish at Intermountain West fish hatcheries. Birds were estimated by scientific study to remove 7% and 0.5% of annual production from two hatcheries, whereas managers estimated losses to birds at 15% at each of these two hatcheries. Managers at private hatcheries estimated losses to birds at 13% of production, whereas managers of state-owned hatcheries estimated losses to birds at only 5%.

In the Mississippi Delta of the United States, catfish farmers estimated loss to cormorants of fish worth US\$3.3 million per year. A bioenergetics model combined with data on cormorant numbers and diet, suggested that double-crested cormorants may have eaten up to 20 million catfish per winter in 1989–1990 and 1990–91 in the Delta region of Mississippi, equating to 4% of standing crop at an estimated replacement cost of US\$2 million per year (Glahn and Brugger, 1995), which is not too far from the farmers' estimate of the cost of bird predation given that farmers are likely to provide estimates that maximise the scale of the problem.

Predation by grey herons and great cormorants was studied by Genard *et al.* (1993) in a dyked area intended for extensive brackish water aquaculture on the French Atlantic coast. A pond was divided in two parts, one of 0.9 ha was protected with nets and wires, while the other of 0.3 ha was left without protection. Fish (Mugilidae) were stocked in November, during the wintering period of the birds. Three weeks later, remaining fish in both parts of the ponds were counted. The number, presence and predation behaviour of birds was observed during the experiment. A spectacular but temporary gathering of birds occurred during the first days following fish stocking. Great cormorants used deeper and grey herons shallower areas of the pond. Intensity of predation

attempts was higher among great cormorants. Features of predation exerted by these birds were the rapid exhaustion of the fish population, the higher predation level by cormorants than by grey herons, although the latter obtained a very high success ratio (number of catches/number of attacks) (Genard *et al.*, 1993). Similar studies of fish ponds that could be drained to count numbers of surviving fish after a period of exposure to bird predation were carried out recently in England to try to quantify the impact of predation by grey herons and especially great cormorants. That study had difficulty with assessment of nocturnal feeding by birds, but also suggested that piscivorous birds can remove stocked fish rather rapidly from small ponds (McKay *et al.*, 1999).

Common eiders, and to a lesser extent other ducks such as common goldeneye and long-tailed ducks, can rapidly deplete mussels from suspended cultures on rafts or long-line mussel farms. Many mussel farms in Scotland and in eastern Canada have been put out of business by duck predation drastically reducing cultivated stock and harvest. Most mussel farmers in Scotland regard common eiders as the main predation problem on mussel farms, and spend considerable amounts of time, and cash, in trying to combat this problem (Ross and Furness, 2000).

Not all studies of bird 'problems' at fish farms support the view of managers that birds cause significant economic losses. For example, Ulenaers and van Vessem (1994) showed that great crested grebes at a freshwater fish farm in the Netherlands only marginally influenced fish populations in the ponds despite farmers' claims that they took large amounts of fish.

#### **6.4 Disturbance of birds by aquacultural activities**

Aquaculture can cause disturbance to wildlife by major modification of habitat, such as removal of habitat required for nesting or feeding sites. It can also cause local disturbance through minor alterations to habitat. For example, oyster culture on intertidal areas involves addition of racks, stakes, culture bags, marker poles and other equipment onto open tidal flats. Some birds are attracted onto such structures. For example, gulls (and some kinds of shorebirds) may use elevated structures as roosts. However, most species of shorebirds tend to avoid oyster culture plots, preferring to feed on open areas of tidal flats. Since shorebird numbers tend to be set by the amount of food in their wintering estuaries, loss of open estuarine foraging habitat to aquaculture is likely to have a negative impact on shorebird populations. This has not been studied in detail as yet. Aquaculture probably has much less impact on shorebirds than does loss of estuarine habitat through land claim or alteration of invertebrate populations due to nutrient pollution, but loss of feeding area to shorebirds through aquaculture will add to these other forms of habitat loss that are concerns in the conservation of shorebird populations.

Human activity at aquaculture facilities can also affect wildlife. Where fish farms are sited in remote areas, as with many salmon farms or mussel farms in Scotland, wildlife may be affected by human disturbance resulting from routine farming activities in a way that animals would not be in places where they have become used to regular human activity. For example, black-throated divers nesting on remote lakes may be severely disturbed by helicopter flights transporting live fish into or from a farm, leaving their eggs exposed to predation by gulls or crows. Seaducks may abandon otherwise profitable foraging areas if the level of disturbance by human activity increases their activity costs and reduces time available for foraging so that they are unable to balance their budget. Smaller birds, nervous of the potential threats presented to them by large gulls, may abandon an area in which large aggregations of gulls have come to gather as a result of feeding opportunities provided by aquaculture.

Disturbance of birds by aquaculture is not necessarily restricted to effects on seabirds. Mooney (1998) reported without detail that populations of some coastal tropical raptors in Australia had been reduced as a result of increased disturbance in the coastal zone associated with aquacultural development as well as other forms of habitat loss and coastal development.

While loss of natural habitat to aquaculture can be a problem, so can conversion of aquaculture sites to other uses. Nelson (1993) reported that many areas of the Mai Po marshes in Hong Kong are used for pond culture of shrimps and fish, and that these provide important habitat for a varied bird and other life, including many migratory bird species. Conversion of many of these aquaculture ponds to rice fields represented a serious threat to these bird populations.

#### **6.5 Persecution of birds by fish farmers**

Many forms of aquaculture attract certain birds and marine mammals to feed on the high concentrations of food being cultivated, as described in the previous section. If no action is taken, birds and mammals can have devastating effects on the viability of farms. Farmers generally adopt one or more of three strategies. They may invest in costly structures to exclude wildlife from farms (Brugger, 1995; Ross and Furness, 2000) – though careful design at the initial farm construction phase might make such 'bolt-on' solutions less necessary. They may reduce local numbers of the problem animals by shooting or by other means (Belant *et al.*, 2000). They may employ non-lethal deterrents that scare damaging animals away from the area (Mott and Boyd, 1995; King 1996). However, many non-lethal methods have proved to be unsuccessful (Mott and Boyd, 1995; Russell *et al.*, 1996; Dorr *et al.* 1998, McKay *et al.*, 1999; Ross and Furness, 2000). Nevertheless, killing of wild predators of farmed fish is generally poor management practice. There is little scientific evidence that killing predators is effective in controlling predation.

There is even less evidence that it produces an economic gain for aquaculture. By killing wildlife, it may be argued that the aquaculture industry paints an unflattering picture of itself as insensitive to the environment.

Some experts argue that there is little evidence that removal of bird or mammal predators has any long-term effect on predator abundance or fish loss at farms, because removed predators are quickly replaced by others attracted to such concentrations of food (McKay *et al.*, 1999). Advocates of lethal control of predators argue that killing a few predators scares others and increases the effectiveness of non-lethal deterrent methods (McKay *et al.*, 1999).

The main wildlife predation problems faced by aquaculture are fish-eating aquatic mammals and birds, especially seals, cormorants and herons (Accord, 1995; Trapp *et al.*, 1995; Russell *et al.*, 1996; McKay *et al.*, 1999). A survey of fish farmers in north-central states of USA found a strong consensus that farmers should be allowed to kill birds on their property without permits, and an unwillingness to invest money in preventative measures. The United States Fish and Wildlife Service (USFWS) reported killing under permit of about 10,000 birds per year in the early 1990s, mostly double-crested cormorants, herons and egrets. These birds were killed to reduce impacts of fish-eating birds in fish farms. Mississippi Delta catfish farmers estimated loss to cormorants of fish worth US\$3.3 million per year, despite shooting birds and spending US\$2.1 million per year on deterring birds. However, although the last example indicates large financial costs, these represent only a few percent of the industry's production value.

Predator control occurs in association with aquaculture in most parts of the world. Two studies of predator control at Scottish salmon and trout farms in the late 1980s estimated that around 800 grey herons, 1600 great cormorants and 1400 European shags were being killed each year in that rapidly developing industry (Carss, 1994). The majority of these birds were killed illegally, only a small proportion being killed under licence from the Scottish Office. What effect does predator control have on bird populations, and can these impacts be justified by the economic gains that result? Data to answer these questions do not exist for most parts of the world. Even in countries where wildlife conservation is given a high priority and populations are carefully monitored, the facts are far from clear. Surprisingly, despite issuing licenses to permit farmers to kill birds, the USFWS has not made any definitive evaluation of the effect of shooting on population trends of cormorants, herons and egrets in the USA. Recent analyses of legal killing suggest that the numbers killed under permit are probably too small to noticeably affect bird population size, though the scale and impact of killing without permit is not evaluated. Blackwell *et al.* (2000) reported that the USFWS issued 26 permits to 9 aquaculture facilities in New York, New Jersey and Pennsylvania from 1985–1997, resulting in a mean of 83 birds killed under permit per year per facility. Most birds

killed were herring gulls or great blue herons, and the authors considered these numbers to be negligible in relation to the bird population sizes. Belant *et al.* (2000) reviewed permit records for predator killing in 9 States in southeast United States from 1987–1995. Over 108,000 birds were authorised to be killed and over 64,000 were reported to have been killed. Half were double-crested cormorants, 21% were great blue herons, and 13% were great egrets. Because numbers reported killed per year represented less than 3% of the continental breeding numbers for each species the authors concluded that the level of legal killing did not adversely affect population sizes of these birds, and Christmas bird-count data showed no evidence of killing affecting numbers of birds present in the region. However, this study did not provide a reliable measure of increase in mortality rate of birds within the states where the killing occurred, and made no allowance for the numbers killed without permits.

Numbers of great cormorants in northwest Scotland have certainly declined in recent decades whereas in every other part of Europe their numbers have been increasing (Lloyd *et al.*, 1991). It is likely, though far from proven, that the decrease in their numbers in north-west Scotland is a direct result of the killing of large numbers of great cormorants at salmon farms; best estimates suggest that 1600 great cormorants were killed each year at Scottish fish farms during the 1980s only a small proportion of these being killed legally under permit (Carss, 1994).

Some common eiders are shot by mussel farmers in Scotland, and some are drowned as a result of becoming entangled in anti-predator nets set around mussel farms. However, numbers of common eiders in Scotland are increasing in all areas except Shetland (where deaths of ducks associated with mussel farming have been negligible up to now), and there is no indication from census data that the rate of increase has been noticeably reduced by the mortality associated with mussel farming in other parts of Scotland (Ross and Furness, 2000).

If it is difficult to establish the impact of, or even the extent of predator control by aquaculturalists in the United States or Scotland, it is far more difficult to assess the situation in parts of the world where the lobby for wildlife protection is less strong and there is little monitoring of bird or aquatic mammal populations.

In addition to the impact of lethal control measures, non-lethal measures may affect wildlife. In this case the effects are probably minor and mainly involve impacts on local distribution and behaviour rather than on population size. Non-lethal scaring measures regularly used in the aquaculture industry include shooting with blanks, use of gas cannons, other acoustic deterrents, scarecrows, chasing by powerboat, flashing lights and pyrotechnics. Regular scaring of birds from aquaculture sites may increase their energy needs through increasing the time they have to spend flying, and may reduce their longer term foraging rates. Seaducks in areas with extensive mussel farming show higher rates of vigilance than ducks in areas without mussel farms, and respond



more rapidly (by flying away) when approached by boat (Ross and Furness, 2000). This relates to the frequent habit of farmers to chase seaducks away from mussel farms by powerboat. Birds learn to be more cautious in places where they are subject to such harassment.

## **6.6 Impacts on birds of the harvest of mussel spat**

Mussel farming often relies on the harvest of wild mussel spat to establish cultures of mussels on ropes or on selected intertidal or subtidal plots. Removal of natural spat can be very extensive and can influence food availability for specialist predators of mussels (Kaiser *et al.*, 1998). It is difficult to estimate the importance of mussel spat harvest for dependent wildlife. In many areas the quantity of spat is so large that harvesting some may have little or no influence on subsequent biomass of natural mussel stocks in the region. However, there is one well-documented case where bird populations have been severely affected by mussel spat harvest and mussel farming, combined with fisheries for other molluscs that might have provided alternative prey. In the Wadden Sea, common eiders and Eurasian oystercatchers occur in hundreds of thousands during winter, and feed predominantly on mussels and cockles. These resources are subject to heavy fishery harvest as well as collection of mussel spat for subtidal mussel farming, despite the fact that the Dutch Wadden Sea has been declared a wetland of international importance as a Ramsar site, a Biosphere Reserve, and under the EC Wild Birds Directive and EC Habitats Directive. Mussel spat stock assessment is carried out prior to harvest, and in years of spat scarcity, intertidal areas are now closed to harvesting in order to provide for the food requirements of protected bird populations (Smaal and Lucas, 2000). However, in 1990, mussel seed collection led to a near-complete stock depletion, at the same time as harvesting of mature cultivated mussels reduced adult stocks to a very low level. Unprecedented thousands of common eiders and Eurasian oystercatchers died that winter and numbers in the Wadden Sea fell and have remained lower than in the 1970s–1980s. Many surviving common eiders moved out of the Wadden Sea and established a new pattern of feeding on another mollusc species, *Spisula*, available in deeper North Sea waters. Continued harvesting of mussel spat and contraction of the area of mussel beds into smaller areas of protected mussel farms, together with fisheries exploiting Wadden Sea cockles and North Sea *Spisula*, was followed in winter 1999/2000 by another mass mortality of common eiders, with over 21,000 birds starving to death along the Wadden Sea coastline (Camphuysen *et al.*, 2002, Ens, 2002).

## **6.7 Impacts of industrial fishing on seabirds**

Industrial fishing, to harvest small fish for production of feed for aquaculture, is an indirect, but important, influence of the aquaculture industry on the environment (Naylor *et al.*, 2000). Not only has aquaculture

production been increasing, but there has also been a trend towards intensification of aquacultural systems. A particularly rapid increase has occurred in the use of aquafeed in production of predatory fish and crustacea. Aquaculture has tended to 'feed up the food chain'. This trend has been most pronounced in prawn culture, staple freshwater fish culture and luxury marine fish culture in countries such as China, Indonesia, the Philippines and India. Globally, the use of artificially compounded feeds ('aquafeeds') to increase production of farmed finfish and crustacea has increased at a rate in excess of 30% per year for the last few years. Production of around 3 million tonnes of farmed fish and crustacea in 1995 required about 1.5 million tonnes of fish meal and oil, manufactured from over 5 million tonnes of pelagic fish (wet weight). By 2000, nearly 10 million tonnes of pelagic fish was being used in aquafeeds, and this increase is projected to continue. These pelagic fish are harvested by 'industrial fisheries', where the entire catch is destined for reduction to fish meal and oil rather than for direct human consumption. The EU is a major importer of global production of fish meal. Most industrial fisheries harvest from stocks of abundant, small, shoaling, pelagic fish such as anchovies, sandeels, capelin, sprats or juvenile herring. Such fish are also food for many predatory fish that are commercially important for human consumption fisheries, as well as supporting many marine mammals and seabirds. Hence concern has been raised over potential for competition between industrial fisheries, commercially important human consumption fisheries, and wildlife conservation interests

The proportion of the global catch of industrial fisheries that is used for aquafeeds has been increasing very rapidly, from 10% in 1988 to 17% in 1994, and 33% in 1997. This gives several causes for concern. The rapid increase in requirement for fish meal and oils could not be met by proportional increases in industrial fish landings, since most major industrial fisheries are fully exploited. By comparison with relatively slowly increasing industrial fish catches, global production of soya has doubled in the last ten years. This has led to increasing prices for fishmeal and fish oil whereas soya meal and oil prices have remained relatively constant; this provides an increasing incentive to replace fish with soya in feeds where this is possible.

Even the current levels of industrial fishing may not be sustainable. For example, Peruvian anchovy catches drop considerably in El Niño years. North Sea sandeel catches may only be sustainable as long as stocks of predatory fish remain greatly depleted (Furness, 1999). Capelin catches may only remain high while cod stocks that feed on those capelin remain depleted. Collapse of the Barents Sea capelin stock during the 1980s seems to have been due predominantly to very high capelin mortality as a result of rapid increases in cod recruitment but with an additional impact of industrial fishing on the capelin stock (Gjosæter, 1997). It has also been argued that industrial fisheries may have adverse effects on the rest of the food web of which the industrial stocks are a

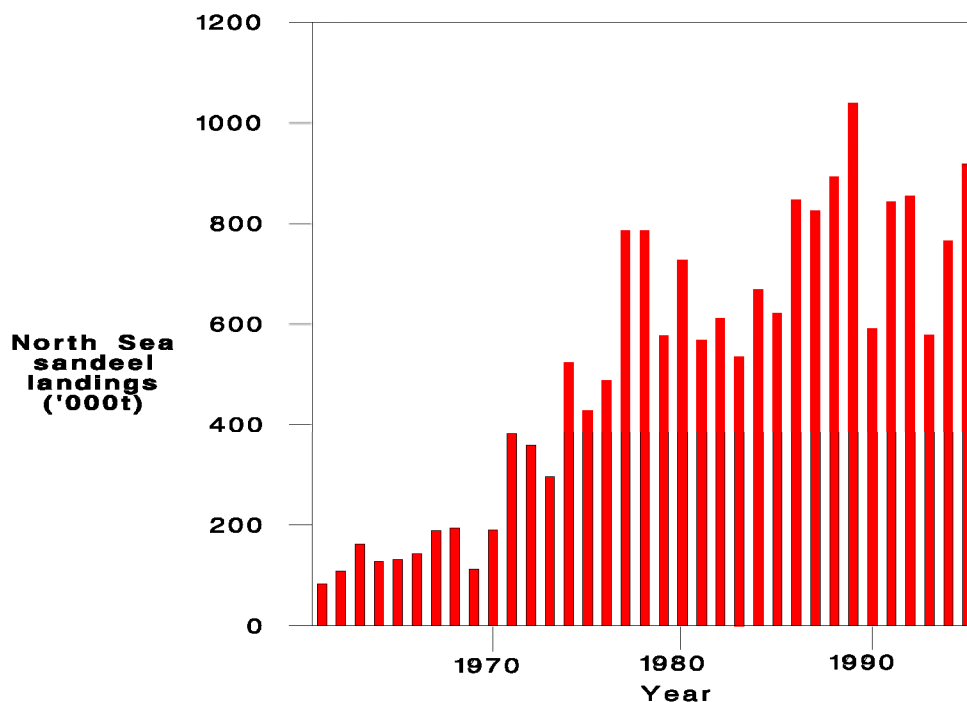


Figure 6.2. Sandeel landings from the North Sea each year from 1961 to 1995. From Furness (1999).

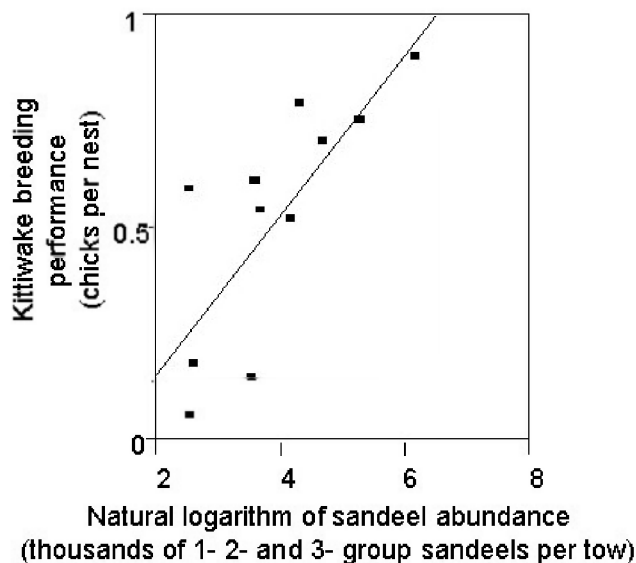


Figure 6.3. Breeding performance of black-legged kittiwakes in relation to numbers of sandeels at Shetland. From Furness (1999).

fundamental component because industrial fishery catches can be very large. For example, the North Sea sandeel fishery has grown very rapidly over the last three decades to become the largest single species fishery in the North Sea (Figure 6.2).

Industrial fisheries might deprive top predators of their food, or may alter energy  $F_{low}$  within a food web to change the balance between fish stocks. Interactions between industrial fisheries and other parts of the

ecosystem have been most intensively studied in the North Sea (sandeel fishery), Norway (herring fishery) and in the Barents Sea (capelin fishery). In the case of North Sea sandeels, many seabirds, marine mammals and predatory fish feed predominantly on sandeels in summer when these fish become available in the upper layers of the sea. We know much more about seabird feeding on sandeels than about marine mammal or predatory fish. Ecological theory predicts that some types of seabirds will be much more vulnerable to reductions in food-fish

abundance than others. Empirical data from Shetland, where sandeel abundance fell to dramatically low levels in the mid-1980s, support theoretical predictions (Furness and Tasker, 2000). One of the species expected to respond most strongly is the black-legged kittiwake. Breeding success of black-legged kittiwakes does indeed correlate with sandeel stock density, both in the North Sea, and in Shetland and over the range of sandeel abundances recorded in these years, the relationship between breeding success and log sandeel abundance is essentially linear (Figure 6.3). The North Sea sandeel fishery was closed from January 2000 in one small area of the North Sea where black-legged kittiwake breeding success had been poor for several seasons following large harvests of sandeels, on the basis that black-legged

kittiwake performance represents an indicator of the availability of sandeels to top predators in general. However, black-legged kittiwake breeding success has been high in most North Sea colonies, and breeding numbers have increased alongside the growing industrial fishery (Furness, 1999). Why has this been possible? It seems that reductions in stocks of predatory fish have more than compensated for the growing industrial fishery. Predatory fish, especially mackerel and gadoids, are by far the largest consumers of sandeels, while the needs of seabirds and marine mammals are very much less (Figure 6.4).

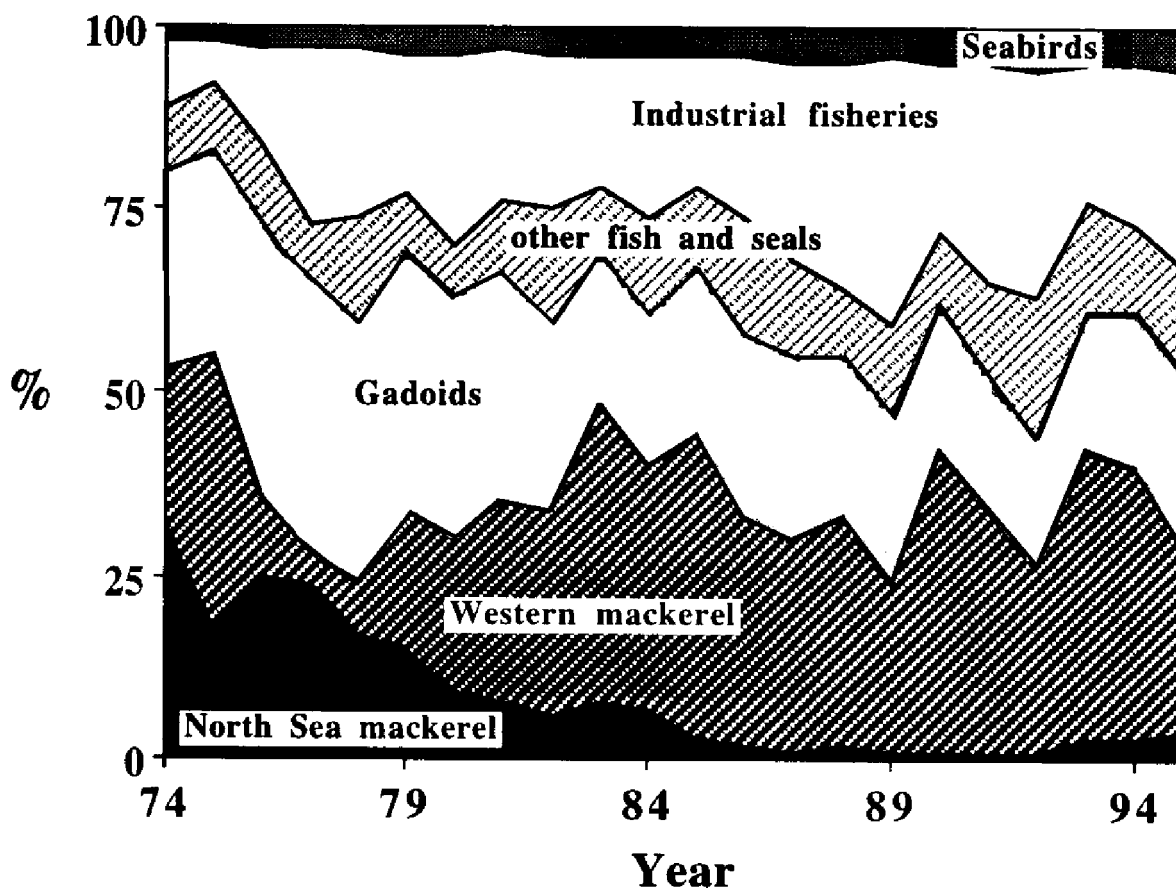


Figure 6.4. Percentages of the annual take of the North Sea stock of sandeels attributable to particular consumer groups; North Sea mackerel stock, east Atlantic mackerel stock known as 'western mackerel', gadoids (primarily whiting, haddock and cod), all other piscivorous fish plus seals, the North Sea industrial fishery, and seabirds. Quantities were estimated by bioenergetics modelling plus data on stomach contents of fish and diet composition of seabirds and seals. Data from ICES (1997).

The implication of this is that changes in abundance of predatory fish probably influence sandeel availability to seabirds and seals more than the industrial fishery does. A large industrial fishery may be compatible with healthy populations of seabirds and marine mammals in the North Sea providing predatory fish stocks remain depleted.

In Norway, long-term studies of Atlantic puffin and black-legged kittiwake breeding ecology show strong correlations between breeding success and herring abundance (Anker-Nilssen and Aarvak, 2001; Anker-Nilssen unpubl.). 0-group herring abundance explains 89% of the variation in fledging success of Atlantic puffins in the largest colony in Norway, but also 84% of annual variation in adult survival rate of Atlantic puffins, indicating a surprisingly strong influence of herring abundance on adult survival rate which is generally believed to be much less sensitive to food abundance than is reproductive success. Furthermore, the clearly sigmoidal nature of these effects is demonstrated (Anker-Nilssen and Aarvak, 2001). Performance remains fairly consistent down to a certain threshold herring abundance, rapidly falling to a low level when herring abundance falls below the threshold value. The extent to which the interannual variations in Norwegian herring abundance are due to effects of industrial fisheries is a matter of debate, but the strong influence of herring abundance on Atlantic puffin and black-legged kittiwake breeding success and on Atlantic puffin survival imply that effects of industrial fishing on herring abundance will feed through to impacts on these seabirds if herring abundance is caused to decline below the threshold level.

In the Barents Sea, stocks of cod and herring have been heavily exploited for human food consumption, and capelin are harvested by an industrial fishery. The capelin stock collapsed in the mid-1980s and again in the early 1990s. These collapses were largely due to high predation rates by cod, resulting from exceptionally large cod year classes, but exacerbated by the industrial fishery for capelin (Bogstad and Mehl, 1997; Gjosæter, 1997). The consequence for seabirds and marine mammals was dramatic. Huge numbers of starving seals from the arctic invaded Norwegian coastal waters in search of food. Almost 90% of common guillemots in the Barents Sea starved to death in winter 1986–1987 because they could not find alternative food in the absence of capelin (Vader *et al.*, 1990; Barrett and Krasnov, 1996). Despite these catastrophes, there are common features between the Barents Sea and North Sea ecosystems. In both, predatory fish are by far the biggest consumer of the food fish, with the industrial fishery and marine mammals taking less, and seabirds less again (Furness and Tasker, 1997; Mehlum and Gabrielsen, 1995). In both, fluctuations in predatory fish stocks appear to influence food fish availability to wildlife more than the industrial fishery does. However, both cases show that top predators such as seabirds and marine mammals are vulnerable to alterations in ecosystem structure, whether induced by natural variation or by fisheries.

Elsewhere, interactions between industrial fisheries, marine mammals and seabirds have not been quantified in detail. However, the loss of millions of seabirds from the Peruvian coast is a well-known example where the industrial fishery for anchoveta seems to have reduced food availability to the birds, thereby inhibiting their populations from recovering after crashes induced by El Niño events (Duffy, 1983).

A trend in aquaculture has been to reduce the quantity of fish meal and oil going into the production of a unit of aquaculture produce. Improvements in the feeding efficiency of aquaculture systems have been taking place, so that the amount of feed wasted in intensive systems has been reduced. This is important in reducing farm waste pollution impacts, but higher conversion efficiency also reduces the requirement for industrial fishing products. At present, most aquafeeds are over-formulated as nutritionally complete diets regardless of stocking density and natural food availability, and this requires adjustment to further reduce wastage. Another trend that may mitigate increased requirements for aquafeeds is alteration of aquafeed composition to contain less fishmeal and oil. Industrial fish protein can to some extent be substituted by soya protein (Thomson, 1990) though this could lead to concerns over use of genetically modified organisms in the human food chain. Fish oil can similarly be substituted by vegetable oil. Specific limiting nutrients can be added to try to compensate for chemical compositional differences between fish and other ingredients. At the moment there is much research by commercial aquafeed companies to investigate these alternatives and the suitability of altered feeds for aquaculture production and product quality. IFOMA has estimated that the use of aquafeed will more than double between 2000 and 2010 (Table 6.1). Despite reductions in the proportions of fish meal and oil in aquafeed, the quantities of these constituents needed to support this growth are anticipated to increase by about 30% over that decade (Table 6.2).

Although FAO estimate that about 25 million tonnes of fish are discarded worldwide each year, mechanisms and facilities to collect these and to convert them to fish meal and oil are rarely available, and it would not be desirable to encourage development of a market for discards as the aim of management should be to reduce to a minimum the amounts of discards generated. Furthermore, many contaminants are stored in fish livers and accumulate to higher concentrations in larger and older fish, so that contaminant problems might arise if discards and offal from continental shelf and enclosed sea fisheries were substantial contributors to fish meal. Most industrial fish are short lived and occur in upwelling regions where pollution is negligible. While it seems that some changes that reduce dependence on fish meal and oils may be practical, it is likely that the overall trend for the foreseeable future will be greater demand for fish meal and oil in aquaculture rather than a reduction, and it is probable that inclusion of discards and offal will represent only a small fraction of future aquafeed production. To the extent that industrial fisheries remove food required by seabirds or deplete prey fish stocks,

they may adversely affect seabird populations. If, as seems to be the case, fishing down the food chain results in a reduction in total mortality imposed on prey fish stocks, then adverse impacts of industrial fisheries seem more likely in upwelling food chains, where fishing down the food chain tends not to be evident. Industrial

fisheries on shelf fish stocks may be less likely to have adverse effects on seabirds because they tend to harvest from stocks of food fish that are subject to reduced mortality because their main (fish) predators have been depleted.

Table 6.1. Projections of aquaculture production by species groups and estimated requirements for aquafeeds (data from IFOMA).

Species	Production in 1996 ('000 t)	Estimated production in 2000 ('000 t)	Predicted production in 2010 ('000 t)	% estimated to be on aquafeed in 2000	% projected to be on aquafeed in 2010
Carp	11,504	13,983	36,268	25	50
Tilapia	801	974	2,526	40	60
Shrimp	1,034	1,034	1,684	80	90
Salmon	644	876	1,569	100	100
Bass, etc <sup>1</sup>	629	856	1,394	60	80
Trout	400	450	733	100	100
Catfish	330	371	604	85	90
Milkfish	364	379	462	40	75
Eel	216	216	263	80	90
Other marine fish	60	105	650	100	100
Total	15,982	19,244	46,153		

Species	Aquafeed use per tonne produced in 2000	Aquafeed use per tonne produced expected in 2010	Aquafeed required in 2000 ('000 t)	Aquafeed required in 2010 ('000 t)
Carp	2	1.5	6,991	27,000
Tilapia	2	1.5	779	2,106
Shrimp	1.8	1.6	1,489	2,425
Salmon	1.2	0.8	1,051	1,255
Bass, etc <sup>1</sup>	1.8	1.5	923	1,670
Trout	1.3	0.8	585	586
Catfish	1.6	1.4	505	761
Milkfish	2.0	1.6	303	554
Eel	2.0	1.2	346	284
Other marine fish	1.2	0.9	126	585
Total			13,098	37,226

<sup>1</sup>Bass, bream, yellowtail, grouper, jacks and mullets.

Table 6.2. Predicted requirements for fish meal and oil for use in aquafeeds in 2010 compared with present use (data from IFOMA).

Species	% Fish meal in feed in 2000	Predicted % fish meal in feed in 2010	% Fish oils in feed in 2000	Predicted % Fish oils in feed in 2010	Fish meal required in 2000 ('000 t)	Fish meal required in 2010 ('000 t)	Fish oil required in 2000 ('000 t)	Fish oil required in 2010 ('000 t)
Carp	5	2.5	1	0.5	350	675	70	135
Tilapia	7	3.5	1	0.5	55	74	8	11
Shrimp	25	20	2	3	372	485	29	73
Salmon	40	30	25	20	454	377	283	251
Bass, etc <sup>1</sup>	45	40	20	15	415	668	185	251
Trout	30	25	15	20	176	147	88	117
Catfish	3	-	1	1	15	-	5	8
Milkfish	12	5	3	2	36	28	10	11
Eel	50	40	5	10	173	114	17	28
Other marine fish	55	45	10	12	69	263	13	70
Total					2,115	2,831	708	955

<sup>1</sup>Bass, bream, yellowtail, grouper, jacks and mullets.

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## Annex 1: Scientific names of species used in this report

King penguin	<i>Aptenodytes patagonius</i>
Black-throated diver	<i>Gavia arctica</i>
Great crested grebe	<i>Podiceps griseigena</i>
Wandering albatross	<i>Diomedea exulans</i>
Northern fulmar	<i>Fulmarus glacialis</i>
Snow petrel	<i>Pagodroma nivea</i>
Blue petrel	<i>Halobaena caerulea</i>
Bulwer's petrel	<i>Bulweria bulwerii</i>
Cory's shearwater	<i>Calonectris diomedea</i>
Short-tailed shearwater	<i>Puffinus tenuirostris</i>
Little shearwater	<i>Puffinus assimilis</i>
Madeiran storm-petrel	<i>Oceanodroma castro</i>
Leach's storm-petrel	<i>Oceanodroma leucorhoa</i>
Northern gannet	<i>Morus bassanus</i>
Great cormorant	<i>Phalacrocorax carbo</i>
Double-crested cormorant	<i>Phalacrocorax auritus</i>
European shag	<i>Phalacrocorax aristotelis</i>
Grey heron	<i>Ardea cinerea</i>
Great blue heron	<i>Ardea herodias</i>
Great egret	<i>Ardea alba</i>
Black-crowned night heron	<i>Nycticorax nycticorax</i>
Common eider	<i>Somateria mollissima</i>
Long-tailed duck	<i>Clangula hyenalis</i>
Black scoter	<i>Melanitta fusca</i>
Velvet scoter	<i>Mellanitta fusca</i>
Common goldeneye	<i>Bucephala clangula</i>
Osprey	<i>Pandion haliaetus</i>
Eurasian oystercatcher	<i>Haemotopus ostralegus</i>
Arctic skua	<i>Stercorarius parasiticus</i>
Great skua	<i>Stercorarius skua</i>
Little gull	<i>Larus minutus</i>
Black-headed gull	<i>Larus ridibundus</i>
Mediterranean gull	<i>Larus melanocephalus</i>
Common gull	<i>Larus canus</i>
Ring-billed gull	<i>Larus delawarensis</i>
California gull	<i>Larus californicus</i>
Lesser black-backed gull	<i>Larus fuscus</i>
Herring gull	<i>Larus argentatus</i>
Yellow-legged gull	<i>Larus michahellis</i>
Great black-backed gull	<i>Larus marinus</i>
Black-legged kittiwake	<i>Rissa tridactyla</i>
Sandwich tern	<i>Sterna sandvicensis</i>
Roseate tern	<i>Sterna dougallii</i>
Common tern	<i>Sterna hirundo</i>
Arctic tern	<i>Sterna paradisaea</i>
Common guillemot	<i>Uria aalge</i>
Brunnich's guillemot	<i>Uria lomvia</i>
Razorbill	<i>Alca torda</i>
Atlantic puffin	<i>Fratercula arctica</i>



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